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Universidade Federal da Paraíba
Departamento de Sistemática e Ecologia
Pós-Graduação em Ciências Biológicas- Zoologia

Franciany Gabriella Braga Pereira

Conhecimento ecológico de populações indígenas e tradicionais sobre animais silvestres e uso de barreiros na Amazônia

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Tese de Doutorado

Conhecimento ecológico de populações indígenas e tradicionais sobre animais silvestres e uso de barreiros na Amazônia

Tese apresentada ao Programa de Pós-Graduação em Ciências Biológicas (Zoologia) da Universidade Federal da Paraíba, como requisito parcial para a obtenção do título de Doutora em Ciências Biológicas.

Franciany Gabriella Braga Pereira

Orientador: Prof. Dr Rômulo Romeu da Nóbrega Alves
Coorientador: Prof. Dr Pedro Mayor
Coorientador: Prof. Dr Carlos Peres

João Pessoa,
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Ata da 151ª Apresentação e Banca de Defesa de Doutorado de Franciany Gabriella Braga Pereira

Aos vinte e cinco dias do mês de fevereiro de dois mil e vinte e dois, às 09:00 horas, no Ambiente Virtual, da Universidade Federal da Paraíba, reuniram-se, em caráter de solenidade pública, membros da banca examinadora para avaliar a tese de doutorado de **Franciany Gabriella Braga Pereira**, candidata ao grau de Doutora em Ciências Biológicas. A banca examinadora foi composta pelos seguintes membros: **Dr. Rômulo Romeu da Nóbrega Alves (UEPB); Dra. Olga Lucia Montenegro Diaz (Univ. Nac. Colombia); Dr. Juarez Pezzuti (UFPA); Dr. Andre Pinassi Antunes (RedeFauna); Dra. Flavia Santoro (Univ. Nac. Córdoba).** Compareceram à solenidade, além da candidata e membros da banca examinadora, alunos e professores do PPGCB. Dando início à sessão, a coordenação fez a abertura dos trabalhos, apresentando a discente e os membros da banca. Foi passada a palavra ao orientador, para que assumisse a posição de presidente da sessão. A partir de então, o presidente, após declarar o objeto da solenidade, concedeu a palavra a **Franciany Gabriella Braga Pereira**, para que dissertasse, oral e sucintamente, a respeito de seu trabalho intitulado **"Conhecimento ecológico de populações indígenas e tradicionais sobre animais silvestres e uso de barreiros na Amazônia".** Passando então a discorrer sobre o aludido tema, dentro do prazo legal, a candidata foi a seguir arguida pelos examinadores na forma regimental. Em seguida, passou a Comissão, em caráter secreto, a proceder à avaliação e julgamento do trabalho, concluindo por atribuir-lhe o conceito APROVADO. Perante o resultado proclamado, os documentos da banca foram preparados para trâmites seguintes. Encerrados os trabalhos, nada mais havendo a tratar, eu, orientador, como presidente, lavrei a presente ata que, lida e aprovada, assino juntamente com os demais membros da banca examinadora.

João Pessoa, 25/02/2022.

Dr. Rômulo Romeu da Nóbrega Alves (UEPB
Orientador

Andre Pinassi Antunes
Dr. Andre Pinassi Antunes (RedeFauna)
Examinador

Olga L Montenegro D
Dra. Olga Lucia Montenegro Diaz (Univ. Nac. Colombia)
Examinadora

Dra. Flavia Santoro (Univ. Nac. Córdoba)
Examinadora

Dr. Juarez Pezzuti (UFPA)
Examinador

Franciany Gabriella Braga Pereira
(discente ciente do resultado)

(Em modo de webconferência, as assinaturas digitalizadas são certificadas pelo presidente da banca)

Dedico aos ribeirinhos e indígenas da Amazônia, maiores protetores e conhecedores desta floresta e de todos os seus mistérios.

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Resumo

A conservação da biodiversidade da Amazônia está vinculada a estimativas de abundância bem como ao monitoramento populacional das espécies silvestres. No capítulo 1 desta desse, apresento os bastidores do doutorado bem como uma introdução geral do que será abordado nesta tese. Existe a necessidade de padronizar métodos eficazes para a estimativa de abundância da fauna. Por isto, no capítulo 2 desta tese, comparamos dados de abundância de 91 espécies silvestres obtidos através de transectos lineares (9.221 km de trilhas) com dados obtidos a partir de 291 entrevistas estruturadas em 18 locais na Amazônia Central e Ocidental. Encontramos uma concordância significativa dos índices de abundância populacional para espécies diurnas e cinegéticas entre os dois métodos. Essa relação também foi positiva independente da sociabilidade da espécie, tamanho corporal e modo de locomoção; e do tipo florestal amostrado (florestas de terra firme e de várzea). No entanto, os transectos lineares não foram eficazes no levantamento de muitas espécies que ocorrem na área, com 40,2% e 39,8% de todas as espécies sendo raramente e nunca detectadas, respectivamente, em pelo menos um dos locais amostrados. Por outro lado, essas espécies foram amplamente relatadas por informantes locais como ocorrendo em abundâncias intermediárias a altas. Demonstramos a eficácia dasdas entrevistas para estimativa de abundância. Entretanto, precisávamos ainda saber quem deveriam ser os entrevistados em estudos de estimativa de abundância e sobre quais espécies poderíamos perguntar. Para isto, no capítulo 3, analisamos o grau de consenso sobre a abundância de 95 espécies entre 333 pessoas com diferentes características sociais e de experiências com a vida selvagem em 20 vilas com características demográficas distintas na Amazônia Ocidental e Central. Encontramos um alto índice de consenso (>0,6) quanto à abundância populacional da espécie para todas as vilas e para 79,64% dos entrevistados. O consenso entre os moradores de cada vila foi significativamente maior quanto menor o tamanho da população da vila. O valor de consenso também foi alto independentemente do tempo de caça dos entrevistados. Considerando todas as 95 espécies, encontramos para 81

(85,26%) um valor de consenso alto em todos os locais amostrados. Espécies com maiores valores de consenso sobre sua abundância são aquelas de grupos de tamanhos maiores, mais abundantes e mais caçadas. Além de informações sobre a fauna na floresta como um todo, moradores locais sabem falar com exatidão sobre a abundância e comportamento das espécies em locais específicos, e foi a partir deste contexto que desenhamos os capítulos 4 e 5 desta tese. Particularmente em relação àà caça, em algumas regiões da Amazônia, 25% das atividades de caça ocorrem nos barreiros (chamados localmente geralmente de *chupador* no Brasil e *colpas* no Peru). Estes são locais com maior concentração de minerais naturais no solo e que estão localizados próximos dos igarapés, portanto sujeitos a inundações periódicas. Nestes locais, os caçadores montam suas redes meio as árvores nos barreiros para esperar os animais cinegéticos que visitam o local para consumo do solo. Este comportamento de consumo do solo pelos animais ocorre com o objetivo de suplementação mineral, bem como de desintoxicação do seu corpo. Em um local visitado por tantas espécies, durante as longas horas que os caçadores passam esperando a espécie alvo no barreiro, estes caçadores podem também observar com atenção o comportamento de outras espécies visitantes do local. No capítulo 4, a partir de entrevistas semi-estruturadas obtivemos informações sobre 31 espécies de vertebrados visitando 56 barreiros em duas regiões da Amazônia Central. Em termos de tipos de barreiros, encontramos três classificações distintas a depender do tamanho do barreiro, riqueza de espécies visitando o local e período de alagamento pela água do igarapé (são eles: barreiro, chupador e canamã). Apesar do consumo de solo e água nos barreiros serem os principais atrativos das espécies silvestres que visitam esses locais, as espécies identificadas nas entrevistas como usuárias dos barreiros tem também como objetivo de visita a predação e outras relações ecológias, bem como banho e outros comportamentos. Em geral, a estação de maior abundância de animais silvestres nos barreiros foi a vazante, quando o nível da água dos igarapés diminui e assim o barreiro fica exposto. Ou seja, apesar de tão importante, a disponibilidade de solo consumível pelas espécies é alterada pelo pulso d'água. Além dos barreiros naturais, a maior concentração de minerais nos solos amazônicos também tem sido gerada pela indústria de extração de petróleo, por meio da contaminação do solo pelas "águas produzidas", principal subproduto dessa indústria e que inclui alta concentração de minerais. No entanto, solos poluídos por petróleo também contêm alta concentração de compostos petrogênicos tóxicos. Através do conhecimento local foram identificados locais contaminados por petróleo na Amazônia onde os animais silvestres consomem o solo. No capítulo 5, investigamos e descrevemos a geofagia do solo e da água poluídos por petróleo por 26 espécies de mamíferos e aves através da análise de 8.623 vídeos gravados a partir de armadilhas fotográficas em três barreiros naturais e em dezesseis barreiros poluídos por petróleo localizados em uma concessão de bloco de petróleo na Amazônia. Documentamos um total de 3.818 visitas independentes de 26 espécies, tendo 62,3% dessas visitas provas de ingestão de solo por 18 espécies diferentes. Considerando as visitas com ingestão de solo, *Tapirus terrestris* foi responsável por 69,58% das visitas, seguida de *Mazama americana* (13,75%). Não encontramos diferença significativa na frequência de visitas em barreiros naturais quando comparadas com barreiros contaminados com petróleo, sendo alta a taxa de visitação em ambos. Esses resultados fornecem dados relevantes para confirmar que o comportamento geofágico pela fauna não é um fenômeno incomum em barreiros contaminados, mas sim um comportamento generalizado em áreas de extração de petróleo na Amazônia. Este resultado é ainda mais preocupante pois esses compostos podem bioacumular nos tecidos dos animais e alguns podem até biomagnificar através da cadeia alimentar, incluindo nos predadores de topo e também nas populações humanas locais. Chegamos ao fim desta tese concluindo que a combinação do conhecimento local e científico é uma ferramenta potencial para aprimorar nosso conhecimento sobre as espécies florestais tropicais e monitorar espécies e ambientes tão ameaçados por atividades de extração de petróleo. Estruturando desta forma estratégias mais eficazes para atingir as metas de conservação da biodiversidade.

Palavras-chave: Armadilha fotográfica, ciência cidadã, conhecimento local, entrevistas, mamíferos, monitoramento participativo, vertebrados

Abstract

Effective estimation of wildlife population abundance is an important component of population monitoring, and ultimately essential for the development of conservation actions. As the conservation research community faces the need to standardise effective methods for estimating fauna abundance, in chapter 2 of this thesis, we compared concomitant abundance data for 91 wild species from diurnal line transects (9,221 km of trails) and a LEK-based method (291 structured interviews) at 18 sites in Central and Western Amazonia. We found a significant agreement of population abundance indices for diurnal and game species. This relationship was also positive regardless of species sociality, body size and locomotion mode; and of sampled forest type (upland and flooded forests). However, line transects were not effective at surveying many species occurring in the area, with 40.2% and 39.8% of all species being rarely and never detected in at least one of the survey sites. On the other hand, these species were widely reported by local informants to occur at intermediate to high abundances. We demonstrate the effectiveness of interviews for abundance estimation. However, we still needed to know who should be interviewed in abundance estimation studies and about which species we could ask. For this, in chapter 3, we analysed the degree of consensus about the abundance of 95 species among 333 people with different social and wildlife experiences characteristics in 20 demographically distinct sites in the Western and Central Amazon. We found that village consensus was significantly higher the lower the population size of a given village. However, a high score of consensus (>0.6) was find regarding the species population abundance for all of the sampled villages and for 79.64% of the interviewees. The consensus value was also high regardless of the interviewees' hunting experience. Considering all 95 species, we found a high consensus score in all 20 sampled sites for 81 species (85.26%). Species that have greater consensus scores about their abundance are those living in larger sized groups, more abundant and more hunted. Specifically in relation to hunting, in some regions of Amazonia, 25% of the hunting activities occur in the salt licks, which consist in places with a higher concentration of natural minerals in the soil and which is

often found on the edges of creeks, therefore they are liable to periodic flooding as the level of water rises. The hunters hang their nets above the salt licks and wait for game animals to visit to ingest the mineral-rich soil. In the Amazon rainforest, animals exhibit geophagical behaviour in salt licks to obtain key mineral supplementation and detoxicate from plant secondary compounds, reducing digestive disorders in their bodies. In a place visited by so many wild species, during the long period that hunters spend there waiting for the target species, hunter can also acquire a high level of knowledge about species that pass-through salt licks during the year. Through LEK- based methods, in chapter 4, we obtained information on 31 species of vertebrates visiting 56 salt licks in two regions of Central Amazon through different seasons. In terms of types of salt licks, we found three distinct categories (*barreiro, chupador* and *canamã*) depending on the salt lick size, animals' visit period, and the diversity of visitors, as well as by the flooding period of the creeks water. Despite soil and water consumption in salt licks being the main attraction of wild species visiting these sites, species identified from the interviews as users of the salt licks also visit the place for bathing, predation and other ecological relationships and behaviours. In general, the season with the highest abundance of wild animals was the receding floodwaters season, because in this period the water level of creeks decreases and so the salt lick is exposed. In addition to natural salt licks, the higher concentration of minerals in Amazon soils has also been generated by the oil extraction industry, contaminating the soil with "produced waters", the main by-product of this industry and which includes a high concentration of minerals. Additionally, oil-polluted soils also contain high concentrations of toxic petrogenic compounds. Through local ecological knowledge, oil-contaminated sites where wild animals consume the soil were identified in Western Amazon. In chapter 5, we investigated the geophagy of oil-polluted soil and water by 26 species of mammals and birds through the analysis of 8,623 videos recorded from a camera trap programme in three natural salt licks and sixteen oil-polluted salt licks located in an oil block concession in Amazon rainforest. We documented a total of 3,818 independent visits from 26 species, with 62.3% of these visits displaying soil ingestion

proofs from 18 different species. Considering visits with soil ingestion, *Tapirus terrestris* accounted for 69,58% of the visits, followed by *Mazama americana* (13,75%). We did not find a significant difference in the visit frequency for natural salt licks when compared to oil-polluted salt licks, with the visitation rate high in both. Our results provide relevant data to confirm that geophagy by wildlife is not an unusual phenomenon in oil-polluted salt licks, but rather a widespread behaviour in oil extractive areas in the Amazon. Even worst, these compounds can bioaccumulate in animals' tissues and some can even biomagnify through the food chain, including top predators and also local human populations (whose subsistence depends on wild meat). This makes the consumption of oil-polluted soil a major concern for conservation biodiversity and public health.

Keywords: Camera trap, interviews, local knowledge, mammal, vertebrates

Sumário

Bastidores da tese

Em 2018, ao finalizar o mestrado eu estava na clássica dúvida de "emendo mestrado e doutorado ou dou uma respirada? " Decidi emendar, mas então me veio outra grande dúvida "Começo a pesquisar sobre a Amazônia, ou continuo na África subsaariana?". Decidi aceitar a sugestão de meu coorientador Carlos Peres e decidi estudar Amazônia. Entretanto, defendido o mestrado, comecei a dividir meu tempo para trabalhar nos artigos frutos da dissertação e nas obrigações do doutorado. Foi aí que veio uma angustia estranha, porque ao mesmo tempo que eu estava muito feliz porque começaria a trabalhar na Amazônia, um dos lugares mais incríveis do planeta, meu coração ainda estava completamente imerso na minha dissertação. Durante o mestrado pesquisei sobre impacto da guerra civil de Angola na biodiversidade e naquele momento de mudança eu completava 4 anos de dedicação a pesquisa sobre a biodiversidade em África.

No segundo mês do meu doutorado eu fui para Inglaterra passar 2 meses trabalhando com meu coorientador Carlos Peres e coloquei essa angústia para ele, que respondeu "assim que você começar seu campo na Amazônia, tal angustia passará". Eu acreditava que isto aconteceria, mas como começar o campo sem financiamento e cheia de disciplinas presenciais para fazer? E para completar, como eu tinha poucas horas por dia para me dedicar a finalizar os artigos oriundos do mestrado, porque tinha as responsabilidades do doutorado para cumprir, fiquei praticamente todo ano de 2018 trabalhando nestes artigos.

No Segundo semestre de 2018, terminei de fazer as disciplinas, mas o financiamento não tinha saído, os três retornos de solicitação que tive de financiamento até então, foram todos negando o pedido. Por fim eu decidi ir do meu bolso mesmo fazer um piloto na Amazônia, porque ou era isso ou a incerteza de escolha de tema não passaria. Lá fui eu para 2 reservas na bacia do Juruá, lugares lindos, mas que ao mesmo tempo me faziam sentir saudades da África.

Enquanto eu fazia o campo nestas reservas, o meu barqueiro, Sr. Joaquim (na capa desta tese), insistia que eu precisava ir trabalhar nos barreiros de um tal de "igarapé Ueré", que também ficava na bacia do Juruá, mas fora das reservas. Sr. Joaquim acabou me convencendo, eu comprei mais 100 litros de gasolina (que no local custa R$8,00 o litro) e fomos para o igarapé do Ueré.

Sabe estes lugares que mexem profundamente com você?

Então, foi este lugar para mim. Passei todos meus dias no Ueré literalmente chorando de emoção e finalmente falei: sim, quero muito estudar a Amazônia! Conheci os indígenas Kulina e outros ribeirinhos não- indígenas moradores daquele lugar encantado.

Photo credit: Franciany Braga-Pereira

Eu voltei de campo para João Pessoa renovada e extremamente motivada e queria começar meu campo de vez, mas os pedidos de financiamento não haviam saído ainda. Então lá se foram mais alguns meses escrevendo mais pedidos, escrevendo artigo e estudando estatística. Finalmente o primeiro aceite de financiamento saiu, e de um dia para o outro eu decidi que em 20 dias eu me mudaria para o Amazonas. Para completar, iniciado meu campo, recebo resposta positiva de outros 2 financiadores. Em 2019, iniciei meu campo na RDS do Amanã, que com certeza me apresentou o barreiro mais lindo da vida. Logo depois, viajei para a bacia do Rio Negro e seguia firme em campos sem fim, meu objetivo era conhecer e pesquisar em diversas partes do estado do Amazonas.

Mas então chega a Pandemia. Eu estava em Manaus quando recebo a notícia do caos que estava por vir e impossibilitada de seguir com o campo comentei com um pesquisador sobre minha preocupação com o futuro do meu doutorado, já que precisei interromper meu campo sem finalizar a coleta de dados para todos os artigos. Este pesquisador, chamado Hani el Bziri, me disse que iria escrever a outros pesquisadores e que poderíamos reunir um banco de dados já coletados. Assim, eu poderia seguir com minha tese durante o período que não pudesse ir para campo. E foi aí que comecei a conhecer pesquisadores que não pararam de me presentear com dados riquíssimos. Thais Morcatti, Pedro Mayor, Marina Vieira, Pedro Perez, Marti Orta e por aí segue a lista de nomes de pessoas responsáveis por muitos dos dados que compartilharei com vocês nesta tese (a lista complete de nomes está na sessão de agradecimentos). Para quem queria pesquisar barreiros e conhecimento tradicional no estado do Amazonas, através desta rede de apoio entre pesquisadores, pude aprender muito não só sobre a Amazônia Central, mas também sobre o Oeste da Amazônia.

Ao final desta tese o que tenho para dizer é que sigo desejando realizar pesquisas com a biodiversidade em África, mas que agora também uma boa parte do meu coração pulsa para e pela Amazônia.

CAPÍTULO 1

Introdução Geral

A abundância é um dos indicadores mais usados para avaliar o status da população de vertebrados silvestres, o que, em última análise, permite que os pesquisadores e gestores avaliem os efeitos das ameaças nas populações e a eficácia das ações de conservação (Kremen et al., 1994; Stephenson, 2019). Consequentemente, os métodos utilizados em campo para a estimativa da abundância são de suma importância para o sucesso do levantamento de dados (Fragoso et al., 2016). Nessa perspectiva, transectos lineares são um dos mais antigos e frequentes métodos utilizados para obtenção de dados de abundância da fauna (Plumptre, 2000; Stephenson, 2019). Por outro lado, métodos baseados no conhecimento ecológico local (LEK, do inglês local ecological knowledge) apesar de contribuir para a pesquisa auxiliando cientistas na localização e coleta de informações sobre plantas e animais desde o século XVI (Alves & Souto, 2015) vem sendo mais intensamente utilizados em pesquisas científicas apenas nas últimas três décadas. Atualmente, métodos baseados no LEK são usados para coletar informações sobre habitats, usos extrativistas da biodiversidade, conflitos humano-vida selvagem, ecologia e comportamento de espécies (Joa et al., 2018; Young et al., 2018), dinâmica populacional ao longo do tempo (Braga -Pereira et al., 2020), e para melhorar a governança local (Joa et al., 2018; Vieira et al., 2019).

O conhecimento ecológico local e o acadêmico surgem de sistemas distintos, porém, ambos possuem uma forma interna (particular de cada grupo) para validação da informação. O conhecimento acadêmico é validado principalmente por meio de revisão por pares por outros cientistas. Existem muitos vieses da mente humana que facilitarão que uma informação seja validada e a consistência de respostas fornecidas é uma das possíveis maneiras de validação. Neste caso, para o conhecimento local uma informação será validada e assim passada para as próximas gerações quando muitas pessoas experientes fornecerem respostas consistentes sobre um assunto específico. A partir da abordagem *ética*[1], as informações da população local sobre determinado assunto podem ser analisadas por meio da abordagem do consenso cultural (Burgess et al, 2018). A análise de consenso nos permitiria determinar os perfis de espécies, pessoas e aldeias com maior padrão de consenso nas estimativas de abundância a partir de dados obtidos por meio de entrevistas. Dessa forma, determinaríamos os perfis mais adequados à formação cultural do conhecimento em relação aos animais silvestres.

As populações locais interagem com animais silvestres de diferentes formas, através de suas atividades cotidianas na comunidade ou na mata, através da observação da natureza, da utilização de produtos oriundos de animais silvestres, de atividades como caça e pesca, do xerimbabo, de relações harmônicas ou conflituosas, entre outras (Braga-Pereira et al., 2021, Zamoner 2018). Como consequência, moradores locais desenvolvem um conhecimento intrínseco, holístico e complexo sobre a fauna e a natureza como um todo desde então. Sobre a atividade de caça, para populações tradicionais e indígenas da Amazônia esta prática é crucial na construção do conhecimento da natureza como um todo e nos aspectos cosmológicos associados a ela, pois os caçadores que percorrem a paisagem em busca de presas vão aprendendo e descobrindo de maneira muito profunda a floresta e seus mistérios. Na Amazônia, alguns locais específicos são considerados pontos de convergência cruciais entre a fauna cinegética[2] e os caçadores, onde o esforço de caça geralmente é concentrado. Em algumas regiões da Amazônia 25% das atividades de caça ocorrem nos barreiros (locais de maior concentração de minerais naturais), onde caçadores instalam suas redes e ficam esperando os animais cinegéticos que vão consumir o solo do local ou passam para verificar se há algum animal durante a caçada de busca ativa (Walschburger e Hildebrand, 1988). Os caçadores passam horas nestes barreiros, sempre com muito cuidado e zelo ao local tido por muitas culturas como sagrado. Por investirem parte de seu tempo esperando nos barreiros, caçadores locais constroem amplo conhecimento sobre as espécies que visitam o local ao longo

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[1] Abordagem ética é a aquela de fora, ou seja, da perspectiva do observador/pesquisador.

[2] Espécies cinegéticas são as espécies alvo de caça

do ano (não só sobre a fauna cinegética) e sobre outros aspectos ecológicos destas paisagens.

Barreiros são formações geológicas naturais com altas concentrações minerais (como Na) onde os animais visitam e consomem o solo ou água (Abrahams e Parsons 1996; Krishnamani e Mahaney 2000). Em inglês estes barreiros são chamadas de salt lick (termo mais comumente empregado) mineral lick e natural lick (Klaus and Schmid, 1998; Montenegro 2004). Nas comunidades tradicionais da Amazônia brasileira um termo muito comum é chupador, pois o animal vai até o local chupar a água, comportamento que inclusive pode ser escutado pelos caçadores enquanto esperam seu alvo de caça na rede.

A principal motivação por trás deste comportamento geofágico parece mudar entre as espécies que visitam o local. Entretanto um frequente atrativo da fauna é a presença de micronutrientes chave que faltam na dieta de espécies herbívoras e onívoras (Atwood e Weeks 2002, 2003; Davies e Baillie 1988; Voros et al. 2001). A deficiência de minerais ocorre porque a maior parte da floresta amazônica é caracterizada por ter solos muito ácidos, com baixo teor de nutrientes disponíveis e alta concentração de Al tóxico. Além disso, a Amazônia ocidental é uma região geograficamente desprovida de sal, pois a deposição de sais em aerossóis diminui com a distância das fontes oceânicas (Dudley et al., 2012). Nesse tipo de ambiente, as espécies herbívoras podem enfrentar limitações minerais se sua única fonte de minerais são os recursos vegetais. Se estas paisagens na região amazônica fornecem alguns minerais de importância nutricional para os herbívoros, sua existência pode reduzir os custos de manter a saúde destes animais, desta forma, pode ser fundamental para a persistência das populações de espécies silvestres.

Na Amazônia peruana, onde os barreiros são chamadas de *colpas*, moradores locais comunicaram aos pesquisadores de nosso grupo de pesquisa que os animais estavam consumindo solo próximo a centros de extração de petróleo e este solo para os moradores locais estava também contaminado com petróleo. O consumo de solo aparentemente contaminado estava preocupando os moradores devido ao potencial impacto à saúde dos animais e também das pessoas que consumiam daqueles animais. Os pesquisadores comunicados então coletaram amostras de solo onde segundo os moradores locais a fauna fazia consumo e após análise laboratorial confirmaram a presença de petróleo no solo contaminado (Orta Martínez et al., 2007).

A atração da fauna para estes locais contaminados por petróleo se dá, pois, um dos principais subprodutos da indústria de extração de petróleo, a chamada "água produzida", apresenta altas concentrações de minerais (Fakhru'l-Razi et al., 2009). O alto teor de salinidade da água produzida pode funcionar como chupadores naturais, atraindo os animais para consumir solos poluídos por petróleo (Emmons e Stark, 1979). No entanto, estes solos poluídos também contêm alta concentração de isótopos radioativos e agentes tóxicos, como hidrocarbonetos (por exemplo, benzeno, xileno ou tolueno) e metais pesados (por exemplo, bário (Ba), arsênico (As) ou mercúrio (Hg))) (IARC, 1988, 2012; Doyle, 1994; Fakhru'l-Razi et al., 2009; Konkel, 2016). Dessa forma, como solos com alta concentração de minerais têm sido procurados por diversas espécies animais, solos contaminados por óleo podem ter se tornado uma fonte mineral atrativa e seu consumo pode iniciar uma rota de integração de metais pesados na cadeia trófica em hotspots altamente biodiversos da Amazônia, impactando também as populações humanas locais.

Diante disto, o objetivo principal desta tese foi, a partir do conhecimento ecológico de populações indígenas e tradicionais da Amazônia, analisar formas de estimativas de abundância de vertebrados silvestres e o uso de barreiros naturais e antropogênicos pela fauna da Amazônia.

Estrutura da Tese

Esta tese está dividida em 6 capítulos. O primeiro capítulo contém a introdução geral, os capítulos 2, 3, 4 e 5 contém os artigos originados da tese e o capítulo 6 contém uma síntese das atividades curriculares e extra curriculares desenvolvidos durante o doutorado (Fig. 1).

O Capítulo 2 foi publicado na revista Methods in Ecology and Evolution. O capítulo 3 será submetido para a revista People and Nature. O capítulo 4 será submetido para a revista Ethnobiology and Ethnomedicine. O capítulo 5será submetido para a revista Nature Sustainability. Cada capítulo está formatado de acordo com as normas das revistas que foi publicado ou a qual será submetido.

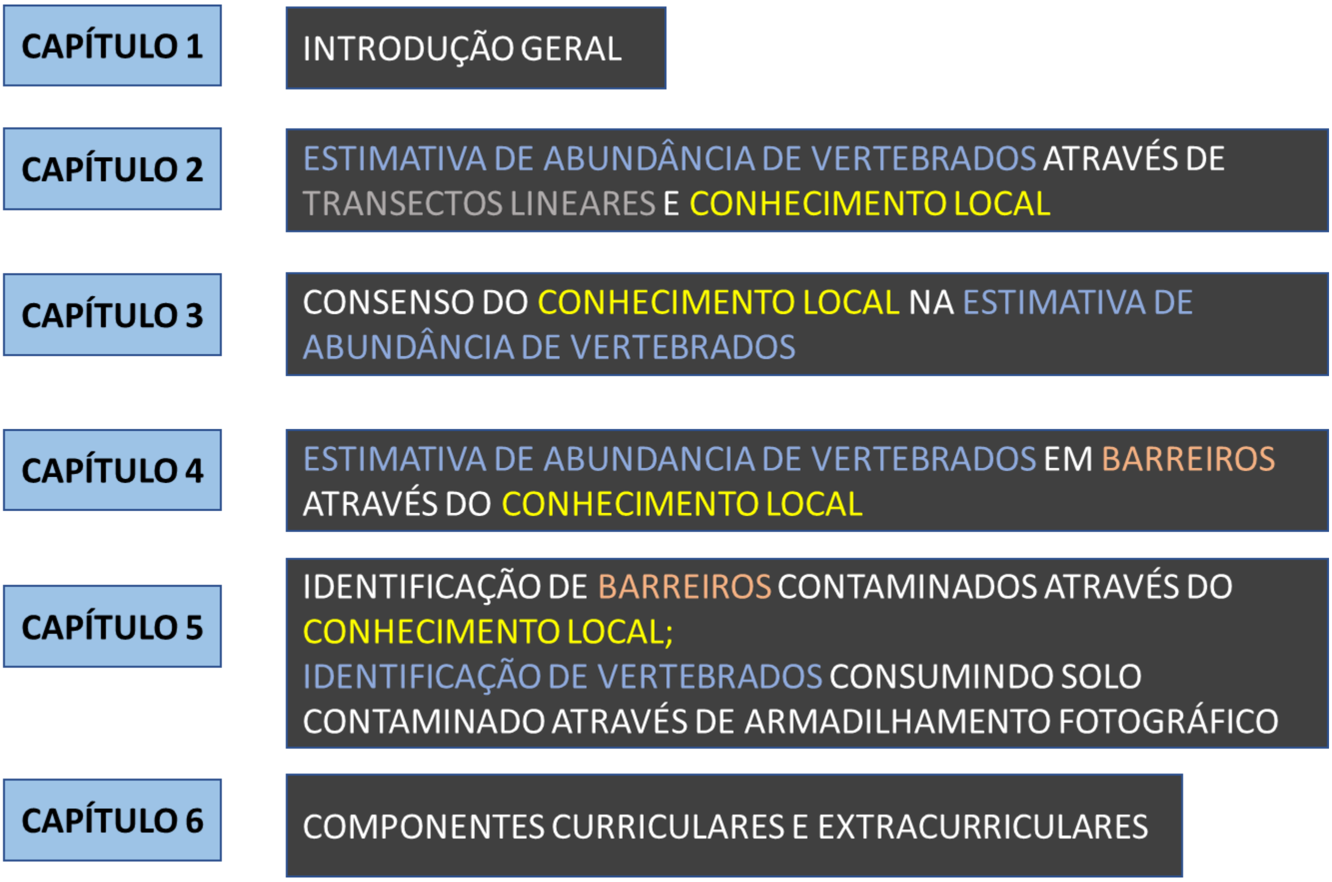


Figura 1. Resumo descritivo de objetivos de cada capítulo

CAPÍTULO 2.

Congruence of local ecological knowledge (LEK)-based methods and line-transect surveys in estimating wildlife abundance in Tropical forests

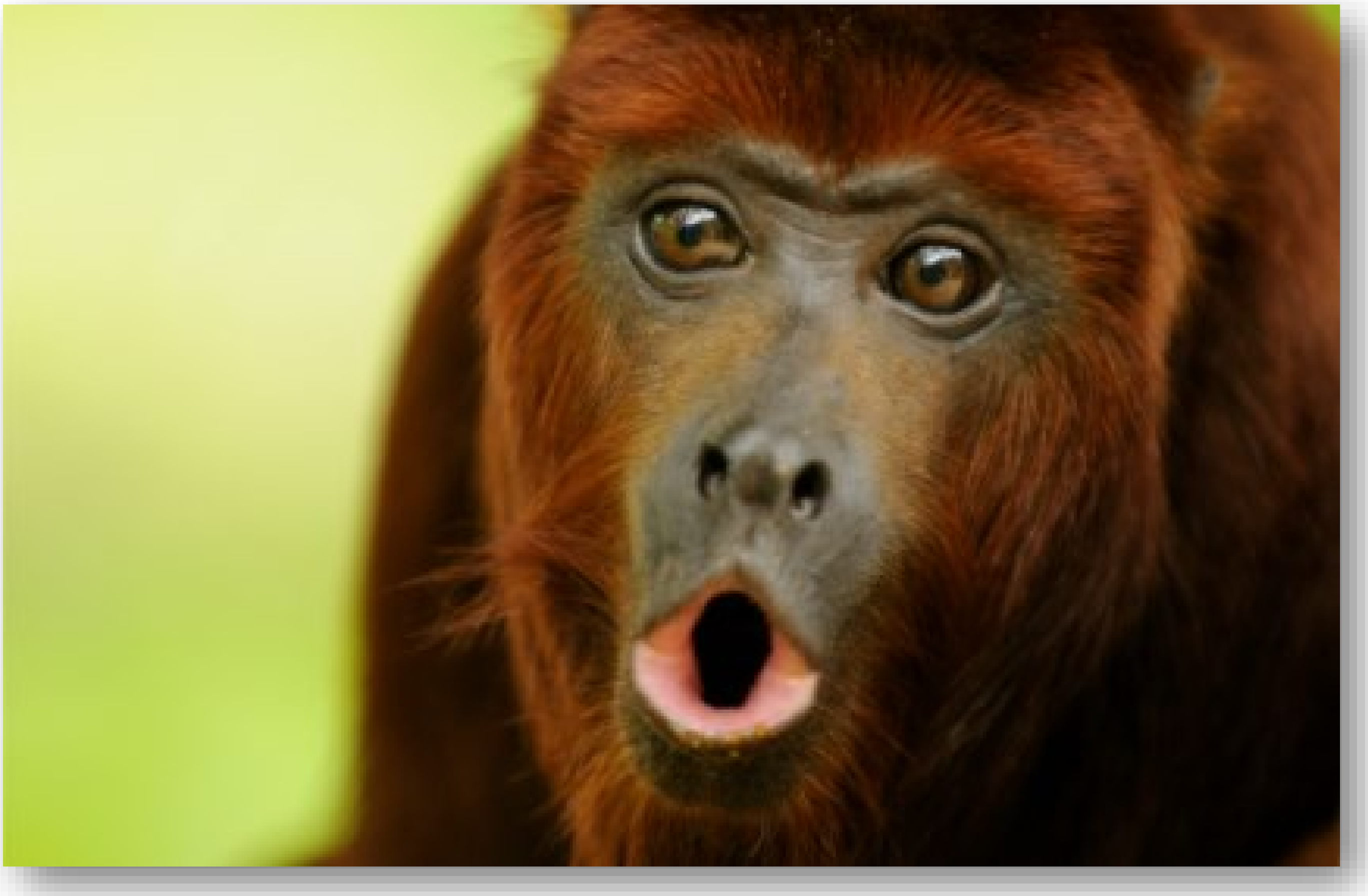


Photo credit: Mark Bowler

Abstract

1)Effective estimation of wildlife population abundance is an important component of population monitoring, and ultimately essential for the development of conservation actions. Diurnal line transect surveys are one of the most applied methods for abundance estimations. Local ecological knowledge (LEK) is empirically acquired through the observation of ecological processes by local people. LEK-based methods have only been recognized as valid scientific methods for surveying fauna abundance in the last three decades. However, the agreement between both methods has not been extensively analysed.

2)We compared concomitant abundance data for 91 wild species (mammals, birds and tortoises) from diurnal line transects (9,221 km of trails) and a LEK-based method (291 structured interviews) at 18 sites in Central and Western Amazonia. We used biological and socioecological factors to assess the agreements and divergences between abundance indices obtained from both methods.

3)We found a significant agreement of population abundance indices for diurnal and game species. This relationship was also positive regardless of species sociality (solitary or social), body size and locomotion mode (terrestrial and arboreal); and of sampled forest type (upland and flooded forests). Conversely, we did not find significant abundance covariances for nocturnal and non-game species. Despite the general agreement between methods, line transects were not effective at surveying many species occurring in the area, with 40.2% and 39.8% of all species being rarely and never detected in at least one of the survey sites. On the other hand, these species were widely reported by local informants to occur at intermediate to high abundances.

4)Although LEK-based methods have been long neglected by ecologists, our comparative study demonstrated their effectiveness for estimating vertebrate abundance of a wide diversity of taxa and forest environments. This can be used simultaneously with line transects surveys to calibrate abundance estimates and record species that are rarely sighted during surveys on foot, but that are often observed by local people during their daily extractive activities.

Thus, the combination of local and scientific knowledge is a potential tool to improve our knowledge of tropical forest species and foster the development of effective strategies to meet biodiversity conservation goals.

Keywords (8): Amazon, citizen science, ethnobiology, ethnozoology, vertebrates, traditional knowledge, subsistence hunting.

Introduction

Abundance is one of the most used indicators to assess wildlife population status, which ultimately enables practitioners to assess the effects of threats on populations and the effectiveness of conservation actions (Kremen et al., 1994; Stephenson, 2019). However, surveying wildlife abundance remains challenging due to financial and logistical limitations, which are more pronounced in long-term studies in poorly accessible areas. In addition, constraints posed by certain species' biological traits may result in underestimated detection through conventional methods (MacKenzie et al., 2006; Nichols and Williams 2006). While abundant species and those with small home ranges require a moderate sampling effort to estimate their abundance, rare species and those with large home ranges may be difficult to detect, decreasing the accuracy of abundance estimations (Plumptre, 2000). Consequently, methods used in the field can determine the success or failure of abundance surveys (Fragoso et al., 2016). The best method should ideally ensure high detection rates of the target species whilst also being cost-effective and accurate (Fragoso et al., 2016; Guillera-Arroita, 2016).

Line transects surveys are frequently used as a method to obtain abundance data of fauna (Stephenson, 2019; Plumptre, 2000). This is mainly because of the broad range of species this method can target, being used to assess the status of populations ranging from whales in the ocean to small invertebrates in forests (Peres and Cunha 2012; Haugaasen and Peres, 2005; de Thoisy et al., 2008). However, line transects surveys require intensive sampling effort (de Thoisy et al. 2008) and are often conducted diurnally, resulting in poor estimates of abundance mainly affecting nocturnal and less abundant species (Munari et al., 2011). Even during night surveys, the efficiency of the technique on monitoring nocturnal species is generally low, given the limited human visual capacity and the inability of observers to move in silence (Munari et al., 2011). Line transects can therefore be costly, time and staff consuming, and require year-round assessments to adjust for seasonal changes in abundance and behaviour (Fashing and Cords, 2000; Van der Hoeven et al., 2004).

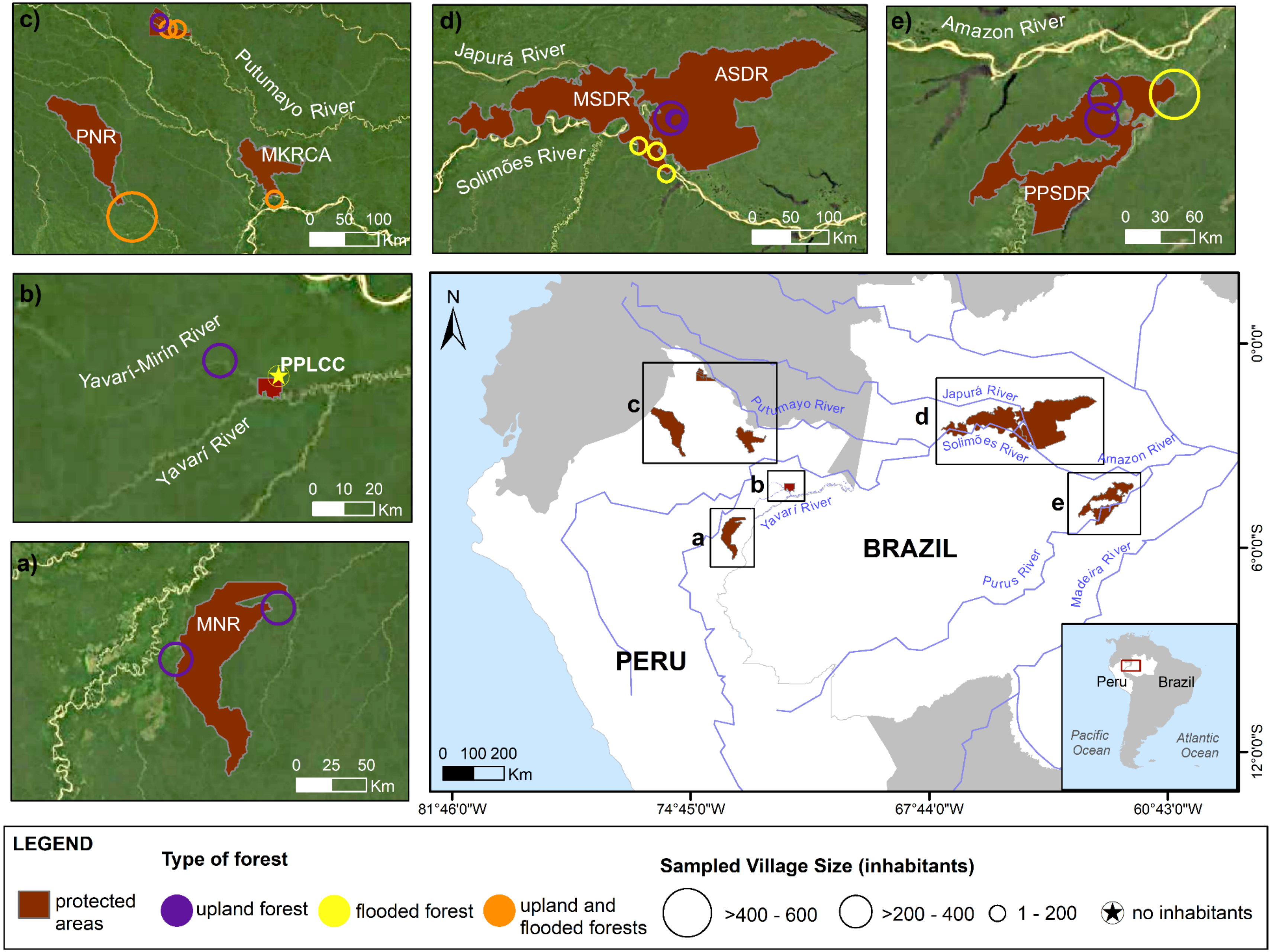
The integration of natural and social science methodologies in conservation studies has gained traction over the past three decades, mainly through the "citizen science" and the "ethnoscience" approaches (Berkes, 2017). Both approaches can involve the use of local ecological knowledge (LEK), which is defined as the knowledge and practices of local people regarding ecological relationships that are gained through extensive personal empirical observations of and interactions with local ecosystems, and shared among local resource users (Charnley et al., 2007). As LEK includes traditional, indigenous and local ecological knowledge, we herein use the term LEK instead of traditional or indigenous ecological knowledge. LEK has contributed to research by assisting scientists in locating and collecting information on plants and animals since the 16th century (Alves and Souto, 2015). Currently, LEK-based methods are used to gather information on habitats, extractive uses of biodiversity, human-wildlife conflicts, species ecology and behaviour (Joa et al., 2018; Young et al., 2018), population dynamics over time (Braga-Pereira et al., 2020), and enhance governance (Joa et al., 2018; Vieira et al., 2019).

LEK-based methods have also been applied to develop new scientific methods to overcome previous methodological hurdles (Morcatty et al. 2020; El Bizri et al. 2016; Parry and Peres 2015) and may provide a cost-effective and robust understanding of natural systems that are likely to equate to or exceed that of conventional scientific knowledge (Gagnon and Berteaux 2009; Meijaard et al., 2011). Hence, the combination of local knowledge and methods conventionally used by wildlife ecologists could improve species' detection rates, facilitate mutual learning and local empowerment, and contribute to enhance conservation goals (Burgess et al., 2018). To date, studies have focused on the comparison of the two methods regarding the species detection rate for one or a few sets of species (see Anadón et al., 2009; Camino et al., 2020; Madsen et al., 2020; Perez-Peña et al., 2012). In this study, we estimated and compared abundance indices of 91 species of wild vertebrates (including mammals, birds and tortoises) using data collected concomitantly through diurnal line transect censuses and perceptions of local people through a LEK-based method at 18 sites around indigenous and non-indigenous riverine villages in the Western and Central Amazon. We also examined some biological and socioecological factors that can explain agreements and divergences between both methods to develop a better understanding of their limitations and potentials.

Methods

Study area and villages

This study was conducted in 18 sites located in the Brazilian (n=9) and the Peruvian (n=9) Amazon. These include eight specific sites in upland forest, four in flooded forest and six in both upland forest and flooded (Fig. 1). Eight locations are within indigenous villages, nine are in non-indigenous riverine villages, and one site has no human settlement (Supplementary Material, Table 1). The non-indigenous riverine villages are located in Sustainable Use Protected Areas, which are a legally recognized category of protected area in which traditional people partake in decision-making on natural resource use and management. Hunting remains an important subsistence activity for the residents living within these areas.


c)
Putumayo River
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0 50 100
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0 30 60
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b)
Yavarí-Mirín River
PPLCC
Yavarí River
0 10 20
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a)
MNR
0 25 50
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Putumayo River
d
Japurá River
Solimões River
Amazon River
b
Yavarí River
e
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BRAZIL
Purus River
Madeira River
PERU
0 100 200
Km
Peru
Brazil
Pacific Ocean
Atlantic Ocean
0°0'0"
6°0'0"S
12°0'0"S
81°46'0"W
74°45'0"W
67°44'0"W
60°43'0"W
LEGEND
protected areas
Type of forest
upland forest
flooded forest
upland and flooded forests
Sampled Village Size (inhabitants)
>400 - 600
>200 - 400
1 - 200
no inhabitants

Figure 1. Map of the study area portraying the 18 sites in Central and Western Amazonia. Brown background areas represent protected areas; MNR: Matsés National; PPLCC: Lago Preto and Paredón Conservation Concession; PNR: Pucacuro National Reserve; MKRCA: Maijuna-Kichwa Regional Conservation Area; MSDR: Mamirauá Sustainable Development Reserve; ASDR: Amanã Sustainable Development Reserve; PPSDR: Piagaçu-Purus Sustainable Development Reserve. Map generated using ArcGIS 10.3.1; Datum: WGS84 Source: ESRI, Edited in Adobe Photoshop and Elaborated by Nadia Zamboni and Franciany Braga-Pereira in December 2020.

Ethics statement

We followed the rules and guidelines for applying Free, Prior and Informed Consent as detailed in Buppert and McKeehan (2013). This research was approved by the Instituto Chico Mendes de Conservação da Biodiversidade from Brazil (License SISBIO 29092-1; SISBIO 2; 29092-3; SISBIO 29092-4; SISBIO 29092-5; SISBIO 29092-6; CEUC 1474/2011, CEUC 003/2013 e CEUC 052/2011) and the Dirección General de Flora y Fauna Silvestre from Peru (License 0350-2012-DGFFS-DGEFFS; 0068-2015-SERFOR-DGGSPFFS). Community meetings and coordination with communal authorities were carried out prior to conducting interviews to agree on procedures.

Data collection

Between 2011 and 2017, we surveyed the abundance of a set of species through line transects, and through interviews with local people from 17 different indigenous and non-indigenous riverine villages. All villages were settled in or near the sites where transects were surveyed, and local people use these sites for different purposes (hunting, harvesting of forest products, etc.); therefore, each village offered information about at least one correspondent site. Sixteen villages informed LEK about a single correspondent site, and 1 village informed LEK for 2 correspondent sites. Interviews and line transects within each study area were conducted in a mean lapse time of 8.6 months, ranging from 0 to 24 months.

The species considered in this study did not necessarily occur in all study sites, and each sampling was conducted considering the species known to occur in a given region from previous studies. In total we surveyed the abundance of 91 species, with a median of 35 species (range = 14 - 45) per site. This number included 45% Primates (number of species = 41), 13.2% birds (n=12), 13.2% Carnivora (n=12), 8.8% Rodentia (n=8), 6.6% Pilosa (n=6), 5.5% Artiodactyla (n= 5), 3.3% Cingulata (n=3), 2.2% Didelphimorphia (n=2), 1.1% Perissodactyla (n=1) and 1.1% Testudines (n=1) (Supplementary Material, Table 2). All bird species considered in this study consist of gamebirds.

Line transects

We estimated the population abundance of each species from direct diurnal sightings conducted on 31 line transects, with a total surveyed distance of 9,221 km (ranging from 42 km to 2,687 km surveyed per site; mean= 512 Km, SD= 707) (Supplementary material, Table 1). Each transect was randomly positioned in all study areas and transects were opened prior to the surveys. Two observers (at least one of them was a local monitor) walked the trails between 6:00 h and 15:00 h at an average speed of 1.5 km/h. When a group of animals was encountered, the number of individuals and species was recorded. From the collected data, we calculated the sighting rates (individuals/km, calculated as the total number of individuals observed divided by the total effort in km travelled on any given transect during all seasons), which were used as our abundance index since higher abundance increases species detections (e.g. Paim et al., 2019). The value of zero was assigned to species whose occurrence is confirmed in the area but that were not detected on any transect sampled near that village.

Local Ecological Knowledge

We interviewed 291 local people from the sampled villages (average interviewees per village =16.16, SD=6.62) using a snowball sampling technique (Bailey, 1994) through the indication by each interviewee of another local expert on fauna. The interviewees' ages ranged between 16 to 75 years old (average = 37.75; SD=13.29). We conducted interviews individually to collect the interviewee perception of the abundance through LEK-based methods of each species that occur in the area in which the interviewee lives. Interviews did not require local translators as both the interviewers and the interviewee, including those from indigenous territories, were fluent in Portuguese or Spanish. All researchers conducting the interviews were already working in each site and had built relationships of trust in the communities.

Data were collected through structured interviews with an illustrated checklist, which provided colour plates of species expected in each study area (Supplementary Material, Table 1). During each interview we asked the local vernacular name for each species illustration, often corresponding to the species common nomenclature in Portuguese or Spanish. For each species, we asked the interviewee to estimate their abundance on a Likert scale; 0 (when the species was "absent"), 1 (low abundance), 2 (medium abundance) and 3 (high abundance) (Van Holt et al., 2010; Van Holt et al., 2016). The value assigned by each interviewee for each species was considered as our abundance index for the LEK-based method. The value of zero was assigned only to species whose occurrence is expected for the area by previous studies but was mentioned as absent by a specific interviewee. We validated the consistency of the responses through a cultural consensus analysis (Borgatti and Halgin 2011), which consists in a multivariate test based on the degree of similarity between respondents' answers. In this case, respondents showed a personal consensus higher than 0.6 (indicating a high consensus) regarding the abundance indices of each species population.

Data compilation

Covariates

Species traits

Species traits were used to help explain the agreement and divergence between abundance values obtained through line transects and the LEK-based method. These included body mass, sociality (solitary/social - with two or more individuals), habit (diurnal/nocturnal) and locomotion mode (arboreal/terrestrial). For some analysis, we also used body mass categories; small (less than 1 kg), medium (between 1 kg and 5 kg) and large-sized (exceeding 5 kg) species, considering adult average body mass (Emmons and Feer, 1990).

Hunting rate

We used data in Peres (2000) to obtain information on the hunting rate (in number of individuals hunted per person per year) of each species across the Brazilian Amazon; if a species was not listed by Peres (2000), we considered a hunting rate of 0. For some analysis, we also divided into quantiles the distribution of hunting rate values of all species, forming the following hunting rate ordinal classes: no hunting (0), low hunting (until 0.05), moderate hunting (between 0.05 and 0.35) and heavy hunting (until 1).

Forest type

We used the forest type of each surveyed site as a covariate. The studied sites were either in upland forest and/or white-water flooded forest. Upland forest (*terra-firme*) is a non- flooded forest located in higher sites of the Amazonian rainforest. White-water flooded forest (*varzea*) is a seasonal floodplain forest inundated by white-water rivers that occurs in the Amazonian rainforest.

Data analysis

Drivers of abundance for line transects and LEK-based method

We examined the effect of species traits, hunting rate and forest type on line transect and LEK-based abundance indices. For the transect data analysis, we used GLMM with the negative binomial distribution. We considered i) line transect abundance index values as a response variable; ii) species traits, hunting rate and forest type as a predictor variable of fixed effects; and iii) species as random variable.

For the LEK-based method data analysis, we used cumulative link mixed model (CLMM) because the data of the perceived abundance are ordinal, ranging from 0 to 3. For this model, we considered i) LEK-based abundance index as a response variable; ii) species traits, hunting rate and forest type as a predictor variable of fixed effects; and iii) sites and species as random variables. In this case, abundance indices collected in each interview were compared per species within a particular site.

There was no collinearity ($p > 0.05$) among predictor variables. For GLMM and CLMM, we used residual checks to verify whether our models were, in principle, suitable or otherwise. We used the Akaike information criterion to select models of interest if ΔAIC values >6 (ΔAIC obtain from the difference between a null and complete model AIC values) (Harrison et al., 2018; Richards, 2008). All analyses were performed in R ver. 3.5.3 (R Core Team

2019) using the ordinal (Christensen, 2019), MuMin e lme4 (Oksanen et al., 2013) packages.

Comparison of abundance indices obtained through line transect and LEK-based method

We conducted generalized linear mixed models (GLMMs) to examine the relationship between the abundance indices from the two methods, comparing the within-species abundance index from each interviewee and the abundance index obtained through line transects at each site. We considered i) line transect abundance index as a response variable; ii) LEK-based method abundance index as a predictor variable of fixed effects; and iii) site and species as random variables following Zuur et al., (2007). In this analysis, we nested the effect of each species within each particular site (see Supplementary material, Table 1).

Firstly, we analysed the entire dataset in one initial model. Then, we stratified the dataset into different sets considering the following groups in different models: diurnal, nocturnal, arboreal, terrestrial, solitary, social, small, medium and large-sized body; none, low, medium and high hunting rate; upland and flooded forests. We did this stratification to clarify the relationship between abundance indices from transect and LEK-based method according to different biological and socioecological factors (Supplementary material, Table 1).

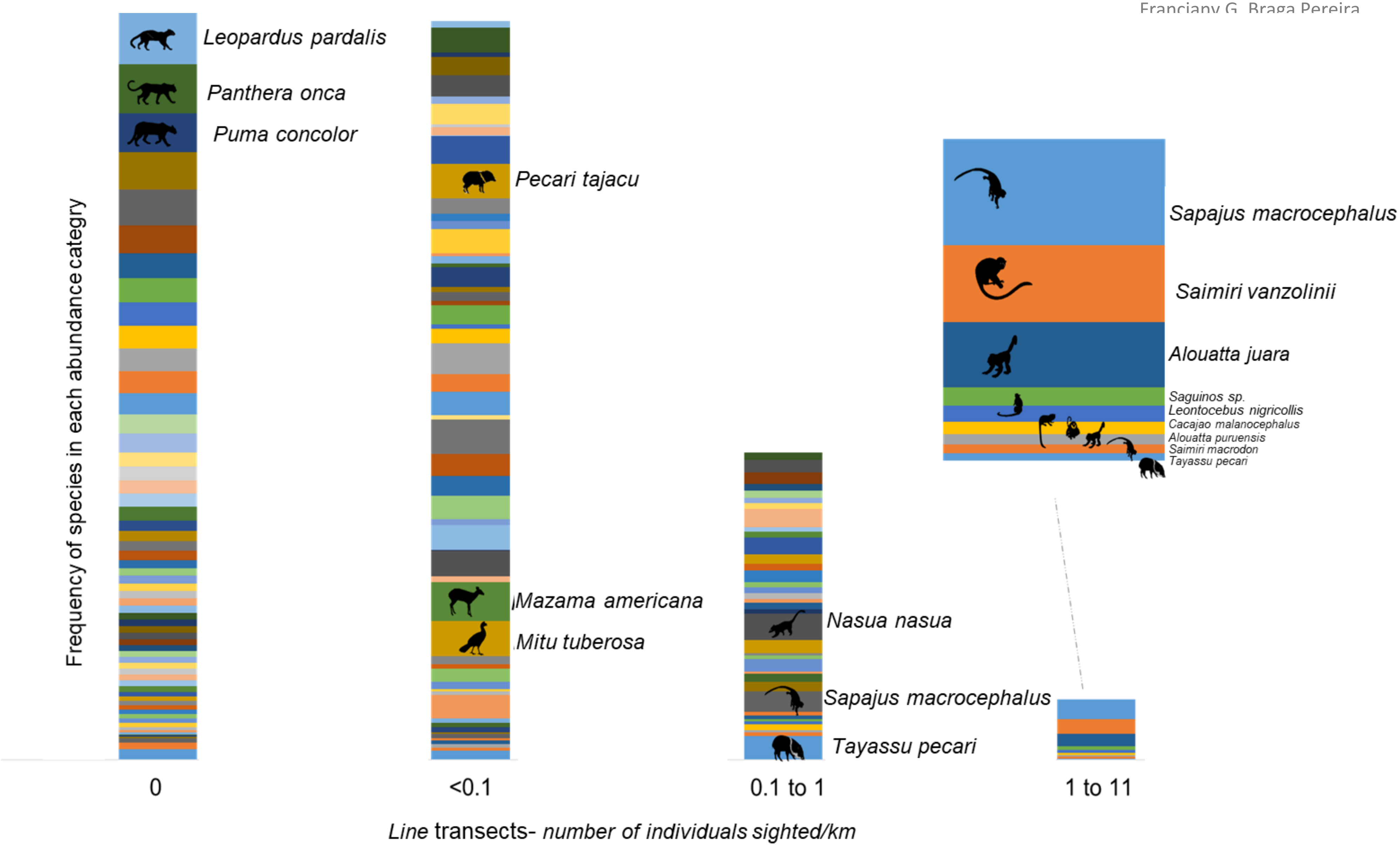
Given that line transect abundances consist of over-dispersed count data, we used the negative binomial distribution. Given that each of these GLMM had only one predictor, we used frequentist statistics to evaluate the relationships between variables, presenting in each case p-values (<0.05), confidence intervals, F- values, degrees of freedom and adjusted r squared values (>0.6).

We used pairwise Pearson correlations to examine the strength and direction of species-specific correlation coefficients between abundance indices based either on the LEK-based method or line-transect censuses. Here, we calculated the mean and the error (± 95% CIs) of correlation coefficients for each species. For these correlations we excluded all species occurring at fewer than four of all 18 sites. To boost sample sizes and the number of species included in the analysis, we pooled all taxonomic species into ecological analogues or functional groups (hereafter, ecospecies), typically defined as closely related parapatric species or congeners that replace one another across geographic boundaries (see Peres & Palacios, 2007). Considering that our sampling unit (number of interviewees) is ~300, a correlation value > 0.113 can be considered highly significant at alpha = 0.05 (see Statistics Solutions, 2021).

Results

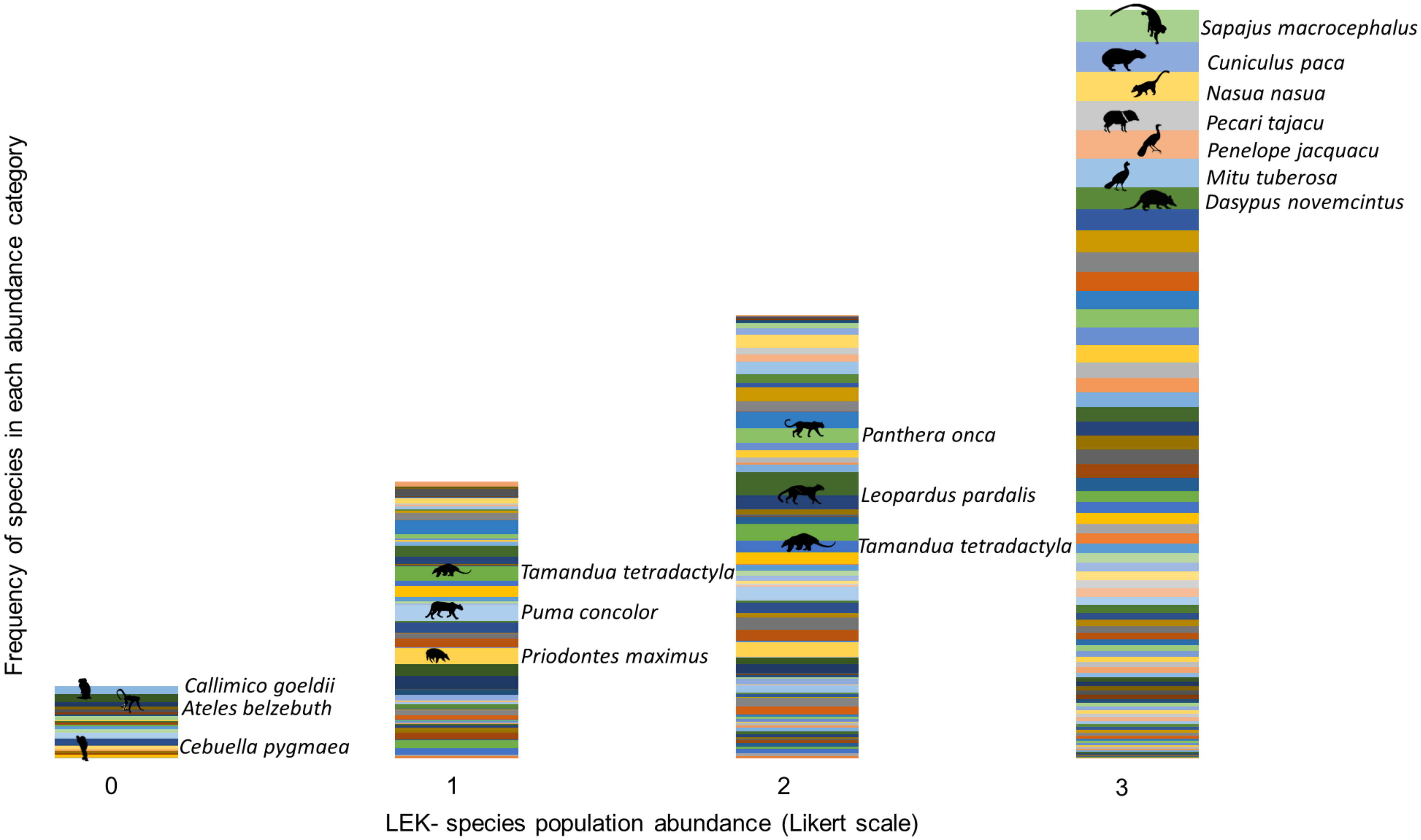
Abundance estimates using line transects and the LEK-based method

For line transect surveys, most species were either undetected (39.8%) or yielded an abundance below 0.1 individuals per km walked (40.2%) (Fig 2; Supplementary material, Fig 2). Many of no- or low-detection species during surveys on foot had been widely recorded through the LEK-based method as having medium (44%) or high (41%) abundance in the area. For example, *Coendou spp.*, *Pipile cumanensis*, *Bradypus tridactylus, Cheracebus torquatus* and *Choleopus didactylus* were not recorded in any transect but were reported as occurring in all interviews. Conversely, the distribution of species abundance indices in the histogram according to LEK-based method had only 4.7% of the data representing species that are supposedly absent in the area, and 17.9%, 28.7% and 48.6%, representing species with low, medium and high abundance indices, respectively (Fig 3). In 90.3% of the occasions when an interviewee reported that a species was absent in the village, the species was also not detected in the area through line-transect surveys. Conversely, only on five occasions did interviewees say that a species recorded during surveys were absent in the area (twice for *Sciurus* sp. and *Pithecia albicans,* and once for *Lagothrix poeppigii*).


Leopardus pardalis
Panthera onca
Puma concolor
Pecari tajacu
Mazama americana
Mitu tuberosa
Nasua nasua
Sapajus macrocephalus
Tayassu pecari
Sapajus macrocephalus
Saimiri vanzolinii
Alouatta juara
Saguinos sp.
Leontocebus nigricollis
Cacajao malanocephalus
Alouatta puruensis
Saimiri macrodon
Tayassu pecari
Frequency of species in each abundance category
0
<0.1
0.1 to 1
1 to 11
Line transects- number of individuals sighted/km

529

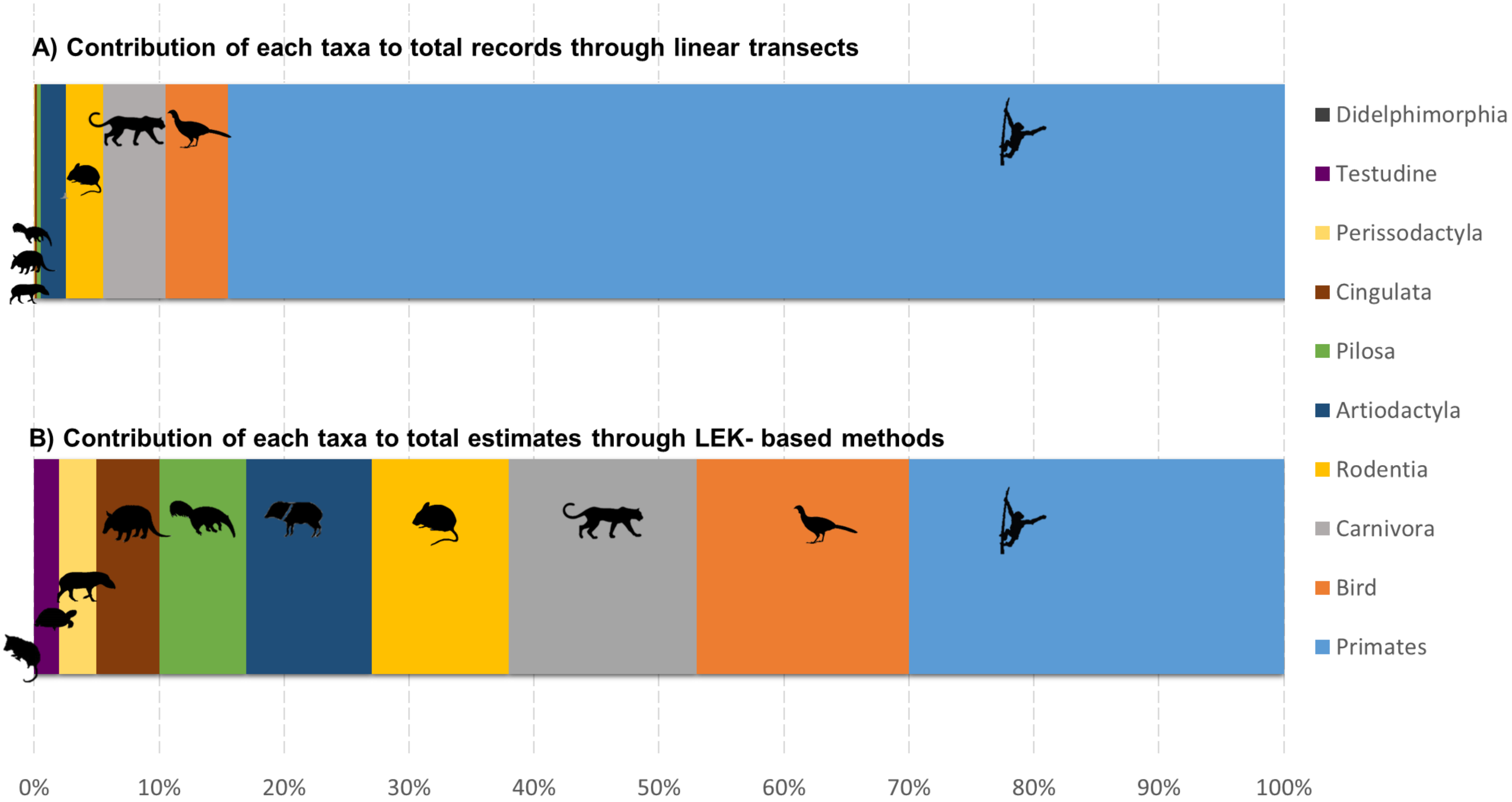
Figure 2. Distribution of species abundance according to linear transect. Abundance values were grouped into 4 categories (x-axis): 0 (when the species was never recorded in transects of a specific village); <0.1 individuals per km; from 0.1 to 1 individual per km; and from 1 to 11 individuals per km. Each coloured box represents a species and box sizes represent the number of sites in which the species was detected with that abundance category. Y-axis represents the frequency with which each species was recorded in each abundance category. A close-up was performed on the last category for better visualization of the most registered species. Species can occur in more than one category because the graph is based on the number of sites with a certain abundance category for each species. We included a silhouette in the box of the most registered species in each category.


Frequency of species in each abundance category
Sapajus macrocephalus
Cuniculus paca
Nasua nasua
Pecari tajacu
Penelope jacquacu
Mitu tuberosa
Dasypus novemcintus
Panthera onca
Leopardus pardalis
Tamandua tetradactyla
Tamandua tetradactyla
Puma concolor
Priodontes maximus
Callimico goeldii
Ateles belzebuth
Cebuella pygmaea
0
1
2
3
LEK- species population abundance (Likert scale)

548

Figure 3. Distribution of species abundance according to LEK-based methods. Abundance indices were grouped into four categories (x-axis): when the species was perceived as “absent”; “low abundance”; “medium abundance”; and “high abundance” by each interviewee. Each coloured box represents a species and box sizes represent the number of interviewees reporting a certain abundance category for each species. Y-axis represents the frequency with which each species was recorded in each abundance category. Species can occur in more than one category because the graph is based on the number of interviewees indicating a certain abundance category for each species. We included a silhouette in the box of species with a higher number of reports in each category.

The most detected taxa on line transects were Primates, followed by gamebirds and Carnivora, which corresponded to 85%, 5% and 5% in the total sum of all detected individuals (N= 22,908), respectively. Tortoises and Didelphimorphia species were never recorded using line transects. In agreement with line transects, the most abundant taxa recorded through LEK-based method were Primates, gamebirds and Carnivora which corresponded to 30%, 17% and 15% in the total sum of all abundance data estimated by all interviewees (N= 20,282), respectively. The abundance of tortoises and Didelphimorphia species within the total of LEK-based abundance indices were 2% and 0.4%, respectively (Fig. 4).


A) Contribution of each taxa to total records through linear transects
B) Contribution of each taxa to total estimates through LEK- based methods
Didelphimorphia
Testudine
Perissodactyla
Cingulata
Pilosa
Artiodactyla
Rodentia
Carnivora
Bird
Primates
0%
10%
20%
30%
40%
50%
60%
70%
80%
90%
100%

566

Figure 4. Representation of each taxa in the total sum of (A) all individuals detected on linear transect, and (B) the abundances estimated by all interviewees through LEK-based method. Both methods include the same number of species per taxa. The percentages for each method were calculated by summing all the abundance indices of all species for each taxon, thereby deriving the percentage of that summed value for the total abundance indices.

Correlates of abundance estimated by line transects and the LEK-based method

When each method was analysed separately, the most abundant species on line transect were those that live in larger groups and with diurnal habit. In addition, the abundance of populations in flooded forests were higher than in upland forests using line transects (Fig 5A; Supplementary Material, Table 4). For the LEK-based method, species that live in larger groups were also estimated to have a higher abundance; all other variables did not have a significant effect on the LEK-based abundance (Fig 5B; Supplementary Material, Table 5).

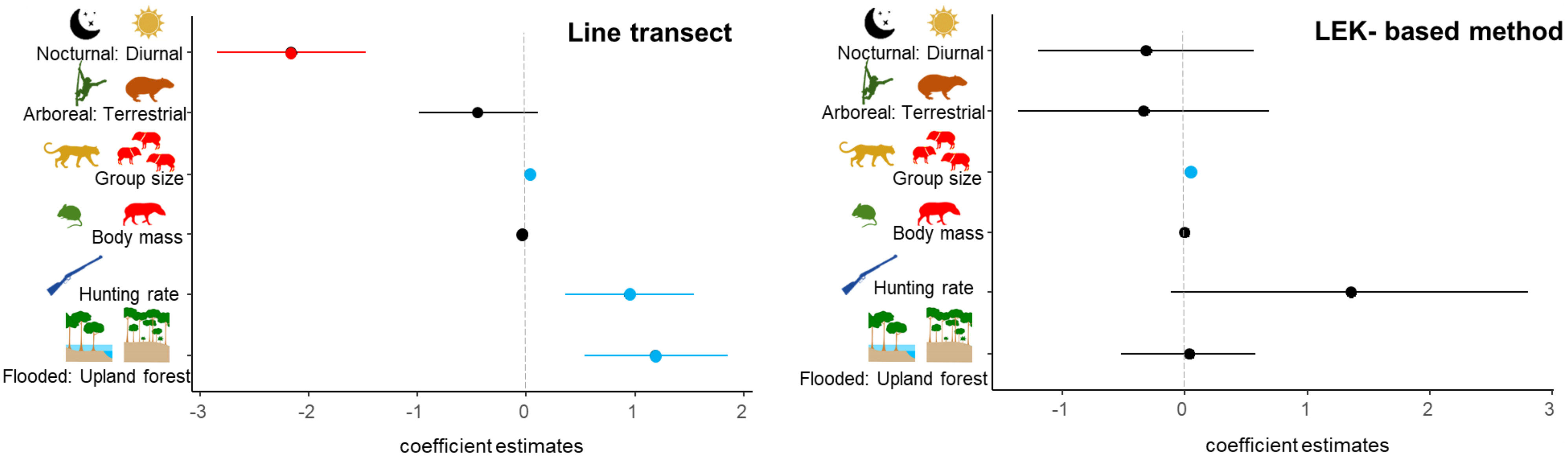


Figure 5. Linear coefficient estimates (± 95% confidence intervals) showing the magnitude and direction of biological and socioecological effects on the abundance indices obtained through the line transect (A) and LEK-based method (B), when analysed separately. Blue and red solid dots represent either significantly positive or significantly negative effects, respectively; and black solid dots represent non-significant effects. Silhouette credits: Franciany Braga-Pereira. Flooded and Upland forest illustration credits: Andrew Abraham.

Comparisons of abundance estimates using line transect and the LEK-based method

For 30 of all 54 ecospecies, we found a significantly positive correlation (≥ 0.113) between abundance indices based on the LEK-based method and direct field surveys, with only two species showing a significantly negative correlation. All group-living species with group sizes ≥ eight individuals showed a positive correlation (Fig. 6).

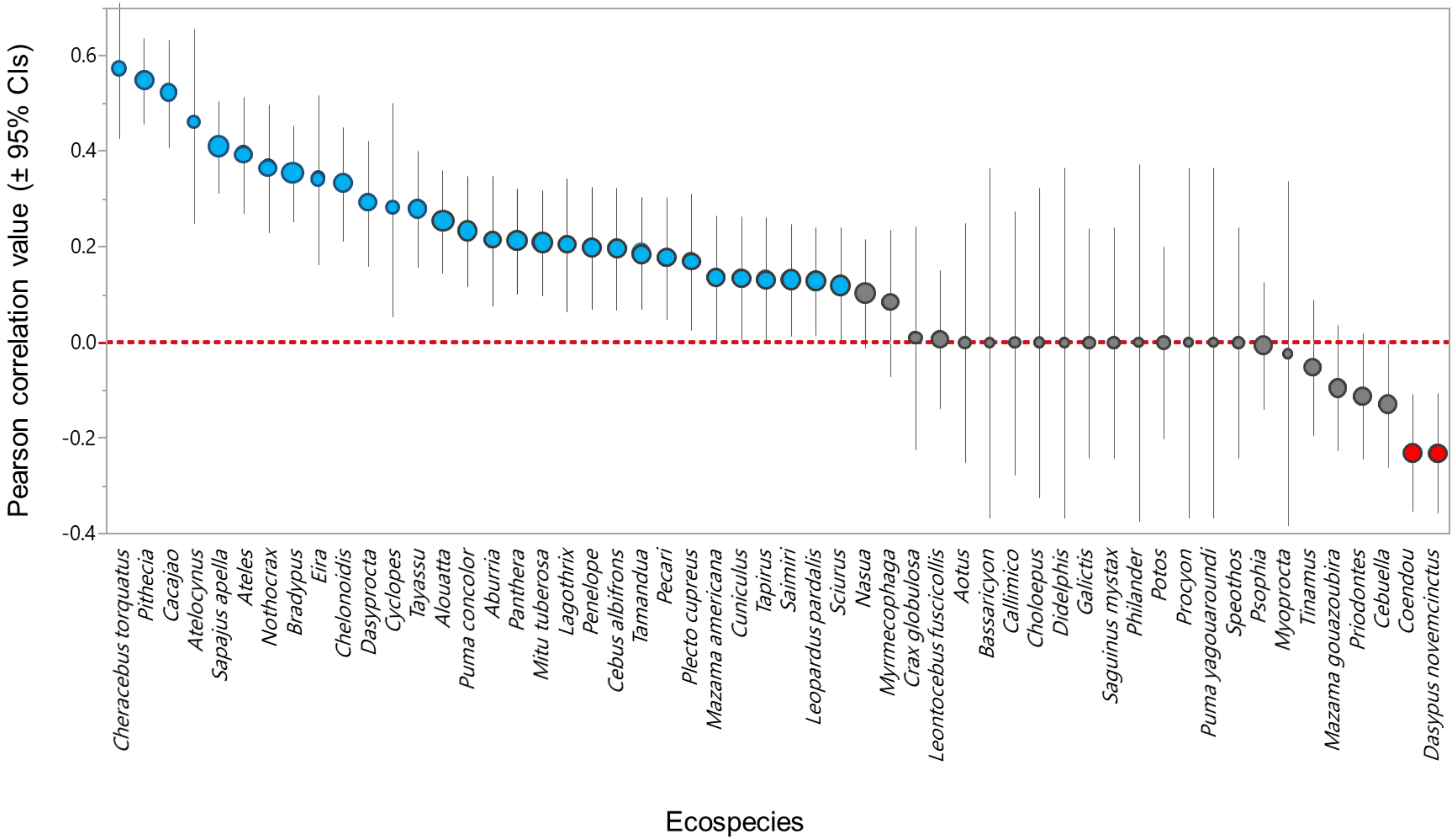

Pearson correlation value (± 95% CIs)
0.6
0.4
0.2
0.0
-0.2
-0.4
Cheracebus torquatus
Pithecia
Cacajao
Atelocynus
Sapajus apella
Ateles
Nothocrax
Bradypus
Eira
Chelonoidis
Dasyprocta
Cyclopes
Tayassu
Alouatta
Puma concolor
Aburria
Panthera
Mitu tuberosa
Lagothrix
Penelope
Cebus albifrons
Tamandua
Pecari
Plecto cupreus
Mazama americana
Cuniculus
Tapirus
Saimiri
Leopardus pardalis
Sciurus
Nasua
Myrmecophaga
Crax globulosa
Leontocebus fuscicollis
Aotus
Bassaricyon
Callimico
Choloepus
Didelphis
Galictis
Saguinus mystax
Philander
Potos
Procyon
Puma yagouaroundi
Speothos
Psophia
Myoprocta
Tinamus
Mazama gouazoubira
Priodontes
Cebuella
Coendou
Dasypus novemcinctus
Ecospecies

Figure 6. Ecospecies-specific abundance correlation between the LEK-based method and line transect. Considering that our sampling unit (the number of interviewees) is ~300, correlations > 0.113 (for $p<0.05$) was considered highly significant. Circle sizes are proportional to counts (number of interviews). Blob colours denote correlation level, in blue positive correlation, grey without correlation and red negative correlation. Here we pooled all taxonomic species into ecological analogues or functional groups (ecospecies).

Considering all species (the entire dataset in our initial model), we found a consistent and significant relationship between the abundance indices obtained through transects and the LEK-based method ($p<0.001$; Supplement material, Table 3). However, when we stratified the dataset, our models revealed that this relationship is dependent on biological and socioecological factors. We found a consistent and significant relationship between the abundance indices for species that are diurnal (Fig 7A; Supplementary material, Fig 3) and hunted at an intermediate level (Fig 7E); and independently of sociality (7B), body size (Fig 7C), locomotion mode (Fig 7D), and forest type (Fig 7F). On the other hand, we did not find a significant relationship for species that are nocturnal (Fig 7A; Supplementary material, Fig 3) and non-hunted or with a high level of hunting (Fig 7D; Supplementary material, Table 3).

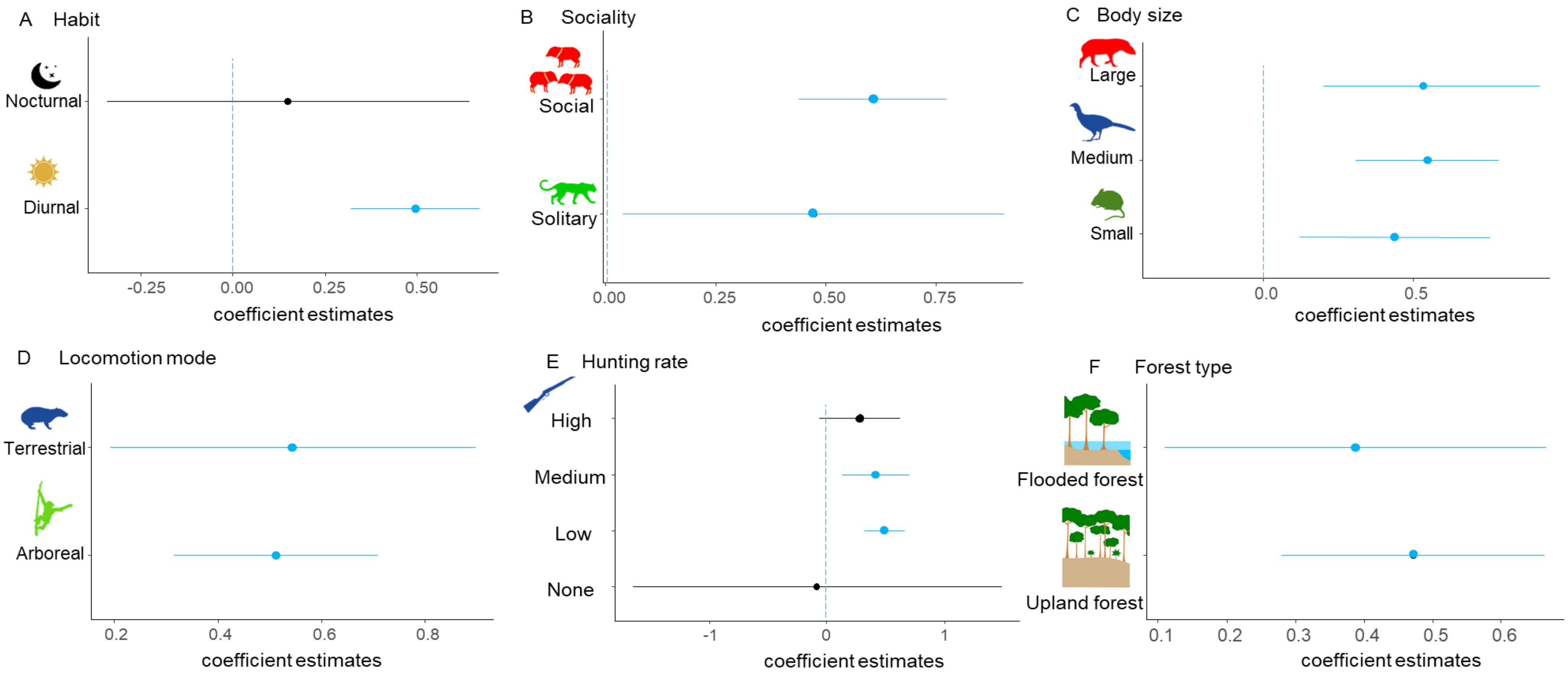


Figure 7. Linear coefficient estimates (± 95% confidence intervals) showing the magnitude and direction of effects on the relationship between the abundance indices obtained through the line transect and the LEK-based method for species regarding its habit (A); sociality (B); body mass (small- to 1 kg; medium- from 1 kg to 5 kg; large- exceeding 5 kg) (C); locomotion mode (D);

hunting rate (no hunting (0), low hunting (until 0.05), medium hunting (between 0.05 and 0.35) and high hunting (until 1) (E); and forest type (F). Blue and black solid dots represent significantly positive and non-significant effects, respectively. Silhouette credits: Franciany Braga-Pereira. Flooded and Upland forest illustration credits: Andrew Abraham.

Discussion

Population abundance assessments are essential to estimate the population status of wild species as well as facilitate decision-making regarding their conservation and management. The effectiveness of management decisions is dependent on the accuracy and timeliness of abundance estimates, meaning that improvements to data collection may herald improved management actions (Hodgson, 2018). In this study, we compared two methods of abundance data acquisition, which arise from two distinct systems of knowledge. Line transect surveys are based on theoretical scientific knowledge, characterised by being systematic, controlled and based on hypotheses; which provides objectivity, verifiability, and, when properly applied, precision and accuracy (Rodríguez and Pérez, 2017). On the other hand, LEK arises from day-to-day practices and empirical knowledge embedded within specific worldviews beyond the nature/culture divide (Congretel and Pinton, 2020; Rodríguez and Pérez, 2017). LEK has direct practical applications and is considered more inductive and tacit (Congretel and Pinton, 2020).

In this study, abundance indices obtained from line transects and a LEK-based method are comparable for species that are diurnal, and independently of the species locomotion mode, sociality, body mass and forest type. The fact that both methods were congruent in terms of abundance estimates shows that conventional survey techniques based on direct sampling of populations can be substituted or, in some circumstances, be improved by LEK-based methods. On the other hand, we found that line transect may underreport, and even fail to report species with specific traits (such as nocturnal or rare species) as, according to previous studies, all species considered in this research potentially occur in the study areas. While LEK widely recorded most species that occurred in the area. In accordance with our results, wildlife abundances estimated by shepherds in Southeastern Spain were similar to those from line-transect surveys, but shepherds’ ecological knowledge yielded abundance estimates across a broader range of species than linear transects, which only detected the species in the upper abundance range (Anadón et al., 2009). Records and memory recalls of neotropical vertebrate species occupancy by long-term residents at dozens to hundreds of forest fragments are also far more complete than those derived from short-term surveys (Peres and Michalski 2006; Canale et al., 2012). Also, the ability to identify the occurrence and variations in populations of some species through the LEK has been found to be more accurate for ungulates compared to line transects and camera trapping (Camino et al., 2020) and for Tayassuidae when comparing LEK to line transect (Pérez-Peña et al., 2017). Overreporting through LEK could be expected for species involved in psycho-attitudes of human-wildlife conflicts, such large felids (Treves and Karanth, 2003). Because of these conflicts, the perceived abundance by locals could be magnified. However, the abundance of large felids was perceived as low or intermediate by most interviewees.

In some cases, the argument for low detection of species through diurnal line-transect surveys may be the reduced effort. However, we highlight that the effort applied during the line transect in our study – including a total of 9,221 km walked – far exceeds the average effort often applied in Neotropical forests, which usually ranges from 40 to 600 km (de Thoisy 2008). Our low record of many species was therefore not a result of limited effort, and we claim that line transects could be an inappropriate method to survey several but not all species. For example, ecospecies yielding non-significant correlation values between the two types of abundance estimates are primarily those that are rarely detected along diurnal surveys on foot. For instance, porcupines were never recorded along transects, and *Dasypus novemcinctus* either failed to be recorded or its survey-based abundance was very low (0.002 ind. per km). Conversely, both small- and large-group-living Pitheciinae primates (*Pithecia* and *Cacajao*) showed highly positive correlation values because in villages where these taxa were not detected, interviewees also perceived them as absent, whereas in villages where they were frequently detected along transects, all informants indicated intermediate to high abundance. Some ecospecies failed to yield significantly positive correlations, but this does not necessarily mean that either one of the two methods is inefficient. For example, large tinamids (*Tinamus*) are frequently recorded along transects but are subject to high variance in detection rates, whereas interviewees consistently reported high abundance values.

In general, line-transect surveys cover less than 0.5% of a given study area, which is often too scarce to reliably estimate species' abundance (Matthews and Matthews, 2002; Van der Hoeven et al., 2004). On the other hand, LEK is arguably a compelling method because the observer performance and overall survey effort of hunters surely exceed those of conventional biodiversity surveys. In addition, the effort of LEK is multi-scale, given that local forest observers are generally present at all times of the day and year around, accounting for different circadian rhythms and seasons, and in multiple areas when conducting their habitual activities, such as hunting, fishing, farming and harvest of timber and non-timber products. Even the same specific activity can include diverse practices. For example, local people use many techniques to hunt nocturnal or diurnal species, such as waiting on trees, traps and baits, sweeping the forest floor or spotlighting along the riverbanks from a canoe across different landscape types (Vieira et al., 2015; Tavares et al., 2020). The repetition of such practices results in a systemic knowledge of their surroundings, including natural environments and the perception of wildlife population changes, which are ultimately reflected on species abundance estimates over different time scales (Braga-Pereira et al., 2020).

We did not find agreement in abundance indices of non-hunted or heavily hunted species comparing the two methods in the same model. The non-agreement about non-hunted species is possibly biased by the non-detection of 83% of these species during transect surveys, while these same species were mentioned as present in 84% of LEK interviews. This lower detectability of non-hunted species, which are generally rare (Bodmer, 1995), during line transect surveys, reinforces the inappropriate use of this method to detect rare species. Regarding hunted species, they can be elusive and therefore less detected during the transect sampling through direct sighting. However, hunters holistically consider other signs left by animals (such as footprints, scratches, urine trails, feces, odors, animal vocalizations and other specific noises) for estimating its abundance. Fragoso et al., (2016) explains that as much as these signs could also be identified during line transects, in most of the studies the record usually just occurs when the individual animal is visualized), which reduces the number of individuals registred during transect sampling.

Upland forests are more species-rich, including more forest habitat specialists than flooded forests, while the average population biomass density is higher in seasonally-flooded forests along white-water rivers (Peres, 1997; Haugaasen and Peres, 2005). Although we expected greater abundance in flooded forests in relation to upland, we did not detect differences on abundance indices related to forest type through LEK when the two methods were analysed separately. We believe that estimates using the nominal ordinal scale of abundance was probably inappropriate to capture difference in wildlife abundance between forest types. This is because nominal ordinal classification is subject to how each nominal level (low, medium and high) is perceived by each person and, therefore, reduces the efficiency of comparisons among different sites. Therefore, the use of LEK-based methods when considering nominal scales, as used in this study, could be used to provide reliable comparisons over time within a site, in a way that abundance trends can be detected, but it is less reliable to make comparisons among people living in different sites, as local people use different levels of reference based on local natural abundance to give their responses. Although this index may be less reliable to make comparisons over long periods of time, as more recent situations may become the new baseline for people's perceptions on animal abundance, LEK not only takes information from one's own experience but also from other individuals (e.g. their ancestors) in their environment over time (Mazzocchi 2006). For this reason, it is possible to ask about animal abundance from long ago or if population abundances have changed over time (Braga-Pereira et al., 2020; Van Holt et al., 2007).

To improve the accuracy of LEK-based methods we recommend the adoption of quantitative methods during interviews for the estimation of wildlife abundance, in which participants would estimate the number of individuals occurring in a certain area. Using quantitative visual scales (Braga-Pereira et al.,

2020) or physical units (i.e. seeds) (Chaves et al., 2020) that allow the informant first-hand indicate the number of specimens he/she perceived in a certain area would therefore be more efficient to detect differences between environments and across long periods of time (see Supplementary material, Fig 4). Quantitative visual scales may be more useful when interviews about animal relative abundance are targeted at a larger number of species, thus optimizing interview time. On the other hand, estimates of numbers of individuals could be used to estimate population density, especially for those species for which interviewees are most effective in measuring their numbers within a given area, as in the case of game species (Van der Hoeven et al., 2004). For species that are not of local people's interest, line transects may provide more accurate population density estimates, because survey effort will be directed to a particular species. Another advantage of line transects is that they can provide accurate information to compare population densities among sites and over time, as they are performed in a systematic way that makes them comparable. However, we advocate that even when using linear transects, LEK-based methods should be used to calibrate and ensure that the non-detection of a given species is not a result of underreporting.

An efficient way to refine population studies would be to first conduct interviews at an early stage of monitoring to obtain a preliminary overview of the area, and improve study design on line-transect surveys. Secondly, studies could involve local people in monitoring line-transects so they can help inform on species that remain undetected during sampling, but may be observed elsewhere. In addition, the perception of local monitors is multisensory, involving hearing, smell and indirect visual signals, such as tracks and scratches, which increase detection probability along transects. Moreover, community-based wildlife monitoring (where locals record and interpret their own data) can provide more than a science contribution, for example in contributing to long-term sustainability by empowering local stakeholders to better manage their own natural resources (Danielsen et al., 2009, Luzar et al., 2011; Constantino et al., 2012), build local capacity and develop legitimate and successful conservation initiatives (Fragoso et al., 2016).

LEK plus training in community-based wildlife monitoring can be an empowering method that can be performed and continued regardless, for example, of international or national crises, such as the COVID-19 pandemic, during which many protected areas remain closed to external researchers. Furthermore, participatory approaches have proven to provide cost-effective monitoring of the distribution and abundance over large spatio-temporal scales even for rare, nocturnal and cryptic species (Silvertown 2009; Damme et al., 2015; Farhadinia et al., 2018). In our study, travel expenses to transect sites from the field stations including food supplies, and considering only the cost of a technician (US$50/day) and a local assistant (US$20/day) (Gardner et al., 2008) to survey a typical transect each day, we estimate that around US$161,368 would be spent to conduct all linear transect surveys in this study. In comparison, considering two technician interviewers (US$50/day) for each of the 17 villages sampled, we estimate that the LEK-based method would cost US$1,700 to obtain comparatively reliable abundance indices for most of species.

Using a large dataset collected at a large spatial scale from different regions we compared vertebrate abundance estimates obtained from two sampling methods for a range of species and environments, and more importantly assessed the effect of several biological factors, hunting level and landscape type on the congruence and divergence between these methods. Given that interviews with local experts optimise sampling effort and reduce monetary costs, this method may overcome the lack of resource for continued and large-scale reassessments, another major constraint in environmental research and conservation projects. We strongly recommend inclusion of LEK-based methods to manage and monitor wildlife populations. As local people have accumulated a profound body of knowledge of Amazonian wildlife, it is urgent that local and scientific knowledge-based methods be combined and shared reciprocally. This combination not only benefits the scale (Gagnon and Berteaux, 2009) and budget of the monitoring (or research) (Silvertown 2009; Damme et al., 2015; Farhadinia et al., 2018), but also promotes the collaboration between local people and external researchers in wildlife management initiatives (Constantino et al., 2012). We claim that this is a leading alternative to develop effective strategies in social-ecological systems to meet biodiversity monitoring and conservation goals.

DATA AVAILABILITY

Data deposited in the Dryad repository:
http://datadryad.org/resource/doi:10.5061/dryad.905qfttms

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Supplementary material

Table 1- Information on areas studied in the Brazilian and Peruvian Amazon, and the period and effort for execution of each method, line transects and LEK.

Country	Area	Village	Forest type	Esffort (km)	Number of trails	Number of total interviews conducted	Ethnicity
Brazil	Mamirauá SDR	Vila Alencar	*Flooded forest*	77	3	15	non- indigenous area
Brazil	Mamirauá SDR	Aiucá	*Flooded forest*	62	3	24	non- indigenous area
Brazil	Mamirauá SDR	Jarauá	*Flooded forest*	42	3	22	non- indigenous area
Brazil	Amanã SDR	Boa Esperança	*Upland forest*	286	4	19	non- indigenous area
Brazil	Amanã SDR	Ubim	*Upland forest*	167	2	7	non- indigenous area
Brazil	Amanã SDR	Bom Jesus do Baré	*Upland forest*	89	1	11	non- indigenous area
Brazil	Piagaçu Purus SDR	Livramento	*Upland forest*	856	8	17	non- indigenous area
Brazil	Piagaçu Purus SDR	São João do Uauacu	*Upland forest*	656	5	11	non- indigenous area
Brazil	Piagaçu Purus SDR	Caua-Cuiana	*Flooded forest*	230	2	10	non- indigenous area
Peru	Yavarí-Mirín River	Lago Preto	*Flooded forest*	466		19	indigenous area
Peru	Yavarí-Mirín River	Esperanza	*Upland forest*	563		19	indigenous area
Peru	Putumayo River	Mashunta	*Upland forest*	230		15	indigenous area

Peru	Putumayo River	Santa Rita	*Upland forest*	208		8	indigenous area
Peru	Putumayo River	Nueva Jerusalém	*Flooded forest*	119		6	indigenous area
Peru	Matsés NR	Alemán	*Upland forest*	238		18	indigenous area
Peru	Matsés NR	Gálvez	*Upland forest*	229		15	indigenous area
Peru	Pucacuro NR	28 de Julio	*Upland forest*	811		33	indigenous area
Peru	Maijuna-Kichwa RCA	Sucusari	*Upland forest*	2126		22	indigenous area
	TOTAL			7454	31	291	

Table 2- List of species considered in this study with their respectively biological and ecological information. The scientific name reported by the cited authors is considered in this table and in some times, this is not the same of the current name.

Order	Species	Habitat	Habit	Body mass (Kg)	Social behaviour	Hunting rate	Reference
Ave	Aburria_cumanensis	Arboreal	Diurnal	1.40	3.56	0.16	Wallace_et_al_2001
Primates	Alouatta_juara	Arboreal	Diurnal	6.00	5.54	0.34	Soini_1992_for_A_seniculus
Primates	Alouatta_puruensis	Arboreal	Diurnal	6.00	5.54	0.34	Soini_1992_for_A_seniculus
Primates	Alouatta_seniculus	Arboreal	Diurnal	6.00	5.54	0.34	Soini_1992_for_A_seniculus
Primates	Aotus_sp	Arboreal	Diurnal	0.87	3.30	0.04	Aquino_and_Encarnacion_1990_for_A_vociferans
Primates	Aotus_vociferans	Arboreal	Diurnal	0.87	3.30	0.04	Aquino_and_Encarnacion_1990_for_A_vociferans
Primates	Ateles_belzebuth	Arboreal	Diurnal	9.00	3.34	0.25	Iwanaga_and_Ferrari_2002_for_A_chamek_subgroups
Primates	Ateles_chamek	Arboreal	Diurnal	9.00	3.34	0.25	Iwanaga_and_Ferrari_2002_for_A_chamek_subgroups
Carnivora	Atelocynus_microtis	Terretrial	Noturnal	9.50	1.00	0	Peres_1991

Carnivora	Bassaricyon_gabbii	Arboreal	Noturnal	1.00	1.90	0	Pontes_and_Chivers_2002
Pilosa	Bradipus_o_Choloepus	Arboreal	Noturnal	4.15	1.00	0	cf_REF
Pilosa	Bradypus_tridactylus	Arboreal	Diurnal	2.40	1.00	0.06	Goffart_1971
Pilosa	Bradypus_variegatus	Arboreal	Diurnal	3.65	1.00	0.06	Goffart_1971
Primates	Cacajao_calvus	Arboreal	Diurnal	3.16	43.50	0.01	Bowler_and_Bodmer_2009_for_C_c _ucayalii
Primates	Cacajao_melanocephalus	Arboreal	Diurnal	3.16	100.00	0.01	Barnett_et_al._2005
Primates	Callicebus_cupreus	Arboreal	Diurnal	1.17	4.10	0.04	Defler_et_al_2010_for_P_caquetensis
Primates	Callicebus_discolor	Arboreal	Diurnal	1.17	4.10	0.04	Defler_et_al_2010_for_P_caquetensis
Primates	Callicebus_lucifer	Arboreal	Diurnal	1.41	4.39	0.04	Kinzey_et_al_1977
Primates	Callicebus_sp.	Arboreal	Diurnal	1.17	4.10	0.04	Defler_et_al_2010_for_P_caquetensis
Primates	Callicebus_torquatus	Arboreal	Diurnal	1.30	4.26	0.04	Defler_et_al_2010_for_P_caquetensis
Primates	Callimico_goeldii	Arboreal	Diurnal	0.36	4.00	0	Porter_et_al_2007

Primates	Cebuella_pygmaea	Arboreal	Diurnal	0.12	5.33	0	Soini_1982_mean_for_last_acount_of_each_group
Primates	Cebus_albifrons	Arboreal	Diurnal	2.00	35.00	1.07	Defler_1982_for_C_albifrons
Primates	Cebus_apella	Arboreal	Diurnal	3.50	7.00	1.07	Peres_1988
Primates	Cebus_unicolor	Arboreal	Diurnal	2.00	35.00	1.07	Defler_1982_for_C_albifrons
Primates	Cebus_yuracus	Arboreal	Diurnal	2.00	35.00	1.07	Defler_1982_for_C_albifrons
Testudine	Chelonoidis_denticulata	Terretrial	Diurnal	8.00	1.00	0.46	Wilkinson_et_al_2010
Pilosa	Choleopus_didactylus	Arboreal	Diurnal	3.37	1.00	0.136756757	cf_REF
Primates	Ciclopes_didactylus	Arboreal	Noturnal	0.40	1.00	0	cf_REF
Rodentia	Coendou_bicolor	Arboreal	Noturnal	5.00	1.00	0.03	Roberts_et_al_1985_for_C_prehensilis
Rodentia	Coendou_prehensilis	Arboreal	Noturnal	3.36	1.00	0.03	cf_REF
Ave	Crax_globulosa	Arboreal	Diurnal	3.00	1.69	0.3	Haugaasen_and_Peres_2008
Rodentia	Cuniculus_paca	Terretrial	Noturnal	8.00	1.00	0.8	Eisenberg_1989
Rodentia	Dasyprocta_fuliginosa	Terretrial	Diurnal	5.00	1.00	0.66	cf_REF

Rodentia	Dasyprocta_sp.	Terretrial	Diurnal	5.00	1.00	0.66	cf_REF
Cingulata	Dasypus_novemcinctus	Terretrial	Noturnal	6.00	1.00	0.67	cf_REF
Cingulata	Dasypus_sp	Terretrial	Noturnal	3.98	1.00	0.009	cf_REF
Didelphimorphia	Didelphis_marsupialis	Arboreal	Noturnal	1.04	1.00	0	cf_REF
Carnivora	Eira_barbara	Terretrial	Noturnal	3.98	4.00	0.009	Gittleman_1989
Carnivora	Galictis_vittata	Terretrial	Noturnal	2.91	2.00	0	Gittleman_1989
Primates	Lagothrix_cana	Arboreal	Diurnal	11.00	46.50	0.56	Peres_1996
Primates	Lagothrix_lagotricha	Arboreal	Diurnal	11.00	46.50	0.56	Peres_1996
Primates	Lagothrix_poeppigii	Arboreal	Diurnal	11.00	46.50	0.56	Peres_1996
Primates	Leontocebus_nigricollis	Arboreal	Diurnal	0.35	5.13	0.01	Goldizen_et_al_1996
Primates	Leontocebus_sp	Arboreal	Diurnal	0.35	5.13	0.01	Goldizen_et_al_1996
Carnivora	Leopardus_pardalis	Terretrial	Noturnal	10.46	1.00	0.02	cf_REF
Artiodactyla	Mazama_americana	Terretrial	Noturnal	20.00	1.00	0.31	cf_REF

Artiodactyla	Mazama_gouasoubira	Terretrial	Noturnal	17.00	1.00	0.31	cf_REF
Artiodactyla	Mazama_nemorivaga	Terretrial	Noturnal	17.00	1.00	0.31	cf_REF
Ave	Mitu_salvini	Arboreal	Diurnal	3.00	1.36	0.3	Haugaasen_and_Peres_2008
Ave	Mitu_tuberosa	Arboreal	Diurnal	3.00	1.36	0.3	Haugaasen_and_Peres_2008
Rodentia	Myoprocta_pratti	Terretrial	Diurnal	5.00	1.00	0.04	NA
Pilosa	Myrmecophaga_tridactyla	Terretrial	Diurnal	27.00	1.00	0.01	cf_REF
Carnivora	Nasua_nasua	Arboreal	Diurnal	5.00	7.40	0.37	Beisiegel_2001
Ave	Nothocrax_urumutum	Arboreal	Diurnal	1.20	1.00	0.009	NA
Carnivora	Panthera_onca	Terretrial	Noturnal	68.75	1.00	0	cf_REF
Artiodactyla	Pecari_tajacu	Terretrial	Diurnal	25.00	26.00	0.61	Mendes_Pontes_2004
Ave	Penelope_jacquacu	Arboreal	Diurnal	2.00	1.75	0.54	Haugaasen_and_Peres_2008
Didelphimorphia	Philander_andersoni	Arboreal	Noturnal	0.40	1.00	0	cf_REF
Ave	Pipile_cumanensis	Arboreal	Diurnal	1.40	3.56	0.16	Wallace_et_al_2001
Primates	Pithecia_aequatorialis	Arboreal	Diurnal	2.35	4.00	0.046	Defler_2003_for_P_monachus
Primates	Pithecia_albicans	Arboreal	Diurnal	2.35	4.00	0.046	Defler_2003_for_P_monachus

Primates	Pithecia_hirsuta	Arboreal	Diurnal	2.35	4.00	0.046	Defler_2003_for_P_monachus
Primates	Pithecia_milleri	Arboreal	Diurnal	2.35	4.00	0.046	Defler_2003_for_P_monachus
Primates	Pithecia_monachus	Arboreal	Diurnal	2.35	4.00	0.046	Defler_2003_for_P_monachus
Primates	Plecturocebus_cupreus	Arboreal	Diurnal	1.17	4.10	0.04	Defler_et_al_2010_for_P_caquetensis
Carnivora	Potos_flavus	Arboreal	Noturnal	2.49	1.00	0.01	cf_REF
Cingulata	Priodontes_maximus	Terretrial	Diurnal	39.40	1.00	0.004	cf_REF
Carnivora	Procyon_concrivorus	Terretrial	Noturnal	10.10	1.00	0	cf_REF
Ave	Psophia_crepitans	Terretrial	Diurnal	1.30	4.42	0.23	Haugaasen_and_Peres_2008
Ave	Psophia_leucoptera	Terretrial	Diurnal	1.30	4.42	0.23	Haugaasen_and_Peres_2008
Carnivora	Puma_concolor	Terretrial	Noturnal	37.00	1.00	0	cf_REF
Carnivora	Puma_yagouaroundi	Terretrial	Noturnal	5.00	1.00	0	cf_REF
Primates	Saguinus_fuscicollis	Arboreal	Diurnal	0.52	4.63	0.01	Moya_et_al_1990
Primates	Saguinus_inustus	Arboreal	Diurnal	0.52	4.63	0.01	Moya_et_al_1990
Primates	Saguinus_mistax	Arboreal	Diurnal	0.52	4.63	0.01	Moya_et_al_1990
Primates	Saguinus_sp.	Arboreal	Diurnal	0.52	4.63	0.01	Moya_et_al_1990

Primates	Saimiri_boliviensis	Arboreal	Diurnal	0.68	31.00	0.008	Peres_1988_for_S_macrodon
Primates	Saimiri_macrodon	Arboreal	Diurnal	0.68	31.00	0.008	Peres_1988_for_S_macrodon
Primates	Saimiri_sciureus	Arboreal	Diurnal	0.68	31.00	0.008	Peres_1988_for_S_macrodon
Primates	Saimiri_sp.	Arboreal	Diurnal	0.68	27.57	0.008	Paim_2008
Primates	Sapajus_macrocephalus	Arboreal	Diurnal	3.50	7.00	1.07	Peres_1988
Rodentia	Sciurus_igniventris	Arboreal	Diurnal	0.34	1.00	0.14	cf_REF
Rodentia	Sciurus_sp	Arboreal	Diurnal	0.34	1.00	0.14	cf_REF
Ave	Sophia_leucoptera	Terretrial	Diurnal	1.30	4.42	0.23	Haugaasen_and_Peres_2008
Carnivora	Speothos_venaticus	Terretrial	Diurnal	5.50	3.00	0	Oliveira_et_al_2018
Pilosa	Tamandua_tetradactyla	Arboreal	Noturnal	4.56	1.00	0.09	cf_REF
Perissodactyla	Tapirus_terrestris	Terretrial	Noturnal	140.00	1.00	0.05	cf_REF
Artiodactyla	Tayassu_pecari	Terretrial	Diurnal	35.00	93.50	0.67	Fragoso_1998_mean_of_two_groups
Ave	Tinamus_major	Terretrial	Diurnal	1.17	1.04	0.25	Haugaasen_and_Peres_2008
Ave	Tinamus_sp	Terretrial	Diurnal	1.17	1.04	0.25	Haugaasen_and_Peres_2008

Table 3. Details of the all models elaborated for verify the influence of Habitat, Habit, Hunting rate, Group size, Body mass and Landscape on the relationship between the abundance values via line transects census and LEK

Predictor variables	Estimate	Std. Error	z value	Pr(>|z|)		AIC	AIC Null model	ΔAIC
Habitat terretrial	0.534	0.179	2.984	0.0029	**	1021.2	1028.8	7.6
Habitat arboreal	0.510	0.100	5.116	3.13E-07	***	3919.6	3945.7	26.1
Habit diurnal	0.493	0.089	5.518	3.44E-08	***	4635.8	4666.5	30.7
Habit noturnal	0.151	0.250	1	0.5440		860.1	858.4	-1.7
Hunting rate- High	0.268	0.174	1.545	0.1220		1705.6	1706.9	1.3
Hunting rate- Medium	0.407	0.144	2.835	0.0046	**	1398.3	1404.8	6.5
Hunting rate- Low	0.481	0.088	5.502	3.75E-08	***	4976.4	5007.1	30.7
Hunting rate- None	0.006	0.740	0	0.9929		40.9	38.9	-2
Body mass- Large	0.522	0.184	2.842	0.0045	**	1326.1	1333.2	7.1
Body mass- Medium	0.541	0.121	4.479	7.49E-06	***	2286.7	2306.2	19.5
Body mass- Small	0.439	0.001	329	<2e-16	***	1337.1	1343.4	6.3
Solitary	0.3824	0.2191	1.745	0.0809	.	524.8	525.4	0.6
Group size- Small	0.458	0.091	5.004	5.62E-07	***	4304.6	4329.5	24.9
Upland forest	0.469	0.098	4.792	1.65E-06	***	3948.5	3971.4	22.9
Flooded forest	0.385	0.170	2.261	0.0237	*	1650.1	1656.3	6.2

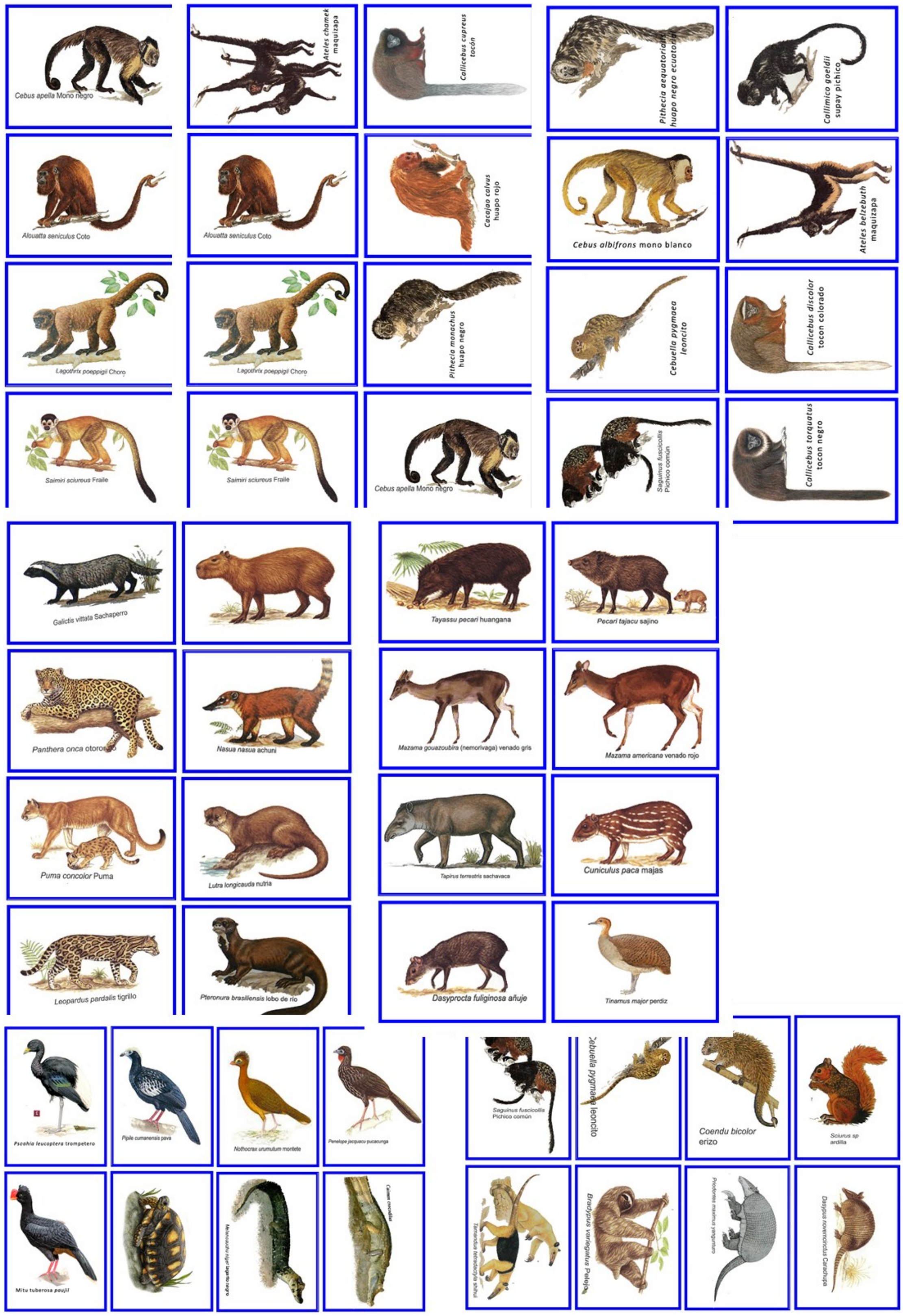
All species	0.482	0.088	5.502	3.76E-08	***	4976.4	5007.1	30.7

Table 4. Details of the complete model and the null model using *generalized linear mixed model* for verify the influence of Habitat (arboreal:terrestrial), Habit (nocturnal:diurnal), Hunting rate, Group size, Body mass and Forest type (Flooded:Upland) on the abundance via Line transect

Predictor variables	Estimate	Std. Error	z value	Pr(>|z|)		AIC	AIC Null model	ΔAIC
Arboreal:Terrestrial	0.480694	0.2816	1.707	0.08784		4800.5	4983.2	182.7
Nocturnal:Diurnal	-2.250.719	0.358	-6.286	3.25E-10	***			
Hunting rate	1.013.868	0.3083	3.289	0.00101	**			
Body mass	-0.02842	0.0138	-2.056	0.03976	*			
Group size	0.032242	0.005	6.472	9.68E-11	***			
Flooded:Upland forest	1.192.865	0.3346	3.565	0.00036	**			

Table 5. Details of the complete model and the null model using *Cumulative* Link Mixed *Models* for verify the influence of Habitat (arboreal:terrestrial), Habit (nocturnal:diurnal), Hunting rate, Group size, Body mass and Forest type (Flooded:Upland) on the abundance perceived via LEK

Predictor variables	Estimate	Std. Error	z value	Pr(>|z|)		AIC	AIC Null model	ΔAIC
Arboreal:Terrestrial	-0.33662	0.5224	-0.644	0.5193		17311	17318	7.66
Nocturnal:Diurnal	-0.31971	0.4495	-0.711	0.4769				
Hunting rate	1.347.180	0.7425	1.814	0.0696	.			
Body mass	-0.00491	0.0131	-0.376	0.707				
Group size	0.039867	0.0129	3.088	0.002	**			
Flooded forest:Upland forest	0.027066	0.2775	0.098	0.9223				



1070 Figure 1. illustrated checklist used during the structured interviews which provided visual stimulation with drawings of species presumably present in each study areas.

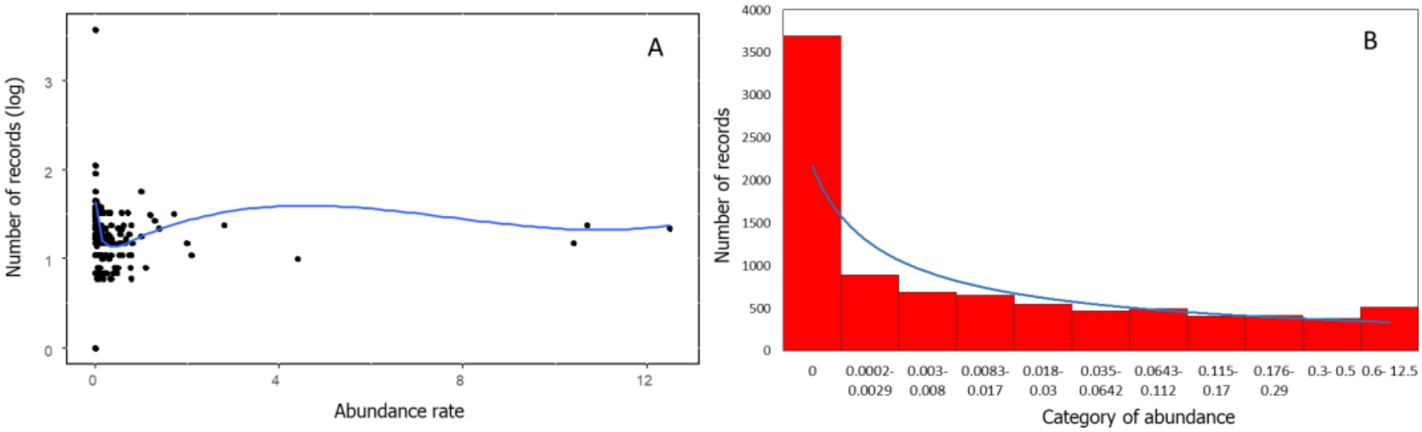


Figure 2. Number of times (in log) each abundance rate (A) or each category of abundance (B) was recorded during linear transect. Most records are for species in the lower abundance rate/categories. Figure A was elaborate using each abundance rate registered in this study. In figure B the abundance values were grouped into 12 categories, similar to figure 2 of the main manuscript (in which the values were grouped into 4 categories).

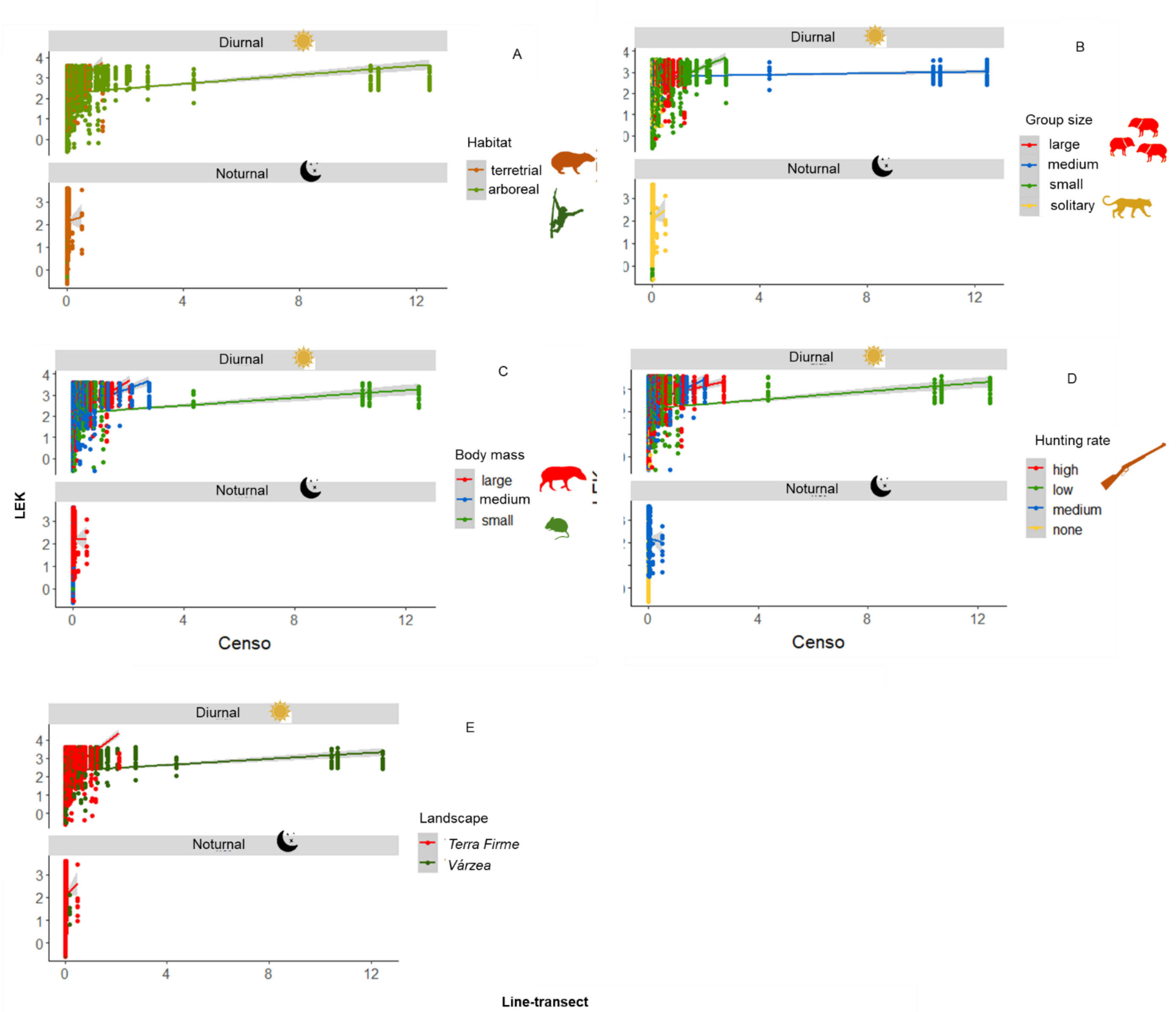


Figure 3. Relationship between the abundance obtained through the line transect and LEK for species regarding its habit (A-E), habitat (A), group size (small- until 10 individuals; medium-from 11 to 30 individuals; large-from 31 to 100 individuals) (B); body mass (small- until 1 kg; medium- from 1 kg to 5 kg; large- exceeding 5 kg) (C), hunting rate (D), forest type (E).

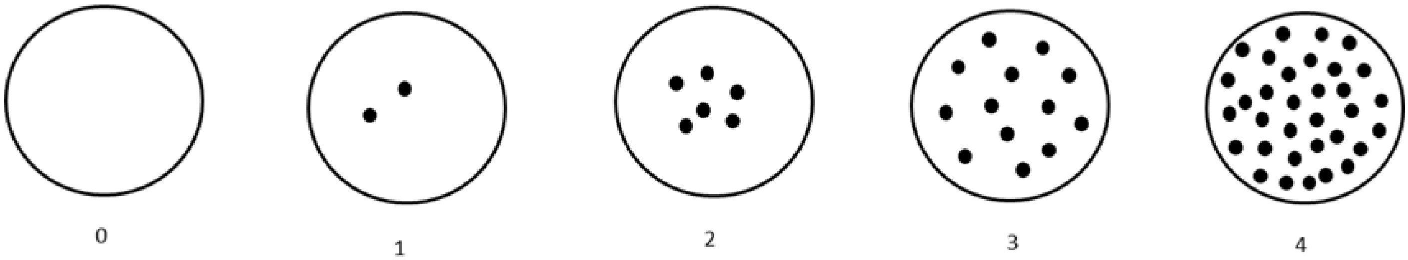


Figure 4. Example of a quantitative visual scales. Graphic depiction of the species abundance scale, ranging from 0 (when the species population is “absent”) to many specimens represented by each small circle.

CAPÍTULO 3

Consensus levels of local ecological knowledge allows collection of robust data on vertebrate abundance in the Amazon

Photo credit: Franciany Braga-Pereira

Abstract

1. Given the ongoing environmental degradation from local to global scales, it is fundamental to develop more efficient means of gathering data on species and ecosystems. Citizen science in which local communities can provide information consistently over time has been shown to be effective. However, there are few assessments of the level of uniformity of the data gathered (consensus) within and between participating human populations.
2. To determine the level of consensus on the abundance of hunted and non-hunted forest species we interviewed 323 persons in 19 villages in Western and Central Amazon. These villages varied in size and socio-economic characteristics and in the experience with wildlife of their dwellers. Interviewees estimated the relative abundance of 101 species using a four-point Likert scale. Answers were evaluated using the social network data analysis software, UCINET. We then compared the answer offered by each interviewee with the most frequently given response within a village; each interviewee was given a consensus score of 0 – 1 (high consensus >0.6). To obtain the consensus level of the abundance for each species we contrasted the average abundance value per species per village with its standard deviation (SD). A high consensus score is when the SD was smaller than the average value and a low consensus score was when the SD value was greater than the average value.
3. High consensus was found for species population abundance in all sampled villages and for 79.6% of interviewees. Village consensus of all species abundance pooled was negatively correlated with village population size. The consensus level was high regardless of the interviewees' hunting experience. Species that are more frequently hunted or had higher abundance as measured in line transects had greater consensus scores; the only two species with low consensus are rare and solitary.
4. We show in our study in the Amazon that information gathered by local peoples, Indigenous as well as non-Indigenous, can be useful in understanding the status of animal species found within their environment. The high level of cultural consensus we describe likely arises from knowledge sharing and from the strong connection between the persons interviewed and the forest environments.

Key-words: Citizen science, hunting, Indigenous People, local communities, tropical forest,

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1. INTRODUCTION

Around 12 million rural forest inhabitants live in the Amazon of which 1.5 million are Indigenous Peoples of 300 ethnic groups. Most of these inhabitants rely on hunting and fishing for their animal protein, an important component of their food security (FAO & FILAC, 2021). Forest peoples directly interact with animals through hunting and fishing. Since childhood, they can identify animals' signs and interpret these in terms of the animals' behaviour or potential environmental changes (Albert, 2016). This interaction with the forest and its biodiversity involves an intricate web of human and extra-human relations linked to kinship, social relations and norms, rituals and cosmology (Ingold, 2000).

An increasing number of wildlife conservation and research projects have used citizen science initiatives to collect data that would otherwise take more time and resources if only scientists were involved (Braga-Pereira et al., 2021; Farhadinia et al., 2018; Ponce-Martins et al., 2022; Van Damme et al., 2015). An example of this is the use of interviews with local hunters to estimate the relative abundance of terrestrial vertebrate species that show there is a significant correlation with abundance estimates obtained using conventional methods such as linear transects and camera traps (Braga-Pereira et al., 2021; Camino et al., 2020; Zayonc & Coomes, 2022). These associations can then be translated into a wider understanding of the species found around hunter settlements and beyond, including the relationship among them and with their environment, across time and space (Braga-Pereira et al., 2021). This broad knowledge base is described as Local Ecological Knowledge (LEK). As LEK includes traditional, Indigenous and local knowledge, herein we use this term instead of traditional or indigenous ecological knowledge.

Although academic knowledge and LEK are distinct systems both rely on an internal process of information validation (Congretel & Pinton, 2020; Cunha, 2007; Rodríguez & Pérez, 2017). The former is primarily corroborated through peer review by other scientists whereas when LEK is used, the level of consensus of answers given by the interviewed persons can be used to validate the information (Burgess et al., 2018). The level of agreement between answers can be considered a proxy of common knowledge and tools such as cultural consensus analysis (CCA) can be used (Burgess et al., 2018). Basic assumptions in the application of CCA are that: (1) responses should be solicited independently (i.e., not in a focus group); (2) only one domain of knowledge is tested at a time (e.g., a test of knowledge about animal abundance and medicinal plants would not be appropriate); and (3) participants share a common culture and there is a single ‘true answer’ to each question (Romney et al., 1986; Weller, 2007; Borgatti & Halgin, 2011).

CCA has been applied in several fields to measure people's perceptions of a given subject and to assess whether individuals share the same cultural concepts. Initially, CCA was widely used in health (Romney et al., 1986, Garro, 1986, Moore et al., 1997, Weller & Baer 2001). In the last decades CCA has been applied to environmental conservation and resource management issues (Miller et al., 2004, Grant & Miller 2004, Vieira 2019), such as to analyse use and classification of plants (Canales et al., 2005, Case et al., 2005; Galeano 2000; Hanazaki, 2010) and animals (Romulo et al., 2011; Volpato et al., 2011; Kent, 2011; Van et al., 2010; Rickenbach, 2015). Since culture is shared knowledge, consensus amongst a group of people assumes that the answer given to a question by a person will corroborate with the answer of another person. On the other hand, people who do not know the answer, due to differences in their experiences, are less likely to agree with the others. Ultimately, the answer appearing at a higher frequency amongst a group of experts would be the valid answer (Borgatti & Halgin, 2011; Romney et al., 1986, Weller, 2007).

In this study, we interviewed inhabitants of several socially and demographically distinct settlements (villages) in Western and Central Amazon about their perception of the abundance of forest species. We determined consensus levels amongst interviewees in each village regarding the abundance of species found in their environment. Given that people with more hunting experience or who have lived longer in a specific village (and not in the urban area) will have had more contact with animals and will have shared and received more information, we expected that hunting experience (Hypothesis 1) and time living in the village (Hypothesis 2) will influence their personal consensus level. As the information and experiences are likely to circulate faster in villages with lower population sizes, we expected that the consensus level will be influenced by the number of inhabitants in each village (Hypothesis 3). People living in non-flooded and flooded areas in the Amazon interact in different ways with the different species in each habitat. This is due to contrasts in peoples’ dependence on resident fauna for food, but because the abrupt seasonal water level changes in flooded habitats affect access to wild species during certain times of the year. Therefore, we expected that the type of environment in which a village is situated can influence people's consensus on the abundance of species (Hypothesis 4). Given that some species will attract greater attention from people due their characteristics and the use made of these animals, we expected to see people would pay more attention and show high levels of consensus for species that are hunted (Hypothesis 5). Considering that more abundant species, living in groups, of medium and large body size are more easily detected in forest, we predicted that species' abundance (Hypothesis 6), body size (Hypothesis 7) and sociability (Hypothesis 8) will influence the consensus of the abundance of these species. Finally, since local populations carry out their daily activities in aquatic and terrestrial environments at different times during the day, we considered that the species' habit (Hypothesis 9) and locomotion mode (Hypothesis 10) will not influence consensus of species abundance.

2. **METHODS**

2.1 Study areas and human populations

This study was conducted in a total of 18 villages in the Brazilian (n=9) and Peruvian (n=9) Amazon. These included eight sites in upland forest, four in flooded forest and six in a mix of upland and flooded forests (Fig. 1). There were nine Indigenous villages and another nine non-Indigenous riverine settlements. Village population sizes were comparable in both countries (Brazil: mean ± SD = 208.4 ±148.1, range = 35-519 inhabitants; Peru: mean ± SD = 224.4 ±170.6, range = 50-559 inhabitants) (see Supplementary Table 1).

Our study villages in Peru were composed of Indigenous groups that included Boras, Huitotos, Kichwas, Maijuna, Matses, Secoyas and Yagua. In Brazil, we worked in non-Indigenous riverine villages inside Sustainable Use Protected Areas (SUPA) inhabited by *caboclos* or *ribeirinhos*. This latter group are peasant populations living along the river margins of the Amazon basin, descendants of intermarriage, and cultural and economic assimilation between remnant Indigenous populations, European descendants, and Afro-Brazilian settlers from north-eastern Brazil (Fraser, 2010).

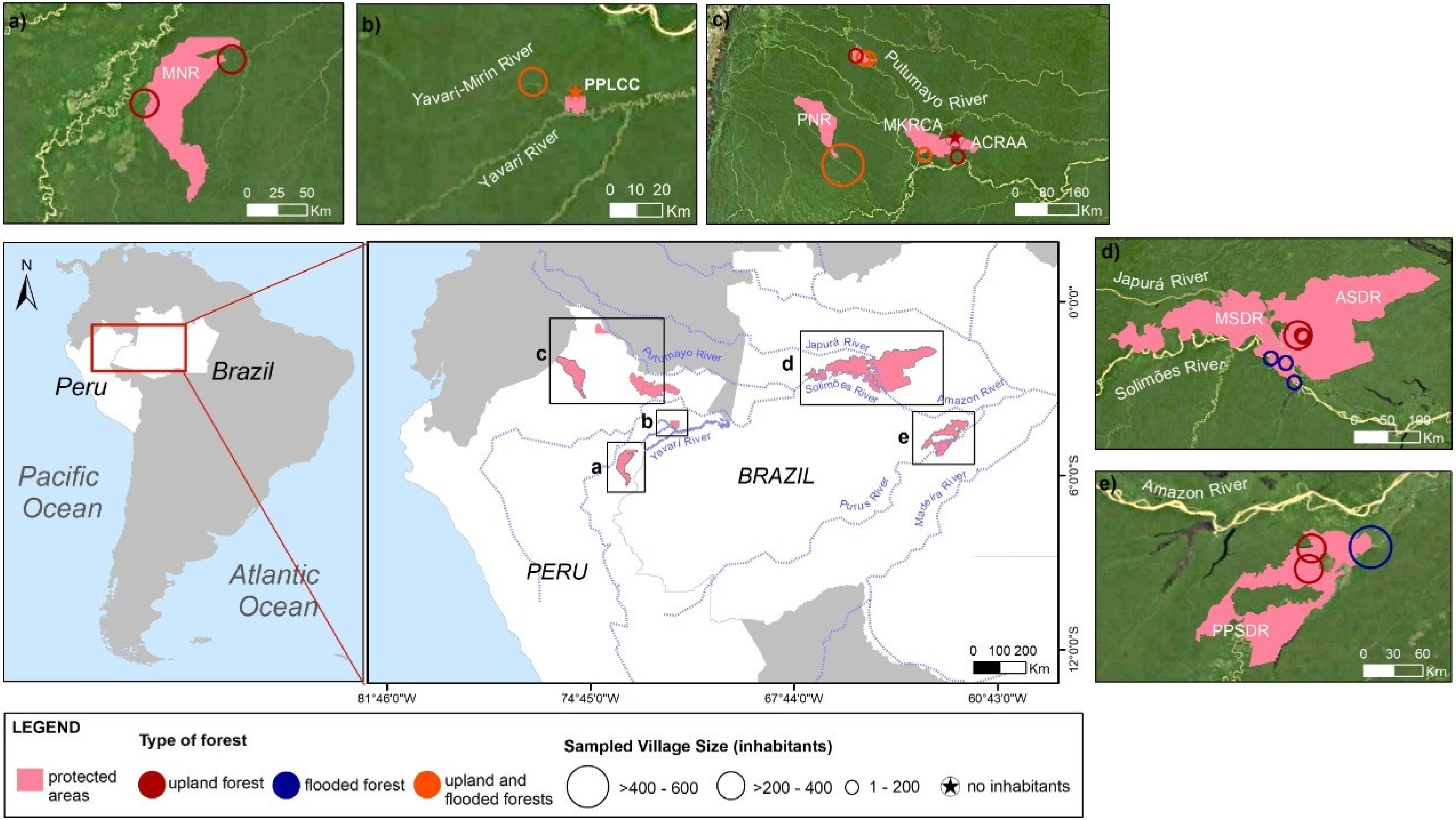
a)
MNR
0 25 50
Km
b)
Yavari-Mirin River
PPLCC
Yavari River
0 10 20
Km
c)
Putumayo River
PNR
MKRCA
ACRAA
0 80 160
Km
N
Peru
Brazil
Pacific Ocean
Atlantic Ocean
c
d
b
a
e
Putumayo River
Japurá River
Solimões River
Amazon River
Yavari River
Purus River
Madeira River
BRAZIL
PERU
0°0'0"
6°0'0"S
12°0'0"S
81°46'0"W
74°45'0"W
67°44'0"W
60°43'0"W
0 100 200
Km
d)
Japurá River
ASDR
MSDR
Solimões River
0 50 100
Km
e)
Amazon River
PPSDR
0 30 60
Km
LEGEND
protected areas
Type of forest
upland forest
flooded forest
upland and flooded forests
Sampled Village Size (inhabitants)
>400 - 600
>200 - 400
1 - 200
no inhabitants

Figure 1. Map of the study area portraying the 18 sites in Central and Western Amazonia. Pink background areas represent protected areas. Letters (a–e) provide close-up views of the sampled regions and study areas; MNR: Matsés National; PPLCC: Lago Preto and Paredón Conservation Concession; PNR: Reserve; Pucacuro National Reserve; MKRCA: Maijuna-Kichwa Regional Conservation Area; ACRAA: Ampiyacu Apayacu Regional Conservation Area. MSDR: Mamirauá Sustainable Development Reserve; ASDR: Amanã Sustainable Development Reserve; PPSDR: Piagaçu-Purus Sustainable Development Reserve. Map generated using ArcGIS 10.3.1; Datum: WGS84 Source: ESRI.

People in villages within SUPAs and Indigenous lands in Brazil and Peru are legally allowed to take part in decision-making on natural resource use and management in their areas of influence. Residents exploit natural resources (e.g., hunting, harvesting forest products and fishing) for food and other purposes. In some villages, timber, fish, wild meat and agricultural products are also opportunistically traded. Accessibility to urban areas is difficult but has increased with more frequent logging traffic, facilitating the introduction of urban customs that require monetary income, such as the use of electrical appliances (Bernárdez-Rodriguez et al., 2021). Fishing and hunting are the most important subsistence activity for all residents. Hunters are mostly men, but boys (when on vacation) will accompany their fathers on hunts from around 11 years of age and can go out on their own at around 15 years of age (Vieira et al., 2019).

2.2 Data collection

Using structured interviews in all villages, between 2013 and 2017, we surveyed abundance perceptions of villagers of a total of 101 species (>1kg). This number of species included 41 primates (45%), 12 birds (13.2%), 12 carnivorans (13.2%), 8 rodents (8.8%), 6 sloths and tamanduas (6.6%), 5 even-toed ungulates (5.5%), 3 armadillos (3.3%), 2 opossums (2.2%), 1 tapir (1.1%) and 1 turtle (1.1%). All birds considered in this study were game birds (Braga- Pereira et al., 2021). A median of 35 species (range = 14–45) were found per site. There was a median of 30.8 ± 8.8 species (range = 17–42) per site. Since the assembly of species of primates, birds, and Testudines varied among sites, we pooled these taxa at genus level so the consensus for these species could be compared (Supplementary material table 2).

We interviewed only persons who were known to be knowledgeable of wild animals and their environments. To select first persons to interview we identified potential interviewees using our previous knowledge of residents in each village. We then employed a snowball sampling technique (Bailey, 1994) and asked the first persons approached to suggest others and asked them to participate in the study and did the same procedure with the new participants. We interviewed each person on their own so that no conferring between interviewees occurred (Kent, 2011). We used an illustrated checklist with colour plates of the main animal species known to occur in each study area (Supplementary Material Fig 1).

For each species, we asked the interviewee to estimate its relative abundance on a Likert scale; 0 (when the species was “absent”), 1 (low abundance), 2 (medium abundance) and 3 (high abundance) (Van Holt et al., 2010, 2016). A zero value was assigned only to species whose occurrence was expected to occur in an area from previous studies, but which was considered absent by an interviewee. The abundance of a species was assessed in comparison to others occurring in the area (the abundance of each new species presented during the interview was compared to the previous ones). We added a species that did not occur locally to test the veracity of the answers. The abundance of a species indicated its status within a radius of approximately 5 km from the centre of the village where each hunter lived. This is the radius that hunters usually carry out their hunting activities.

Interviews did not require local translators as interviewers and interviewee, including those from Indigenous territories were fluent in Portuguese or Spanish. Research groups involved in this study who conducted the interviews were already working in each site for 10-20 years and had built relationships of trust in the communities prior to data collection.

2.3 Predictor variables

We used the following socioeconomic predictor variables:
1) number of years an interviewee had dedicated to hunting as a measure of his hunting experience; 2) percentage time each respondent spent outside the village in which they currently live compared to time spent within the village; and 3) village population size.

Biological and ecological predictor variables included:
Species traits: 1) body mass; 2) sociality (solitary/social – with two or more individuals); 3) habit (diurnal/nocturnal); 4) locomotion mode (arboreal/terrestrial) (see Supplementary material table 3); and 5) abundance data collected through line transects in each sampled villages (obtained from Braga- Pereira et al., 2021).

Habitat type: 1) upland forest (non-flooded forest located in sites with higher elevation within the Amazon rainforest) and 2) white-water flooded forest (a seasonal floodplain forest inundated by white-water rivers that flow within the Amazon rainforest).

Hunting rate: the number of individuals hunted per person per year across the Brazilian Amazon was taken from data in Peres (2000); if a species was not listed by Peres (2000), we considered it as having a hunting rate of 0.

2.4 Response variables

Using the software UCINET Version 6.511 (Borgatti et al., 2002) we generated the following consensus scores:

Personal (interviewee) consensus: we created a matrix using the answers of perceived abundance level of each species by each interviewee in each village. From this answers matrix the software determined the *valid answer* on species abundance as the answer appearing with higher frequency in each village (Borgatti & Halgin, 2011; Romney et al., 1986; Weller, 2007). Comparing the answer of each interviewee to the valid answer, each interviewee received a consensus score (ranging from 0 to 1). People providing responses that are similar to the *valid answer* received a score near to 1 and those people providing responses that were different received a score near to zero (Romney et al., 1986).

Village consensus: we created a matrix inputting each personal consensus score. So, the software calculated the village consensus comparing the personal consensus of people belonging to the same village. Scores above 0.5 are considered high consensus scores, or in the case of more conservative authors, scores above 0.6. Here, we used a conservative estimate of cultural competence of 0.6 (Romney *et al.,* 1986).

Consensus per species: first we calculated the average of abundance value per species per village (hereafter, average value). With this average value we calculated a standard deviation per species per village (hereafter, SD value). The SD values generated is a convergence measure, the lower the deviation values, the higher the agreement. Then we compared the difference between the average and SD of the abundance values, to obtain a consensus level per species (hereafter, consensus score per species):

$$\text{Consensus score per species} = \text{average value} - \text{SD value}$$

where a high consensus score is when the SD value is smaller than the average value (positive value) and a low consensus score is when the SD value is larger than the average value (negative value) i.e., there is a high variation in the abundance values reported by the interviewees.

2.5 Data analysis

Using the consensus results obtained, we developed sets of models to assess the following:

Personal consensus: we conducted generalized linear models (GLM), using hunting experience of each interviewee (Hypothesis 1) and time each interviewee lived outside the village they currently lived in (Hypothesis 2) as predictor variables.

Village consensus: we conducted GLM using village population size (Hypothesis 3) and landscape type (flooded and upland forests) (Hypothesis 4) of each village as predictor variables.

Consensus per species: we conducted generalized linear mixed models (GLMM). Species' hunting level (Hypothesis 5), body mass (Hypothesis 7), sociality (solitary/social - with two or more individuals) (Hypothesis 8), habit (diurnal/nocturnal) (Hypothesis 9), and locomotion mode (arboreal/terrestrial/aquatic) (Hypothesis 10) as predictor variable of fixed effects; and village as random variables (Zuur et al., 2007).

Using linear regression, we tested the influence of abundance obtained through linear transects on the consensus for each species (Hypothesis 6). Abundance cannot be included as a predictor variable in the above GLMM as it correlates with the species' habit. In addition, we did not have line transects abundance information for 1 of the 18 sampled sites, so we used a dataset with 18 sites in this model.

We found no collinearity ($p > 0.05$) among predictor variables in the GLMM. For GLM and GLMM we used the Beta-Inflated family of distribution, based on type of data. We used residual checks to verify whether our models were, in principle, suitable or not. We used the Akaike information criterion (AIC) to select models of interest. The model with the lowest AIC was retained, and the remaining competing models were ordered according to their Akaike differences (ΔAIC) with respect to the best model (lowest AIC). All analyses were performed in R ver. 3.5.3 (R Core Team 2019) using the ordinal (Christensen, 2019), MuMin e lme4 (Oksanen et al., 2013) packages.

We did not analyse the difference in consensus between Indigenous and non-Indigenous villages because this variable (village ethnicity) was influenced by the variable "village population size". This is because most Indigenous and non-Indigenous villages coincide with villages of greater and smaller population size, respectively. As the number of women interviewed compared to the men was small we could not verify if gender influenced consensus level either.

2.6 Ethics statement

We followed the rules and guidelines for applying Free, Prior and Informed Consent as detailed in Buppert & McKeehan (2013). This research was approved by the Instituto Chico Mendes de Conservação da Biodiversidade from Brazil (License SISBIO 29092-1; SISBIO 2; 29092-3; SISBIO 29092-4; SISBIO 29092-5; SISBIO 29092-6; CEUC 1474/2011, CEUC 003/2013 e CEUC 052/2011) and the Dirección General de Flora y Fauna Silvestre from Peru (License 0350-2012-DGFFS-DGEFFS; 0068-2015-SERFOR-DGGSPFFS). Community meetings and coordination with communal authorities were carried out prior to conducting interviews to agree on procedures.

3. RESULTS

We interviewed a total of 323 inhabitants in the study villages. The average (± SD) number of interviewees for all villages was 17.8 ± 9.6, range 6 – 42 (15.1 ± 5.8 in Brazilian villages and 20.6 ± 12.1 in the Peruvian villages). We interviewed a total of 2 women and 321 men. Ages ranged from 16 to 75 years old (37.8 ± 14.0). Hunting experience of interviewees varied from 0 to 64 years (21.7 ± 14.8 years). Percentage time respondents were absent from their villages (either in another village or in the urban area) compared to time living in the village ranged from 0 to 95 (16.4 ± 25.8%).

3.1 Personal and village consensus baseline

The overall mean competence score was 0.7 ± 0.01 (range = 0.6 - 0.9) for village consensus and 0.7 ± 0.1 (range = 0.1 - 0.9) for personal consensus. These results indicate that amongst most people there was high agreement on the abundance of vertebrate populations occurring around their village. For all villages and 79.6% of interviewees, we found consensus scores of species population abundance to be above the 0.6 threshold (Fig 2A e B).

3.2 Consensus per species

Of the 101 species, we found a high consensus score for 81 species (85.3 %) in villages pooled (Fig 2C). We found a high consensus level among all interviewees for the abundance of *Dasyprocta* sp. and *Saimiri vazoline* in the sampled villages. We found low consensus in at least 50% of the villages for two species, *Puma yagouaroundi* and *Procyon cancrivorus* (Fig 3).

3.3 Consensus model results

We did not find any effect of hunting experience and time living outside the village on personal consensus (Table 1; Supplementary material Fig 2 A and B). Thus, Hypotheses 1 and 2 were refuted. Village consensus on the other hand was significantly higher the lower the population size of a given village (Fig 4), but we did not find difference among landscape, so our Hypotheses 3 was accepted, and Hypothesis 4 was refuted (Supplementary material Fig 3). Finally, we found that consensus was higher for species with higher abundance in line transects and that are hunted more often (Fig 5A and B). So, Hypotheses 5 and 6 were also accepted. The other variables (species' body size, sociability, habit and locomotion mode) showed no effect on the level of consensus per species (Table 2; Supplementary material Fig 4 A, B, C, D). The predictor variables with significant effect on the consensus are showed in detail in Fig. 4 and Fig. 5A and B.

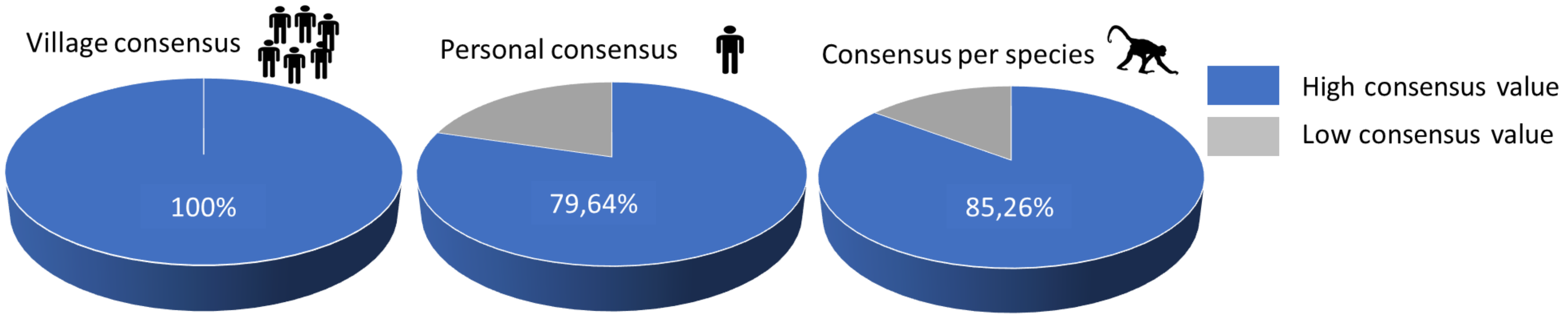

Figure 2. Percentage of villages (A), interviewee (B) and species (C) presenting high and low scores of cultural consensus. For village and personal consensus, a high consensus level is that above of 0.6. For consensus per species a high and low consensus level is when the SD value is smaller and greater, respectively, than the average value.

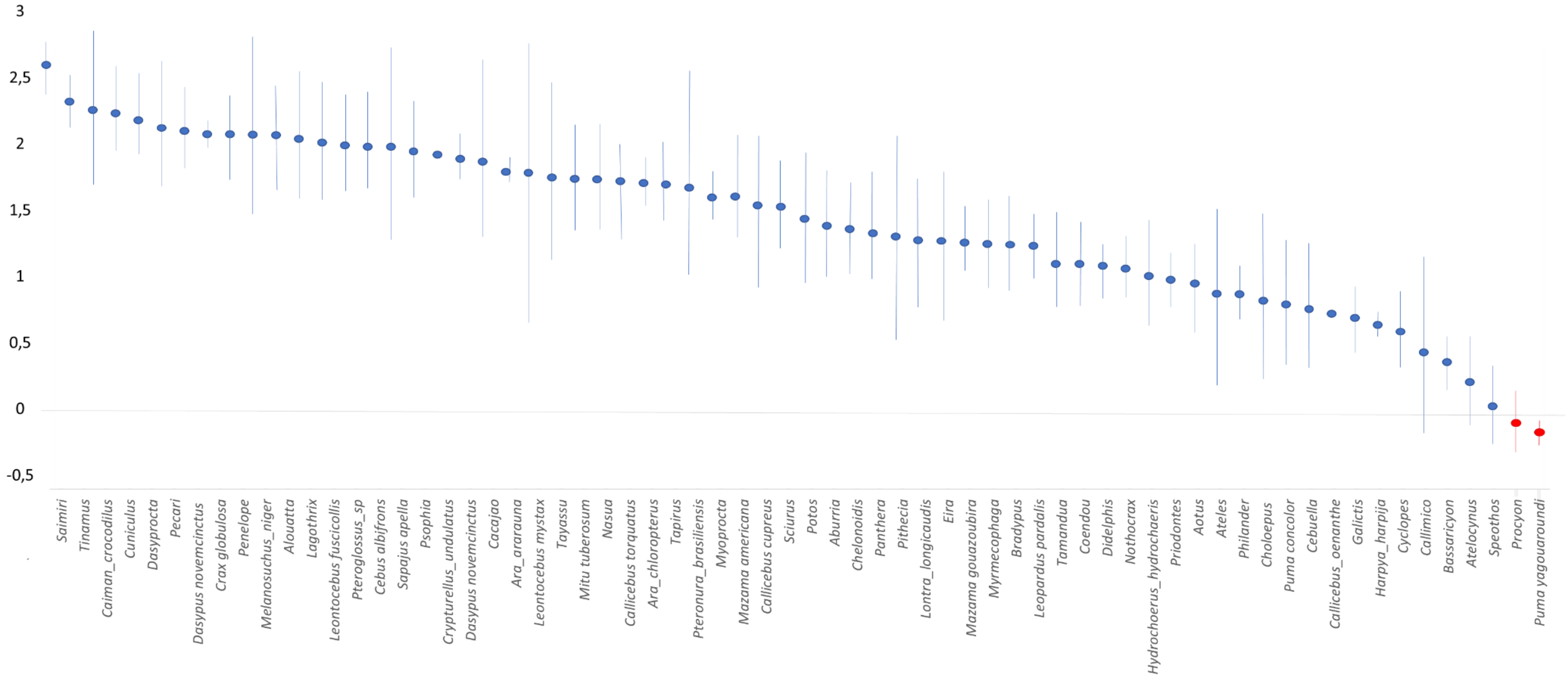

Figure 3.Consensus scores per ecospecies. The circles represent the average value of consensus per ecospecies (calculated from the difference between the average and SD abundance values per species per village). The bars represent the maximum and minimum consensus scores obtained for each species in different villages. Blues circles represent high consensus score (SD value smaller than the average value). Red circles represent low consensus score (SD value greater than the average value).

Table 1. Details of the all generalized linear mixed models and linear models elaborated for verify the influence of social factors on the cultural consensus obtained by each interviewee (personal consensus) and by each village (village consensus). Estimated values indicate the coefficients associated with the variable listed on the left. This represents the estimated amount by which the odds (that each response variable would increase if each explanatory variable were one unit higher). Z-values indicate the degree to which explanatory variables exert a significant effect. Pr (>|z) denote significance levels as following: ns $P > 0.05$; ** $P \leq 0.01$; *** $P \leq 0.001$. AIC Akaike Information Criterion; ΔAIC difference of AIC of the selected model in relation to the null model. Variables with significant effect are in bold.

Response variable	Predictors	Estimate	Std. Error	z value	Pr(>|z|)		AIC	AIC Null model	ΔAIC
Personal consensus	Time hunting	-5E-04	0.0032	-0.152	0.879		-275.45	-278.95	-3.5
	Time far away from the village	0.1249	0.1809	0.69	0.491				
Village consensus	**Number of inhabitants per village**	**-4E-04**	**1E-04**	**-3.075**	**0.007**	**	**-1132.32**	**-1096.21**	**36.103**
	Flooded: Flooded and Upland	0.04421	0.032	1.382	0.1675				
	Flooded: Upland	0.0694	0.03187	2.178	0.0797				

Table 2. Details of all generalized linear mixed models and linear models elaborated for verify the influence of biological variables, hunting level and landscape type on the cultural consensus obtained for each species (consensus per species). Estimated values indicate the coefficients associated with the variable listed on the left. This represents the estimated amount by which the odds (that each response variable would increase if each explanatory variable were one unit higher). Z-values indicate the degree to which explanatory variables exert a significant effect. Pr (>|z) denote significance levels as following: ns P > 0.05; ** P ≤ 0.01; *** P ≤ 0.001. AIC Akaike Information Criterion; ΔAIC difference of AIC of the selected model in relation to the null model. Variables with significant effect are in bold.

	Response variable	Predictors	Estimate	Std. Error	z value	Pr(>|z|)		AIC	ΔAIC	Model
Model 1	Consensus per species	**Hunting rate**	**2.77E-01**	**4.33E-02**	**6.394**	**2.94E-10**	***	-1130.9		GLMM
		Body mass	-2.33E-02	1.73E-02	-1.346	0.1786				
		Group size	-6.45E-05	1.45E-04	-0.444	0.6574				
		Nocturnal: diurnal	-1.74E-02	3.37E-02	-0.517	0.6056				
		Arboreal: aquatic	-1.13E-02	5.13E-02	-0.219	0.8264				
		Terrestrial: aquatic	-7.03E-03	5.18E-02	-0.136	0.8922				
Null model		Mo						-1096.2	37.631	
Model 2	Consensus per species	**Individuals per km**	**0.043249**	**0.00583**	**7.421**	**3.76E-13**	***			**LM**

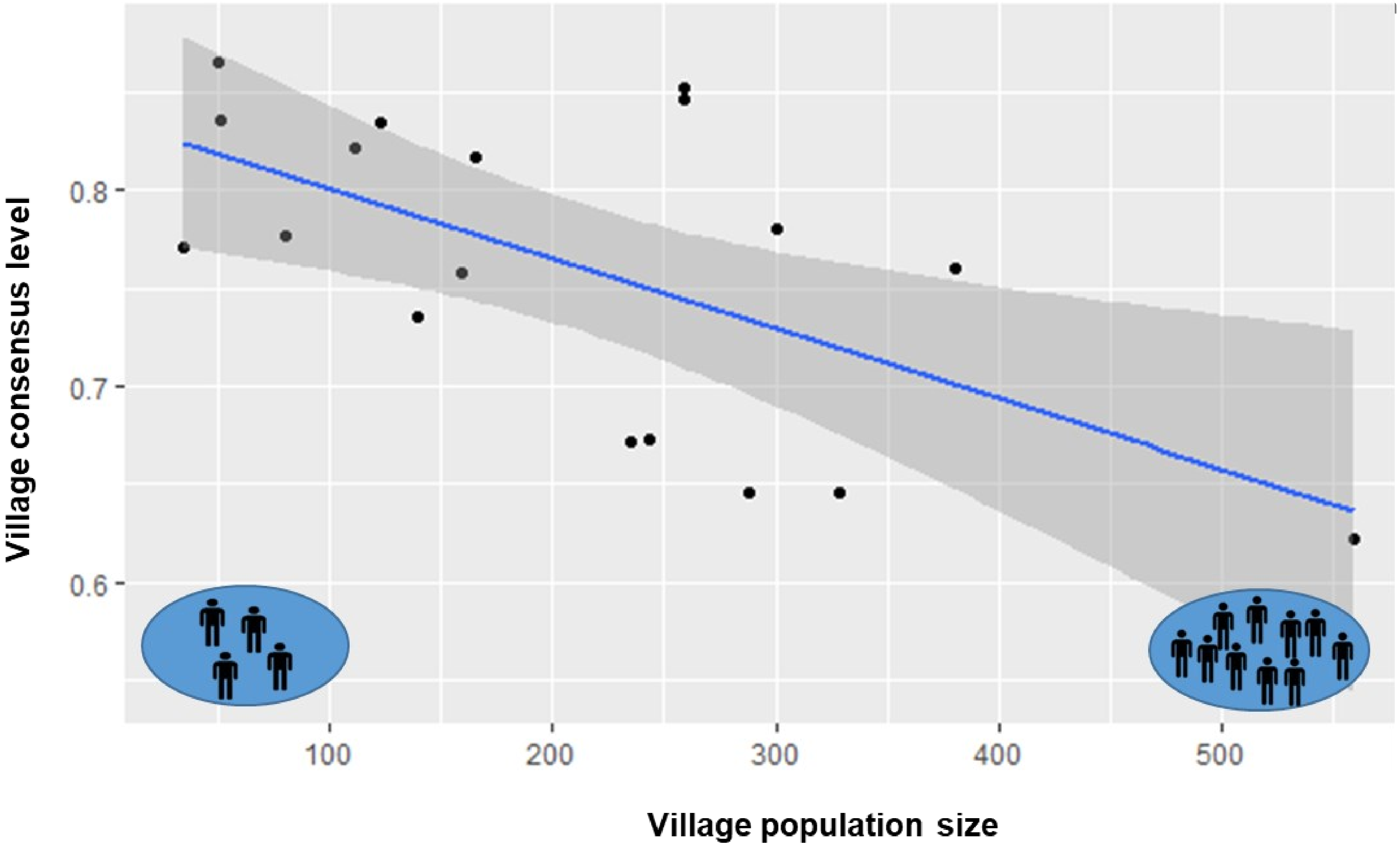

Figure 4. Relationship between village consensus and village population size. Values of village consensus >0.6 indicate a high efficiency in perceiving abundance value. The gray area represents 95% CIs.

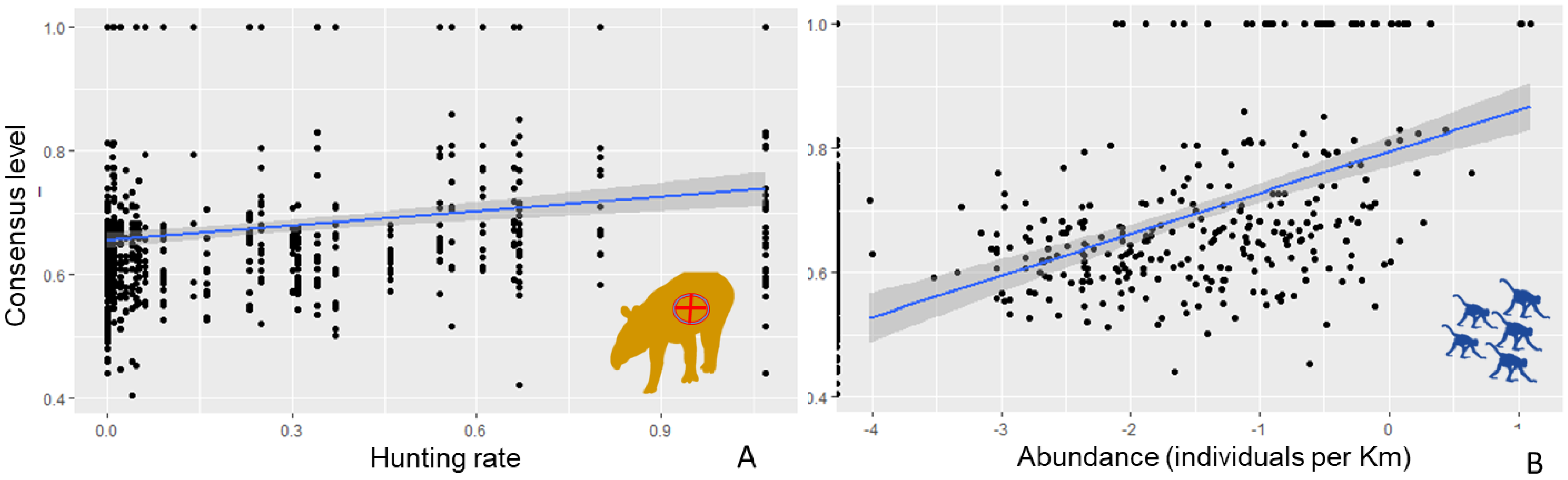

Figure 5. Relationship between (A) consensus per species and hunting rate; and (B) consensus per species and abundance ($\log_{10}$). Values near of 1.0 indicate consistence in the answers regarding the perceived abundance value. The grey area represents 95% Cis.

4. DISCUSSION

We found a high consensus of species abundance for most species, by most people and in all villages. Given these results, we can conclude with some confidence that local knowledge of biodiversity experts can be valuable to estimate vertebrate population abundance, even by people with little hunting experience and for rare, solitary, nocturnal and non-hunted species. In this study, *Saimiri, Tinamus, Caiman crocodilos, Cuniculus paca, Dasyprocta* and *Pecari tajacu* represent the species with higher consensus level. This may be related to typically higher abundance of these species in the sampled villages and therefore greater familiarity of interviewees with these species around their sites. It is probable also that this high consensus could have been affected by the fact that we only used one high abundance category in the Likert scale. A way to improve the accuracy of species abundance estimates would be to include a larger number of categories using a quantitative, not a nominal, scale (Fig 6) and asking questions with a clear a reference point for abundance estimates (Braga-Pereira et al., 2020). For more details to improve the accuracy of species abundance estimates using LEK see Box 1 “Recommendations for improve the data collection on species abundance”.

The correlation found between higher consensus levels and more abundant species from linear transect data also supports the hypothesis that visibility of an animal (through its abundance) will influence its detection probability, increasing the chances of encounter by people (Hanazaki, 2010). As pointed by Hunn (1999), more abundant and conspicuous species are more likely to be noticed than less abundant and secretive ones. In accordance with our work, highest concordances were linked only to species’ abundance and not size in a study of consensus in naming trees (Hanazaki, 2010).

Several studies have indicated that abundance of animals of a species is a variable of importance in human-wildlife relationships (Tamburini et al., 2021, Zamudio & Hilgert, 2018, Wajner et al., 2019, Gosler, 2017). The ecological significance given to the abundance or conspicuousness of an animal is considered greater than its cultural importance (Bentley & Rodriguez, 2001), but interestingly cultural importance can be maintained even when abundance of the species decreases (Gosler, 2017). In this regard, it should be noted that, in addition to abundance, utilitarian and cultural aspects also affect the cultural importance of a species (Lucena et al., 2012; Wajneret al., 2019). For example, the six species with the highest consensus levels in the studied areas are also used as pets or hunted for their meat and for other by-products in the region. The usefulness value of these species will increase peoples' interest. This would explain why a higher consensus were typical of the more hunted species. Villagers have an interest where they see animals they can hunt, how many there are and how they behave. This curiosity is shared not only by hunters, but among everybody in a village, as hunted species are an important source of food and because not only hunters (and usually adult men) are involved in the wild meat trade (El Bizri et al., 2020). This can also explain why species such as *P. cancrivorus, P. yagouaroundi* and *Speothos venaticus* had the lowest consensus values, as these species are rare and unusually hunted.

Since morphological, ecological and behavioural characteristics of a species affect their observability, many species go unobserved not just because they are rare, but because they are very small, solitary, cryptic, silent, or nocturnal (Atran et al., 2002; Bentley & Rodriguez, 2001; Gosler, 2017; Hunn, 1999; Zamudio & Hilgert, 2018). All three species that had a low consensus level in our study fit into at least two of these characteristics. *P. cancrivorus* is a solitary and nocturnal animal, *P. yagouaroundi* is a solitary species, silent and furtive, and *S. venaticus*, in addition to being extremely rare and cryptic, is a solitary, silent and nocturnal species. We emphasise that even though abundance and the utilitarian factor are variables found in our study that significantly influence consensus levels, other factors, although less prominent, may also be influencing our results.

Indigenous Peoples and Local Communities (IPLC) understand the ecology and behaviour of non-hunted animals, since we found a good consensus level even for these species. This is because IPLC also observe non-hunted species in their daily lives and activities. For example, when spending time at salt licks, a common practice performed by hunters in the Amazon to hunt animals such as tapir and deer, hunters also observe many other non-game species that visit the site (Montenegro, 2004). There is also indirect hunter observation of non-hunted species. For example, in northern Brazil, the Yanomami people continuously turn toward and listen to the biophony of the forest. They are also constantly involved in decoding an elaborate system of sounds connected

Box 1- Recommendations for improve the data collection on species abundance.

Use quantitative visual scales (not a nominal one) with many levels of relative abundance (Fig. 6) (Braga-Pereira et al., 2020)

- Delimitate a clear abundance reference point: e.g., i) ask for different years ("are peccaries more abundant 10 years ago or now?"), ii) ask for different areas ("are peccaries more abundant in area A or B?"), iii) compare different species ("are peccaries or tapir more abundant?").
- Start asking for key species (for example one that you know is very abundant and another that is absent).
- Delimitate the research area: use of maps for the interviewee to describe more accurately the area about which he/she is answering

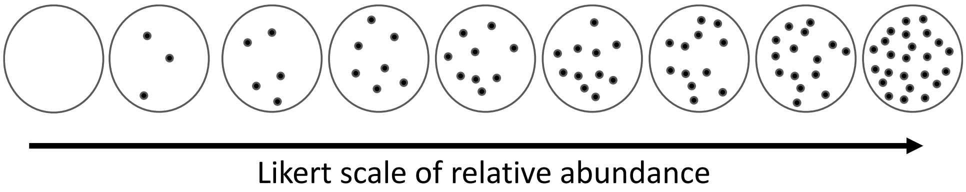

Figure 6. example of quantitative Likert scale with broad abundance ratings

Ask for the number of individuals perceived by the expert.

- Focusing on fewer target species and in specific sites would allow this more refined questioning (e.g., "how many tapirs visit the salt lick A?").
- To facilitate quantification, provide seeds to the interviewee and ask her/him to place the seed number referring to the number of individuals of each species in a site, on the image of the animal (Fig. 7) (Chaves et al., 2020).

Figure 7. example of how interviewees can indicate the number of individuals of each species in a specific site

- For social species, ask for the size of the group.

Measure the hunter effort

- Ask how long time/ how far walking is need to find a target animal (Coomes et al 2020; Zayonc & Coomes, 2022);
- Ask how frequently an expert encounters a species in a particular area.

Have information of the difference between the years of hunting experience and how many years ago the interviewee stopped with the activity.

with the notion of songs, cries, and calls of many birds, amphibians and certain insects that they interpret as acoustic clues such as the possible presence of the prey, fruits or plants associated with them (Albert, 2016) indirectly gathering information on numerous species.

Amazonian hunters have developed other skills to identify animals, besides direct sightings. This is because Amazon rainforest is dense and dark often impeding the hunter to see the animal (Cormier, 2000) other than at very short distances. This means that hunters must rely on clues such as sound or smell to detect the presence or movement of animals in the undergrowth or in the treetops (Albert, 2016). Hunting experience and success is therefore closely linked to an intricate knowledge of animals, where hunters can tell the sex, age and spatial location of an animal just from its vocalization, odour intensity, body shape or signs left by the animal on foliage.

Given that hunters use time-acquired skills to detect animals in the forest, we expected that experienced hunters would have higher consensus compared to non-hunters. However, we did not detect any difference between these. This result coincides with Zayonc & Coomes (2022) and it may be related to the interest that all villagers have in talking about hunted species. Among some Indigenous Peoples, it is common for hunters to share their histories with all the community after coming back from their hunts (Aparicio, 2014). Older hunters are more experienced, as we show in the positive correlation between interviewee's age and time hunting ($p < 2e-16$) (supplementary material Fig 2C), but knowledge of elderly and younger interviewees did not differ, being high for all age groups. Measuring the difference between years of hunting experience and how many years ago the interviewee stopped the activity would be interesting to investigate.

We also expected to find a significant and negative association between time absent from the village and personal consensus, explained by the hypothesis of empirical knowledge and erosion of the LEK: more time in the city means less time in the forest and in dialogue with other local members who know about the forest. Therefore, less is learned and updated about animals in practice. After all, traditional knowledge resides as much or more in its investigation processes as in the ready-made collections transmitted by previous generations. Regarding on that, studies have shown that both the migration of young people to urban centres or migration of even adult residents from their home communities (Bonsi, 1980) can trigger or accelerate the processes of LEK erosion (Ohmagari & Berkes, 1997; Reyes-García et al., 2010; Zent, 1999;

Prado & Murrieta, 2018), because it deprives people of exchanging and updating information with other people in the family and village. The high consensus level found independently of time away from the village may be a result of the selection of interviewees through the snowball technique (persons pointed out to us to interview were primally persons who were known to have a vast knowledge of wildlife).

We found that the consensus among members of villages of smaller population size was greater, which was expected, as the lower the human density in the area, the greater the proportion of residents that the same information reaches. Because we interviewed experts, perhaps personal experience influences the answers provided by each interviewee more than the transmission of knowledge. However, we expected that the sharing of the experiences of each resident will result in greater knowledge for all persons living in the village resulting from the sum of the experiences that each person can share. We highlight that although consensus was lower in villages of larger population sizes, the consensus level was high (>0.6) in all villages in our study. This might be explained because in all sampled villages personal orality is the main way of disseminating information, so local knowledge is shared, even in the larger villages. In addition, accessibility to urban areas from most villages is difficult, which reduces the introduction of urban information in the area and keep the local knowledge sharing as one of the main knowledge sources. Finally, it is expected that there is a dilution effect operating in larger villages, however none of the sampled villages significantly large if we compare these villages, for example, with the population of neighbourhoods in the nearby cities.

Worldwide, IPLCs manage large tracts of land; Indigenous Peoples alone influence around 38 million km^2 of the world which safeguard biodiversity and mitigate against climate change (Garnett et al., 2020; O' Bryan et al., 2020; Estrada et al., 2022). As a result, there is a growing recognition among researchers and conservationists that local knowledge systems, perspectives, and histories hold globally important conservation lessons due the substantial proportion of the world's biodiversity inhabits lands managed by local peoples (Indigenous and non-indigenous) (Fletcher et al., 2021; Schuster et al., 2019; Piperno et al., 2021). Based on a large database gathered over a wide and diverse geographical and cultural scale, we can conclude that cultural consensus level among villagers is overall high arising from a long-established and intricate connection between local people and their forest environments. Our findings reinforce the importance of LEK-based methods for research such as the estimation of the status of fauna. Furthermore, the high personal consensus found, regardless of the social characteristics of the interviewees, also shows that not only the hunters are efficient in estimating the abundance of the fauna, but also other people who in somehow observed wildlife and so developed knowledge on them.

All peoples living in Amazon rainforests have accumulated a profound body of experience of Amazonian wildlife and their environments over very long periods of time. The immense value of this knowledge base must be recognised by academics and management agencies to improve their integration in research and conservation programs, as increasingly demanded by international bodies (IUCN, 2021; United Nations, 2013). The inclusion of local and Indigenous Peoples in protecting areas of global importance such as the Amazon must go beyond information gathering and move into clear joint decision-making processes that encourage decoloniality (Trisos et al., 2021) and incorporate the varied perspectives, approaches and interpretations by Indigenous Peoples and local communities from, with and within the natural environment.

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CAPÍTULO 4

Estimating the spatio-temporal availability and ecological importance of salt licks in Amazon rainforest through local ecological knowledge

Salt lick at Ueré.

Photo credit: Joaquim Gomes Lima

Abstract

In some regions of Amazonia, 25% of the hunting activities occur in salt licks, which are locations with a higher concentration of natural minerals in the soil and which are often found on the edges of creeks, therefore they are liable to periodic flooding as the water level rises. The hunters hang their nets above the salt licks and wait for game animals to visit to ingest the mineral-rich soil. In the Amazon rainforest, animals exhibit geophagical behaviour in salt licks to obtain key mineral supplementation and detoxicate from plant secondary compounds, reducing digestive disorders in their bodies. In a place visited by so many species, during the long period that hunters spend there waiting for the target species, hunters can also acquire a high level of knowledge about species that pass through there during the year and also about the ecological aspects of salt licks. Here, we evaluate the seasonal availability of salt licks in Amazonia and their use by wild animals throughout the year. For this, we used LEK-based methods and obtained information on 31 species of vertebrates visiting 56 salt licks in two regions of the Central Amazon. We also obtained information on the abundance rate in salt licks with different ecological characteristics, as well as on the behaviour pattern of the species visiting salt licks. In terms of the names given to the categories of salt lick, the one most frequently registered was *chupador*, followed by *barreiro* and *canamã,* with the name given according to the salt lick size, animals' visit period and the diversity of visitors, as well as by the flooding period of the creek water. Despite soil and water consumption in salt licks being the main attraction of wild species visiting these sites, species identified from the interviews as users of the salt licks also visit salt licks for bathing, predation and other ecological relationships and behaviours. In general, the season with the highest wild species abundance was the receding floodwaters season, because the creeks' water level decreases and so the salt lick is exposed. Conversely, during the flood pulse, interviewees perceived that the majority of salt licks are not visited by most species because they are often covered by water. Most of the interviewees (74,46%) said that salt licks are one of the main places where they hunt, however the hunting at salt licks is only done during the receding floodwaters season.

Introduction

In Amazonia, some specific places are considered crucial convergence points between the hunted fauna and the hunters, where the hunting effort is usually concentrated. In some regions of Amazonia, 25% of the hunting activities occur in the salt licks (places with a higher concentration of natural minerals), where the hunters hang their nets for waiting game animals to visit salt licks to ingest the mineral-rich soil (Walschburger & Hildebrand, 1988). Because of the time spent in these sites, hunters acquired a high level of knowledge about species that pass through there during the year and also about the ecological aspects of these landscapes.

Salt licks are natural geologic formations where animals visit and exhibit geophagical behaviour (Klaus et al. 1998; Lee et al. 2010; Panichev et al. 2013), in this case consuming soil or drinking water (Abrahams & Parsons 1996; Krishnamani & Mahaney 2000). Salt licks are widely used by animals around the world (Atwood & Weeks, 2002; Blake et al. 2011; Matsubayashi et al. 2007), and the main motivation behind geophagy seems to change through species. Most of animals consuming soil in mineral licks, has as propose to obtain key micronutrients missing in their diets (Atwood & Weeks 2002, 2003; Davis & Baillie 1988; Voros et al. 2001). However, since in some salt licks the minerals analysed present concentrations equal or lower than the concentrations found in untouched surrounding soil (Hladik & Gueguen, 1974; Arthur & Alldredge, 1979), geophagy cannot be uniquely explained by mineral supplementation (Griffiths, et al. 2020). For example, clay (e.g., bentonite, zeolite), present in higher concentrations in this type of environment, help to adsorb toxins from secondary plant compounds (e.g.: alkaloids) and alleviate digestive disorders in the animals' body through the increase of the buffering capacity (Bravo et al., 2008; Brightsmith et al., 2008; Ghanem et al., 2013; Matsubayashi et al., 2007). The soil consumed also helps to reduce parasites in the animal's body (Oates 1978; Mahaney et. al., 1997; Gilardi et. al., 1999). Thus, geophagy has multiple causes that may vary geographically, seasonally, and among groups (Davies & Baillie 1988; Setz et al. 1999).

Mineral deficiency occurs because most of the Amazon rainforest is characterized by acid soils with a low level of available nutrients and a high concentration of toxic Al (Griffiths, et al. 2020). Moreover, the western Amazon is a region deprived geographically of salt, as aerosol deposition of salt declines with distance from oceanic sources (Dudley et al., 2012). In this type of environment, herbivorous species may face mineral limitations if their only source of minerals is plant resources. If salt licks in the Amazonian region provides some minerals with nutritional importance for herbivores, their existence can reduce the cost of maintaining health and/or obtaining adequate nutrition and, thus, can be fundamental for the sufferance of wild species.

In addition to the benefits for the herbivorous and omnivorous' nutrition and health, these landscapes may also work as places for carnivores to easily find their prey, among other social encounters (Griffiths et al., 2020; Link & Fiore, 2013; Matsuda & Izawa, 2008). Also, the presence of salt licks in some regions may affect population density and structure, and influence the carrying capacity of a population (Klaus & Schmid, 1998). Since salt licks may have more than one function for wildlife species, and those functions may vary across species, licks may represent a resource whose ecological importance goes beyond the particular benefits for individual species reaching community-wide level with a broader ecological perspective (Montenegro, 2004).

Although geophagy seems to offer diverse benefits for the animals, it also entails some costs. For instance, animals are more exposed to predation, poaching and hunting in salt licks since these sites are frequently visited by predators seeking easy preys (Griffiths et al., 2020; Montenegro, 1998; Varanashi, 2014) and are also important hunting sites for local people (Montenegro, 1998; Tobler et al. 2009; Blake et al. 2011) and poachers (Seidensticker & McNeely, 1975; Klaus & Schmidg, 1998; Klaus et al. 1998). Exposure to diseases is also high, since at salt licks there is a large contact between animals (Hebert & Cowan, 1971; Henshaw & Ayeni, 1971), including in some cases to wild from domesticated animals (Plummer et al. 2018). Moreover, the ingestion of clay may provoke tooth wear (Mayland et al. 1975) and soil can also contain excessive concentrations of otherwise essential minerals that would lead to mineral imbalances (Kreulen, 1985), or even contain toxic elements such as Pb, Cd, Hg, As or radionuclides (Mayland et al., 1975; Arthur & Alldredge, 1979; Kreulen, 1985). Despite all of these potential negative consequences, the animals spent high energy seeking out and visiting licks and may walk long distances and even exceed their home ranges to visit a salt lick (Tobler, 2008).

This ecological importance of salt licks by Amazonian vertebrates can provide context to seasonal changes in species occupancy and movement. Visitation rates and behaviours at salt licks might be affected by environmental variables, such as the lunar cycle and seasonality (e.g., Blake et al. 2010; Griffiths, et al. 2020). In the dry season an increased in salt lick visit was find for red howler monkeys (Blake, et al. 2010). Seasonal salt lick use could be due to differential use of habitats throughout the year, particularly as access to and movement across some regions is restricted by rising waters in creeks and rivers during the rainy season (Griffiths, et al. 2020). In this context, since, in Amazonia, salt licks are located on the edges of creeks, they are liable to periodic flooding as the level of water rises. A question we raise, therefore, is how long is this key resource available for consumption throughout the year?

In this chapter, we evaluate the temporal availability of salt licks in Amazonia and their use by the fauna through different seasons. For this, we used LEK- based methods and obtained information on 31 species of vertebrates visiting 56 salt licks in the Central Amazon. We also obtained information on the abundance rate in salt licks with different ecological characteristics, as well as on the behaviour pattern of the species visiting salt lick and on hunting practice in salt licks.

Methods

Study area

Data were collected in 9 villages in Protected Areas of Sustainable Use and in 2 villages in Indigenous lands in demarcation process: Amanã Sustainable Development Reserve (-2.19722, -64.39923); Médio Juruá Extractive Reserve (-5.22274, -67.54154); Uacari Sustainable Development Reserve (-5.74645, -

67.67361); and Igarapé Ueré, territory occupied by the indigenous people of the Kulina ethnic group.

All sampled villages were in upland *terra firme* primary rainforest. The region has a wet, tropical climate with a mean annual rainfall of 3,679 mm (2008–2010; Bauana Field Station; 5°26′19″S, 67°17′12″W). Regarding the precipitation patterns, the dry season consists of the month May to October; and the rainy season of November to April (Hawes & Peres, 2016).

Although only two seasons (wet and dry) are often used the in the literature, people interviewed for this study described the licks visitation as occurring in four different seasons: receding floodwaters, dry phase, rising floodwaters and flood pulse (wet season), being receding floodwaters the period when floodwaters recede and the area near the creeks (where the salt licks are usually found) get exposed. The dry phase, or dry season, is the period with low water levels and lower pluviometric indexes. Rising floodwaters represents the season when when floodwaters rise and the area near the creeks, therefore, begin to be covered by water. Lastly, it is called the flood pulse phase, the period of higher river water levels and rain rates.

Ethics statement

For access to Federal protected areas, this study was submitted and authorized by SISBIO (Sistema Nacional de Informação sobre Biodiversidade) (license number: 65028-2). For access to Indigenous Lands, this study was authorized by FUNAI (Fundação Nacional Indígena) (license number: 65028). For access to state protected areas, we obtain authorization from DEMUC (Departamento de Mudanças Climáticas e Unidades de Conservação), (license number:113/2019). In addition, this project is registered on Plataforma Brasil and was approved by the Research Ethics Committee of the Health Sciences Center of the Paraíba Federal University (license number:: 59846816.3.0000.5188). In order to access the area occupied by the indigenous people of the Kulina ethnic group at Ueré creek, authorization from the local leadership was used (in addition to the SISBio license), since it is not a demarcated area, FUNAI cannot give permission for entry into the place. In addition, in all the communities visited, I first requested authorization from the leaders.

Data collection

Between 2018 and 2019, we interviewed 47 local people from the sampled villages (average interviewees per village = 3, *SD* = 6) using a snowball sampling technique (Bailey, 1994) through the indication by each interviewee of another local expert on fauna visiting salt lick.

The questions asked during the interviews were:
1) What is the name given to the sites where animals go to consume soil?
2) Make a list of these sites in your hunting area.
For each site utilized by the hunter, an answer:
3) What is the distance of the salt lick from the nearest creeks?
4) In which month, on average, does the site start to be uncovered by water? And covered?
5) List the species that frequent the site.
6) What does each species do on the site?
7) For each listed salt lick and species that the interviewee feels comfortable describing its relative abundance, it was presented a logarithmic scale of relative abundance (Fig. 1) so that the hunter can point to the population abundance that he/she perceives over the four seasons for each species in each salt lick.

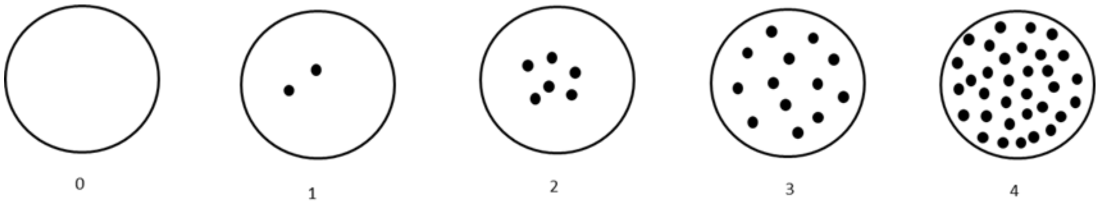

Figure 1quantitative visual scales presented during the interview for species abundance estimates, ranging from 0 (when the species population is "absent") to many specimens represented by each small circle.

Data compilation

The salt licks were divided into 3 categories according to how they are flooded by the creek water during the flood pulse season: non-flooding (salt licks that even in the peak of flood pulse season are not covered by water); Gradual flooding (salt licks which will become slowly covered by water); and rapid flooding (salt licks that became covered by water even during the rising floodwater season.

Results and Discussion

Lick classification

Through the interviews, we recorded 59 salt licks, of which 56 are natural and 3 anthropogenic, in the visited villages of Amanã Sustainable Development Reserve and Juruá basin. Considering the anthropogenic salt licks, none of them was intentionally made by the hunters (by the practice of putting salt in the soil). In these cases, two locations had oil extraction industry which provide as main waste product the called produced water, that presents a high concentration of minerals (Fakhru'l-Razi *et al.*, 2009) and so that can attract the wildlife for soil ingestion. In another case of anthropogenic salt lick, it appeared after some weeks of local people putting the fish in salt to dry on a wood drying rack. As the fish dried up, the salty liquid dripped onto the ground and an artificial salt lick started to be formatted.

In terms of the names given to the salt licks in the sampled villages, the one most frequently recorded was *chupador*, followed by *barreiro* and *canamã* for referencing sodium-rich environments, where animals go to consume soil and water. *Canamã* is the name given to a salt lick which is large in size, as well as visited by a high diversity of animals (including parrots) through the whole year because it's not completely covered by water in the flood pulse season (non-floodable salt licks). Also, canamã are often found near to the head of creeks. On the other hand, *chupador* and *barreiro* are smaller sites with lower animal diversity, and are usually covered by water during the flood pulse season (presenting rapid or gradual flooding in this season), because they are often found closer to the main part or to the mouth of creeks channel. In addition, there is also a difference between the *chupador* and *barreiro*, as *barreiro* constituting the smallest and less diverse salt lick, which is only available during the receding floodwaters period. In addition, *barreiros* are also less used by hunters. One of the interviewees, when asked to define *chupador* and *canamã*, and explain the difference between them, in a very didactic way, answered: "both serve to treat animals´ health, but while the *chupador* is a primary care centre in our village, the *canamã* is more like a hospital in Manaus".

Richness and behaviour of species visiting salt licks

We identified 31 species visiting salt licks through interviews, which perform the visit for different purposes (Table 1). Despite soil and water consumption in salt licks being the main attraction for wild species visiting these sites, some species visit salt licks for bathing, predation and other ecological relationships and behaviours. For example, Dasypodidae (armadillo) species were only seen digging burrows in the salt lick, but not consuming the soil at the site. Some other animals indirectly consume the salt lick water even though, according to the interviewees, they do not visit the place for this main purpose. For example, the *Tayassu pecari* uses the site for bathing. For this reason, the *Tapirus terrestres* does not visit the place to consume soil on the same day of the *T. pecari*. Conversely, tapirs visit the salt licks on the same day as the *Pecari tajacu*, because this animal does not stir the water as much as the *T. pecari* does. Besides, the *P. tajacu* looks for fruits and seeds in the salt licks for a short to medium time and then leaves, while the *T. pecari* spends much more time bathing there.

Table 1. Motivations perceived by interviews for animal visits in salt licks

Species	Soil ingestion	Passing	Bathing	Foraging	Predation	Gnawing tree roots on site	Other
Alouatta seniculus	X	X		X			
Aotus infulatus	X	X		X			
Ateles chamek	X						
Callicebus torquatus	X						
Chelonoidis	X	X					
Cuniculus paca	X	X		X		X	
Dasyprocta fuliginosa	X	X		X			
Mazama nemorivaga	X						
Mitu tuberosum	X			X			
Quiropteros	X						
Sapajus macrocephalus	X			X			
Tapirus terrestris	X						
Tayassu pecari	X	X	X	X			
Tinamus guttatus	X			X			

Psophia leucoptera		X			
Aramides cajaneus					X
Cacajao melanocephalus		X			
Cebus albifrons		X			
Coendou prehensilis				X	
Leopardus wiedii			X		
Leopardus pardalis			X		
Nothocrax urumutum		X			
Panthera onca			X		
Pecari tajacu		X		X	
Penelope jacquacu	X	X			
Pipile cujubi	X	X			
Potos flavus					X
Priodontes maximus	X				X
Puma concolor			X		
Saimiri sciureus		X			

Variation in abundance across the seasons

According to the interviewees, the season with the highest abundance of wild animals was the receding floodwaters season, followed by a dry phase. This is because during the receding floodwaters season the animals come from the centre of the forest to near the creeks as the water level itself decreases and the salt lick is finally exposed. However, the interviewees perceived that, during the dry season, the animals also move to places far from the creek to eat more fruits. Consequently, the visitation rate for some species may decreases in this salt licks. In the rising floodwaters season, most of the salt licks presented a low abundance for all species. During the flood pulse, the majority of the salt licks were not visited by any specimen, with the exception of *T. terresstris*, Quiropteros and *C. paca*, for which some interviewees noticed, although with low abundance, the presence of individuals of these taxa in the licks.

The diet of the *T. terrestris* is made up of fruit and foliage (Montenegro, 2004) and a main food source for the them is aguaje palm fruit (*Mauritia flexuosa*) (Bodmer, 1990; Virapongse et al. 2017). Similarly to our result, a study conducted in Maijuna-Kichwa Regional Conservation Area (in Loreto, Peru) where the aguaje palm fruits from approximately May to August (Gilmore et al. 2013) shows that possible during this time, *T. terrestris* are consuming fruit as a larger proportion of their diet and so, they visit salt licks less frequently (Griffiths, et al. 2020). However, Griffiths, et al. (2020) found a higher abundance of tapir in the salt licks during flood pulse season. Although we recorded that *T. terrestris* is one of the few species visiting salt lick during flood pulse season, we found that receding floodwaters season is the most important visit period of salt licks by *T. terrestris*. Griffiths, et al. (2020) however, considered the year as having only two seasons and the fact that the analyses were not carried out for the receding floodwater season separately of the dry season may have influenced this difference in the results.

On the other hand, our result is in accordance with a study carried out at the Madre de Dios River in Southern Peruvian Amazon, in which a very high visitation rate to the salt licks was recorded for *T. terrestris* during the dry season (Montenegro 1998). Similar results were also recorded for *A. seniculus* visiting salt licks often in the dry season, with a visit peak between June and July (Blake et al. (2011; Griffiths, et al. 2020) and *Pipela sp* also showed a visitation increase in April through May (Griffiths, et al. 2020). The higher visit rate recorded for *A. seniculus* in the dry season might be related to a shift in its diet when a greater proportion of leaves is consumed (Blake et al. 2010). In the case of *Mazama nemorivaga* the low visit rate recorded during the interviews is might because species of *M. americana* (species close to that recorded in our study) avoids flooded forest during the wet season, and those located in floodplain forest shift their diet to include woodier foods during that time due to resource scarcity (Bodmer, 1990).

At the peak of the dry season, salt lick of rapid flooding (often found closer to the main part of creeks channel or to the creek's mounth) are more visited when compared to non- floodable salt licks (often placed near to the head of creeks). This occurs because salt licks distant from the creeks become very dry during the dry season, which makes it difficult for the animals to consume the soil. On the other hand, during the flood pulse season non-floodable salt licks (*canamãs)* have higher species abundance when compared to the other salt licks. This is because, once in the flood pulse season the majority of salt licks are covered by water, non-floodable salt licks are the only to provide mineral sources for animals that depend on this key source. However, the interviewees explain that during the flood pulse season, even in these salt licks, *T. terrestris* and *C. paca* visits flow decrease because minerals concentration in the soil reduces, as a consequence of higher leaching caused by the rain (Fig. 2).

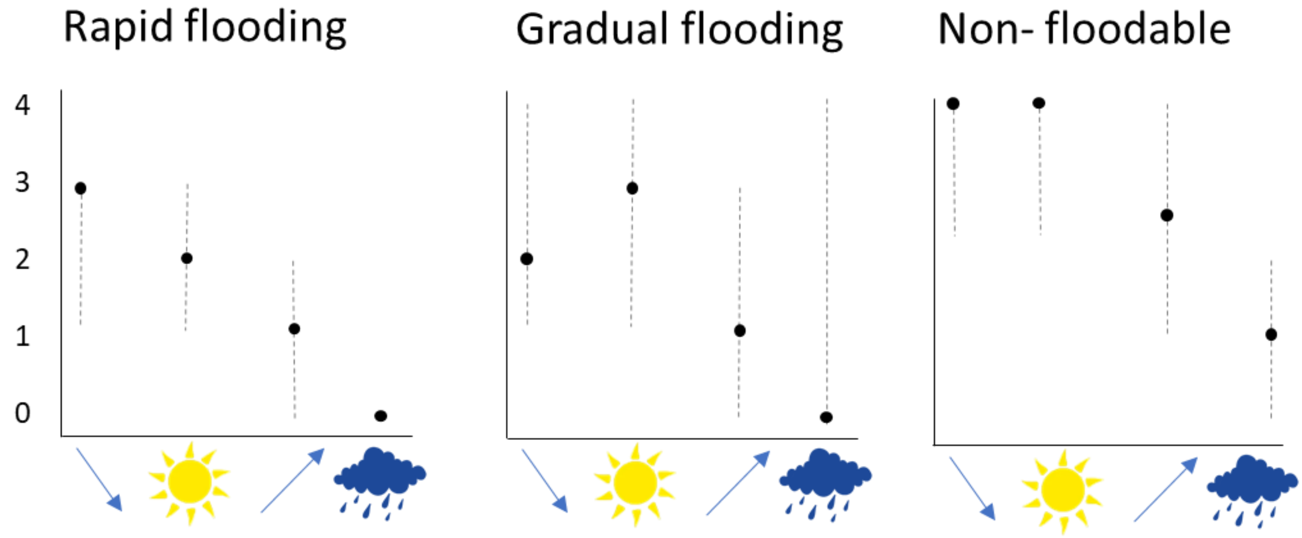

Figure 2. Abundance of species (x axis) visiting different types of salt lick as defined by flooding receding type when floodwaters recede (down arrow symbol), dry phase (sun symbol), rising floodwaters (up arrow symbol) and flood pulse seasons (raincloud symbol).

Game species visiting salt licks

The interviewed hunters listed the following species as those hunted in salt licks: *Tapirus terrestris* (VU); *Priodontes maximus* (VU); *Tayassu pecari* (VU); *Ateles chamek* (EN); *Pecari tajacu* (LC); *Cuniculus paca* (LC); *Mazama nemorivaga* (LC).

Most of the interviewees (74,46%) said that salt licks are one of the main places where they hunt. However, the hunting at salt licks is only performed during the receding floodwaters season, when it is easier to find the animals in the salt licks. In addition, as during this season it rains less, it is possible for the hunter to stay dry during the long hours of waiting in the salt lick. Also, fishing is more difficult during this season and game meat becomes the main source of protein for local residents. We hypothesise, however, that in more defaunated or regions of lower species abundance (like clearwater basins in Amazonia), the hunting practice of waiting at salt licks will be of a greater importance. That is why actively finding animals is difficult in environments of low availability of game species, and waiting at a salt lick becomes a less energy-intensive way to find and kill the target animal. Regarding that, a study conducted in the Piagaçu-Purus Sustainable Development Reserve revealed a high number of anthropogenic salt licks built by hunters (adding salt to the soil) in order to facilitate encounters with game mammals (Vieira et al. 2016). Since the Purus basin is historically more exploited than the Juruá basin and Amanã Sustainable Development Reserve, this data supports our hypothesis, which will be confirmed with future studies in additional areas. Furthermore, the practice of hunting that consists of waiting in trees whose falling fruits are eaten by animals, or in trees whose roots are gnawed by animals like *T. terrestris*, C; *paca* and *C. prehensilis* is common in lowland places with low species richness, where salt licks are uncommon.

Overall, our results showed that based on local ecological knowledge salt licks have numerous socioecological functions for many species of birds and mammals. Visits at these sites were linked to soil consumption, but also for other ecological relationships. Another important result is that the visit in different salt licks will depend not only the species needs, but also of the salt lick flooding period.

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CAPÍTULO 5

Geophagy as a new route of oil-pollution ingestion by Amazonian wild life

Photo from camera trap

Abstract

The deliberate ingestion of soil (intentional geophagy) in salt licks is a widespread behaviour that is frequently observed in herbivores and omnivorous wildlife, arguably, for mineral supplementation in nutrient-poor ecosystems such as the Amazon. Previous studies have suggested that Amazonian wildlife might be redirecting geophagy from salt licks to oil-polluted sites and, therefore, becoming an important route for contaminant exposure, posing a risk to animal's health. In here, we investigated the geophagy of oil-polluted soils through the analysis of 8,623 videos recorded from a camera trap programme in three natural salt licks and in sixteen oil-polluted sites located inoil block 192, one of the longest running oil project in the Peruvian rainforest and the most productive one in the Peruvian Amazon. We documented a total of 3,821 independent visits during 1,641 camera-days from 26 species of mammals and birds, with 66.5% of these visits displaying soil ingestion proofs from 18 different species. Considering visits with soil ingestion, *Tapirus terrestris* accounted for 62,39% of the visits, followed by *Mazama americana* (14,73%). The visit duration in natural salt licks was higher than in oil-polluted sites and in sites with lower accessibilty. Also, in sites with lower access, the proportion of diurnal visits was higher compared to nocturnal visits. However, we did not find a significant difference in the visit frequency between natural salt licks and to oil-polluted sites. These results provide relevant data to confirm that geophagy by wildlife in artificial oil-polluted salt licks is not an unusual phenomenon, but rather a widespread behaviour in the Amazon. Finally, as toxic petrogenic compounds may be bioaccumulating in animals' tissues and biomagnifying through the food chain. This points out redirected geophagy to oil-polluted soils as an important route for oil-related contaminant exposure for Amazonian wildlife and indigenous people that rely on subsistence hunting, making the consumption of oil-polluted soil a major concern for conservation biodiversity and public health.

Key-words: Camera trap; Community-based monitoring; Indigenous health; mineral licks; Oil extraction; salt licks; soil consumption.

Introduction

The deliberate ingestion of soil (intentional geophagy) by wildlife in salt licks is a widespread behaviour that is frequently observed in herbivores and omnivorous mammals and birds. Arguably, this behaviour is for detoxicate from plant secondary compounds and mineral supplementation in nutrient-poor ecosystems such as the Amazon, having each of these functions a greater or lesser degree of importance depending on factors such as species, animal reproductive period, sex and diet (Voigt et al, 2018; Klaus et al., 1998; Lee et al., 2010; Panichev et al., 2013) (Atwood & Weeks 2002, 2003; Davies & Baillie 1988; Mahaney et al. 1995; Voros et al. 2001).

Soil ingestion plays an important work in animal health, and the presence of salt licks in some regions may affect animal population density and structure (Klaus & Schmid, 1998). Salt licks are places with high concentrations of clay and essential minerals such as Na, Ca and Fe (March and Sadleir 1975, Brightsmith and Munoz-Najar 2004) and they can be naturally occurring or artificial. In the Amazon, the presence of natural salt licks is associated to young geological formations (<65 millions of yr) (Lee et al, 2010). Regarding artificial salt licks, their intentional creation is often performed by farmers for their cattle, horses and other herbivores to encourage health growth and development (Lameed and Adetola, 2012). In addition to livestock, artificial salt licks are used to attract wildlife for hunting, ecoturism (eg.: wildlife watching) and for wildlife conservation and management (Simpson et al, 2020; Ang and Chan, 2010; Lim and Mojiol, 2020). However, it is worrying that mining and soil polluted have been also used as artificial salt licks by the animals (Sheppard,1998; Orem, 2019). For example, overburden from the abandoned coal mine have been used by Denali caribou (Rangifer tarandus) as a lick in Alaska (Boertje, 1981) and).oil-polluted sites have been used by Amazonian fauna in the Peru (Orta-Martínez et al 2018).

It has been suggested that the disposal of produced water, the main by-product of oil extraction industry that can have a very high salinity (up to

190,000 mg/L), might attract wildlife to these oil polluted sites (Orta-Martínez et al 2018). However, produced water can also contain high concentration of a number of potentially toxic agents, including radioactive isotopes, dispersed hydrocarbons (i.e. phenolic and polyaromatic molecules among others), and heavy metals (i.e. cadmium, chromium, lead and barium among others) (Fakhru'l-Razi *et al.*, 2009). Many of this compounds are mutagenic and carcinogenic and bioaccumulate (Monteiro et al, 2016; Vaikosen et al, 2014). In fact, high average concentration of lead (0.49 mg kg−1 wet weight) were reported in livers from wild game in oil extraction areas from the Northern Peruvian Amazon and their lead isotopic fingerprints indicated that produced water is a major source of lead for wildlife (Cartró-Sabaté et al., 2019). These results uncovered important health risks from geophagy in oil-polluted soils to tropical wildlife and local communities that rely on subsistence hunting.

Produced or formation water originates from the oil and gas reservoirs and is brought to the surface during oil and gas extraction operations (Long et al., 2013; UNEP Technical Publication & E&P Forum, 1997). Although many regulations for onshore oil operations prohibit entirely the untreated discharge of produced water and require its reinjection back to the reservoirs, oil companies have often discharged them onto land and surface waters in Low and Middle Income Countries (LMICs) for several decades (Jernelöv, 2010; Orta-Martínez, in press). Oil exploration in the western Amazon started as early as the 1920s in Ecuador (Sawyer, 2004) and the late 1930s in Peru (Orta-Martínez 2010), reaching its production boom in the 1980s (Finer 2010). In the subsequent four decades numerous large projects have been developed in Amazon, such as the *ITT project* in Ecuador, the Urucu gas project in Brazil and the Camisea gas project in Peru. It should be noted that ~733,000 km^2 of rainforest in the western Amazon are covered by oil and gas blocks (Finer et al., 2008). To make it even worst, 39.4% of the Amazonian rainforests overlap with conventional oil and gas reserves, a percentage that resembles the ~30% of the estimated worldwide rainforests overlapping with oil and gas reserves (Orta-Martínez in press). In addition, many of the oil and gas blocks overlap indigenous territories, both titled and non-titled lands, as well as areas utilized by peoples in voluntary isolation (Finer et al., 2008).

Nowadays, oil and gas blocks cover more than two-thirds of the Amazon in Ecuador and Peru, and major exploration activities are set to increase rapidly in Bolivia and western Brazil (Finer et al., 2008). Moreover, this overlap is expected to increase since Amazonian countries are strongly promoting hydrocarbon exploration, as a consequence of the globally growing oil demand (Finer et al., 2008, 2015). Indeed, oil demand increased from 85.3 to 96.6 million BPD between 2006 and 2016 (BP, 2017), pushing the hydrocarbon frontier towards more remote territories and towards unconventional sources whose exploitation is generally associated to higher costs, risks and impacts (Orta-Martínez & Finer, 2010).

Oil and gas blocks are currently filling these remote areas and increasing the concern that the new oil and gas projects could bring a proliferation of new access routes throughout the western Amazon (Sierra, 2000). Increasing access and the consequent integration of indigenous people to the market economy, which would result in over hunting (Laurance et al, 2009). Similarly, the noise pollution produced by oil and gas operations may be a relevant environmental stressor for wildlife behaviour, ecology and physiology (Francis & Barber, 2013). In here, we present the results of a 3-years camera trap programme to study geophagy of oil-polluted sites in a major oil concession in the Peruvian Amazon. To do so, we compare natural salt licks and artificial oil-polluted salt licks, describing the taxa and the behaviour of the wildlife visiting both types of salt licks, including visit frequency, daytime and duration, and ingestion evidences. Finally, we assessed the effect of the noise disturbance and accessibility on the behaviour of wildlife in each salt lick.

Methods

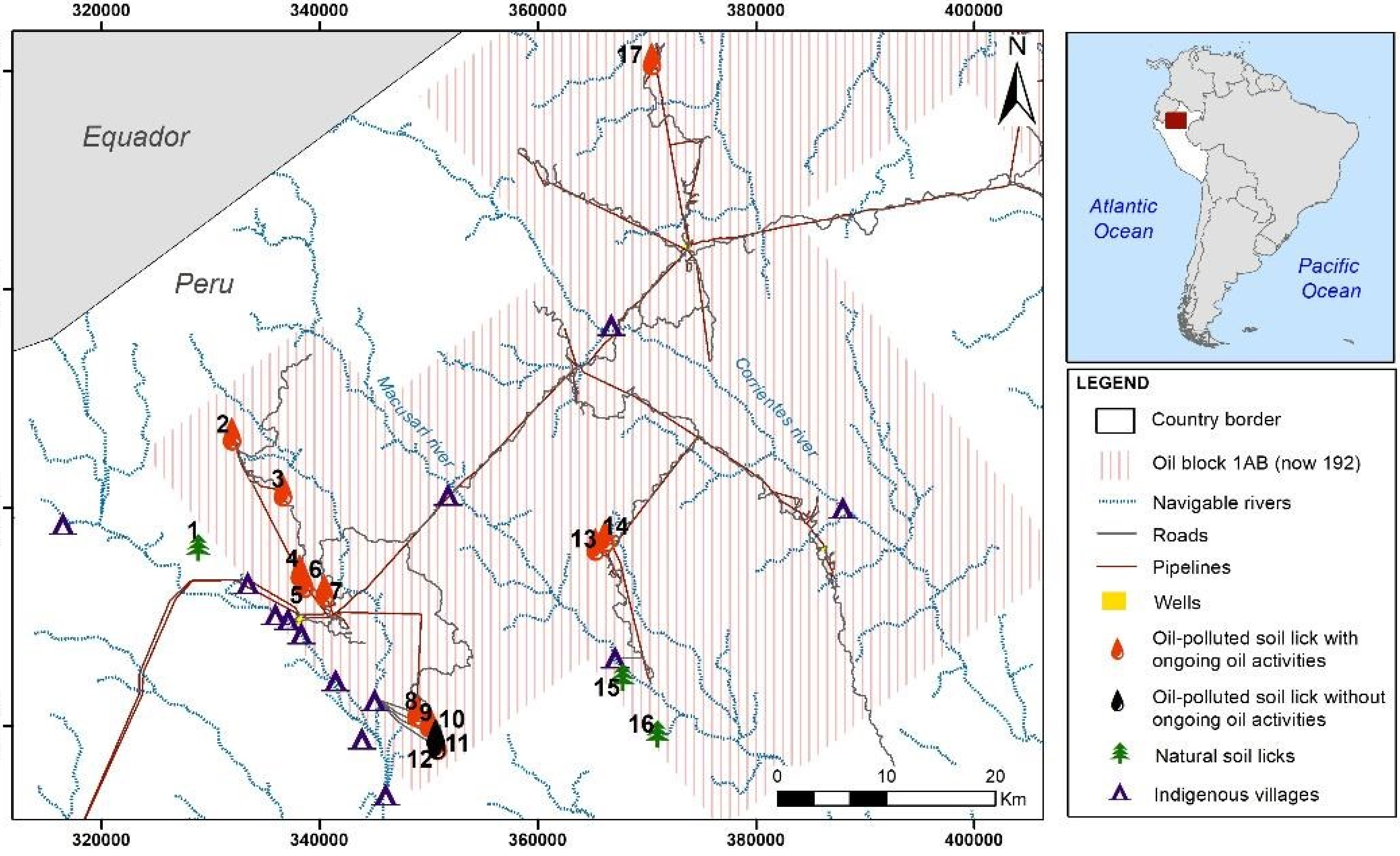
Study area

This study was conducted in 19 natural (n=3) and artificial (n=16) salt licks located in the Corrientes and Pastaza River basins in the Northern Peruvian

Amazon, the ancestral territories of the Achuar and Kichwa indigenous people. Energy corporations have extracted oil from these area, the oil block 192 (formerly block 1AB), since the early 1970s (O'Callaghan-Gordo et al, 2021). This oil block, together with the adjacent oil block 8, have come to be one of the longest running oil project in the Peruvian rainforest and the most productive one in Peru (Orta-Martínez & Finer 2010). A total of 7,090 million barrels of produced water have been discharged into the rivers of the area and 2,014 oil-polluted sites have been reported by the operating oil company in block 192 (Orta-Martínez in press), resulting in severe environmental pollution and adverse impacts to public health. Because of that, on 2 May 2005 the Achuar people from the Corrientes River filed a petition asking to urgently conduct analysis to verify the presence of heavy metals and prevalence of oil-related diseases in people of Trompeteros district (FECONACO, 2005). As a result, alarming values were found, showing that 99.20% and 79.20% of adults exceeded the acceptable limits for cadmium and lead in blood, respectively (DIGESA 2006). After more than one year of the results publication an outcome was codified in what is known as the Dorissa Accords, that includes a modification of the environmental management programs to reinject all the produced water in blocks 1AB/192 and 8 by 31 December 2007. However, this commitment was only applied to Achuar territories and the Corrientes River within it (Orta-Martínez, 2018).

The salt licks included in this study are inside or only x kilometres away 1 natural salt lickfrom the actual limits of the oil block 192 (Fig. 1). All oil-polluted salt licks are located close to ongoing or abandoned oil infrastructure and were selected by local indigenous environmental monitors based on an *in situ* organoleptic assessment (Cartró-Sabaté, 2019) and subsequently confirmed by chemical detection of steranes and hopanes (Rosell-Melé et al., 2018). The presence of hopanes and steranes in environmental samples indicates the occurrence of oil derived products (Volkman et al., 1997; Wang et al., 2006) (see details of soil analysis in Supplementary material S1).

Around 10,000 Indigenous Achuar, Quechua and Kichwa people inhabit this oil block. The subsistence activities of these indigenous communities include hunting, fishing and small-scale agriculture.


320000
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9740000
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9700000
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Equador
Peru
Macusari river
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1
2
3
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9
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11
12
13
14
15
16
17
0
10
20
Km
Atlantic Ocean
Pacific Ocean
LEGEND
Country border
Oil block 1AB (now 192)
Navigable rivers
Roads
Pipelines
Wells
Oil-polluted soil lick with ongoing oil activities
Oil-polluted soil lick without ongoing oil activities
Natural soil licks
Indigenous villages

Figure 1. Map of the study area showing the 19 salt licks studied in the Corrientes and Pastaza River basins in the Northern Peruvian Amazon. Map generated using ArcGIS 10.3.1; Datum: WGS84 Source: ESRI,.

indigenous They rely on and,

Data Collection

Between July 2013 and October 2015, a participatory soil and camera trap survey were conducted in cooperation with a locally run environmental monitoring programme , which has played a crucial role in the identification, documentation, and mapping of oil activity impacts since 2005 (Orta-Martínez, 2010).

Soil sample collection and analysis

Soil samples from all salt lick studied were also collected. In each salt lick, three soil replicates separated by 1.5 m, removing the superficial soil (0-20 cm depth), were collected using a methacrylate tube of 7 cm diameter. after removing the superficial dead leaves. Each subsample was sent at ambient temperature to the laboratory and stored in the freezer upon arrival. Soil samples were classified as natural or oil-polluted according to visual and olfactory *in situ* inspection and chemical detection of sterane and hopane residues (Rosell-Melé et al., 2018).

Petroleum biomarkers analysis

To determine the presence of oil-related pollution in the collected soils, the occurrence of petroleum biomarkers (sterans and hopanes) was analysed in the Environmental Forensics Laboratory from the ICTA-UAB. The presence of hopanes and steranes compounds in environmental samples indicates the existence of oil derived products (Volkman et al., 1997; Wang et al., 2006). The determination of hopanes and steranes compounds was carried out using the analytical method of gas chromatography–mass spectrometry (GC-MS). First, the oil residues from soil samples were extracted. To remove high-boiling compounds that are not eluted from the GC column and may influence the performance of the instrument (Albaigés et al., 2015), the hydrocarbon fractions were separated by column chromatography. Extracts were injected in a GC-MS instrument, where selected ion monitoring was used to retrieve the distribution of hopanes and steranes from the whole GC-MS chromatograph. In this case, the monitoring of ions with a mass-to-charge ratio (m/z) of 191 was selected to obtain the distribution of hopanes, and 217 and 218 for steranes. The whole analysis was performed for the seven soil samples, two blancs and one reference sample. Squalene (25 ng/µg) was added as internal standard. The extraction of the samples with organic solvents in a microwave (MarsX-CELL) was done using 10 Teflon digestion vessels that were previously cleaned using the same solvent. 5 g of soil for all the samples were extracted, except for soil samples from sites 6, where only 2 g were used, because of the appreciable higher oil pollution load. 25mL of trace analysis grade n-hexane–acetone (1:1, v/v) (Merck, Darmstadt, Germany), a magnetic stirrer and 150 mL of internal standard were added to each vessel. The extraction method started with a 12 minutes temperature ramp from room temperature to 115 ºC and then it kept this temperature for 2 minutes. After the extraction, the vessels' content was transferred to 50 mL test tubes for its centrifugation. Samples were centrifuged during 10 minutes at 2200 rpm in a centrifuge (Rotofix32-Hettich) and the supernatant liquid was transferred to 50 mL pear bottom flasks. Approximately 2mL of hexane were added to each test tube, which were agitated in a pulse-vortexing and then centrifuged again. This process was repeated three times in order to accurate as much as possible the collection of extract. Then the extract was evaporated to 0.5 mL using a rotary evaporator (Büchi Heating bath B-490 and Büchi Rotavapor R-200). The extracts were 61 fractionated by adsorption chromatography with glass columns containing 2.5 g of silica (Scharlau, Barcelona, Spain), 2.5 g of aluminium oxide (Sigma–Aldrich, St. Louis, USA) previously activated at 110 ºC and deactivated 5% with ultrapure water (MiliQ/Millipore, Cork, Ireland), and 1 g of sodium sulphate (Merck, Darmstadt, Germany). The first fraction of eluate was collected corresponding to aliphatic hydrocarbons eluting in 6 mL of n-hexane. The extracts were concentrated first by rotary evaporation and finally with a gentle stream of N2 to near dryness. The biomarkers identification was carried out in an Agilent 7890A gas chromatograph (GC) coupled to an Agilent 5975C mass spectrometer (MS) operated in electron impact ionization mode (70 eV) and equipped with a 30 m x 0.25 mm x 0.25 µm DB-5ms capillary column (J&W Scientific, CA, USA) and a 5 m guard column. The instrument was operated in splitless mode. The oven temperature program started at 60 ºC (held for 1 min) then increased to 320 ºC at a rate of 4 ºC min-1 and held for 10 min. Injector, transfer line and ion source temperatures were 310 ºC, 320 ºC and 250 ºC respectively. Helium was used as the carrier gas at constant flow of 2 mL min-1 . The MS was used in both scan mode and single ion mass mode at a time, monitoring m/z= 191 for the hopanes and m/z= 217-218 for the steranes.

Camera trapping survey

One camera trap (Bushnell 8MP Trophy Cam HD I model 2) was placed one meter above ground surface at each salt lick. Cameras were set in video mode and they recorded one minute when triggered by the infrared motion-and-heat detector. A period of one minute of inaction between videos was established and cameras were continuously activated until batteries were discharged. Batteries were replaced as soon as possible. Thus, camera traps were not active for all the study period in all salt licks (Supplementary material, Fig 1), and the total data collection effort was 1641 camera days.

Video data compilation

7858 videos were recorded during 1,641 camera-days in 16 oil-polluted salt licks (6018 videos) and 3 natural salt licks (1840 videos). From all videos obtained, 5,961 (75,8%) correspond to animal records, 778 (9.9%) videos were triggered by rain or, probably, by abrupt light changes, 622 corrupted videos (7.9%), and 118 videos (1,5%) were triggered by people (usually monitors or hunters). All recorded videos were labelled with site id, camera id, video id, date and time. We extracted the information above using *Timelapse software* (Greenberg & Godin, 2015). Species, number of biological units per species per record, sex, daytime (diurnal/ nocturnal/crepuscular) and ingestion of soil and water were identified per each video by 2 scientists. Biological unit was defined as one individual when considering solitary species and a group of individuals when considering social species. Chiroptera were identified at the order level, but they were not considered for analysis.

Several variables were calculated per each salt lick based on the data mentioned above :

- 'Independent visits':Each independent visit was defined as videos recorded in an interval of one hour (Tobler et al., 2008). If several videos were recorded in an interval below one hour, we consider a new visit when: i) a new species appears, ii) no animal appears (eg.: videos triggered by rain or abrupt light changes), iii) an animal of the same species but of different sex appears, iv) in the same video more than one specimen of the same species appears, if dealing with a solitary species. In that case, each individual in a different visit.
- Species' visit frequency: calculated as the total of independent visits per species divided by the number of active camera-days in each salt lick.
- Ingestion evidence: independent visits with recorded acts of chewing, eating, drooling or licking soil and water, or licking the oil infrastructure .
- Visit duration: time between the first and the last recorded video of an independent visit (following Link et al. (2011). As this formula could overestimate visit duration in cases in which there is a long (but < 1 hr) gap between videos, we calculated visit duration only for species with large enough sample size (>30 videos), in this case for 11 species (Griffts et al., 2018) (see details in Supplementary material S3). In the species studied, visits composed by one video were assigned a duration of two minutes, since that was the minimum delay for the camera's trigger.
- Daytime activity patern: percentage of diurnal, nocturnal and crepuscular visits per specie and salt lick. Crepuscular visits were defined as visits occurring from 5:30h to 6h30min and 17h30min to 18:30h, diurnal visits, from 6h30 to 17h30min and, nocturnal visits, from 18h30min to 5:30h (Griffts et al., 2018). We just considered the activity partner only for

species with large enough sample size of recorded visits (>30 videos), with exception of *Ara chloropterus* and *Ara macao,* for which we could not access the correct time of their visits.

Noise disturbance and salt lick access

To further study the impact of oil activity on Amazonian wildlife, we also examined the effects of oil operation noise and hunting access provided by oil roads and infrastructure. Euclidean distance of each salt lick from the nearest active oil infrastructure (i.e. pipelines, roads, wells and production facilities) was measured using ArcGIS 10.3.1 as a proxy for "noise disturbance". (Fig 1; Supplementary material, Table 1). Travel time of each salt lick from the nearest indigenous village was used as a proxy for "hunting access", as previous studies have reported the strong relation between hunting pressure and travel distance from villages (Vickers, 1991; Parish, 2001). Travel time was calculated adding road, river (Strahler order> 3) and forest distance and considering walking, boat and car/motorcycle speeds (4, 10 and 30 km/h, respectively) (Fig 1; Supplementary material, Table 1) (Gleyzer et al., 2004).

Statistical analysis

To compare the difference of visit frequency and species richness between natural and oil-polluted salt licks, we used generalized linear mixed models (GLMM) with negative binomial distribution for: i) all species, ii) species ingesting soil, iii) each species. We considered visit frequency values as a response variable and the type of salt lick (natural or oil-polluted salt lick) as a predictor variable. Considering that the species distribution in the area and the structure of the salt lick might affect the number of species visiting the studied salt licks we considered the species ID as a random variable. In addition, in this type of model the effect of the number of oil-polluted salt licks being greater than the number of natural ones is also controlled.

We also calculated the proportion of diurnal visits compared to nocturnal visit through the reason of the number of diurnal/nocturnal visits per species per salt lick.

GLMM with the negative binomial distribution was also used to assess relationships between oil infrastructure disturbance (“noise disturbance” and “hunting access”) and i) visit frequency, ii) visit duration, and iii) number of diurnal/nocturnal visits . In these models, the type of salt licks (oil-polluted or natural) and the species ID were used as a variable of random effect in the models, to reduce bias regarding the activity partner particular of each species and of the number of species visiting each particular salt lick.

Residual checks were used to verify model suitability. We used the Akaike information criterion to select models of interest if ΔAIC values >6 (ΔAIC obtained from the difference between a null and complete model AIC values (Harrison et al., 2018; Richards, 2008). All analyses were performed in R ver. 3.5.3 (R Development Core Team, 2019). GLMMs were based on the MuMIn e lme4 packages (Oksanen et al., 2013) and MFAs, on the *FactoMineR* package (Husson et al., 2018).

Results

A total of 3,821 visits from 26 species have been documented, 2,979 visits from 24 species in oil-polluted salt licks and 843 visits from 15 species in natural salt licks.

In 66.5% of these visits (2,541), ingestion evidences were recorded from 15 different species, 15 species in oil-polluted salt licks and 7 in natural salt licks. We recorded in average 3.9 species ingesting soil (range 1-8) in each salt lick , 4.0 (range 3-8) per oil-polluted salt lick, and 3.7 (range 2-6) per natural salt lick, but this difference was not statistically significant (Fig 2; Supplementary material, Table 2).

Tapirus terrestris accounted for 62.39% (n=2,384 visits) of the visits, followed by *Mazama americana* (14.73%, n=549), *Ortalis guttata* (5.23%, n= 200), *Cuniculus paca* (3.71%, n=142), and other species 14.28% (n=546). , *Tapirus terrestris* accounted for 69.65% (n=1,770) of the visits with ingestion evidences, followed by *Mazama americana* (13.65%; n=347), *Ortalis guttata* (6.53%, n=166), *Patagioenas cayennensis* (2.75%, n=70), and other species 7.39% (n=188).

Иsit frequency was 5.07 visits/camera day (ranging from 0.56 to 31.61). When considering all species, The visit frequency in natural salt licks (9.6; ranging from 3.54 to 21.55) was higher than in oil-polluted salt licks (4.21; ranging from 0.56 to 31.61), but this difference was non-significant (p= 0.0958). However, when analysing each species separately, we found the visit frequency of *Tapirus terrestris* was significantly higher in natural salt licks than in oil-polluted salt licks (p=0.02985). The abundance of *Ortalis guttata* was also higher in natural salt licks, however this difference was not significant. For all the other species visiting both oil-polluted and natural salt licks, oil-polluted salt licks showed a non-significant higher visit rate when compared to natural salt licks (Fig 2, Supplementary material, Table 3).

The visit frequency in oil-polluted salt licks in abandoned oil-infrastructures, was significantly higher, when compared to active oil-infrastructures (p= 0.00851), and trended to be higher than in natural salt licks (p=0.36343). The visit frequency was higher for salt licks more distant of active oil-structure (proxy of noise disturbance) (p<2e-16). Distance from indigenous villages (proxy of “hunting access”), did not show a significant effect on visit frequency (p=0.1073; Table 1).

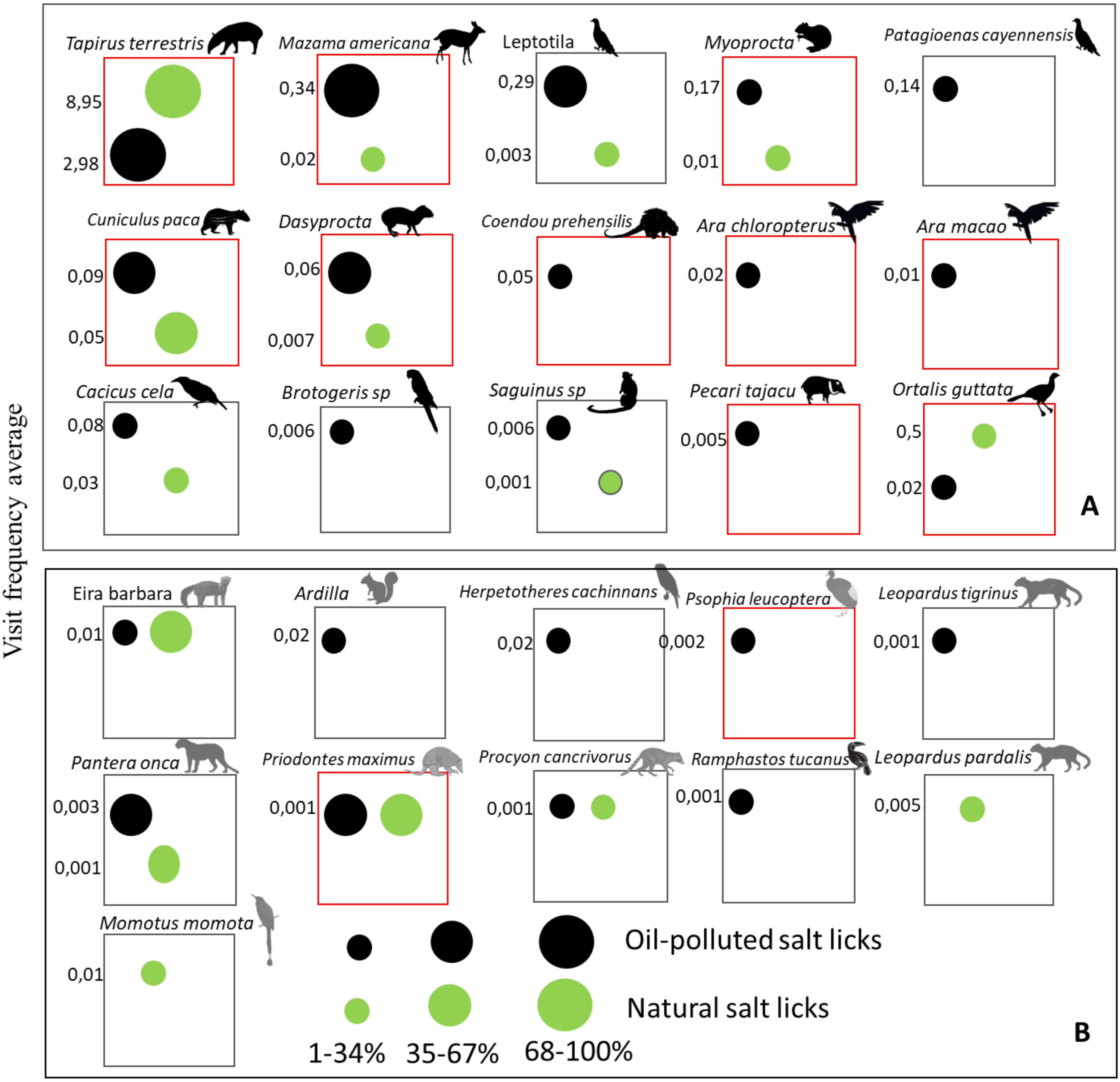


Figure 2. Average visit frequency (in visits/camera day) of (A) species recorded in salt licks consuming soils and waters, and (B) species recorded visiting in salt licks but not consuming soils and waters. Symbol sizes are proportional to the percentage of salt licks in which the species was identified in oil-polluted (n=16) and natural (n=3) salt licks. Game species for subsistence hunting are

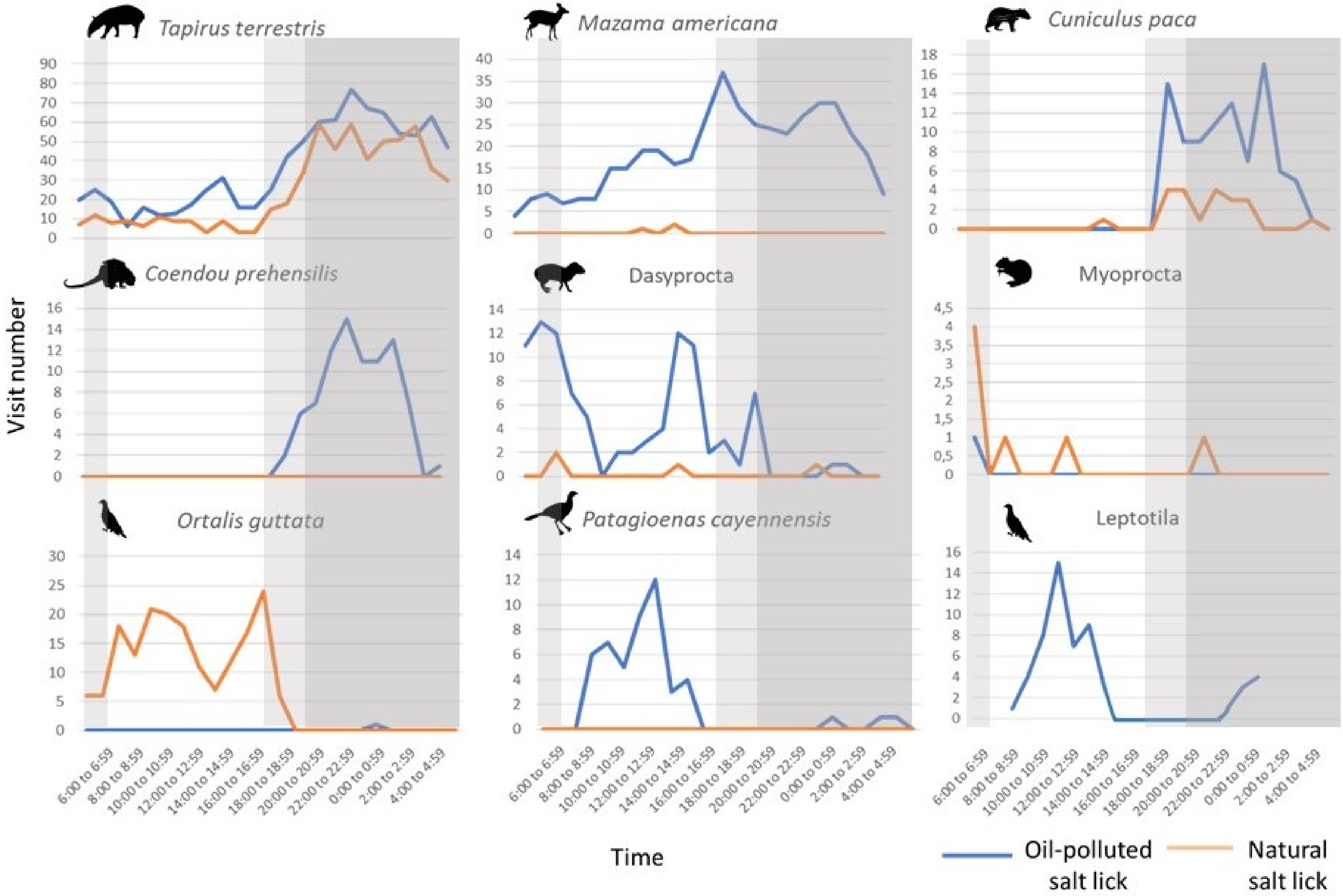
shown in red boxes (Peres, 2000). Species are ordered left to right and top to bottom from the highest to the lowest visit frequency in oil-polluted salt licks.

Table 1. Models using GLMMs to examine the effects on species visit frequency of type of salt lick, distance from active oil activities and distance from indigenous villages. Estimate represents the amount by which each response variable would increase if each explanatory variable were one unit higher. Z-values indicate the degree to which explanatory variables exert a significant effect. Pr (>|z) denote significance levels as following: ns P > 0.05; ** P ≤ 0.01; *** P ≤ 0.001. AIC Akaike Information Criterion; ΔAIC difference of AIC with respect to the selected model in comparison to null model.

	Response variable	Predictors	Estimate	Std. Error	z value	Pr(>|z|)		AIC	AIC null model	Δ AIC
model 1	Visit Frequency	1.Natural salt lick vs Oil-polluted salt lick in abandoned oil infrastructures	-0.8837	0.9723	-0.909	0.36343			213.9	6,20
		2.Oil-polluted salt lick in active oil-infrastructures vs Oil-polluted salt lick in abandoned oil-infrastructures	-21.510	0.8175	-2.631	0.00851	**			
model 2	Visit Frequency	Distance from active oil-infrastructures (noise disturbance)	0.491259	0.006207	79.14	<2e-16	***	207.7	213.9	6,20
		Distance from indigenous village (hunting pressure)	0.6162	0.3827	1.610	0.1073				

Overall, 66.9% of the species for which daytime activity patterns were analysed exhibited nocturnal activity, 30.1% exhibited diurnal activity, and 2.9% crepuscular activity. We found a high variation of the activity partner among the species (Fig 3; Supplementary material, Table 6). The number of diurnal visits compared to nocturnal visits decreased significantly with the distance from indigenous communities (p=0.0033) and increased with the distance from oil infrastructure (p= 0.00036) (Supplementary material, Table 7 and 8). We did not find difference in the activity partner when comparing the visits in oil-polluted to natural salt licks.

Average visit duration was 11.03min (max=60; min=2) and increased significantly with the distance from indigenous communities (E=- 0.33691; p= 5.21e-09) and the distance from oil infrastructure (E=- 0.12100, p=0.00441). In addition, visit duration was significantly longer in natural salt licks (average=10.3min, max=60min, min=2min) when compared to the oil-polluted ones (average =6.1min, max= 52min, min=2min) (p=7.17e-08). *Tapirus terrestris* had a higher visit duration in both natural (13.95 min) and oil-polluted (10.79 min) salt licks when compared to the other species (Supplementary material, Table 9).



Tapirus terrestris
Mazama americana
Cuniculus paca
Coendou prehensilis
Dasyprocta
Myoprocta
Ortalis guttata
Patagioenas cayennensis
Leptotila
Visit number
Time
6:00 to 6:59
8:00 to 8:59
10:00 to 10:59
12:00 to 12:59
14:00 to 14:59
16:00 to 16:59
18:00 to 18:59
20:00 to 20:59
22:00 to 22:59
0:00 to 0:59
2:00 to 2:59
4:00 to 4:59
Oil-polluted salt lick
Natural salt lick

Figure 3. Daytime activity pattern the different species with more than 30 visits (with exception of *Ara chloropterus* and *Ara macao*).

Discussion

There is some concern about the impacts of noise produced during seismic oil exploration on wildlife behaviour in tropical rainforests (Rabanal 2010; kolowski 2010, 2012). To our knowledge there are no studies regarding the effects on wildlife activity and abundance from noise disturbance of oil extraction activities. The wide range of noise types and intensity from oil extraction operations, noise pollution may be a relevant environmental stressor.

Geophagy in natural salt licks among several species have been related to their need to overcome nutritional deficiencies, to alleviate digestive disorders, and/or to increase their buffering capacity (Emmons and Stark, 1979; Kreulen, 1985; Klaus and Schmidg, 1998; Blake *et al.*, 2011). Our results confirm that geophagy in oil-polluted sites is a widespread behaviour in oil extractive areas in the Amazon. We identified 26 species visiting these oil-polluted salt licks, 16 of them repeatedly ingesting oil-polluted soils and waters during a total of 2,541 visits, in 16 different oil-polluted salt licks over 1,641 camera-days. All the species recorded ingesting soils and waters in the oil-polluted sites are frugivorous and herbivorous, and have been frequently observed in natural salt licks (Blake *et al.*, 2011; Varanashi, 2014).

In the oil-polluted salt licks included in this study, we found a high values (µg/g) of heavy metals (in accordance to XXXX citar fonte onde posso encontrar valores seguros de metais pesados), including Ba (667,9 ±1185,6), Cd (61,7 ± 229,9), Hg (0,9 ± 3,2) and Pb (23,7 ± 10,7) (Braga-Pereira et al in press). High concentrations of petrogenic pollutants (petroleum hydrocarbons and/or oil-related heavy metals) have been reported in these oil-polluted salt licks, confirming that redirection of intentional geophagy to oil-polluted sites is a new and important source of exposure to petrogenic compounds for wildlife. Previous research showed that some vertebrate species visiting salt licks ingest large quantities of soil (up to ~30% of digesta) (Beyer *et al.*, 1994; Hui, 2004). The ingestion of soils has already been pointed out as an increasingly important route for contaminant exposure for both livestock and free-ranging wildlife in industrialized countries (Weeks & Kirkpatrick, 1976; Weeks, 1978; Arthur & Alldredge, 1979; Fries, 1982; Fries *et al.*, 1982).

As oil-polluted soils contain toxic and carcinogenic compounds, such as heavy metals and hydrocarbons (Rosell-Melé et al., 2018; O'Callaghan-Gordo et al., 2021), its ingestion might also be an even dangerous exposure for pollutant contamination. In addition, and the ingestion could potentially result in bioaccumulation and biomagnification of hydrocarbons and heavy metals, and affecting even top predators in the tropical rainforests (Jorgensen & Fath, 2008). In fact, as previously observed in natural salt licks (Montenegro, 2004; Blake *et al.*, 2011; Varanashi, 2014), the oil-polluted and natural salt licks of our research are also visited by predators, such as the felines *Leopardus tigrinus, Leopardus pardalis* and *Panthera onca*.

Moreover, we identified 10 game species, including the 4 species most frequently consumed in indigenous people's diets (*Tapirus terrestris, Cuniculus paca, Mazama americana* and *Pecari tajacu*) (Bodmer & Lozano, 2001), consuming oil-polluted soils. Amazon societies depends on subsistence hunting as major source of protein and income (Bizri et al., 2020). Thus, local human populations might also be exposed to petroleum through the wild meat consumption. Regarding on that, high blood lead levels were detected among indigenous people living in the studied area (the largest onshore oil extracting area of Peru). The highest levels were found among participants from the Corrientes River basin, where most of the oil extraction activities were concentrated and the highest amount of produced water had been released.

The risks posed to human health from wildlife geophagy in oil-polluted sites might reach beyond the local communities as wild meat trade to urban markets has been reported in this and other oil concessions in the global tropical rainforests (Orta Martínez *et al.*, 2007; Suarez 2013). This risk is even more potentiated, since oil roads constructed by oil companies increases access and boosts wild meat trade from oil concessions (Suarez 2013). In Iquitos (one of the largest wild meat market of the entire Amazon), six species which are reported consuming oil-polluted soils in this study (*Pecari tajacu, Cuniculus paca, Dasyprocta fuliginosa, Mazama americana*, *Tapirus terresris* and *Psophia leucoptera*) represent 51.8% of the total wild meat sold (Mayor et al, 2021).

In general, the frequency of visits in natural salt licks tended to be higher than in oil-polluted sites. However, when comparing only natural salt licks to oil-polluted salt licks in abandoned infrastructure, we found a tendency for the frequency of visits to be higher on oil-polluted sites, which might confirm our hypothesis that the noise from oil-infrastructure is affecting the behaviour of the species too. The major concern here is that leakage from abandoned oil wells it is an unresolved problem, because a high proportion of seals placed in wells may be faulty (Davies et al., 2014). In addition, an important fraction of oil spilled (i.e., components with alkanes and PAHs) will remain in the soil micropores for years due to their resistance towards biodegradation (Yavari et al., 2015).

Considering that the main difference between oil-polluted salt licks in abandoned and in ongoing infrastructure is the presence of noise, and that in salt licks near to active oil-structures, we observed lower visit frequency and shorter visit duration, we highlighted that noise disturbance is another important impact of oil extraction activities on wildlife.

There is some concern about the impacts of noise produced during seismic oil exploration on wildlife behaviour in tropical rainforests (Rabanal 2010; kolowski 2010, 2012). To our knowledge there are no studies regarding the effects on wildlife activity and abundance from noise disturbance of oil extraction activities. The wide range of noise types and intensity from oil extraction operations, noise pollution may be a relevant environmental stressor.

Oil road network drastically changes access to previously remote salt licks in the rainforests. For example, the average travel time spent to reach the natural salt licks from an indigenous community through the roads constructed by the oil-extraction industry is 1.7 hours (ranging from 0.3 to 2.2). While without these roads the average expend time would be much lower. In addition, the average travel time spent to reach the studied oil-polluted salt licks from an indigenous community is 0.8 hours (ranging from 0.2 to 1.3), that is, with even greater access than to natural salt licks. Além do maior acesso a áreas que concentram espécies cinegéticas (como é o caso de salt licks), nós encontramos que a presença de indústrias petroleiras afeta também o comportamento das espécies. Por exemplo, we found a significant higher proportion of nocturnal visits and a shorter visit duration in salt licks with greater access. In addition, although it was not significant, we also found a trended of a lower visit frequency in salt licks with greater access.

Over the last decade, there has been a huge controversy in trying to estimate the area affected by oil extraction activities in the western Amazon. In 2010, the Peruvian Ministry of Environment declared that only 20.37 Km^2 (0.40%) of the 1AB/192 oil concession was impacted by oil activities. In the calculations, it was only taking into account the area where the installations were placed (MINAM, 2010). Considering the mobility of the species recorded in oil-polluted salt licks, it can be argued that the area exposed to oil pollution is much larger. In view of the mobility of the most frequent visitor to the studied sites *Tapirus terrestris* (Tobler, 2008), we concluded that up to 5,820 Km^2 and up to 83.47% of the area occupied by the oil block might be exposed to oil pollution (Cartró-Sabaté, 2018). Moreover, the oil concession overlaps with Indigenous communities who largely depend on subsistence hunting, and whose hunting grounds are mostly located inside the oil concession limits (Fernández-Llamazares et al., 2019).

The redirection of geophagy to oil-polluted soils by wild Amazonian species might be an important exposure route to highly toxic, carcinogenic, and mutagenic compounds associated with the oil industry. To make it even worst, these compounds can bioaccumulate in animals' tissues and biomagnify through the food chain, posing at risk the health of top predators and, , local human populations that rely on subsistence hunting. Considering that oil hydrocarbon reservoirs overlap with 30% of world tropical forests (Orta-Martínez et al., 2018), the relevance of the widespread behaviour describe here exceed the frontiers of the study region. Indeed, oil and gas projects are the primary threat to remote and well-preserved areas in the eastern Ecuador (blocks 31 and ITT in the Yasuní National Park and Biosphere Reserve), the northern Peruvian Amazon (blocks 39 and 67 in the proposed Napo-Tigre Territorial Reserve for indigenous people in voluntary isolation), the Brazil's Urucu region (**block name and natural protected area name**) and Bolivia's Madidi region (**block name and natural protected area name**), among many others. The use of sub-standard technologies in LMICs (i.e. discharge of produced water and reocurring oil spills) adds a further twist to this concern for tropical conservation and public health. The improvement of operational practices or a moratoria on oil and gas extraction in areas that overlap with highly biodiverse regions are crucial to prevent environmental harm in these critically important ecosystems and to protect the health of local indigenous communities.

Around 1 million barrels/day of produced water have been directly released on soils and rivers in the study area, between the beginning of oil extraction and 2010, when re-injection of produced water back to the oil reservoir was implemented in the area (Orta Martínez et al., 2007). This discharge has led to an increase of 12% and 20–30% in sodium and chloride concentrations in the Amazon river (Óbidos, Pará), thousands of kilometers downstream in Brazil (Yusta-García et al., 2017). The dumping of produced water on the environment has been a common practice in oil operations in tropical countries, where the oil industry uses substandard technology not in accordance with the state-of-the-art employed in its home countries (Jernelöv, 2010). Moreover, worldwide,

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Supplementary Material

Table 1. Information on sampled salt licks

Salt lick code	X	Y	Nearest indigenous village	Distance from the village (km)	Travel time (hours)	Distance from ongoing oil activities (km)	Category
1	329303	9696322	Wararai	16,13	2,01625	2,9	natural salt lick
2	331902	9706704	Nuevo Andoas	28,62	0,954	0	oil-polluted salt lick in active oil-infrastructures
3	336596	9701584	Nuevo Andoas	19,91	0,663667	0	oil-polluted salt lick in active oil-infrastructures
4	338129	9694197	Nuevo Andoas	10,45	0,348333	0	oil-polluted salt lick in active oil-infrastructures
5	338569	9693173	Nuevo Andoas	9,1	0,303333	0	oil-polluted salt lick in active oil-infrastructures
6	338569	9693173	Nuevo Andoas	9,1	0,303333	0	oil-polluted salt lick in active oil-

							infrastructures
7	340409	9692361	Nuevo Andoas	7,23	0,241	0	oil-polluted salt lick in active oil-infrastructures
8	348811	9681344	Nueva Alianza de Capahuari	3,42	0,855	0	oil-polluted salt lick in active oil-infrastructures
9	350004	9680393	Nueva Alianza de Capahuari	5,06	1,265	0	oil-polluted salt lick in active oil-infrastructures
10	350590	9679070	Nueva Alianza de Capahuari	6,13	1,5325	1,54	oil-polluted salt lick with disabled oil-infrastructures
11	350755	9678276	Nueva Alianza de Capahuari	6,61	1,6525	2,2	oil-polluted salt lick with disabled oil-infrastructures
12	350777	9678313	Nueva Alianza de Capahuari	6,61	1,6525	0	oil-polluted salt lick with disabled oil-infrastructures
13	365262	9696595	Nueva Jerusalen	17,67	0,589	0	oil-polluted salt lick in active oil-infrastructures
14	366141	9697346	Nueva Jerusalen	16,25	0,541667	0	oil-polluted salt lick in active oil-infrastructures
15	368240	9684409	Nueva Jerusalen	2,64	0,33	2,29	natural salt lick
16	371420	9679240	Nueva Jerusalen	23,39	2,92375	4,66	natural salt lick

17	370415 9741077	José Olaya	39,31	1,310333	0	oil-polluted salt lick in active oil-infrastructures

Table 2. Number of species recorded in each salt lick visiting and consuming soil and water

Salt lick	Type	Number of species visiting	Number of species consuming soil	Average species visiting		Average species consuming	
2	oil-polluted salt lick in active oil-infrastructures	3	3	5,842105	all salt licks	3,941176	all salt licks
5	oil-polluted salt lick in active oil-infrastructures	9	6	5,5625	oil-polluted salt licks	3,733333	oil-polluted salt licks
3	oil-polluted salt lick in active oil-infrastructures	4	3	7,333333	natural salt lick	4	natural salt lick
6	oil-polluted salt lick in active oil-infrastructures	9	6				
18	oil-polluted salt lick in active oil-infrastructures	3	3				
4	oil-polluted salt lick in active oil-infrastructures	11	5				
1	oil-polluted salt lick in active oil-infrastructures	15	8				
16	natural salt lick	5	3				
15	natural salt lick	2	1				

19	oil-polluted salt lick in active oil-infrastructures	4	2
12	oil-polluted salt lick in active oil-infrastructures	3	3
14	oil-polluted salt lick in active oil-infrastructures	3	3
6	oil-polluted salt lick in active oil-infrastructures	8	6
7	oil-polluted salt lick in active oil-infrastructures	9	6
8	oil-polluted salt lick in active oil-infrastructures	2	2
9	oil-polluted salt lick in active oil-infrastructures	6	5
12	oil-polluted salt lick in active oil-infrastructures	3	2
11	oil-polluted salt lick in active oil-infrastructures	7	6
10	oil-polluted salt lick in active oil-infrastructures	5	0

Table 3. Visit frequency of each species in each sampled salt lick measured through the total of visits of each species in each salt lick divided by the number of recording days in each specific salt licks. Records of Quiropteros are not considered here.

	oil-polluted salt lick in active oil-infrastructures													oil-polluted salt lick in disabled oil-infrastructures			Measure for oil-polluted salt lick			natural salt lick			Measure for natural sal lick		
Salt lick code	**2**	**5**	**3**	**6**	**18**	**4**	**19**	**13**	**14**	**17**	**7**	**8**	**9**	**12**	**11**	**10**	**Average**	**Max**	**Min**	**1**	**16**	**15**	**Average**	**Max**	**Min**
number of recording days	32	96	200	181	23	139	22	80	67	105	155	12	38	67	64	63	84	200	12	213	47	37	99	213	37
All species	0,6	2,1	6,3	1,4	3,7	2,1	2,3	1,9	1,6	0,8	1,0	4,2	2,2	0,6	5,0	31,6	4,2	31,6	0,6	3,5	21,6	3,7	9,6	21,6	3,5
Ara chloropterus	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,4	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,4	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Ara macao	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,2	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,2	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Ardilla	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Brotogeris sp.	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Cacicus cela	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Coendou prehensilis	0,0	0,1	0,0	0,5	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,2	0,1	0,5	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Cuniculus paca	0,4	0,1	0,0	0,4	0,0	0,1	0,3	0,0	0,0	0,0	0,0	0,1	0,0	0,0	0,1	0,0	0,1	0,4	0,0	0,1	0,1	0,0	0,1	0,1	0,0
Dasyprocta	0,0	0,0	0,0	0,1	0,0	0,5	0,0	0,0	0,1	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,1	0,5	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Eira barbara	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Herpetotheres cachinnans	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Leopardus pardalis	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Leopardus sp.	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Leopardus tigrinus	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Leptotila	0,0	0,0	0,0	0,2	0,0	0,0	0,0	0,1	0,0	0,0	0,1	4,1	0,0	0,0	0,0	0,1	0,3	4,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Mazama americana	0,1	0,9	2,5	0,0	0,1	1,3	0,0	0,1	0,2	0,0	0,1	0,0	0,2	0,0	0,0	0,0	0,3	2,5	0,0	0,0	0,1	0,0	0,0	0,1	0,0
Momotus momota	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Myoprocta	0,0	0,0	0,0	0,0	2,8	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,2	2,8	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Ortalis guttata	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	1,5	0,0	0,0	0,5	1,5	0,0

Pantera onca	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Patagioenas cayennensis	0,0	0,0	0,0	0,1	0,8	0,0	0,0	0,0	0,0	0,0	0,1	0,0	0,2	0,0	0,4	0,7	0,1	0,8	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Pecari tajacu	0,0	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,1	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Priodontes maximus	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Procyon cancrivorus	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Psophia leucoptera	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Ramphastos tucanus	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Saguinus sp.	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0	0,0
Tapirus terrestris	0,1	0,9	3,8	0,0	0,0	0,0	2,0	1,7	1,3	0,0	0,6	0,0	1,7	0,6	4,5	30,6	3,0	30,6	0,0	1,8	21,4	3,7	9,0	21,4	1,8

CAPÍTULO 6

Instituições que forneceram apoio logístico para a pesquisa

Este projeto conta com o empréstimo pelo Projeto Médio Juruá de 01 barco de alumínio equipado com motor de popa 15hp, 120 armadilhas fotográficas (Bushnell, modelo 119876, e Browning, modelo BTC-6HD), 2 unidades GPS (Garmin, modelo GPSMap 64), equipamento de escalada em árvores e acesso a escritórios na cidade de Carauari. O Instituto Mamirauá, Instituto Chico Mendes, ASPROC e Fundação Amazonas Sustentável estão fornecendo apoio com "caronas" em seus barcos, quando já estiverem sendo utilizados por alguém de sua equipe e com acesso à internet e acomodação em suas bases.ICTA UAB

PROJETO
MÉDIO JURUÁ

Instituto de Desenvolvimento
Sustentável Mamirauá

ICMBio
MMA

ASPROC
Associação dos Produtores
Rurais de Carauari

Fundação
Amazonas
Sustentável

Financiamento

Um total de 9 aplicações foram feitas, destas recebi 3 respostas positivas.

Instituições financiadoras da pesquisa

DSC
DALLAS SAFARI CLUB

BOLSAS FUNBIO
CONSERVANDO
O FUTURO

The
Rufford
Foundation

Instituições que negaram o pedido de financiamento

NATIONAL
GEOGRAPHIC

PITTSBURGH ZOO
& PPG AQUARIUM

ROGER
WILLIAMS PARK
ZOO

Mitsubishi Corporation Foundation
FOR THE AMERICAS

CONSERVATION
LEADERSHIP
PROGRAMME

Componentes Curriculares

Disciplinas cursadas no PPGCB

Conservação da Biodiversidade (Nota final 9.8);

Programa R (Nota final 9.8);

Análise de redes para a compreensão de sistemas etnobiológicos (Nota final 10.0);

Seminários III (Nota final 9.0);

Seminários IV (Nota final 10.0);

Estágio curricular de docência no ensino superior

Disciplina: Ecologia de Comunidade (Aprovada);

Disciplina: Metodologia cientifica (Aprovada).

Componentes Extracurriculares

Artigos publicados durante o doutorado (9)

1. Franciany Braga-Pereira, Thais Q. Morcatty, Hani R. El Bizri (...) Pedro Mayor. Congruence of local ecological knowledge (LEK)-based methods and line-transect surveys in estimating wildlife abundance in Tropical forests. **Methods in Ecology and Evolution**. (2021). https://doi.org/10.1111/2041-210X.13773

2. Franciany Braga-Pereira, Carlos A. Peres, Rômulo Romeu Nóbrega Alves, Carmén Van-Dúnem Santos. Intrinsic and extrinsic motivations governing prey choice by hunters in a post-war African forest-savannah macromosaic**. Plos One**. (2021). https://doi.org/10.1371/journal.pone.0261198

3. Franciany Braga-Pereira, Carmen Van-Dúnem Santos, Rômulo Romeu Nóbrega
Alves and Luke Hunter. Persistence of wild felids after a protracted civil war in Quiçama National Park and Quiçama Game Reserve, Angola. **African Journal of Ecology**. (2021). https://doi.org/10.1111/aje.12946

4. Braga-Pereira, F., Peres, C.A., Campos-Silva, J.V. et al. Warfare-induced mammal population declines in Southwestern Africa are mediated by species life history, habitat type and hunter preferences. **Nature Scientific Reports.** 10, 15428 (2020). https://www.nature.com/articles/s41598-020-71501-0

5. Braga-Pereira, F., Bogoni, J. A. and Alves, R. R. N. From spears to automatic rifles: the shift in hunting techniques as a mammal depletion driver during the Angolan civil war. **Biological Conservation.** 249, 108744 (2020). https://www.sciencedirect.com/science/article/abs/pii/S0006320720308028

6. Karolina Martins Borges, Tacyana Pereira Ribeiro Oliveira, **Franciany BragaPereira**, Ierecê Lucena Rosa, Henrique Anatole Cardoso Ramos, Luiz Alves Rocha, Rômulo Romeu Nóbrega Alves Caught in the (inter)net: Online trade of ornamental fish in Brazil. **Biological Conservation**, 263. (2021). https://doi.org/10.1016/j.biocon.2021.109344

7. Bernárdez-Rodriguez, G. F., Bowler, M., **Braga-Pereira, F**., McNaughton, M., & Mayor, P. Conservation education promotes positive short- and medium-term changes in perceptions and attitudes towards a threatened primate species. **Ethnobiology and Conservation**, *10*. (2021). https://doi.org/10.15451/ec2021-09-10.31-1-16

8. da Silva, M.X.G., **Braga-Pereira, F**., da Silva, M.C. *et al.* What are the factors influencing the aversion of students towards reptiles?. **J Ethnobiology Ethnomedicine 17,** 35. (2021). https://doi.org/10.1186/s13002-021-00462-z

9. Braga Pereira, Franciany, D. Franzini, Lissa, Mota, Heliene, Santos, Carmen Alves, Rômulo. Ebook-MAMÍFEROS DO PARQUE NACIONAL DA QUIÇAMA -ANGOLA Unindo o conhecimento tradicional ao científico. Mamíferos do Parque Nacional da Quiçama- Unindo o conhecimento tradicionalao científico. (2018). **Editora UFPB.** http://www.editora.ufpb.br/sistema/press5/index.php/UFPB/catalog/book/116

Artigos em revisão em revista preparados durante o doutorado (3)

1. André Pinassi Antunes, Daniel Munari, Marina A R de Mattos Vieira, **Franciany Braga-Pereira,** et al. Community-based wildlife management to prevent Amazonian pandemics. **The Lancet***.*

2. Claudileide Pereira dos Santos, **Franciany Braga- Pereira***, Rômulo Romeu Nobrega Alves, Domestic and wildmeat consumption and preferences by indigenous students in the Brazilian coast. **Environment, Development and Sustainability***.*

3- Vincent Nijman, Thais Q. Morcatty, Hani R. El Bizri, Hasan Al-Razi, Andie Ang, Ahmad Ardiansyah, Sadek Atoussi, Daniel Bergin, Sarah Bell, **Franciany Braga-Pereira**, (...) K. Anne-Isola Nekaris. Global Online Trade in Primates for Pets. **Biological Conservation.**

Artigos desenvolvidos durante o doutorado sendo finalizados para submissão (4)

1.Franciany Braga-Pereira, RAYNNER RILKE DUARTE BARBOZA and Rômulo R. N. Alves. Motivadores para caçailegal na maior floresta seca da América do Sul

2. Heliene Mota, **Franciany Braga-Pereira**, Luane Maria Azeredo, Luiz Carlos Serramo Lopez and Rômulo R. N. Alves. Attitudes of undergraduate students towards wild vertebrates: A Relationship between Aesthetic Preferences, Utilitarian, Fear and Preservation.

3.Amanda Rozendo da Silva, **Franciany Braga-Pereira**, José Valberto de Oliveira, Moacyr Xavier Gomes da Silva and R. and Rômulo R. N. Alves. Percepção e atitudes de estudantes sobre vertebrados silvestres na região do nordeste brasileiro: implicações para a conservação.

4. Tall Levi, **Franciany Braga-Pereira**, Carlos Peres. Modelagem da sustentabilidade da caça para autoconsumo em florestas tropicais

Participação em banca de monografia (2)

1. Aryane Rosa Da Costa. Population Estimative and Management of Domestic Dogs (*Canis lupus familiaris*) in two Atlantic Forest Protect Areas in Brazil. 2018. Undergraduate in Ecology - Universidade Federal da Paraíba.

2. Carlos Eduardo Neves. Local Ecological Knowledge of mammals by school students, and rural and urban adult populations in the Largest Tropical Dry Forest Region in South America. 2021. Undergraduate in Biology. Universidade Federal da Paraíba

Participação em Congresso (3)

European Congress of Conservation Biology, 12 a 15 de junho, em Jyväskylä, Finlândia

XVI Congress of the International Society of Ethnobiology and XII Brazilian Symposium on Ethnobiology and Ethnoecology, 7 a 10 de Agosto de 2018, em Belém, Brasil

X Simpósio Nordestino de Etnobiologia e Etnoecologia, 22 a 26 de abril de 2019, em João Pessoa, Brasil

Organização de Simpósio (1)

IV Simpósio de Zoologia da UFPB: Ciência tradicional, aplicada e de ponta. De 06 a 10 de agosto de 2018, em João Pessoa, Brasil

Premiações em Congressos (2)

Marked warfare-induced mammal population declines in West Africa are mediated by species life history, habitat type and hunters preferences

Prey selectivity and drivers of illegal hunting by West African game hunters

Representante Discente (março 2018 até abril 2019)