Universidade Federal de Minas Gerais

Instituto de Ciência Biológicas

Departamento de Genética, Ecologia e Evolução

Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre

Paloma Marques Santos

Avaliação da ocupação e da adequabilidade ambiental para a preguiça-de-coleira (*Bradypus torquatus*, Illiger 1811): uma abordagem multi-espacial e multi-temporal com contribuições para a conservação da biodiversidade.

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Tese apresentada ao Instituto de Ciências Biológicas da Universidade Federal de Minas Gerais, para obtenção do título de Doutora em Ecologia, Conservação e Manejo da Vida Silvestre.

Orientador: Dr. Adriano Pereira Paglia

Coorientadores: Dr.Adriano Garcia Chiarello e Dr. Milton Cezar Ribeiro.

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043 Santos, Paloma Marques.

Avaliação da ocupação e da adequabilidade ambiental para a preguiça-de-coleira (*Bradypus torquatus*, Illiger 1811): uma abordagem multi-espacial e multi-temporal com contribuições para a conservação da biodiversidade. [manuscrito] / Paloma Marques Santos – 2020.

246 f. : il. ; 29,5 cm.

Orientador: Dr. Adriano Pereira Paglia. Co-orientadores: Dr. Adriano Garcia Chiarello e Dr. Milton Cezar Ribeiro.

Tese (doutorado) – Universidade Federal de Minas Gerais, Instituto de Ciências Biológicas. Programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre.

1. Ecologia. 2. Florestas. 3. Biodiversidade. 4. Conservação dos Recursos Naturais. 5. Bichos-Preguiça. 6. Mudança Climática. I. Paglia, Adriano Pereira. II. Chiarello, Adriano Garcia. III. Ribeiro, Milton Cezar. IV. Universidade Federal de Minas Gerais. Instituto de Ciências Biológicas. V. Título.

CDU: 502.7

Ficha catalográfica elaborada pela bibliotecária Rosilene Moreira Coelho de Sá – CRB 6 – 2726

UFMG

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ECMVS

Tese defendida em 4 de junho de 2020 e aprovada pela banca examinadora constituída pelos membros:

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Doutor Adriano Pereira Paglia
(Presidente da Banca)

À Maria Luiza e à Maria Angélica,

por sempre me mostrarem

a força que há dentro de mim.

AGRADECIMENTOS

Eu demorei 1 mês para conseguir escrever esses agradecimentos. Não é fácil, definitivamente, transpor em linhas os sentimentos acumulados nesses 7 anos de pós-graduação, entre mestrado e doutorado. Há uma grande possibilidade de eu ter esquecido de alguém. Não por mal, mas escrever essa parte da tese enche meus olhos de lágrimas.

Em um momento de ataques, desvalorização e desmonte da ciência e educação brasileira, gostaria de começar agradecendo imensamente à CAPES e ao CNPq. Se não fossem pelas bolsas que recebida ao longo desses anos é evidente que pouco provavelmente, eu estaria aqui. Graças à CAPES (Processo 88881.188493/2018-01), eu realizei um sonho de morar 6 meses nos EUA para realizar parte de meu doutorado, e ir além em minha formação profissional. Gostaria de agradecer a Rufford Small Grants (21498-1), por financiar meus trabalhos de campo.

Tive sorte de ter em meu caminho, um super time de orientadores que se fizeram presentes nos processos dessa tese. Quero agradecer profundamente o meu orientador, Adriano Pereira Paglia, que abriu os braços a mim em 2013, e que desde então, vem me apoiado, em ensinado e me guiado. Também gostaria de agradecer aos meus coorientadores: Milton Cezar Ribeiro e Adriano Garcia Chiarello. Suas diferentes visões de ciência e de mundo me fizeram pensar muito além do que eu poderia imaginar. E claro, não poderia finalizar sem agradecer à Larissa Lynn Bailey, à qual eu tive a oportunidade de conviver por 6 meses, e que mostrou a simplicidades das coisas, a tranquilidade e me apresentou à uma ecologia nunca antes desbravada por mim.

Gostaria de agradecer à banca, pelos ensinamentos e valiosas discussões não só acerca do conteúdo dessa tese, mas também de toda a ciência, ecologia e conservação. Ao Paulo de Marco, a Renata Frederico, ao grande Fabiano Rodrigues de Melo (Bião!), à incrível Camila Cassano, deixo aqui meu muito obrigada. Agradeço também ao suplente Pedro Giovani, ao Ubirajara de Oliveira (que não pôde participar). E por fim, agradeço a Kátia Ferraz e ao Flávio Rodrigues, que diante da burocracia que se instaurou, compreenderam perfeitamente a situação.

À UFMG, pelo suporte incrível, por me oferecer uma biblioteca incrível, por me alimentar com uma comida boa e de qualidade. Ao programa de Pós-Graduação em Ecologia, Conservação e Manejo da Vida Silvestre (ECMVS), pelo suporte ao longo desses 7 anos, e por me ensinarem que tipo de profissional eu devo e não devo ser. À Vila Parentoni, pela amizade, companheirismo e alegria, que me acompanharam todo o tempo. Ao Laboratório de Ecologia e Conservação (LEC – UFMG) – Lud, Pão, Rodrigo, Aninha, Rodolfo, Léo, Augusto, Arthur, Dani, pelo companheirismo, alegrias, e ciência compartilhada, em especial à Gabi, por todo o suporte ao longo desses anos, todo aprendizado, todo toró de ideia, e amizade. Ao Laboratório de Ecologia Espacial e Conservação (LEEC – UNESP) – Rê, Milene, Ju Assis, Ju Oshima, Ju Silveira, Felipe Martelo, Erison, Ber, André Regolin, Thadeu Sobral, Laurinha, João Penna, Vinicius e demais, por todo aprendizado, por sempre me receberem tão bem nas minhas idas à Rio Claro. E por fim, ao Laboratório de Ecologia e Conservação (LAEC – FFLCRP/USP) - Roberta, Nielson, Vinicius, Natalia, Victor - que mesmo não tão frequente, me ajudaram de diversas formas nessa tese. A todos os meus amigos em Fort Collins, que me fizeram sentir em casa: Abbey, Emma, John, Paul, Kim, Gillian, Melody, Kris, Danny, Bennet, obrigada por fazer minha estadia nos EUA leve.

Ao ICMBio, pelas licenças e pelo apoio para a realização dessa pesquisa, principalmente ao Centro Nacional de Pesquisa e Conservação de Primatas Brasileiros (CPB). Ao Instituto de Pesquisa e Conservação de Tamanduás no Brasil – Instituto Tamanduá, por terem me recebido tão bem a pouco mais de um ano, e que desde então, me fez entender o verdadeiro de significado de Conservação da Natureza. Em especial à Flávia Miranda, heroína da conservação, que tanto me inspira e me incentiva a trilhar meu caminho junto à conservação aplicada.

A todo pessoal capixaba – ao IEMA/ES, pelas autorizações de pesquisa e suporte ao longo desses anos. Ao Professor Sérgio Lucena Mendes, que desde o mestrado tem me ajudado de diversas formas, mesmo com recursos limitados. Ao pessoal da UFES, pelo apoio e suporte logístico em meus campos: Andressa Gatti, Dani Moreira, Joana Zorzal e principalmente, à Nila Rássia, por me ajudar em campo, sem hesitar. Pela companhia, amizade nos mais de 8 meses de campo. A todos o moradores da Região Serrana do Espírito Santo – Seu Floriano, Dona Deolinda, Seu José, Seu Milton, Dona Laura, Leonardo Salvador, Edson Simer, Gerson Beger, Seu Futrica, Família Espíndola, João Michio, Seu Fio Módulo, Família Del Puppo, e todas as outras famílias pela assistências, prosa boa, cafés e acolhimento durante essas andanças. Vocês foram fundamentais!

À minha família, que sempre me apoiou e que foram (e são!) fundamentais nessa caminhada. Aos meus dois irmãos – Eduardo e Allyson, pelo carinho, suporte, e atenção comigo. Ao meu pai Eduardo, que sempre esteve comigo, do seu jeito calado, mas amoroso. Ao meu padrasto Flávio por toda preocupação e atenção. Às minhas sobrinhas Flavinha e Ana Marcela, por toda alegria e pureza. Às minhas cunhadas Clécia e Riana, pelos sorrisos e palavras de afeto. À minha sogra, dona Tuca, pela tranquilidade e paz.

Dedico esse parágrafo a somente ela. Ela que desde cedo me falou para eu ser forte, e sempre me mostrou onde buscar essas forças. Ela que matava 3 leões por dia para me dar os melhores cuidados. Ela que enfrentou de cabeça erguida uma sociedade machista e racista. Ela que não mediu esforços para proteger e educar os seus 3 filhos. À minha mãe, Maria Luiza, muito obrigada. Palavras não cabem aqui, o amor incondicional é o que prevalece.

Ao Candomblé e à minha família de santo, que me mostrou um novo sentido à vida quando eu mais precisava. À Eduardo de Omolu e à Alessandra Gomes, por todo amor, carinho, apoio, compreensão e ensinamentos. À Loy, por todos os conselhos, amor e abrigo. Aos meus queridos irmãos de santo por todo amor e amizade: Ti, Carioca, Bruna, Mari, Guto, Flávia, Richard, Roberto, Mirna, Philip, Rômulo, Gabriel, Iara, João, Vini, vocês são incríveis! À mãe da minha cabeça, Oyá, por me dá forças sempre que eu me mostro fraca. Ao meu pai Ogum, por me ajudar a abrir meus caminhos. A Exu, por sempre me guiar e me mostrar o caminho. A todos os orixás e entidades que me acompanham e não me deixam só, à toda minha ancestralidade, que me nutre com uma força inimaginável: Obrigada!

Aos meus amigos fora da academia e da universidade, que estão presentes em diversos momentos: Alysson, Mateus, Pedro, Marcinho, Rayana, Mayra, Sassá, Lu, Ciba, Patilda e em especial às novas baianas Kele, Alice, Jana e Cacá, pela amizade e amor desde que cheguei em BH. Aos meus tambores que nunca se calaram! E por sempre me mostrarem a força que há dentro de mim. Ao Maracatu Ilê Asé Sòpònnòn, ao Baque mulher, ao mulheraço, ao Babadan e ao Magia negra, ressignificar o sentido de resistência!

E por fim, mas não menos importante. Esse parágrafo é dedicado ao Alê, mas poderia ser um agradecimento à parte. Meu companheiro, meu amor, meu melhor amigo. Concordou em ficar 6 meses longe de mim quando fui para os EUA. Me abraçou forte nos meus momentos de tristeza, nas incertezas do futuro. Me dá conforto, compreensão. Me ensina que a vida precisa ser mais calma. Me ensina a respirar. Me ensina o que é amor. A você, Alê, um muito obrigada. Palavras não são suficientes para descrever o que você e para mim.

Enfim, a todos que contribuíram não só para essa tese, mas para a minha formação pessoal. Obrigada. Olorum Odupé.

“*A única coisa que separa as mulheres de cor de qualquer outra pessoa é a oportunidade.”*

Viola Davis

“*Os (...) entomologistas Bert Hölldobler e Ed Wilson fizeram uma afirmação intrigante no livro* Journey to the ants. *(...) há dois tipos de cientistas. Um é o teórico, que se interessa por uma questão específica e procura o melhor organismo para resolvê-la. (...). O outro tipo é o naturalista, que se interessa por uma classe específica de animais em si, percebendo que cada animal tem sua história para contar, que se revelará dotada de interesse teórico se for suficientemente estudada. Hölldobler e Wilson consideram-se pertencentes à segunda classe, assim como eu. “*

Frans De Waal, no livro “*Eu, primata*”

Resumo

As áreas antrópicas relacionadas à produção de alimentos são uma das grandes responsáveis pelas mudanças climáticas, uma vez que a conversão de habitats naturais em paisagens agropastoris cumpre uma função fundamental no aumento da liberação do CO_2 no mundo. A crescente perda de habitat interrompe fluxos gênicos e processos ecológicos importantes para a biodiversidade e assim, as mudanças no uso da terra e seus efeitos nas mudanças climáticas correspondem a uma das principais causas de extinção de espécies. Espécies arborícolas são as maiores atingidas pela perda de habitat, como é o caso da Preguiça-de-Coleira (*Bradypus torquatus*), exclusiva da ameaçada Mata Atlântica. Assim, a presente tese teve como objetivo principal entender a influência de fatores ambientais na probabilidade de ocupação da preguiça-de-coleira e avaliar a adequabilidade ambiental atual e futura para a espécie, além de contribuir com a geração de dados de biodiversidade de uso livre. O primeiro capítulo buscou compilar e disponibilizar dados de ocorrência sobre os xenartros na região neotropical, no intuito de facilitar análises e modelagens biológicas e ecológicas diversas. O segundo capítulo objetivou identificar a influência da cobertura florestal e outras varáveis na probabilidade de ocupação da preguiça-de-coleira. O terceiro capítulo investigou a adequabilidade ambiental para a espécie, levando em consideração cenários de regeneração e cenários de mudanças climáticas. Como principais resultados podemos citar: 1) a compilação de mais 30 mil registros de xenartros – incluindo da preguiça-de-coleira, de diferentes instituições de todo mundo e de diferentes fontes – dados estes que, em sua maioria, até então estavam indisponíveis para o acesso público; 2) a descoberta de que preguiça-de-coleira responde a um limiar de 35% de cobertura florestal, abaixo do qual a probabilidade de ocupação desta espécie declina consideravelmente, chegando a quase zero em regiões com menos de 20% de floresta. Além disso, verificamos que a porcentagem de áreas abertas, como pastos, pastos abandonados e áreas de solo expostos, afetam negativamente a probabilidade de ocupação desta preguiça, enquanto áreas com presença de árvores importantes para a sua dieta tem influência positiva; 3) cenários de regeneração natural podem mitigar os efeitos das mudanças climáticas na adequabilidade ambiental para a preguiça. De modo geral, essa tese reforça a importância de áreas florestais como preditoras da ocorrência da espécie, sendo locais com baixa cobertura florestal inadequadas para a sua presença. Além disso, a disponibilização de dados de ocorrência dos xenartros irá facilitar o desenvolvimento de diversas pesquisas e avaliações conservacionistas voltadas a este grupo relativamente ainda pouco estudado. Os produtos aqui gerados irão contribuir com metas e ações estabelecidos em planos de ações nacionais, visando a conservação a longo-prazo da espécie.

Palavras-chave: Mata Atlântica, desmatamento, biodiversidade, ecologia de paisagem, conservação

Abstract

Food production is one of the main responsible for climate change, once converting natural habitats into large agricultural landscapes plays a fundamental role in global warming. Additionally, the habitat loss may interrupt gene flows and important ecological processes. Therefore, changes in land use and its effects on climate change correspond to one of the main causes of species extinction. Arboreal species are the most affected by habitat loss, due to their high dependence on forests, as is the case of the maned sloth (*Bradypus torquatus*), exclusive of the threatened Atlantic Forest. Thus, the main objective of this present thesis was to understand the influence of environmental factors in the occupancy probability of *B.torquatus* and to evaluate the current and future environmental suitability for the species, besides contributing to the generation of open biodiversity data. The first chapter sought to compile and make available occurrence data on the xenarthrans, in order to contribute with several analyzes and ecological modeling. The second chapter aimed to identify the influence of forest cover and other variables on the occupancy probability of *B.torquatus*. The third chapter evaluated the environmental suitability for the species, considering regeneration and climate change scenarios. As a result: 1) We compiled over 30,000 xenarthran records, from different institutions around the world and from different sources - including data that until then were unavailable for public access; 2) The occupancy probability of the maned sloth is 0.97, but it decreases abruptly at 35% of forest cover, reaching zero in areas with less than 20% of forest cover. In addition, open areas cover, such as pastures, abandoned pastures and exposed soil, has a negative effect on *B.torquatus* occupancy probability, whereas areas with the presence of important trees to their diet affect positively; 3) Natural regeneration can mitigate the effects of climate change on environmental suitability for maned sloth. Overall, this thesis reinforces that forest areas are important predictors for species, and their low coverage implies in unsuitable areas for their presence. Additionally, the provision of xenarthran data will favor the development of several research and conservationist assessments focused on the group. The products generated here will contribute to goals and actions established in national action plans, aiming the long-term conservation of the species.

Keywords: Atlantic Forest, deforestation, biodiversity, landscape ecology, conservation

Sumário

Introdução geral

De acordo com o último relatório do IPCC (Painel Intergovernamental sobre Mudanças Climáticas; em inglês *Intergovernmental Panel on Climate Change* 2019), cerca de 56% das áreas terrestres mundiais são utilizadas para fins econômicos (pecuária, agricultura, silvicultura, extração de madeira e infraestrutura), e apenas 16% são ocupadas por vegetação nativa, entre florestas e áreas naturais não-florestais. Essa intensa conversão de ambientes naturais em extensas áreas antrópicas cria ambientes cada vez mais hostis para diversas espécies, ameaçando fortemente a conservação da biodiversidade mundial (BELLARD *et al.*, 2014; HADDAD *et al.*, 2015; KOUMARIS; FAHRIG, 2016).

A quantidade de habitat remanescente possui um papel fundamental na dinâmica de populações e comunidades (FAHRIG, 2013). Alguns grupos reagem linearmente à perda de habitat (BECA *et al.*, 2017), enquanto outros exibem uma resposta não-linear (BOESING; NICHOLS; METZGER, 2018; MUYLAERT; STEVEN; RIBEIRO, 2016). Este último relaciona-se aos limiares críticos (*critical thresholds),* e seu conceito sugere que pequenas modificações na proporção de habitat disponível podem levar a mudanças abruptas nas respostas ecológicas (SWIFT; HANNON, 2010). Assim, tais limiares (também chamados de Limiares de fragmentação) prevê uma quantidade mínima de habitat – que poderá variar entre as espécies (SWIFT; HANNON, 2010; MUYLAERT; STEVEN; RIBEIRO, 2016; BOESING; NICHOLS; METZGER, 2018) - no qual abaixo dela a conectividade da paisagem estará prejudicada e as espécies tornam-se mais dependentes de fatores como tamanho e isolamento dos fragmentos. Além da quantidade de habitat, as espécies – principalmente as florestais – necessitam de áreas naturais adequadas e bem estruturadas para desempenhar suas atividades diárias. Habitats complexos consiste em locais com estratificação florestal bem definida e variada (AUGUST, 1983). Sendo assim, fatores como altura do dossel, abertura da copa, composição florística, estratificação florestal e complexidade de habitat são essenciais para manter a estrutura e a viabilidade de populações e comunidades em uma mancha de habitat (GRELLE, 2003; PAIM, FERNANDA P. *et al.*, 2017; PAIM, FERNANDA POZZAN *et al.*, 2018; SMITH *et al.*, 2019).

Áreas florestais também são grandes responsáveis por estocar grande parte do carbono mundial, e transformá-las em áreas agropastoris - liberando o carbono armazenado – cumprem papel fundamental no aquecimento global (IPCC, 2018). Portanto, diminuir – ou zerar – o desmatamento em larga escala é uma das formas mais eficazes de mitigar os efeitos das mudanças climáticas. Além disso, torna-se necessário o estímulo à regeneração e à restauração florestal, uma vez que árvores jovens em crescimento são importantes sumidouros de carbono (CASPERSEN *et al.*, 2000). Além de ajudar na regulação climática, o crescimento das áreas florestais viabiliza a manutenção conservação da biodiversidade (METZGER *et al.,* 2019). Atualmente, o Brasil conta com a Política Nacional de Recuperação da Vegetação Nativa (Planaveg - Decreto nº 8.972, de 23 de janeiro de 2017) , cujo o principal objetivo é "*articular, integrar e promover políticas, programas e ações indutoras da recuperação de florestas e demais formas de vegetação nativa e de impulsionar a regularização ambiental das propriedades rurais brasileiras, nos termos da Lei nº 12.651, de 25 de maio de 2012, em área total de, no mínimo, doze milhões de hectares, até 31 de dezembro de 2030.*" . De acordo com o plano, tal recuperação visa principalmente as Áreas de Preservação Permanente (APP) e as Reservas Legais (RL), por meio de inciativas que envolvam a sensibilização de toda a cadeia produtiva para a recuperação das áreas naturais.

Além da recuperação da vegetação nativa, outros mecanismos conservacionistas se destacam para frear a perda de habitat e garantir a preservação de áreas naturais, como por exemplo o estabelecimento de áreas protegidas. No Brasil, as áreas protegidas são conhecidas por Unidades de Conservação e estão garantidas por lei. O Sistema Nacional de Unidades de Conservação (SNUC – Lei 9,985/2000) foi instituído como forma de potencializar o papel das diferentes categorias de UCs na conservação da biodiversidade. Complementando as funções das UCs no Brasil, a Lei de Preservação da Vegetação Nativa (LPVN, Lei 12,651/2012) estabelece a proteção de áreas ambientais áreas sensíveis, como topos e encostas de morros, nascentes e margens de rios, lagos e lagoas – as chamadas Áreas de Preservação Permanente (APPs), além da proteção de áreas de vegetação nativa localizadas em propriedades particulares – Reservas Legais (RL). Tais mecanismos visam a preservação de áreas não englobadas pelo SNUC, de forma a garantir diversos serviços ecossistêmicos. Porém, de modo geral, algumas dessas áreas definidas e protegidas vêm sendo constantemente ameaçadas por propostas de leis que flexibilizam a supressão vegetal para fins agrícolas – no caso das RLs (Projeto de Lei n. 2362/2019, o qual já fora retirado de votação do Senado Federal), ou seja por propostas de leis que permitem a abertura e uso de estradas dentro de Parques Nacionais (Projetos de Lei 984/2019 - em trâmite - e 61/2013 - este último foi barrado pelo Superior Tribunal Federal recentemente).

A Mata Atlântica brasileira ocupa hoje um território de 1,08 milhão km^2 (MUYLAERT *et al.*, 2018). Com sua distribuição restrita a pouco mais de 28% da cobertura original (REZENDE *et al.*, 2018), o bioma possui um alto déficit de Unidades de Conservação, APPs e Reservas Legais (SOARES-FILHO *et al.*, 2014). Considerado um *hotspot* para a conservação da biodiversidade (MYERS *et al.*, 2000), diversas espécies endêmicas e ameaçadas habitam a área do bioma, e a deficiência das áreas legalmente protegidas pode agravar a situação de ameaça. Dentre as espécies endêmicas, encontra-se a Preguiça-de-Coleira *Bradypus torquatus* (Pilosa: Bradypodidae). Habitando quase que exclusivamente florestas ombrófilas densa, entre os estados de Sergipe e Rio de Janeiro (HIRSCH; CHIARELLO, 2012), a Preguiça-de-Coleira se alimenta basicamente folhas e brotos (CHIARELLO, 1998b; MONTGOMERY; SUNQUIST, 1975), fato intrinsicamente relacionado à baixa taxa metabólica (NAGY; MONTGOMERY, 1980). Em função de seu hábito estritamente arborícola e folívoro, a espécie é profundamente afetada pela fragmentação, perda e destruição do habitat (CHIARELLO *et al.*, 2018; CHIARELLO; MORAES-BARROS, 2014; SANTOS *et al.*, 2019). Tais fatores, associados a uma restrita e descontínua distribuição tornam essa espécie ameaçada, e atualmente ela está classificada como vulnerável tanto na lista nacional (Portaria MMA n°444/2014; CHIARELLO *et al.*, 2018), quanto na internacional (CHIARELLO; MORAES-BARROS, 2014) de espécies ameaçadas. A preguiça-de-coleira está, atualmente, no Plano de Ação Nacional para a Conservação dos Primatas da Mata Atlântica e da Preguiça-de-Coleira (PAN PPMA), o qual institui metas para a conservação da preguiça. Essa tese contribui diretamente na execução de importantes ações presentes no PAN PPMA.

Além da perda de habitat, as mudanças climáticas podem exercer um papel fundamental no declínio das populações de Preguiça-de-Coleira. Tal fator pode impactar a espécie de duas formas: direta e indireta, ambas podendo atuar em sinergia. A forma direta está relacionada com a biologia e fisiologia da espécie: As preguiças, de modo geral, possuem uma temperatura bastante instável (CHIARELLO, 1998a; GINÉ *et al*., 2015), devido ao seu baixo metabolismo (McNAB, 1978). Portanto, condições climáticas extremas (e.g. temperaturas muito baixas/altas, precipitação baixa ou irregular, alta sazonalidade etc. – condições que poderão ser acentuadas pelas mudanças climáticas – IPCC, 2018) podem representar uma grande ameaça a espécie. Por outro lado, a forma indireta relaciona-se ao seu ambiente. As preguiças precisam da copa das árvores para desempenhar diversas atividades diárias (CHIARELLO, 1998a, 1998b), incluindo o comportamento termorregulador, o qual é responsável por evitar mudanças bruscas em sua temperatura (MONTGOMERY; SUNQUIST, 1975; CHIARELLO, 1998a, GINÉ *et al*., 2015). As mudanças climáticas poderão afetar a estrutura, a cobertura e a composição das áreas naturais (VELAZCO *et al.*, 2018; PREVEDELLO *et al.*, 2019), impactando diretamente a oferta de recurso e as condições ambientais – essenciais para a manutenção das populações de preguiças (CHIARELLO, 2008; SANTOS *et al*., 2016).

As preguiças pertencem à superordem Xenarthra juntamente com os tamanduás e tamanduaí – que compõe a Ordem Pilosa - e os tatus – da Ordem Cingulata (GIBB *et al.*, 2016). Os xenarthras é um dos grupos de mamíferos mais antigos existentes – sendo um dos quatro clados basais dos mamíferos placentários (DELSUC *et al.*, 2002). Ademais, é o único grupo com origem na América do Sul, onde sua história teve início, por volta de 65 milhões de anos atrás (GIBB *et al*., 2016; porém ver DELSUC; DOUZERY, 2008), e hoje conta com 37 espécies existentes – a grande maioria pertencentes aos tatus (SANTOS *et al.*, 2019).

Embora os xenarthras possam prover enormes descobertas acerca da evolução dos primeiros mamíferos placentários, de forma geral, muitos aspectos do grupo permanecem sem respostas (SUPERINA; LOUGHRY, 2015). As espécies de Xenarthra podem ser bastante crípticos ao ambiente em que vivem, tornando difíceis pesquisas que abordem parâmetros demográficos. Felizmente, a ciência tem avançado de forma contínua, permitindo o desenvolvimento de modelos estatísticos e matemáticos que possibilitam construir análises levando em consideração a baixa detectabilidade das espécies em seu meio natural. A modelagem de ocupação é uma alternativa para avaliar aspectos ecológicos e biológicos de espécies crípticas (MACKENZIE *et al.*, 2006). Tal abordagem estima a proporção da área, de fragmentos ou de unidades de amostra que é ocupada, corrigida pela detecção imperfeita ou falsa ausência, i.e., a espécie pode estar lá, mas não foi possível detectá-la, em consequência de diversos fatores, como comportamento das espécies ou características do habitat (MACKENZIE *et al.*, 2006).

Adicionalmente, por meio da obtenção de dados já existentes em banco de dados e bibliotecas virtuais, é possível desenvolver outros tipos de modelos que forneçam importantes informações ecológicas acerca de um grupo. Aqui inclui-se a Modelagem de Distribuição de Espécies (em inglês *Species Distribution Models* - SDM). Tal abordagem fundamenta-se ecologicamente no conceito de nicho ecológico - o "*hiper-volume n-dimensional*"

(HUTCHINSON, 1957), ou o conjunto de condições e recursos que permitem à espécie suprir suas necessidades mínimas para sobreviver (CHASE; LEIBOLD, 2003). Dessa forma, o SDM é capaz de prever áreas potenciais para a distribuição de uma espécie ou avaliar áreas adequadas para um grupo (FERRAZ *et al.*, 2012; GUISAN *et al.*, 2013; JOSE V; NAMEER, 2020) através de dados de ocorrência e de varáveis ambientais, acessando a relação entre a distribuição de uma determinada espécie (espaço geográfico) as condições ambientais (espaço ambiental) (PULLIAM, 2000; MARTÍNEZ-MEYER *et al.*, 2006; PETERSON, 2006, SOBERÓN, 2007), com amplo uso biológico (MORATO *et al.*, 2014; PORTUGAL *et al.*, 2019).

Diversas plataformas on-line, como *SpeciesLink* (http://splink.cria.org.br/), GBIF (https://www.gbif.org/) e GIFT (http://gift.uni-goettingen.de/home), disponibilizam dados de biodiversidade, os quais podem ser utilizados em variadas análises. Se somam a essas plataformas os projetos de *datapapers* das séries ATLANTIC SERIES e NEOTROPICAL SERIES, gerados e publicados nos três últimos anos. Ao compilar, integrar e disponibilizar dados diversos de biodiversidade (muitos deles não publicados e presentes apenas em dissertações, teses e relatórios de consultorias) de acesso gratuito e livre, os *datapapers* integra-se ao *Open data movement* (tradução livre: movimento (ou iniciativa) por dados abertos) que promove a disponibilização e uso de dados científicos de forma livre e irrestrita (BAACK, 2015).

Utilizando modelos de ocupação e modelos de distribuição de espécies, a presente tese buscou entender a influência de fatores locais e espaciais sobre a probabilidade de ocupação da preguiça-de-coleira e avaliar a adequabilidade ambiental presente e futura para a espécie ao longo de sua distribuição. Essa tese buscou, ainda, a compilação e a disponibilização de dados biológicos, se alinhando ao *open data movement*. A estrutura da tese se encontra dividida em três capítulos. O primeiro capítulo – já publicado no periódico internacional *Ecology* - consistiu em reunir, padronizar e disponibilizar dados de ocorrência da superordem Xenarthra em toda região Neotropical (SANTOS *et al.*, 2019). Como resultado, compilamos mais de 35 mil registros de ocorrência, distribuídos desde o Sul da Argentina/Chile, até a região Sudeste do Estados Unidos. O *datapaper* contou com colaboração de 505 pesquisadores de 282 instituições de várias partes do mundo que nos enviaram dados de xenartros de diversos, anos, tipo e locais. Além dos dados enviados pelos colaboradores, foram compilados também dados de literatura – artigos, tese, dissertação, relatórios. Constatou-se que dados de xenartros já publicados e disponíveis correspondem a menos de 20% do *dataset*, e, portanto, o *datapaper* está sendo fundamental na integração e disponibilização de dados exclusivos. O presente capítulo se junta a iniciativas como ATLANTIC CAMTRAP (LIMA, F. *et al.*, 2017), ATLANTIC BATS (MUYLAERT *et al.*, 2017) , ATLANTIC PRIMATES (CULOT *et al.*, 2018) e ATLANTIC SMALL MAMMALS (BOVENDORP *et al.*, 2017), sendo o primeiro da série NEOTROPICAL a ser publicado (SANTOS *et al.*, 2019).

O segundo capítulo - publicado no periódico internacional *Biological Conservation* - buscou identificar e compreender como variáveis locais e espaciais influenciam a probabilidade de ocupação da preguiça-de-coleira. Para isso, foram realizadas campanhas de campo a fim de se detectar a espécies e coletar variáveis locais, em mais de 60 pontos amostrais localizados na região Serrana do Espírito Santo. Por meio de ferramentas de SIG, realizamos análises espaciais para calcular a porcentagem de floresta e de áreas abertas, utilizando a abordagem de multi-escala. Por meio de seleção de modelos, selecionamos a escala de efeito mais apropriada e utilizamos a modelagem de ocupação para identificar limiares críticos para a presença da preguiça-de-coleira (SANTOS *et al.*, 2019).

Já o terceiro capítulo consiste na utilização de Modelos de Distribuição de Espécies e dos dados compilados nos dois capítulos anteriores para avaliar a adequabilidade ambiental presente e futura para a preguiça-de-coleira ao longo de sua distribuição. Para isso, levamos em consideração um cenário atual e três cenários futuros relacionados à porcentagem de cobertura florestal, e em cada cenário incluímos variáveis climáticas em diferentes cenários de mudanças climáticas. Particularidades genéticas das preguiças-de-coleira foram consideradas, analisando as duas Unidades Evolutivas Significativas separadamente.

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Capítulo 1

NEOTROPICAL XENARTHRANS: a dataset of occurrence of xenarthran species in the Neotropics

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** Manuscript, data and complete list of authors and institutions are published in Ecology: https://doi.org/10.1002/ecy.2663

Introduction

The superorder Xenarthra *(xenon* = stranger; *arthros* = articulations) is one of the four major clades of placental mammals (Madsen et al. 2001, Delsuc et al. 2002, Delsuc and Douzery, 2008). This group, which includes the orders Cingulata (armadillos) and Pilosa (anteaters and sloths), began its radiation in South America and later dispersed to Central and North America through the establishment of land bridges (Vizcaíno and Loughry, 2008). Molecular dating estimates Xenarthra split into Cingulata and Pilosa about 67 million years ago (Gibb et al. 2016). However, molecular dating estimate that the group originated earlier, approximately 105 million years ago (Springer 2003, Delsuc et al. 2004, Delsuc and Douzery 2008).

Xenarthrans encompass more than 150 described genera through Earth's history, including recently extinct taxa such as glyptodonts and giant ground sloths (Fariña 1996). This number decreased to only 13 extant genera, represented by 37 species (21 armadillos, six sloths, ten anteaters) classified in six families: Dasypodidae (long-nosed armadillos), Chlamyphoridae (fairy, hairy, three-banded, naked-tailed, and giant armadillos), Myrmecophagidae (lesser and giant anteaters), Cyclopedidae (pygmy anteaters), Bradypodidae (three-toed sloths), and Megalonychidae (two-toed sloths) (Abba et al. 2015, Gibb et al. 2016, Feijó and Cordeiro-Estrela 2016, Miranda et al. 2017, Feijó et al. 2018). They are distributed from the southeastern United States of America to southern Patagonia, with more diversity in South America (Gardner 2008, Vizcaíno and Loughry 2008, Taulman and Robbins 2014).

Each family has very distinctive morphological and ecological characteristics, making Xenarthra a unique and morphologically diverse group. The bodies of armadillos are covered by a carapace comprised of mobile articulated plates, which offers protection against predators (McDonough and Loughry 2008). Anteaters possess a protrusible tongue and have a long and tubular skull characterized by the complete loss of teeth, as well as large front claws used to feed on ants and termites (Wilson and Mittermeier 2018). Three-toed sloths have three long digits on each forelimb, whereas two-toed sloths only have two. Both are strictly arboreal and possess long claws that they use to hang on to tree branches. Most mammals have seven cervical vertebrae; however, the living sloths have a variable number: 5–8 in two-toed sloths (*Choloepus*) and 8–10 in three-toed sloths (*Bradypus*) (Buchholtz and Stepien 2009).

Besides their unique anatomical structures and ecology, the xenarthrans have an exceptional physiology, with a lower metabolism than expected from similar-sized mammals, and variable body temperature, characteristics that may be related to a low energy intake diet and the ingestion of non-nutritive material during feeding (McNab 1984). Their diets are mainly based on ants and termites (anteaters and armadillos), arthropods (armadillos), and leaves (sloths). The living sloths are arboreal, feeding on leaves from trees, vines, and lianas, and their low metabolism might assist in reducing the absorption of toxic leaves (Gilmore et al. 2001).

Armadillos use burrows as shelter and to buffer their body temperature from environmental variation (Attias et al. 2018), as well as the Anteaters (Camilo-Alves and Mourão, 2006; Mourão and Medri 2007). Additionally, xenarthrans play important ecological roles. Armadillos act as ecosystem engineers with many species associated with their burrows (Desbiez and Kluyber 2013, Aya-Cuero et al. 2017). Armadillos and anteaters impact ant and termite populations contributing, in turn, to the structure and diversity of plant communities (Rao 2000, Terborgh et al. 2006). Sloths are important components of the arboreal vertebrate biomass, being top prey for large raptors and contributing to the nutrient cycling of tropical forests. Their pelage is also the substrate to a diverse array of microorganisms and invertebrates (Montgomery and Sunquist 1978; Higginbotham et al. 2014).

According to the International Union for Conservation of Nature (IUCN), five xenarthran species are at some risk of extinction – *Bradypus pygmaeus, Bradypus torquatus, Myrmecophaga tridactyla, Priodontes maximus*, and *Tolypeutes tricinctus*. Another four are near threatened – *Cabassous chacoensis, Dasypus sabanicola, Tolypeutes matacus*, and *Zaedyus pichiy* – all showing decreasing population trends. Additionally, because of recent taxonomic revisions, three species of *Dasypus* and seven species of *Cyclopes* have pending conservation status evaluations (Feijó and Cordeiro-Estrela 2016, Miranda et al. 2017, Feijó et al. 2018). For all xenarthran species, the major threats are habitat loss resulting from fragmentation (Chiarello and Moraes-Barros 2014, ICMBio 2015), domestic and feral dog attacks, roadkill (Chiarello and Moraes-Barros 2014, Ascenção et al. 2017), subsistence hunting, illegal capture (ICMBio 2015), and fires (Miranda et al. 2014).

Despite the increasing interest in living xenarthran species (Diniz and Brito 2012, Superina et al. 2014), they remain a relatively understudied group when compared to other groups of mammals (Superina and Loughry 2015). Therefore, many aspects of their ecology are data deficient, making the correct assessment of population trends a difficult task. Conservation of xenarthrans is becoming more pressing as this taxon faces increasing negative impacts from anthropogenic actions. The Anteater, Sloth and Armadillo Specialist Group is a network of mammalogists and conservationists within the IUCN. Its main goal is to comprehend and provide information to aid in conservation action for those species. There are a growing number of specific xenarthran conservation initiatives, including rehabilitation, education, monitoring programs, and scientific research, supported by non-governmental organizations such as Fundacion Aiunau (www.aiunau.org/), Colombia; Asociación Panamericana para la Conservación (http://www.appcpanama.org/), Panama; Instituto de Pesquisa e Conservação de Tamanduás do Brazil; Projeto Tamanduá (http://www.tamandua.org/), Brazil; Projeto Bandeiras e Rodovias (www.tamanduabandeira.org), Brazil; Proyecto Iberá - Oso Hormiguero, (http://www.proyectoibera.org/especiesamenazadas_osohormiguero.htm), Argentina; Proyecto Hormigueros (https://www.cunaguaro.co/proyecto-hormigueros), Colombia; Programa Tatu-bola (http://tatubola.org.br/), Brazil; Armadillos de Chile (http://www.armadilloschile.cl/), Chile; The Sloth Conservation Foundation (https://slothconservation.com/), United Kingdom; and Projeto Tatu-canastra (www.icasconservation.org.br/o), Brazil. Nonetheless, relatively few species are considered in those programs (such as *Myrmecophaga tridactyla*, *Priodontes maximus*, *Cyclopes* sp. and *Tolypeutes tricinctus*).

Most xenarthran data are unpublished or have been published in the grey literature, including reports, Master and PhD theses in different languages, with low accessibility to the broad scientific community. In an effort to gather this dispersed and unavailable data in one place, we compiled published and unpublished data on xenarthrans, from the years 1855 to 2018, including occurrence and quantitative data. A total of 24,847 unique georeferenced records were assembled, ranging from the northern frontier of the Neotropics (Mexico, southern USA, and Caribbean regions), through to the austral south of South America (Chile, Argentina, Uruguay, and Paraguay). Neotropical Xenarthrans is by far the largest dataset ever assembled with respect to armadillos, sloths, and anteaters.

This work is part of the Neotropical, Atlantic, Brazil, and Amazonia series initiative, the objective of which is to make data on the biodiversity of these regions publicly available. Until now, the following data-papers of these series have been published: Atlantic Frugivory (Bello et al. 2017), Atlantic Camtraps (Lima et al. 2017), Atlantic Small Mammals (Bovendorp et al. 2017), Atlantic Bats (Muylaert et al. 2017), Atlantic Birds (Hasui et al. 2018), Atlantic Mammal Traits (Gonçalves et al. 2018), Atlantic Amphibians (Vancine et al. 2018), Non-volant mammals from the Upper Paraná River Basin (Gonçalves et al. 2018), Atlantic Primates (Culot et al. 2018), Jaguar GPS movement (Morato et al. 2018), and Brazil roadkill (Grillo et al. 2018).

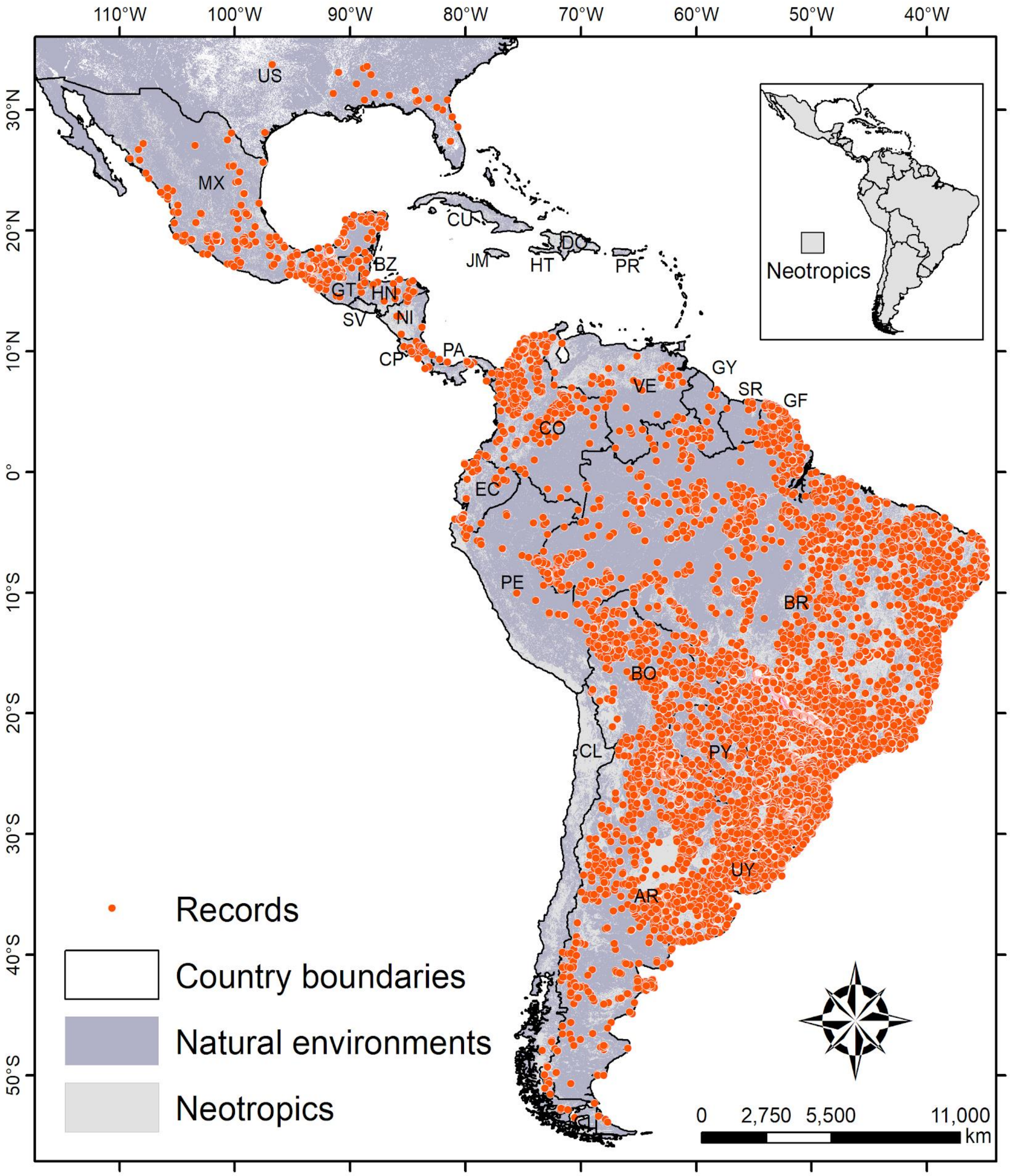

Fig. 1: Distribution of the Xenarthra records of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from years of 1855 to 2018; several populations may thus be extinct today.

METADATA

Class I - Data set descriptors

A. Data set identity

Tile: NEOTROPICAL XENARTHRANS: A DATASET OF OCCURRENCE OF XENARTHRAN SPECIES IN THE NEOTROPICS

B. Data set and metadata identification code

Suggested data set identity codes:

NEOTROPICAL_XENARTHRANS_QUANTITATIVE.csv

NEOTROPICAL_XENARTHRANS_QUALITATIVE.csv

NEOTROPICAL_XENARTHRANS_REFERENCES.csv

C. Data set description

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Abstract:

Xenarthrans – anteaters, sloths, and armadillos – have essential functions for ecosystem maintenance, such as insect control and nutrient cycling, playing key roles as ecosystem engineers. Because of habitat loss and fragmentation, hunting pressure, and conflicts with domestic dogs, these species have been threatened locally, regionally, or even across their full distribution ranges. The Neotropics harbor 21 species of armadillos, ten anteaters, and six sloths. Our dataset includes the families Chlamyphoridae (13), Dasypodidae (7), Myrmecophagidae (3), Bradypodidae (4), and Megalonychidae (2). We have no occurrence data on *Dasypus pilosus* (Dasypodidae). Regarding Cyclopedidae, until recently, only one species was recognized, but new genetic studies have revealed that the group is represented by seven species. In this data-paper, we compiled a total of 42,528 records of 31 species, represented by occurrence and quantitative data, totaling 24,847 unique georeferenced records. The geographic range is from the south of the USA, Mexico, and Caribbean countries at the northern portion of the Neotropics, to its austral distribution in Argentina, Paraguay, Chile, and Uruguay. Regarding anteaters, *Myrmecophaga tridactyla* has the most records (n=5,941), and *Cyclopes* sp. has the fewest (n=240). The armadillo species with the most data is *Dasypus novemcinctus* (n=11,588), and the least recorded for *Calyptophractus retusus* (n=33). With regards to sloth species, *Bradypus*

variegatus has the most records (n=962), and *Bradypus pygmaeus* has the fewest (n=12). Our main objective with Neotropical Xenarthrans is to make occurrence and quantitative data available to facilitate more ecological research, particularly if we integrate the xenarthran data with other datasets of Neotropical Series which will become available very soon (i.e. Neotropical Carnivores, Neotropical Invasive Mammals, and Neotropical Hunters and Dogs). Therefore, studies on trophic cascades, hunting pressure, habitat loss, fragmentation effects, species invasion, and climate change effects will be possible with the Neotropical Xenarthrans dataset.

D. Key words

Xenarthra, Pilosa, Cingulata, Biodiversity Hotspot, Neotropical region, Neotropical mammals, forest fragmentation, habitat loss

E. Description

To organize the dataset into the Neotropical realm, we selected all the countries and islands indicated in the literature as the Neotropical area and merged all polygons into a single shapefile. Our study extent includes the southern USA, Mexico, Central America, the Caribbean islands and South America (Figure 1). The dataset is comprised of 42,528 records (Figure 1; Table 1), including 40,794 (95.92%) with taxonomic certainty down to species-level identification, and 1,734 (4.08%) records of six genera with taxonomic uncertainty, with the majority of those records belonging to *Dasypus* sp. (1,359 records, 78.37%). In some cases, we taxonomically corrected some of those uncertain records, based on the distribution, on available literature, or on evidence material sent to us by the authors and after consulting specialists (more details, Table 2). When correction was not possible, we opted to maintain only the genus. A special case includes the genus *Cyclopes* sp. (240 records): at the beginning of this study, the genus was monospecific, *Cyclopes didactylus*. In the middle of the process, the taxon was subjected to a taxonomic review, which split the previously single species into seven (Miranda et al. 2017). To avoid misinterpretation of the data, we decided to maintain *Cyclopes* sp.

The two armadillo families appear most frequently in terms of both the number of records and species; Dasypodidae is responsible for the most records (n=14,790), followed by Chlamyphoridae (n=13,887), and they are represented by seven (two subspecies) and 13 species, respectively (Figures 3–5). Two of these species are the most common in the dataset (Figure 2): *Dasypus novemcinctus* (Dasypodidae, n=11,588) and *Euphractus sexcinctus* (Chlamyphoridae, n=7,325), both of which are species with wide distributions (Figures 3 and 4, respectively). Only one species, *Cyclopes* sp., represents the family Cyclopedidae (Figure 9; see the recent taxonomic review above). The other anteater species, Myrmecophagidae, is the third-most numerous, (n=11,349), represented by its three species *Myrmecophaga didactyla* (n=5,941), *Tamandua tetradactyla* (n=5,089), and *Tamandua mexicana* (n=319). The two sloth families are the least numerous, mainly the Megalonychidae, with only 414 records (Figure 7). *Bradypus variegatus*, from the family Bradypodidae, is the most-common sloth species, with 962 records (Figure 8). *Bradypus pygmaeus*, also from the family Bradypodidae, is the least-common species (n=12), both among sloth species and the dataset as a whole, reflecting its restricted insular distribution (Figure 8).

Approximately 99.94% of the records are from native species, and only 0.06% are from introduced (*Priodontes maximus*) and potentially introduced (*Bradypus* spp.) species. According to the most-recent IUCN Red List classification (IUCN 2018) and considering taxonomically confirmed species and the genus *Cyclopes*, 29,731 records (72.45%, 14 species and the genus *Cyclopes*) are from Least Concern (LC) species, 2,020 (4.92%, five species) are from Near Threatened (NT) species, 8,451 (20.6%, four species) are from Vulnerable (VU) species, and 12 records (0.03%, one species) are from Critically Endangered (CR) species. An additional 343 records (0.84%, four species) are from Data Deficient (DD) species, and another 477 records (1.16%, two species) are from Not Evaluated (NE) species (Table 1).

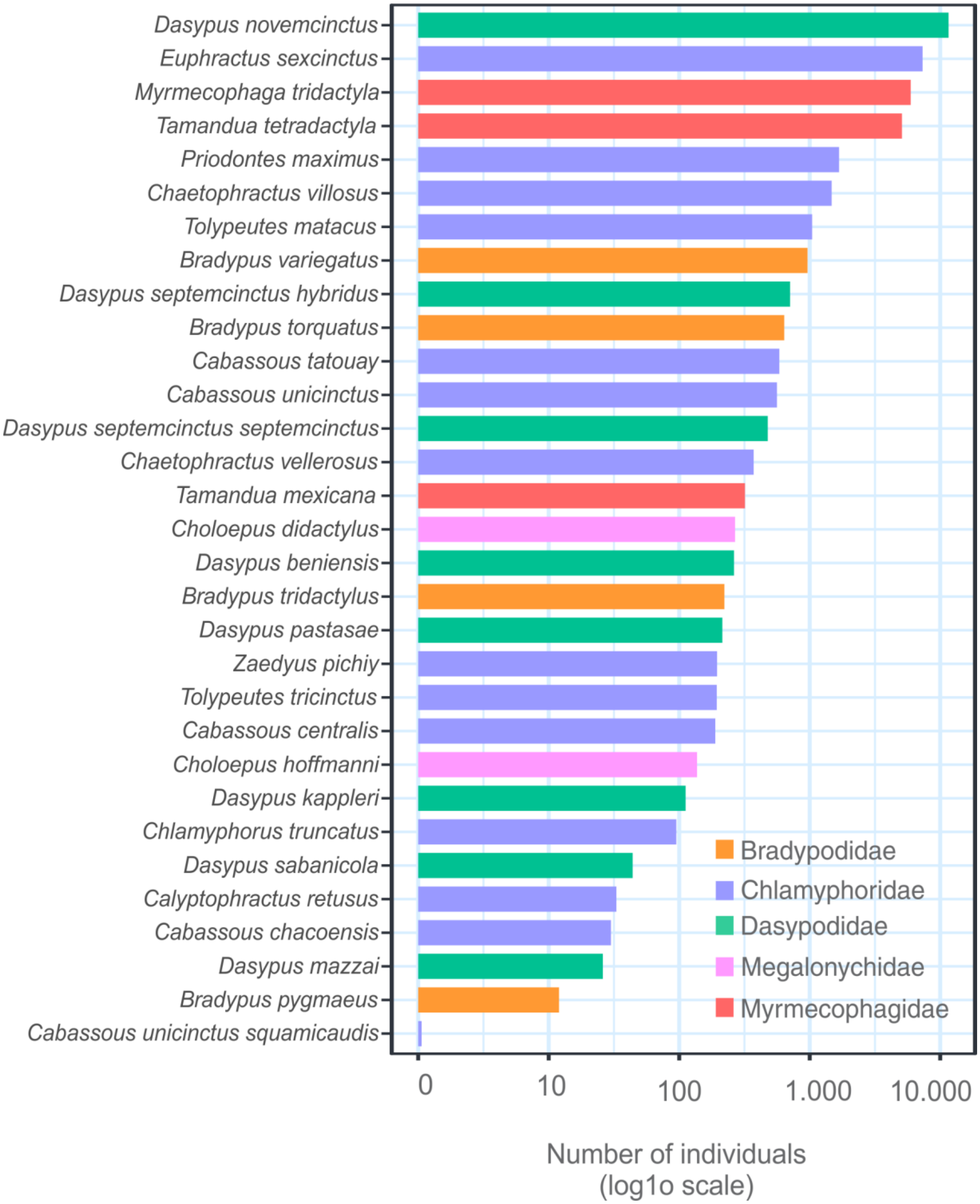

Fig. 2: Number of records per species of armadillos, anteaters and sloths within NEOTROPICAL XENARTHRANS dataset. Species with unconfirmed identification are not represented. Each color refers to a xenarthran family.

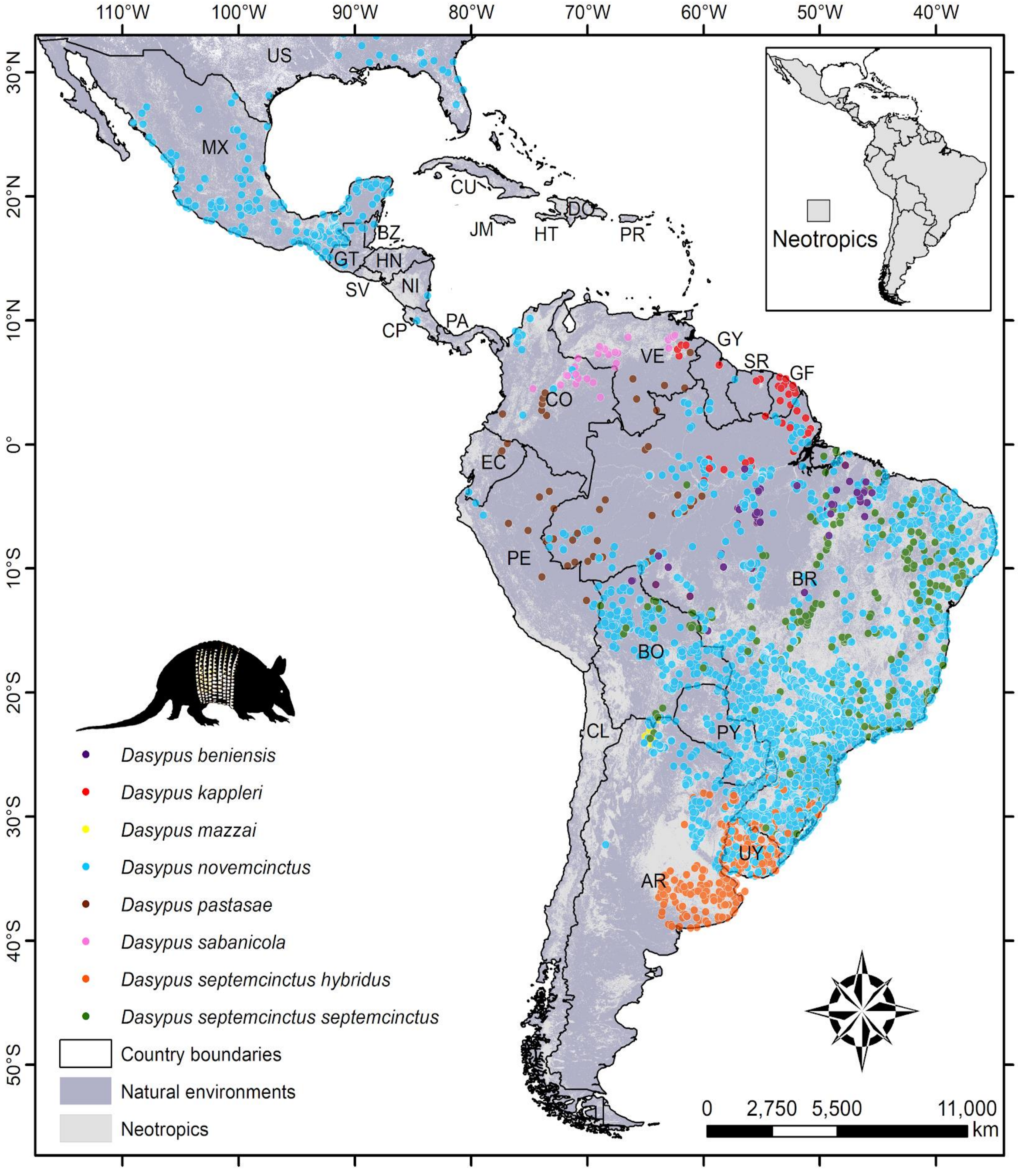

Fig. 3: Distribution of the records of the family Dasypodidae of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1896 to 2018, thus several populations may now be extinct. Unconfirmed species with “sp.” are not included.

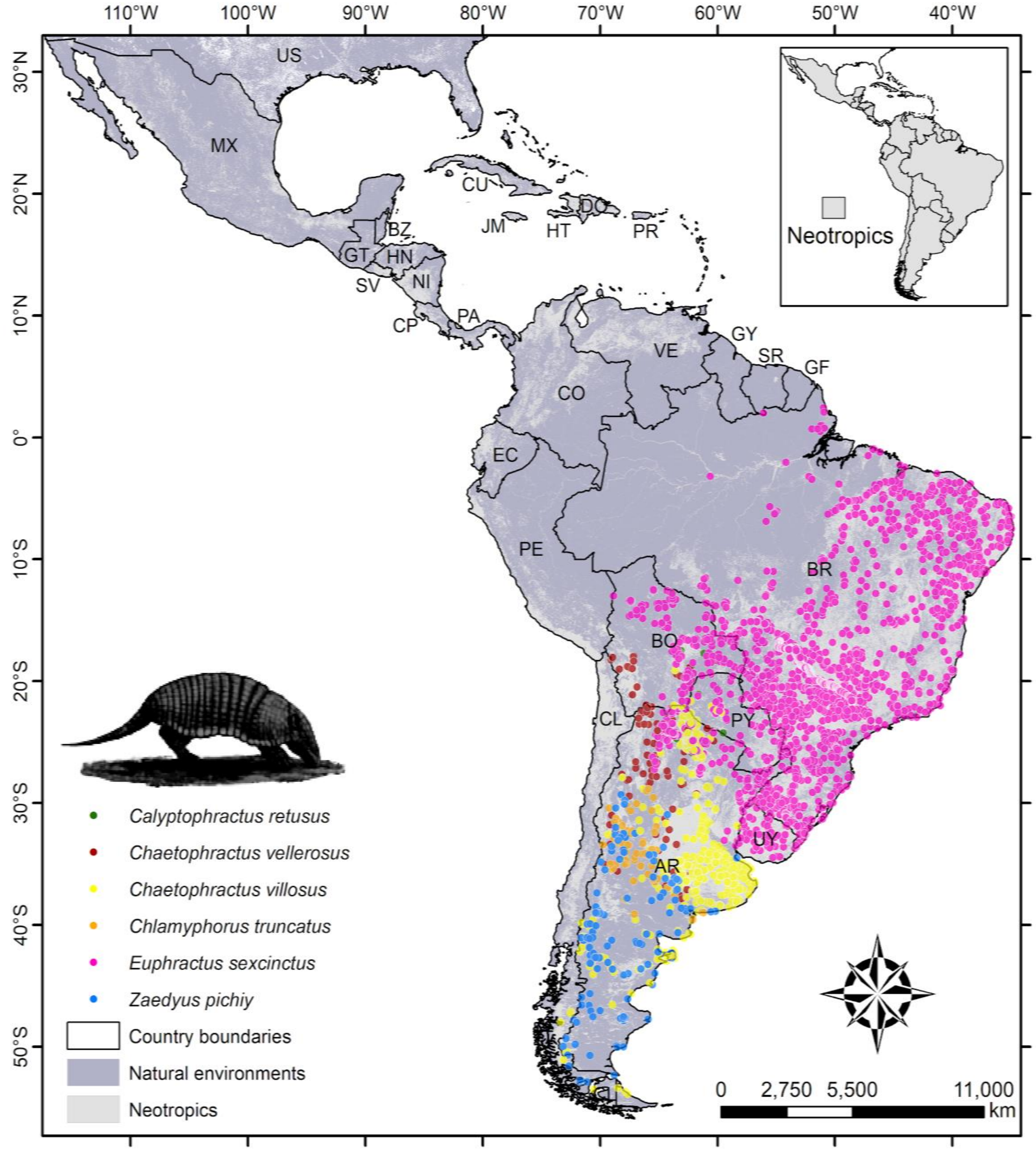

Fig. 4: Distribution of the records of the family Chlamyphoridae (Subfamilies Chlamyphoniae and Euphractinae) of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1910 to 2018, thus several populations may now be extinct. Unconfirmed species with "sp." are not included. Additional species in this family are shown in Figure 5.

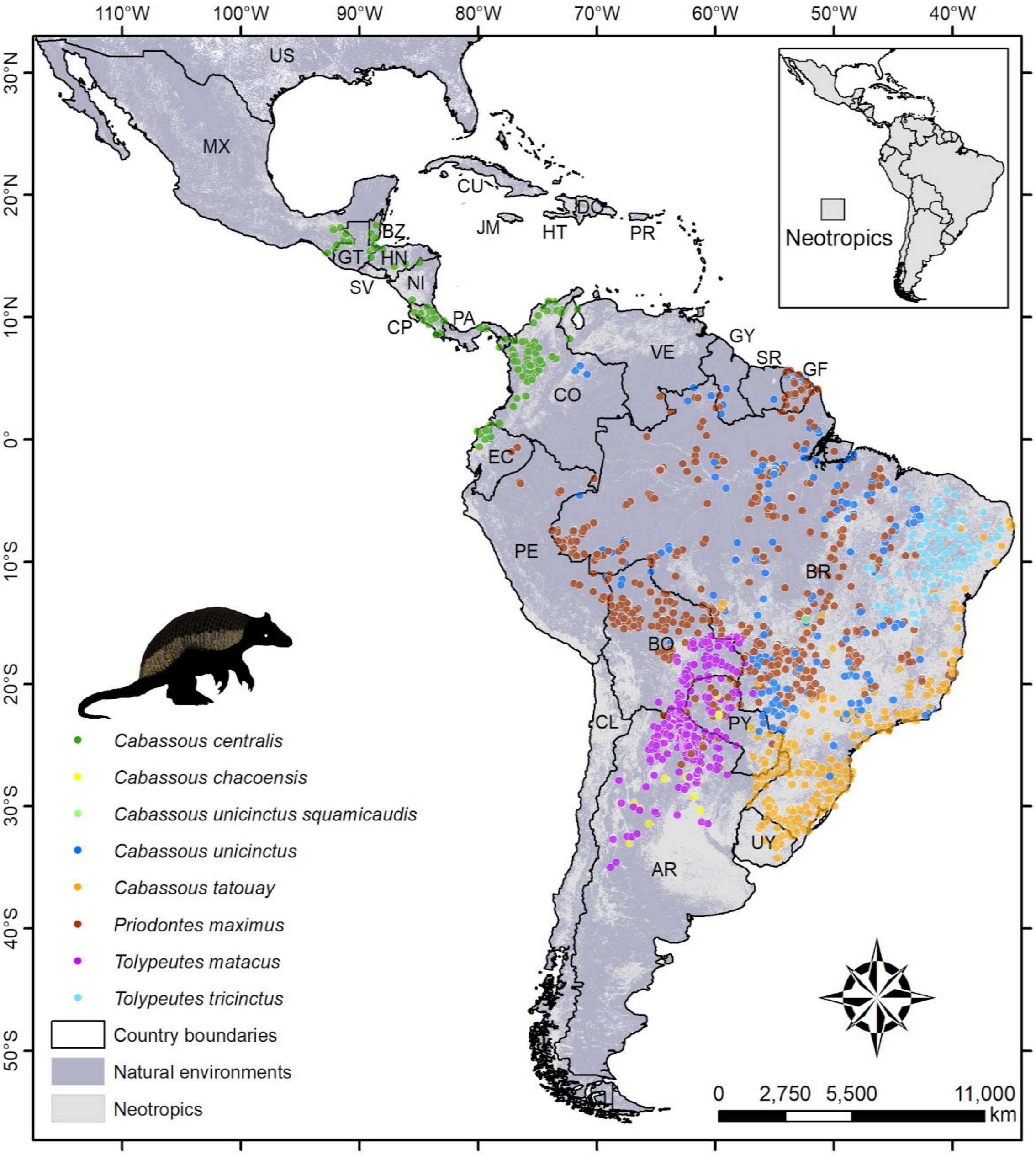

Fig. 5: Distribution of the records of the family Chlamyphoridae (Subfamily Tolypeutinae) of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1855 to 2018, thus several populations may now thus be extinct. Unconfirmed species with "sp." are not included.

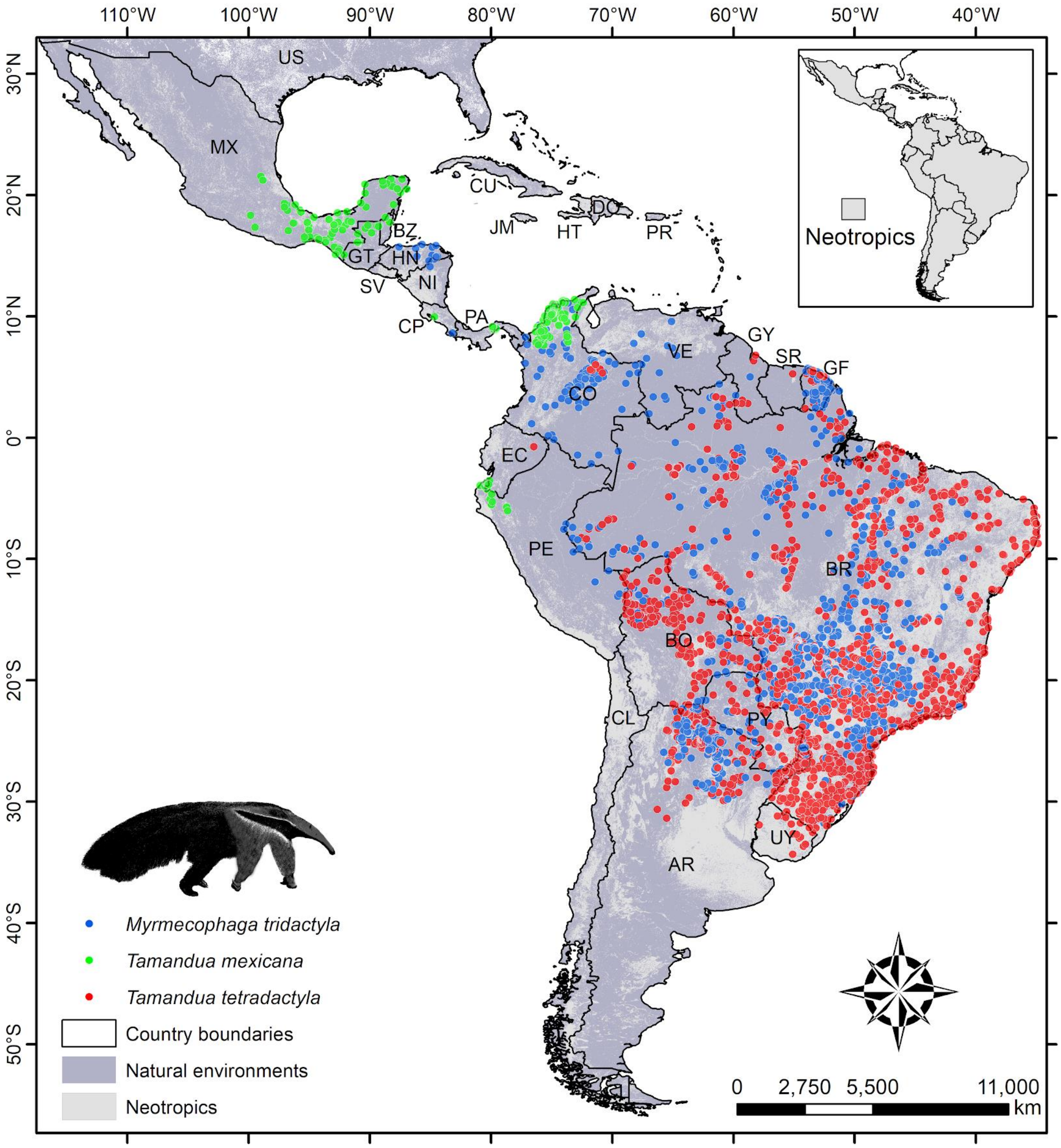

Fig. 6: Distribution of the records of the family Myrmecophagidae of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1899 to 2018, thus several populations may now be extinct. Unconfirmed species with "sp." are not included.

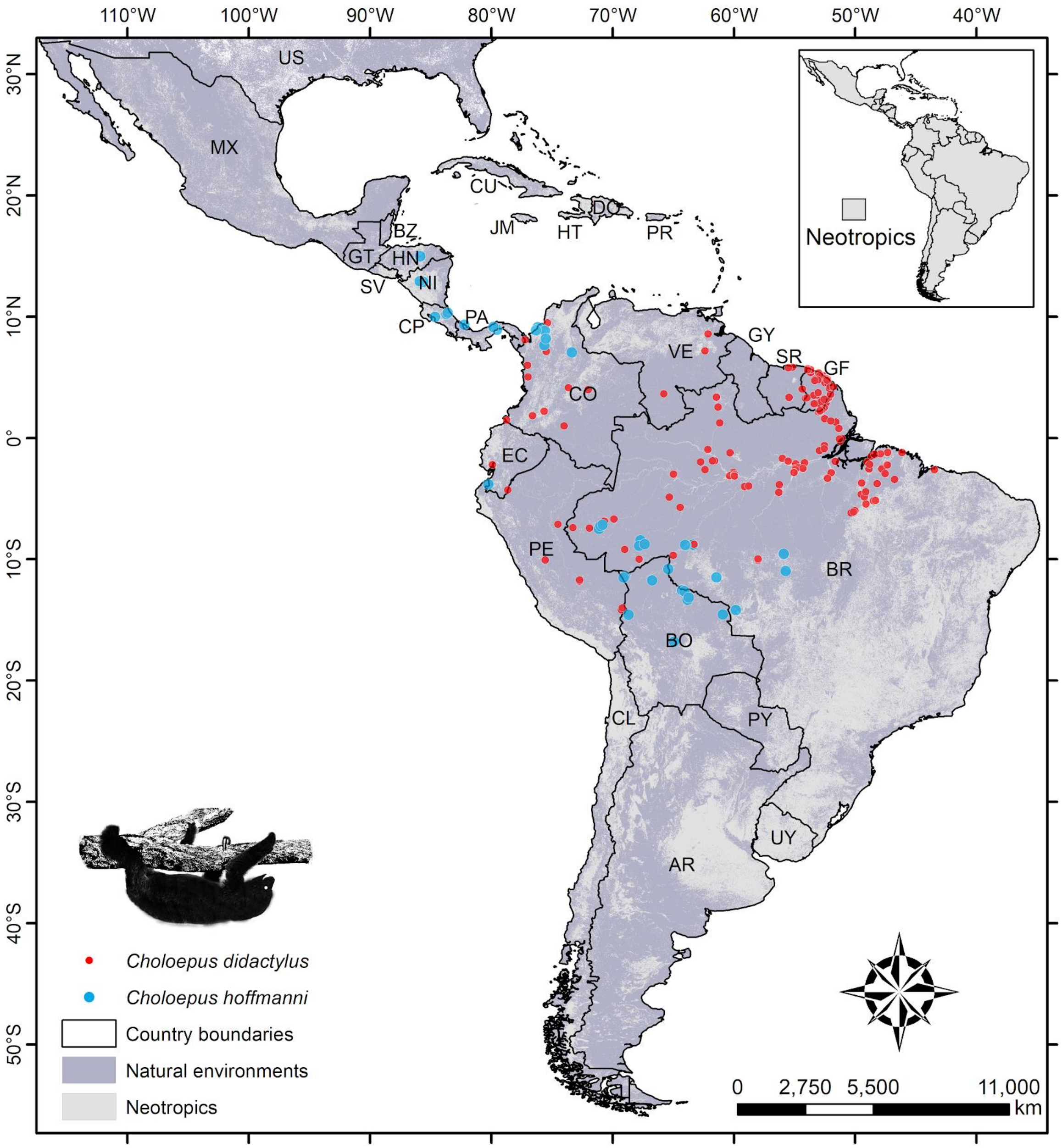

Fig. 7: Distribution of the records of the family Megalonychidae of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1919 to 2017, thus several populations may now be extinct. Unconfirmed species with "sp." are not included.

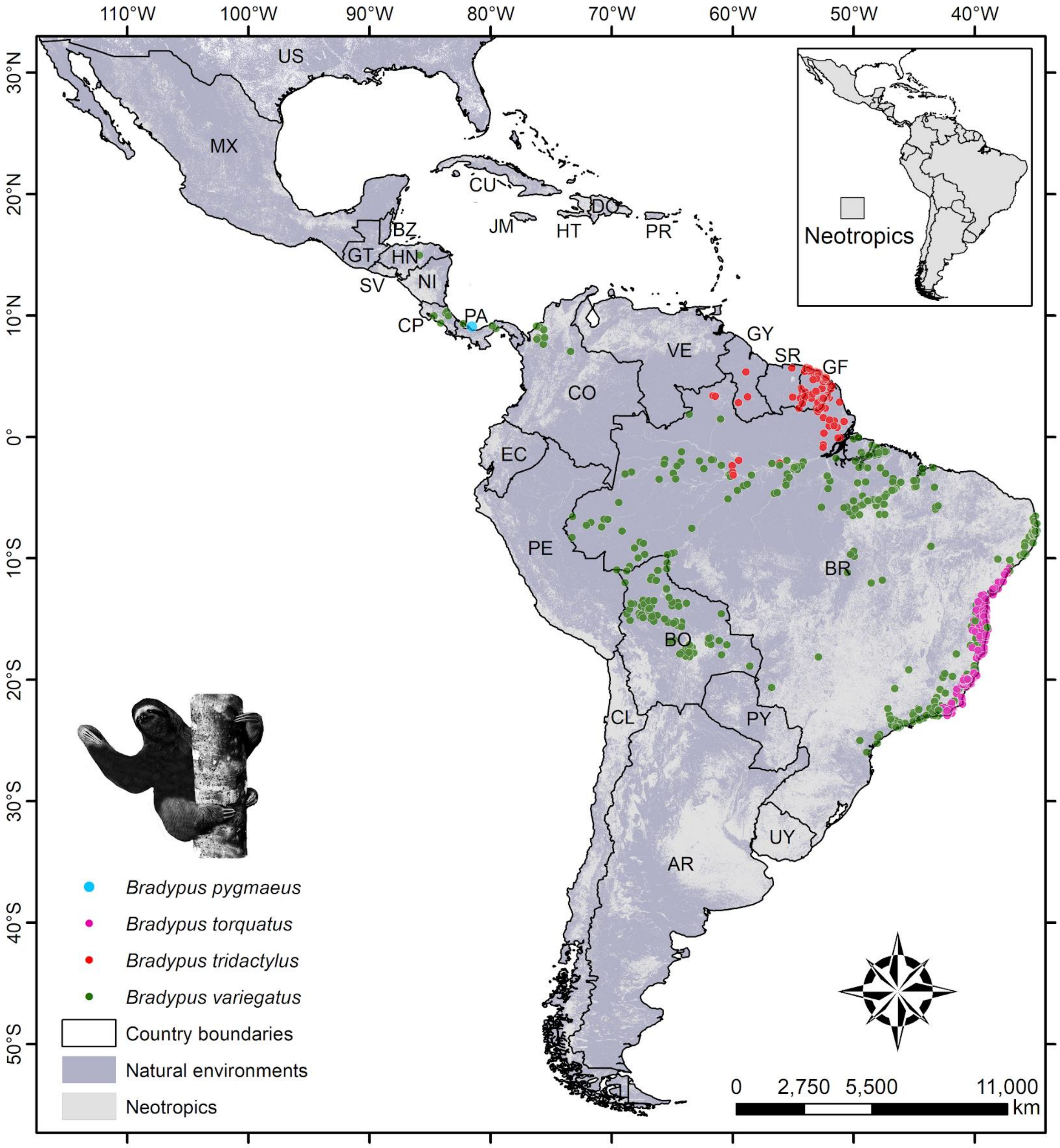

Fig. 8: Distribution of the records of the family Bradypodidae of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1911 to 2018, thus several populations may now be extinct. Unconfirmed species with "sp." are not included.

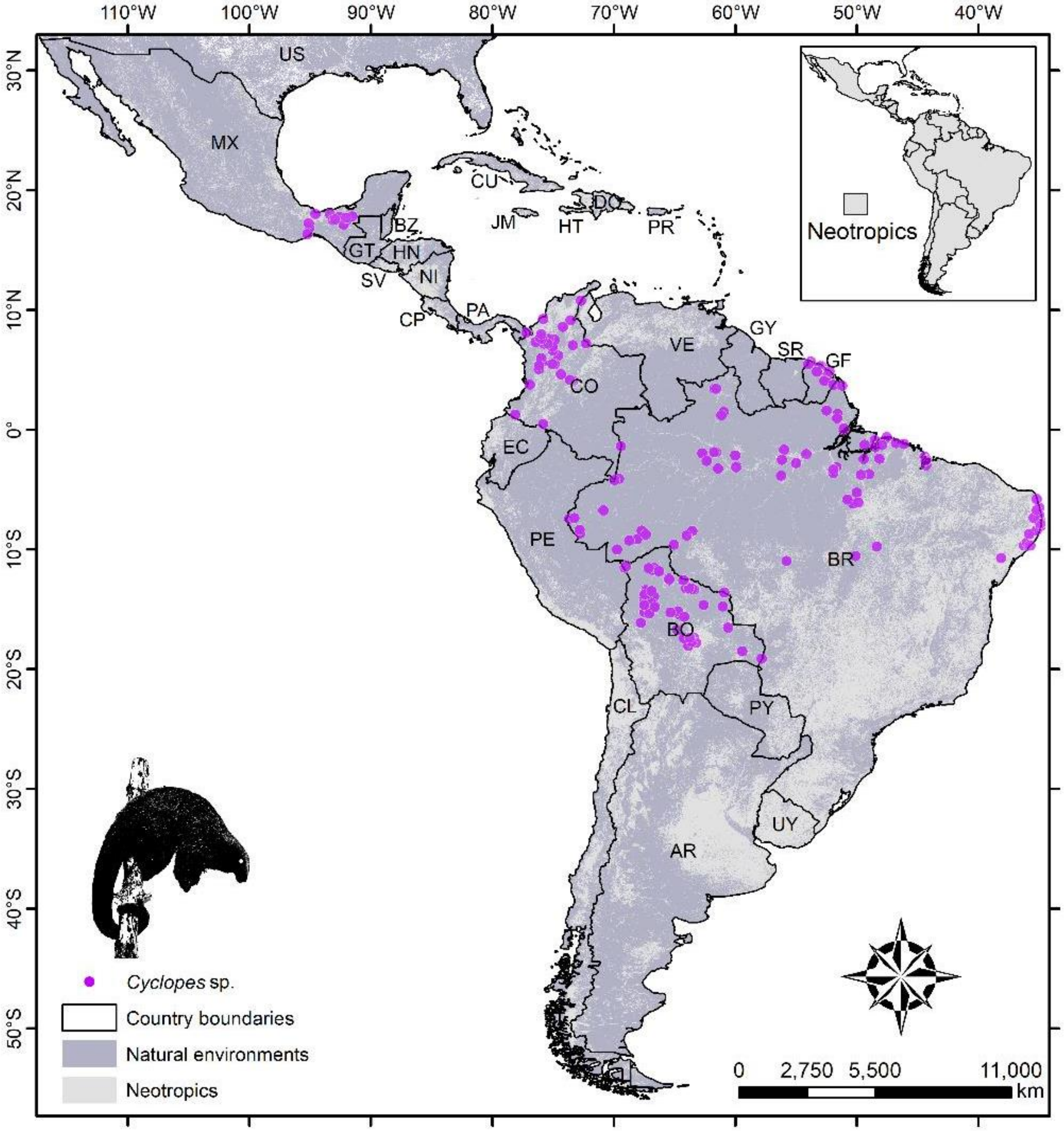

Fig. 9: Distribution of the records of the family Cyclopedidae of the NEOTROPICAL XENARTHRANS dataset in Neotropical realm. Records are from 1931 to 2017, thus several populations may now be extinct.

Of the 19 different methods identified, camera trapping most-frequently documents xenarthran species, responsible for 13,866 of the records (38.42%), especially when used in conjunction with other methods (Table 3). But even when used as the sole sampling method, camera trapping is still the most-frequently method (32.4%). Roadkill is the second most-successful method, accountable for 14.6% of the records. Active search is mostly used in combination with other methods, followed by interview. Other methods include line transect, vestiges, museum collections, telemetry, bibliographical survey, live trapping, sand plots, track plots, car monitoring, and opportunistic encounters (Table 3). About 83.90% of the records correspond to presence-only data, 10.02% to presence-absence data, and 6.08% to quantitative data.

According to our database, studies on Xenarthra have been increasing in recent years (Figure 10). Most of the data (< 10,000 records) were collected between 2010 and 2014, and this is the major period for most species (Figure 11). There are less than 100 records from a less-studied period from 1970 to 1974 (Figure 10) and 24 of the 31 species has records on this period (Figure 11).

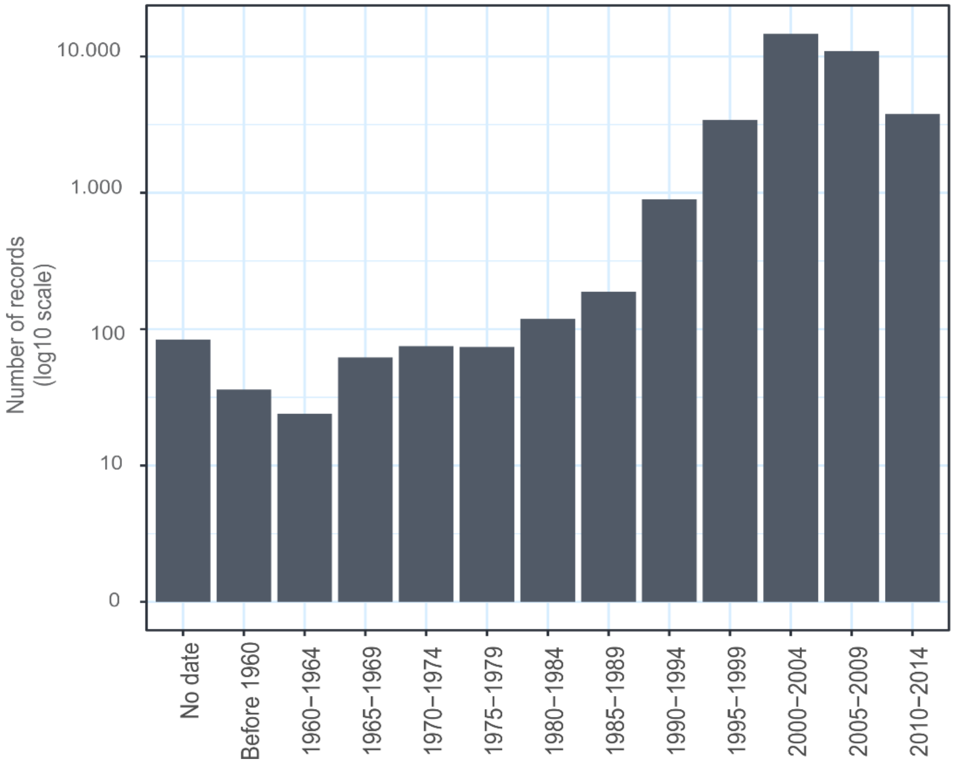

Fig.10: Number of xenarthran records over time.

Fig.11: Proportion of the records over time for each xenarthran species.

The records are concentrated in some regions along the Neotropical realm (Figure 1). On one hand, Bolivia, southeastern Brazil, northern Argentina, Mexico, and Guiana hold the most records. Alternatively, some regions, such as the Amazon - mainly due the low accessibility to this area (Figure 1) - and some countries in South and Central America such as Chile, Peru, Costa Rica, and Panama, have few records - mainly due the lack of contact with researchers of theses area; Figure 1; More details - Class II, Topic C: Data Limitations and Potential Enhancements).

To characterize the environmental conditions of the Neotropical area, we overlapped all data locations with the environment maps, using the function “extract” of the raster package in the R program. The raster files corresponding to temperature and precipitation values were obtained from the WorldClim database 1.4 (http://www.worldclim.org/version1). The altitude of most records is below 1,000 m, with a peak around lowland regions (between 200 and 300 m; Fig. 12). The average annual temperature of the sampling sites varies mainly between 20 and 27° C, and few sampling records show temperatures below this range (which is expected for the neotropical region; Fig. 12). Lastly, annual precipitation varies mostly between 1,500 and 2,000 mm, and few sampling sites show values above 2,000 mm. Elevation data were obtained from the digital elevation map from the CGIARCSI database version v4.1 (http://www.cgiar-csi.org/data/srtm-90m-digital-elevation-database-v4-1). The digital elevation model has 1 km of spatial resolution and is available in a unique file in “.tif” format.

Natural environment layers for the maps were obtained from the land-use map from the http://due.esrin.esa.int/page_globcover.php. We used the description of Appendix III to identify only the areas described as natural areas that overlapped with the Neotropics realm boundary (http://due.esrin.esa.int/files/GLOBCOVER_Products_Description_Validation_Report_I2.1.1.pdf). The legend codes classified as natural environment areas were: 40, 50, 60, 70, 90, 100, 110, 120, 130, 140, 150, 160, 170, and 180.

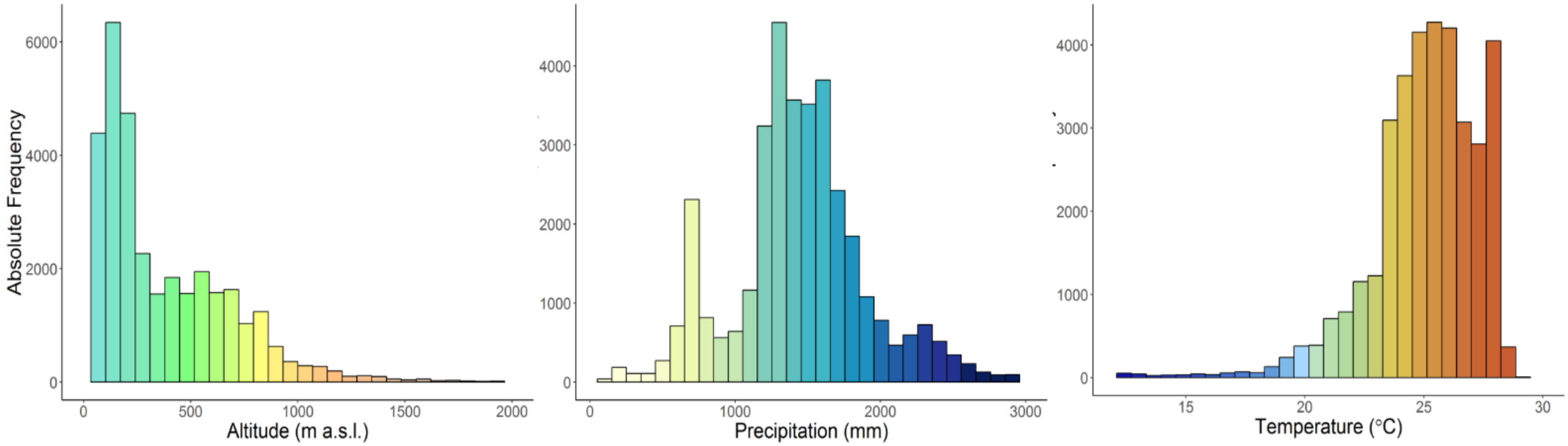

Fig. 12: Variation of altitudes, annual mean temperature, and annual precipitation of the study sites in the NEOTROPICAL XENARTHRAN dataset.

Class II - Research Origin Descriptors

A. Overall project description

Identity

A compilation of xenarthran (armadillos, anteaters, and sloths) occurrence and quantitative data in Neotropical Regions.

Originator (s)

The Neotropical Xenarthrans project was coordinated by Paloma M. Santos at the Universidade Federal de Minas Gerais (UFMG) and all authors contributed in the assembly of the database. This research is part of the NEOTROPICAL, ATLANTIC, BRAZIL, AMAZONIA, and NEOTROPICAL series, which is led by Mauro Galetti and Milton Cezar Ribeiro, Universidade Estadual Paulista (UNESP).

Period of the study

Years of source publications range from 1855 – 2018

Objectives

With this paper, we aimed (i) to compile all raw and unpublished xenarthran data collected in recent decades by researchers, projects, and consultants; (ii) to compile the information available in the Portuguese, Spanish, and English literature regarding xenarthran inventories in Neotropical regions, both occurrence only, population, and community data, including the corresponding methods and sampling efforts.

Abstract

Same as above

Source(s) of funding

The compilation of this dataset was supported by grants, fellowships, and scholarships from the Agence Nationale de la Recherche (National Research Agency from France) CEBA, ref.

ANR-10-LABX-25-01, Agencia Nacional de Promoción Científica y Técnica de Argentina (National Agency for Scientific and Technical Promotion of Argentina, AGENCIA), Alexander Koenig Society, Amazonas Distribuidora de Energia S.A, Conselho Nacional de Desenvolvimento Científico e Tecnológico (Brazilian Research Council, CNPq) numbers 307016/2015-3, 106186/79, 300982/82, 153423/2016-1, 301061/2007-6, 425746/2016-0, 207195/2014-5, 457434/2012-0, 563134/2010-0, 457497/2012-2, 457491/2012-4, 401887/2014-6, 459735/2014-4, 161089/2014-3, 301652/2015-5, 484350/2013-7, 503372/2014-5, 150123/2018-3, 472802/2010-0, 141057/2008-4, 158990/2014-5, 303006/2014-5, 141263/2016-4, 306392/2013-5, 140730/2010-9, 140039/2018-1, 216938/2014-7, 457458/2012-7, 141041/2008-0, 307781/2014-3, 307303/2017-9, 306695/2015-4 and 421361/2017-4, Casadinho/PROCAD 552198/2011-0, Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (Coordination for the Improvement of Higher Education Personnel, CAPES) numbers 1840/2015, 817737/2015, 1537137, 303757/2012-4, 080410/0, 004/2012 and 3261/2013, PNPD/CAPES number 283-60/2013, COFECUB/CAPES numbers 88887.130202/201701 and 88887.143361/201700, American Society of Mammalogists, Anglo American, Associação Comunidade Waimiri Atroari, Associação de Defesa Etnoambiental Kanindé, BioFaces, Blue Moon, Brehm Foundation for International Bird Conservation, BRVias, Cambuhy Agrícola Ltda., Chinese Academy of Sciences President’s International Fellowship Initiative number 2018PB0040, Christian Rossell, Conservation International Brazil (CI Brazil), Cleveland Metropolitan Zoo, Club de amigos del yaguareté, CNEC WorleyParsons resources and energy, COFUSA Forestry company, Departamento Administrativo de Ciencia, Tecnología e Innovación, Estudios de Doctorado en el Exterior (COLCIENCIAS), Comisión Nacional Forestal, Secretaría de Medio Ambiente y Recursos Naturales, Comisión Nacional de Áreas Naturales Protegidas-Reserva de la Biosfera Pantanos de Centla; Área de Protección de Flora y Fauna Laguna de Términos, Companhia de Concessão Rodoviária de Juiz de Fora – RJ (CONCER), Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Consejo Nacional de Ciencia y Tecnología de México (CONACYT), Conservation International's Margot Marsh Primate Action Fund numbers 1001257, 6002856, Conservation Leadership Programme number F02304217, Conservation Program of Endangered Species (PROCER-Mexico) of the National, Commission of Protected Areas (CONANP-Mexico), Conservation, Food and Health Foundation, Consorcio del Mayab, Cota-Robles Graduate Fellowship, China Three Gorges Corporation (CTG-Brazil), Darwin Initiative, DICE Small Grant Program, Durrell

Wildlife Conservation Trust, Earthwatch Institute, El Colegio de la Frontera Sur, Unidad Campeche ECOSUR, Fauna and Flora International, Fazenda Barranco Alto and Fazenda Vera Lúcia, Fundação para a Ciência e a Tecnologia (FCT)-Portugal (SFRH/BD/40638/2007), Fibria Celulose S.A., Fundação de Amparo à Pesquisa do Estado do Amazonas (FAPEAM), Fundação de Amparo à Pesquisa do Estado de Goiás (FAPEG) number 11/2013, Fundação de Amparo à Pesquisa e ao Desenvolvimento Científico e Tecnológico do Maranhão (FAPEMA), Fundação de Amparo à Pesquisa do Estado de Minas Gerais (FAPEMIG) numbers CRA- RDP-00104-10; CRA 00357/07; APQ 01145-10, APQ-03315-16 and APQ-00604-17, Fundação de Amparo à Pesquisa do Estado do Rio Grande do Sul (FAPERGS), Fundação Carlos Chagas Filho de Amparo à Pesquisa do Estado do Rio de Janeiro (FAPERJ) numbers E-26/102.960/2012 and E26 /201.267/ 2014, Fundação de Amparo à Pesquisa e Inovação do Espírito Santo (FAPES) numbers 0607/2015 and 0510/2016, Fundação de Amparo à Pesquisa do Estado da Bahia (FAPESB), Fundação de Amparo à Pesquisa e Inovação do Estado de Santa Catarina (FAPESC) numbers 06/2016 and 189/2016; Programa de Apoio à Pesquisa (PAP/FAPESC) number 2017TR744, Fundação de Amparo à Pesquisa do Estado de São Paulo (FAPESP) numbers 2013/04957-8, 2012/14245-2, 2013/18526-9, 2014/16320-7, 2013/50421-2, 2016/11595-3, 2014/01986-0, 2015/18381-6, 2014/23095-0, 2014/23132-2, 2013/50421-2, 2014/10192-7, 2015/25742-5, 2015/11521-7, 2011/22449-4, 2014/01986-0, 2013/10029-6, 2014/24921-0, 2016/16433-1, 2015/22844-1, 2017/12925-0, 2014/50434-0, 2014/09300-0, 2013/50421-2, 2015/10778 and 2014/16320-7, Fundação de Amparo à Ciência e Tecnologia do Estado de Pernambuco (FACEPE), Fundação de Amparo ao Ensino e Pesquisa (FAEP), Fundação de Tecnologia do Estado do Acre (FUNTAC), Fundação Estadual do Meio Ambiente e Recursos Hídricos de Roraima number 01683/09-01, Fundação Grupo Boticário de Proteção à Natureza, Fundación Vida Silvestre Argentina, Fundação de Desenvolvimento Científico e Cultural (FUNDECC) , Fundação de Apoio ao Desenvolvimento do Ensino, Ciência e Tecnologia do Estado de Mato Grosso do Sul (FUNDECT) numbers 0223/2014 and 06/2016, Fundo Brasileiro para a Biodiversidade – Tropical Forest Conservation Act agreement (FUNBio/TFCA), Global Heritage Fund, Grupo de Ecología de Paisaje y Medio Ambiente de la Universidad de Buenos Aires, Grupo Zema, Hamerton Zoo Park, Heinrich Hertz Foundation, Hotel Belmond, Idea Wild, Instituto Estadual de Florestas (IEF/MG) number 2101010400410, Ingenieros Civiles Asociados división Infraestructura (ICAi), Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA), Instituto Chico Mendes de Conservação da Biodiversidade

(ICMBio), International Foundation for Science, International Paper Co. of Brazil, Instituto de Pesquisas e Estudos Florestais (IPEF), Kirschbaum Foundation, Lincoln Park Zoo, Liz Claiborne and Art Ortenberg Jaguar Small Grants (Panthera Foundation), Lwarcel Celulose Ltda, Max Planck Institute, Ministerio de Ecología y Recursos Naturales de la Provincia de Misiones, Ministério Público de Minas Gerais, Ministry of Culture and Science of North Rhine-Westphalia , Mohamed bin Zayed Species Conservation Fund numbers 12055114 and 0925815, National Council of Science and Technology of Mexico, National Geographic Society grant number W314-14, National Institute for Science and Technology in Wetlands (INAU / UFMT), National Science Foundation (NSF) Pre-doctoral Fellowship grant number 0709598, Overbrook, Programa de auxílio ao Pesquisador (PAPESQ/UNIFAP) number 015/2015, Pays de la Loire - Strategies Internationales (CASEST), Pesquisas Ecológicas de Longa Duração (PELD/CNPQ) numbers 88887.140649/2017-00 and 441278/2016-7 (JSS), PELD Parque Nacional da Serra da Bodoquena (PELD/PNSB), People’s Trust For Endangered Species (PTES), President's International Fellowship Initiative grant number 2018PB0040, Primate Conservation Inc. number 1158, Projeto de Conservação e Utilização Sustentável da Diversidade Biológica Brasileira do Ministério do Meio Ambiente (PROBIO/MMA), Programa Áreas Protegidas da Amazônia (ARPA), Programa Beca, Programa Clima, Naturaleza y Comunidades en Guatemala (CNCG), Programa das Nações Unidas para o Desenvolvimento (PNUD) , Programa de Concessão de Bolsa de Incentivo ao Pesquisador da Universidade Estadual de Goiás (PROBIP/UEG) number 009/2016, Programa Nacional de Incentivos a Investigadores, Projeto Estrada Ecológica and Novo Colégio, Projeto Onçafari, Pró-Reitoria de Pesquisa e Pós-Graduação da Universidade Estadual de Santa Cruz–(PROPP/UESC) number 00220.1100.1735, Pró - Reitoria de Assuntos Estudantis da Universidade Federal do Rio Grande do Sul (PRAE, UFRGS), Pró -Vida Brazil, Rainforest Alliance, Red de Macro Universidades de América Latina y el Caribe and Santander Bank , Resource Award, Ridgeway Trust , Royal Ontario Museum (ROM Governors), Rufford Foundation grants number 18442 -1, 12658 -1, 20144 -2 and 21498 -1, San Diego Zoo Global, Segré Fondation, Secretaria do Meio Ambiente (SEMA), Secretaría de Ciencia y Técnica, Serviço Social do Comércio (SESC), Sistemas Estratégicos para la Gestión Ambiental (SEGA), Sistema Nacional de Pesquisa em Biodiversidade (SISBIOTA/CNPq) numbers 563134/2010 -0 and 563216/2010 - 7, Smithsonian Tropical Research Institute, Sophie Danforth Conservation Biology Fund, SOS Mata Atlântica Foundation, Studienstiftung des Deutschen Volkes, Superintendência da Zona Franca de Manaus

(SUFRAMA), Suzano Papel e Celulose S.A., The Ashoka Foundation, The Conservation, Food and Health Foundation, The Nature Conservancy, The Rolex Awards, The Scott Neotropical Fund of the Lincoln Park Zoological Society, The Species Survival Fund from Wildlife Trust, The Whitley Awards, The Woodland Park Zoo, Tijoá Energy, Tropical Ecology Assessment and Monitoring (TEAM), United States Agency for International Development (USAID), United States Department of the Interior (USDOI), Universidade Vila Velha, University of California, University of Puerto Rico, Unidad para el Cambio Rural (UCAR), Ministerio de Agroindustria, Argentina (PIA) numbers 2011/10102 and 2014/14061, Universidad Nacional de San Luis, Facultad de Química, Bioquímica y Farmacia., Usina Caeté, Usina Coruripe LTDA, Wildlife Conservation Society (WCS) Bolivia, WCS Brazil, WCS Guatemala, WCS Jaguar Conservation Program, WCS Research Fellowship Program Grant , Wild Felid Legacy Scholarship, Wildlife Trust, World Wide Fund for Nature (WWF) United States of America, WWF Brazil, WWF EFN Scholarship Program, Zoological Gardens of Dortmund and Cologne, Zoological Society of London; EDGE of Existence (EDGE , ZSL), Zoological Society of San Diego, Consórcio Capim Branco Energia (CCBE). MCR thanks FAPESP (process #2013/50421-2), CNPq (processes # 312045/2013-1; #312292/2016-3), and PROCAD/CAPES (project # 88881.068425/2014-01) for their financial support.

B. Specific Subproject description

Site Description

The target area comprises all the Neotropical realm and part of North America (Mexico and southeastern United States). We included those areas to cover the entirety of Xenarthra's distribution. The Neotropical region is one of the eight biogeographic terrestrial realms (i.e., "ecozones") of the Earth's surface (Olson and Dinerstein 1998, Olson et al. 2001). It extends from parts of Mexico, southeast USA (southern Florida and coastal central Florida), Central America, and Caribbean countries, to the austral distribution of South America. Including the Central USA and the highlands of Mexico, the coordinates extend from 30ºN to 50ºS, and 110ºW to 40ºW, encompassing approximately 51% of the American continent territory – about 21 million km² of territorial extension - with most parts located in South America. The vegetation type is composed of different forests, savannas, mangroves, chaco, sandy vegetation, xeric vegetation, altitudinal open and rocky fields, cloud forest, floodplains, freshwater marshes, Andean and lowland environments, etc. The region includes more than half of the American continent population (approximately 55%), residing mainly in important and large metropolitan areas, such as São Paulo, Mexico City, Buenos Aires, and Rio de Janeiro (IBGE 2018, INEGI 2015, INDEC 2010). These figures suggest a strong and severe anthropogenic impact on natural areas, and consequently on biodiversity. The region has a high rate of endemism and a unique but threatened biodiversity. Therefore, the target area includes seven of the 35 world biodiversity hotspots: the Cerrado, the Atlantic Forest, Tropical Andes, Tumbes-Choco-Magdalena, Valdivian Forests, Mesoamerica and Caribbean Islands (Myers et al. 2000, Mittermeier et al. 2011), beyond the Amazon Forest, one of the largest and most important tropical forest in the world.

Data Compilation

We created a collaborative network of researchers who had collected and published data on xenarthran occurrence and community composition in the Neotropics and invited everyone to contribute to this NEOTROPICAL XENARTHRANS data-paper (i.e., their data would be added to our dataset). In addition, we collected occurrence data from the following museums and institutions: Museu Nacional/Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil (MNRJ), Museu de Zoologia da Universidade de São Paulo, São Paulo, Brazil (MZUSP), Centro de Primatologia do Rio de Janeiro, Rio de Janeiro, Brazil (CPRJ), Museu da Fauna, Rio de Janeiro, Brazil (MFRJ), Museu Paraense "Emílio Goeldi", Pará, Brazil (MPEG), Fundação Zoobotânica do Rio Grande do Sul (FZBRS), Museu de História Natural do Capão da Imbuia, Paraná, Brazil (MHNCI), Centro Universitário Barra Mansa, Rio de Janeiro, Brazil (CUBM), Pontifícia Universidade Católica do Rio Grande do Sul, Rio Grande do Sul, Brazil (PUC-RS), Universidade Federal da Paraíba (UFPB), Paraíba, Brazil, Universidade Federal do Rio Grande do Norte (UFRN), Rio Grande do Norte, Brazil, Universidade Federal de Viçosa (UFV), Minas Gerais, Brazil, Universidade de Brasília (UnB), Brasília, Brazil, Universidade Federal de Santa Catarina (UFSC), Santa Catarina, Brazil, Museu Arquidiocesano Dom Joaquim, Santa Catarina, Brazil (MADJ), Museu do Colégio Agrícola de Camboriú Gert Hering, Santa Catarina, Brazil (MCAGH), Museu do Seminário Coração de Jesus, Santa Catarina, Brazil (MSCJ), Museu de Ciências Naturais, Rio Grande do Sul, Brazil (MCN), Museu de História Natural, Lisbon, Portugal (MHNL), Museu Bocage, Lisbon, Portugal (MBL), Museo de Historia Natural de Valparaíso, Chile (MHNV), Museu Argentino de Ciências Naturais Bernardino Rivadavia, Argentina (MACN), Museum Zoologicum Bogoriense (MZB), American Museum of Natural

History, New York (AMNH), Field Museum of Natural History, Chicago (FMNH), British Museum (Natural History), London (BMNH), Smithsonian Institution National Museum of Natural History, Washington, D. C. (USNM), and Kansas University Natural History Museum, Lawrence, Kansas (KUNHM).

After the first round of data compilation, we performed a systematic review of published literature on occurrence records and abundance of Xenarthra in the Neotropics using the following electronic databases: Web of Science (http://scientific.thomson.com/isi/), Science Direct (https://www.sciencedirect.com/), PubMed (https://www.ncbi.nlm.nih.gov/pubmed/), and Google Scholar (https://scholar.google.com.br/). We searched these databases for relevant articles using combinations of the following search terms within the title, abstract, and/or keywords: genus of Neotropical xenarthrans and location names (countries, states, and provinces or biomes and ecosystems) that lacked information in the first round of data compilation. Search terms were used in both English, Portuguese, and Spanish.

Research Methods

All studies that report location coordinates of occurrence or abundance of xenarthrans were included in NEOTROPICAL XENARTHRANS, independent of sampling method or effort. All the recorded coordinates are in decimal degrees, and the datum is WGS 84. Although most of such data includes the precise location of species records, a small portion of the data refers to municipalities, roads, protected areas, or farms (i.e., the location is not precise). However, we have a PRECISION attribute on the database that allows users to track the geographic coordinate precision and identify whether the level of precision satisfies their requirements.

The focus of our data-paper is Neotropical regions, and we delineated this boundary considering the literature description of the realm boundary, encompassing Central America including Mexico, the Caribbean islands, and South America. For the complete database, however, we also allowed contributors to send data for the transitional zone between the Neotropics and neighboring regions, both at northern and southern parts of the Neotropical range. Therefore, if users are interested in the stricto sensu Neotropical regions, we suggest that they use their boundaries of interest, overlap that data, and extract which data they want to use.

We organized the entire dataset into two different sub-datasets, separated according to the data type: quantitative information (*i.e.*, more than one record per point) and qualitative information (i.e., presence and absence and presence-only data; only one record per point). The quantitative information (Table 4) contains all quantitative data (*i.e.* number of records per point) obtained by different methods, such as linear transect, telemetry, and camera trapping. Qualitative data (Table 5) includes all records of occurrence data (individual and occasional records of xenarthran species, independent of the method used), including interviews, museum collections, line transects, camera traps, roadkill, active searches, vestiges, and opportunistic visualization. All the missing information was labeled as “NA” in the dataset.

Data were compiled from Darwin 1842, Bard 1855, Lahille 1899, Miller 1899, Allen 1905, Thomas 1919, Yepes 1938a, 1938b, 1939, 1944, Minoprio 1945, Giai 1950, Vieira 1953, Zeballos 1960, Roig 1965, Rood 1970, Crespo 1974, 1982, Olrog 1976, 1979, Orcés and Albuja 1976, Wetzel and Avila-Pires 1980, Daciuk et al. 1981, Mares et al. 1981, 1996, McCarthy 1982, Massoia and Chebez 1985, Montgomery 1986, Navone 1987, 1988, George et al. 1988, Mascarenhas and Puorto 1988, Cardich 1988, Ojeda and Mares 1989, Cuaron et al. 1989, Stallings et al. 1990, Janzen 1991, Márquez et al. 1991, Carrillo-Jiménez and Wong-Reyes 1992, Del Vitto et al. 1993, Administración de Parques Nacionales 1993, Parker et al. 1993, Heinonen and Bosso 1994, Juliá et al. 1994, Barnnet and Cunha 1994, Vidoz and González 1994, NEA 1995, Pujalte et al. 1995, Brooks 1995, Bolkovi et al. 1995, Rodríguez and Chinchilla 1996, Capllonch et al. 1996, Cherem and Perez 1996, Heinonen Fortabat and Chebez 1997, Gaia 1998a, 1998b, 1998c, 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Taxonomic data

Taxonomic identity was checked by experts of each family:

a) **Chlamyphoridae and Dasypodidae:** A. Desbiez, G. Mourão, A. Feijó, A. Bocchiglieri, N. Attias.
b) **Myrmecophagidae and Cyclopedidae:** F. Miranda, V. Tavares, F. Rodrigues, A. Paglia, A. Bertassoni.
c) **Bradypodidae and Megalonychidae:** A. Chiarello, P. Santos, G. Guiné, S. Silva.

All data were therefore changed with respect to the identity of most recent nomenclature and distribution, following Gibb et al. (2016), Miranda et al. (2017), Feijó et al. (2018), and other available literature (Table 2). The taxonomic uncertainties were maintained, using sp.

Validation

Specialists checked the localities for each xenarthran species and excluded the points that were erroneous (Table 2). Due to changes in taxonomic classification, several records of *Cyclopes didactylus* in North and South America were registered as *Cyclopes* sp., following Miranda et al. (2017).

C. Data Limitations and Potential Enhancements

The extant xenarthrans are a fascinating study group of interest, partly due to their ecology, biology, and taxonomy. Some species have a nocturnal lifestyle, others only inhabit tree canopies, and, in addition, they also show a great ability to camouflage themselves in the environment in which they live. This ability reflects directly in the number of studies; the super order Xenarthra has much less research than other mammal groups and therefore, gathering data from the various species is a significant challenge. Nevertheless, we recognize the many efforts that researchers made to collect and identify the xenarthran data compiled here, given the cryptic habits of several species.

The xenarthran taxonomy, particularly for armadillos, is unusual and usually requires detailed examination for a correct classification. Therefore, some of the species are prone to misidentification, mainly due the similar morphological characteristics and similar common names in different regions. The misidentification and the uncertainty apply notably when it comes to data from interviews, or when the data are very old. In all cases, we resorted to the latest taxonomic classification and known distribution (Abba et al. 2015; Gibb et al 2016; Feijó and Cordeiro-Estrela, 2016, Miranda et al. 2017, Feijó et al. 2018; Table 2), besides an extensive consultation with xenarthran specialists (for more details, see item 4, section B, Class II and Table 2). Some of the records, mainly those from museums or interviews, were excluded due to a lack of certainty regarding occurrences. Two special cases should be highlighted. The recent taxonomic review of the genus *Cyclopes* (Miranda *et al.* 2017) led to uncertainty regarding the data that we received. In this case, after consulting key specialists on these species, we decided to adopt *Cyclopes* sp., thus avoiding errors in the current distribution of the species. The same approach applies to the genus *Dasypus*. We followed a recent review of the taxonomic classification, which splits the species *Dasypus kappleri* into two other species, *D. beniensis* and *D. pastasae* (Feijó *et al.* 2018).

The geographical accuracy of data is another important topic which deserves attention. We received several types of coordinates, some of them referring to the municipality or cities, while others referred to the fragment or point sampled. This difference could represent a bias in our dataset. Therefore, we strongly recommended that co-authors add the precision (in meters). In case of lack of clarity, we sent a protocol created by us to assist in estimating precision.

We made significant efforts to compile data from all different regions of the Neotropics and indeed, data from the different biomes and regions were assembled. There was, however, a concentration of records in some areas (e.g., the southeastern region of Brazil) in contrast to regions with few records (e.g., the Amazon, some countries of Central and South America). To fill this gap, we focused the literature search on those regions (see 2. Data Compilation section B, Class II), even though there are some areas with few or no studies. This pattern was found in other data-papers (such as Lima et al. 2017 and Vancine et al. 2017) and seems to reflect the frequency with which the different natural areas are being studied, often related to accessibility, mainly in the Amazon region. Additionally, we were unable to contact researchers who are developing (or have developed) xenarthran research in some neotropical countries, such as Chile, Peru, Costa Rica, and Panama. This gap is unrelated to lack of studies, but is related to lack of contact with local researchers, since the literature reports xenarthran data on those areas (Suutari 2010, Peery and Pauli 2012, Ruiz-Aravena and 2012),

We highlight that the data come from different studies, and consequently, the records have different sampling efforts, methods, taxonomic scope, and collection date (including some records without the collection date). Therefore, we strongly recommend the use of the data to be carried out with a confiable selection criteria. The lack of quantitative data and the prevalence of occurrence (presence-absence and presence-only) data reflect the challenges involved when working with this group. A xenarthran survey may be complex and require alternative methods and significant efforts. In some cases, it may be necessary to use two or three different methods if questions aside from the presence of the species (e.g., abundance or habitat use) are to be investigated.

Despite these limitations, we compiled the largest and most-complete dataset of xenarthran species of the Neotropical region to date. The merit belongs to the researchers who have provided us with a great deal of information from different kinds of studies. With this data-paper, we are providing data often hidden in the gray literature, or in databases inaccessible to the public. We hope that the public who will use the database will be able to answer questions, such as: 1) general patterns of distribution, 2) gaps in xenarthran studies and which species deserve more attention, 3) poorly studied regions and where to focus more surveys.

Class III - Data set Status and Accessibility

A. Status

Latest update

December 2018.

Latest Archive date

December 2018.

Metadata Status

Latest update December 2018, which refers to the submitted version of revision process

B. Accessibility

Original NEOTROPICAL XENARTHRANS dataset can be accessed on the ECOLOGY repository. All the data – in its updated version and complementary material – are fully available for both public use and research purposes.

Storage location and medium:

The dataset can be accessed on the GitHub Inc. repository (https://github.com/LEEClab/Neotropical_Xenarthans) in .CSV format. A mirror of this repository will also be available at https://github.com/LEEClab/Neotropical_series, where all the other data-papers of NEOTROPICAL SERIES will be available.

Contact persons:

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Copyright restrictions:

None

Proprietary restrictions:

Please cite this data-paper when using its data in publications. We also request that researchers and teachers inform us of how they are using this data.

Costs:

None

Class IV - Data Structural Descriptors

A. Data set File

Identity:

NEOTROPICAL_XENARTHRANS_QUANTITATIVE.csv;

NEOTROPICAL_XENARTHRANS_QUALITATIVE.csv;

NEOTROPICAL_XENARTHRANS_REFERENCES.csv.

Size:

NEOTROPICAL_XENARTHRANS_QUANTITATIVE.csv, 2,585 records, 1, 169KB;

NEOTROPICAL_XENARTHRANS_QUALITATIVE.csv, 39,943 records, 14,631 KB;

NEOTROPICAL_XENARTHRANS_REFERENCES.csv, 1,669 references, 302 KB.

Format and storage mode:

comma-separated values (.csv).

Header Information:

See column descriptors in section B.

Alphanumeric attributes:

Mixed.

Data anomalies:

If no information is available for a given record, this is indicated as 'NA'.

B. Variable Information

1) Table 4. Reference Information
2) Table 5. Quantitative information
3) Table 6. Qualitative information

Class V - Supplemental Descriptors

A. Data acquisition

1. Data request history: None

2. Data set updates history: None

3. Data entry/verification procedures

B. History of dataset usage

Acknowledgements

This paper is part of the Neotropical, Amazonia, Atlantic, and Brazil series of data-papers. We thank the NeoXen team, that made this work possible and to Abigail Feuka and Emma Hanslowe (Colorado State University, USA) for proof reading the manuscript.We thank the companies that provided sources and the colleagues that offered any kind of field assistance: Daniel Martíne Nambo, Cristina López Fuerte, Ricardo Gaudarrama, Juína villages, Cintia Brazão, Luiz Gustavo, Carolina Potter, ICMBio workers from Sena Madureira, Parque Nacional Chapada do Guimarães, Reserva Biológica das Araucárias and Parque Nacional dos Campos Gerais, RESEX Cazumbá-Iracema landowners, Tayanna Mendonça da Silva Godim, Fernanda Guedes da Silva, Marcela Alvares Oliveira, Marcela Alvares Oliveira, Armando Muniz Calouro, Associação de Defesa Etnoambiental Kanindé, Felipe B. R. Gomes, Rodolfo A. Salm, Thiago Cavalcante, Ricardo P. Souza José Ricardo Falconi, Fabio Nunes dos Santos from Fazenda Cambuhy, Nambiquara Indians from Alantesu, Comissão Executiva do Plano da Lavoura Cacaueira (CEPLAC), Área Particular de Preservação Ambiental São Francisco, Programa de Pesquisa em Biodiversidade (PPBio), o Instituto Nacional de Pesquisas da Amazônia (INPA), Biolaw Consultoria Ambiental, National Council of Science and Technology of Mexico, The National Council of Protected Areas of Guatemala and Wildlife Conservation Society Guatemala Program, Ministry of Ecology of Misiones province, the National Parks Administration of Argentina, Fundación Vida Silvestre Argentina, National Institute of Agricultural Technology (INTA), Cataratas del Iguazú International Airport, Argentine Army, Conservation Land Trust, Fundación Amado Bonpland, and property owners, especially ARAUCO S.A., Forestal Bosques del Plata S.A, EVASA, Masisa Forestal S.A., GMF Latinoamericana S.A., Forestal Las Marías, Altinia S.A. and the several farmers and landowners that provided support and allowed us to conduct camera trap surveys on their properties, Estação Ambiental de Volta Grande, MG, Valdênio Marques, Fernando de Miranda Ramos, FLONA de São Francisco de Paula workers. Companhia Paranaense de Energia (COPEL), Hugo C. M. Costa, Leandro A. Abade, Frederico Souza, Daniel G. Rocha, Caio F. M. Lima, Mr. Lenilson, Botanical Garden Benjamin Maranhão staff, UFAL staff, Adriana Ribeiro de Oliveira, Paula Vianna, RPPN SESC Pantanal, Instituto Estadual de Florestas de Minas Gerais, (IEF) for their support during fieldwork at Rio Doce State Park, the management of the Chandless State Park and the Extractive Reserve of Cazumbá Iracema, AMATA S.A, Floresta Nacional do Jamari, Rondônia, Brazil, Agência de Transporte do Estado de São Paulo (ARTESP), Dr. Ramachrisna Teixeira and Mr. Sisinho Paes de Santana, Institute of Astronomy, Geophysics and Atmospheric Sciences of the University of São Paulo (IAG-USP), Instituto de Desenvolvimento Sustentável Mamirauá (IDSM), the owners and workers of Fazenda San Francisco, Fernando Alvarenga Reis from EMBRAPA Gado de Corte, Lerrane de Fatima Cunha, Fundação de Amparo à Pesquisa do Estado de Minas Gerais – FAPEMIG, Novo Colégio – SP, the Parque Nacional Sierra de las Quijadas- APN/ARG park guards, especially Daniel Figueroa, volunteers and students from PROICO 2-2314: Mamíferos del Parque Nacional Sierra de las Quijadas- UNSL-Arg, the Instituto Geográfico Cartográfico do Estado de São Paulo for providing the orthophoto mosaic images; the University of São Paulo, the Instituto Florestal, and the Fundação Florestal for logistical support, the whole team of Tijuca National Park, the Laboratory of Ecology and Conservation of Forests - LECF/UFRRJ, in particular to Thayssa and AS Pires, and of Laboratory of Ecology and Conservation of Populations – LECP / UFRJ, especially to César, Raíssa, Marcelo, FAS Fernandez, and all who directly or indirectly contributed to the Neotropical Xenarthrans data-paper. Most of the Mexican records were kindly made available by the Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. 2016. EncicloVida (Online database: http://www.enciclovida.mx). CONABIO. Mexico. All co-authors from the Bolivian medium to large sized mammal volume and associated database, especially Heidy Lopez-Strauss, Nohelia Mercado and Zulia Porcel. We dedicate this work to G. Gene Montgomery, M.E. Sunquist, Laurenz Pinder, and all the great names associated with Xenarthrans, that certainly contributed and inspired all current research with this incredible group. We also appreciate all the comments and suggestions from the reviewers and editor. We dedicate this work to the National Museum of Rio de Janeiro, which strongly contributes to research, education, and science not only in Brazil, but for the entire Neotropic realm and for the whole world.

TABLES

Table 1. Species information. Family, IUCN conservation status and number of records of the armadillos, anteaters and sloth species reported in the NEOTROPICAL XENARTHRANS dataset. Species without confirmed identification (sp.; excluding *Cyclopes* sp.) were not included in the species counting. DD=Data Deficient, NE=Not Evaluated, LC=Least Concern, NT=Near Threatened, VU=Vulnerable, CR=Critically Endangered.

Group	**Familly**	**Species***	**IUCN status**	**Number of records**
Armadillos	Chlamyphoridae	*Cabassous centralis*	DD	189
		Cabassous chacoensis	NT	30
		Cabassous tatouay	LC	585
		*Cabassous unicinctus**	LC	562
		Calyptophractus retusus	DD	33
		Chaetophractus vellerosus	LC	372
		Chaetophractus villosus	LC	1,473
		Chlamyphorus truncatus	DD	95
		Euphractus sexcinctus	LC	7,325
		Priodontes maximus	VU	1,678
		Tolypeutes matacus	NT	1,044
		Tolypeutes tricinctus	VU	194
		Zaedyus pichiy	NT	195
	Dasypodidae	*Dasypus beniensis*	NE	278
		*Dasypus kappleri***	LC	94

		*Dasypus mazzai****	DD	26
		Dasypus novemcinctus	LC	11,588
		Dasypus pastasae	NE	217
		Dasypus sabanicola	NT	44
		*Dasypus septemcinctus hybridus***	NT	707
		Dasypus septemcinctus septemcinctus **	LC	477
Anteaters	Cyclopedidae	*Cyclopes* sp.**	NT*	240
	Myrmecophagidae	*Myrmecophaga tridactyla*	VU	5,941
		Tamandua mexicana	LC	319
		Tamandua tetradactyla	LC	5,089
Sloths	Bradypodidae	*Bradypus pygmaeus*	CR	12
		Bradypus torquatus	VU	638
		Bradypus tridactylus	LC	222
		Bradypus variegatus	LC	962
	Megalonychidae	*Choloepus didactylus*	LC	268
		Choloepus hoffmanni	LC	137

*Including the registers of the subspecies.

**A re-evaluation is required because of recent taxonomic changes.

*** Taxonomic synonym *Dasypus yepesi*.

Table 2: List of species registered in the data paper, the synonym (If applicable), reference we use to confirm the record and justification of why we corrected the species (If applicable). - means “no information”.

Species	**Synonyms**	**Reference**	**Justification**
Bradypus pygmaeus	-	Anderson and Handley 2001, Pauli 2018	-
Bradypus torquatus	-	Hirsch and Chiarello 2012, Gardner 2008, Pauli 2018	-
Bradypus tridactylus	-	Moraes-Barros et al 2010, Gardner 2008, Pauli 2019	Some records of *Bradypus tridactylus* were corrected to *B.variegatus*, since *B.tridactylus* occurs only north of the Amazon River and east of the Rio Negro, and some points were well outside the known distribution for the species
Bradypus variegatus	-	Moraes-Barros et al 2010, Gardner 2008, Pauli 2020	-
Cabassous centralis	-	Abba and Superina 2010	-

Cabassous chacoensis	-	Abba and Superina 2010	-
Cabassous tatouay	-	Abba and Superina 2010, Feijó and Langguth 2013	-
Cabassous unicinctus	-	Abba and Superina 2010, Feijó and Langguth 2013	The genus *Cabassus* is represented by *C. tatouay* in northeastern Brazil. Historically these records were erroneously applied to *C.unicinctus.* Thus, all the records from northeastern Brazil were classified as *C. tatouay*
Calyptophractus retusus	*Chlamyphorus retusus; Burmeisteria retusa*	Abba and Superina 2010, Delsuc 2009	-
Chaetophractus vellerosus	*Chaetophractus nationi; Dasypus vellerosus*	Abba and Superina 2010, Abba et al. 2015	All the records of *Chaetophractus nationi* were considered as *C. vellerosus*
Chaetophractus villosus	*Dasypus villosus*	Abba and Superina 2010	-

Chlamyphorus truncatus	-	Abba and Superina 2010	-
Choloepus didactylus	-	Chiarello and Plese 2014, Moraes-Barros 2018	-
Choloepus hoffmanni	-	Plese and Chiarello 2014, Moraes-Barros 2018	-
Cyclopes sp.	*Cyclopes didactylus*	Miranda et al. 2017	All the records were considered as *Cyclopes* sp., due to the recent taxonomic review
Dasypus beniensis	*Dasypus kappleri*	Feijó and Cordeiro-Estrela 2016	The records south of the Amazon River were considered *Dasypus beniensis*
Dasypus kappleri	-	Feijó and Cordeiro-Estrela 2017	*Dasypus kappleri* was divided into three species. With the new arrangement, *D. kappleri* is restricted to the region of the Guiana shield. The records south of the Amazon River refer

			to *D. beniensis*, and to the west of the Guiana shield and north of the Amazon River refer to *D. pastasae*.
Dasypus mazzai	*Dasypus yepesi*	Feijó and Cordeiro-Estrela 2014	-
Dasypus novemcinctus	-	Abba and Superina 2010, Feijó et al. 2018	-
Dasypus pastasae	*Dasypus kappleri*	Feijó and Cordeiro-Estrela 2016	The records of west Guiana shield and north Amazon River were considered *Dasypus pastasae*
Dasypus sabanicola	-	Abba and Superina 2010, Feijó et al. 2018	-
Dasypus septemcinctus hybridus	*Dasypus hybridus*	Abba and Superina 2010, Feijó et al. 2018	All the records of *Dasypus hybridus* were re-classified as *D. septemcinctus hybridus* due to recent taxonomic

			changes, which considered *D. hybridus* as a subspecies of *D. septemcinctus*
Dasypus septemcinctus septemcinctus	*Dasypus septemcinctus*	Abba and Superina 2010, Feijó et al. 2018	For all records of *D. septemcinctus* we added the subspecies *D. septemcinctus septemcinctus*, due to the recent taxonomic review
Euphractus sexcinctus	-	Abba and Superina 2010	-
Myrmecophaga tridactyla	-	Miranda et al. 2014, Bertassoni 2018	-
Priodontes maximus	*Priodontes giganteus*	Abba and Superina 2010	All the records of *Priodontes giganteus* were considered as *P. maximus*, since the name *P. giganteus* is not used anymore, according to specialists.
Tamandua mexicana	-	Ortega et al. 2014, Bertassoni 2018	-

Tamandua tetradactyla	-	Miranda et al. 2014. Bertassoni 2018	-
Tolypeutes matacus	-	Abba and Superina 2010, IUCN 2018	-
Tolypeutes tricinctus	-	Abba and Superina 2010, Feijó et al. 2015	-
Zaedyus pichiy	-	Abba and Superina 2010	-

Table 3. Method information. All the methods identified, their absolute frequency and percentage.

Method	**Frequency**	**Percentage**
Camera trap	13,783	32.4
Road kill	6,225	14.6
Opportunistic	4,339	10.2
Line transect	2,551	6
Active search and interview	2,141	5.03
Bibliographical survey	1,609	3.78
Monitoring	1,358	3.19
Museum scientific collection	780	1.83
Vestige	673	1.58
Active search	624	1.47
Interview	523	1.23
Telemetry	343	0.81
Active search and vestige	243	0.57
Capture	238	0.56
Tracks plot	99	0.23
Car monitoring	74	0.17
Line transect and camera trap	74	0.17
Line transect and live trap	74	0.17
Data base	65	0.15
Burrows sampling	60	0.14
Apprehension	46	0.11

Active search and opportunistic	41	0.1
Live trap	39	0.09
Interview and vestige	21	0.05
Active search, vestige and interview	11	0.03
Live trap and vestige	11	0.03
Interview and bibliographical survey	9	0.02
Active search and roadkill	4	0.01
Camera trap and vestige	4	0.01
Interview, museum scientific collection and bibliographical survey	4	0.01
Vestige and road kill	4	0.01
Active search and camera trap	3	0.01
Active search and live trap	3	0.01
Line transect and vestige	3	0.01
Active search, vestige and camera trap	2	0
Capture and interview	2	0
Active search and bibliographical survey	1	0
Active search and roadkill collection	1	0
Active search, bibliographical survey and interview	1	0
Interview and museum scientific collection	1	0
Interview, museum scientific collection and vestige	1	0
Sand plot	1	0

Vestige and museum scientific collection	1	0
NA*	6,438	15.1
Total	42,528	100

*No information

Table 4. Reference information. Description of the fields related to references.

Type of information	Field	Description	Levels	Example
REFERENCE INFORMATI-ON	REF_ID	Reference identification that links the references to a specific record in the dataset.	NEOXEN_1001 – NEOXEN_2669	NEOXEN_1703
	REF_TYPE	Type of the publication: “Thesis” refers to theses, dissertations and monographs; “Technical document” refer to reports and wildlife management programs.	Abstract Book Book chapter Database In prep In press Management plan Personnal Report Scientific paper Submitted Thesis Unpublished	Scientific paper

	REFERENCE	Study reference in *Ecology* style.	1554 references	Santos, P. M., A. G. Chiarello, M. C. Ribeiro, J. W. Ribeiro, and A. P. Paglia. 2016. Local and landscape influences on the habitat occupancy of the endangered maned sloth *Bradypus torquatus* within fragmented landscapes. Mammalian Biology - Zeitschrift für Säugetierkunde 81:447–454.
	Ref_STATUS	Located and unlocated references	Located Not Located	Located
	CITATION_in_text	Citation format in text: author and year		Santos et al. 2016

Table 5. Quantitative information. Description of the fields related to NEOTROPICAL XENARTHRANS abundance data.

Field	**Description**	**Levels**	**Example**
ORDEMBD	Identification code of each species record. Each code is exclusive and represents the record in the sampling area and the dataset of origin. Records made in the same area in different studies received different codes.	ABBA_00001 - ABBA_02128; ACSA_00001 - ACSA_00060; PAMS_00001 - PAMS_00033	ACSA0001
REFERENCE	Study reference for the published records.		Carrillo-Bilbao G, Cuircama V, Ruíz R, Martin-Solano S (2016) Área de vida, dieta, preferência de estrato vertical y uso del tiempo de Bradypus variegatus, liberados
STUDY_AIM	Information about the objective of the study.		ABD-COMM
DATA_TYPE	Type of collected data.	Presence only Presence-absence Abundance	Abundance

SPECIES	The name of the registered species according to the current taxonomy.	See Table 1	*Bradypus torquatus*
GENUS	The name of the genus according to the current taxonomy.		*Bradypus*
FAMILY	The name of the family according to the current taxonomy.		Bradypodidae
ORDER	The name of the ORDER according to the current taxonomy.		Pilosa
IUCN_STATUS	Information about the threatened status according to IUCN red list.	NE DD LC NT VU EN CR EW EX	VU

SP_ORIGIN	Information about the origin of that species in that sampling area.	Native Potentially introduced Introduced	Native
SITE	Name of the sampling area.		La Estación Científica Agroecológica Fátima
AREA_HA	Area of the sampling site in hectares.	0.05 - 5.400.000	3096
MUNICIPALITY	Municipality of the sampling site.		Fatima
STATE	State or province of the sampling site.		Pastaza
COUNTRY	Country of the sampling site.		Ecuador
LONG_X	Longitude corrected and transformed into decimal degrees (datum WGS84).	Decimal Degree	-56.11345
LAT_Y	Latitude corrected and transformed into decimal degrees (datum WGS84).	Decimal Degree	-19.566064
PRECISION	Coordinate precision, in meters, of the sampling site.	3 - 290000	30

UC	Information about the protection of the sampling area. Yes: if the area is a conservation unit or is within a conservation unit; No: If the area is outside a conservation unit.	Yes - No	Yes
VEG_TYPE	Vegetation type of the sampling area location, as described in the reference paper, or described by the data collector.		Amazon forest
ANNUAL_RAIN	Annual rainfall WorldClim v. 1.4., in mm, available in http://www.worldclim.org/version1. Access on August 2nd, 2018.		164
ANNUAL_TEMP	Average annual temperature WorldClim v. 1.4., in Celsius degrees, available in http://www.worldclim.org/version1. Access on August 2nd, 2018.		25.1

ALTITUDE	Altitude in meters above sea level, from the Hydro-1K dataset (United States Geological Survey – USGS, 2001. HYDRO 1K: Elevation Derivative Database. Available from: <http://edc.usgs.gov/products/elevation/gtopo30/hydro/namerica.html>). Access on August 2nd, 2018.		469
Olsoneconame	ECO_NAME column of the shapefile wwf_terr_ecos available in WWF website (Olson et al. 2001).		Caatinga
OlsonG200r	Olson bioregion (Olson et al. 2001), represented by G200_REGIO column of the shapefile wwf_terr_ecos		Cerrado Woodlands and Savannas
METHOD	Sampling method described in the reference paper or by the data collector.		Line transect
N_POINTS	Number of sampling points.	1 - 1401	500

D_POINTS	Distance, in kilometers, between the sampling points.	0 - 1.6	1.2
LTR_EFF	Effort in kilometers walked on transects in the sampling area.	0.5 – 5000	618
LTR_ABD_10KM	Abundance of individuals registered on transects. Calculated as the number of individuals/10km.	0 – 25.71	0.32
CAM_EFF	Survey effort in the sampling area using camera traps. Measured in hours.	1.03 - 36000	3600
CAM_ABD	Number of records registered on camera traps.	0 - 94	25
SURV_EFF	Survey effort in the sampling area. Measured in days, hours and kilometers.	8.8h – 24.3h	24.1h
SURV_ABD	Number of records registered on survey. Calculated as the number of counts/day.	0 - 85	33
COL_STRT_MO	Month when data collection started	January - December	May
COL_STRT_YR	Year when data	1896- 2018	1996

	collection started.		
COL_END_MO	Month when data collection ended.	January - December	December
COL_END_YR	Year when data collection ended.	1855- 2018	1996

Table 6. Qualitative information. Description of the fields related to NEOTROPICAL XENARTHRANS occurrence data.

Field	**Description**	**Levels**	**Example**
ORDEMBD	Identification code of each species record. Each code is exclusive and represents the record in the sampling area and the dataset of origin. Records made in the same area in different studies received different codes.	ABBA_00001 - ABBA_02128; ACSA_00001 - ACSA_00060; PAMS_00001 - PAMS_00033	PAMS_0001
REFERENCE	Study reference for the published records.		Carrillo-Bilbao G, Cuircama V, Ruíz R, Martin-Solano S (2016) Área de vida, dieta, preferência de estrato vertical y uso del tiempo de Bradypus variegatus, liberados
STUDY_AIM	Information about the objective of the study.		Occur
DATA_TYPE	Type of collected data.	Presence only Presence-absence	Presence-absence

SPECIES	The name of the registered species according to the current taxonomy.	See Table 1	*Bradypus variegatus*
GENUS	The name of the genus according to the current taxonomy.		*Bradypus*
FAMILY	The name of the family according to the current taxonomy.		Bradypodidae
ORDER	The name of the ORDER according to the current taxonomy.		Pilosa
IUCN_STATUS	Information about the threatened status according to IUCN red list.	NE DD LC NT VU EN CR EW EX	LC
SP_ORIGIN	Information about the origin of that species in that sampling area.	Native Potentially introduced	Native

SITE	Name of the sampling area.		La Estación Científica Agroecológica Fátima
AREA_HA	Area of the sampling site in hectares.	0.05 - 5.400.000	3096
MUNICIPALITY	Municipality of the sampling site.		Fatima
STATE	State or province of the sampling site.		Pastaza
COUNTRY	Country of the sampling site.		Ecuador
LONG_X	Longitude corrected and transformed into decimal degrees (datum WGS84).	Decimal Degree	-56.11345
LAT_Y	Latitude corrected and transformed into decimal degrees (datum WGS84).	Decimal Degree	-19.566064
PRECISION	Coordinate precision, in meters, of the sampling site.	3 - 290000	30
UC	Information about the protection of the sampling area. Yes: if the area is a conservation unit or is within a conservation unit; No: If the area is outside a conservation unit.	Yes - No	Yes

VEG_TYPE	Vegetation type of the sampling area location, as described in the reference paper, or described by the data collector.		Amazon forest
ANNUAL_RAIN	Annual rainfall WorldClim v. 1.4., in mm, available in http://www.worldclim.org/version1. Access on August 2nd, 2018.		164
ANNUAL_TEMP	Average annual temperature WorldClim v. 1.4., in Celsius degrees, available in http://www.worldclim.org/version1. Access on August 2nd, 2018.		25.1
ALTITUDE	Altitude in meters above sea level, from the Hydro-1K dataset (United States Geological Survey – USGS, 2001. HYDRO 1K: Elevation Derivative Database. Available from: <http://edc.usgs.gov/products/elevation/gtopo30/hydro/namerica.html>) Access on August 2nd, 2018.		469

Olsoneconame	ECO_NAME column of the shapefile wwf_terr_ecos available in WWF website (Olson et al. 2001).		Uruguayan savanna
OlsonG200r	Olson bioregion (Olson et al. 2001), represented by G200_REGIO column of the shapefile wwf_terr_ecos		Cerrado Woodlands and Savannas
METHOD	Sampling method described in the reference paper or by the data collector.		Line transect
N_POINTS	Number of sampling points.	1 - 1401	500
D_POINTS	Distance, in kilometers, between the sampling points.	0 - 1.6	1.2
LTR_EFF	Effort in kilometers walked on transects in the sampling area.	0.5 – 5000	618
CAM_EFF	Survey effort in the sampling area using camera traps. Measured in hours.	1.03 - 36000	3600
SURV_EFF	Survey effort in the sampling area. Measured in days, hours and	8.8h – 24.3h	24.1h

	kilometers.		
COL_STRT_MO	Month when data collection started	January - December	May
COL_STRT_YR	Year when data collection started.	1896- 2018	1996
COL_END_MO	Month when data collection ended.	January - December	December
COL_END_YR	Year when data collection ended.	1855- 2018	1996

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Capítulo 2

Living on the Edge: Forest cover threshold effect on endangered maned sloth occurrence in Atlantic Forest

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Manuscript published at Biological Conservation:
https://doi.org/10.1016/j.biocon.2019.108264

Abstract

Habitat loss and the isolation of remaining habitats are undoubtedly the two greatest threats to biodiversity conservation, especially for the maned sloth, due to its ecological restrictions. In this study, we identified a critical threshold of forest cover for maned sloth occurrence and explored the effects of other local and landscape variables. We sampled 68 sites, where we searched for the maned sloth and collected local habitat variables. We calculated the percentage of forest cover and open areas, assessing the appropriated scale through model selection. We used occupancy models and model selection methods to identify the threshold and assess occupancy and detection probabilities. The occupancy probability of the maned sloth is 0.97, but it decreases abruptly at 35% of forest cover, reaching zero in areas with less than 20% of forest cover. The two landscape variables are the most important predictors of sloth occupancy, based on the cumulative weight of evidence, were: Forest cover (78%) and Open areas cover (46%); the latter influencing negatively maned sloth occupancy. This is the first attempt to identify the habitat requirements of the threatened maned sloth in a fragmented area using landscape and local variables. Our results imply that conservation of maned sloth will benefit from an increase in the amount of native forest at the landscape scale. Given difficulties in the creation of new public protected areas, this improvement could be achieved via restoration of areas located in private properties which are protected by the Brazilian Forest Code.

Keywords: *Bradypus torquatus;* Pilosa; Threatened; Threshold; Atlantic Forest; Habitat loss

1. Introduction

Human-dominated areas have expanded throughout the world and are primarily responsible for increased deforestation (LEBLOIS *et al*., 2017). Habitat fragmentation and loss leads to patch size reduction, increases habitat isolation and edge effects, and have several negative consequences for biodiversity (FAHRIG, 2003). Worldwide, most of Earth's native ecosystems are within 1 km of the habitat edge (HADDAD *et al.*, 2015), embedded in anthropogenic matrix that is often inhospitable to biodiversity (CASTELLÓN; SIEVING, 2006). These anthropogenic matrices can act as a filter, restricting dynamic processes such as dispersion and movement (MORAES *et al.*, 2018; MORATO *et al.*, 2018), gene flow (DIXO *et al.*, 2009), and pollination (PAVAGEAU *et al.*, 2017). Additionally, deforested areas have limited shelter or refuge for native fauna, exposing species to opportunist predation by domestic dogs (DOHERTY *et al.*, 2017), poaching (FERREGUETTI *et al*., 2018) and roadkill (ASCENSAO *et al.*, 2017; ASSIS; GIACOMINI; RIBEIRO, 2019). All these threats are commonly related with human activities in a non-habitat matrix areas (DOHERTY *et al*., 2017; FERREGUETTI *et al*., 2018).

Among the various attributes intrinsic to anthropized environments, the amount of remaining habitat is perhaps one of the most influential in the dynamics of populations and communities (FAHRIG, 2013). When habitat loss increases and reaches a critical threshold, small modifications in the remaining habitat can produce more severe changes in biological responses (WITH; KING, 1999; HUGGETT, 2005). Theory predicts that structural and functional connectivity suffers when the amount of remaining habitat drops below this critical threshold, and a species distribution becomes more dependent on patch size and isolation (LANDE, 1987; FAHRIG, 2003). The threshold value will vary among species, following the species' habitat requirements (MUYLAERT *et al*., 2016), patch size (MAGIOLI *et al.*, 2015), matrix composition (BOESING; NICHOLS; METZGER, 2018), and biome (MELO *et al.*, 2018).

As the proportion of suitable habitat increases (*i.e.* above a given threshold), other variables may drive the ecological dynamics, besides the landscape context, such as habitat quality. The local habitat structure and composition plays an important role in occurrence and persistence of animal communities (AUGUST, 1983; SANTOS *et al*., 2016). Increasing habitat complexity creates a safe and dynamic environment, providing different resources and conditions necessary for reproduction (SHENBROT *et al.*, 2002) and foraging success

(JACOB; MATTER; CAMERON, 2017). In addition, a more complex habitat structure may increase the alpha and beta diversities by providing suitable habitats for a higher number of species (CAMARGO; SANO; VIEIRA, 2018; GRELLE, 2003).

The Atlantic Forest is a biodiversity hotspot (ZACHOS; HABEL, 2011) and one of the most threatened biomes in the world and its area has been reduced to 28% of its original cover (REZENDE *et al.*, 2018). Most remaining forest patches are smaller than 50 ha and about half of the remaining forest is within 100 meters of any forest edge (RIBEIRO *et al.*, 2009). Landscape composition and configuration pose a threat to several species, including the endemic maned sloth (*Bradypus torquatus*; Bradypodidae). According to Santos *et al*. (2019), in a data paper that compiled the largest data set on abundance and occurrence of Neotropical Xenarthrans, maned sloth (with 638 records) are distributed along Atlantic Forest cost from Sergipe to Rio Janeiro, Brazil. The species inhabits dense ombrophylous forests (HIRSCH; CHIARELLO, 2012) and feeds mainly on leaves and shoots (MONTGOMERY; SUNQUIST, 1975; CHIARELLO, 1998b). The species rarely descends to the ground, moving mostly through the forest canopy (MONTGOMERY; SUNQUIST, 1975). Due to its strictly arboreal habit and its low capacity to move in non-forested areas, the species is highly affected by the habitat loss (CHIARELLO; MORAES-BARROS, 2014). Its restricted distribution and the dependence on forest led the International Union for Conservation of Nature (IUCN) to classify the maned sloth as Vulnerable (VU) in 2014 (CHIARELLO; MORAES-BARROS, 2014).

There is strong evidence that habitat loss and fragmentation negatively influence maned sloth occurrence (CHIARELLO; MORAES-BARROS, 2014). Intuitively, increasing the connection between fragments would be a logical strategy to improve the conservation of the species. Nevertheless, what exactly does “improve connection” mean for the maned sloth? What are the minimum habitat amount requirements to maintain this species? Is there a critical threshold, below which landscape variables would most likely influence the species occurrence? These questions are relevant to decision-makers responsible for applying resources to direct conservation efforts; these efforts are led by the National Center for Research and Conservation of Brazilian Primates (CPB) and linked to Chico Mendes Institute for Biodiversity Conservation (ICMBio), a Brazilian federal agency responsible for the National Action Plans.

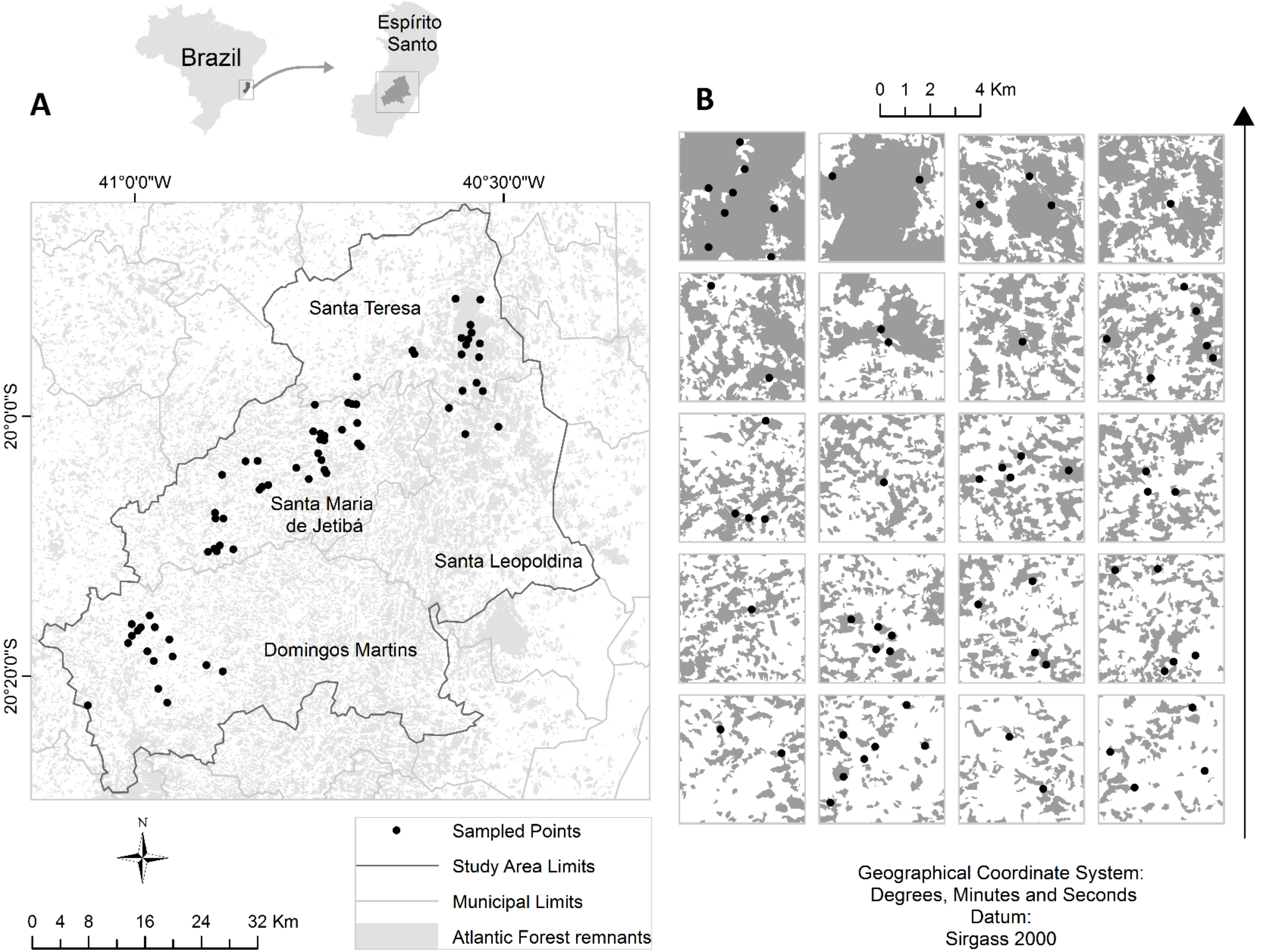
In previously study, Santos *et al*. (2016) found a strong positive relationship between local variables (e.g., canopy height and proportion of important trees) and maned sloth occurrence – these local variables were more important than factors related to habitat fragmentation (e.g., patch size and isolation metrics). However, the studied area encompassed a region with >36% of forest cover, a level that exceeds forest thresholds found in other ecosystems with different taxa (thresholds ~30 – 35% forest cover; ANDRÉN, 1994; ESTAVILLO *et al.*, 2013). Motivated by our previous experience, here, we expanded the study region to encompass a broader area covering the full gradient of available habitat (5 to 100% forest cover) to identify and assess larger-scale variables for maned sloth occurrence. In this study, we aimed to better understand what affects maned sloth occupancy probability at the landscape level and in a local context to improve the subsidies for landscape and local management, respectively. We also contribute to the understanding of maned sloth habitat requirements, which are still unclear. We hypothesize there is a threshold relationship between the forest cover (%) and maned sloth occurrence: above this threshold maned sloth occurrence should be high and governed primarily by local variables and below this threshold maned sloth occupancy should decline rapidly regardless of local variables.

2. Material and Methods

2.1. Study area

We conducted this study in the mountainous region of Espírito Santo State in southeast Brazil (Fig. 1), which historically consisted entirely of the Atlantic Forest (MUYLAERT *et al.*, 2018). The study region encompasses an area of approximately 187,000 ha (Fig. 1), composed primarily of dense lower-montane and montane ombrophylous forests (INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA, 2012). The elevation ranges from 600 to 1200 m.a.s.l. and temperatures are mild throughout the year (mean annual temperature = 22.1°C). The annual mean precipitation is 1232 mm, with most rain concentrated between November and March (INCAPER, 2013a, b, d, c).

The study region has 34% of native forests, concentrated (> 85%) in the northeast, where the largest protected areas are located, while the southern part has less than 20% of forest cover (Fig. 1). Most forest areas are privately owned (97.2%), mainly by descendents from European settlers, who arrived at the region in the early 19th century, and established small agricultral properties and a family farming system – which remains the economic base today (INCAPER, 2013a, b, d, c).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Sirgass 2000

Fig. 1: Study area with the location of the 68 sampled points for recording maned sloth within Atlantic Forest remnants, Espírito Santo, Brazil (A). Example of the forest cover gradient throughout the region, in which the sampled points are inserted (B). The right-side arrow on Panel B indicated the north-south forest cover gradient arrangement.

2.2. Sampling design and covariates selection

We selected 68 sampling points (i.e., sites) using GrassGis 7.4.1 software (GRASS Development Team, 2018) across a range of forest cover (5-100%) within the study region. Specifically, we select 19 sites on northeast (higher percentage of forest cover), 16 sites in the south (lower percentage of forest cover), and 33 sites from the central Santa Maria de Jetibá region (medium percentage of forest cover). To minimize spatial autocorrelation, sites were at least 500 m away from each other, a distance that is large enough to accommodate the home ranges of multiple individuals – the average sloth home range was estimated at 4.5 ha in previous studies conducted in this same area (CHIARELLO, 1998a). Sloth surveys were conducted during the dry season to avoid effects of rain on species detection. Specifically, we surveyed 33 sites in the central Santa Maria de Jetibá region from April to August in 2014 (SANTOS et al., 2016), and 35 different sites from April to October in 2017 and from March to July in 2018. The sites surveyed in 2014 were not surveyed in 2017/2018 and vice-versa. Surveys within a given dry season were spaced at approximately 1-month intervals such that each site was surveyed six times. During each survey, we established two perpendicular 200-m transects, forming a cross (Figure A.1). We walked the resulting four 100-m long segments at a speed of about 0.24 km/h, searching the canopies for maned sloths. To maximize detection, we walked the transects between 7 am - 5 pm, when the species is most active (CHIARELLO, 1998a; GINÉ *et al.*, 2015). We measured two local variables, canopy height (m) and the proportion of important trees (hereafter important trees) at five locations, including the center point (intersection of the two perpendicular transects) and at the halfway point of each 100-m transects (Figure A.1). At each location, we measured the canopy height with a rangefinder (Bushnell model Sport 850, 4 x 20 mm) and we recorded the presence/absence of important tree species (*Micropholis venulosa*, *Cupania furfuraceae*, *Crecopia* spp., and *Ficus* spp.) within a 5 m radius of each location. Collectively, these species represent more than 40% of the maned sloth diet (CHIARELLO, 1998b). We combined the measurements at the five locations for each site to determine the site-specific mean canopy height and the proportion of locations with at least one of the important trees.

We also calculated two compositional landscape strucutre variables that we felt may influence maned sloth probability of occurrence: forest cover (%) and open areas cover (%). We expected each covariate of interest to influence maned sloth ecology and biology differently (Table 1). For example, forest cover is important to reproduction, foraging success, and other aspects of maned sloth ecology and behavior. Therefore, we expected a positive relationship between the forest cover and maned sloth occurrence. Conversely, open areas cover may deter or hamper the species' dispersal, since moving through these areas may expose sloths to predation, starvation, temperature stress, and other hazards. As the open areas cover increases, we expect maned sloth occupancy to decrease. At the local level, important trees are a necessary food source and consequently, there should be more maned sloth in areas with a high proportion of those trees, leading to higher species occurrence and detection probabilities. The relationship with detection should also be positive since it may be easier to detect the species if individuals spend more time in or around these food sources. Many trees also serve as shelter where sloths can safely rest and thermoregulate appropriately, which is usually done in the upper strata of the forest canopy. Accordingly, we expect occupancy probability to increase with increasing canopy height. However, detection is more difficult the higher the sloth is in the canopy since vegetation density also increase with canopy height, obstructing our viewing capacity.

Table 1: Local and landscape level variables thought to influence maned sloth occupancy probability (Ψ) or detection probability (p). Hypothesized positive (+) or negative (-) effects are given for each parameter.

Variable	Level	Ecological Importance	Parameter	Effect on Ψ	Effect on p
Forest cover (%)	Landscape	Breeding Foraging success Shelter	Ψ	+	
Open areas cover (%)	Landscape	Dispersal Exposure to threats	Ψ	-	
Important trees (Proportion of locations)	Local (5 m)	Food source	Ψ/p	+	+
Mean canopy height (m)	Local (5 m)	Shelter Thermoregulation	Ψ/p	+	-

To calculate landscape variables for each sampled site, we used a 2015 land use and land cover map with 10-m resolution, georeferenced in the UTM Projection System, SIRGAS

2000 Datum, Zone 24s, provided by Geobases (IEMA-ES, https://geobases.es.gov.br/). We ran the landscape analyses with Fragstats 4.2 (McGarigal et al., 2012) and considered six buffer sizes (i.e. scale of effect or spatial extension; MIGUET *et al.*, 2015) for each sampled sites: 100 m, 200 m, 300 m, 400 m, 500 m and 600 m. The usage of multiple spatial extents when evaluating the scale of effect of landscape structure influence on species occurrence, abundance or species richness is of utmost importance in landscape ecology studies (GESTICH *et al.*, 2019). Within each buffer, we calculated the percentage of forest cover and the percentage of open areas cover – the latter including existing and abandoned pasture and exposed soil.

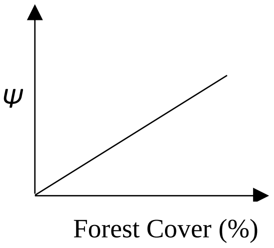
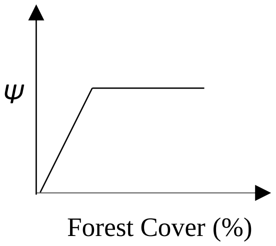
2.3. Occupancy and detection probabilities

We used a step-wise approach (four steps) to evaluate factors influencing maned sloth occupancy and detection probabilities. First, we determined the scale of effect (buffer size) for each of our landscape covariates: percentage of forest cover and percentage of open areas cover. Using a general model structure where both occupancy and detection probabilities are modeled as additive functions of local variables (important trees and canopy height), we fitted 36 models, where sloth occupancy was modeled as a linear function (on the logit scale) of all possible combinations of the six spatial extents for percent forest cover and percent open areas cover (Table A.1). We used the best-supported scale of effect for each of the landscape variables in subsequent analyses. We also performed a correlation analysis using the best-supported scale of effect for forest cover (500 m) and open area cover (600 m) and our local covariates to ensure that none of our covariates were highly correlated ($|r| < 0.6$; Table A.2).

Second, we explored if there was evidence of a threshold relationship between the probability of sloth occurrence and the percent of forest cover, using a general detection probability structure and an occupancy structure that accounted for potential variation associated with our other covariates. Specifically, we built eight occupancy structures: one corresponding to the linear relation (on the logit scale) between the sloth occupancy probability and percent forest cover and the other seven models reflected threshold effects at specified values of forest cover (e.g., 20%, 25%, 30%,…, 50% forest cover; Tables 2 and A.4). For example, a model with a specified threshold of 30% forest cover suggests that sloth occupancy starts to decline when the forest cover falls below 30%, but sloth occupancy is approximately equal at sites above the threshold (see Table 2 for a graphical representation of these relationships). We modelled thresholds from 20% to 50% forest cover (increasing in 5%

increments) and used a model selection criteria (AICc) to determine which threshold level was best supported by our data (Table A.4).

Table 2: Graphical representation for different threshold relations between maned sloth occupancy probability (Ψ) and forest cover (%) in Atlantic Forest, Espírito Santo, Brazil. Eight threshold relationships were fit to the data including a linear relationship (no threshold) and seven threshold models where the specified threshold value (x) varied among models and ranged from x = 20% forest cover to x = 50% forest cover, vary by increments of 5%. All threshold models had eight parameters, allowing sloth occupancy probability to vary with other considered variables, Ψ (Open areas cover + Important Trees + Canopy Height + *Threshold Relationship*) and detection probability was modeled as function of both local variables, p (Canopy Height + Important Trees).

Threshold Relationship	Graphical Representation
Linear (no threshold)	
Threshold specified at 7 different Forest Cover levels (x) x = 20%, 25%, …, 50%	

Third, using the best supported threshold occupancy structure, we explored local factors that could influence sloth detection probability. Specifically, we fitted four detection probability structures that included all additive combinations of canopy height and important tree (Table 3). Finally, we considered occupancy structures to explore additive (16 models) and interactive effects (6 models) of our local and landscape variables, fitting models with up to 7 parameters to avoid the overparameterization. We explored interactive relationships between forest cover and the other occupancy covariates because the effect of some variables (e.g., important trees and open areas cover) may change as the forest cover increases.

All models were fitted using the single-season occupancy model (MACKENZIE *et al.*, 2006) incorporated in program MARK (WHITE; BURHNAM 1999). Using our global model, we conducted a goodness-of-fit test and estimated overdispersion ($\hat{c}$) using the parametric bootstrap procedure (MACKENZIE; BAILEY, 2004) implemented in program PRESENCE (software version 2.12.20; HINES, 2006). We evaluated model selection using second-order Akaike Information Criterion (AICc) or Quasi-Akaike Information Criterion (QAICc; BURNHAM and ANDERSON, 2002), selecting only those models with ΔAICc< 2 (MACKENZIE *et al*., 2006). In addition, we calculated cumulative model weights (w_+) for each explanatory variable in our balanced set of models associated with each of the four steps described above (BURNHAM; ANDERSON, 2002). Then, we used estimated effects from our best-supported model to generate predict maps for the maned sloth distribution at the study area.

3. Results

We detected the maned sloth 53 times at 32 out 68 sampled sites (naïve occupancy estimate = 47%). Among the sites with sloth detections, 53% had only one detection, 34% had two detections, and 6% had three or four detections. The forest cover around the sampled sites range from 5% to 100%, open areas cover ranges from 0 to 69%, the proportion of important trees range from 0 to 1 and canopy height range from 8.9 to 30.5 m (Table A.3).

The evaluation of the scale of effect (buffer size) for our two landscape variables suggested that a model that included forest cover at 500m and open area cover at 600m was approximately twice as likely as any other model in candidate set (Table A.1). These scales of effects were used in all subsequent modeling. The parametric bootstrap goodness of fit revealed some evidence of overdispersion ($\hat{c}$=2.59), so we used Quasi-AICc (BURNHAM; ANDERSON, 2002) for model selection procedures. Our best-supported forest threshold relationship suggested that maned sloth occupancy declined when forest cover was below 30-35% (Tables 3A, A.4).We found some evidence that detection probability was positively affected by canopy height (w_+ = 0.32; $\hat{\beta}$ = 0.09, SE($\hat{\beta}$) = 0.07; Fig. 2), but a model with constant detection probability was more parsimonious (model weight = 0.44, Table 3B). To ensure less bias in our occupancy estimates, we used the more general structure where detection could vary among sites with different canopy height in our evaluation of factors influencing sloth occupancy.

Table 3: Model selection results for supported models of: (A) thresholds effects sloth occupancy and forest cover (%), (B) detection probability structure and (C) occupancy probability structure. The general detection structure, *p* (Canopy Height+ ImpTrees), was used for all threshold models (A). The best-supported forest cover threshold value and additional covariates were included in the general occupancy structure (Ψ (Open areas cover + ImpTrees + Canopy + 35% Forest Cover Threshold) used to evaluate the four detection probability structures (B). To reduce potential bias, we used the supported *p* (Canopy Height) detection structure when evaluating factors influencing sloth occupancy (C). Measurements of model fit (-2Log(L)), number of parameters, and Quasi-AICc values are comparable across all model evaluation steps (A, B, and C). Model weights and difference in QAICc (ΔQAICc) are calculated within steps only to help identify the best supported model structure within a given step. The bolded model represents the most parsimonious model in the candidate model set and the constant occupancy structure is given as a reference. ImpTrees = Important Trees.

Model	QAICc	ΔQAICc	*w*	-2Log(L)	*K*
(A) Threshold Relationship: Ψ (Open areas cover + ImpTrees + Canopy Height + *Threshold Relationship*)					
35% Forest Cover Threshold	**117.32**	**0**	**0.21**	**250.07**	**8**
30% Forest Cover Threshold	117.70	0.37	0.18	251.02	8
40% Forest Cover Threshold	118.09	0.77	0.15	252.01	8
(B) Detection probability structure					
p (.) – constant detection structure	114.30	0	0.44	255.22	6
***p* (Canopy Height)**	**114.98**	**0.68**	**0.32**	**250.64**	**7**
p (ImpTrees)	116.56	2.26	0.14	254.63	7
p (Canopy Height + ImpTrees)	117.33	3.03	0.01	250.07	8
(C) Occupancy structure					

Ψ (35% Forest Cover Threshold)	**110.24**	**0**	**0.20**	**255.92**	**4**
Ψ (35% Forest Cover Threshold+ Open area cover)	111.09	0.86	0.13	252.11	5
Ψ (35% Forest Cover Threshold+ ImpTrees)	111.35	1.12	0.12	252.748	5
Ψ (.) – constant occupancy structure	115.01	4.77	0.02	273.98	3

Among competing occupancy models, three had the ΔQAICc <2 (Tables 3C, A.5). Our best-supported model suggests that sloth occurrence is extremely high (near 1.0) when there is over 35% forest cover surrounding a site, but sloth occurrence drops rapidly when forest cover is below this threshold (Fig. 2). Our data also suggested that sloths do not occur at sites with ≤20% of forest cover (Fig 2). As expected, the proportion of important trees showed a positive relationship with occupancy probability and the percentage of open areas cover had a negative influence on sloth occurrence (Fig. 2). We found no evidence of interactive relationship between the forest cover and the other three variables (Table A.5), possibly due to our distribution of sampling sites. Landscape variables have the most pronounced importance: forest cover had the highest cumulative weight ($w_+ = 0.78$), followed by the open area cover ($w_+ = 0.46$). The two local variables had the lowest cumulative weights $w_+ < 0.40$ (Fig. 3).

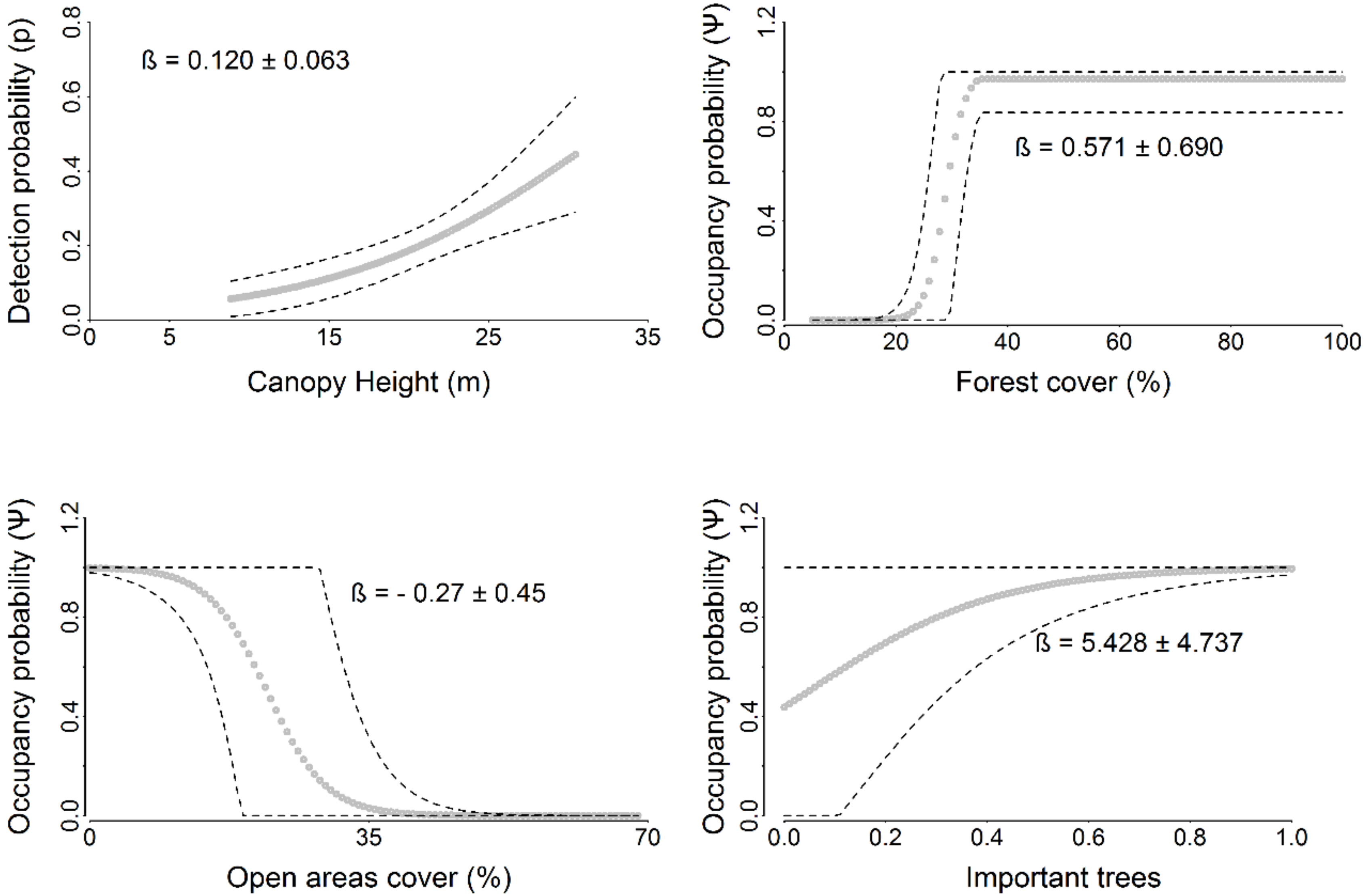

Detection probability (p)
0.0
0.2
0.4
0.6
0.8
ß = 0.120 ± 0.063
0
5
15
25
35
Canopy Height (m)
Occupancy probability (Ψ)
0.0
0.4
0.8
1.2
ß = 0.571 ± 0.690
0
20
40
60
80
100
Forest cover (%)
Occupancy probability (Ψ)
0.0
0.4
0.8
1.2
ß = - 0.27 ± 0.45
0
35
70
Open areas cover (%)
Occupancy probability (Ψ)
0.0
0.4
0.8
1.2
ß = 5.428 ± 4.737
0.0
0.2
0.4
0.6
0.8
1.0
Important trees

Fig. 2: Maned sloth detection and occupancy probabilities at 68 sampled sites within Atlantic Forest, Espírito Santo, Brazil, predicted by the three best-supported models. The slope parameters (betas) estimated for the covariates “forest cover”, “open areas cover” (second ranked model) and “important trees” (third ranked model) are also shown. The gray line indicates the estimated covariate relationship and the black dash indicate the standard error. The relationships between open areas and important trees are graphed using the mean forest cover (%).

The slope parameters (betas) estimated for the covariates “forest cover”, “open areas cover” (second ranked model) and “important trees” (third ranked model) are also shown. The gray line indicates the estimated covariate relationship and the black dash indicate the standard error. The relationships between open areas and important trees are graphed using the mean forest cover (%).

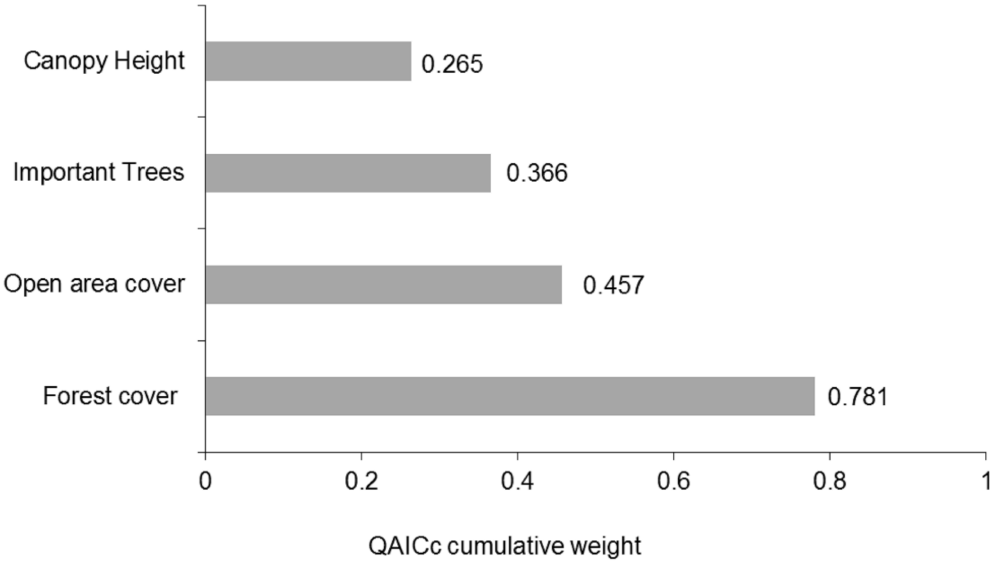


Fig. 3: Cumulative weight of evidence (QAICc cumulative weight) for each covariate used to model maned sloth occupancy within Atlantic Forest, Espírito Santo, Brazil.

Our predictive maps generated from the top three occupancy models show the maned sloth occurring with higher probabilities in a relatively wide band that stretches approximately north-southward in the study region (Fig. 4). In all three predictions, the maned sloth has a high occupancy probability in the northeast and central part of the study area. Conversely, the sloth is almost absent in the areas in the south, north and east edge – which corresponds to the areas with low forest cover.

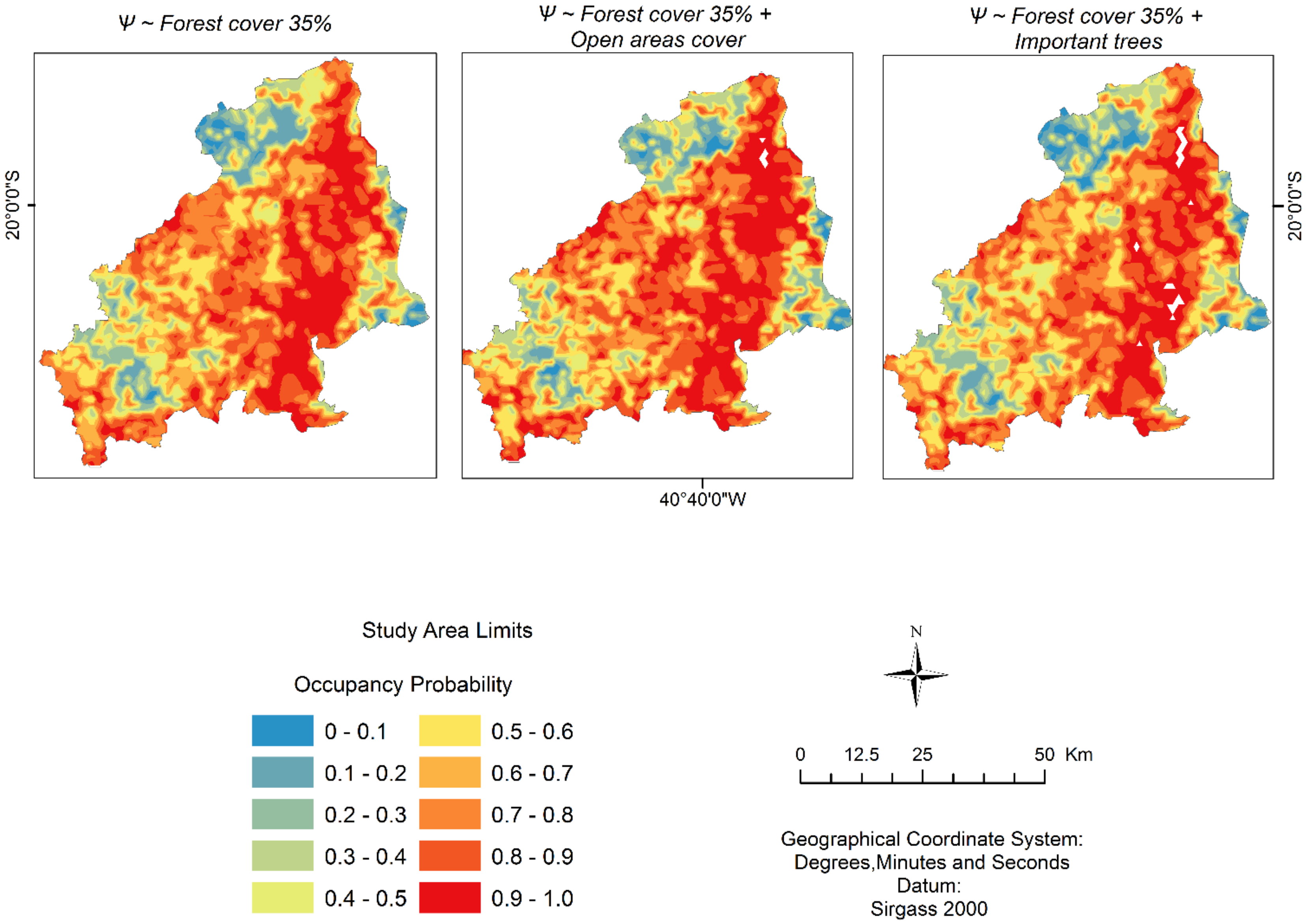

Ψ ~ Forest cover 35%
Ψ ~ Forest cover 35% + Open areas cover
Ψ ~ Forest cover 35% + Important trees
20°0'0"S
20°0'0"S
40°40'0"W
Study Area Limits
Occupancy Probability
0 - 0.1
0.1 - 0.2
0.2 - 0.3
0.3 - 0.4
0.4 - 0.5
0.5 - 0.6
0.6 - 0.7
0.7 - 0.8
0.8 - 0.9
0.9 - 1.0
N
0
12.5
25
50 Km
Geographical Coordinate System: Degrees,Minutes and Seconds
Datum: Sirgass 2000

Fig. 4: Predicted estimates of maned sloth occupancy probability in the study area within Atlantic Forest, Espírito Santo, Brazil; the three best-supported occupancy models were used.

4. Discussion

We found a threshold effect for the occupancy probability of the threatened and cryptic maned sloth. In contrast to previous study (SANTOS *et al.*, 2016), our results suggest that both forest cover and open areas cover strongly affect maned sloth occurrence across forest fragmentation gradients. The effect of local variables on detection probability at occupied sites was similar to our previous study, with canopy height positively influencing the maned sloth’s detection. This positive relationship was contrary to our expectation and may be related to the higher abundance of the maned sloth in taller forests which, apparently, overcompensates the difficulties in finding sloths high up in the canopy.

4.1 – Thresholds effects on occupancy probability

The best-supported model predicts that areas with more than 35% of forest cover (65% of the study area) are likely to support maned sloth ($\hat{\psi}$ = 0.97, SE ($\hat{\psi}$) = 0.13), but occupancy estimates decrease abruptly in areas below that threshold (Fig. 2), which corresponds to 35% of the study area (Fig. 4). Other habitat specialist groups exhibit similar responses to changes in habitat amount, including small to medium-sized terrestrial mammals (ESTAVILLO; PARDINI; DA ROCHA, 2013; OCHOA-QUINTERO *et al.*, 2015), woody plants (LIMA; MARIANO-NETO, 2014), and birds (ZUCKERBERG; PORTER, 2010). We also observed the absence of the maned sloth in areas with less than 20% of forest cover (15% of the study area; Fig.4). Forest loss in the region is already severe (originally 100% forest cover) and additional forest loss will increase the isolation of remaining forest patches and decrease patch size. These processes are likely to lead to the local extinction of sloth in some patches and reduce sloth movement between patches making recolonization difficult.

As a forest specialist, habitat loss represents one of the major threats for the maned sloth. Landscapes with high concentration of natural habitats have smaller inter-patch distances and large numbers of patches (number of patches peaks at 30%, see GUSTAFSON; PARKER, 1992). This configuration allows the maned sloth – a strictly arboreal species – to move easily between forest patches. As the forest cover decreases, patch isolation and matrix resistance increase, followed by a decrease in patch size (ANDRÉN, 1994; SWIFT; HANNON, 2010). Isolated and small fragments might no longer support viable populations of maned sloth. In general, community composition may exhibit a shift (PARDINI *et al.*, 2010;

BANKS-LEITE *et al*., 2014; MAGIOLI *et al*., 2015), holding mainly generalists species, since they can exploit different types of habitat to complement their diet and behavior or resource needs (ESTAVILLO; PARDINI; DA ROCHA, 2013; LIMA; MARIANO-NETO, 2014).

4.2 – Interactions between Forest Cover and local/landscape levels

Contrary to our expectations, we found no evidence of interaction between the forest cover and the local variables, though our findings may be influenced by our limited number of sites with forest cover values between 20-35%, where sloth occupancy estimates are greater than 0, but less than 1 (Fig 2). Forest structure and composition play an unquestionable role in sustaining high biodiversity (AUGUST, 1983). Habitat loss may cause important changes in the habitat structure of remaining patches, with an increase of light demanding arboreal species (early successional), and canopy openness, and a decrease of overall basal area (ROCHA-SANTOS *et al.*, 2016; BENCHIMOL *et al*., 2017). Species richness may decrease within patches (ANDRADE *et a*l., 2015; MAGIOLI *et al.*, 2015), and the compositional dissimilarity of plants between fragments may increase (ARROYO-RODRÍGUEZ *et al.*, 2013; BENCHIMOL *et al.*, 2017). This simplification of habitat composition and structure can limit food resource for the maned sloth, since in general, the sloths have a very restrictive diet, feeding on a few arboreal species (CHIARELLO, 1998b; MONTGOMERY; SUNQUIST, 1975), including some shade tolerant species (CHIARELLO, 1998b). The reduction of canopy height and increase of canopy openness can also influence sloths, since they spend most of their time on high canopy strata resting, feeding, breeding, and executing thermoregulatory behavior (MONTGOMERY; SUNQUIST, 1975; PAULI *et al*., 2016; GARCÉS-RESTREPO *et al*., 2017).

Our second-best model suggests that sloth occupancy is influenced by both the open areas cover and forest cover, with a strong negative effect of open areas (Fig 2). Therefore, open areas may limit maned sloth movements, and the species may avoid it – a pattern also found previously (FALCONI *et al*., 2015). Sloths can survive in small fragments of forest surrounded by an intermediate forest cover (20-35%) due to their relatively small home range (CHIARELLO, 1998b; FALCONI *et al.,* 2015). They can exploit living fences (*e.g.* narrow linear strips of planted trees; LEÓN; HARVEY, 2006), isolated trees, forestry systems, and agriculture crops if these habitat patches provide food resources or are located between potentially accessible forest fragments (CASSANO; KIERULFF; CHIARELLO, 2011;

PEERY; PAULI, 2014; VAUGHAN *et al*., 2007). This is also true for other arboreal mammal, such as *Alouatta palliata mexicana* (ASENSIO *et al*., 2009), *Colobus anglensis palliatus* (ANDERSON; ROWCLIFFE; COWLISHAW, 2007), and *Callicebus personatus* found in our study area (Pers. Observation). The use of living fences, isolated trees, forestry systems and agriculture crops – habitats with poor resources and conditions – by animals that are considered strictly arboreal provides evidence of the species adaptive capacity to maximize resource consumption in more anthropogenic landscapes due to the lower quality of existing forest fragments. However, the use of open areas may represent a severe predation hazard since the sloths, and other arboreal species, are frequently exposed to attacks by domestic and feral dogs *(*VAUGHAN *et al., 2007;* OLIVEIRA *et al*., 2008; GARCÉS-RESTREPO *et al*., 2018). Similarly, since disturbed forests are of low quality, the sloths would be foraging on a lower and more open canopy, becoming more exposed to predation by ocelot (*Leopardus pardalis* - DELIBES *et al*., 2011) and tayra (*Eira Barbara* - SÁENZ-BOLAÑOS *et al*., 2018). In addition, the energy expenditure to cross larger open areas may be higher than its putative resource gains, since the sloth anatomy is not adapted to movements on the ground (GOFFART, 1971).

4.3 – Implications for conservation

We provide the first assessment of the maned sloth habitat requirements in a broader landscape context, representing the full spectrum of forest cover gradient, and with a multi-scale perspective covering a wide range of scale of effects. This understanding can lead to better targeted conservation actions, including a more accurate assessment of the species conservation status and threats. The use of thresholds in conservation and management actions might be debated, mainly because the threshold value may exclude species that require more contiguous habitat (JOHNSON, 2013; VAN DER HOEK; ZUCKERBERG; MANNE, 2015). Additionally, when habitat loss is severe, biodiversity recovery may require different conservation approaches and the previous conservation state may never be obtained (JOHNSON, 2013; VAN DER HOEK; ZUCKERBERG; MANNE, 2015). Still, maintaining 30% to 35% of forest cover in an area can be important for a wide diversity of species and may conserve several ecological process essential to ecosystem function(BANKS-LEITE *et al.*, 2014; MARTENSEN *et al.*, 2012; MUYLAERT; STEVEN; RIBEIRO, 2016).

To improve maned sloth conservation, our results support the need for an increase in forest cover, which will also increase habitat connectivity, by: 1) restoring and increasing riparian vegetation and 2) increasing matrix permeability with agroforestry systems and living fences. The Atlantic Forest has lost much of its riparian vegetation, and if this vegetation could be restored, the forest cover could increase to 30% (REZENDE *et al.*, 2018), approaching the threshold level found in our study. Sloths typically use riparian forest for foraging and dispersal (RAMIREZ *et al.*, 2011; GARCÉS-RESTREPO *et al.*, 2018) and an early study observed maned sloths using swamp forests in Poço das Antas Reserve (PINDER, 1985). These areas are also useful for birds (TREMBLAY; ST. CLAIR, 2011) butterflies (VAN HALDER *et al.*, 2015), and bats (MUYLAERT; STEVEN; RIBEIRO, 2016; WORDLEY *et al.*, 2015). Additionally, riparian vegetation has a positive impact on water quality by reducing the nutrient and sediment load in streams (DOSSKEY *et al.*, 2010).

Still, conserving and restoring riparian vegetation may not be enough to ensure the conservation of maned sloth. According to the Brazilian Native Vegetation Protection Law (Federal Law 12,727/2012), 20% of forest areas in rural properties must be set aside as legal forest reserves. However, this value is far below our threshold of 35%, and we found that maned sloth is unlikely to exist in areas with less than 20% of forest cover (Fig. 2). Thereby, we reinforce that areas with less than 20% forest cover are unlikely to sustain viable populations of sloths. Additionally, the current law incorporates riparian vegetation in the calculation of the legal reserves, resulting in future deficits in the overall vegetation (SOARES-FILHO *et al.*, 2014), including our study area where 97.3% of forest cover are privately owned. To guarantee more consistent conservation, the restoration of riparian forest must be followed by forest reforestation in other areas.

By mixing native vegetation with agricultural production, the agroforestry systems may be a sustainable way of reducing the contrast between the forest cover and the matrix, along with increasing forest area (SCHROTH; HARVEY, 2007; SANTOS *et al.*, 2019). When combined with other forested areas, these systems can offer more suitable habitat for maned sloths than the traditional agricultural system (CASSANO; KIERULFF; CHIARELLO, 2011) Moreover, providing different trees in the form of living fences yield new resource areas, resting refugia, and can act as movement corridors between habitat patches for the sloths, enhancing the landscape connectivity and reducing the pressure to traverse the anthropogenic open matrix (CASTELLÓN; SIEVING, 2006; LEÓN; HARVEY, 2006). Together, the restoration and increase of riparian vegetation and the creation of living fences and biodiverse agroforestry system may provide useful corridors and habitats for the maned sloth and for a wide range of species, thus allowing a long-term biodiversity conservation.

In this paper, we found a threshold relationship between maned sloth occurrence and forest cover using an occupancy modeling framework. This framework is recommended for species with low detection probabilities, such as sloth species. Sloth occurrence declined quickly in areas with less than 35% forest cover and sloth were unlikely to occur in areas with less than 20% forest cover. Future studies aiming at understanding the potential interaction between landscape and local variables for this species may concentrate effort within this forest cover range and could provide valuable information for conservation efforts in degraded forest areas. In addition, we strongly recommend long-term monitoring of sloth in our study region – as well as in other key distribution areas of the species –incorporating movement monitoring, to estimate changes in occupancy over time and determine sloth movements and resource use within and between forest patches surrounded by different types of non-habitat and poor-habitat matrix. Such studies would make it possible to better infer how the maned sloth uses and selects their habitats in a heterogeneous and anthropogenic landscape, improving the management and conservation of this lesser known species.

Acknowledgements

We thank LB`s Lab for useful comments on earlier versions of the manuscript and the two anonymous reviewers for the useful comments on an early version. PMS research grants from The Rufford Foundation (21498-1) and scholarship from CNPq (141571/2017-9) and CAPES (88881.188493/2018-01). MCR thanks FAPESP (process #2013/50421-2), CNPq (processes # 312045/2013-1; #312292/2016-3) and PROCAD/CAPES (project # 88881.068425/2014-01) for their financial support. AGC has a Research Productivity Grant from CNPq (303101/2017-2). APP research support from FAPEMIG and CAPES. We thank to Sérgio L. Mendes and all his lab team, to the field work support, specially Nila R.C. Gontijo. We thank all the landowners for the authorization of research on their areas, and to ICMBio/SISBIO (License 57979), MNRJ and SEMAST (License 78343860) for the authorizations of research on protected areas.

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Appendix A. Supplementary data

Table A.1: Model selection results for supported models of scale of effect. The same general model structure was used for sloth occupancy, Ψ (Canopy Height + ImpTrees + Forest cover *scale of effect* + Open areas cover *scale of effect*), varying the scale (buffer, m) of each landscape variable. Detection probability was modeled as a function of both local covariates, *p* (Canopy Height +ImpTrees). For each model we report: AICc = Akaike information criteria corrected for small samples; *w* = Model Weight; -2Log(L) = Measurements of model fit; K = number of parameters. The bolded model represents the most parsimonious model in the candidate model set.

Model	AICc	ΔAICc	*w*	*K*	-2log(L)
General structure: Ψ (Canopy Height + ImpTrees + Forest cover scale of effect + Open areas cover scale of effect), p (Canopy Height +ImpTrees)					
Forest cover 500 + Open areas cover 600	**271.02**	**0.00**	**0.09**	**8**	**252.58**
Forest cover 600 + Open areas cover 600	272.33	1.31	0.05	8	253.89
Forest cover 500 + Open areas cover 400	272.47	1.45	0.04	8	254.03
Forest cover 600 + Open areas cover 400	272.47	1.45	0.04	8	254.03
Forest cover 100 + Open areas cover 400	272.56	1.54	0.04	8	254.12
Forest cover 400 + Open areas cover 400	272.59	1.57	0.04	8	254.15
Forest cover 600 + Open areas cover 500	272.72	1.69	0.04	8	254.28
Forest cover 300 + Open areas cover 400	272.74	1.72	0.04	8	254.30
Forest cover 500 + Open areas cover 500	272.79	1.77	0.04	8	254.35
Forest cover 100 + Open areas cover 300	272.79	1.77	0.04	8	254.35
Forest cover 200 + Open areas cover 400	272.90	1.87	0.04	8	254.45
Forest cover 500 + Open areas cover 300	273.08	2.06	0.03	8	254.64
Forest cover 600 + Open areas cover 300	273.09	2.07	0.03	8	254.65
Forest cover 100 + Open areas cover 200	273.10	2.07	0.03	8	254.66
Forest cover 400 + Open areas cover 600	273.10	2.08	0.03	8	254.66
Forest cover 400 + Open areas cover 300	273.19	2.16	0.03	8	254.75
Forest cover 300 + Open areas cover 300	273.23	2.21	0.03	8	254.79
Forest cover 400 + Open areas cover 500	273.24	2.22	0.03	8	254.80
Forest cover 200 + Open areas cover 300	273.28	2.26	0.03	8	254.84
Forest cover 100 + Open areas cover 500	273.43	2.41	0.03	8	254.99

Forest cover 300 + Open areas cover 600	273.50	2.48	0.03	8	255.06
Forest cover 300 + Open areas cover 500	273.60	2.58	0.02	8	255.16
Forest cover 100 + Open areas cover 600	273.81	2.78	0.02	8	255.37
Forest cover 200 + Open areas cover 500	273.81	2.79	0.02	8	255.37
Forest cover 300 + Open areas cover 200	273.81	2.79	0.02	8	255.37
Forest cover 400 + Open areas cover 200	273.83	2.81	0.02	8	255.39
Forest cover 600 + Open areas cover 200	273.86	2.83	0.02	8	255.42
Forest cover 200 + Open areas cover 200	273.86	2.83	0.02	8	255.42
Forest cover 500 + Open areas cover 200	273.86	2.83	0.02	8	255.42
Forest cover 200 + Open areas cover 600	273.95	2.93	0.02	8	255.51
Forest cover 100 + Open areas cover 100	278.60	7.58	0.00	8	260.16
Forest cover 300 + Open areas cover 100	279.27	8.25	0.00	8	260.83
Forest cover 200 + Open areas cover 100	279.48	8.46	0.00	8	261.04
Forest cover 500 + Open areas cover 100	279.63	8.61	0.00	8	261.19
Forest cover 400 + Open areas cover 100	280.82	9.80	0.00	8	262.38
Forest cover 600 + Open areas cover 100	280.85	9.83	0.00	8	262.41

Table A.2: Pearson Correlation Matrix for selected covariates.

	Canopy Height (m)	Important Trees	Open areas cover (%) in 600 m	Forest Cover (%) in 500 m
Canopy Height (m)	**1**			
Important Trees	0.39	**1**		
Open areas cover (%) in 600 m	-0.46	-0.32	**1**	
Forest Cover (%) in 500 m	0.25	0.27	-0.50	**1**

Table A.3: General characteristics for the sampled sites in study area. Where: Total mean – mean of the respective variable in al sampled sites; Non-detected = mean of the respective variable in all sites which the maned sloth were not detected; Detected = mean of the respective variable in all sites which the maned sloth were detected; m = meters; SD = Standard Deviation.

Variable	Total Mean (SD)	Non-detected (SD)	Detected (SD)
Forest cover (%)	48.72 (±21.05)	41.47 (±10.49)	56.87 (±20.96)
Open areas cover (%)	10.46 (±11.75)	16.59 (±17.29)	3.56 (±3.47)
Important Trees (0-1)	0.63 (±0.27)	0.53 (±0.31)	0.73 (±0.20)
Canopy Height (m)	19.59 (±3.34)	18 (±3.12)	21.38 (±3.41)
Total of the sampled Sites	68	36	32

Table A.4: Model selection results for eight models exploring thresholds effect between forest cover and the Maned sloth occupancy probability. All models include a general detection probability structure, *p* (Canopy Height+ ImpTrees), and an occupancy structure that accounted for potential variation associated with our other covariates, Ψ (Open areas cover + ImpTrees + Canopy Height + ***Threshold Relationship***). For each model we report: QAICc = Quasi-AICc; *w* = Model Weight; -2Log(L) = Measurements of model fit; *K* = number of parameters.

Model	QAICc	ΔQAICc	*w*	-2log(L)	*K*
Ψ ~ Forest cover_35% + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	117.33	0	0.213	250.07	8
Ψ ~ Forest cover_30% + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	117.7	0.37	0.18	251.02	8
Ψ ~ Forest cover_40% + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	118.09	0.77	0.15	252.02	8
Ψ ~ Forest cover_45% + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	118.35	1.02	0.13	252.65	8
Ψ ~ Forest cover_50% + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	118.51	1.20	0.12	253.12	8
Ψ ~ Forest cover_25% + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	118.93	1.61	0.1	254.13	8
Ψ ~ Forest cover + Open areas cover + Important trees + Canopy Height, *p* ~ Canopy Height + Important Trees	119.57	2.24	0.07	255.74	8

Ψ ~ Forest cover_20% + Open areas cover + Important trees + Canopy Height, p ~ Canopy Height + Important Trees	119.99	2.66	0.06	256.81	8

Table A.5: Model selection results for occupancy structures that included additive (16 models) and interactive effects (6 models) of our local and landscape variables on Maned sloth occupancy probability. All models include the supported detection probability structure, *p* (Canopy Height). For each model we report: QAICc = Quasi-AICc; *w* = Model Weight; -*2Log(L)* = -2 Log Likelihood; *K* = number of parameters; + = Additive models; * = Interactive models. The bolded model represents the three better models in the candidate model set (ΔQAICc < 2).

Model	QAICc	ΔQAICc	*w*	*-2Log(L)*	*K*
***Ψ* Forest cover_35%**	**110.23**	**0**	**0.20**	255.92	**4**
***Ψ* Forest cover_35% + Open areas cover**	**111.10**	**0.86**	**0.13**	252.11	**5**
***Ψ* Forest cover_35% + Important trees**	**111.35**	**1.12**	**0.12**	252.74	**5**
Ψ Forest Cover_35% + Canopy height	112.36	2.12	0.07	255.36	5
Ψ Open areas cover	112.46	2.22	0.07	261.58	4
Ψ Forest Cover_35% + Open areas cover + Important trees	112.66	2.43	0.06	249.91	6
Ψ Forest Cover_35% + Open areas cover+ Canopy height	113.40	3.17	0.04	251.83	6
Ψ Forest Cover_35% + Canopy Height + Important trees	113.47	3.23	0.04	251.96	6
Ψ Forest Cover_35% * Open areas cover	113.49	3.25	0.04	252.08	6

Ψ Open areas cover+ Important trees	113.639	3.40	0.04	258.56	5
Ψ Forest Cover_35% * Important trees	113.71	3.48	0.04	252.61	6
Ψ Open areas cover + Canopy height	114.62	4.39	0.02	261.09	5
Ψ Forest Cover_35% * Canopy height	114.68	4.44	0.022	255.14	6
Global model	115.00	4.74	0.02	249.42	7
Null model	115.01	4.77	0.019	273.98	3
Ψ Open areas cover+ Canopy height + Important trees	115.36	5.12	0.02	256.74	6
Ψ Open areas cover* Important trees	115.72	5.48	0.01	257.73	6
Ψ Canopy height + Important trees	115.75	5.517	0.01	263.91	5
Ψ Important trees	115.94	5.71	0.01	270.44	4
Ψ Canopy height	116.56	6.33	0.01	260.10	4
Ψ Open areas cover* Canopy height	116.62	6.38	0.01	272.12	6

Ψ Canopy height	118.13	7.89	0.00	263.80	6

*Important trees

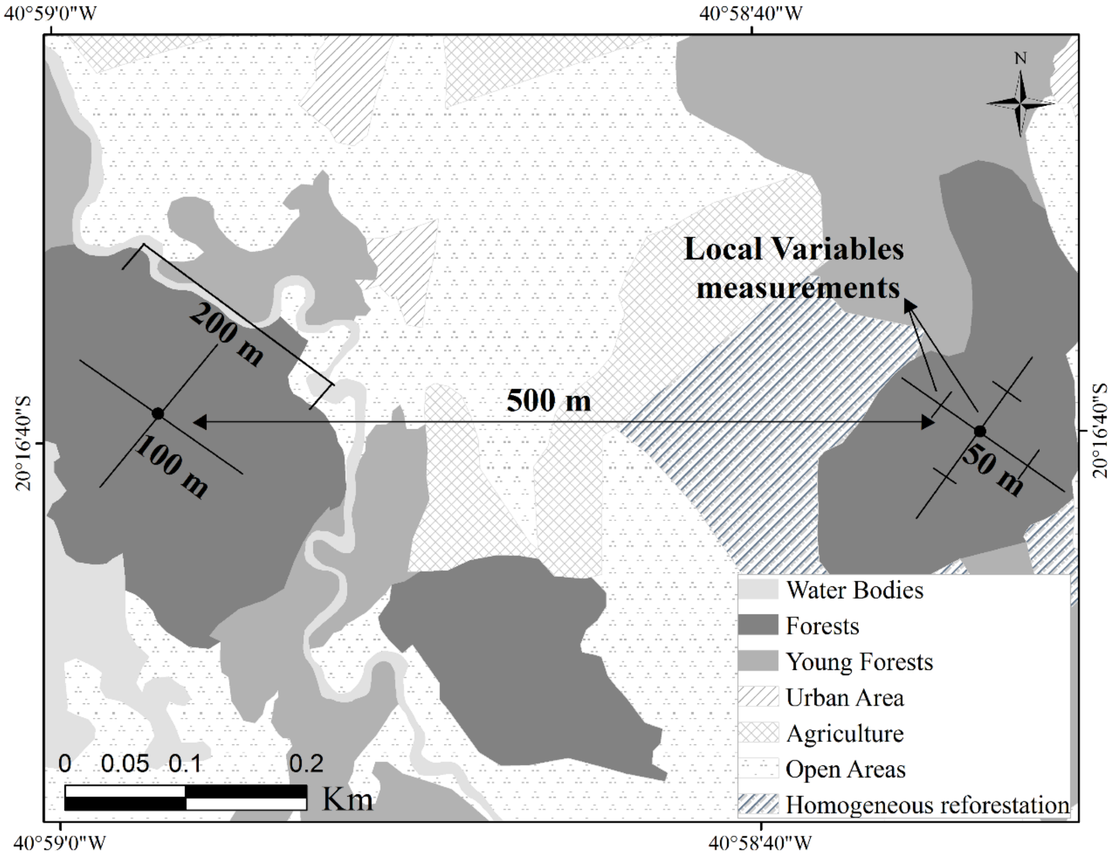

Fig. A.1: Schematic representation of sample design.

Capítulo 3

Natural regeneration can mitigate the climate change effects on suitable areas for the endangered maned sloth in Atlantic Forest, Brazil.

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Manuscript to be submitted in Global Change Biology

Abstract

Climate changes and habitat loss are two of the major factors responsible for the species extinction. Urgent conservation measures are needed to mitigate these global threats, such as forest regeneration. The maned sloth (*Bradypus torquatus*) is a arboreal and folivore species, with populations of the northern lineages (Sergipe and Bahia) being divergent from southern ones (Espírito Santo and Rio de Janeiro), being considered two distincts Evolutionary Significant Units (ESUs). This article sought to evaluate the environmental suitability for the endangered maned sloth using Species Distribution Models (SDM tools), and modeling the two evolutionary units separately – ESU01 (North) and ESU02 (South). We performed the analyses using the algorithm Maxent to model future scenarios, taking into account climate change in 2050 and three forest scenarios – BAU (without regeneration), scenario 02 (less optimistc – less regeneration) and scenario 03 (more optimisct – more regeneration), using four climate variables (two for each ESU) and two landscape variables. We analysed the models according to the brazilian legislation, to understand the conservation and legal protection of suitable areas. Suitable areas correspond to 17.71 % of the geographical space for ESU01, and 7.0% for ESU02. ESU01 exhibits a decrease of suitable areas in scenarios BAU and 02, but a stability in scenario 03. On the other hand, ESU02 exhibits a increase of suitable areas in all scenarios. For both populations, forest cover play and important role in the current environmental suitability, but this pattern may change in future scenarios, mainly in ESU02, in which the climate variable gain importance. Most part of suitable areas are outside from any legal protection, and this patterns continue in future scenarios. The future conservaion of *B.torquatus* will depend on landscape planning, in order to include different types of forest regeneration.

Keywords: Conservation, Forest regeneration, Landscape ecology, Global warming, Pilosa

1. Introduction

Successful biodiversity conservation is intrinsically linked to the our capacity to foresee how or if a given habitat will support viable populations of different species in the future. Climate change is expected to act as the primary actor in the following years, with several impacts on biodiversity distribution (ARAÚJO; RAHBEK, 2006; MORAES *et al.*, 2019), species richness and composition (GOUVEIA *et al.*, 2016; LIMA *et al.*, 2019), and suitable areas for viable populations (PRÉAU *et al.*, 2019; ZHANG *et al.*, 2020). Human activities, such as agriculture, forestry, and other land uses are responsible for 23% of global greenhouse gas emissions (IPCC, 2018). In addition, the conversion of native vegetation into anthropogenic areas directly affects the biodiversity, creating inhospitable enviroments for a variety of species (MORAES *et al*., 2018). Several biomes are experiencing high rates of habitat loss and decreasing biodiversity, since small and isolated habitats may be inadequate to sustain viable populations (FAHRIG, 2003; SILVA *et al.*, 2015).

Globally, forest areas play a fundamental role in regulating climate, mitigating the effects caused by climate (PREVEDELLO *et al.*, 2019; SILVÉRIO *et al.*, 2015); thus, reducing deforestation is imperative and urgent. Additionally, the regeneration of natural habitats is crucial to reduce the impacts of global warming (BAPTISTA; RUDEL, 2006; RUDEL *et al.*, 2005; RUDEL; BATES; MACHINGUIASHI, 2002), since growing young forests are excellent carbon sinks (CASPERSEN *et al.*, 2000; STRASSBURG *et al.*, 2014). Deforested areas can be recovered in two ways: 1) active restoration and 2) passive — or natural — regeneration (CHAZDON, 2012). The latter is considered the cheapest form of forest recovery, since it is simply a natural process of ecological succession (CHAZDON, 2012). The former involves human labor for planting seedlings and seeding directly and is therefore costlier, especially for small landowners (LAMB; ERSKINE; PARROTTA, 2005; but see BANKS-LEITE *et al*., 2014). Besides the forest recovery and regeneration, protect the current natural areas is of utmost importance to ensure some reduction of climatic effects and guarantee the conservation of biodiversity (THOMAS, 2010; VELASCO *et al.*, 2019). Some conservations initiatives are fundamental in order to ensure the long-term preservation of natural environments and the species that inhabit them, such as the establishment of a protected areas system (NAUGHTON-TREVES; HOLLAND; BRANDON, 2005) and the conservation of natural areas in private lands (BOWERS, 1999; METZGER *et al.*, 2019).

The Brazilian Atlantic Forest has been reduced to just over 28% of its original size (REZENDE *et al.*, 2018), and is composed mainly of fragments smaller than 50 ha that are immersed in heterogeneous matrix (RIBEIRO *et al.*, 2009), and highly vulnerable to climate change (BELLARD *et al.*, 2014; SOBRAL-SOUZA *et al.*, 2018). Besides, the biome has been experiencing a increase in deforestation rates of 27%, between 2018 and 2019 (FUNDAÇÃO SOS MATA ATLÂNTICA, 2020). Nevertheless, some regions report an increase in forest cover, mainly due to natural succession processes (BAPTISTA; RUDEL, 2006; DE REZENDE *et al.*, 2015; TEIXEIRA *et al.*, 2009).

Even with its long history of deforestation, the biome is still home to numerous endemic and threatened species, such as the maned sloth (*Bradypus torquatus),* according to the national red list (MMA Ordinance No. 444/2014; CHIARELLO *et al*., 2018) and by the international list (IUCN. CHIARELLO ; MORAES-BARROS, 2014). Due to their strictly arboreal and folivorous habit (CHIARELLO, 1998a, b), maned sloths need sizeable forest cover for their survival and are practically absent in areas with less than 20% forest cover (SANTOS *et al*., 2019b). Additionally, the presence of large open areas such as abandoned pasture lands has a detrimental effect on them (FALCONI *et al.*, 2015; SANTOS *et al.*, 2019b). Since they are highly adapted and restricted to the arboreal environment, a very high energy expenditure is required to cross these open areas and doing so exposes them to opportunistic predation events (PEERY; PAULI, 2014; VAUGHAN *et al.*, 2007). Additionally, the activity patterns of maned sloths might be strongly influenced by the variations in environment temperature, due to a labile body temperature (CHIARELLO 1998a; GINÉ *et al.*, 2015). Climate change may affect the structure, coverage and composition of natural areas (VELAZCO *et al*., 2018; PREVEDELLO *et al*., 2019), impacting the supply of resources and conditions - essential for the maintenance of sloth populations (CHIARELLO, 2008; SANTOS *et al*., 2016). *Bradypus torquatus* occurs in environments from sea level to almost 1300 m, predominantly in altitudes from 0 to 299 m (HIRSCH; CHIARELLO, 2012). Higher elevations are usually colder - a limiting factor for the *B. torquatus* (McNAB, 1985; HIRSCH; CHIARELLO, 2012).

Bradypus torquatus has a restricted and discontinuous distribution in the Atlantic Coast forest, (HIRSCH; CHIARELLO, 2012; SANTOS *et al*., 2019a), inhabiting fragments in four Brazilian states—Sergipe, Bahia, Espírito Santo, and Rio de Janeiro (HIRSCH; CHIARELLO, 2012; SANTOS *et al.*, 2019a). Overall, populations of *B. torquatus* are genetically structured, with northern lineages (Sergipe and Bahia) being divergent from southern ones (Espírito Santo and Rio de Janeiro) to the point of being considered two distincts Evolutionary Significant Units (ESUs) (SCHETINO; COIMBRA; SANTOS, 2017). This diversification probably occurred during the Miocene-Pliocene transition (SCHETINO; COIMBRA; SANTOS, 2017), and the current natural boundaries between the two lineages is located in northern Espírito Santo (from the limit with Bahia, to the north of Rio Doce). Consequently, it is extremely important to treat these two lineages as if they were different species, especially in relation to conservation strategies (MOREIRA *et al.*, 2014; SCHETINO; COIMBRA; SANTOS, 2017).

Species Distribution Models (SDM) are an important tool given its capacity for predicting areas where the species in question is likely to be found (ELITH; LEATHWICK, 2009). Through occurrence data and environmental variables, this tool allows researchers to understand the relationship between the distribution of a given species (e.g. geographical space) and the environmental conditions (e.g. environmental space), and thus to map environmentally suitable areas for the species (PULLIAM, 2000; SOBERÓN, 2007; SOBERÓN; NAKAMURA 2009; FERRAZ *et al.*, 2012). The approach has been widely used to identify priority areas for species conservation (CARRASCO *et al.*, 2020; PORTUGAL *et al.*, 2019; RODRIGUES *et al.*, 2004), define areas for stablishing ecological corridors (MORATO *et al.*, 2014), predict potential hybridization zones between native and invasive species (MORAES *et al.*, 2019), and identify possible effects of habitat regeneration on species conservation (ANGELIERI *et al.*, 2016), among other uses.

Due to its physiological characteristics (CHIARELLO 1998a; GINÉ *et al.*, 2015) and its dependence to forest environments (SANTOS *et al.*, 2019b), climatic stability (low seasonality, mild temperatures, etc.) and increasing forest areas act positively for the occurrence of *B. torquatus* (MOREIRA *et al.*, 2014; SANTOS *et al.*, 2019b). Yet, it is uncertain how climate and landscape changes will interact in the future. Using the Species Distribution Models (SDM) and considering the two evolutionary units, this novel study aimed to 1) Evaluate the environmental suitability for *B. torquatus* throughout its distribution; 2) Assess environmental suitability in the future, using natural forest regeneration and climate change scenarios; and 3) Analyze the resulting scenarios - present and future - in accordance with the Brazilian environmental legislation. Thus, we evaluated whether suitable areas are currently legally protected and whether the gains of suitable areas will be protected in the future.

2. Material and Methods

2.1 Geographical space

The Atlantic Forest covers part of the Brazilian east coast, extending to more inland areas in its southern distribution (MUYLAERT *et al.*, 2018). Throughout its extent, the biome encompasses different vegetation types, and the ombrophilous formations - preferred by *B.torquatus* - are located near coastal areas and in the eastern slope of Serra do Mar (OLIVEIRA-FILHO; FONTES, 2000). We delimited the geographical space using the WWF terrestrial ecoregions database (OLSON *et al.*, 2001), selecting only the vegetation types present in the current known distribution of *B.torquatus* (HIRSCH; CHIARELLO, 2012). We also included ecoregions present in the most northerly areas of ecoregions, comprising the states of Pernambuco (PE), Paraiba (PB), and Rio Grande do Norte (RN, Fig. 1). Such coverage is justified by the fact that there are reports of the species in such areas, although this has never been confirmed (HIRSCH; CHIARELLO, 2012). The defined area is 237,936 km² (ESU01 – 123,516 km²; ESU02 – 114,420 km²), encompassing coastal areas in eight states and consisting mainly of perennial vegetation, sandbanks, and mangroves (Fig. 1).

2.2 Modeling framework

In this study, we opted to model current and future suitable areas separately for each of the Evolutionary Significative Units (ESU) of *B. torquatus* (SCHETINO; COIMBRA; SANTOS, 2017). We used ESU01 and ESU02 to refer to the two modeled areas (Fig. 1): ESU01 corresponding to the northern population ,located in Sergipe and Bahia, and ESU02 to the southern population, present in Espírito Santo and Rio de Janeiro (SCHETINO; COIMBRA; SANTOS, 2017).

First, we modelled the current scenario, using current climate data and current landscape data. Then, we modeled the future environmental suitability surfaces in 2050, according to three scenarios: 1) Scenario 01, Business as Usual (BAU), projected future climate, but without changes in landscape. In this case we analyzed the losses/gains of suitable areas only from the perspective of climate change. The other two scenarios correspond to the analysis of losses/gains of suitable areas from the perspective of climate change and of regeneration scenarios: 2) Scenario 02, less optimistic, projected future climate and changes in landscape, with a small rate of regeneration, only enough to maintain the species, i.e., 35% of forest cover, a threshold level of forest cover we detected previously (Santos *et al*., 2019b); and 3) Scenario 03, more optimistic, projected future climate and changes in landscape with high rate of regeneration, taking into account all areas with some potential for regeneration. We used the same variables for present and for future scenarios.

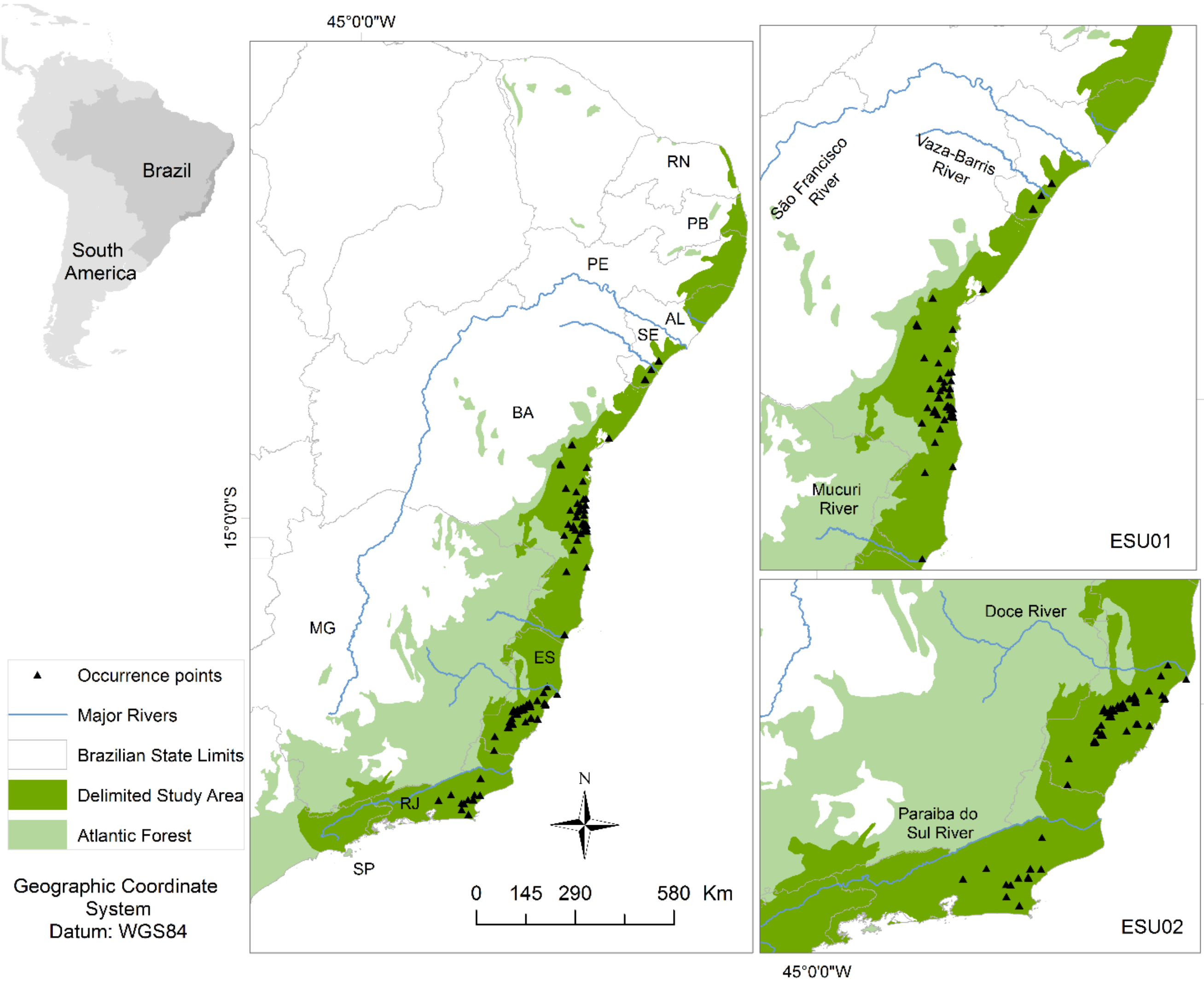


Fig. 1 Study area delimitated by WWF Ecorregions (Olson *et al*., 2001) and distribution of presence data in each Evolutionary Significant Units (ESU).

2.3 Presence data and sampling bias

To construct the environmental suitability models we compiled *B. torquatus* presence data available from Neotropical Xenarthrans datapapers (SANTOS *et al*., 2019a). This database is a vast compilation of data from various sources and researchers, including data that were previously unknown and inaccessible, totaling 638 records. Subsequently, we filtered all presence points in four steps: 1) all points with coordinate accuracy above 100 meters were eliminated. The data paper included the information about coordinate precision, allowing us select only location points, and not locality; 2) all points related to apprehension, interview, and roadkill were eliminated, since these are often not representative of the real location of the species and are often inaccurate; 3) only data obtained from 2009 to present were selected, to be temporally consistent with the environmental data used; 4) we maintain all points with a minimum distance of 500 m to avoid spatial correlation (SANTOS, *et al.*, 2016, 2019b). At the end, a database with 164 points were generated, consisting of 94 points from ESU01 and 70 points from ESU02 (Fig. 01). Finally, we developed a sampling probability surface, called bias grid, to reduce the sampling bias in areas more intensely sampled than others (PHILLIPS *et al.*, 2009) by decreasing the weight of these locations. The bias grid was created by calculating the Gaussian Kernel density of the sampling locations with a distance of 30 km. This value best represents the two large clusters of sampling points - in the south of Bahia and in the mountain region of Espírito Santo.

2.4 Environmental space: Environmental variables and future scenarios

We obtained the bioclimatic data through the WorldClim data base (WordClim V.2) and elevation data through USGS (United States Geological Service) (Table A.1). Since the bioclimatic variables are generally correlated with each other, a factor analysis with maximum variation rotation was performed, separately for each ESU, to check for multicollinearity. We selected only 2 bioclimatic variables for the ESU: Precipitation of Wettest Month (Bio13) and Precipitation Seasonality (Bio15) for ESU01 and Precipitation of Driest Month (Bio14) and Temperature Seasonality (Bio04), for ESU02. For future climate scenarios and to evaluate the effect of climate change, we selected the bioclimatic variables from two Representative Concentration Pathways (RCPs), related to different scenarios of greenhouse gas emissions for 2050 (mean for the period from 2041 and 2060). The RCP 4.5 represent an optimistic climate scenario, and RCP 8.5 represents a pessimistic climate scenario (van VUUREN *et al.*, 2011). For each RCP, we considered three Global Climate Model (GCM) – GCMS4, HadGEM2-ES and MIROC-ESM. used in other studies of primates in the Atlantic Forest (GOUVEIA *et al.*, 2016).

The two landscape variables chosen for modeling was forest cover and pasture cover, which are important predictors for the presence of *B. torquatus* (SANTOS *et al.*, 2019b). We used the spatial data available from The Brazilian Foundation for Sustainable Development (*In Portuguese* Fundação Brasileira para o Desenvolvimento Sustentável - FBDS) and the Laboratory of Processing Images and Geoprocessing (*In Portuguese* Laboratório de Processamento de Imagens e Geoprocessamento - LAPIG (Table A.1). The software GrassGis 7.4.4 (Grass, 2018) was used to perform all landscape analyses.

To create the regeneration scenarios, we used a map of natural regeneration potential in pastures areas, ranging from 0 to 100, where higher values represented greater potential for regeneration (NIEBUHR *et al.*, 2017). These maps were created based on the landscape structure (e.g. fragment size) and movement of the frugivorous fauna (NIEBUHR *et al.*, 2017), predicting natural regeneration based on the chance of seed dispersal in non-forest matrices of tropical forests. First, we calculated the minimum area that would need to be regenerated to ensure the presence of *B. torquatus* in the delimited area, based on Santos *et al.*, (2019b), which reported a critical threshold on habitat amount to *B.torquatus* presence: the occupancy probability start to decline in areas with less than 35% of forest cover. Currently, only 25.45% of the area for maned sloth in ESU01 is covered by the Atlantic Forest remnants, and 27.61% in ESU02. To guarantee future populations of the species, it would be necessary to restore 9.55% (11,794.67 km²) and 7.39% (8,455.20 km²), respectively, to reach 35% of forest cover. So, for each ESU, we selected the pixels necessary to reach this goal (Table 01).

Second, we selected the pixels in the regeneration map to create the different scenario – except for Scenario 01 (BAU). For Scenario 02, we selected all pixels with regeneration potential higher than 10 for ESU01 and selected all pixels with regeneration potential higher than 40 for ESU02 (Table 01). For Scenario 03, we selected all pixels with some regeneration potential for both ESUs (Table 1). The third step consisted of mosaicking the raster created with the current forest cover raster (Table A.1), creating forest cover maps for future scenarios. The reverse process was done with the pasture cover, in which we subtract from the current raster pasture areas that will be potentially regenerated, creating the future pasture cover maps. The fourth step was to calculate the percentage of forest and pasture in each pixel of 1km.

Table 1 Accumulated area of pixels in the regeneration map for both ESU (central and left column). The bold lines correspond to the minimum pixel values selected to create the Scenario 02 map.

Regeneration Potential	Accumulated area (Km²) of pixels - ESU01	Accumulated area (Km²) of pixels - ESU02
90	393.55	600.81
80	1227	1976.47
70	2041.61	3364.14
60	3210.95	5372.95
50	4600.42	7795
40	6226.5	**10606.72**
30	8007.82	13743.87
20	10382.57	17815.78
10	**13216.83**	22714.19
0	22800.23	39661.32

To rule out possible correlations between all selected variables, we performed a correlation analysis using a Pearson correlation matrix, for which the results were p> 0.7 indicating that none of the variables were correlated. For the modeling, the chosen environmental variables were all standardized to a spatial resolution of 0.0083 decimal degrees (approximately 1 km).

2.5 Species Distribution Models (SDM)

We use the Maximum Entropy algorithm to generate the distribution models through the Maxent 3.4.1 software (PHILLIPS; DUDI, 2008). Maxent is a modeling technique with high accuracy and better performance than other methods (ROURA-PASCUAL *et al.*, 2009). MaxEnt uses only presence points and background points – also namely as pseudo-absence points, environmental variables and sampling points to calculates an environmental suitability index, indicating where the species is most likely to occur, with values ranging from 0 to 1.

We calibrate the current and future models using bootstrapping methods with 10 random partitions, in which 70% of the data set went for training and 30% went for testing. All models were generated with a convergence threshold of 1.0^{E-5} with 500 interactions and

10,000 background points. We use a Jackknife test to measure the relative importance of each variable in model performance. To evaluate the performance of resulting models, we calculate the Area Under the Curve (AUC) of the Receiving Operator Characteristic (ROC), in which values above 0.75 correspond with great discriminatory capacity between the generated models and random models.

The 38 final models (19 models for each ESU) for current (2 models) and future scenarios (36 models – 18 from each RCP; Fig S1) were converted into binary maps to discriminate between suitable and unsuitable maps, applying a threshold rule defined by the 10th percentile of training presence points. This threshold removes suitability values that are less than the 10% higher suitability values (MUKHERJEE *et al.*, 2020). Binomial probability and omission error were also evaluated. We create consensus models by summing all the GCMs resulting from each RCP, through the extension BioDinamica, from DinamicaEgo software (SOARES-FILHO; RODRIGUES; COSTA, 2009). The results of each GCM are available in the supplementary material (Fig S1).

2.6 Conservation of Suitable areas

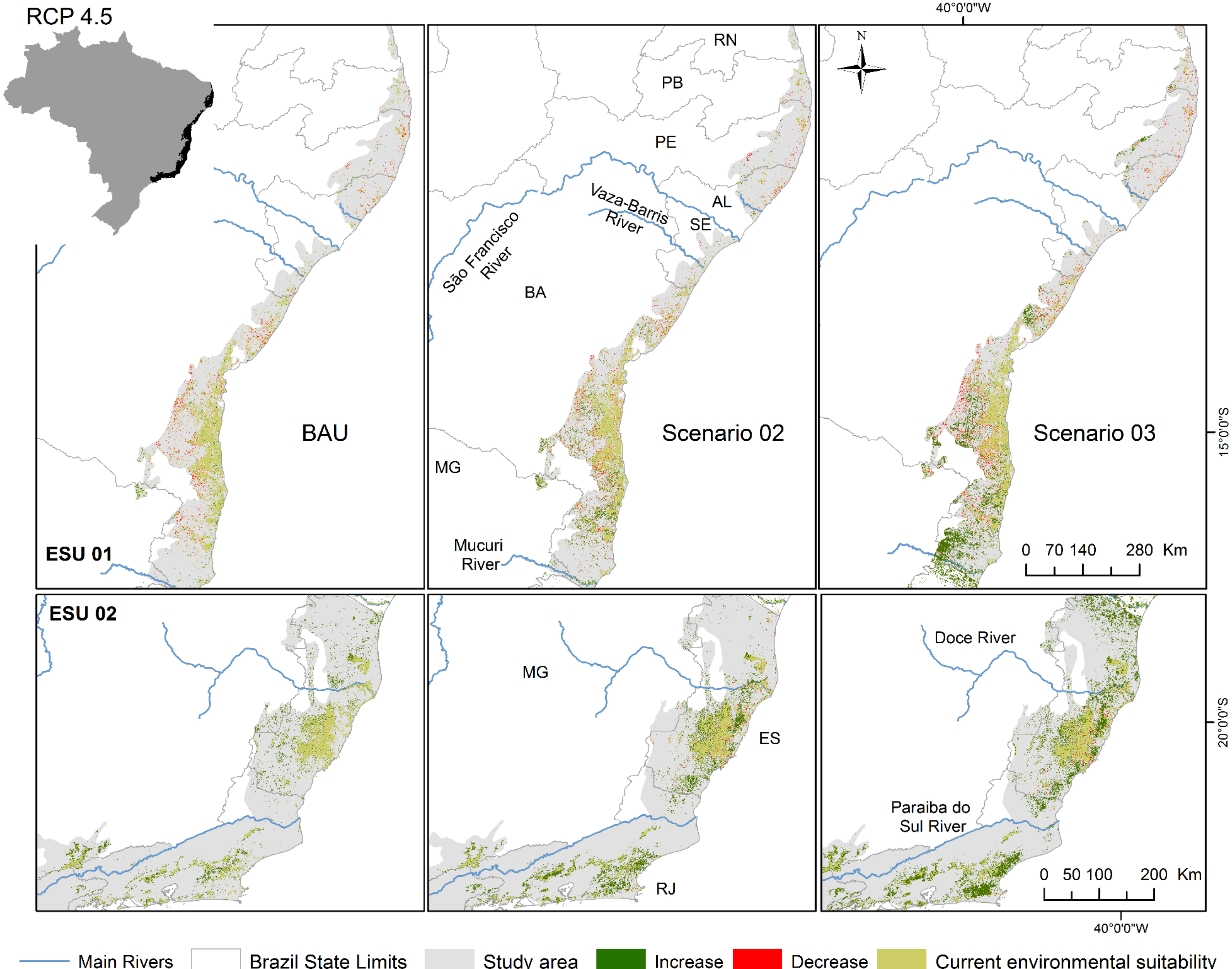
After applying the threshold in all resulting maps, we analyzed the different scenarios according to Brazilian Native Vegetation Protection Law (*In portuguese* LPVN. Law 12,727/2012) and to the National System of Conservation Units *(In portuguese* Sistema Nacional de Unidades de Conservação – SNUC. Law 9,985/2000). The LPVN established the Legal Reserve (RL, from Reserva Legal, *in portuguese*) of 20% of native vegetation in private properties and the Permanent Protection Area (APP from Área de Preservação Permanente, in Portuguese) focused on preservation of river springs, river banks, lakes, lagoons, hilltops and steep slopes. The SNUC establish two types of protected areas – Strictly protected areas and Protected areas of sustainable use. We obtained a GIS database available from different official sources to calculate the percentage of suitable areas in three different conservation categories: riparian APP, Strictly Protected Area and Protected Area of Sustainable Use (Table A.1).

3. Results

All resulting models had good discriminatory power according to AUC values - ESU01: 0.929±0.017 and ESU02: 0.961±0.009 (Table A.2). The current model predicts that

29,886.07 km² - only 4.15% of the study area – are suitable for *B.torquatus*. ESU01 was found to have higher cover of suitable areas (17.71 % or - 21,872.38 km²) then ESU02 (7.0% or- 8,013.69 km²). Two areas of the range of *B. torquatus* stand out for their high suitability: South Bahia and Center-South Espírito Santo (Fig. 2).

In future scenarios, both ESU01 and ESU02 show differences in future land cover changes (Table 02, Fig. 02). The ESU01 would experience a loss of suitable areas, even in the regeneration scenarios, mainly further inland and more northern areas (Table 02; Fig. 02). In the BAU scenario, the suitable areas would decrease more abruptly, reaching 4033.02 km² (4.0%) in losses by 2050 (Table 02). The loss would be subtler in Scenario 02, and in Scenario 03, a stability in the suitable areas would take place only in RCP 4.5 (Table 2). On the other hand, ESU02 would experience an increase in the amount of suitable areas, including in the scenario without regeneration (Table 2; Fig 02).


RCP 4.5
40°0'0"W
BAU
Scenario 02
Scenario 03
ESU 01
ESU 02
RN
PB
PE
AL
SE
BA
MG
ES
RJ
São Francisco River
Vaza-Barris River
Mucuri River
Doce River
Paraiba do Sul River
N
0 70 140 280 Km
0 50 100 200 Km
15°0'0"S
20°0'0"S
40°0'0"W
Main Rivers
Brazil State Limits
Study area
Increase
Decrease
Current environmental suitability

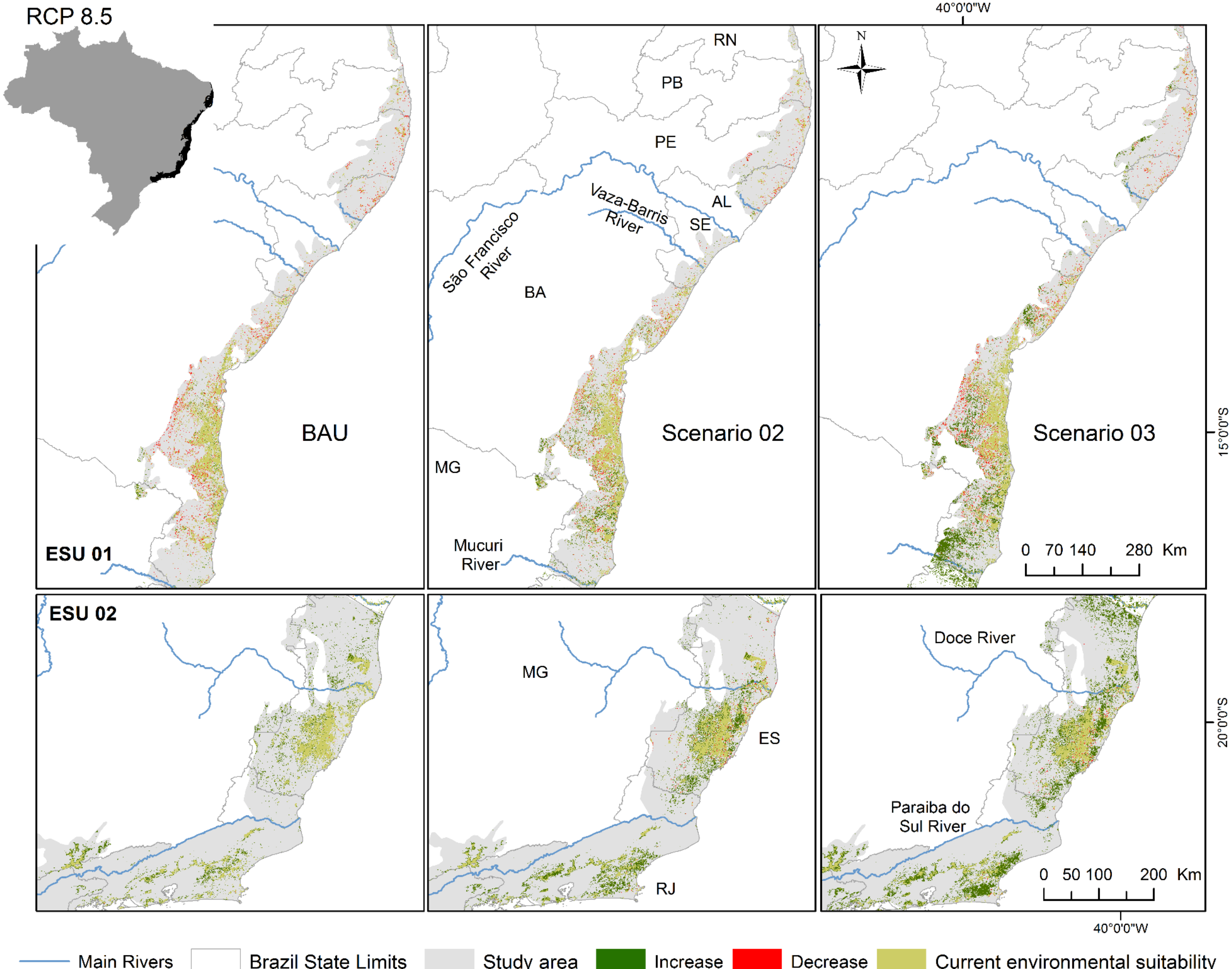

RCP 8.5
40°0'0"W
RN
PB
PE
AL
SE
Vaza-Barris River
São Francisco River
BA
MG
Mucuri River
N
BAU
Scenario 02
Scenario 03
15°0'0"S
0 70 140 280 Km
ESU 01
ESU 02
Doce River
MG
ES
RJ
Paraiba do Sul River
20°0'0"S
0 50 100 200 Km
40°0'0"W
Main Rivers
Brazil State Limits
Study area
Increase
Decrease
Current environmental suitability

Fig. 02 Increase and decrease of suitable areas in relation to current environmental suitability for both populations of *Bradypus torquatus*, considering regeneration scenarios and climate changes. BAU – Business As Usual; RN – Rio Grande do Norte; PB – Paraíba; PE – Pernambuco; AL – Alagoas; SE – Sergipe’; BA – Bahia; MG – Minas Gerais; ES – Espírito Santo; RJ – Rio de Janeiro.

Table 02 Suitable areas in km² for the two populations in the different scenarios and the deacrese (-) or increase (+) of suitable areas (%). S01 – Scenario 01 (BAU), S02 - Scenario 02, S03 - Scenario 03. 45 and 85 – RCP 4.5 and 8.5, respectively.

Scenario	ESU01 (Km²)	ESU02 (Km²)	Total área (Km²)
Current	21872.38	8013.69	29886.07
S01-45	16939.36 (- 4.0 %)	9450.87 (+ 1.25 %)	26390.23
S01-85	16130.52 (- 4.65 %)	10366.02 (+ 2.06 %)	26496.53
S02-45	18595.45 (- 2.65 %)	12525.23 (+ 3.94 %)	31120.68
S02-85	19382.81(- 2.06 %)	13002.29 (+ 4.36 %)	32385.10
S03-45	22030.83 (+ 0.13 %)	14055.82 (+ 5.28 %)	36086.65
S03-85	21115.40 (- 0.61 %)	14270.71 (+ 5.47 %)	35386.11

According to the Jackknife test, the environmental parameter with the highest influence on environmental suitability was forest cover for ESU01 (Fig.3), which positively influenced the suitability (Fig.04). However, ESU02 showed a different pattern: the forest cover was only important in scenarios without regeneration (Fig.3). In both regeneration scenarios, Bio 04 (Temperature Seasonality) and pasture cover had more influence on environmental suitability.

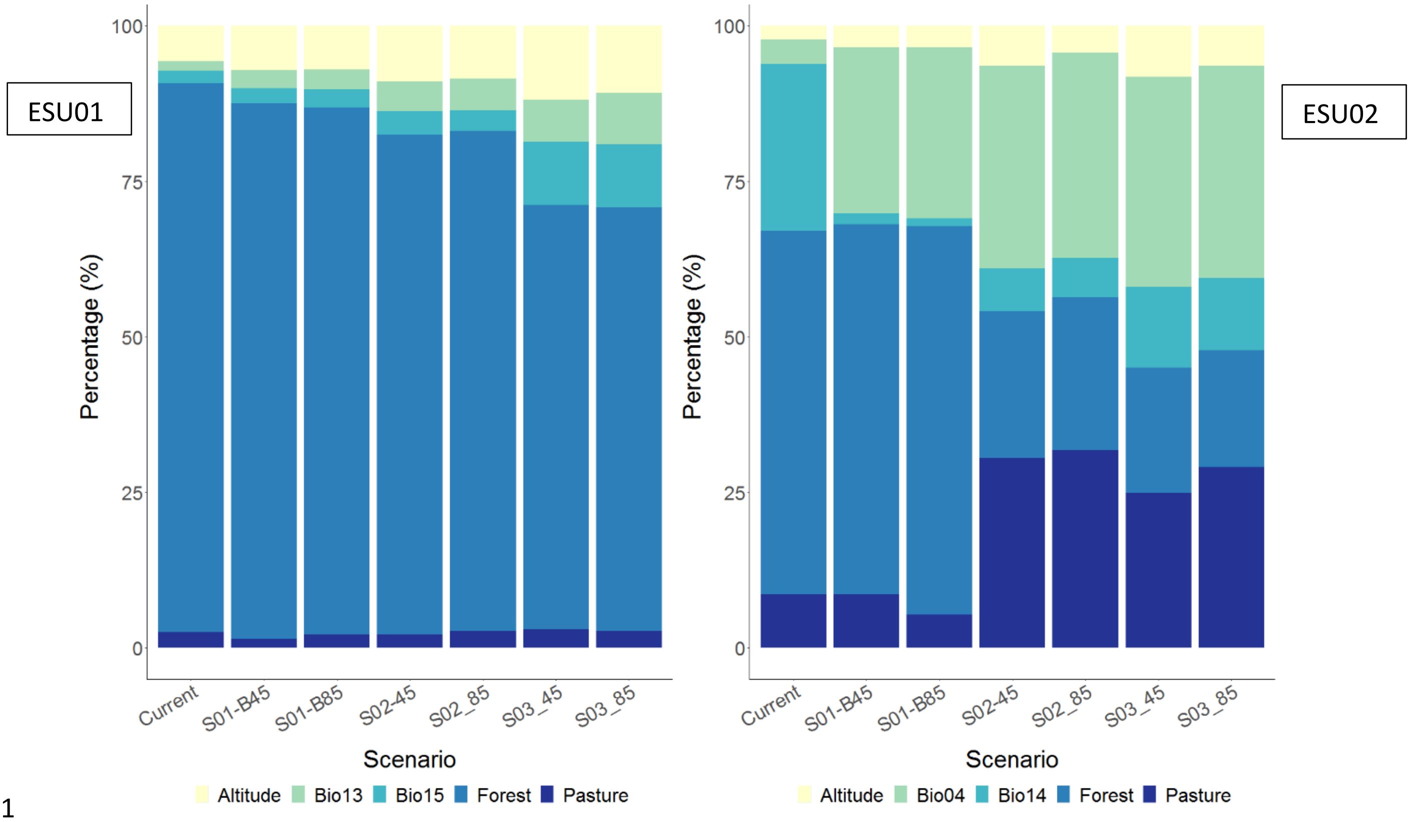


Fig. 03 Results in percentage for the Jackknife test for variable of environmental variables in the development of Maxent model.

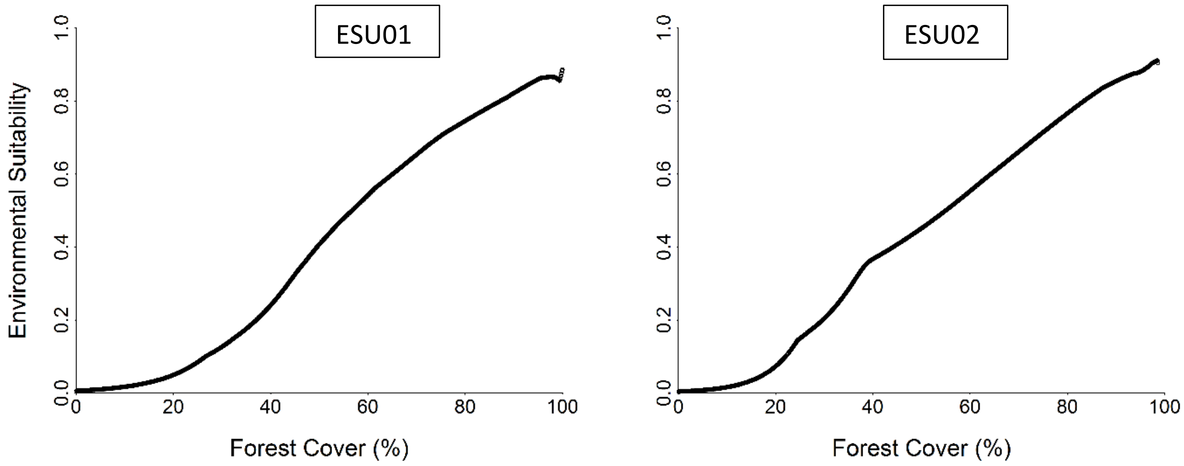


Fig. 04 Response curves of the most important variable and the environmental suitability for the current scenario.

Currently, most part of these suitable areas in both ESUs are excluded from any legal protection (ESU01 – 65.03% and ESU02 – 56%; Fig.5). Among the protected regions, areas of Sustainable use protection encompass more suitable areas in ESU01 (21.4%), followed by APP (7.27%) and Strictly protected areas (6.30%). For ESU02, suitable areas are present almost in same proportion in the three conservation categories (15.32%, 14.44%, and 14.25, respectively - Fig.5). The same trend for both ESUs remains throughout the future scenarios, with the majority of the suitable areas remaining unprotected (Fig.5).

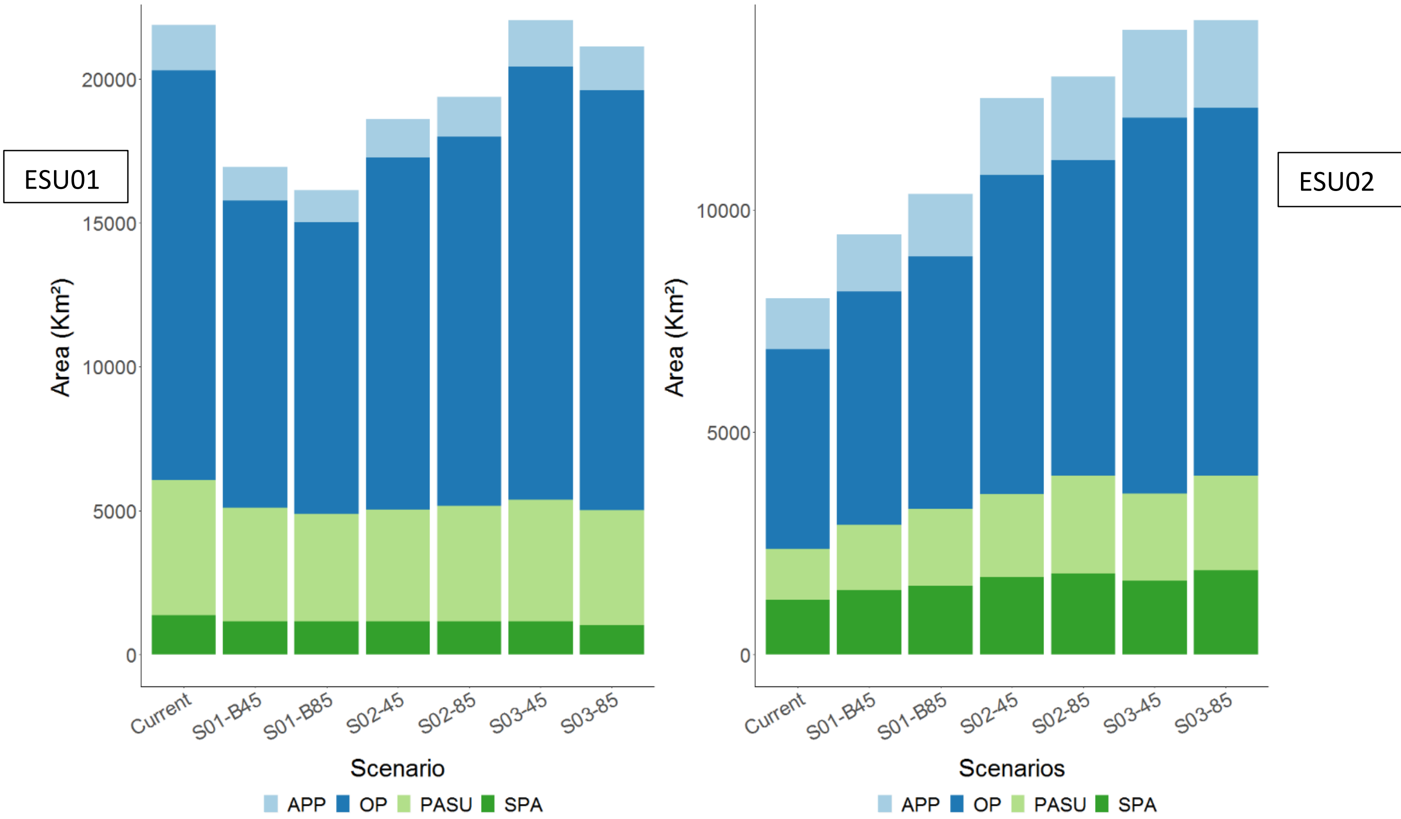


Fig. 5 Distribution of Suitable Areas Outside Protected Areas (OP) and in different conservation categories, according to conservation Categories, where: APP - Permanent Protection Areas; PASU – Protected Areas of Sustainable Use; SPA – Strictly Protected Areas.

4. Discussion

4.1 Overview of environmental Suitability models

The strong influence of climate change on the species distribution and the size of suitable areas is undeniable (GOUVEIA *et al.*, 2016b; STRUEBIG *et al.*, 2015). However, as we demonstrate here, and in accordance with other works (PREVEDELLO *et al.*, 2019; SILVÉRIO *et al.*, 2015), forest regeneration can mitigate the negative influence of climate change. We also highlight the importance of considering the genetic structure of species in ecological analyses: Genetically distinct populations of the same species present some particularities and distinct relations within the inhabited environment, thus, producing different results.

South Bahia and Center-South of Espírito Santo represents two important strongholds for *B.torquatus* (Fig 02). Both locations present well-preserved forest cover and include important protected areas, such as UNA Wildlife Refuge (BA), four National Parks - Descobrimento, Pau Brasil, Monte Pascoal and Serra das Lontras (BA) and two Biological Reserves- Una (BA) and Augusto Ruscchi (ES). Our models correctly predict the current low environmental suitability already indicated by gaps in the distribution of *B. torquatus*, in North Espírito Santo and North Rio de Janeiro (Fig.2). These areas present a low forest cover, which reduces the environmental suitability for the species. Yet, two large blocks of forest in the distribution gap between Bahia and Espírito Santo have been classified as suitable, which leads us to believe the existence of other undetected influences. Other studies highlight past epirogenic and climatic events which resulted in forest shrinkage, fragmentation and vegetation change (SCHETINO; COIMBRA; SANTOS, 2017; VASCONCELOS *et al.*, 1992). This gap region is distinct from the rest of the ombrophilous Atlantic Forest, with high seasonality and the presence of a semi-deciduous forest (JESUS; ROLIM, 2005). Similarly, previous studies carried out in North of Rio de Janeiro state highlights the low tree diversity of the semi-deciduous forests existing there (OLIVEIRA-FILHO; FONTES, 2000), linked to a strongly seasonal climate (HIRSCH; CHIARELLO, 2012; OLIVEIRA-FILHO *et al.*, 2005), which likely acts as a limiting factor for the presence of strictly folivorous species.

The northernmost part of *B. torquatus*’ range, the regions of Pernambuco (PE), Paraiba (PB) and Rio Grande do Norte (RN - Fig. 02), present low environmental suitability. Previous research indicates similar result (MOREIRA *et al.*, 2014), evidencing the current absence of the species in the region. In contrast, Sergipe (SE), which represent the known northern limit of *B. torquatus'* range, still presents important though isolated areas suitable for the species, especially near the border of this state with Bahia (Fig. 2, Fig. S1). These populations deserve special attention; since the confirmation of this species in the region in 2009 (CHAGAS *et al.*, 2009), only one unpublished study has been conducted on this population (SANTOS *et al.*, 2019). Thus, several ecological aspects remain unknow. Furthermore, the genetics of this population have not yet been studied (SCHETINO; COIMBRA; SANTOS, 2017), and it has not been included in any national conservation plan.

4.2 Present and future scenarios

The ESU01 concentrates large and continuous suitable areas for *B. torquatus,* with a stability trend only in the most positive scenarios (Table 02). Additionally, most current and future suitable areas in ESU01 represent one of the most climatically stable regions of the Atlantic Forest (CARNAVAL *et al.*, 2009; SOBRAL-SOUZA *et al.*, 2018). Overall, the Atlantic forest presents an increase in precipitation seasonality from east to west, and also from south to north (OLIVEIRA-FILHO; FONTES, 2000), and this trend will become stronger with climate changes (IPCC, 2018). The increase of precipitation seasonality and the decrease in precipitation of the wettest month, combined with a low forest cover, might explain why the environmental suitability declined mainly further inland and in northern areas - more vulnerable to climate changes (CARNAVAL *et al*., 2009). Fluctuations in the precipitation regimes, may affect the production of young leaves, a preferred food item that becomes more abundant during the wet season (CHIARELLO, 1998b).

The BAU scenario – which represents the climate changes effects without the forest regeneration - presents a sharp decrease in suitable areas. This trend decreased in the less optimist scenario (Scenario 02 - Fig 02; Table 02), ratifying the importance of forest regeneration in mitigating the effects of climate change. Even though, the decrease in suitable areas in practically all scenarios (Scenario 02 – Fig 02, Table 02) lights a warning for the species conservation, since the region was one of the most deforested between 2017 and 2018 (e.g. Bahia and Sergipe States), counteracting the national trend of decreasing and stabilizing deforestation in the Atlantic Forest (FUNDAÇÃO SOS MATA ATLÂNTICA, 2018).

In the ESU02 region, the environmental suitability depends on other factors, besides the percentage of forest cover, such as the presence of pasture and climate conditions. A slight improvement in climatic conditions and a large increase in forest cover may result in a continuous increase on suitable areas, even in the BAU scenario (Fig 2, Table 02). The southeast Atlantic Forest is considered climatically unstable (CARNAVAL *et al.*, 2009), presenting a strong seasonality in precipitation and in temperature throughout the year (OLIVERA - FILHO; FONTES, 2000). With global warming, the temperature seasonality and the precipitation of the driest month tends to decrease (Table S3). More constant temperatures might benefits the species, since their body temperature varies with the environment and activity decreases during cold periods (CHIARELLO, 1998a; GINÉ *et al.*, 2015). Differently, the decrease of precipitation of the driest month can be harmful to the species, since it may exacerbate water deficit during the driest months, affecting leaf production in the dry season (LIEBERMAN; LIEBERMAN, 1984; REICH, 1995). The maned sloths from ESU02 are able to find and consume young leaves throughout the year (CHIARELLO, 1998b), despite the seasonal variation that is typical of this region today, but this flexibility might be compromised if precipitation seasonally increases. As we highlighted above, the two distribution gaps of this species have stronger seasonality in precipitation than non-gap areas.

Regarding the forest remnants, the increase in suitable areas is concentrated in specific locations, around the largest forest areas (Fig. 02). The region englobes a few large forest patches, as well as regions with medium-sized fragments, that are functionally connected (SANTOS; JÚNIOR; EUGENIO, 2012; SANTOS *et al.*, 2016), and also a high incidence of pastures. This combination creates ideal conditions for increasing forest cover, since larger fragments encompass a greater diversity of species – including important animals seed dispersal (WUNDERLE, 1997) - and therefore, their surrounding matrices are more exposed to seed rain (CHAZDON, 2012; CROUZEILLES *et al.*, 2016; TAMBOSI *et al.*, 2014). Additionally, the region experiment lower rates of habitat loss, presenting the lowest deforestation rate in the last 30 years (FUNDAÇÃO SOS MATA ATLÂNTICA, 2018).

4.3 Conservation Status of Suitable areas

In both ESUs, the percentage of suitable areas legally protected in the three categories we analyzed is consistently low (Fig. 05). Part of this unprotected area is somewhat protected by the Atlantic Forest Law (Law 11. 428/2006), which prohibits the suppression of primary and late secondary forest areas, except in rare cases, in which environmental compensation is given. However, it allows the suppression of early secondary areas and of the initial stage, threatening, for example, the incidence of the natural regeneration of an area (VARJABEDIAN, 2010). Thus, it`s quite necessary to guarantee the conservation of these areas, in order to ensure a substantial increase in forest cover, and consequently the environmental suitability for several forest species endemic to the Atlantic Forest.

The other areas lying outside of the aforementioned conservation categories may belong to areas of Legal Reserves. According to LPVN, Legal Reserves is an instrument for preserving areas of natural vegetation in private areas, and in the Atlantic Forest this area occupies 20% of private property. The Legal Reserves system, however, has been constantly threatened with the argument that it is necessary to increase agricultural production areas (METZGER *et al.*, 2019), and its presence in the future is unknown. Legal Reserves have been neglected since the approval of the LPVN in 2012, which ruled that APP areas also must enter into the calculation of the RL areas. This was, in addition to forgiving the debt of small landowners who deforested their RL areas before 2008, a decision that cost the regeneration of thousands of hectares (BRANCALION *et al.*, 2016; SOARES-FILHO *et al.*, 2014). Legal Reserves have fundamental ecological roles, including improving regional and global climate regulation (METZGER *et al.*, 2019). They compose important parts of the landscape, and along with APPs may assist in the regional increase of forest areas, facilitating the flow of organisms between protected areas and maintaining the genetic viability of populations (METZGER *et al.*, 2019; PADOVEZI *et al.*, 2018).

5. Conclusions

Our models evaluate the environmental suitability for *B. torquatus*, which does not necessarily indicate that the species is currently present in the area. This approach is highly useful for conservation and management purposes (MUKHERJEE *et al.*, 2020; PORTUGAL *et al.*, 2019; PRÉAU *et al.*, 2020), and in particular, helps to identify and prioritize areas for reintroduction (DANKS; KLEIN, 2002; MARTÍNEZ-MEYER *et al.*, 2006; PANT *et al.*, 2020; PETERSON, 2006) , and to identify areas to create potential wildlife corridors to link suitable yet noncontiguous areas (LIU *et al.*, 2018; MORATO *et al.*, 2014).

Natural regeneration is a reality in Atlantic Forest, and as we show here, it can guarantee a decrease in local effects of climate change effects on environmental suitability for a forest dependent species. The majority of pasture lands are economically unproductive

(STRASSBURG *et al.*, 2014), and therefore present an opportunity for forest recovery (LATAWIEC *et al.*, 2015), especially if there is an associated monetary return (BANKS-LEITE *et al.*, 2019). Additionally, encouraging the natural regeneration of these areas may contribute to actions instituted by the National Policy for the Recovery of Native vegetation (*In portuguese* Planaveg, Decree nº 8.972/2017), whose objective is to recover up to 12 million hectares of native vegetation by 2030. Our models will contribute to several conservation actions for *B. torquatus* and are especially useful for our different analyses for the two ESUs. Even though its conservation status is threatened, mainly due to habitat loss, there remain adequate areas to sustain viable populations.

In this sense, it is extremely important to ensure the maintenance of areas that are already protected, mainly those in South Bahia, Espírito Santo and Rio de Janeiro, by decreasing external pressures. Promoting compliance with environmental law in the entire analyzed area will also be necessary to preserve APPs and Legal Reserves. Lastly, the natural regeneration of abandoned areas should be encouraged, especially in the more northerly areas (Sergipe and North Bahia) that currently contain suitable but isolated areas. In this case, it is also necessary to collaborate for their protection, to minimize the suppression of regrowth, and to monitor the forests' successional stages, interfering when necessary to increase its quality.

Acknowledgements

PMS research grants from The Rufford Foundation (21498-1) and scholarship from CNPq (141571/2017-9) and CAPES (88881.188493/2018-01). MCR thanks FAPESP (process #2013/50421-2), CNPq (processes # 312045/2013-1; #312292/2016-3) and PROCAD/CAPES (project # 88881.068425/2014-01) for their financial support. AGC has a Research Productivity Grant from CNPq (303101/2017-2). APP research support from FAPEMIG and CAPES.

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Appendix A. Supplementary data

Table S1 Characterícts of variables used to delineate environmental suitability for *Bradypus torquatus*.

Use	Variable	Description	Original Resolution	Year	Source	ESU 01	ESU 02
Environmental Suitability Models	Forest Cover Percentage	Forest Cover percentage in 1x1km pixel	5 x 5 m	2018	Brazilian Foundation for Sustainable Development (In Portuguese Fundação Brasileira para o Desenvolvimento Sustentável - FBDS) Source: https://www.fbds.org.br	X	X
	Pasture cover Percentage	Pasture Cover percentage in 1x1km pixel	30x30m	2015	Image Processing and Geoprocessing Lab (In portuguese: Laboratório Processamento de Imagens e Geoprocessamento - LAPIG). Source: www.pastagem.org	X	X
	Elevation	Global elevation data	30 arc-second (1x1km)	2004	NASA Shuttle Radar Topography Mission Source: https://www2.jpl.nasa.gov/srtm/	X	X

	Bioclimatic Variables	Bio 04 = Temperature Seasonality (standard deviation *100)	30 arc-second (1x1km)	2005	Worldclim global climate variables V1. Source: https://www.worldclim.org/		X
		Bio 13 = Precipitation of Wettest Month				X	
		Bio 14 = Precipitation of Driest Month					X
		Bio 15 = Precipitation Seasonality (Coefficient of Variation)				X	
Conservation Analyses	hydric APPs	Areas of water APPs, which include rivers, water springs, lakes, lagoons and artificial reservoirs	5 x 5 m	2018	Brazilian Foundation for Sustainable Development (In Portuguese Fundação Brasileira para o Desenvolvimento Sustentável - FBDS) Source: https://www.fbds.org.br	X	X
	Strictly Protected areas (SPA) and	Shapefile with the distribution of protected areas of integral			Ministry of the Environment (In Portuguese: Minstério do Meio Ambiente - MMA). Source:	X	X

	Protected Areas of Sustainable use (PASU)	conservation and sustainable use		http://mapas.mma.gov.br/i3geo/datadownload.htm

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Table S2 AUC scores, test omission, and variable contribution for current scenario. All binomial probability - $p < 0.05$.

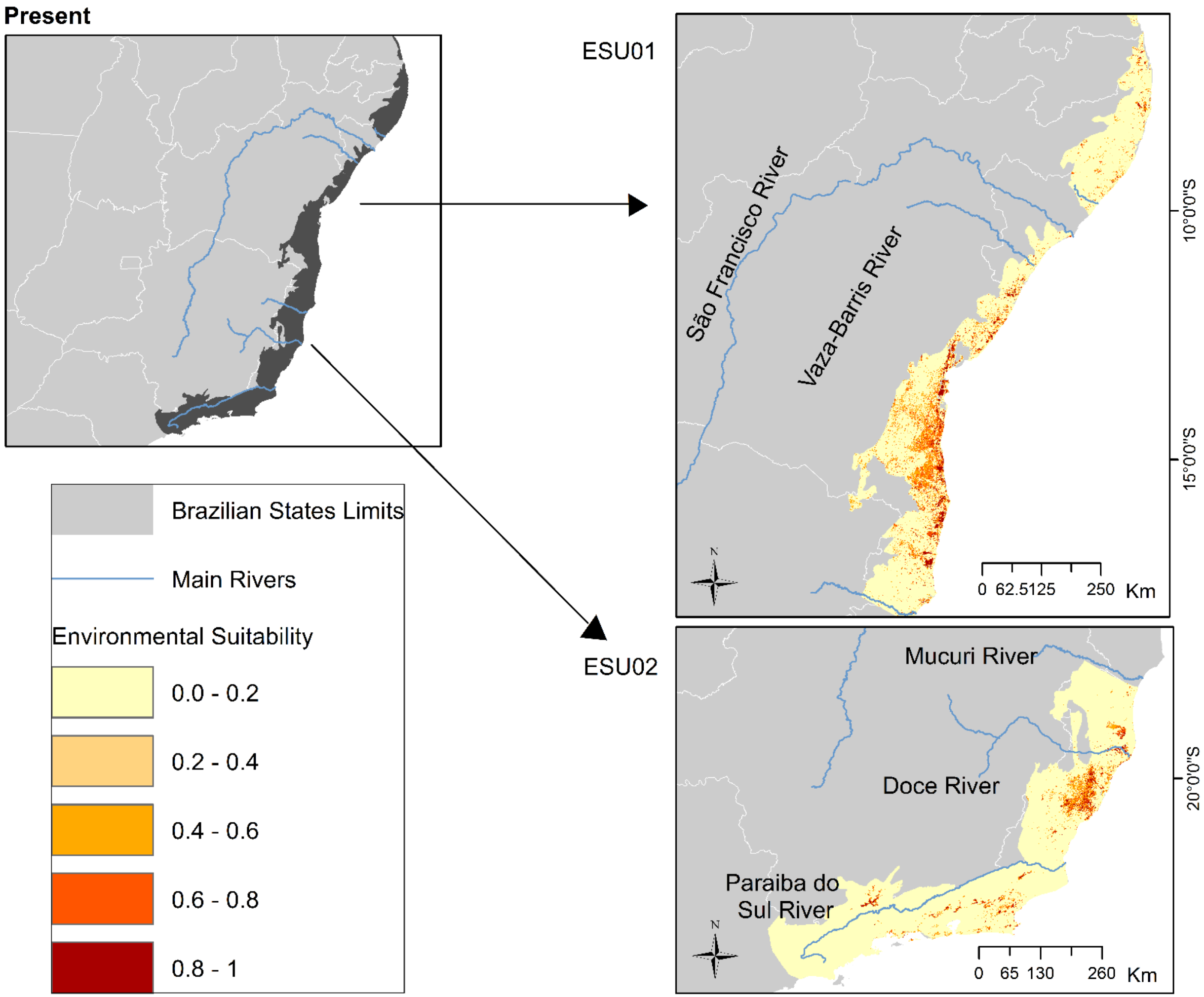
Scenarios	Model Evaluation				Variable Contibution				
	Test AUC	**AUC SD**	**10 percentil**	**Omission test**	**Altitude**	**Bio13**	**Bio15**	**Forest cover**	**Pasture cover**
ESU01	0.929	0.017	0.367	0.094	5.701	1.563	2.032	88.177	2.527
ESU02	0.961	0.009	0.407	0.171	2.168	26.888	3.959	58.351	8.634

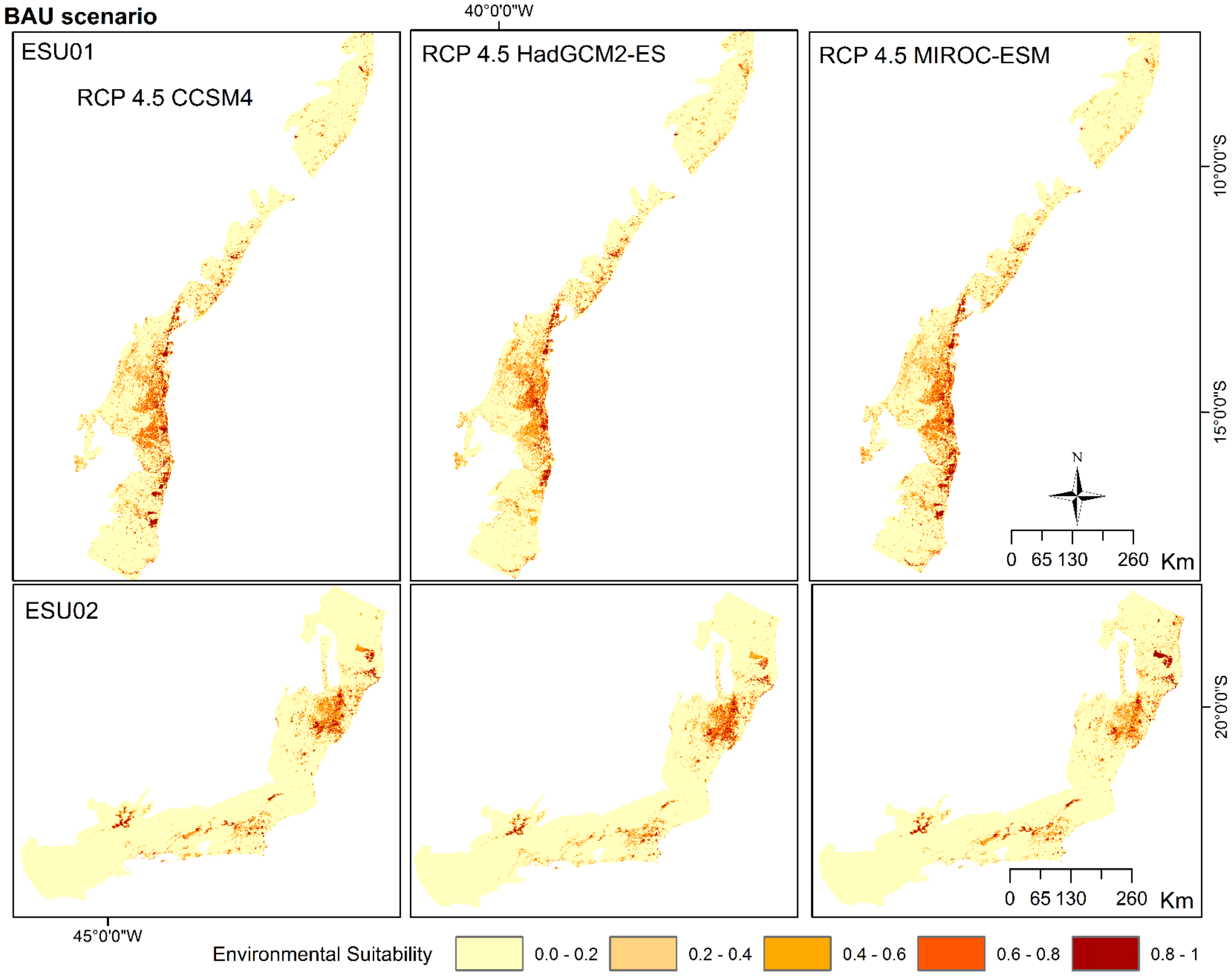
Table S3 Maximum, minimum and mean values of the variables for both populations and in all scenarios.

ESU01						
	Bio13 - Precipitation of the Wettest month (mm)			**Bio15 - Precipitation Seasonality**		
Scenario	**Max**	**Min**	**Mean**	**Max**	**Min**	**Mean**
Current Scenario	336	4	186.412	76	9	40.128
RCP45 CCSM4	275	84	162.629	78	14	42.200
RCP45 HadGCM2-ES	280	95	166.433	78	18	45.138
RCP45 MIROC-ESM	265	74	157.197	80	14	43.088
RCP85 CCSM4	249	76	156.710	78	11	42.600
RCP85 HadGCM2-ES	306	105	184.003	79	23	50.040
RCP85 MIROC-ESM	234	75	156.672	85	18	44.721
ESU02						
	Bio04 - Temperature Seasonality			**Bio14 - Precipitation of the driest month (mm)**		
Scenario	**Max**	**Min**	**Mean**	**Max**	**Min**	**Mean**
Current Scenario	2525	1444	1960.396	176	13	32.613
RCP45 CCSM4	2423	1427	1847.344	178	12	30.334
RCP45 HadGCM2-ES	2534	1515	2009.255	195	13	31.569
RCP45 MIROC-ESM	2354	1312	1786.010	159	10	27.033
RCP85 CCSM4	2380	1393	1908.077	159	12	28.866

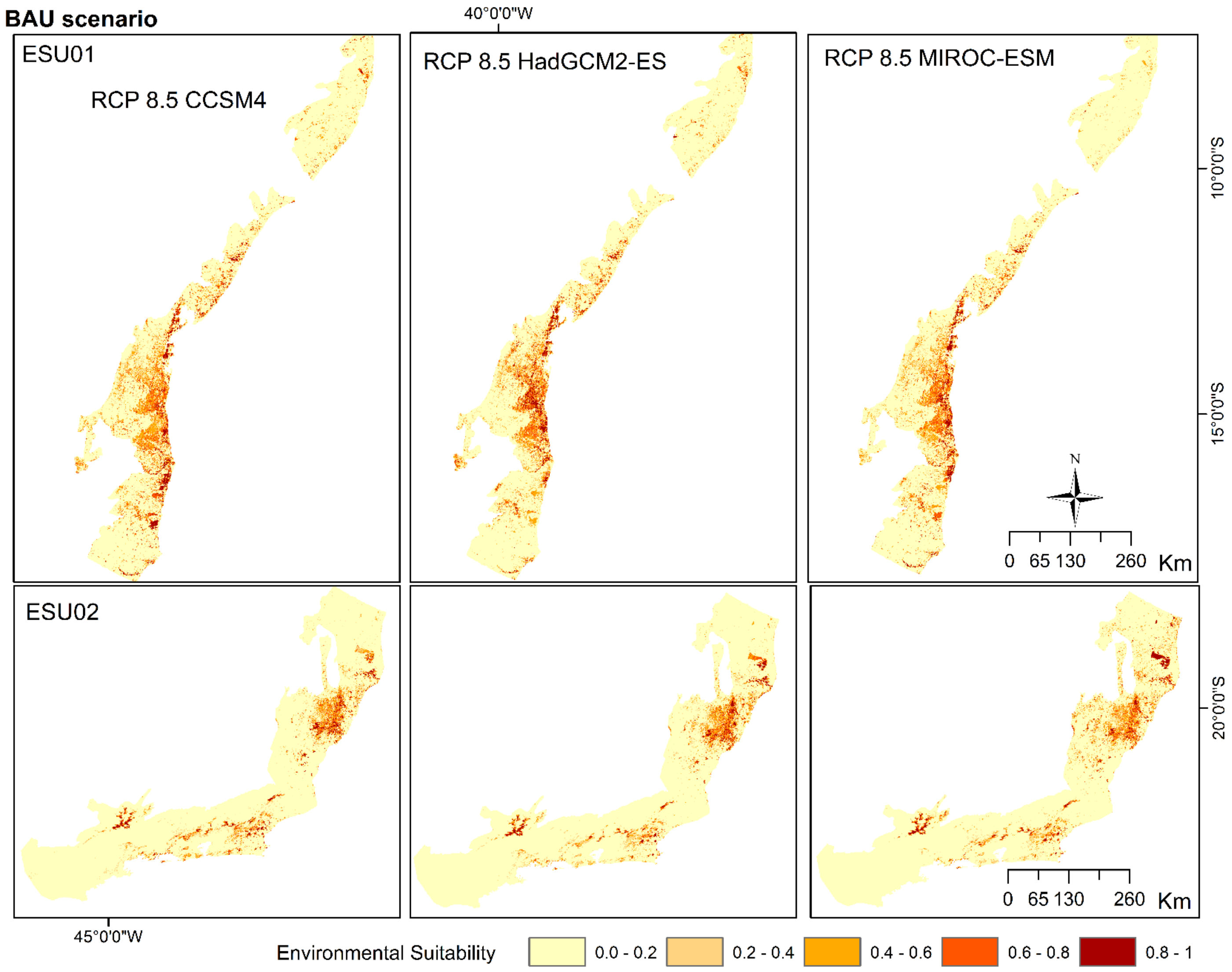
RCP85 HadGCM2-ES	2602	1575	2088.178	197	12	29.145
RCP85 MIROC-ESM	2324	1343	1822.031	120	7	22.561

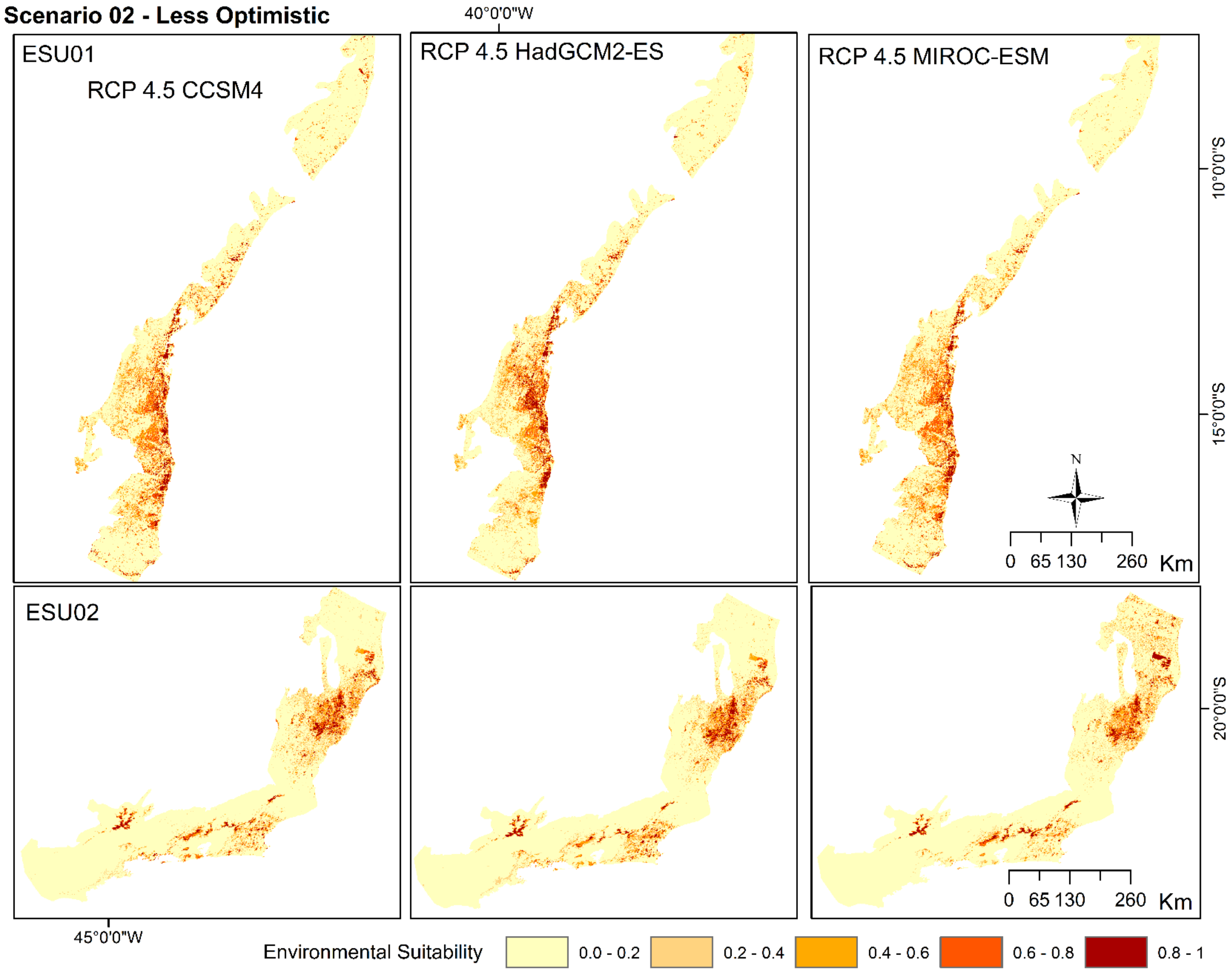
Fig S1

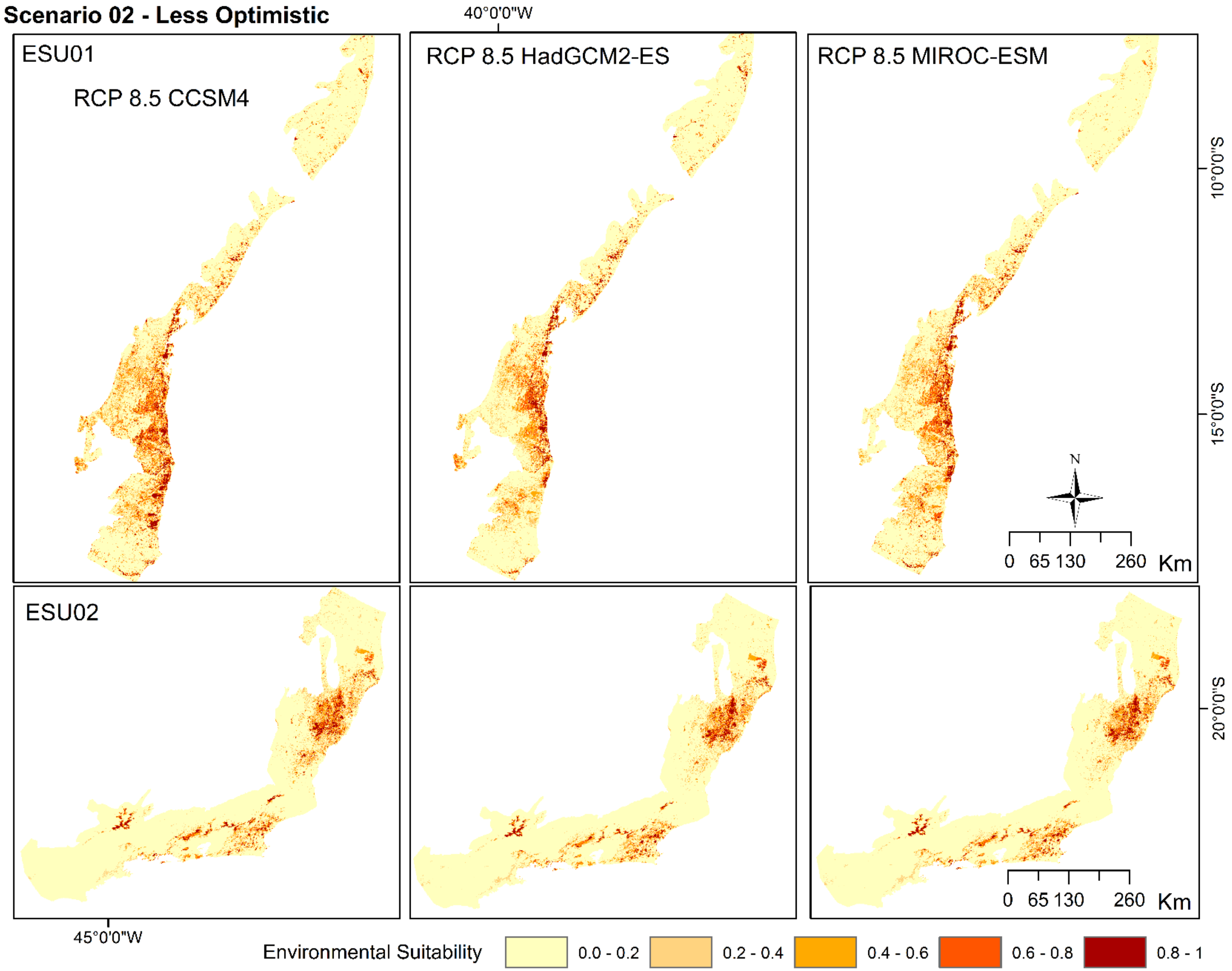

Present
ESU01
ESU02
São Francisco River
Vaza-Barris River
Mucuri River
Doce River
Paraiba do Sul River
10°0'0"S
15°0'0"S
20°0'0"S
N
0 62.5125 250 Km
0 65 130 260 Km
Brazilian States Limits
Main Rivers
Environmental Suitability
0.0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1

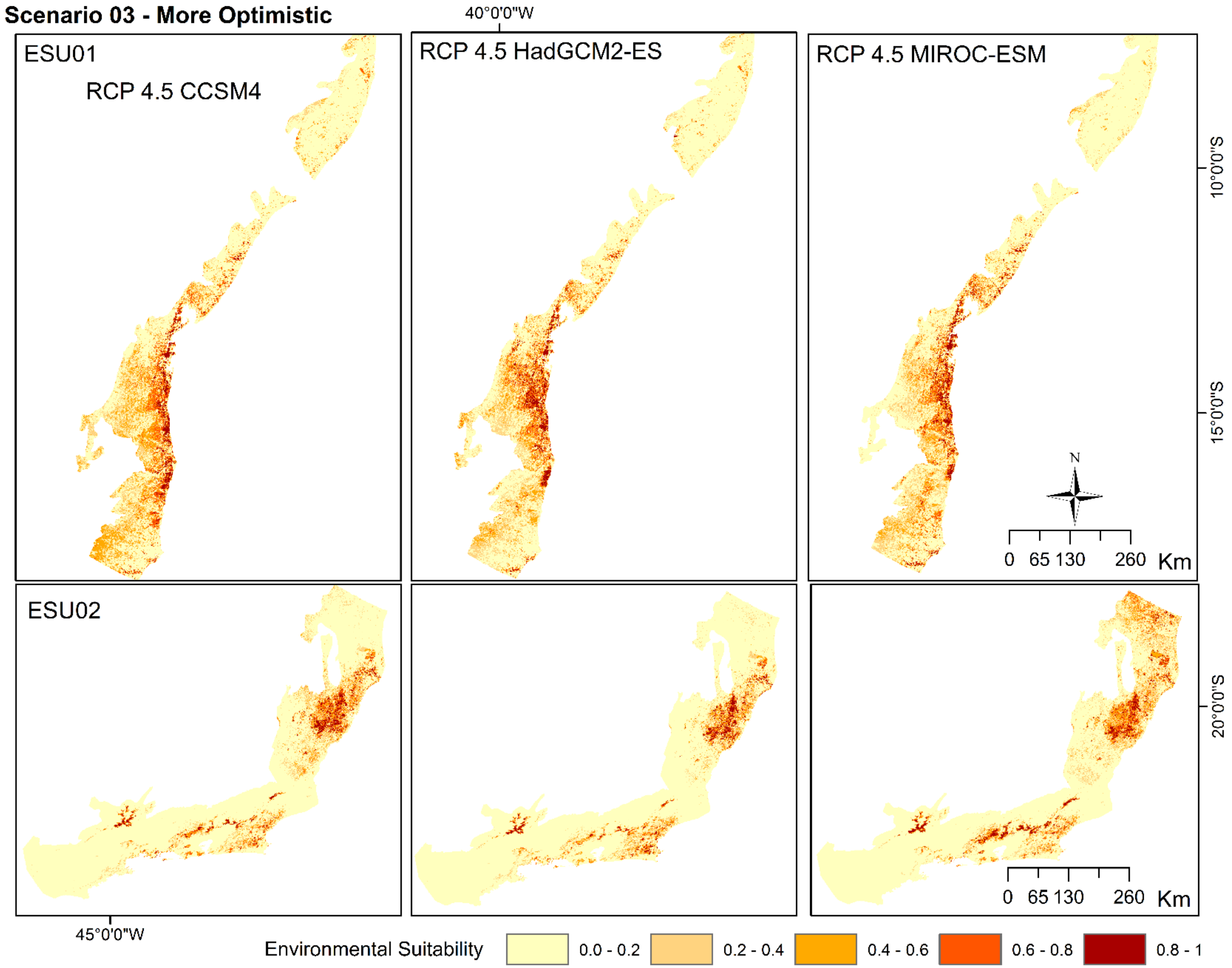

BAU scenario
ESU01
RCP 4.5 CCSM4
40°0'0"W
RCP 4.5 HadGCM2-ES
RCP 4.5 MIROC-ESM
10°0'0"S
15°0'0"S
N
0 65 130 260 Km
ESU02
20°0'0"S
0 65 130 260 Km
45°0'0"W
Environmental Suitability
0.0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1

3756


BAU scenario
40°0'0"W
ESU01
RCP 8.5 CCSM4
RCP 8.5 HadGCM2-ES
RCP 8.5 MIROC-ESM
10°0'0"S
15°0'0"S
N
0 65 130 260 Km
ESU02
20°0'0"S
0 65 130 260 Km
45°0'0"W
Environmental Suitability
0.0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1


Scenario 02 - Less Optimistic
40°0'0"W
ESU01
RCP 4.5 CCSM4
RCP 4.5 HadGCM2-ES
RCP 4.5 MIROC-ESM
10°0'0"S
15°0'0"S
N
0 65 130 260 Km
ESU02
20°0'0"S
0 65 130 260 Km
45°0'0"W
Environmental Suitability
0.0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1


Scenario 02 - Less Optimistic
40°0'0"W
ESU01
RCP 8.5 CCSM4
RCP 8.5 HadGCM2-ES
RCP 8.5 MIROC-ESM
10°0'0"S
15°0'0"S
N
0 65 130 260 Km
ESU02
20°0'0"S
0 65 130 260 Km
45°0'0"W
Environmental Suitability
0.0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1


Scenario 03 - More Optimistic
40°0'0"W
ESU01
RCP 4.5 CCSM4
RCP 4.5 HadGCM2-ES
RCP 4.5 MIROC-ESM
10°0'0"S
15°0'0"S
N
0 65 130 260 Km
ESU02
20°0'0"S
0 65 130 260 Km
45°0'0"W
Environmental Suitability
0.0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1

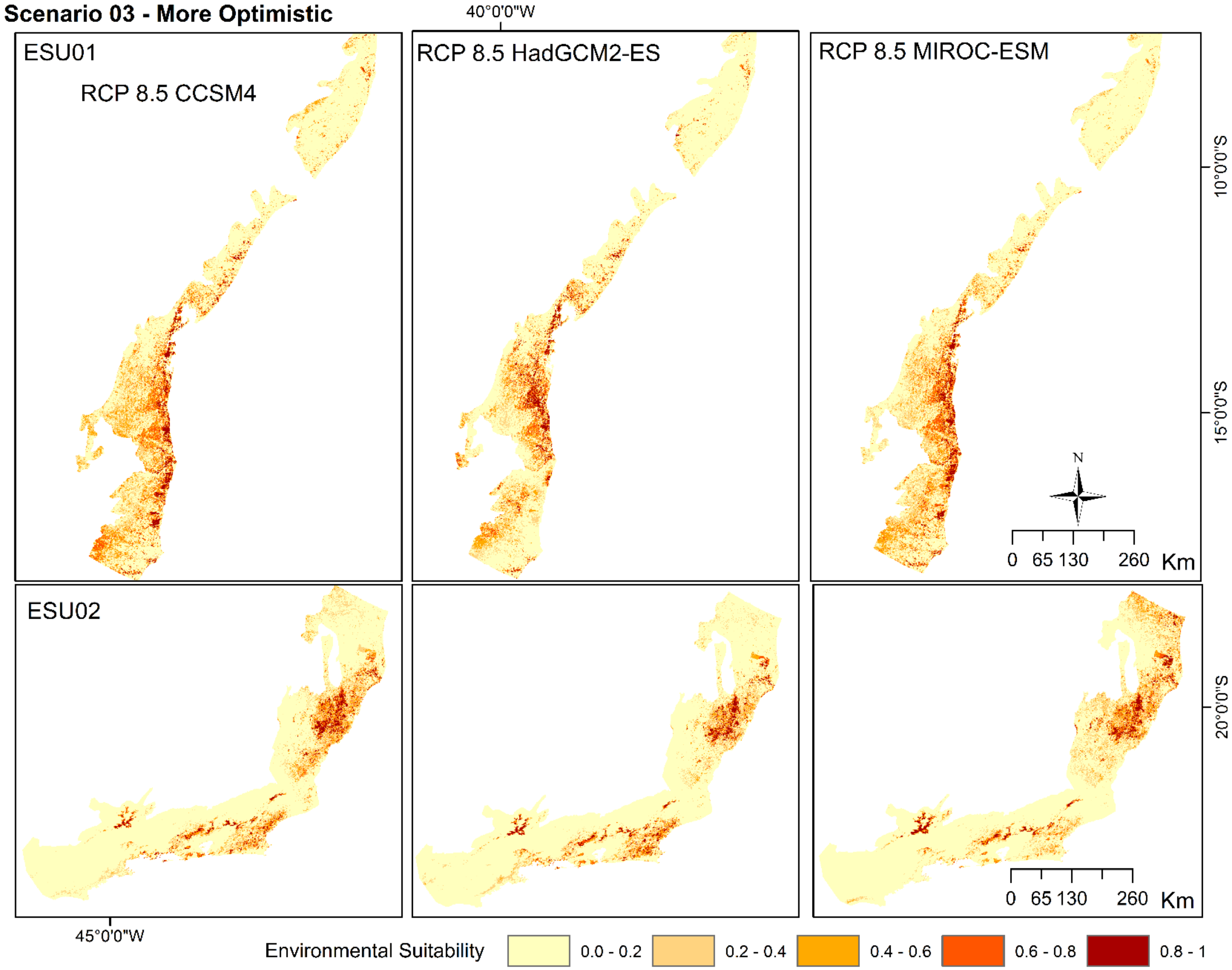


Fig S1 Current and future environmental suitability for both populations of *B.torquatus*. The environmental suitability is separated in different classes, and all results from the different circulations models are represented here.

Conclusões Gerais

Nesta tese, buscamos identificar a influências de variáveis ambientais na ocupação de remanescentes florestais pela preguiça de coleira e na disposição das áreas mais adequadas para a espécies por meio de análises multi-espaciais e temporais. De modo geral, a ocupação da espécie em uma dada região irá depender de fatores múltiplos, como o ambiente local e a composição espacial dos elementos da paisagem. O futuro da espécie irá depender principalmente da dinâmica florestal e da climática. Além disso, a tese resultou em uma grande contribuição com a geração e o uso de dados abertos.

Embora a preguiça ocupe ambientes alterados – porém com boa cobertura floresta (> 35%), a sua ausência em áreas com menos de 20% de cobertura florestal acende um alerta em relação a sua conservação. A presença de áreas abertas – mesmo em menor proporção – inviabiliza a presença da espécie em determinadas áreas. O alto requerimento energético para atravessar tais locais e as ameaças intrínsecas nesses ambientes, como a predação oportunística, são as principais barreiras. Por outro lado, como já salientado em estudo anterior (SANTOS *et al.*, 2016), as preguiças preferem determinados tipos de ambientes florestais, como locais que possua um dossel alto e conectado e que abrange árvores importantes para a sua dieta. As preguiças passam boa parte de sua vida nos dosséis e utiliza a conexão da copa entre as árvores para o seu deslocamento (MONTGOMERY; SUNQUIST, 1975). Assim, árvores altas são essenciais, por cumprirem função fundamental no comportamento termo regulatórios desempenhado pelas preguiças, de subir no topo das árvores para se aquecer (MONTGOMERY; SUNQUIST, 1975; NAGY; MONTGOMERY, 1980). Assim, é possível concluir que existe uma série de fatores multi-escala responsáveis pela ocupação da espécie em determinado fragmento de mata.

A necessidade de se obter mais dados sobre a preguiça-de-coleira nos levou a um importante passo nessa tese – o NEOTROPICAL XENARTHRAS. A compilação e padronização das informações de ocorrência de espécies de xernatros, com mais de 30 mil registros de tatus, tamanduás e preguiças, produziu o maior banco de dados existentes para esse grupo ainda tão desconhecido. A análise desse banco de dados deixa clara a grande densidade de registros que se encontravam indisponíveis para o acesso do público em geral e o quanto ainda precisamos avançar nas pesquisas sobre o grupo. Todos os dados do NeoXen estão disponíveis no Github (https://github.com/LEEClab/Neotropical_Xenarthra) e a base de dados será alimentada à medida em que novos registros forem surgindo.

Com os resultados do capítulo 01 e com a compilação de dados do capítulo 02 foi possível gerar modelos de distribuição de espécie voltados para a avaliação da adequabilidade do habitat para as preguiças. A presença de florestas, mais uma vez, se mostrou como a principal preditora da espécie, uma vez que a disponibilidade de boas áreas está relacionada com a presença de uma boa cobertura florestal. Adicionalmente, a alta cobertura de florestas também se mostra bastante importante para mitigar os efeitos das mudanças climáticas sobre a espécie – contribuindo na conservação de uma boa proporção de áreas adequadas.

O estabelecimento de novas Unidades de conservação e a manutenção das já existentes é um importante passo para garantir a proteção das áreas naturais. Além disso, as Reservas Legais podem ser cruciais para resguardar áreas naturais, uma vez que, conforme mostrado nos capítulos 02 e 03, grande parte das áreas ocupadas e adequadas para a *B.torquatus* estão localizadas em propriedades particulares. Porém, para potencializar as suas funções, precisaria destituir as Áreas de Preservação Permanente do cálculo das RLs, uma vez que as duas categorias atuam em complementariedade na conservação e preservação de florestas. Da mesma forma, é necessário garantir o amparo legal das áreas potenciais para regeneração natural – ou as que já estão em processo de regeneração, a fim de evitar seu corte e a supressão

Por fim, uma característica dessa tese é a sua ação conservacionista aplicada. Todo o escopo dessa tese está relacionado com diversas metas do Plano de Ação Nacional para a Conservação dos Primatas da Mata Atlântica e da Preguiça-de-Coleira (PAN PPMA) - identificar áreas importantes, ampliar o conhecimento de ocorrência e ocupação e seus preditores ambientais - e os resultados gerados aqui irão integrar a avaliação da espécie. Além disso, todos os dados compilados no NEOTROPICAL XENARTHRAS já estão sendo utilizados pelos órgãos ambientais. O Centro Nacional de Pesquisa e Conservação de Primatas Brasileiros (CPB/ICMBio), responsável por avaliar os xenartros, tem utilizado os dados de ocorrência do *datapaper* para desenvolver as fichas de avaliação das espécies, por meio de seu portal SALVE. O Instituto Nacional da Mata Atlântica (INMA) também utilizou os dados referentes aos xenartros no estado do Espírito Santo para elaborar o a lista estadual de espécie ameaçadas (INMA, 2019). Assim, essa tese vem cumprindo seu papel na conservação da biodiversidade, ao integrar academia, ações de organizações conservacionistas e órgãos públicos, federal e estaduais, utilizando a ecolgia de uma espécie aplicada a sua conservação.

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