UNIVERSIDADE ESTADUAL DE SANTA CRUZ

PRÓ-REITORIA DE PESQUISA E PÓS-GRADUAÇÃO

PÓS-GRADUAÇÃO EM ECOLOGIA E CONSERVAÇÃO DA BIODIVERSIDADE

YASMIN MARIA SAMPAIO DOS REIS

MONITORAMENTO COMUNITÁRIO DA FAUNA CINEGÉTICA EM FLORESTAS TROPICAIS: EFETIVIDADE DE PROJETOS E SUSTENTABILIDADE DE CAÇA EM UMA ÁREA PROTEGIDA NA AMAZÔNIA

ILHÉUS – BAHIA

2023

YASMIN MARIA SAMPAIO DOS REIS

MONITORAMENTO COMUNITÁRIO DA FAUNA CINEGÉTICA EM FLORESTAS TROPICAIS: EFETIVIDADE DE PROJETOS E SUSTENTABILIDADE DE CAÇA EM UMA ÁREA PROTEGIDA NA AMAZÔNIA

Tese apresentada à Universidade Estadual de Santa Cruz, como parte das exigências para obtenção do título de Doutor em Ecologia e Conservação da Biodiversidade.
Área de concentração: Ecologia e Conservação da Biodiversidade
Discente: Yasmin Maria Sampaio dos Reis
Orientadora: Drª Maíra Benchimol

ILHÉUS -BAHIA

2023

R375 Reis, Yasmin Maria Sampaio dos.

Monitoramento comunitário da fauna cinegética em florestas tropicais: efetividade de projetos e sustentabilidade de caça em uma área protegida na Amazônia / Yasmin Maria Sampaio dos Reis. – Ilhéus, BA: UESC, 2023.

159 f. : il.

Orientadora: Maíra Benchimol.

Tese (doutorado) – Universidade Estadual de Santa Cruz. Programa de Pós-Graduação em Ecologia e Conservação da Biodiversidade.

Inclui referências.

1. Biodiversidade – Monitoramento. 2. Mamíferos. 3. Fauna. 4. Caça – Proteção. 5. Sustentabilidade. I. Título.

CDD 333.95

YASMIN MARIA SAMPAIO DOS REIS

MONITORAMENTO COMUNITÁRIO DA FAUNA CINEGÉTICA EM FLORESTAS TROPICAIS: EFETIVIDADE DE PROJETOS E SUSTENTABILIDADE DE CAÇA EM UMA ÁREA PROTEGIDA NA AMAZÔNIA

Tese apresentada à Universidade Estadual de Santa Cruz, como parte das exigências para obtenção do título de Doutor em Ecologia e Conservação da Biodiversidade

Ilhéus, 27 de setembro de 2023.

Dr. Elildo de Carvalho Junior
ICMBio

Prof. Dr. Juarez Pezzuti
UFPA

Prof. Dra. Franciany Braga Pereira
University of Barcelona

Prof. Dr. Whaldener Endo
UFRR

Prof. Dra. Maíra Benchimol
(Orientadora)
UESC

Dedico esta tese aos habitantes da Reserva Extrativista Tapajós-Arapiuns, guardiões da floresta. À memória de meu pai, cuja vida foi dedicada à conservação da biodiversidade e à melhoria da qualidade de vida desses povos. À minha mãe, meu alicerce constante, com sincera gratidão por seu amor incondicional.

AGRADECIMENTOS

“Quando o nosso sonho é a vontade de Deus para a nossa vida”. Com estas palavras, celebrei a minha aprovação no programa de doutorado em 2019. Hoje, ao chegar ao final desta importante etapa da minha jornada acadêmica, minha convicção na vontade de Deus para mim se torna ainda mais sólida. Por essa razão, agradeço à Deus por ter me acompanhado, por ter infundido em mim força e perseverança para nunca desistir do meu objetivo.

E falando em força, agradeço à minha família, que se mostrou uma fonte inesgotável de fortaleza durante esse período. Minha mãe, Gorete, sempre me incentivou com um orgulho profundo a seguir por esse caminho, e foi ela quem orou fervorosamente a cada desafio que enfrentei durante o doutorado. À minha irmã Isabela, agradeço pelas conversas edificantes e por me lembrar constantemente da bênção que era viver essa etapa da minha vida. A meu irmão Nivaldo, agradeço por demonstrar amor e apoio ao me presentear com deliciosos lanches e sobremesas. E a minha irmã Izabel, nossa eterna criança, agradeço por ser o abraço de Deus no meu dia-a-dia em inúmeras ocasiões.

Uma especial e carinhosa menção às minhas leais companheiras de quatro patas, Lise, Bela e Belinha. Sua companhia trouxe conforto e alegria inigualáveis durante esta jornada.

Minha gratidão às minhas amigas Delaine, Amora, Fernanda, Priscila, Nayane, Jhersyka, Alcimara, Heveline, Rosiana, Cássia, Letícia e Juliana por estarem sempre dispostas a ouvir meus anseios, compartilhar minhas alegrias e aliviar minhas aflições. Queridas amigas, suas palavras de encorajamento foram um apoio inestimável. Reconheço o quanto pude contar com cada uma de vocês e sou grata pela amizade e gentileza que demonstraram.

Agradeço aos amigos e irmãos da igreja, especialmente da Comunidade Católica Vida no Espírito Santo, por todo o apoio em forma de conversas e orações ao longo desses anos.

Aos servidores e voluntários do ICMBio/Santarém-PA na Reserva Extrativista Tapajós-Arapiuns. Às #MeninasdafaunaRTA, começo agradecendo a Jackeline, por me receber tão bem no Programa Monitora. Seu acolhimento não se limitou ao papel de gestora da Reserva, mas estendeu-se à amizade e confiança depositadas, além de ter proporcionado inúmeras oportunidades acadêmicas e profissionais. À Stephanie, cujo apoio durante as viagens de campo e demais atividades do projeto foi fundamental. E não posso deixar de mencionar Rúbia e Bianca, que foram verdadeiros pilares durante o meu doutorado. Elas sempre me incentivaram, e juntas compartilhamos ideias e a vontade de garantir que o programa Monitora na Reserva Tapajós-Arapiuns realmente fornecesse informações úteis para as decisões locais. Além disso, sempre que necessário, elas compartilharam materiais e dados, e com paciência me relembravam informações que frequentemente escapavam da memória.

Um agradecimento mais que especial à minha "mãe científica", Maíra, por ter aceitado me orientar e, principalmente, por nunca ter soltado minha mão durante esses anos. Sua forma de fazer pesquisa, lecionar e orientar, seu compromisso e a dedicação e alegria com os quais realiza tudo isso, redefiniram minha perspectiva em relação à academia. Maíra, você não apenas foi um apoio e uma fonte de inspiração, mas também assumiu o papel de uma verdadeira psicóloga, oferecendo conselhos valiosos e compreendendo meu ritmo. Hoje, ao finalizar meu doutorado, me sinto mais emocionalmente segura e com habilidades ampliadas. E isso é também graças a você. Obrigada por tudo!

Esta tese é fruto de um trabalho construído com as mãos unidas de diversas pessoas e instituições preocupadas com a conservação da biodiversidade e com a qualidade de vida dos povos tradicionais e indígenas. Dentre as quais posso citar: Organização das Associações e Moradores da Reserva Extrativista Tapajós-Arapiuns (Tapajoara), Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio), Coordenação Geral de Pesquisa e Monitoramento da Biodiversidade (COMOB), Centro Nacional de Pesquisa e Conservação de Mamíferos Carnívoros (CENAP), Centro Nacional de Pesquisa e Conservação de Aves Silvestres (CEMAVE) e Instituto de Pesquisas Ecológicas (IPÊ). O apoio dos financiadores foi imprescindível, entre eles: Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ), Gordon & Betty Moore Foundation, Fundo Amazônia, Programa ARPA e United States Agency for International Development (USAID).

Agradeço ao Conselho de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) pela bolsa de doutorado. À Universidade Estadual de Santa Cruz, a Pró-Reitora de Pesquisa e Pós-Graduação (PROPP UESC), ao Programa de Pós-graduação de Ecologia e Conservação da Biodiversidade (PPGECB) e ao Laboratório de Ecologia Aplicada à Conservação (LEAC) pelo suporte e infraestrutura fornecidos para o desenvolvimento desta pesquisa.

Aos professores do PPGECB pelos conhecimentos compartilhados durante as disciplinas e fora delas. Em particular, sou grata a professora Camila Cassano pelos valiosos comentários nas versões do manuscrito do Capítulo 2 desta tese.

Aos pesquisadores Dr. Hani El Bizri, Dr. Milton de Paula, Dr. Elildo Carvalho que forneceram ajuda com conhecimento analítico. À banca de defesa, composta por Dr. Elildo Carvalho, Prof. Dr. Juarez Pezzuti, Prof. Dra. Franciany Braga Pereira e Prof. Dr. Whaldener Endo, pelos valiosos comentários e sugestões.

A todos os colegas da pós-graduação, pelos momentos de lazer e pela troca de conhecimentos, seja na copa da Base Ambiental, no karaokê do Iate ou no pastel da Dona Gel. Em especial, sou grata aos meus "irmãos científicos" Matheus, Victor, Flávia, Carla, Carolina e Débora, assim como os amigos que tive a honra de morar: Karol, Claudia, Hermana, Lorena, Lucas, Diogo, Letícia, Ketlen e Júlia.

Finalizo fazendo um agradecimento muito especial aos moradores da Reserva Tapajós-Arapiuns, cujas portas de suas comunidades e casas se abriram generosamente para mim. Com eles, aprendi muito sobre a fauna e o cotidiano daqueles que não apenas habitam, mas também lutam para salvaguardar a floresta. Agradeço especialmente aos monitores locais do programa Monitora, pois sem eles, este trabalho não seria viável. Hoje, é por meio do trabalho com essas comunidades e monitores dedicados que deposito todas as minhas esperanças na efetiva conservação da biodiversidade e na manutenção da segurança alimentar dos povos e comunidades da região do Tapajós e Arapiuns.

SUMÁRIO

RESUMO

Os projetos de monitoramento da biodiversidade são ferramentas essenciais para compreender o impacto das atividades antrópicas na perda atual de biodiversidade. Neste sentido, iniciativas de monitoramento comunitário (i.e., com o envolvimento de pessoas locais) focadas em vertebrados terrestres *in situ* e da caça têm se expandindo em países tropicais, especialmente dentro de Unidades de Conservação (UC) de uso sustentável. Estes programas permitem avaliar a manutenção das populações de espécies cinegéticas (i.e., alvo de caça) dentro de UCs, além de avaliar se a caça de subsistência tem afetado estas populações. Assim, o objetivo desta tese foi: (i) discutir o potencial dos projetos de monitoramento comunitário da fauna cinegética em fornecer informações de longo prazo sobre as populações, empoderar as comunidades locais e implementar ações de manejo; (ii) avaliar a influência da pressão antropogênica sobre as populações de espécies terrestres florestais e sobre os padrões de caça ao longo de uma série temporal; e assim (iii) contribuir para a gestão da caça em uma UC amazônica de uso sustentável, a Reserva Extrativista Tapajós-Arapiuns (RESEX-TA). No primeiro capítulo, realizamos uma revisão sistemática da literatura para identificar projetos de monitoramento comunitário da fauna terrestre cinegética nos trópicos. Analisamos especificamente dezessete projetos e revelamos que essas iniciativas têm potencial para gerar informações contínuas, empoderar as comunidades locais e implementar ações de manejo. Contudo, observamos desafios como interrupções por falta de financiamento e a necessidade de maior participação local para um empoderamento efetivo. Isso destaca a importância do engajamento comunitário em todas as etapas do monitoramento, de parcerias sólidas para financiamento a longo prazo e da tradução dos resultados em ações de manejo e conservação. Os dois capítulos seguintes basearam-se nos dados de seis anos do monitoramento comunitário das espécies caçadas na RESEX-TA, parte do Programa Nacional de Monitoramento da Biodiversidade (Monitora) do Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio). No segundo capítulo, avaliamos a influência de variáveis antrópicas (relacionadas à pressão de caça) e anos monitorados nos padrões de densidade e biomassa de espécies terrestres cinegéticas, e as tendências temporais nas estimativas de densidade. Mostramos que a pressão antropogênica não afetou a densidade e a biomassa dos vertebrados de médio e grande porte, com exceção do táxon Tinamidae, influenciado negativamente. Além disso, as densidades de todos os táxons analisados se mantiveram estáveis ao longo da série temporal. No entanto, os baixos registos de algumas espécies podem indicar perdas populacionais passadas, refletindo em um cenário atual de semi-defaunação. Nossos resultados destacam a importância do monitoramento comunitário na obtenção de dados e orientação de ações de manejo. No terceiro capítulo analisamos a influência das mesmas variáveis antrópicas e anos do Capítulo II sobre o perfil e produtividade de caça (i.e., CPUE), e as tendências temporais nas estimativas de CPUE. Nossos resultados demonstram que a pressão antropogênica não afetou negativamente as variáveis respostas e a CPUE se manteve estável. Por fim, nossas descobertas sugerem que a caça é provavelmente sustentável para a maioria das espécies, mas insustentável para espécies ameaçadas como *Tapirus terrestris* e *Tayassu pecari*. Assim, esta tese contribui no entendimento dos desafios e potencialidades do monitoramento comunitário, fornecendo novas evidências sobre a importância dessas iniciativas no fornecimento de informações de longo prazo que subsidiam ações de gestão. Por fim, sugerimos o protocolo da Tapajós-Arapiuns como modelo para outras áreas protegidas amazônicas interessadas em monitorar a caça de subsistência.

Palavras-chave: Monitoramento participativo, mamíferos, aves, fauna cinegética, sustentabilidade da caça.

ABSTRACT

Biodiversity monitoring projects are essential tools for understanding the impact of anthropogenic activities on current biodiversity loss. In this context, community monitoring initiatives (i.e., involving local people) focused on monitoring terrestrial vertebrates *in situ* and game harvest have been expanding in tropical countries, especially within sustainable-use protected area (PA). These programs allow for assessing the maintenance of game species populations (i.e., species targeted for hunting) within PAs, as well as evaluating whether subsistence hunting has impacted these populations. Thus, the objective of this thesis was: (i) to discuss the potential of community monitoring projects for game fauna to provide long-term information on populations, empower local communities and implement management actions; (ii) to assess the influence of anthropogenic pressure on terrestrial forest species populations and hunting patterns over a time series; and (iii) to contribute to the management of hunting in an Amazonian sustainable-use PA, the Tapajós-Arapiuns Extractive Reserve (RESEX-TA). In the first chapter, we conducted a systematic literature review to identify community-based monitoring projects of terrestrial game fauna in the tropics. We specifically examined seventeen of those projects and revealed that these initiatives have the potential to provide continuous information, empower local people, and implement management actions. However, we observed challenges such as interruptions due to lack of funding and the need for greater local participation for effective empowerment. This highlights the importance of community engagement at all monitoring stages, solid partnerships for long-term funding, and translating results into management and conservation actions. The next two chapters were based on database from 6-years of community monitoring of game species and game harvest conducted within in the RESEX-TA, part of the National Biodiversity Monitoring Program (Monitora) of the Chico Mendes Institute for Biodiversity Conservation (ICMBio). In the second chapter, we evaluated the influence of anthropogenic stressors (related to hunting pressure) and monitored year on the on the patterns of density and biomass of terrestrial game species and the temporal trends in density estimates along the time series. We demonstrated that anthropogenic pressure had no significant impact on the density and biomass of medium and large vertebrates, with the exception of the taxon Tinamidae, negatively influenced. Additionally, the densities of all the analyzed taxa remained stable throughout the time series. However, the low records for some species may indicate past population losses which would reflect in a semi-defaunated current scenario. Our results highlight the importance of community monitoring in obtaining data and guiding management actions. In the third chapter, we analyzed the influence of the same anthropogenic variables and years as in Chapter II on hunting profile and hunting productivity (i.e., CPUE), and further assessed trends in CPUE. Our results demonstrate that anthropogenic pressure did not negatively affect response variables, and CPUE remained stable along the time series. Finally, our findings suggest subsistence hunting is likely sustainable for most species but unsustainable for threatened species like *Tapirus terrestris* and *Tayassu pecari*. Thus, this thesis contributes to understanding the challenges and potential of community monitoring, providing new evidence on the importance of these initiatives in providing long-term information that supports management actions. Finally, we suggest the Tapajós-Arapiuns protocol as a model for other Amazonian protected areas interested in monitoring subsistence hunting.

Key words: participatory monitoring, mammals, birds, game fauna, hunting sustainability.

INTRODUÇÃO GERAL

As florestas tropicais são habitats essenciais para a persistência de uma ampla gama de espécies, e possuem importância única ao abrigar a mais alta biodiversidade da Terra e fornecer uma ampla gama de serviços ecossistêmicos (BRANDON, 2014; BOGONI *et al.*, 2020). Além disso, muitas florestas tropicais são habitadas por diversos povos indígenas e comunidades tradicionais que dependem de seus recursos para subsistência. No entanto, os ecossistemas tropicais enfrentam ameaças crescentes devido às atividades humanas, incluindo a destruição de habitats, o aquecimento climático e a exploração da fauna silvestre (VEIGA & EHLERS, 2009). Como resultado, as florestas tropicais estão perdendo espécies a uma taxa alarmante, com mais de 60% das espécies endêmicas ameaçadas de extinção devido apenas às mudanças climáticas (MANES *et al.*, 2021). Em resposta a esses desafios, diversas estratégias de conservação têm sido propostas, como a demarcação e gestão de áreas protegidas. Em particular, programas sistemáticos de monitoramento da biodiversidade têm sido estabelecidos em um número crescente destas áreas protegidas, com o intuito de subsidiar, avaliar e acompanhar as distribuições de diferentes populações biológicas e assim contribuir no planejamento de estratégias de conservação espécie-específicas (CRONEMBERGER *et al.*, 2023b).

O monitoramento da biodiversidade compreende um passo fundamental para a conservação, pois, quando adequadamente planejado com objetivos e delineamento amostral claros (YOCCOZ *et al.*, 2003) e amostragens eficientes, padronizadas e confiáveis (PONCE-MARTINS *et al.*, 2022), pode fornecer informações de longo prazo sobre o status e as tendências das espécies-alvo (DANIELSEN *et al.*, 2005; IPBES 2015). Especialmente em áreas protegidas, detentora de altos níveis de biodiversidade, essa ferramenta torna-se essencial para acompanhar o estado dos ecossistemas, espécies e processos naturais. Além disso, fornece dados baseados em evidências sobre a forma como os alvos monitorados respondem às mudanças ambientais e às intervenções de manejo (LINDENMAYER & LIKENS, 2009; LOVETT *et al.*, 2007).

Os projetos de monitoramento da biodiversidade de base comunitária (i.e., com o envolvimento de pessoas locais) têm se expandido significativamente em áreas protegidas, particularmente em Unidades de Conservação de Uso Sustentável. Em particular, essas áreas protegidas visam conciliar a conservação da biodiversidade com a exploração sustentável dos recursos naturais por parte de comunidades indígenas ou tradicionais que habitam a região (LIMA & POZZOBON, 2005). No entanto, ainda há poucas informações disponíveis sobre a efetividade destas áreas em alcançar essa sinergia, visto a dificuldade em avaliar a sustentabilidade no uso dos recursos. Nesse contexto, os projetos comunitários de monitoramento, focados na biodiversidade e no uso de recursos naturais, podem contribuir para esse entendimento.

Monitoramento comunitário

Apesar de haver um consenso em relação a importância do monitoramento, programas de base comunitária tem gerado discordância entre os pesquisadores em relação ao protagonismo dos atores locais. Alguns questionam a capacidade dos comunitários de lidar com erros amostrais e, por consequência, gerar inferências confiáveis sobre os alvos monitorados (YOCCOZ *et al.*, 2003; BURTON, 2012). Porém, outros reconhecem o vasto conhecimento e habilidades da população local (ESBACH, 2023) e defendem o potencial desses projetos em promover três pilares essenciais para a conservação: (i) fornecer informações contínuas sobre o recurso monitorado, (ii) empoderar as comunidades locais e (iii) implementar ações de manejo (DANIELSEN *et al.*, 2021, 2022). Nesse contexto, é de suma importância avaliar as contribuições e os desafios dos projetos de monitoramento de base comunitária, visando aprimorar a eficácia dos projetos atuais e futuros, além de aprofundar nosso entendimento e melhorar a gestão sustentável nas áreas protegidas.

Como em qualquer iniciativa de conservação a longo prazo, um dos principais desafios enfrentados pelos programas de monitoramento é a obtenção de financiamento contínuo (DANIELSEN *et al.*, 2009; CRONEMBERGER *et al.*, 2023b). Especialmente em países emergentes e biodiversos como é o caso do Brasil, com políticas ambientais desafiadoras (DIELE-VIEGAS *et al.*, 2020), o monitoramento baseado na comunidade emerge como uma iniciativa econômica que busca promover o envolvimento das comunidades locais, fortalecer os sistemas locais existentes e aprimorar a tomada de decisões na gestão dos recursos naturais (DANIELSEN *et al.*, 2003; DANIELSEN *et al.*, 2014).

De fato, iniciativas comunitárias estão ganhando destaque em áreas protegidas gerenciadas de forma colaborativa com a população local, onde a compreensão das principais ameaças às espécies e/ou recursos naturais utilizados pelas comunidades é crucial para propor novas ações de mitigação (LUZAR *et al.*, 2011). Por exemplo, na Amazônia Ocidental, o monitoramento comunitário do pirarucu gigante (*Arapaima* cf. *gigas*) resultou na recuperação das populações dessa espécie ao longo do tempo, evidenciando o impacto positivo do manejo comunitário na conservação da espécie e no empoderamento local (CAMPOS-SILVA *et al.*, 2019). Em outra região, no sudoeste da Guiana, técnicos de campo indígenas, treinados por cientistas, coletaram dados sociais, de caça e ecológicos em 23 aldeias Makushi e Wapishana ao longo de três anos (LUZAR *et al.*, 2012; IWAMURA *et al.*, 2014; FRAGOSO *et al.*, 2016). Esses dados revelaram indícios de práticas de caça sustentável no território (BRODIE & FRAGOSO, 2021), apontando que as terras indígenas desempenham um papel crucial na manutenção das populações de espécies cinegéticas, ao mesmo tempo que garantem a segurança alimentar das comunidades indígenas. Esses exemplos demonstram o potencial e a relevância das abordagens de monitoramento baseadas na comunidade na promoção da conservação e bem-estar das populações locais.

Diversas iniciativas comunitárias têm sido implementadas na bacia Amazônica para monitorar os vertebrados terrestres cinegéticos *in situ* e a atividade de caça, visando avaliar a sustentabilidade da caça (ZAPATA-RÍOS *et al.*, 2009; LUZAR *et al.*, 2011; CONSTANTINO *et al.*, 2012; MAYOR *et al.*, 2017; EL BIZRI *et al.*, 2020; OLIVEIRA & CALOURO, 2020; DE PAULA *et al.*, 2022). Uma vez que mamíferos e aves de médio e grande porte representam os principais alvos da caça (BENÍTEZ-LÓPEZ *et al.*, 2017, 2019; OLIVEIRA *et al.*, 2023), compreendem o grupo focal nestes programas de monitoramento. Em particular, espécies de grande porte são preferencialmente caçadas devido ao sabor da carne e ao maior retorno por unidade de esforço. Entretanto, também são mais suscetíveis à extinção ou redução populacional devido às suas baixas taxas reprodutivas, longa expectativa de vida e extensos tempos de gestação (BODMER *et al.*, 1997; CARDILLO *et al.*, 2005). Além disso, tais espécies estão sujeitas a influências negativas de fatores externos, como atividades antrópicas (CARDILLO *et al.*, 2005; RIPPLE *et al.*, 2016).

Áreas sob pressão antrópica generalizada são usualmente associadas à alta pressão de caça em florestas tropicais (SAMPAIO *et al.*, 2023). Nestes locais, é comum observar uma redução na densidade de espécies de grande porte, resultando em mudanças na estrutura das comunidades de vertebrados e, consequentemente, reduzindo a biomassa desses grupos (PERES, 2000; JEROZOLIMSKI & PERES, 2003). Em resposta a essas mudanças, os caçadores são incentivados a direcionar seus esforços para a captura de espécies de tamanho médio que estão mais disponíveis ou a buscar locais mais distantes de suas comunidades para forragear (STAFFORD *et al.*, 2017). Essas mudanças nas estratégias de caça geralmente resultam em menor produtividade de caça, refletida por uma diminuição na taxa de carne obtida por caçador por hora caçada (WEINBAUM *et al*., 2013; IWAMURA *et al*., 2014).

Sustentabilidade da caça: realidade ou utopia?

Para analisar a sustentabilidade da caça, é comum utilizar métodos que avaliam as tendências temporais e estimam as densidades e biomassa de vertebrados, além da produtividade da caça (medida através da captura por unidade de esforço - CPUE) em áreas com diferentes intensidades de pressão de caça (IWAMURA *et al.*, 2014; SOUZA-MAZUREK *et al.*, 2000; PARRY *et al.*, 2009). Nesse contexto, diversos fatores antropogênicos têm sido identificados como altamente associados à pressão de caça sobre espécies cinegéticas florestais, incluindo pontos de acesso de caçadores (por exemplo, comunidades e estrada), densidade populacional humana (BEIRNE *et al.*, 2019; BENÍTEZ-LÓPEZ *et al.*, 2017, 2019) e idade da comunidade (CONSTANTINO, 2015). Por exemplo, SCABIN & PERES (2021) mostraram que a distância das comunidades e o tamanho da população humana representam bons indicativos de ameaças antropogênicas para grandes vertebrados de caça em uma área protegida de uso sustentável da Amazônia, já que declínios acentuados de biomassa foram registrados em áreas urbanas que apresentam maior pressão humana. Já JEROZOLIMSKI & PERES (2003) demonstraram que as comunidades mais antigas exerceram maior pressão de caça sobre os grandes vertebrados em assentamentos de florestas neotropicais, nos quais os caçadores precisavam mudar suas espécies-alvo de grande porte para numerosas de pequeno porte à medida que as aldeias envelheciam. Avaliar a influência de variáveis antrópicas sobre as populações de vertebrados e os padrões da caça torna-se assim muito útil, especialmente em áreas protegidas de uso sustentável, que podem sofrer esgotamento populacional de algumas espécies severamente caçadas.

Nas regiões tropicais, a caça de subsistência tem sido uma prática tradicionalmente difundida por gerações de populações locais (FA *et al*., 2022). Para as comunidades tradicionais, a fauna silvestre representa uma das principais fontes de proteína, gordura e nutrientes essenciais, sendo superada apenas pela pesca (REDFORD & ROBINSON, 1987, PEZZUTI *et al.,* 2004; VLIET *et al.,* 2017). Quanto mais distantes dos centros urbanos, maior é a dependência dessas comunidades em relação à caça de subsistência (ROBINSON & BENNETT, 2000). No entanto, à medida que as populações humanas crescem e as atividades antrópicas avançam, a caça insustentável emerge como uma das principais ameaças à vida selvagem global. Estima-se que a prática afete cerca de um quinto das espécies ameaçadas da Lista Vermelha (INGRAM *et al.*, 2021), resultando em defaunação (i.e., extinção ou a diminuição populacional de espécies animais em seu habitat) generalizadas, intensificadas pela perda de habitat (DIRZO *et al.*, 2014; YOUNG *et al.*, 2016). Segundo Bogoni e colaboradores (2022), a sinergia entre essas duas atividades antrópicas foi responsável pela extinção local de aproximadamente 70% de espécies de mamíferos em florestas neotropicais. De fato, o declínio generalizado das populações de vertebrados cinegéticos, especialmente os de grande porte, tem sido atribuído à caça excessiva, com estudos apontando para essa situação particularmente na África como na Ásia (DAVIDSON *et al.*, 2009, RIPPLE *et al.*, 2016, BENÍTEZ-LÓPEZ *et al.*, 2019). Tais processos, por sua vez, desencadeiam consequências significativas no funcionamento e na dinâmica dos ecossistemas (PIRES *et al.*, 2022). Além disso, para comunidades que dependem da carne selvagem para subsistência, o esgotamento da vida selvagem pode comprometer a segurança alimentar e desencadear conflitos sociais (BRASHARES *et al.*, 2014).

Na bacia Amazônica, diversos estudos demonstraram o impacto generalizado da caça de subsistência sobre as populações de vertebrados terrestres (BODMER *et al.*, 1994 ; ROBINSON & BENNETT, 2000; PERES, 2000). No entanto, evidências mais recentes indicam que algumas espécies podem ser mais resistentes à caça (e.g., IWAMURA *et al.*, 2014; ANTUNES *et al.*, 2019), visto que diversos povos tradicionais as caçam intensamente há milhares de anos, porém suas populações permanecem estáveis. Isso ocorre provavelmente porque vastas extensões de floresta inacessíveis a caçadores reabastecem as populações de vertebrados terrestres em áreas adjacentes, onde a caça intensa atua fortemente deplecionando determinadas populações (NOVARO *et al.*, 2000; SIRÉN *et al.*, 2004; PERES & NASCIMENTO, 2006). Essa dinâmica, conhecida como fonte-sumidouro (*source-sink*), se apresenta como um dos principais mecanismos de sustentabilidade da caça na região Amazônica (ANTUNES *et al.*, 2016, 2019). Por outro lado, alguns locais da floresta amazônica sofreram declínios populacionais no passado devido à pressão antrópica e atualmente são reconhecidos por se encontrarem em um estado de semi-defaunação (PERES *et al.*, 2003). Assim, para melhor entender esta dinâmica dentro de uma área protegida de uso sustentável amazônica torna-se necessário monitorar a fauna cinegética *in situ* e a atividade de caça com participação comunitária e gerar informações espaço-temporais sobre as espécies e o recurso monitorados e, assim, avaliar a sustentabilidade da caça.

O Programa Monitora

No Brasil, o governo federal estabeleceu em 2013 e formalizou em 2017 o Programa Nacional de Monitoramento da Biodiversidade (Monitora) através do Ministério do Meio Ambiente (MMA) e do Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio). O programa tem como finalidade monitorar a biodiversidade e os serviços ecossistêmicos associados nas Unidades de Conservação (UCs) administradas pelo ICMBio, visando avaliar o estado de conservação das espécies, do uso sustentável dos recursos naturais e da efetividade do sistema de UCs gerenciado pelo Instituto (RIBEIRO, 2018).

O Monitora é um programa contínuo, de longo prazo e larga escala, e já foi implementado em 113 das 334 UCs geridas pelo ICMBio, sendo em sua maioria representada na região amazônica. O programa é composto por três subprogramas (terrestre, aquático continental e marinho costeiro), que apresentam diferentes componentes (e.g.: terrestre florestal, terrestre campestre e savânico) e tipos de alvos (globais, complementar regional e complementar local) (RIBEIRO, 2018). Uma das principais premissas do programa é a participação social em todas as etapas do monitoramento (PEREIRA *et al.*, 2013; CRONEMBERGER *et al.*, 2023b).

Na Amazônia, o Monitora desenvolveu projetos de monitoramento específicos para contextos locais, focados em uma única UC ou grupo, com as chamadas de "metas complementares", que consistem em monitorar os recursos naturais utilizados pelas comunidades locais (CRONEMBERGER *et al.*, 2023b). Dezesseis UCs fazem parte dessa iniciativa e seis protocolos foram desenvolvidos para monitorar recursos escolhidos pelos próprios comunitários, incluindo a castanha-da-Amazônia na Reserva Extrativista (RESEX) Cazumbá-Iracema (AC), peixes na Reserva Biológica do Abufari (AM) e mamíferos em áreas de concessão florestal na Floresta Nacional do Jamari (RO) (CRONEMBERGER *et al.*, 2023a). As questões a serem respondidas foram definidas coletivamente entre as comunidades locais, gestores de UCs e cientistas, que sugeriram protocolos amostrais adequados para alcançar os objetivos (SOUZA *et al.*, 2019). Uma dessas UCs-alvo foi a RESEX Tapajós-Arapiuns (RESEX-TA), localizada no estado do Pará, que optou por priorizar o monitoramento da madeira e das espécies caçadas para subsistência. No entanto, após as primeiras coletas de dados e a constatação da inviabilidade de continuar a amostragem do alvo complementar madeira, a RESEX-TA decidiu concentrar-se exclusivamente no monitoramento dos animais caçados para subsistência. O objetivo deste monitoramento na RESEX-TA é avaliar o status populacional e a pressão de caça sobre várias espécies cinegéticas terrestres, selecionadas pelos comunitários. Por fim, é importante destacar que o monitoramento da caça de subsistência, conduzido no âmbito do Monitora, também foi implementado em outras duas RESEXs paraenses, a Rio Iriri e Riozinho do Anfrísio. No entanto, a abordagem e o método de coleta de dados diferiram dos adotados na RESEX-TA. Infelizmente, o monitoramento nessas outras duas UCs foi interrompido em 2020 devido à falta de recursos financeiros (DE PAULA *et al.*, 2022).

A Reserva Extrativista Tapajós-Arapiuns

A RESEX-TA é uma UC Federal de Uso Sustentável criada inicialmente com objetivo de resguardar a população local dos impactos causados pela expansão de empresas madeireiras sobre os recursos naturais de suas áreas (OLIVEIRA *et al.*, 2005; SILVA, 2008; REIS, 2017). Localizada no oeste do Pará e criada em 1998, a Tapajós-Arapiuns foi a primeira RESEX brasileira a elaborar um plano de manejo. Atualmente é a RESEX mais populosa do país, abrigando cerca de 23 mil moradores em uma área de 647.610 ha, entre as coordenadas geográficas 02º 20’ a 03º 40’ Sul, e 55º 00’ a 56º 00’ Oeste (Fig. 1) (ICMBio, 2014; SILVA *et al.*, 2022).

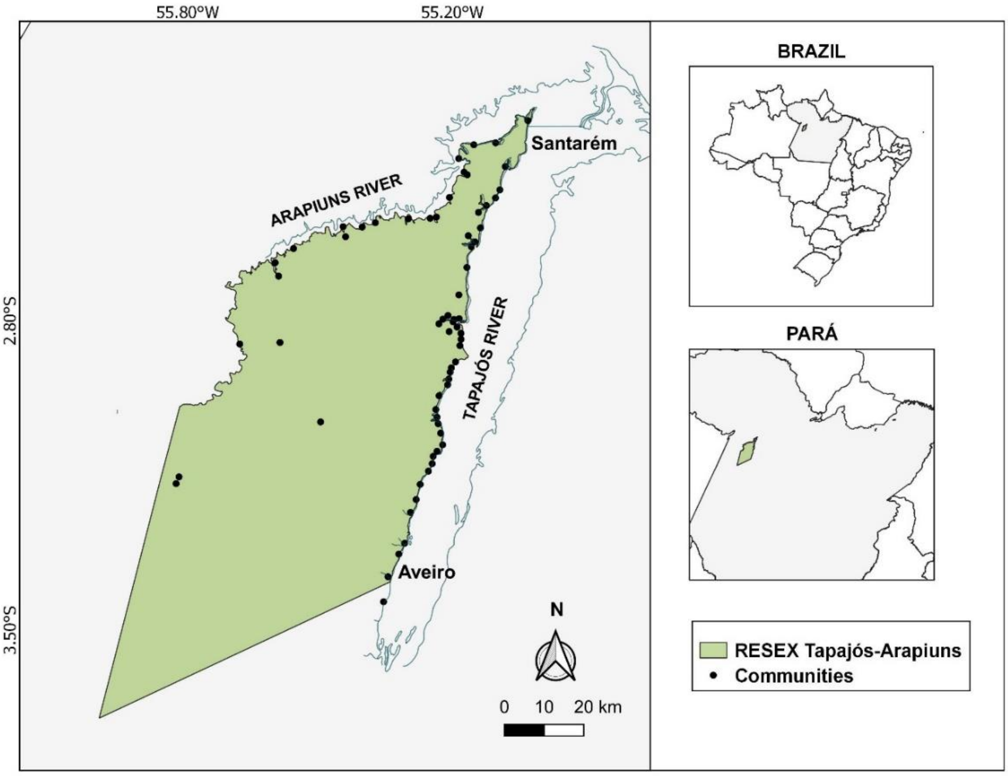


Fig. 1. Mapa da RESEX Tapajós-Arapiuns, com suas respectivas comunidades, localizada os municípios de Santarém e Aveiro (Pará).

A vegetação predominante é a floresta ombrófila densa, caracterizada por árvores de grande porte, com grande abundância de lenhosas, epífitas e lianas (VELOSO *et al.*, 1991, IBGE, 1992), em ambientes de *terra firme* (i.e., florestas não inundáveis). Outras tipologias vegetais comuns incluem cerrado, floresta secundária e florestas alagadas (CARVALHO-JUNIOR, 2008; MONTAG *et al.*, 2012). O clima é tropical úmido (Ami), com estação chuvosa de novembro a maio e seca de agosto a outubro (INPE & CPTEC, 2023).

Atualmente, a RESEX-TA possui 78 comunidades, distribuídas em áreas às margens dos rios Tapajós e Arapiuns (ANDRADE & SPÍNOLA, 2022), porém também existe uma minoria ao longo dos rios Maró, Inhambú e Igarapés do Mentai e Amorim, localizados mais ao interior (OLIVEIRA *et al.*, 2005; ICMBio, 2014). Todas as comunidades são estabelecidas em ambientes de *terra firme* (Fig. 2), nas quais os moradores dependem diretamente da exploração de recursos naturais existentes na área (caça, pesca e extração de produtos florestais madeireiros e não madeireiros) e/ou agricultura familiar e criação de pequenos animais (SAÚDE & ALEGRIA, 2012). A caça é permitida para fins de subsistência, e seu manejo depende de acordos internos firmados entre as comunidades e os gestores da reserva. Em meados de 2021, houve uma tentativa de extração de madeira por meio de um plano de manejo florestal comunitário, mas a falta de consulta popular prévia resultou na paralisação das atividades.

Fig. 2. Acima, comunidade ribeirinha, situada em ambiente de *terra firme*, na RESEX Tapajós-Arapiuns (Foto: Pollyana de Lemos) e abaixo, vista aérea da comunidade Vila de Anã, situada em ambiente de *terra firme* às margens do Rio Arapiuns, na RESEX Tapajós-Arapiuns. (Fonte: Monitora/PMPB/RESEX-TA, ICMBio).

Monitora e Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas na RESEX TA

O subprograma terrestre do Monitora prevê a implantação de, no mínimo, três estações amostrais por UC, onde são realizadas amostragens de quatro grupos bioindicadores (alvos globais): mamíferos de médio e grande porte, aves cinegéticas, borboletas frugívoras e plantas arbóreas (RIBEIRO, 2018). As estações amostrais para mamíferos e aves são trilhas de 5 km de comprimento, onde são realizadas diferentes metodologias de amostragem (NOBRE *et al.*, 2014).

Umas das primeiras UCs a integrar o Monitora foi a RESEX-TA. Devido as suas particularidades ambientais (mais de 90% de sua área preservada) e sociais (a RESEX mais populosa do Brasil), a implementação do Monitora somente teria fundamento para a gestão e beneficiários, se gerasse informações úteis para a tomada de decisão local (SPÍNOLA, 2015). Foi nesse sentido que oficinas foram realizadas com comunitários, a fim de questioná-los sobre quais outros componentes, além dos alvos globais, tinham interesse em monitorar. Foi então que surgiu a demanda pelo monitoramento dos animais caçados para subsistência, visto que os comunitários estavam preocupados com o impacto da atividade sobre as espécies cinegéticas e queriam responder uma importante pergunta: “Teremos bicho no futuro?”. Nestas mesmas oficinas, estabeleceram a necessidade da construção de um protocolo de monitoramento que respondesse a esses questionamentos (Fig. 3).

Fig. 3. Reunião realizada para a seleção dos alvos de monitoramento na RESEX Tapajós-Arapiuns ocorrida em 2014 no Sindicato dos Trabalhadores Rurais, em Santarém-PA. (Fonte: Monitora/PMPB/RESEX-TA, ICMBio).

Alguns meses depois, o protocolo foi elaborado e apresentado por uma especialista da área, discutido e validado por comunitários, gestores, consultores e demais envolvidos (CHIARAVALLOTI *et al.*, 2018). Na mesma ocasião, foi definido que 25 espécies terrestres, incluindo mamíferos de médio e grande porte, aves e répteis, seriam as espécies focais do chamado Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas na RESEX TA (Fig. 4). A implantação deste protocolo foi realizada via Projeto de Monitoramento Participativo da Biodiversidade (PMPB) em parceria firmada com o Instituto de Pesquisas Ecológicas (IPÊ) para viabilizar a implementação do Monitora, com seus alvos globais e complementares (RIBEIRO, 2018).

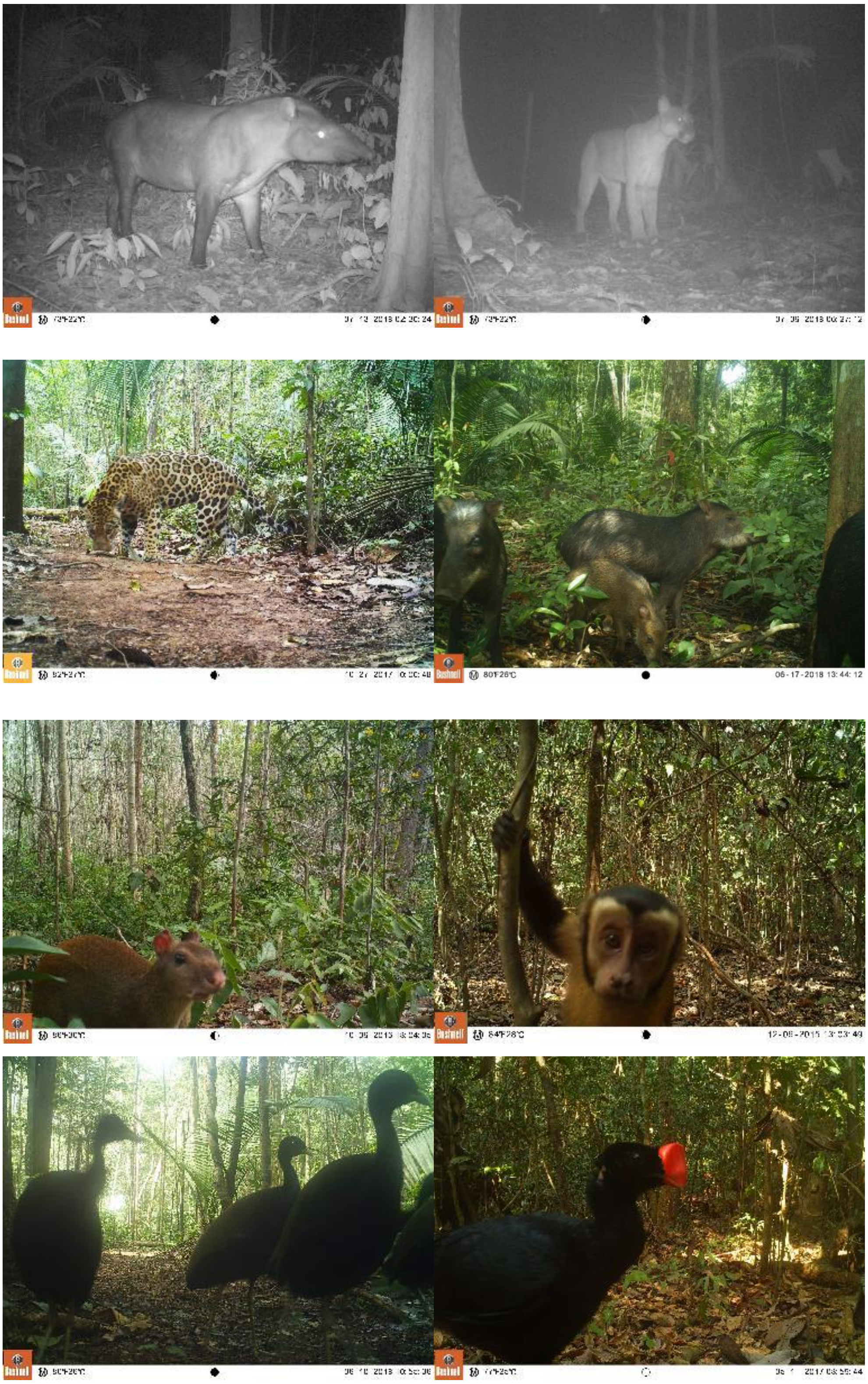


Fig. 4. Algumas espécies-alvos do Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas na RESEX-TA, registradas com o uso de armadilhas fotográficas. De cima, para baixo, da esquerda para direita: *Tapirus terrestres* (anta), *Puma concolor* (onça-parda), *Panthera onca* (onça-pintada), *Tayassu pecari* (queixada), *Dasyprocta croconota* (cutia), *Sapajus apela* (macaco-prego), *Psophia* sp. (jacamim) e *Pauxi tuberosa* (mutum). (Fonte: Monitora/PMPB/RESEX-TA, ICMBio).

Segundo a recomendação do protocolo, a RESEX-TA deveria adotar simultaneamente dois sub-protocolos de monitoramento: 1. Ocorrência e Abundância de espécies cinegéticas; e 2. Pressão de caça sobre as populações cinegéticas terrestres. Em ambos, o monitoramento deveria ocorrer de forma sistemática realizado por moradores locais (monitor da biodiversidade) devidamente capacitados e residentes das comunidades participantes do monitoramento (BENCHIMOL, 2014).

Para a realização do protocolo, a RESEX-TA teria que instalar no mínimo nove trilhas, de 5 km cada, inseridas em áreas próximas à 9 comunidades distintas, categorizadas em diferentes níveis de intensidade de caça (alta, média e baixa) (BENCHIMOL, 2014). Essa classificação foi baseada no mapeamento participativo do uso dos recursos naturais da RESEX-TA (ICMBio, 2011), documento que apresenta um mapa das áreas de caça e indica classes de intensidade de uso com base em informações de espécies caçadas.

Para a seleção das nove comunidades-polo foi primeiramente considerada a existência de duas trilhas previamente estabelecidas em duas comunidades, Boim e Cametá. As demais foram selecionadas conforme interesse da comunidade, categoria de intensidade de caça a qual pertenciam e viabilidade de estabelecimento das trilhas. Após a seleção, implantação do programa e início das coletas em 2014, houve entretanto uma desistência, restando assim oito comunidades-polo (Fig. 5, Tabela 1). Além disso, houve também a incorporação de quatro comunidades-anexas no programa, que passaram a contribuir apenas com o protocolo complementar de caça (Tabela 1).

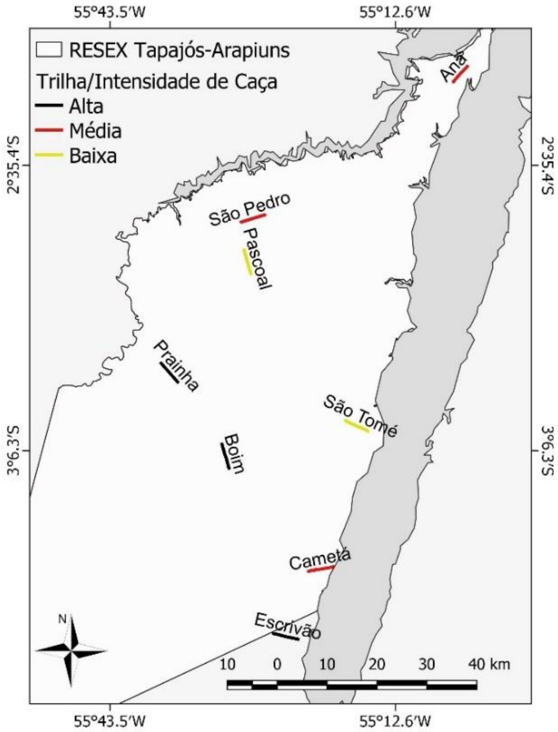


Fig. 5. Mapa da RESEX-TA com as trilhas (transectos) do Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas na RESEX-TA nas oito comunidades-polo, representadas de acordo com as categorias de intensidade de caça.

Tabela 1. Comunidades-polo e comunidades-anexa, com as respectivas categorias de intensidade de caça, números de famílias e monitores, participantes do Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas na RESEX-TA.

Categoria	**Especificação**	**Comunidade**	**N° famílias**	**Nº monitores**
Alta	Polo	Boim	113	1
Alta	Polo	Escrivão	91	4
Alta	Polo	Prainha do Maró	75	4
Alta	Anexa	Rosário	29	1
Alta	Anexa	Tucumatuba	86	1
Média	Polo	Cametá	207	3
Média	Polo	São Pedro	155	4
Média	Polo	Vila de Anã	86	2
Média	Anexa	Maripá	70	2
Média	Anexa	Pinhel	89	1
Baixa	Polo	São Tomé	65	4
Baixa	Polo	Pascoal	20	4

Para a seleção dos monitores, cada comunidade-polo indicou no mínimo três comunitários locais para participar do primeiro curso de capacitação de monitores, realizado em 2014. Esse curso teve a duração de cinco dias e resultou na formação de 32 monitores capacitados para atuar nas coletas de dados dos alvos globais e alvos complementares do programa Monitora na RESEX.

Hoje, após a primeira capacitação e mais três cursos de reciclagem (realizados em 2016, 2019 e 2022; Fig. 6), o protocolo da RESEX-TA envolve a participação de 31 monitores que recebem diárias de campo para execução das atividades do monitoramento em suas respectivas comunidades (Tabela 1).

Fig. 6. Capacitação dos monitores da biodiversidade. Foto oficial do curso de 2014 (esquerda) e do curso de reciclagem de 2022 (direita) (Fonte: Monitora/PMPB/RESEX-TA, ICMBio).

Todos os dados coletados são encaminhados à equipe técnica do Monitora/PMPB na RESEX. Informo que desde o início do PMPB, eu faço parte da equipe técnica do programa, tendo participados dos cursos, tabulação e análises dos dados. A seguir, descrevo brevemente os dois sub-protocolos do protocolo de monitoramento das espécies caçadas para subsistência na RESEX-TA.

Sub-protocolo I: Ocorrência e Abundância de espécies cinegéticas

Neste primeiro sub-protocolo, pares de monitores realizam nas proximidades de suas comunidades a técnica de amostragem de transecção linear. Esta metodologia tem sido amplamente utilizada em florestas tropicais (PERES, 1999; BENCHIMOL, 2016) e consiste em caminhadas em velocidade lenta e constante (em média 1 km por hora) ao longo de um transecto linear previamente estabelecido, buscando registros visuais das espécies alvo.

As amostragens têm sido realizadas nas oito trilhas localizadas nas comunidades-polo, apenas pela manhã (6h30 – 11h30) durante dez dias por ano, sempre no final do período chuvoso (junho e julho) (entre 2015 até hoje) e não ocorreram em dias chuvosos (Fig. 7). Sempre que um indivíduo ou grupo da espécie-alvo é registrado, as seguintes informações são obtidas: nome da espécie, tipo de registro, hora do encontro, localização do animal no transecto, número de indivíduos e distância perpendicular ao primeiro indivíduo detectado, subgrupo ou grupo (medido com trena) (Anexo 2). Tendo em vista que os monitores locais possuem ampla experiência em caminhadas na floresta, busca e identificação de animais, e cursos de treinamento apropriados foram fornecidos, assumimos que a coleta de dados tem sido bem realizada.

Fig. 7. Monitores da biodiversidade em campo preenchendo a ficha de dados enquanto realizam as amostragens de transecção linear do Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas na RESEX-TA (Fonte: Monitora/PMPB/RESEX-TA, ICMBio).

Sub-protocolo II: Pressão de caça sobre as populações cinegéticas terrestres

A amostragem das espécies que são caçadas localmente ocorre por meio do preenchimento de fichas de registros de evento de caça. De fácil entendimento e compreensão, a ficha pode ser preenchida por qualquer membro familiar após um evento independente de caça executada por algum integrante da família (Anexo 4) (BENCHIMOL, 2014).

Para esclarecer eventuais dúvidas de preenchimento das fichas, um monitor de cada comunidade-polo recebe treinamento específico, sendo também o responsável pela entrega e recebimento mensal das fichas de consumo de caça. As famílias participantes não são identificadas através de nomes e sim de números, para evitar qualquer constrangimento pela declaração dos eventos de caça.

Em particular, o formulário contém campos em branco para preenchimento de informações sobre o nome e a quantidade de espécies caçadas, a data e hora de início e término de cada evento, o número de caçadores envolvidos, o sucesso da caça e se a caça foi intencional ou oportunista (i.e., o caçador foi capaz de matar um animal enquanto estava envolvido em outras atividades, como a agricultura). A fim de manter a confidencialidade das informações fornecidas, cada família recebeu um termo de compromisso da equipe técnica do Monitora/PMPB (BENCHIMOL, 2014). Desde 2019, as amostragens deste sub-protocolo estão paralisadas devido à mudança de abordagem da gestão da RESEX e falta de recursos financeiros.

Estrutura da Tese

O objetivo desta tese foi avaliar projetos de monitoramento comunitários voltados aos vertebrados caçados em florestas tropicais, primeiro identificando estes projetos em uma escala global, e então concentrando-se em um único projeto e avaliando a influência da pressão antropogênica sobre as populações de vertebrados terrestres cinegéticos e os padrões de caça ao longo de uma série temporal. Neste sentido, busco fornecer uma contribuição substancial na discussão sobre o potencial de projetos comunitários em promover os três pilares da conservação: (i) fornecer informações a longo prazo sobre o recurso monitorado, (ii) empoderar as comunidades locais e (iii) implementar ações de manejo para a conservação da biodiversidade. Além disso, avaliamos se e como populações de vertebrados *in situ* e caçados são afetados por diferentes variáveis antropogênicas na RESEX-TA. Como resultado direto, esperamos contribuir com a gestão de uso da caça na RESEX-TA e avaliar a efetividade desta área protegida em proteger as populações animais, a fim de garantir sua persistência e uso sustentável ao longo do tempo. Por fim, como este é o primeiro protocolo de monitoramento do efeito da caça de subsistência em espécies cinegéticas do ICMBio, pretendemos contribuir para o seu aprimoramento a partir das nossas sugestões, a fim de que possa ser utilizado como um modelo de protocolo de monitoramento da caça de subsistência por outras áreas protegidas que tenham interesse em monitorar esse recurso.

Assim, esta tese está estruturada em 3 capítulos. O primeiro capítulo, intitulado "**A efetividade de projetos de monitoramento de base comunitária da fauna terrestre alvo de caça nos trópicos: uma revisão global"** (*Effectiveness of community-based monitoring projects of terrestrial game fauna in the tropics: a global review*) consiste em uma revisão sistemática da literatura. Nesta revisão, buscamos identificar todos os projetos, passados e vigentes, de monitoramento comunitário da fauna alvo de caça terrestre nos trópicos e, examinamos especificamente dezessete desses projetos em termos de custos, interrupção e efetividade. Este capítulo foi publicado na revista *Perspectives in Ecology and Conservation* em 2023 (21: 172–179).

O segundo capítulo, intitulado "**Monitoramento baseado na comunidade revela baixa pressão antrópica sobre a população de vertebrados cinegéticos em uma área protegida de uso sustentável na Amazônia"** (*Community-based monitoring reveals low anthropogenic pressure on game vertebrate population in a sustainable-use Amazonian protected area*) tem como objetivo avaliar como: (i) variáveis antrópicas (relacionadas à pressão de caça) e anos modelam os padrões de densidade geral e individual e biomassa geral de espécies terrestres alvos de caça, e (ii) as tendências temporais nas estimativas de densidade flutuam ao longo de uma série temporal. Os dados são oriundos do banco de dados de seis anos do Protocolo Complementar do Efeito da Caça de Subsistência sobre espécies cinegéticas do Monitora/PMPB na RESEX-TA. O manuscrito se encontra em revisão na revista *Environmental Conservation*.

O terceiro capítulo, intitulado "**A caça na área protegida de uso sustentável mais populosa da Amazônia: insights de 5 anos de monitoramento comunitário"** (*Game harvest in the most populated Amazonian sustainable-use protected area: insights from 5-years of community-based monitoring*) avalia (i) os efeitos das mesmas variáveis antrópicas e ano do Capítulo II sobre o perfil de caça e os padrões de produtividade de caça (i.e., CPUE), e (ii) as tendências temporais nas estimativas de CPUE na mesma área protegida.

REFERÊNCIAS

ANDRADE, D. F. C.; SPÍNOLA, J. N. **O Rio que nos une,** pp 205-216. Brasília. Brasil: ICMBio. 2022.

ANTUNES, A. P. *et al*. A conspiracy of silence: Subsistence hunting rights in the Brazilian Amazon. **Land use policy** 84: 1-11. 2019.

ANTUNES, A. P. *et al*. Empty forest or empty rivers? A century of commercial hunting in Amazonia. **Science Advances** 2: e1600936. 2016.

BRASHARES, J. S. *et al*. Wildlife decline and social conflict. **Science** 345(6195): 376-378. 2014.

BEIRNE, C. *et al*. Participatory monitoring reveals village-centered gradients of mammalian defaunation in central Africa. **Biological conservation** 233: 228-238. 2019.

BENCHIMOL, M. S. **Protocolo Complementar do Efeito da Caça de Subsistência sobre Espécies Cinegéticas na RESEX Tapajós-Arapiuns**. Programa de Monitoramento da Biodiversidade com relevância para o Clima, Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio. Brasília: ICMBio. 2014.

BENCHIMOL, M. Medium and Large-sized Mammals. In: LARSEN, T. H. (Ed.). **Core Standardized Methods for Rapid Biological Field Assessment**. Conservation International. USA: Arlington, VA, pp. 37-48. 2016.

BENÍTEZ-LÓPEZ, A. *et al*. The impact of hunting on tropical mammal and bird populations. **Science** 356: 180–183. 2017.

BENÍTEZ-LÓPEZ, A. *et al*. Intact but empty forests? Patterns of hunting-induced mammal defaunation in the tropics. **PLoS biology** *17*(5): e3000247. 2019.

BRANDON, K. Ecosystem services from tropical forests: review of current science. **Center for Global Development Working Paper** 380. 2014.

BODMER, R. E. *et al*. Managing wildlife to conserve Amazonian forests: population biology and economic considerations of game hunting. **Biological conservation** 67(1), 29-35. 1994.

BODMER, R. E.; EISENBERG, J. F.; REDFORD, K. H. Hunting and the Likelihood of Extinction of Amazonian Mammals: Caza y Probabilidad de Extinción de Mamiferos Amazónicos. **Conservation Biology** 11(2), 460-466. 1997.

BOGONI, J. A.; FERRAZ, K. M.; PERES, C. A. Continental-scale local extinctions in mammal assemblages are synergistically induced by habitat loss and hunting pressure. **Biological Conservation** 272: 109635. 2022.

BOGONI, J. A.; PERES, C. A.; FERRAZ, M. P. M. B. Effects of mammal defaunation on natural ecosystem services and human well being throughout the entire Neotropical realm. **Ecosystem Services** 45: 101173. 2020.

BRODIE, J. F.; FRAGOSO, J. M. V. Understanding the distribution of bushmeat hunting effort across landscapes by testing hypotheses about human foraging. **Conservation Biology** 35(3): 1009-1018. 2021.

BURTON, A. C. Critical evaluation of a long-term, locally-based wildlife monitoring program in West Africa. **Biodiversity and Conservation** 21: 3079-3094. 2012.

CAMPOS-SILVA, J. V.; HAWES, J. E.; PERES, C.A. Population recovery, seasonal site fidelity, and daily activity of pirarucu (*Arapaima* spp.) in an Amazonian floodplain mosaic. **Freshwater Biology** 64(7): 1255-1264. 2019.

CARDILLO, M. *et al*. Multiple causes of high extinction risk in large mammal species. **Science** 309: 1239–1241. 2005.

CARVALHO JUNIOR, E. A. R. **Caçando onças na Reserva Extrativista Tapajós-Arapiuns**. Relatório de atividades: Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio). Santarém: ICMBio. 2008.

CHIARAVALLOTI, R. M. *et al*. Monitoramento participativo de caça de subsistência da Reserva Extrativista Tapajós-Arapiuns (PA): caminhos para uma participação social efetiva. **Biodiversidade Brasileira** 9(2), 203-218. 2018.

CONSTANTINO, P. *et al*. Empowering local people through community-based resource monitoring: a comparison of Brazil and Namibia. **Ecology and Society** 17(4): 22. 2012.

CONSTANTINO, P. A. L. Dynamics of hunting territories and prey distribution in Amazonian Indigenous Lands. **Applied Geography** 56: 222-231. 2015.

CRONEMBERGER, C. *et al*. **Monitoramento da biodiversidade como instrumento de apoio à evolução da gestão em unidades de conservação da Amazônia**. In: TÓFOLI CF, *et al.* Diálogos da Conservação – Monitoramento Participativo da Biodiversidade: Contribuições para Conservação das Áreas Protegidas da Amazônia. São Paulo: IPÊ -Instituto de Pesquisas Ecológicas, 18–33. 2023a. Disponível em: https://www.ipe.org.br/publicacoes/ipe.

CRONEMBERGER, C. *et al*. Social Participation in the Brazilian National Biodiversity Monitoring Program Leads to Multiple Socioenvironmental Outcomes. **Citizen Science: Theory and Practice** 8(1): 32, 1–15. 2023b.

DANIELSEN, F. *et al*. On participatory biodiversity monitoring and its applicability: a response to Yoccoz et al. and Rodrıguez. **Oryx** 37(4). 2003.

DANIELSEN, F.; BURGESS, N. D.; BALMFORD, A. Monitoring matters: examining the potential of locally-based approaches. **Biodiversity & Conservation** 14: 2507-2542. 2005.

DANIELSEN, F. *et al*. Local participation in natural resource monitoring: a characterization of approaches. **Conservation Biology** 23(1): 31-42. 2009.

DANIELSEN, F. *et al*. Testing focus groups as a tool for connecting indigenous and local knowledge on abundance of natural resources with science-based land management systems. **Conservation Letters** 7: 380-389. 2014.

DANIELSEN, F. *et al*. The concept, practice, application, and results of locally based monitoring of the environment. **BioScience** 71(5): 484-502. 2021.

DANIELSEN, F. *et al*. Community monitoring of natural resource systems and the environment. **Annual Review of Environment and Resources** 47: 637-670. 2022.

DAVIDSON, A. D. *et al*. Multiple ecological pathways to extinction in mammals. **Proceedings of the National Academy of Sciences** 106(26): 10702-10705. 2009.

DE PAULA M. J. *et al*. Hunting sustainability within two eastern Amazon Extractive Reserves. **Environmental Conservation** 49(2):90-98. 2022.

DIELE-VIEGAS, L. M.; PEREIRA, E. J. D. A. L.; ROCHA, C. F. D. The new Brazilian gold rush: Is Amazonia at risk?. **Forest Policy and Economics** 119: 102270. 2020.

DIRZO, R. *et al*. Defaunation in the Anthropocene. **Science** 345(6195): 401-406. 2014.

FA, J. E.; FUNK, S. M.; NASI, R. **Hunting wildlife in the tropics and subtropics**. Cambridge University Press. 2022.

FRAGOSO, J. M. V. *et al*. 2016. Line transect surveys underdetect terrestrial mammals: Implications for the sustainability of subsistence hunting. **PloS one** 11(4): e0152659. 2016.

EL BIZRI, H. R. *et al*. Involving local communities for effective citizen science: Determining game species' reproductive status to assess hunting effects in tropical forests. **Journal of Applied Ecology** 58: 224-235. 2020.

ESBACH, M. S. Estimating mammal density from track counts collected by Indigenous Amazonian hunters. **Perspectives in Ecology and Conservation**. 2023.

RIPPLE, W. J. *et al*. Bushmeat hunting and extinction risk to the world's mammals. **Royal Society open science** 3(10): 160498.

INGRAM, D. J.; *et al*. Wild meat is still on the menu: Progress in wild meat research, policy, and practice from 2002 to 2020. Annual Review of Environment and Resources 46: 221-254. 2021.

INSTITUTO NACIONAL DE PESQUISAS ESPACIAIS, CENTRO DE PREVISÃO DE TEMPO E ESTUDOS CLIMÁTICOS. **Precipitação diária 2023** [www document]. URL http://clima1.cptec.inpe.br/estacaochuvosa/pt. 2023.
INSTITUTO BRASILEIRO DE GEOGRAFIA E ESTATÍSTICA**. Manual técnico da vegetação brasileira**. IBGE - Instituto Brasileiro de Geografia e Estatística. Rio de Janeiro. 1992.

INSTITUTO CHICO MENDES DE CONSERVAÇÃO DA BIODIVERSIDADE. **Plano de Manejo da Reserva Extrativista Tapajós-Arapiuns**. Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio. Brasília: ICMBio. 2014.

INSTITUTO CHICO MENDES DE CONSERVAÇÃO DA BIODIVERSIDADE. **Mapeamento participativo do uso dos recursos naturais na RESEX Tapajós-Arapiuns**. Conservação Internacional-CI, Associações da Reserva Extrativista Tapajós-Arapiuns-Tapajoara, Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio. Santarém: ICMBio. 2011.

INTRAGOVERNMENTAL PLATAFORM ON BIODIVERSITY AND ECOSYSTEM SERVICES (IPBES). http://www.ipbes.net/. 2015.

IWAMURA, T. *et al*. Agent-based modeling of hunting and subsistence agriculture on indigenous lands: Understanding interactions between social and ecological systems. **Environmental Modelling & Software** 58: 109-127. 2014.

JEROZOLIMSKI, A.; PERES, C. A. Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. **Biological Conservation** 111(3) : 415-425. 2003.

LIMA, D.; POZZOBON J. Amazônia socioambiental: sustentabilidade ecológica e diversidade social. **Estudos Avançados** 19: 45–76. 2005.

LINDENMAYER, D. B.; LIKENS, G. E. The science and application of ecological monitoring. **Biological conservation** 143(6): 1317-1328. 2010.

LOVETT, G. M. *et al*. Who needs environmental monitoring?. **Frontiers in Ecology and the Environment** 5(5): 253-260. 2007.

LUZAR, J. B. *et al*. Large-scale environmental monitoring by indigenous peoples. **BioScience** 61: 771-781. 2011.

LUZAR, J. B.; SILVIUS, K. M.; FRAGOSO, J. M. Church affiliation and meat taboos in indigenous communities of Guyanese Amazonia. **Human Ecology** 40: 833-845. 2012.

MAYOR, P.; EL BIZRI, H.; BODMER, R. E. *et al*. Assessment of mammal reproduction for hunting sustainability through community-based sampling of species in the wild. **Conservation Biology** 31(4): 912-923. 2017.

MANES, S., *et al*. Endemism increases species' climate change risk in areas of global biodiversity importance. **Biological Conservation** *257*: 109070. 2021

MONTAG, F. F. A. *et al*. **Ictiofauna na Reserva Extrativista Tapajós-Arapiuns, Estado do Pará, Brasil**. Universidade Federal do Pará (UFPA). 2012.

NOBRE, R. D. A. *et al*. **Monitoramento da biodiversidade: roteiro metodológico de aplicação**. Brasília: ICMBio. 2014.

NOVARO, A. J.; REDFORD, K. H.; BODMER, R. E. Effect of hunting in source–sink systems in the Neotropics. **Conservation Biology** 14: 713–721. 2000.

OLIVEIRA, A. C. M.; JUNIOR, O. C.; CHAVES, R. Gestão participativa e a atividade de caça na Reserva Extrativista do Tapajós – Arapiuns, Santarém, PA. **Raízes** 23: 42–51. 2005.

OLIVEIRA, M. A.; CALOURO, A. M. Medium-sized and large mammals of the Cazumbá-Iracema Extractivist Reserve, Acre, Brazil. **Check List** 16(1): 127-136. 2020.

OLIVEIRA, M. A., *et al*. Comparing hunting practices of urban and rural inhabitants in Western Amazonia: the role of religion, wealth, and livelihoods. **Research Square**. 2023.

PARRY, L.; BARLOW, J. O. S.; PERES, C. A. Hunting for sustainability in tropical secondary forests. **Conservation Biology** 23(5): 1270-1280. 2009.

PEREIRA, R. C. *et al*. Monitoramento *in situ* da biodiversidade: Proposta para um sistema brasileiro de monitoramento da biodiversidade. Brasília: ICMBio. 2013.

PERES, C. A. General guidelines for standardizing line-transect surveys of tropical forest primates. **Neotropical primates** 7(1): 11-16. 1999.

PERES, C. A. Effects of subsistence hunting on vertebrate community structure in amazonian forests. **Conservation Biology** 14: 240–253. 2000.

PERES, C., BARLOW, J., HAUGAASEN, T. Vertebrate responses to surface wildfires in a central Amazonian forest. **Oryx** 37(1): 97-109. 2003.

PERES, C. A.; NASCIMENTO, H. S. Impact of game hunting by the Kayapó of south-eastern Amazonia: implications for wildlife conservation in tropical forest indigenous reserves. **Biodiversity & Conservation** 15(8): 2627-2653. 2006.

PEZZUTI, J. C. B. *et al*. **A caça e a pesca no Parque Nacional do Jaú, Amazonas**. In: BORGES, S. H. *et al*. (eds.). Janelas para a Biodiversidade no Parque Nacional do Jaú. Fundação Vitória amazônica, Manaus, 213-230. 2004.

PIRES, M. M.; GALETTI, M. Beyond the “empty forest”: The defaunation syndromes of Neotropical forests in the Anthropocene. **Global Ecology and Conservation** e02362. 2022.

PONCE-MARTINS, M. *et al*. Assessing the contribution of local experts in monitoring Neotropical vertebrates with camera traps, linear transects and track and sign surveys in the Amazon. **Perspectives in Ecology and Conservation**. 2022.

REIS Y. **Aspectos da caça de subsistência em comunidades extrativistas no oeste do estado do Pará (Amazônia, Brasil)**. 2017. 150f. Dissertação (Mestrado em Zoologia) - Museu Emilio Goeldi, Pará, Brasil, 2017.

RIBEIRO, K. T. Estratégia do Programa Nacional de Monitoramento da Biodiversidade– Programa Monitora: estrutura, articulações, perspectivas. Brasília: ICMBio. 2018.

REDFORD, K. H.; ROBINSON, J. G. The game of choice: patterns of Indian and colonist hunting in the Neotropics. **American anthropologist** 89(3): 650-667. 1978.

RIPPLE, W. J. *et al*. Caça de carne de caça e risco de extinção para os mamíferos do mundo. **Royal Society open science** 3(10): 160498. 2016.

ROBINSON, J.; BENNETT, E. L. **Hunting for sustainability in tropical forests**. Columbia University Press. 2000.

SAMPAIO, R. *et al*. Vertebrate population changes induced by hunting in Amazonian sustainable-use protected areas. **Biological Conservation** 284: 110206. 2023.

SAÚDE & ALEGRIA. Mapeamentos Participativos e elaboração de Planos de Uso: uma experiência nas comunidades de Anã, Arimum, Atodi e Vila Amazonas nas margens do Rio Arapiuns. SAÚDE & ALEGRIA, Santarém-PA. 2012.

SCABIN, A. B.; PERES, C. A. Hunting pressure modulates the composition and size structure of terrestrial and arboreal vertebrates in Amazonian forests. **Biodiversity and Conservation** 30(12): 3613–3632. 2021.

SILVA, A. F. S. **O Uso da Fauna Cinegética e o Consumo de Proteína Animal em Comunidades Rurais na Amazônia Oriental**. 2008. 81f. Dissertação (Mestrado em Zoologia). Museu Emilio Goeldi, Pará, Brasil. 2008.

SIRÉN, A.; MACHOA, J. Fish, wildlife, and human nutrition in tropical forests: a fat gap? **Interciencia** 33:186–93. 2008.

SOUZA-MAZUREK, R. R. *et al*. Subsistence hunting among the Waimiri Atroari indians in central Amazonia, Brazil. **Biodiversity & Conservation** 9(5), 579-596. 2000.

SOUZA, T. T. *et al*. Monitoramento da biodiversidade para as estratégias de conservação: experiência do Programa Monitora. In Tófoli, C. F.; *et al*. (Eds.). Monitoramento participativo da biodiversidade. Aprendizados em evolução. 2nd ed. São Paulo: IPÊ – Instituto de Pesquisas Ecológicas/MEMNON, 68–83. 2019.

SPÍNOLA, J. N. A Construção do Programa de Monitoramento Participativo da Biodiversidade na Reserva Extrativista Tapajós-Arapiuns, In: VII Seminário de Pesquisa e Iniciação Científica do Instituto Chico Mendes de Conservação da Biodiversidade. Anais do VII Seminário de Pesquisa e Iniciação Científica do Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio. Brasília: ICMBio. p. 00-000. 2015.

STAFFORD, C. A.; PREZIOSI, R. F.; SELLERS, W. I. A pan-neotropical analysis of hunting preferences. **Biodiversity and Conservation** 26: 1877-1897. 2017.

VEIGA, J. E. ; EHLERS, E. Diversidade biológica e dinamismo econômico no meio rural. **Economia do meio ambiente: teoria e prática. Rio de Janeiro**, Ed. Campus, 271-290. 2003.

VELOSO, H. P.; RANGEL-FILHO, A. L. R.; LIMA, J. C. A. **Classificação da vegetação brasileira, adaptada a um sistema universal**. IBGE. 1991.

VAN VLIET, N. *et al*. Bushmeat and human health: assessing the evidence in tropical and sub-tropical forests. **Ethnobiology and Conservation** 6(3):1–45. 2017.

YOCCOZ, N. G.; NICHOLS, J. D.; BOULINIER, T. Monitoring of biological diversity–a response to Danielsen et al. **Oryx** 37(4): 410-410. 2003.

YOUNG, H. S. *et al*. Patterns, causes, and consequences of anthropocene defaunation. **Annual review of ecology, evolution, and systematics** 47:333-358. 2016.

WEINBAUM, K. Z. *et al*. Searching for sustainability: are assessments of wildlife harvests behind the times?. **Ecology letters** 16(1): 99-111. 2013.

CAPÍTULO 1

Effectiveness of community-based monitoring projects of terrestrial game fauna in the tropics: a global review

Article published in *Perspectives in Ecology and Conservation* in May 2023
(https://doi.org/10.1016/j.pecon.2023.03.005)


Foto: Monitora, ICMBio

Effectiveness of community-based monitoring projects of terrestrial game fauna in the tropics: a global review

Yasmin Maria Sampaio dos Reis[1*], Maíra Benchimol[2]

[1,2]Applied Ecology & Conservation Lab, Programa de Pós-graduação em Ecologia e Conservação da Biodiversidade, Universidade Estadual de Santa Cruz, Rodovia Ilhéus-Itabuna, km 16, Salobrinho, 45662-000 Ilhéus, Bahia, Brazil

E-mail addresses: yasmin.sampaioreis@gmail.com (Y. Reis)*, mairabs02@gmail.com (M. Benchimol)

*Corresponding author.

Abstract

Biodiversity monitoring projects comprise key conservation strategies established to minimize biodiversity loss. Particularly, community-based monitoring projects have recently been implemented worldwide. This approach favors three conservation pillars: provision of information on monitored resource through time, local people's empowerment, and management practices. We conducted a systematic literature review to identify all past and current community-based monitoring projects of terrestrial game fauna in the tropics, and specifically examined seventeen of those projects in terms of costs, interruption and effectiveness. We identified a total of 52 projects, mostly located in the Amazon. We revealed an annual cost of US$0.24/hectare/project, with most of these initiatives interrupted due the lack of funding. We also noticed that the absence of data analyses comprised the main obstacle for the assessment on monitored game fauna through time, while empowerment was hampered by the lack of intensive local participation at different stages of monitoring. Finally, we observed that most management actions resulted in community rules and applications, including local bylaws governing resource use. We highlight that community-based programs can be more effective if they engage local people at all monitoring stages, build solid partnerships to ensure long-term funding and translate the outcomes into management practices for the monitored fauna.

Key words: Biodiversity monitoring, locally based monitoring, natural resource management, mammals, birds, tropics.

Introduction

Biodiversity monitoring comprises a fundamental step towards species conservation worldwide, as it allows to detect population changes over time (Lindenmayer & Likens, 2010). Those well elaborated monitoring projects, in which goals and study design are clearly defined (Yoccoz et al., 2003), can provide long-term information on the status and trends of target species and/or natural resources through time, and inform appropriate management actions (Danielsen et al., 2005). Specially in tropical forests, which hold the richest biodiversity on Earth, biodiversity monitoring is one of the main conservation tools established to detain the current biodiversity loss resulting from a myriad of anthropogenic activities, including habitat destruction, climate warming and the overexploitation of wildlife (Veiga & Ehlers, 2009). Assessing the main obstacles and contributions of monitoring projects throughout systematic review can provide useful information to enhance success for both existing and future programs across the tropics.

Over the last two decades, community-based monitoring projects (i.e., conducted by local people) have been implemented in several tropical ecosystems (Danielsen et al., 2021). This approach has been debated among researchers who question their ability to deal with sampling error and thus producing reliable inference to monitored populations (Yoccoz et al., 2003; Burton, 2012), and those who recognize the extensive knowledge of local people and defend its potential as could favor three conservation pillars−(i) provision of information on monitored resource through time, (ii) empowerment of local stakeholders and (iii) implementation of management actions (Danielsen et al., 2021). Empowerment can be defined as a participatory process through which local people gain greater influence over their lives and acquire improved management over used natural resources (Maton, 2008).

In tropical protected areas, which are mainly located in emerging countries, the establishment of biodiversity monitoring projects has been hampered by low financial and human resources (Danielsen et al., 2009). In addition, there are cases where the government still cuts funding to scientific and academic endeavors, as recently seen in Brazil (Tollefson, 2019). In such realities, the community-based approach poses as a cost-effective conservation initiative that seeks to encourage the community involvement, meet local needs, and strengthen existing local systems for the monitoring and managing of natural resources (Danielsen et al., 2003). Indeed, community-based initiatives are gaining prominence in those protected areas managed collaboratively with local people, where understanding the main threats to species and/or natural resources used by residents is fundamental for proposing further mitigation actions (Luzar et al., 2011). In Tanzania, for instance, 181 management interventions were recommended based on outcomes provided by a community-based monitoring program focused on fauna and flora (Topp-Jørgensen et al., 2005). In western Amazonia, the community-based monitoring of the world's largest scaled freshwater fish (*Arapaima* cf. *gigas*) was reflected in the population recovery of the species even outside protected areas, revealing that management performed by local communities can effectively promote biodiversity conservation and empower local people (Campos-Silva et al., 2019).

Medium to large-sized mammals and birds comprise two of the main target groups in monitoring programs across tropical forests. Both represent most of forest vertebrate biomass (Peres, 2000), play key roles in forest functionality, and provide several ecosystem services (Bogoni et al., 2020). Yet they are also the main targets of hunters, with several species having succumbed locally until population depletion in hunted forest sites, mainly in Africa and Asia (Benítez-López et al., 2019). Indeed, mammal species diversity has been locally reduced in nearby villages of Gabon due to hunting, with large and hunted species being most frequently recorded far from the villages (Beirne et al., 2019). Conversely, studies in the Amazon have shown that terrestrial vertebrate populations can be more resilient to hunting (e.g., Iwamura et al., 2014), as humans have been intensively hunting for many years but animal populations are not extirpated. This is likely because many vast upland areas remain inaccessible to hunters, generating a positive source-sink dynamic that can rescue overharvested populations in heavily hunted areas (Antunes et al., 2016; Pereira et al., 2017). For this, Fragoso and colleagues (2016) emphasized the need to use robust and accurate methodologies to properly assess hunted vertebrate populations and propose adequate management actions.

Analyzing the extent to which community-based wildlife monitoring is effective in achieving the three conservation pillars (provision of information on monitored resource through time, empowerment of local people and implementation of management actions) is fundamental to identify both limitations and positive outcomes from already performed projects. We contribute to this discussion by conducting a systematic literature review to identify, map and examine both existing and previous community-based monitoring projects (i.e., minimum of one year of monitoring) of terrestrial game forest species in the tropics. In particular, we assess the main causes of monitoring interruptions, compared the annual costs and analyzed the effectiveness of each project through the analysis of strategies used to promote the three conservation pillars (see our objectives in the Table

S1). In addition, we identify the obstacles faced by each initiative and discuss how community-based monitoring projects focused on terrestrial game fauna can be improved and therefore better achieve success.

Material and methods

Data source and analysis

We conducted a literature search to identify published and unpublished studies on community-based monitoring projects for terrestrial game fauna (i.e., frequently hunted medium to large-sized forest mammals and/or birds) in the tropics. We considered community-based monitoring projects, those occurring for at least one year, where local people (traditional groups or local rangers) were directly involved in data collection. Using Scopus and Google Scholar bibliographic databases, we performed searches until March 2022 using different keywords in English: program OR project AND participatory OR community OR community base OR citizen science AND monitoring AND mammal * OR vertebrate * OR bird * OR hunting. Subsequently, searches were made on Google Scholar using the same keywords translated into Portuguese and Spanish. Both scientific articles and grey literature (i.e., technical reports, dissertations and thesis) were searched. Given that the literature search failed in identifying some studies, we further included additional studies that we were previously aware of. As a criterion, the study needed to explicitly provide information on either existing or previous community-based monitoring projects focused on terrestrial game fauna (i.e., at least one game mammal or bird species). We thus carefully examined the title and abstract of each study and excluded duplicate references and non-related studies (Fig. 1). During this screening process, when more than one study for the same project was obtained, all were selected. Therefore, we ended up with 62 studies (55 publications, 2 book chapters, 1 technical series, 2 master dissertation and 2 PhD thesis) referring to 52 projects (Fig. 1). We thus extracted several information from the selected studies to achieve our objectives (Table S1).

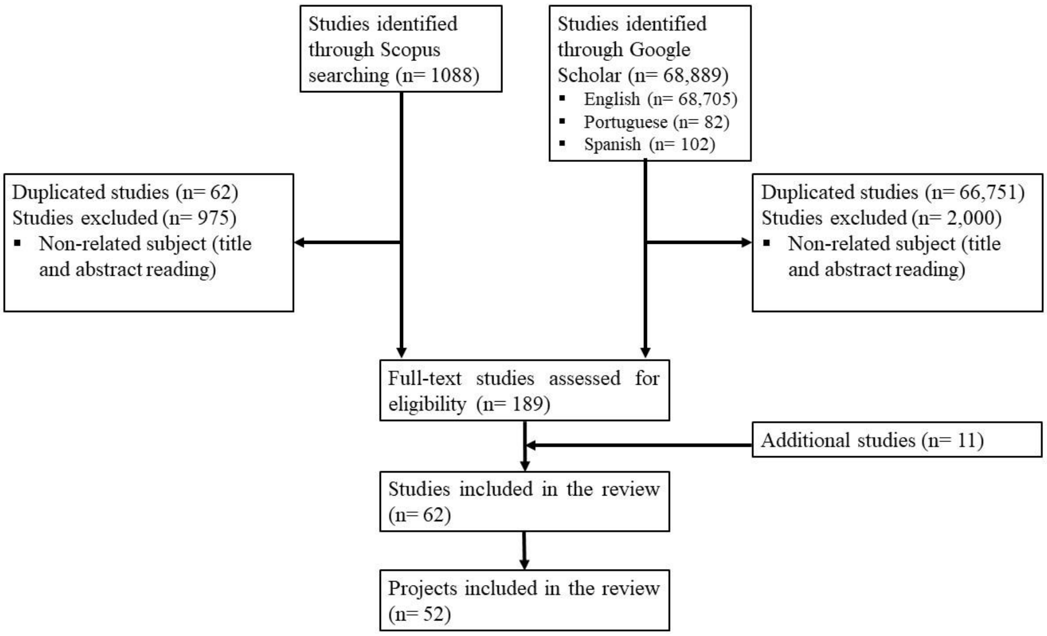


Fig. 1. Flow diagram of the dataset selection process used in our study to identify existing and past community-based monitoring projects of terrestrial game fauna in the tropics.

We finally contacted by e-mail the authors and/or professionals involved on each project to obtain further information (see form consisted of open and closed questions in the Table S2). Out of 52 contacted projects, we obtained responses from 17 monitoring programs. We then carefully examined these projects in terms of interruption, average annual cost US$/project, and effectiveness. We defined that a project was effective when it adopted strategies that promoted the three conservation pillars (see Table S1). We are aware that both the definition and evaluation of ‘effectiveness’ were based on our perspective (i.e., researcher’s viewpoint), which might not follow the conception of local people engaged on the monitored project.

Regarding the first pillar, we evaluated the following strategies adopted by the 17 projects: (i) spatio-temporal data analyses, (ii) percentage of tabulated data and (iii) percentage of data analyzed (using any type of analysis). The strategy (i) was scored as 1 if 'exists' and 0 if 'does not exist'. The strategies (ii) and (iii) were categorized into 4 classes (0-25%, 25-50%, 50-75%, 75-100%), and scored from 1 to 4, respectively. In addition, we evaluated the number of publications resulted from each project.

In relation to the empowerment of local stakeholders, we examined if each of the 17 projects integrated the six main strategies related to empowerment according to the literature (see Constantino et al., 2012 and Costa, 2019) − if (i) local people participated in the elaboration of the project (species definition and criteria to select local monitors),

(ii) the monitored resource is a source of meat and income, (iii) adequate training (i.e., theoretical and practices classes) was provided to the monitors and (iv) local people directly participated in data entry, (v) analysis and (vi) return of the results. Each strategy was scored as 1 if ‘it was adopted’ and 0 if ‘it was not adopted’. Lastly, we evaluated if management actions were performed based on outcomes from each program (Danielsen et al., 2021). In this case, we scored as 1 if ‘exist’ and 0 if ‘do not exist’. Finally, we ranked the 17 monitoring projects in terms of effectiveness, based on the sum of scores considering the three conservation pillars. To obtain the total effectiveness score, the score for each pillar received a weight of 1.

Results

Of the 52 projects identified, 58% monitored both the game fauna (including mammals and birds) *in situ* and harvest aspects (Table S3). Most of these initiatives are located in South America (n = 32; Fig. 2), especially in Amazonia (n = 24), whereas other 13 projects occur in Africa, three in Asia and four in North America. In particular, we highlight that 11 Amazonian projects were identified from 10 additional studies that we were previously aware of (Fig. 1). The majority of projects recorded in our study (83%) were established in protected areas, whereas eleven were identified within unprotected areas. Moreover, we found that all projects relied on the effort of local people in data collection and most started after 2000 (n = 37). Other details are shown in Table S3.

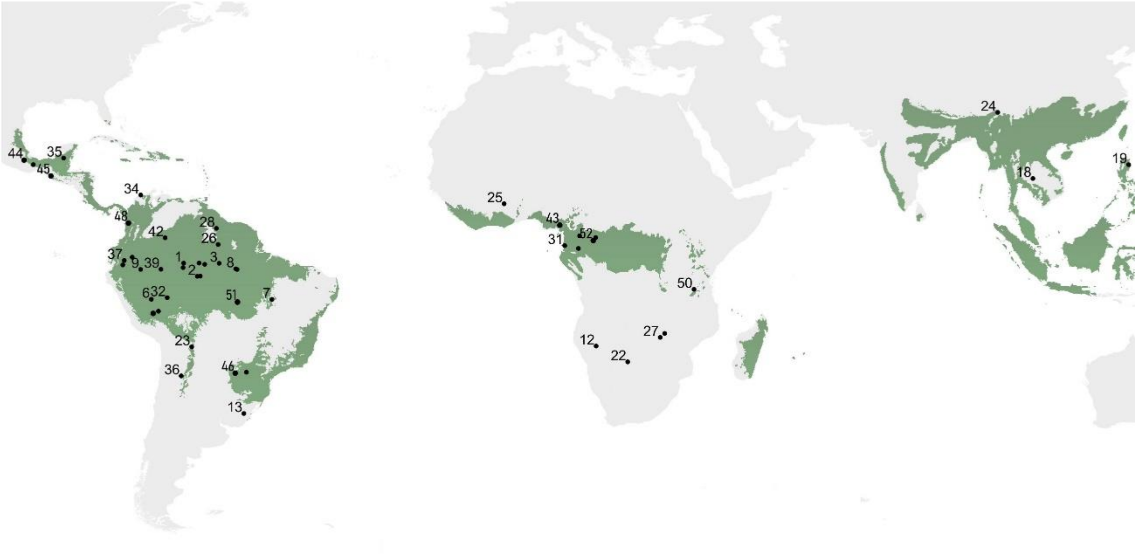
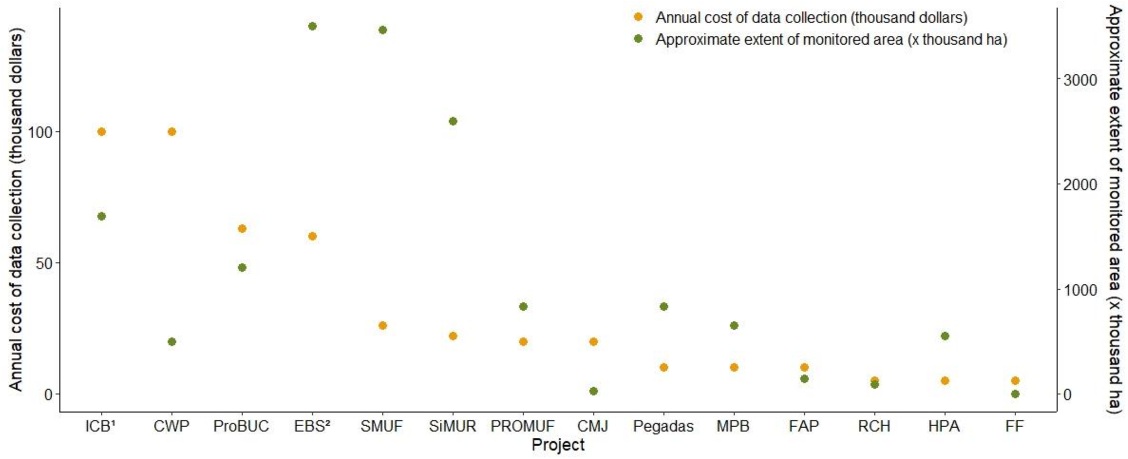


Fig. 2. Location for the 52 existing and past community-based monitoring projects (black dots) of terrestrial game fauna in the tropics (colored in green). Each number represents one project (the respective names are described in Table S3).

Interruption, annual costs and effectiveness

Considering those 17 projects for which we obtained further information (see Table S4), we found that most were interrupted (65%), either after the end of the project deadline (36%) or for unexpected reasons (64%). The lack of financial resources was the main reason for unplanned interruption (86%) and temporary suspension (67%) of projects. Other reported reasons were related to COVID-19 pandemic (14%), changes in priority of project managing (7%), conflicts between community and the project's managers (7%) and change of responsible technician (7%).

Fourteen of the 17 projects provided annual costs information (Fig. 3). The mean annual cost for collecting monitoring data was US$0.24/hectare (SD ± 0.54) per project. Yet, the annual costs substantially varied among projects, ranging from US$5,000.00 (e.g. HPA) to US$300,000.00 (the case of ICB). We found no relationship between the annual cost of data collection (US$) and the approximate extent of monitored area (ha). For example, the most cost-effective project (EBS) invested US$60,000.00 to cover an area of 16,604,500 hectares (Fig. 3).



ICB[1] – Annual cost of data collection (x3)
EBS[2] - Approximate extent of monitored area referring to 16,604,500 hectares

Fig. 3. Relationship between the annual cost of data collection (US$) and approximate extent of monitored area (ha) by each analyzed project. Data were not available for URIL, HXIL and Monitora. See Table S2 for a detailed description of each project.

All 17 projects used strategies to provide information on the monitored resource through time (Table S5). In particular, we found that 53% performed spatio-temporal data analysis, and 94% and 65% of projects had, respectively, performed data entry and analysis for at least 75% of data.

All 17 projects published scientific studies. A total of 200 publications, including articles, book or book chapters and gray literature were identified (Fig. 4). We observed that there is no relationship between the monitoring time and the number of published studies, although some long-term initiatives have published more (e.g. the HPA has been monitoring hunting for 30 years and published 92 studies), other short-term initiatives have published less (e.g. the CMJ monitored the fauna for a year and published one study) (Fig. S1).

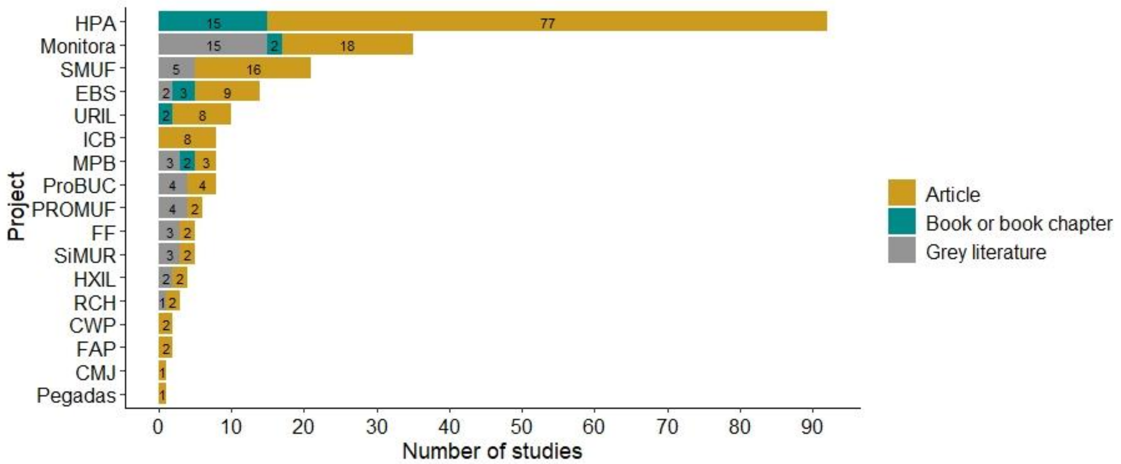


Fig. 4. Number of published studies (article, book or book chapter and grey literature) resulting from each of the 17 community-based monitoring projects of terrestrial game fauna in the tropics.

Despite two exceptions, 15 projects demonstrated that local citizens were likely empowered (Table S6), given that local people were directly involved in project elaboration (target species definition and criteria to select monitors) and provided adequate training for local monitors. Moreover, the effective participation of local people in returning project outcomes also promoted empowerment in some initiatives (29%). Conversely, three strategies were not usually adopted because they were not the focus of most projects – the effective participation of local people in data entry (12%), data analysis (12%) and the importance of the resource as a source of meat and income (18%). Nine initiatives (53%) resulted in management actions (Table S7), mostly encompassed by community rules and applications, such as local bylaws governing resource use. We observed that 44% of projects created slaughter rules; for instance, the SMUF in Brazilian Amazonia contributed to the creation of a quota for the subsistence hunting of the lowland paca (*Cuniculus paca*). In addition, 33% contributed to the creation of management plans, 22% implemented a zoning of hunting areas within protected areas, 22% banned hunting of at least one game species and 11% put in practice the existing rules of the wildlife management plans. Out of the nine projects that endorsed management actions, 33% provided evidence that resulted in the monitoring of further species. For instance, game species including the lowland paca and agouti (*Dasyprocta* spp.) started to be monitored through specific sampling programs in the SMUF, aiming to understand their life-history and ecology. Further, one initiative discouraged the creation of new human settlements aiming to improve the sustainability of hunting in the region. We finally observed that 22% of projects provided management actions at a wider scale. These included communitarian management of natural resources in Namíbia and the establishment of a protected area in the Paso Centurión region, in Uruguay.

By ranking the 17 projects in terms of effectiveness, based on the sum of scores obtained considering all conservation pillars, we noticed that almost half of projects were not fully effective because did not promote this triad (Table S8). The determining factor for this was the failure in conducting management actions (47% of initiatives). Still in terms of effectiveness, we showed no relationship between the average annual cost (dollars/ha) and project effectiveness, as costliest (e.g. FF and CWP) and cheaper (EBS and SMUF) projects were considered efficient (Fig. S2).

Discussion

This study provides the first systematic literature review of community-based monitoring projects of terrestrial game fauna across the entire tropical region. These initiatives have been expanding over the last two decades to provide management strategies toward species conservation in the long-term (Danielsen et al., 2021), yet an evaluation of their effectiveness has never been assessed. We identified 52 projects, which mostly monitored both game fauna and harvest, and relied on the effort of local stakeholders in data collection. We revealed an annual cost of US$0.24/hectare/project, with most of these initiatives being interrupted or suspended due to the lack of funding. Finally, we observed that the absence of management actions precluded most projects to be fully effective. Based on our data compilation, we provide recommendations for ongoing monitoring projects in addition to highlight which aspects should be prioritized when planning monitoring projects focused on terrestrial game species across the tropics.

Spatial distribution of monitoring projects

Most community-based monitoring projects are located in South America, mainly in Amazonian protected areas. Although the Amazon faces an under-sampling of terrestrial vertebrate inventories carried out in protected areas (Bogoni et al., 2021), with a noticeable knowledge gap especially for carnivores and xenarthras (Cruz et al., 2021; Feijó et al., 2022), this biome hosts about half of the tropical forests on Earth (Hansen et al., 2013) mainly concentrated in Brazil. In addition, the Amazon is also home to a great number of indigenous and local people who rely on forests to survive (Lima & Pozzobon, 2005). Government institutions and collaborative networks between researchers/managers/participants of NGOs and OSCIPs are also frequent, which also explains this result. Indeed, the Amazon has been an example of community-management, where local people have been monitoring faunal species and playing a central role in management (e.g. Castello et al., 2009; Petersen et al., 2016). For instance, the community-based monitoring program focused on the pirarucu fish (*Arapaima gigas*) along the Juruá River has resulted on its population recovery and contributed to the development of traditional *ribeirinho* communities (Campos-Silva et al., 2020).

A great number of projects was also recorded in Afrotropical forests, which was expected given the high biodiversity in this continent and the annual meat harvests, higher than in other tropical areas (Fa et al., 2002). In contrast, only three programs were identified in the Brazilian Atlantic Forest. This biome, one of the global biodiversity hotspots, has been historically deforested and transformed into urban and agricultural environment – almost 75% of the Brazilian population inhabits this biome, most in urban centers. This explains the lower number of recorded community-based projects, with monitoring data coming essentially from scientists (e.g. Chiarello, 2000; Kaizer et al., 2021). Finally, the small number of initiatives recorded in Asia, probably caused by the low research effort for mammal species threatened by hunting (Riplle et al., 2016), calls attention to the importance of establishing novel programs in this region. In fact, this continent retains several threatened species, and illegal hunting constitutes the greatest current threat to wild vertebrates in Asia (Harrison et al., 2016).

Interruption and annual costs

Our results revealed that several projects were interrupted. Although interruptions in some projects were planned, the majority were stopped unexpectedly when funding had finished or due to changes in project management and conflicts between the community and the project's managers. Moreover, the absence of solid partnerships to ensure long-term funding is currently the main threat to the interruption of ProBUC and MPB. For example, the MPB in Brazil has always received funding from a nongovernmental organization (IPÊ) and the Amazon Region Protected Areas Program (ARPA), but the former is no longer contributing financially from mid-2022 on. As a result, the project will become exclusively dependent on ARPA and therefore vulnerable to government's temporarily suspension, which has already occurred in early 2021. Indeed, our findings reveal that the scarcity of ongoing financial support poses as the main obstacle to the continuity of community-based monitoring projects, which has also been shown in other studies (Rijsoort & Jinfeng, 2005; Costa et al., 2019). Moreover, we also recognize that other barriers can directly affect the progress of projects. For instance, the COVID-19 pandemic was responsible for the total or partial suspension of activities of some investigated projects (e.g. Monitora).

There is a wide range of total cost among the investigated projects (from US$5,000.00 to US$300,000.00/project or from US$ 0,004 to US$ 2,000/hectare), regardless the monitored area. Moreover, we found no relationship between the annual cost x extent of monitored area, and annual cost x project effectiveness. It is likely that other factors substantially influence costs, including the sampling methods used and the payment for services (Bucheli & Marinelli, 2013).

Effectiveness of monitoring projects

Similar to other studies (e.g. Danielsen et al., 2014; 2021), our results reveal that community-based monitoring provides useful information about monitored game fauna.

We demonstrated that more than half of the projects performed spatio-temporal data analysis and were successful in providing information on the monitored resource through time (e.g. the case of EBS, SMUF and SiMUR). Conversely, failures in spatio-temporal data analyses of some projects comprised the main obstacles for providing these information (e.g. Pegadas, PROMUF and FF). This might be related to the prioritization of investments and human efforts to perform data collection, overlooking the post-collection data management and analyses (Bucheli & Marinelli, 2013). As already pointed out by several researchers, we emphasize the importance of carefully planning each stage of the monitoring project, including the spatial and temporal delimitation of the resource to be monitored (Yoccoz et al., 2003). Although data entry was prioritized by most programs, we noticed that many data sets have not yet been analyzed for some projects, which therefore hampers the assessment of species trends over time~~.~~

Our findings evidence that most projects adopted strategies intended to empower local people, demonstrating their importance in strengthen communities. In particular, these strategies allowed empowerment to occur mainly in the psychological or cognitive dimension (Maton, 2008). Individuals engaged in project elaboration feel proud to get involved in building relationships with external researchers (Constantino et al., 2012). Furthermore, the adequate training led monitors to acquire technical and biological knowledge about the monitored resources (Danielsen et al., 2009). Their participation in data entry, data analysis, and in returns of the results, relevant to psychological and social empowerment, was not often in most projects. These strategies are important to build trust with communities (Luzar et al., 2011), increase the alignment between community and project goals (Noss et al., 2005), and increase the sense of ownership of the project and outcomes, thereby enhancing the local influence on faunal recourse decisions (Danielsen et al., 2021). This was likely the case of the CWP and URIL.

Finally, our results reveal that almost half of the initiatives failed to subsidize management practices, although the recognition of its potential (Villaseñor et al., 2016; Danielsen et al., 2021). Specifically, the duration of projects can be decisive for the creation of management actions, as longer projects implemented more management strategies. In addition, the degree of involvement of local members can determine how information will be converted into management actions (Fernandez-Gimenez et al., 2008; Danielsen et al., 2010), because it may lead to ownership of the natural resource management process (Marrocoli et al., 2018). Moreover, some projects failed in performing spatio-temporal data analyses which would be able to subsidize management practices. Conversely, projects such as Monitora and FAP were able to provide findings about monitored species through time, but local managers of their respectively protected areas did not use the information to implement management actions. In this sense, Danielsen et al. (2005) emphasize that decision policy makers should be directly involved in all stages of monitoring, and therefore establish management actions towards wildlife conservation.

Although not all projects have contributed to the management of wild fauna, we noticed that important management actions were conducted, including the establishment of local hunting grounds and slaughter rules, creation of management plans, temporary or permanent ban of hunting on vulnerable species and support for national policies. Overall, our study shows that management actions are based on rules and applications in the community, such as local statutes governing the use of resources (Van Rijsoort & Jinfeng, 2005) and zoning of hunting areas. In particular, when discussing with local people, these actions become encouraging management strategies, locally respected (Oliveira & Calouro, 2019), and aim not only to protect species, but also to ensure long-term benefits to local communities (Danielsen et al., 2014).

Conclusions and recommendations

Although an emerging number of community-based monitoring projects has been established in tropical forests over the last two decades, several obstacles are hampering their effectiveness in promoting the three pillars of conservation. These include the lack of funding, intensive local participation at different stages of monitoring, spatio-temporal data analyses and management actions for game fauna associated with the projects. We therefore recommend that on-going and novel programs should (i) build solid partnerships with universities, research centers, conservation NGOs or community associations that guarantee long-term funding, solving the problem of project interruption due to lack of financial resources; (ii) engage local people in all stages of monitoring, as empowering people can enhance their interest to continuing monitoring the fauna, even in limited financial circumstances; (iii) invest in human resources to perform spatio-temporal data analysis, which are fundamental to evaluate species fluctuations through space and time. We particularly emphasize the importance of building partnerships with universities, higher education institutions and research centers, specialized in research design, statistics course and article writing, to assist in both data analyses and dissemination of the results to the academia. Finally, we recommend (iv) translate the program outcomes into management actions and thus effectively contribute to safeguard species and, in those sustainable use protected areas, guarantee their long-term sustainability.

Conflict of interest

The authors declare no conflict of interest.

Funding

This study was supported by the Coordination for the Improvement of Higher Education Personnel (CAPES) - Brazil (CAPES - Financing code 001).

Acknowledgements

YMSR was supported by a scholarship from the Coordination for the Improvement of Higher Education Personnel (CAPES) - Brazil (CAPES). We are grateful to the authors and professionals involved on each project analyzed, which provide additional information, in special to Hani El Bizri who discussed some ideas during project conceptualization. We also thank anonymous reviewers who provided useful and constructive comments to the manuscript.

References

Antunes, A. P., Fewster, R. M., Venticinque, E. M., Peres, C. A., Levi, T., Rohe, F., & Shepard Jr, G. H. (2016). Empty forest or empty rivers? A century of commercial hunting in Amazonia. Science advances, 2(10), e1600936. DOI: 10.1126/sciadv.1600936.

Bennett, E.L., Robinson, J.G.,1999. Hunting for sustainability: The Start of a Syntesis. In: Robinson, J.G., Bennett, E.L. (Eds), Hunting for sustainability in tropical forests (biology and resource series). Columbia University Press, New York, pp. 499-519.

Benítez-López, A., Alkemade, R., Schipper, A.M., Ingram, D.J., Verweij, P.A., Eikelboom, J.A.J., Huijbregts, M.A.J., 2017. The impact of hunting on tropical mammal and bird populations. Science. 356, 180-183. http://sro.sussex.ac.uk/67430/.

Bogoni, J.A., Peres, C.A., Ferraz, K.M., 2020. Effects of mammal defaunation on natural ecosystem services and human well being throughout the entire Neotropical realm. Ecosystem Services. 45, 101173. https://doi.org/10.1016/j.ecoser.2020.101173.

Bogoni, J.A., Peres, C.A., Ferraz, K.M, 2021. Medium-to large-bodied mammal surveys across the Neotropics are heavily biased against the most faunally intact assemblages. Mammal Review. https://doi.org/10.1111/mam.12274

Bucheli, M.G.A., Marinelli, C.E., 2014. Custo-eficiência de programas de monitoramento participativo da biodiversidade: o caso do PROBUC (Programa de monitoramento da biodiversidade e do uso de recursos naturais em unidades de conservação estaduais do Amazonas). Biodiversidade Brasileira-BioBrasil. 1, 47-68. https://doi.org/10.37002/biobrasil.v%25vi%25i.332.

Campos-Silva, J.V., Hawes, J.E., Peres, C.A., 2019. Population recovery, seasonal site fidelity, and daily activity of pirarucu (*Arapaima* spp.) in an Amazonian floodplain mosaic. Freshwater Biology, 64(7), pp.1255-1264. https://doi.org/10.1111/fwb.13301.

Campos-Silva, J.V., Hawes, J.E., Freitas, C.T., Andrade, P., Peres, C.A., 2020. Community-based management of Amazonian biodiversity assets. In Participatory Biodiversity Conservation (pp. 99-111). Springer, Cham.

Castello L., Viana J.P., Watkins G., et al., 2009. Lessons from integrating fishers of Arapaima in small-scale fisheries Management at the Mamirauá Reserve, Amazon. Environ Manag. 43, 197–209. https://doi.org/10.1007/s00267-008-9220-5.

Chiarello, A.G., 2000. Density and population size of mammals in remnants of Brazilian Atlantic Forest. Conservation Biology. 14 (6), 1649-1657. https://doi.org/10.1111/j.1523-1739.2000.99071.x.

Constantino, P.L., Fortini, L.B., Kaxinawa, F.R.S., Kaxinawa, A.M., Kaxinawa, E.S., Kaxinawa, A.P., Kaxinawa, L.S., Kaxinawa, J.M., Kaxinawa, J.P., 2008. Indigenous collaborative research for wildlife management in Amazonia: The case of the Kaxinawá, Acre, Brazil. Biological Conservation. 141, 2718-2719. https://doi.org/10.1016/j.biocon.2008.08.008.

Constantino, P.A.L., Carlos, H. S. A., Ramalho, E.E., Rostant, L., Marinelli, C.E., Teles, D., Fonseca-Junior, S.F., Fernandes, R.B., Valsecchi, J., 2012. Empowering local people through community-based resource monitoring: a comparison of Brazil and Namibia. *Ecology and Society*, *17*(4). https://www.jstor.org/stable/26269211.

Costa, D.C., (PhD Thesis) 2019. Monitoramento participativo como subsídio para a gestão adaptativa de unidades de conservação no Amazonas. Universidade Federal do Amazonas, Amazonas, Brasil.

Danielsen, F., Burgess, N.D., Balmford, A., 2005. Monitoring matters: examining the potential of locally-based approaches. Biodiversity & Conservation. 14, 2507-2542. https://doi.org/10.1007/s10531-005-8375-0.

Danielsen, F., Burgess, N.D., Balmford, A., Donald, P.F., Funder, M., Jones, J. P., Alviola, P., Balete, D.S., Blomley, T., Brashares, J., Child, B., Enghoff, M., Fjeldså, J., Holt, S., Hübertz, H., Jensen, A.E., Jensen, P.M., Massao, J., Mendoza, M.M., Ngaga, Y., Poulsen, M. K., Rueda, R., Sam, M., Skielboe, T., Stuart-Hill, G., Topp-Jørgensen, E., Yonten, D., 2009. Local participation in natural resource monitoring: a characterization of approaches. Conservation biology. 23, 31-42. https://doi.org/10.1111/j.1523-1739.2008.01063.x.

Danielsen, F., Burgess, N.D., Jensen, P.M., Pirhofer-Walzl, K., 2010. Environmental monitoring: the scale and speed of implementation varies according to the degree of peoples involvement. J Applied Ecol. 47, 1166–1168. https://doi.org/10.1111/j.1365-2664.2010.01874.x.

Danielsen, F., Jensen, P.M., Burgess, N.D., Coronado, I., Holt, S., Poulsen, M.K., Rueda, R.M., Skielboe, T., Enghoff, M., Hemmingsen, L.H., Sørensen, M., Pirhofer-Walzl, K., 2014. Testing focus groups as a tool for connecting indigenous and local knowledge on abundance of natural resources with science-based land management systems. Conservation Letters. 7, 380-389. https://doi.org/10.1111/conl.12100.

Danielsen, F., Enghoff, M., Poulsen, M.K., Funder, M., Jensen, P.M., Burgess, N.D., 2021. The concept, practice, application, and results of locally based monitoring of the environment. BioScience, 71(5), 484-502. https://doi.org/10.1093/biosci/biab021.

Diduck, A., 1999. Critical education in resource and environmental management: learning and empowerment for a sustainable future. Journal of Environmental Management. 57, 85-97. http://dx.doi.org/10.1006/jema.1999.0289.

El Bizri, H.R., Fa, J.E., Lemos, L.P., Campos-Silva, J.V., Vasconcelos Neto, C.F., Valsecchi, J., Mayor, P., 2020. Involving local communities for effective citizen science: Determining game species' reproductive status to assess hunting effects in tropical forests. Journal of Applied Ecology. 58, 224-235. https://doi.org/10.1111/1365-2664.13633.

Fa, J.E., Peres, C.A., Meeuwig, J., 2002. Bushmeat exploitation in tropical forests: an intercontinental comparison. Conservation biology. 16 (1), 232-237. https://doi.org/10.1046/j.1523-1739.2002.00275.x.

Fernandez-Gimenez, M.E., Ballard, H., Sturvtevant, V.E., 2008. Adaptive management and social learning in collaborative and community-based monitoring: a study of five community-based forestry organizations in the western USA. Ecol Soc. 13, 4. http://www.ecologyandsociety.org/vol13/iss2/art4/.

Fragoso, J.M.V., Levi, T., Oliveira, L.F., Luzar, J.B., Overman, H., Read, J.M. and Silvius, K.M., 2016. Line transect surveys underdetect terrestrial mammals: Implications for the sustainability of subsistence hunting. PloS one, 11(4), p. e0152659. https://doi.org/10.1371/journal.pone.0152659.

Hansen, M.C., Potapov, P.V., Moore, R. et al., 2013. High-resolution global maps of 21st-century forest cover change. Science. 342 (6160), 850-853. https://doi.org/10.1126/science.1244693.

Harrison, RD, Sreekar, R., Brodie, JF, Brook, S., Luskin, M., O'Kelly, H., ... & Velho, N., 2016. Impactos da caça nas florestas tropicais do Sudeste Asiático. Conservation Biology, 30 (5), 972-981. https://doi.org/10.1111/cobi.12785.

Holck, M., 2007. Participatory forest monitoring: an assessment of the accuracy of simple cost-effective methods. Biodiversity Conservation. 17, 2023-2036. https://doi.org/10.1007/s10531-007-9273-4.

Hockley, N.J., Jones, J.P.G., Andriahajaina, F.B., Manica, A., Ranambitsoa, E.H., Randriamboahary, J.A., 2005. When should communities and conservationists monitor exploited resources? Biodiversity and Conservation. 14, 2795-2806. http://dx.doi.org/10.1007/s10531-005-8416-8.

Iwamura, T., Lambin, E. F., Silvius, K. M., Luzar, J. B., Fragoso, J. M. 2014. Agent-based modeling of hunting and subsistence agriculture on indigenous lands: Understanding interactions between social and ecological systems. Environmental Modelling & Software. 58, 109-127. https://doi.org/10.1016/j.envsoft.2014.03.008.

Izurieta, A., Sithole, B., Stacey, N., Hunter-Xenie, H., Campbell, B., Donohoe, P., Brown, J., Wilson, L., 2011. Developing Indicators for monitoring and evaluating joint management effectiveness in protected areas in the Northern Territory, Australia. Ecol Soc. 16, 9. http://www.ecologyandsociety.org/vol16/iss3/art9/main.html.

Kaizer, M.C., Alvim, T.H., Novaes, C.L., McDevitt, A.D., Young, R.J., 2021. Snapshot of the Atlantic Forest canopy: surveying arboreal mammals in a biodiversity hotspot. bioRxiv. https://doi.org/10.1101/2021.03.08.434374.

Lima, D., Pozzobon J., 2005. Amazônia socioambiental: sustentabilidade ecológica e diversidade social. Estud Avançados. 19, 45–76. https://doi.org/10.1590/s0103-40142005000200004.

Luzar, J.B., Silvius, K.M., Overman, H., Giery, S.T., Read, J.M., Fragoso, J.M., 2011. Large-scale environmental monitoring by indigenous peoples. BioScience. 61, 771-781. https://doi.org/10.1525/bio.2011.61.10.7.

Maton K.I., 2008. Empowering community settings: agents of individual development, community betterment, and positive social change. American Journal of Community Psychology. 41, 4–21. https://doi.org/10.1007/s10464-007-9148-6.

Noss, A.J., Oetting, I., Cuéllar, R.L., 2005. Hunter self-monitoring by the Isoseño-Guaraní in the Bolivian Chaco. Biodiversity & Conservation. 14, 2679-2693. https://doi.org/10.1007/s10531-005-8401-2.

Oliveira, M.Á., Calouro, A.M., 2019. Hunting agreements as a strategy for the conservation of species: the case of the Cazumbá-Iracema Extractive Reserve, state of Acre, Brazil. Oecologia Australis, 23(2), 357-366. https://doi.org/10.4257/oeco.2019.2302.13.

Peres, C.A., 2000. Effects of subsistence hunting on vertebrate community structure in Amazonian forests. Conservation Biology. 14, 240-253. http://dx.doi.org/10.1046/j.1523-1739.2000.98485.x.

Pereira, P.M., Valsecchi, J., Queiroz, H., 2019. Spatial patterns of primate hunting in riverine communities in Central Amazonia. Oryx, 53(1), 165-173. https://doi.org/10.1017/S0030605317000199.

Petersen T.A., Brum S.M., Rossoni, F. et al., 2016. Recovery of Arapaima sp. populations by community-based management in floodplains of the Purus River, Amazon. J Fish Biol. 89, 241–248. https://doi.org/10.1111/jfb.12968.

Pound, B., Snapp, S., McDougall, C., Braun, A., 2003. Managing natural resources for sustainable livelihoods: uniting science and participation, Earthscan, London, UK.

Ripple, WJ, Abernethy, K., Betts, MG, Chapron, G., Dirzo, R., Galetti, M., ... & Young, H., 2016. Caça de carne de caça e risco de extinção para os mamíferos do mundo. Royal Society open science, 3 (10), 160498. https://doi.org/10.1098/rsos.160498.

Tollefson, J., 2019. 'Tropical Trump'sparks unprecedented crisis for Brazilian science. Nature, 572(7768), 161-163.

Wenny, D.G., 2000. Seed dispersal, seed predation, and seedling recruitment of a neotropical montane tree. Ecolog. Monogr. 70, 331-351. https://doi.org/10.1890/0012-9615(2000)070[0331:SDSPAS]2.0.CO;2

Van Rijsoort J., Jinfeng Z., 2005. Participatory resource monitoring as a means for promoting social change in Yunnan, China. Biodivers. Conserv. 14, 2543–2573. https://doi.org/10.1007/s10531-005-8377-y.

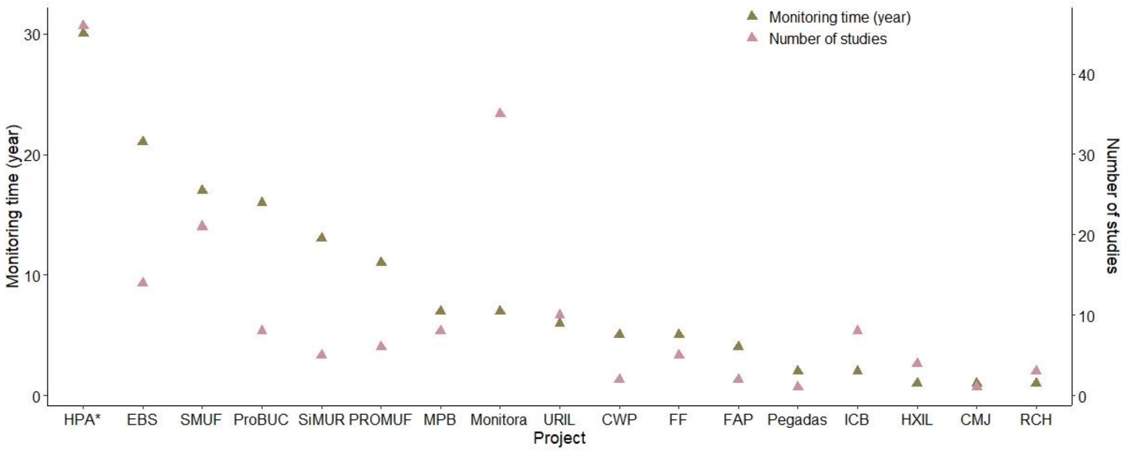
Veiga, J.E., Ehlers, E., 2003. Diversidade biológica e dinamismo econômico no meio rural, em Peter, M et al. (Eds.), Economia do meio ambiente: teoria e prática. Rio de Janeiro, Ed. Campus, pp.271-290.

Yoccoz, N.G., Nichols, J.D., Boulinier, T., 2003. Monitoring of biological diversity–a response to Danielsen et al. Oryx. 37(4), 410-410. https://doi.org/10.1017/S0030605303000747.

Villasenor, E., Porter-Bolland, L., Escobar, F., Guariguata, M.R., Moreno-Casasola, P., 2016. Characteristics of participatory monitoring projects and their relationship to decision-making in biological resource management: a review. Biodiversity and conservation. 25, 2001-2019. https://doi.org/10.1007/s10531-016-1184-9.

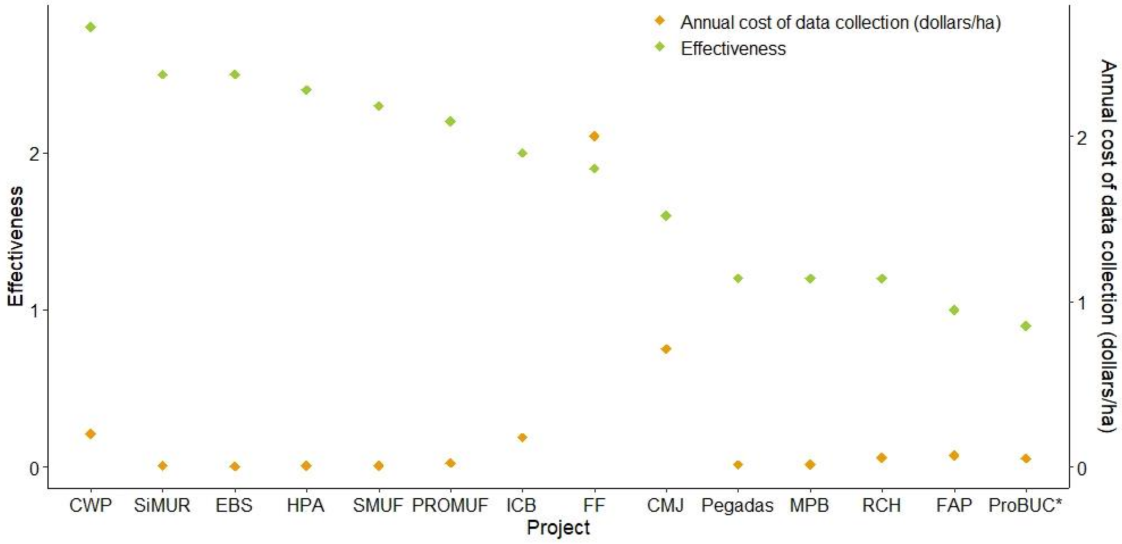
Appendix A. Supplementary data

Figure captions



HPA* - Number of studies (x2)

Fig. S1. Relationship between the monitoring time (year) and number of studies published by each analyzed project.



* some information about effectiveness was not obtained through articles and forms.

Fig. S2. Relationship between the effectiveness and annual cost of data collection (dollars/ha) of each analyzed project.

Table captions

Table S1. Information extracted from the selected studies in relation to objectives of this review: (a) projects identification and mapping, (b) assessment of the main causes of monitoring interruptions, (c) comparison of the annual costs and (d) effectiveness analysis based on strategies used by each program to promote each conservation pillar.

Objective	Information collected
(a) Project identification and mapping	- year of project implementation - geographic coordinates - biome, state and country - inside or outside protected area - number of communities/villages and human population - number of communities/villages and average number of people attended by the project - number of community members and non-community members (technicians, collaborating researchers, managers, etc.) involved in the project - management duty - source of funding - sampling techniques and sampling effort per year - data collector's profile (i.e., done by local people and/or researchers)
(b) Interruptions	- period (year of project implementation and conclusion) - reason for interruption (e.g., the project deadline has finished, lack of financial resources, lack of communities' interest, conflicts between communities and the project's managers)
(c) Annual cost	- annual cost of data collection - US$ (including daily rates/remuneration, costs with field logistics and project material)
(d) Effectiveness	
1º pillar (provision of information on monitored resource through time)	- data entry and data analysis (how much was planned and analysed) - list of publications related to the project
2º pillar (empowerment of local stakeholders)	- if adequate training was provided to the monitors (with theoretical and practices classes) - return of results, data entry and data analysis process (actors involved – communitarian, trainee without higher education, trainee with higher education or researcher) - stakeholders who participated in the elaboration of the project (definition of targets and criteria for the choice of monitors) - importance of the monitored resource
3º pillar (implementation of management actions)	- species that start to be monitored afterwards from project information - species banned from being hunted due to the obtained results - results led to management strategies - results led to protective policy

Table S2. Form in English sent to researchers of community-based monitoring projects of terrestrial game species in the tropics.

UNIVERSIDADE ESTADUAL DE SANTA CRUZ
PRÓ-REITORIA DE PESQUISA E PÓS-GRADUAÇÃO
PÓS-GRADUAÇÃO EM ECOLOGIA E CONSERVAÇÃO DA BIODIVERSIDADE

Project identification and mapping

Project name: ____.

Year of creation: ______________. Year of implementation: ___________.

Location (if possible, with geographic coordinates): ______________________

Biome: ________________. State: _____________. Country: _____________.

Is it a protected area? yes ( ) no ( )

If so, what´s the name? ____.

Number of communities / villages in the protected area: ________________.

Protected area population: __________.

Number of communities / villages attended by the project: ________.

Average number of the population attended by the project: _________.

Number of community members __________ and non-community members (technicians, collaborating researchers, managers, etc.) ____________ involved in the project.

Project management and identification of the main obstacles in proposing wildlife management strategies

Main objective of the project:

Assess the efficiency of the protected area ( ) promote environmental education ( ) generate information to assist the management of the monitored resource ( ) conduct scientific research ( ) other ( )__________

Management duty: communities ( ) NGO ( ) CSO ( ) state government ( ) federal government ( ) other ( ) ____________________________________.

Origin of funding: state government ( ) federal government ( ) international ( ) other ( ) _____________________________________.

Who collects the data receives daily rates/remuneration? yes ( ) no ( )

Annual cost of data collection:

0-5 thousand dollars ( ) 6-10 thousand dollars ( ) 11-15 thousand dollars ( )

16-20 thousand dollars ( ) other ( ): ____________________________.

Sampling techniques: line-transect ( ) sign surveys ( ) camera-trapping ( ) hunting calendar ( ) other ( ): ____.

Sampling effort per year of each technique:

____.

How was the training for monitoring carried out?

With theoretical classes ( ) with theoretical and practical classes ( ) with the presence of specialized researchers ( )

Has the project been interrupted? yes ( ) no ( )

If so, for how long? ______________________.

Is the monitoring still occurring? yes ( ) no ( )

If not, when did it end? ____________________________________.

If not, why did it end?:

The project deadline has finished ( ) lack of financial resources ( ) lack of communities interest ( ) conflicts between communities and the project's managers ( ) other ( ):

____.

Who is in charge for data entry?

Communitarian ( ) trainee without higher education ( ) trainee with higher education ( ) researcher ( ) other ( ): ____.

Who performs data analyses?

Communitarian ( ) trainee without higher education ( ) trainee with higher education ( ) researcher ( ) other ( ): ____.

How much has been tabulated? 0-25% ( ) 25-50% ( ) 50-75% ( ) 75-100% ( )

How much has been analysed? 0-25% ( ) 25-50% ( ) 50-75% ( ) 75-100% ( )

Is the data publicly accessible? yes ( ) no ( )

If so, how to proceed to access the data (request)?:

____.

How are the results made available?

Meeting with the community council / deliberative council ( ) community meeting ( ) monitors meeting ( ) newsletters ( )

other ( ) _____________________________________.

Who returns the results?

Communitarian ( ) researchers ( ) protected area management ( ) project technicians ( )

Year of the last return of the results: ___________________.

Stakeholders who participated in the elaboration of the program (definition of targets and criteria for the choice of monitors):

Communities ( ) researchers ( ) NGO ( ) CSO ( ) state government ( ) federal government ( ) other ( ) ____________________________________.

Main project contributions in proposing strategies for managing wildlife

Does the project work as initially proposed? yes ( ) no ( )

What is the importance of the monitored resource for the communities / villages?

Conservation ( ) main source of meat ( ) main source of income ( ) other ( )______

Is there a temporal and spatial monitoring of the resource? yes ( ) no ( )

Have you published the monitoring program? yes ( ) no ( )

Which source? article ( ) dissertation ( ) thesis ( ) report ( ) other ( )

Publication(s) reference(s):

____

____

Has any species or resource start to be monitored afterwards from program information?

yes ( ) no ( )

The species or monitored resource in interested to:

Researchers ( ) community ( ) protected area management ( ) program technicians ( )

Has any species or resource been banned from being hunted due to the results of the program?

yes ( ) no ( )

Did results promote any resource management strategies?

____.

Did results promote any protective policy? If so, which one?

____.

Table S3. Characteristics of the 52 existing and past community-based monitoring projects of terrestrial game fauna in the tropics identified in our study. The first 17 projects were examined in terms of interruption, costs and effectiveness given that researchers, technicians or protected area manager answered a detailed form.

Project	Location	Protected area	Period	Resource monitored	Game fauna sampling methods	References
1 - Fauna Use Monitoring System (SMUF)	Brazil, Amazonas	Sustainable Development Reserve (SDR) Mamirauá and Amanã	2002-2019	Game species and harvest	hunting form, collection of biological material	EL Bizri et al. (2020), Valsecchi et al. (2014), Constantino et al. (2012)
2 – Pegadas Project	Brazil, Amazonas	Piagaçu-Purus Sustainable Development Reserve (PP-SDR)	2010-2012	Game species	line-transect	Benchimol et al. (2017)
3 - Monitoring Program for Biodiversity and Use of Natural Resources in Amazonas Conservation Units (ProBUC)	Brazil, Amazonas	State Park Rio Negro Setor Norte, Uacari and Uatumã Sustainable Development Reserves (SDR's)	2005-*	Game species and harvest	line-transect, hunting form	Constantino et al. (2012), Costa (2019)
4 - Fauna Use and Management Program (PROMUF)	Brazil, Amazonas	Piagaçu-Purus Sustainable Development Reserve (PP-SDR)	2006-2017	Game species and harvest, semi-terrestrial birds, big felids	line-transect, camera-trapping, hunting form	Costa (2019)
5 - Rio Unini Natural Resource Use Monitoring System (SiMUR)	Brazil, Amazonas	Rio Unini Extractive Reserve, Jaú National Park and Amanã Sustainable Development Reserve (SDR)	2008-*	Game species and harvest	hunting memories (register of hunted animals / month), memories of visual records and traces	Costa (2019)
6 - Monitoring the Use of Natural Resources in Indigenous Lands in Acre (URIL)	Brazil, Acre	Indigenous Lands (IL), mainly Kaxinawá and Katukina	2004-2010	Game species and harvest	Hunting form	Constantino et al. (2012)
7 - Game fauna and hunting sustainability in the Xerente Indigenous Land, Brazilian Cerrado (HXIL)	Brazil, Tocantins	Xerente Indigenous Land (XIL)	2014-2015	Game species and harvest	line-transect, hunting form, interviews with participatory mapping	de Paula et al. (2017)
8 - Participatory Monitoring of Biodiversity in Protected Areas of the Amazon (MPB)	Brazil, Pará	Extractive Reserve Tapajós-Arapiuns	2014-*	Game species and harvest	line-transect, sign surveys, camera-	Reis et al. (2019)

					trapping, hunting form, hunting census	
9 - Monitoring of hunting in the Peruvian Amazon (HPA)	Peru, Loreto	Tamshiyacu Tahuayo Regional Conservation Area, Pacaya–Samiria National Reserve and Yavari-Mirin River (unprotected area)	1991-*	Game species and harvest	line-transect, sign surveys, camera-trapping, hunting form, collection of biological material	Mayor et al. (2017)
10 - Nsombou Abalghe-Dzal Community Wildlife Project Gabon (CWP)	Africa, Gabon	Ogooué-Ivindo Province (unprotected area)	2015-2020	Game species and harvest	line-transect, sign surveys, camera-trapping, hunting form, ‘village transects’	Beirne et al. (2019)
11 - Monitoring faunal recovery in a former illegal logging hotspot in Amazonian Peru (FAP)	Peru, Madre de Dios	Los Amigos Conservation Concession	2004-2008	Game species	line-transect	Pitman et al. (2011)
12 - Event Book System (EBS)	Africa, Namíbia	Namibian Conservancies and Bwabwata National Park	2000-*	Game species and harvest	line –transect, ‘distance’ sampling, hunting form, others that record stochastic events	Constantino et al. (2012)
13 - Fogones de Fauna Project (FF)	Uruguay, Cerro Largo	Paisaje Protegido Paso Centurión and Sierra de Ríos	2012-2017	Game species (mammals)	sign surveys, camera-trapping, interviews, collection of biological material	Grattarola & Tricot (2020)
14 - Community monitoring of the jaguar (CMJ)	Mexico, Oaxaca	Chinantla region (unprotected area)	2015-2016	Game specie (jaguar)	camera-trapping	Lavariega et al. (2020)
15 – Crossing ecologies with the Rio Cueiras hunters: knowledge and hunting strategies in the Lower Rio Negro, Amazonas (RCH)	Brazil, Amazonas	State Park Rio Negro Setor Sul, Environmental Protection Area Left Bank of the Rio Negro and National Park Anavilhanas	2006-2007	Game species and harvest	memories of hunted animals, direct	Campos (2008)

					observation, interviews	
16. National Biodiversity Monitoring Program (Monitora) / terrestrial sub-program, forest component	Brazil, Brazilian states	Brazilian protected areas	2014-*	Game species	line –transect, camera-trapping, interviews	Roque et al. (2018)
17- People versus Parks: Can indigenous peoples coexist with tropical biodiversity? (ICB)	Peru, Madre de Dios	Manu National Park	2003-2005	Game species and harvest	hunting form, collection of biological material	Ohl-Schacherer et al. (2007)
18. Participatory monitoring in Lao People's Democratic Republic (PDR)	Asia, Lao PDR	Xe Pian, Dong Phou Vieng, Xe Sap National Protect Areas and Phou Hin Poun National Biodiversity Conservation Area	1999-NA	Game species	logbook records, village reports of signs/sightings and search effort, repeat surveys and sign transect surveys	Steinmetz (2000)
19. Philippine biodiversity monitoring system	Asia, Philippines	Northern Sierra Madre, Bataan and Mt. Kitanglad Range Natural Parks	1996-1998	Game species and harvest	Transect walk , field diary, photo documentation, focus group discussion	Danielsen et al. (2000)
20 - Monitoring of species of large mammals common in the Zambezi alluvium	Africa, Zimbábue	Mana Pools National Park	1993-NA	Game species (mammals)	line-transect	Dunham & Toit (2012)
21 - Zambia's Community-Based Wildlife Program (ADMADE)	Africa, Zambia	Game Management Areas (GMAs)	1987-NA	Game species and harvest, illegal hunting	anti-poaching foot patrols, recorded sightings of live animals, population trends form	Marks (1999), Gibson & Marks (1995), Marks (2001)
22 – Botswana CBNRM Programme	Africa, Botswana	Kalahari and Okwa Wildlife Management Areas (GMAs)	1996-NA	Game species and harvest, illegal hunting	sign surveys, hunting forms, anti-poaching	Twyman (2000)

					foot patrols, wildlife sighting	
23 - Kaa-Iya Project	Bolivia, Gran Chaco Boliviano	Isoso Indigenous Land (IL)	1996-2003	Game species and harvest	hunting form, collection of biological material	Noss (2004), Noss et al. (2005), Noss et al. (2003)
24 - Community-based conservation Programme	Asia, India	Namdapha National Park	2004-NA	Game species (carnivore, prey species)	camera-trapping	Datta et al. (2008)
25 - Ghana Wildlife Division (GWD) monitoring Program	Africa, Ghana	Mole National Park	1968-2008	Game species (mammals), illegal hunting	daytime foot anti-poaching patrols, record sightings of mammal species and hunters	Burton (2012)
26 - Participatory Hunter Self-monitoring Program	Guyana, Kanashen	Konashen Community-Owned Conservation Area (KCOCA)	2014-2015	Game species and harvest	hunting form	Shaffer et al. (2017)
27 - Biodiversity Project	Africa, Zimbabwe	Zambezi Valley (unprotected area)	1996-NA	Game species (mammals)	daylight and night car counts, bicycle counts, foot counts, water point counts	Gaidet et al. (2006)
28 - Coupled Human and Natural Systems Project	Guyana, Rupununi region	Makushi and Wapishana Indigenous Lands (IL)	2007-2010	Game species and harvest	line-transect, hunting form	Luzar et al. (2011)
29 - Jaguar Project Monitoring Network	Argentina / Brazil / Paraguay, Atlantic Forest of Alto Paraná	Alto Paraná Atlantic forests	2002-2008	Game species (pumas, jaguars)	sighting of felines, faecal samples	De Angelo et al. (2011)
30 - Hunting monitoring in the BaAka village	Africa, Republic, Dzanga–Sangha region	Dzanga-Sangha Special Reserve	1993-1994	Game species and harvest	sighting of game species on hunts	Noss (1999) cited by Danielsen et al. (2014)

31 - Hunting monitoring in the Equatorial Guinea	Africa, Equatorial Guinea	Midyobo Anvom village (unprotected area)	2005-2006	Game species and harvest	hunting form, interviews	Rist J et al. (2010) cited by Danielsen et al. (2014)
32 - Medium-sized and large mammals of the Cazumbá-Iracema Extractivist Reserve, Acre, Brazil	Brazil, Acre	Cazumbá-Iracema Extractivist Reserve	2011-2012	Game species (mammals) and harvest	line -transect, opportunistic sightings, camera-trapping, hunting form, interviews	Oliveira & Calouro (2020)
33 - Long-term trends in wildlife community	Africa, Cameroon	Malen V, Doumo Pierre and Mimpala villages (unprotected area)	2002-2016	Game species (mammals)	line –transect	Tagg et al. (2020)
34 - Medium and large-sized mammals in dry forests of the Colombian Caribbean	Colombia, Magdalena	Tayrona National Natural Park	2012-2017	Game species (mammals)	camera-trapping	Pineda-Cendales et al. (2020)
35 - COMBIOSERVE Project	Mexico, Campeche	Once de Mayo community and Calakmul (unprotected area)	2012-2015	Game species	camera-trapping	Villaseñor et al. (2020)
36 - Using local ecological knowledge to improve large terrestrial mammal surveys, build local capacity and increase conservation opportunities	Argentine, Dry Chaco	Salta, Formosa and Chaco provinces (unprotected area)	2011-2017	Game species	line -transect, opportunistic sightings, sign surveys, camera-trapping, interviews	Camino et al. (2020)
37 - Including Spatial Heterogeneity and Animal Dispersal When Evaluating Hunting: a Model Analysis and an Empirical Assessment in an Amazonian Community	Ecuador, Pastaza	Kichwa community of Sarayaku	1999-2000	Game species and harvest	hunting form	Siren et al. (2004)
38 - Mammal hunting by the Shuar of the Ecuadorian Amazon: is it sustainable?	Ecuador, Morona-Santiago	Miasal, western margin of the Amazon basin	2001-2003	Game species and harvest	line -transect, direct observation, hunting form, collection of biological material,	Zapata-Ríos et al. (2009)

					participatory mapping, interviews	
39 - The impact of subsistence hunting by Tikunas on game species in Amacayacu National Park, Colombian Amazon	Colombia, Amazonas Department	Amacayacu National Park	2005-2009	Game species and harvest	line -transect, hunting form	Maldonado Rodriguez (2010)
40 - Road Development and the Geography of Hunting by an Amazonian Indigenous Group: Consequences for Wildlife Conservation	Ecuador, Napo and Pastaza	Yasuní Biosphere Reserve	2008-2009	Game species and harvest	hunting form	Espinosa et al. (2014)
41 - Subsistence hunting among the Waimiri Atroari Indians in central Amazonia, Brazil	Brazil, Roraima and Amazonas	Waimiri Atroari Indigenous Reserve	1993-1994	Game species and harvest	hunting form	Souza-Mazurek et al. (2000)
42 - Evaluación de la Sostenibilidad de la Cacería De Mamíferos en la Comunidad De Zancudo, Reserva Nacional Natural Puinawai, Guainía-Colombia	Colombia, Guainía	Puinawai Natural National Reserve	2005-2009	Game species and harvest	hunting form, direct observation, participatory mapping, interviews	Tafur Guarín (2010)
43 - Korup Project	Africa, Cameroon	Korup National Park, Rumpi Hills, Nta Ali, and Ejagham Forests, and two logging concessions (unprotected area)	1988-	Game species (primates)	line -transect	Waltert et al. (2002)
44- Fortalecimiento de la Red de Monitoreo de Fauna Silvestre en la Reserva de la Biosfera de Tehuacán-Cuicatlán	Mexico, Oaxaca and Puebla	Tehuacán-Cuicatlán Biosphere Reserve	2009-	Game species	camera-trapping	Botello et al. (2013)
45- Community-Based Bird Monitoring Project	Mexico, Chiapas	Tacaná Volcano Biosphere Reserve	2010-	Game species (birds)	line -transect	Ortega-Álvarez & Calderón-Parra (2021)
46- Impact of Hunting on Large Vertebrates in the Mbaracayu Reserve, Paraguay	Paraguay	Mbaracayú Forest Nature Reserve	1980-	Game species	line -transect, sign surveys	Hill et al. (2003) cited by Luzar et al. (2011)

47- Manejo de Fauna na Reserva Xavante Rio das Mortes: Cultura Indigena e Método Cientifico Integrados Para Conservação	Brazil, Mato Grosso	Xavante Rio das Mortes Indigenous Reserve	1991-	Game species (mammals) and harvest	Trace sampling and hunting form	Prada & Filho (2004) and Fragoso et al. (2000) cited by Luzar et al. (2011)
48- Grupo de cazadores de la comunidad negra de El Valle: hacia la construcción de una estrategia local para el manejo de la vida silvestre en la cuenca del río Valle, Chocó, Colombia	Colombia, Chocó	Lands of the Negra Community of the Valle River basin (unprotected area)	2001-	Game species and harvest	line -transect, sign surveys, hunting form	Trespalacios-González et al. (2003) in Campos-Rozo & Ulloa (2003)
49- La investigación participativa y su utilidad para el manejo de la fauna silvestre en Bolivia	Bolivia	Biosphere Reserve and Community Land of Origin Pilón Lajas	2001-2002	Game species and harvest	hunting form	Townsend (2003) in Campos-Rozo & Ulloa (2003)
50. Community-based monitoring system of village forests in Tanzania	Africa, Tanzania	Kitapilimwa, North Nyang'oro, South Nyang'oro, New Dabaga/Ulongambi and West Kilombero Scarp Forest Reserves	2002-2004	Game species and harvest	patrol, interview and meetings	Topp-Jørgensen (2005)
51- Citizen Science for Monitoring Primates in the Brazilian Atlantic Forest: Preliminary Results from a Critical Conservation Tool	Brazil, Minas Gerais	District of Santo Antônio do Manhuaçu/ Caratinga (unprotected area)	2018-2020	Game species (primates)	Sightings form	Nery et al. (2021)
52- Hunting Techniques, Wildlife Offtake and Market Integration. A Perspective from Individual Variations among the Baka (Cameroon)	Africa, Cameroon	Boumba-Bek and the Nki National Parks and Dja Biosphere Reserve	2012-2013	Game species and harvest	hunting memories (register of hunted animals / week), census of hunters	Romain et al. (2017)

NA = information was not provided in the publication.
* The project/program remains active.

Table S4. Additional information (management duty of the protected area and the project, status, cause of end [interruption] or temporary suspension, origin of funding, number of communities/villages [average population], number of monitors and technical team members and origin of information) on the 17 community-based monitoring projects of terrestrial game fauna in the tropical forests, examined in terms of interruption, costs and effectiveness in our study.

	Management duty of the protected area	**Management duty**	**Status**	**Cause of end (interruption) or temporary suspension**	**Origin of funding**	**Number of villages (average population)**	**Number of monitors and technical team members**	**Origin of information**
SMUF	State government	NGO, CSO	inactive	lack of financial resources	Federal government	10 (1378)	10, 5	Researcher
Pegadas	State government	NGO	inactive	lack of financial resources	State government	5 (1000)	30, 6	Researcher
ProBUC	State government	State government	active	lack of financial resources	Federal government	44(2170)[1]	42[2], NA	Project technician, literature (Constantino et al., 2012 and Costa, 2019)
PROMUF	State government	NGO	inactive	lack of financial resources and conflicts between communities and the project's managers	State government	9 (NA)	25, 11	Researcher, literature (Costa, 2019)
SiMUR	State and federal government	NGO	active	*	International	10 (560[3])	11, 2	Project technician, literature (Costa, 2019)

URIL	Federal government	NGO, communities	inactive	lack of financial resources and change in priority of the project's managers	International	45 (4500)	40, 4	Researcher
HXIL	Federal government	University	inactive	project deadline has finished	International	10 (936[4])	NA, 4	Researcher
MPB	Federal government	Federal government, NGO	active	lack of financial resources, COVID-19 pandemic	Federal government, international	13 (1600)	30, 12	Protected area manager
HPA	State and federal government	NGO, CSO	active	*	International	17 (2000)	50, 10	Researcher
CWP	-	Communities	inactive	lack of financial resources	International	20 (2000)	20, 3	Researcher
FAP	Federal government	NGO	inactive	NA	International	0 (NA)	2, 6	Researcher
EBS	State and federal government	Federal government, communities, donors to community based projects	active	*	State government, international	87[5] (250,000)	650, 4	Researcher
FF	Federal government	NGO, Communities, University	inactive	lack of financial resources	International	1 (60)	50-60, 10	Researcher
CMJ	-	Federal government	inactive	project deadline has finished	Federal government	5 (50)	10, 8	Researcher

RCH	State and federal government	Federal government	inactive	project deadline has finished	Federal government	5(200[6])	19, 2	Researcher
Monitora	State and federal government	State and federal government	active	change of responsible technician, COVID-19 pandemic	State and federal government, international	50[7](NA)	NA	Project technician
ICB	Federal government	Communities	inactive	project deadline has finished	International	2 (400)	NA, NA	Researcher

NA = information was not obtained through articles and forms.
- Unprotected area
* The project was not interrupted or temporarily suspended.
[1] Number referring to the year 2012.
[2] Number referring to the year 2014.
[3] Approximate number representing 140 families.
[4] Approximate number representing 234 families.
[5] Number referring to 86 Namibian Conservancies and 1 National Park.
[6]Approximate number representing 50 families.
[7] Number of monitored conservation units.

Table S5. Strategies used to provide information about the monitored resource through time related to the 17 community-based monitoring projects of terrestrial game fauna in the tropical forests, examined in terms of interruption, costs and effectiveness in our study. The spatio-temporal data analyses were scored as 1 if 'exist' and 0 if 'does not exist'; and the percentages of tabulated data and data analyzed, categorized into 4 classes (0-25%, 25-50%, 50-75%, 75-100%), were scored from 1 to 4, respectively.

Project	Spatio-temporal data analyses	Tabulated data	Analyzed data	Scores
SMUF	1	4	4	9
Pegadas	0	4	4	8
ProBUC	0	4	NA	4
PROMUF	0	4	4	8
SiMUR	1	4	4	9
URIL	1	4	4	9
HXIL	0	4	4	8
MPB	1	4	3	8
HPA	1	4	3	8
CWP	0	4	3	7
FAP	1	4	4	9
EBS	1	4	4	9
FF	0	2	3	5
CMJ	0	4	4	8
RCH	0	4	4	8
Monitora	1	4	1	6
ICB	1	4	4	9

NA = information was not obtained through articles and forms.

Table S6. Strategies used to promote local empowerment related to the 17 community-based monitoring projects of terrestrial game fauna in the tropical forests, examined in terms of interruption, costs and effectiveness in our study. We scored as 1 if 'it was used' and 0 if 'it was not used'.

	Local people participated in the elaboration of the project	monitored resource is a source of meat and income	Adequate training of monitors	Local people participate in data entry	Local people perform data analyses	Local people return the results	Score
SMUF	1	0	1	0	0	0	2
Pegadas	1	0	1	0	0	0	2
ProBUC	1	0	1	0	0	1	3
PROMUF	1	0	1	0	0	0	2
SiMUR	1	0	1	0	0	1	3
URIL	1	0	1	0	1	1	4
HXIL	1	0	1	0	0	0	2
MPB	1	0	1	0	0	0	2
HPA	1	1	1	0	0	0	3
CWP	1	1	1	1	1	1	6
FAP	0	0	0	0	0	0	0
EBS	1	1	1	0	0	0	3
FF	1	0	1	0	0	0	2
CMJ	1	0	1	1	0	1	4
RCH	1	0	1	0	0	0	2
Monitora	1	0	1	0	0	0	2
ICB	0	0	0	0	0	0	0

Table S7. Management actions promoted by the results of the 17 community-based monitoring projects of terrestrial game fauna in the tropical forests, examined in terms of interruption, costs and effectiveness in our study. We score as 1 if 'exist' and 0 if 'does not exist'.

	Results promoted any resource management action	Which ones?	Score
SMUF	Yes	Species or resource start to be monitored and creation of hunting rules.	1
Pegadas	No	-	0
ProBUC	No	-	0
PROMUF	Yes	Species or resource start to be monitored, zoning of hunting areas and creation of hunting rules	1
SiMUR	Yes	Creation of management plans	1
URIL	Yes	Species been banned from being hunted and creation of management plans	1
HXIL	No	-	0
MPB	No	-	0
HPA	Yes	Species or resource start to be monitored, zoning of hunting areas, creation of hunting rules and fisheries management plans	1
CWP	Yes	Comply with management plan rules	1
FAP	No	-	0
EBS	Yes	Species been banned from being hunted, creation of hunting rules and support the policy on sustainable use and community based natural resource management	1
FF	Yes	Support for inclusion of region in the National System of Conservation Units	1
CMJ	No	-	0
RCH	No	-	0
Monitora	No	-	0

ICB	Yes	Avoided the creation of new human settlements	1

Table S8. Ranking of the 17 community-based monitoring projects of terrestrial game fauna in the tropical forests, examined in our study in terms of interruption, costs and effectiveness, from the most to the least effective. To obtain the total effectiveness score, the score for each pillar (shown in tables S5, S6 and S7) received a weight of 1.

Project	first pillar score	second pillar score	third pillar score	Total score (%)
CWP	0,8	1,0	1	2,8
URIL	1,0	0,7	1	2,7
SiMUR	1,0	0,5	1	2,5
EBS	1,0	0,5	1	2,5
HPA	0,9	0,5	1	2,4
SMUF	1,0	0,3	1	2,3
PROMUF	0,9	0,3	1	2,2
ICB	1,0	0,0	1	2,0
FF	0,6	0,3	1	1,9
CMJ	0,9	0,7	0	1,6
Pegadas	0,9	0,3	0	1,2
HXIL	0,9	0,3	0	1,2
MPB	0,9	0,3	0	1,2
RCH	0,9	0,3	0	1,2
FAP	1,0	0,0	0	1,0
Monitora	0,7	0,3	0	1,0
ProBUC*	0,4	0,5	0	0,9

* some information about first pillar was not obtained through articles and forms.

CAPÍTULO 2

Community-based monitoring reveals low anthropogenic pressure on game vertebrate population in a sustainable-use Amazonian protected area

Article under review in *Environmental Conservation* (A2, IF 4.075).

Foto: Pollyana de Lemos

Community-based monitoring reveals low anthropogenic pressure on game vertebrate population in a sustainable-use Amazonian protected area

YASMIN MARIA SAMPAIO DOS REIS[1*], MAÍRA BENCHIMOL[1]

[1]Applied Ecology & Conservation Lab, Programa de Pós-graduação em Ecologia e Conservação da Biodiversidade, Universidade Estadual de Santa Cruz, Rodovia Ilhéus-Itabuna, km 16, Salobrinho, 45662-000 Ilhéus, Bahia, Brazil

E-mail addresses: yasmin.sampaioreis@gmail.com (Y. Reis)*, mairabs02@gmail.com (M. Benchimol)

*Corresponding author.

Abstract

Biodiversity systematic monitoring programs have been expanding across the globe, especially in protected areas (PAs). Among sustainable-use PAs, medium to large-sized mammals and birds comprise crucial groups to be monitored, given their importance to ensure forest functionality and subsistence for local residents. Here, we used a database from 6-years of community-based monitoring conducted within a sustainable-use PA in the Brazilian Amazonia (Tapajós-Arapiuns Extractive Reserve) to examine the influence of anthropogenic stressors (human intensity and distance to nearest road) and the monitored year on patterns of density and biomass of mammal and bird forest game species. We further assessed trends in population density of target groups along the time series. A total of 1,915 km of line-transect surveys were completed by trained local monitors along eight established transects, providing data from 12 medium-sized and 5 large-sized game genera. We performed Generalized Linear Mixed Models considering (a) all medium-sized species; (b) all large-sized species; (c) and four taxa (Tinamidae, Dasyproctidae, Primates and Cervidae) individually, whereas other species failed in exhibiting model convergence. Results showed that some species exhibited great density (such as Dasyprocta croconota), whereas others were rarely detected (e.g., Tapirus terrestris and Tayassu pecari). We showed that anthropogenic variables did not affect the density and biomass of the overall medium-sized and large-sized vertebrates. Dasyproctidae, Tinamidae and Primates were the only taxa influenced by anthropogenic stressors, with negative influence only for Tinamidae. Moreover, density of groups and taxa remained stable through the evaluated time in the Reserve. Yet the low records of some species may indicate past population losses, which would reflect in a semi-defaunated current scenario. Therefore, the continuity of this monitoring program is required to improve our understanding on patterns of population fluctuations in the long-term, but at least during the evaluated years, game populations were not reduced due to anthropogenic stressors.

Key words: Participatory monitoring, game fauna, mammals, birds, hunting pressure

Introduction

The implementation of biodiversity systematic monitoring programs has been expanding across the globe, especially in protected areas (Schmeller et al. 2017; Reis & Benchimol 2023). As these programs provide information on the status and trends of natural resources, including wildlife populations, they can greatly contribute to curbing the current biodiversity worldwide crisis (Danielsen et al. 2022). In addition, monitoring programs can evaluate the impacts of a myriad of anthropogenic activities on biodiversity, including habitat destruction, climate change and wildlife overexploitation (Veiga & Ehlers 2009). In Brazil, for instance, the federal government established a long-term and large-scale biodiversity monitoring program across different protected areas, focused on assessing the conservation status of fauna and flora species in addition to evaluating their responses to anthropogenic pressures (Ribeiro 2018, Cronemberger et al. 2023). Implementing such initiatives is especially needed within sustainable-use protected areas, which retain high biodiversity and are home for indigenous and other traditional communities, who rely on forest resources to survive (Lima & Pozzobon 2005).

Community-based, participatory monitoring programs have been widely established to monitor game fauna in sustainable-use protected areas across tropical countries (Reis & Benchimol 2023). Besides providing spatio-temporal information on monitored game species and informing appropriate management actions (Danielsen et al. 2021), participatory programs value experiences and point of view of local people who have extensive knowledge on wildlife species. In particular, large and medium-sized forest species, including mammals and birds, are among the main target game fauna and represent most of the vertebrate biomass in tropical forests (Benítez-López et al. 2019). Species weighing more than 5 kg (hereafter, large-sized species) comprise the favorite targets of hunters (Constantino et al. 2008), due to the meat flavor and the greater return per unit of effort. In addition, large-sized vertebrate species are more vulnerable as they usually exhibit lower reproductive rates and are affected by external factors such as anthropogenic activities (Cardillo et al. 2005). Therefore, the density of large-sized species usually decreases in areas under pervasive anthropogenic pressure (potentially linked to hunting pressure), therefore altering the structure of vertebrate communities and reducing vertebrate biomass (Peres 2000, Jerozolimski and Peres 2003). In addition, the decay in vertebrate biomass often exceeds the changes in overall species abundance and species loss driven by anthropogenic activities, given that small-sized species may be favored and tend to proliferate in areas where large species were extirpated (Peres & Palacios 2007), a phenomenon named density compensation (MacArthur et al. 1972).

Several anthropogenic factors are highly associated with hunting pressure for forest game vertebrates, including hunter access points (e.g. communities and roads) and human population density (Beirne et al. 2019). For instance, Scabin & Peres (2021) showed that distance from villages and human population size represent good proxies of anthropogenic threats to large game vertebrates in a sustainable-use Amazonia protected area, as steep biomass declines were recorded near urban sites presenting greatest human pressure. Therefore, assessing the influence of anthropogenic variables on patterns of vertebrate density and overall biomass poses vastly useful in sustainable-use protected areas, given their potential in leading vertebrates to population depletion.

Declines in large-sized forest vertebrate populations driven by overhunting occur widely in tropical countries, in particular in Africa and Asia (Benítez-López et al. 2019). Thus, the diversity and relative abundance of large mammal species, such as primates and ungulates, decreased significantly near villages in Gabon due to high levels of hunting pressure (Koerner et al. 2017, Beirne et al. 2019). In Amazonian forests, heavily hunted forest sites also experience reduction in the abundance of large-sized mammals (Peres 2000). Nevertheless, some studies reported that terrestrial vertebrate populations can be resilient to hunting in Amazonian forest sites (e.g., Iwamura et al. 2014, Antunes et al. 2016), as humans have been intensively hunting medium and large-sized species for many years, but animal populations have not been extirpated. This is likely because many vast upland areas remain inaccessible to hunters, establishing a source-sink dynamic that can rescue over harvested populations in heavily hunted areas (Antunes et al. 2016). Conversely, some Amazonian forest sites have experienced past population declines due to anthropogenic pressure, and currently are recognized to exhibit a state of semi-defaunation (Peres et al. 2003). However, to better understand this dynamic it becomes necessary to monitor the game fauna through time and generate spatio-temporal information on the monitored species.

Here, we used a comprehensive database from a 6-year community-based systematic monitoring study in a sustainable-use protected area in the Brazilian Amazonia to evaluate how (i) anthropogenic stressors and year modulate patterns of density and biomass of mammal and bird forest game species; and (ii) game populations fluctuate along the years, i.e., if the density of target groups and taxa are stable, increasing or decreasing through the evaluated time series. We extracted two variables related to hunting pressure (i.e., a proxy of human intensity and the distance to nearest road) and estimated the overall density and biomass of large-sized (≥5 kg) and medium-sized (< 5 kg) game species pooled together, in addition to densities of the four taxa most frequently recorded within eight established linear-transects per year across the Tapajós-Arapiuns Extractive Reserve (hereafter, RESEX-TA). We hypothesize that the interaction between our two anthropogenic stressors would best explain the overall patterns of density and biomass of large-sized game taxa, given the greater access of hunters and their higher preference for large-sized species (Peres 2000). As a result, we expect an increase in the overall densities and biomass of medium-sized game taxa in areas under higher human influence, caused by the phenomena of density compensation (MacArthur et al. 1972). For individual taxa, we expected that Dasyproctidae, Primates and Tinamidae densities (medium-sized taxa) would be positively influenced by anthropogenic stressors, as a result of the compensatory effect, while for Cervidae (large-sized taxa), we expected a negative influence. As a result, we expected that population densities of Tinamidae, Psophiidae, Cracidae, Dasyproctidae, Primates e Procyonidae would increase over the evaluated time, while the intensive hunting would be leading to a decrease in Atelidae, Cervidae and Tayassuidae.

Materials and methods

Study area

This study was conducted at the RESEX-TA, a terrestrial extractive protected area located in western Pará, Brazilian Amazonia (02º 20' to 03º 40' S, 55º 00' to 56º 00' W; Fig. 1). This sustainable-use protected area was created in 1998, aiming to protect local population people against the impacts induced by the expansion of logging companies in the area (Oliveira et al. 2005). Covering a total area of nearly 650,000 ha, the RESEX-TA is currently the most populous extractive reserve in Brazil, housing about 23,000 residents (Silva et al. 2022). It comprises the first Brazilian protected area of this category to develop a management plan (i.e., a technical document that establishes the zoning and norms that guide the use of the protected area).

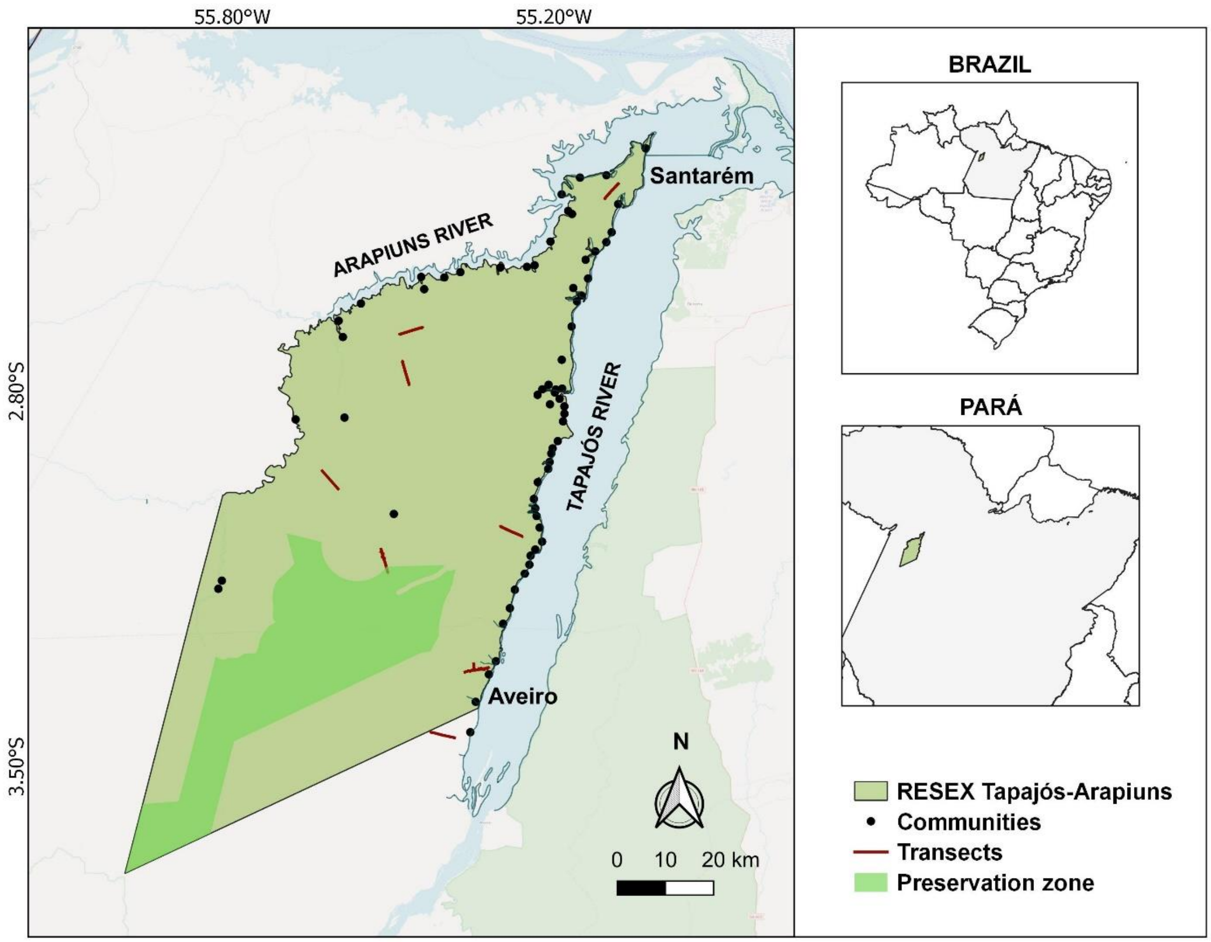


Fig.1. Location of the Tapajós-Arapiuns Extractive Reserve in Santarém and Aveiro cities (Pará), Brazil.

The predominant vegetation is the ombrophilous forest (66.19% of the territory), characterized by large trees, with large abundance of woody, epiphytic and lianas (Veloso et al. 1991, IBGE 1992), in terra firme environments (i.e., unflooded forests). Other common plant typologies encompass secondary forest (8.88%), savannah (0.62%), and flooded forests (Carvalho-Junior 2008, Montag et al. 2012). The climate is humid tropical (Ami), with a rainy season between November to May, and a dry season from August to October (INPE & CPTEC 2023).

The RESEX-TA has currently 78 communities, inhabiting areas along the Tapajós and Arapiuns rivers (Silva et al. 2022), in addition to few families living along small streams (Oliveira et al. 2004, ICMBio 2014; Fig. 1). All communities are established in *terra firme* forests, in which the residents directly depend on the exploitation of natural resources (hunting, fishing and extraction of timber and non-timber forest products), and/or family farming and small animal breeding (Saúde & Alegria 2012). Hunting is allowed for subsistence purposes, and its management depends on internal agreements signed between communities and reserve managers. In mid-2021, there was an attempt at logging through a community forest management plan, but the lack of prior popular consultation resulted in the paralysis of activities.

Participatory Biodiversity Monitoring Project (PBMP)

In 2014, the Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio; belonging to the Brazilian federal government) implemented the Brazilian Biodiversity Monitoring Program (Monitora Program) in the RESEX-TA. Given that local residents expressed their willingness in understanding the current and future population status of game terrestrial fauna (i.e., medium to large-sized mammals and terrestrial birds), the monitoring program in the RESEX-TA focused on species belonging to these groups, as well reptiles (totalizing 25 target species, carefully chosen by local people based on their hunting activities, see Table S1). The implementation of this program was carried out via the Participatory Biodiversity Monitoring Project (PMPB), a partnership between ICMBio and the Instituto de Pesquisas Ecológicas (IPÊ) (Ribeiro 2018).

Firstly, different meetings were carried out in the RESEX-TA aiming to identify different communities willing to participate in the monitoring program – i.e., potential monitors belonging to these communities that would become responsible for transect establishment and data collection. In particular, transects needed to be placed in accessible areas by foot or boat/canoe by the monitors from communities involved in the monitoring program, in areas without direct anthropic interference (apart from hunting activities). Furthermore, we placed transects at different distances from the nearest community, as nearby transects would likely be more affected by hunting pressure (Sampaio et al. 2023). Prior to effective implementation, potential areas were identified through map analysis, taking into account elevation profiles, obtaining coordinates from maps, and on-site recognition of the areas. Finally, transects needed to be spaced by at least 5 km from each other, and placed in non-flooded forests, as comprised the main land cover inside the reserve. A total of nine linear transects of 5 km in length were thus established across the RESEX-TA. However, after beginning data collection in 2014, one community left the program, and eight communities/transects remained in the program (Fig. 1). Each target community selected 3 to 4 monitors older than 18 years old, based on their willingness to participate combined with vast experience in walking along the forest. Subsequent training courses were provided by ICMBio and partners to all monitors, focused on providing instructions for game fauna surveys. Since 2022, data collection is performed by 31 previously trained monitors.

Data collection

On each transect, pairs of monitors were responsible to conduct linear-transect surveys in the nearby transect from their community. This methodology has been widely used in tropical forests (Peres 1999; Benchimol 2016) and consists of walks at a slow and constant speed (an average 1 km per hour) along a linear transect, searching for both visual and acoustic records of the target species. Surveys were conducted only in the morning (6:30 a.m.– 11:30 a.m.) by a pair of monitors and did not occur on rainy days. Whenever an individual or group of the target species was recorded, the following information were obtained: species name, type of record, meeting time, location of the animal on the transect, number of individuals and perpendicular distance to the first detected individual, subgroup or group (measured with a tape). Given that local monitors have extensive experience in walking in the forest, searching and identifying animals, and appropriate training courses were provided, we assume that data collection was well performed. Surveys were conducted between 2015-2020, with an average of nine (SD = 2.59) survey days per transect per year, totaling 384 sampling days across the eight established transects. Although initially the protocol aimed to obtain data from the dry and wet period for 10 days per year per transect, data were collected in different months, according to the availability of the monitors, and resulted in different sampling efforts among transects. Furthermore, there were situations where the linear transects extended throughout the day or rainy periods that only enable partial sampling, leading to the exclusion of data from some days. Therefore, the cumulative distance sampled in each transect varied from 5 to 65 km per year, totalizing 1,915 km (mean ± SD = 239 ± 45 km per transect; Table S2) considering the six years of data collection.

Target taxa

We used data collected from nine game forest taxa (i.e., nine Order or Family of hunted medium and large-sized forest mammals and birds) in the RESEX-TA (Reis et al. 2018, 2022) (Table S1). The selection of focal taxa is related to two criteria: (i) focal species of the Monitora program monitoring protocol, which considered the target hunting species in the region, based on the knowledge of local people; and (ii) species recorded in hunting events (not covered in this study) from the Monitora program (Reis et al. 2018 and 2022). Specifically, for our study, we selected game species from the mentioned categories that could be recorded within RESEX-TA through the use of the linear transect technique (Table S1). In addition, conducting a species-level analysis was not feasible due to the limited number of records for most animals. Consequently, we conducted our analyses at the taxon level. These comprise six taxa of medium-sized (<5 kg): Tinamidae (*Tinamus* spp., *Crypturellus* spp.); Psophiidae (*Psophia viridis*); Cracidae (*Penelope* spp., *Crax* sp. and *Pauxi tuberosa*); Dasyproctidae (*Dasyprocta croconota*); Primates (Cebus unicolor, *Sapajus apella*, *Pithecia irrorata*, *Callicebus hoffmannsi*) and Procyonidae (*Nasua nasua*); and three large-sized taxa (>5 kg): Atelidae (*Alouatta nigerrima*), Cervidae (*Mazama americana* and *M. nemorivaga*) and Tayassuidae (*Dicotyles tajacu* and *Tayassu pecari*). Despite some species are not frequently hunted at other Amazonian sites (e.g., *Pithecia irrorata* and *Callicebus hoffmannsi*), they are captured for consumption at the RESEX-TA (unpublished data). In addition, we did not include Ateles marginatus as focal species since this species is not hunted in the RESEX-TA (Reis et al. 2018).

Density and biomass

We estimated the animal density for each of the nine taxa for each transect and year using the equation:

$$D = \frac{N}{2 * EW * L}$$

where D represents the density (individuals/km^2); N represents the total number of sightings; EW represents the effective width of the transect (in km); and L represents the total distance traveled (Bernardo & Galetti 2004). To obtain EW, we grouped the visualization events per taxa and constructed six competing models: a uniform key function with either cosine or simple polynomial series expansion, a half-normal key function with either cosine or hermite polynomial series expansion, and a hazard rate key function with cosine or simple polynomial series expansion, using the Distance 4.1 software (Thomas et al. 2010) (models in Table S3). We selected the best fit model based on the lowest Akaike Information Criterion values (AIC; Burnham & Anderson 2002) and coefficient of variation, and highest GOF Chi-p values. With the estimated density per taxa, we evaluated the effect of hunting on animal densities considering: (i) group (medium- and large-sized) density and biomass and (ii) “individual” densities of the four most frequent taxa

(Tinamidae, Dasyproctidae, Primates and Cervidae), for each transect per each year. Given the low number of records, we were unable to conduct individual analyses for the other taxa. Biomass was calculated as the sum of densities of each species multiplied by the mean body weight of the species in each group (based on Robson & Redford 1986, Ayres & Ayres 1979 and Valsecchi 2013).

Anthropogenic stressors

We obtained two metrics related to anthropogenic pressure. Firstly, we used a proxy of human intensity (HI) based on the distance to nearest community and human population size of its community (adapted from Scabin & Peres 2021), according to the equation:

$$HI = \sum \frac{S(com)}{\sqrt{d(com)}}$$

where *S* represents the human population size at each community (*com*); and *d* represents the Euclidean distance from the center of each transect to the center of each nearby community. For this, we considered all communities distant up to 10 km from the center of the surveyed transect. This distance was based on the evidence that hunters forage in areas located up to 10 km from their homes at the RESEX-TA (unpublished data from the participatory mapping of hunting activity, carried out in 2018). Distances were calculated in QGIS 2.18.9 (QGIS Development Team 2017) and human population size of each community was obtained from the local ICMBio database from 2022. We assume that population size did not substantially change among surveyed years, based on a comparison between the most current list of protected area beneficiaries (from 2022) and what we had previously known (from 2018) (Spearman's correlation, r = 0.95, p = 0.001).

We also measured the distance to nearest road, defined as the distance from the middle of each transect to the nearest dirt road. Because dirt roads are the main access ways in *terra firme* in the RESEX-TA, we assumed that transects closer to roads will be more accessed by humans than isolated transects (Benítez-López et al. 2017, Beirne et al. 2019).

Data analyses

We used Generalized Linear Mixed Model (GLMMs) to relate variation in human intensity, distance to the nearest road and year of data collection (with the random term 'transect') to overall densities and biomass of both medium-sized and large-sized groups, separately, and to the densities of the four selected taxa individually. We emphasize that it was only possible to analyze diurnal species commonly recorded (i.e., that present greater detectability and therefore provided a suitable number of visual records to enable data analysis); therefore, we excluded those game species that are rarer or more difficult to be recorded in linear transects. We previously tested the correlation among our predictor variables (i.e., human intensity, distance to the nearest road and year) using a Spearman's test, and since none was correlated at (> |0.50|) to each other, we maintained all variables in the global model. Mean, standard deviation, minimum and maximum values of anthropogenic variables can be assessed in Table S4.

For each response variable, we constructed different models: the null model, models presenting a single or two predictors, models with all possible additive combinations of them, and models with the interaction effects between the human intensity and distance to the nearest road. We selected the best model(s) based on the Akaike Information Criterion (AIC; Burnham & Anderson 2002) values, considering as parsimonious those presenting $\Delta AICc \leq 2.0$. When the null model appeared amongst the parsimonious models, we considered that no other model best explained the specific pattern than the chance. Finally, we used the confidence interval (CI) of the coefficient of each variable included in the best models (95% CI) and considered its influence when the CI did not include 0. Due to convergence problems, we were unable to consider the interaction model for Primates and Tinamidae. We transformed the data using log (for groups) or square root (for taxa) and confirmed data normality using the Shapiro-Wilk test. We used the Gaussian distribution family ("*identity*" linkage function) in all models. The analyses were performed in the package *glmmTMB* (Brooks et al. 2022) in R 4.2.1 (R Development Core Team 2022).

To assess population trends over the years, we built state-space models that describe the stochastic and deterministic relationships between the observable and unobserved values using a Bayesian approach (Royle & Kéry 2007, Kéry 2010). We used as a response variable the annual mean density value of the medium-sized, large-sized group and nine taxa considering all transects (as we were able to perform this analysis with all taxa). We also transformed the data using log. The assessment was made by estimating the exponential growth rate *r* of density; in which *r* value is a measure of change in population size that assumes positive, negative and zero values in increasing, declining and stable populations, respectively, and is calculated using the equation:

$$r = \frac{log(N_t/N_0)}{t}$$

where *N0* is the population size at the beginning of the period and *Nt* is the population size after *t* time units (in this case, years; Caughley & Sinclair 1994). A time-series mean *r* value equal or greater than 0 may suggest that populations are stable or growing, but values lower than 0 can be interpreted as evidence of a decline in the evaluated parameter (de Paula et al. 2022). Thus, the *r* value was considered as indicative of the behavior of density over the evaluated years. Therefore, if the mean density *r* values over the time series is equal to 0 or positive for medium-sized, large-sized group and the nine "individual" taxa, we assume that populations are not declining;

decreasing *r* values were taken to indicate otherwise. We use R version 4.2.1 (R Development Core Team 2022) and the *R2jags* package (Su & Yajima 2012) to fit the state-space model in *JAGS* (Plummer 2015). Non-informative priors were used for the density, and 200,000 Markov chain Monte Carlo iterations were run in two independent chains with a 100,000 burn-in and a thinning factor of 0.06. Further, the Gelman-Rubin (*Rhat*) diagnosis was used to evaluate parameter convergence (Gelman & Shirley 2011). *Rhat* is a measure of convergence where values of 1.001 indicate a satisfactory model (the closer to 1, the better the model) and where 1.1 is the acceptable limit. If the 95% Bayesian confidence interval did not include 0, the *r* value was deemed substantially different from 0. The results are expressed as means ± standard deviations.

Results

The mean density of medium-sized group was 44.0 (± 18.97) ind./km², with transects varying from 24.3 to 86.5 ind./km², whereas mean biomass was 101.0 (± 63.43) kg/km² and varied from 51.0 to 258.6 kg/km² (Table S5). Among this group, the taxa with the highest mean densities were the Dasyproctidae (*Dasyprocta croconota*) with 15.9 ind./km² and the medium-sized Primates (considering the four species grouped: *Cebus unicolor*, *Sapajus apella*, *Pithecia irrorata*, *Callicebus hoffmannsi*) with 12.8 ind./km², and the lowest density was 2.1 ind./km² for Tinamidae (*Tinamus* spp. and *Crypturellus* spp.) (Table S5). Some frequently hunted species, such as *Tapirus terrestris*, were rarely recorded (Table S1).

The mean density of large-sized group was 6.5 (± 3.01) ind./km², and varied among transects, from 2.7 to 10.3 ind./km², whereas mean biomass was 121.8 (± 65.66) kg/km² and varied from 31.9 to 223.9 kg/km² (Table S5). The taxa with the highest mean densities were the Tayassuidae (*D. tajacu* and *T. pecari*) with 2.5 (± 2.37) ind./km² and the Atelidae (*Alouatta* *nigerrima*) with 2.1 (± 1.63) ind./km². The Cervidae, with 1.9 (± 0.78) ind./km², showed the lowest mean density among the large-sized group (Table S5).

Influence of anthropogenic stressors on density and biomass of game fauna

Mean density and biomass of the medium-sized and large-sized groups were neither influenced by human intensity nor distance to nearest road, and did not vary among years. Indeed, the null model appeared among the parsimonious in the group analyses (Table 1). However, different results were observed for densities of each investigated taxa (Table 2). For Cervidae, only the null model was retained in the set of best models, whereas the predictor variables were included in the set of best models for the other three taxa. In particular, distance to the nearest road was included in the best model for both Dasyproctidae and Tinamidae, whereas human intensity and monitored year were included in the best model explaining patterns of primate density (Table 2). The density of Dasyproctidae and Tinamidae decreased and increased, respectively, in transects further from roads, whereas the primate density increased in transects with high human intensity and also along the monitored years (Table S6, Fig. 2).

Table 1. Model selection table based on a candidate set of 'best' models ($\Delta AIC \leq 2.00$) predicting the density and biomass patterns for overall medium-sized and large-sized game species on eight transects across the Tapajós-Arapiuns Extractive Reserve. Models ranked by the AICc and ΔAICc values.

Model	**df**	**logLik**	**AICc**	**ΔAICc**	**weight**
Medium-sized group					
Density					
~ null	3	-29.88	66.36	0.00	0.30
~ year	4	-29.10	67.22	0.86	0.19
~ distance to nearest road	4	-29.38	67.78	1.42	0.15
Biomass					
~ null	3	-28.34	63.29	0.00	0.27
~ distance to nearest road	4	-27.57	64.17	0.88	0.18
~ year	4	-27.75	64.53	1.24	0.15
~ human intensity	4	-28.10	65.23	1.95	0.10
Large-sized group					
Density					
~ distance to nearest road	4	-61.19	131.40	0.00	0.29
~ null	3	-62.51	131.62	0.22	0.26
Biomass					
~ null	3	-64.18	134.95	0.00	0.42
~ distance to nearest road	4	-63.93	136.88	1.93	0.16

Table 2. Model selection table based on a candidate set of 'best' models (ΔAIC < 2) predicting density patterns for Cervidae, Dasyproctidae, Tinamidae and Primates on eight transects across the Tapajós-Arapiuns Extractive Reserve. Models ranked by the AICc and ΔAICc values.

Model	Df	logLik	AICc	ΔAICc	weight
Cervidae					
~ null	3	-40.58	87.76	0.00	0.36
~ human intensity	4	-40.17	89.37	1.60	0.16
~ year	4	-40.19	89.40	1.64	0.16
Dasyproctidae					
~ distance to nearest road	4	-50.21	109.44	0.00	0.32
~ distance to nearest road + human intensity	5	-49.32	110.21	0.77	0.22
~ distance to nearest road + human intensity + distance to nearest road:human intensity	6	-48.39	111.05	1.61	0.14
~ year + distance to nearest road	5	-49.91	111.40	1.96	0.12
Tinamidae					
~ distance to nearest road	4	-44.41	97.84	0.00	0.49
Primates					
~ year + human intensity	5	-71.72	155.03	0.00	0.48
~ year + distance to nearest road + human intensity	6	-71.07	156.40	1.38	0.24
~ year	4	-74.00	157.03	2.00	0.17

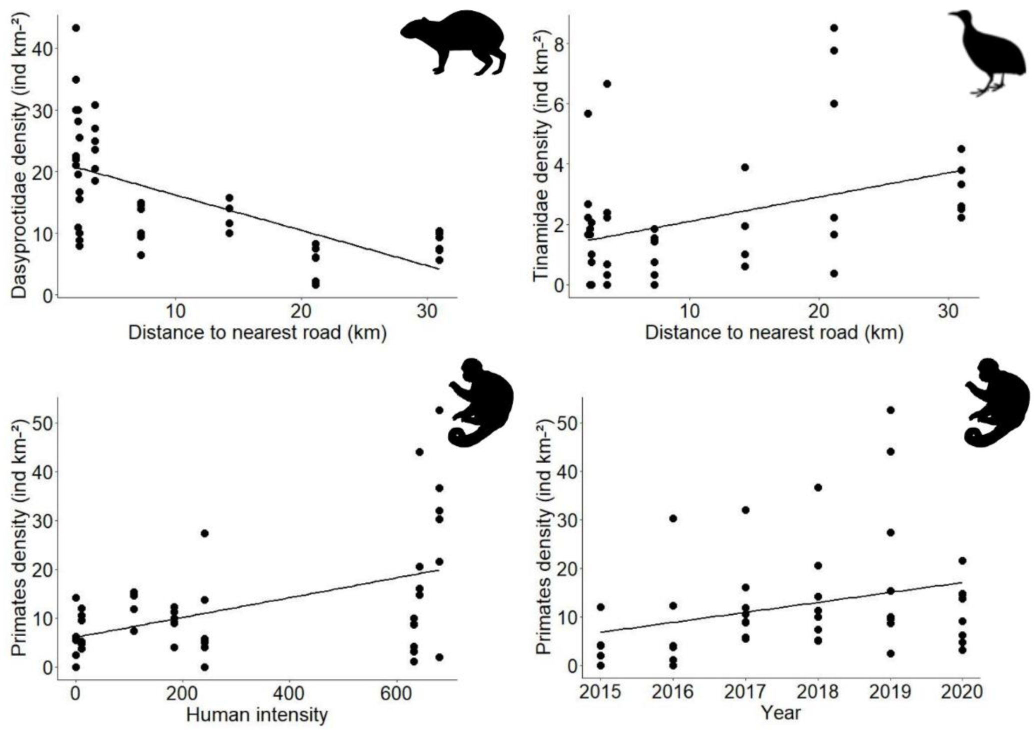
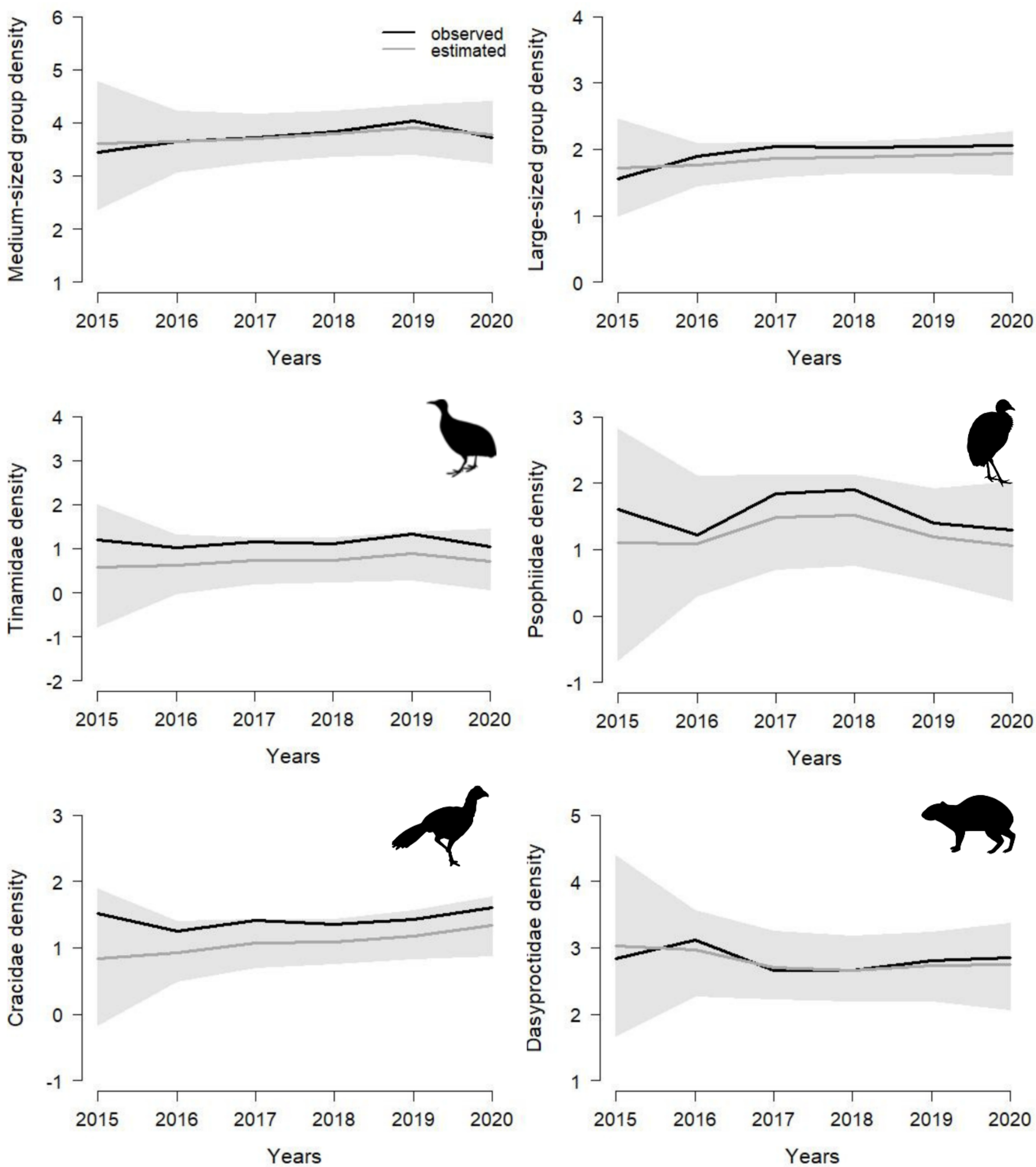


Fig. 2. Mean density of Dasyproctidae and Tinamidae in relation to distance to nearest road, and average density of Primates in relation to human intensity proxy and year, in eight transects across the Tapajós-Arapiuns Extractive Reserve.

Population trends

The parameter convergence was satisfactory for all Bayesian models (Table S7) and the exponential growth rate r was not different from 0 for density averages for the game group and taxa. These temporal trends show that density remained stable for both the medium-sized and large-sized groups, as well as for nine investigated taxa (Tinamidae, Psophiidae, Cracidae, Dasyproctidae, Primates, Procyonidae, Atelidae, Cervidae and Tayassuidae) across all sampled transects in the Reserve, indicating that the game populations were not declining through the evaluated years (Fig. 3).



observed
estimated
Medium-sized group density
Large-sized group density
Tinamidae density
Psophiidae density
Cracidae density
Dasyproctidae density
Years
2015
2016
2017
2018
2019
2020

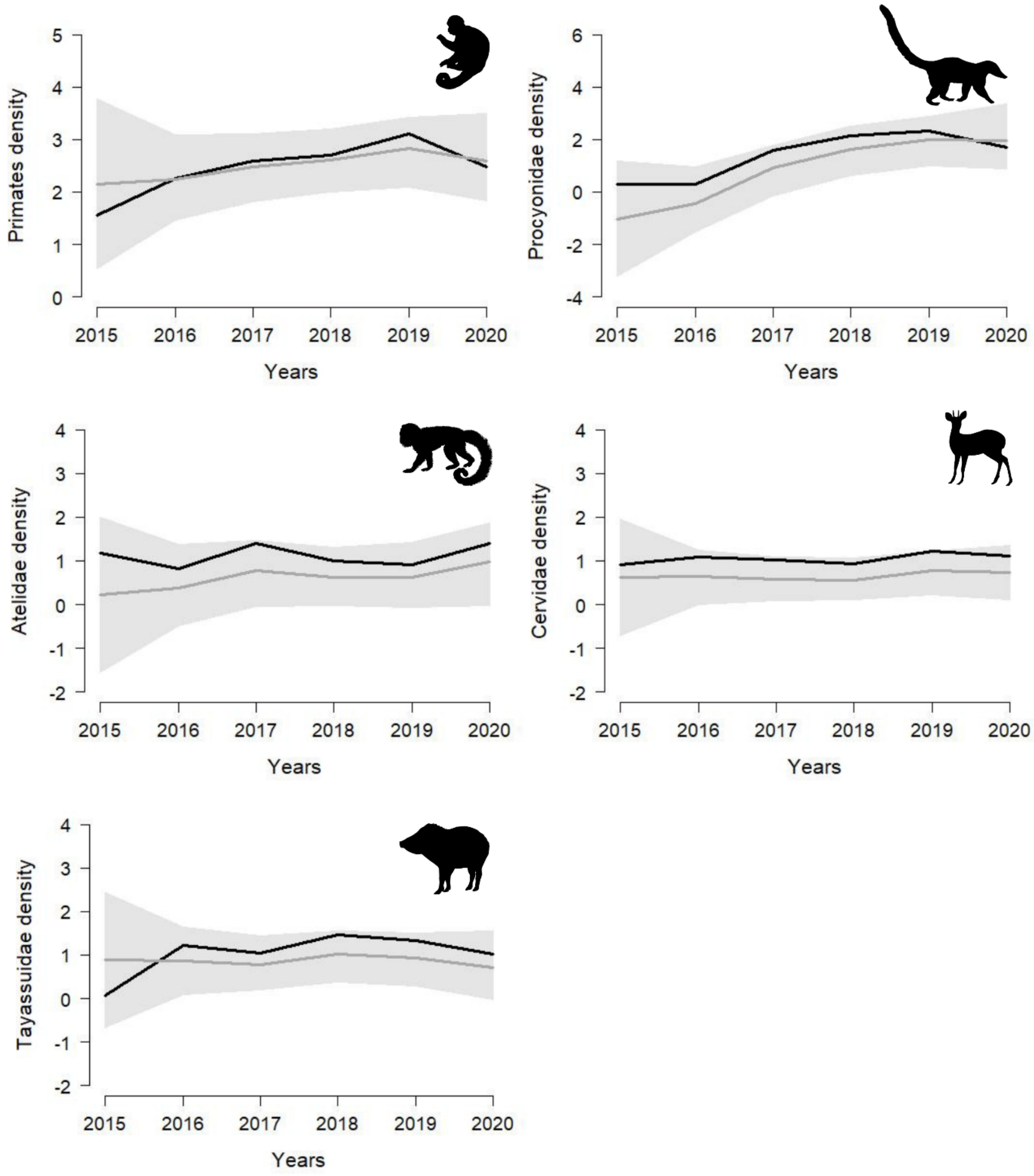


Fig. 3. Annual average density, expressed as log (x + 1), of the medium-sized and large-sized game groups and nine individual taxa (Tinamidae, Psophiidae, Cracidae, Dasyproctidae, Primates, Procyonidae, Atelidae, Cervidae and Tayassuidae) in eight study transects in the Tapajós-Arapiuns Extractive Reserve from 2015 to 2020. Lines represent observed and estimated densities.

Discussion

This comprises the first study assessing the responses of vertebrate game species over a temporal series based on the Brazilian Monitora participatory monitoring program. Contrary to our initial expectations, anthropogenic variables failed in predicting overall density and biomass patterns of both medium and large-sized groups. However, the density of both Dasyproctidae and Tinamidae per transect were influenced by distance to the nearest road, whereas Primates were affected by monitored year and human intensity. Finally, mammal and bird populations remained stable over the six studied years considering all transects pooled together, indicating no decline or increase in the game taxa across the Tapajós-Arapiuns sustainable-use reserve over the time series. However, we recognize some of the limitations intrinsically related to the data. Nevertheless, the obtained information provided by this study is extremely valuable to both enhance our understanding on the patterns of terrestrial vertebrate densities in a populated forest reserve according to an anthropogenic gradient and recognize the importance of implementing long-term participatory monitoring programs in Amazonian reserves.

Our study reveals that overall large-sized and medium-sized game vertebrates included in our analysis were unaffected by anthropogenic pressure through the years, which might be explained due to socioeconomic and environmental context of the RESEX-TA. Although local people have historically and continuously hunted the game fauna across the RESEX-TA, a previous study in the RESEX showed that the preferred species still comprise the most frequently hunted (Reis et al. 2022). Preferred game species in heavily hunted areas usually exhibit low densities, become rare or can even be locally extinct and therefore dropped out of the hunting profiles (Peres & Palacios 2007). Alternative sources of animal protein also occur in the reserve, mainly composed of fish stocks (Oliveira et al 2004, Braga et al. 2022) and small domestic animals (e.g. pig and chicken). In addition, commercial hunting to supply urban centers is non-existent. Finally, the preservation zone, where subsistence hunting and other anthropogenic activity is prohibited, occupies more than 20% of the total area of the RESEX-TA (ICMBio 2014) and can act as 'source' of animals to the 'sink' areas (Novaro et al. 2000).

Anthropogenic stressors presented a substantial influence on the density of some species or taxa, yet exhibited contrasting patterns. In particular, agoutis (*D. croconota*) were substantially affected by the distance to the nearest road, although greater densities were observed in transects near the road. In fact, agoutis can cope with disturbed forested areas, such as forest edges, which are often related to road creation (Peres 2001). For instance, forest fragmentation exerted a positive effect on agouti density in forest patches located in the Central Brazilian Amazonia, and individuals were frequently recorded in forest edges adjacent to open areas and dirt roads (Jorge 2008). Agoutis are granivorous caviomorph rodents, but despite their diet mainly comprises fruits and seeds, it is also complemented by leaves, roots, fungi and even small invertebrates (Henry 1999, Silvius & Fragoso 2003, Dubost & Henry 2006). It is likely that due to this adaptable diet, agoutis are benefiting from a variety of food resources present at forest edges, including natural resources and anthropogenic ones, thus increasing their density. Furthermore, agoutis may be benefiting from food resources from farms (Abrahams et al 2018) or, alternatively, they may be benefiting from the absence or low presence of predators, such as jaguars, close to communities (Carvalho & Pezzuti 2010). In addition, we noted that the recorded density of agoutis in the RESEX-TA is substantially higher (mean ± SD = 15.9 ± 7.74) compared to other Amazonian protected areas (Rosas 2006, Calouro & Marinho-Filho 2006, Endo et al. 2010, Mayor et al. 2015). Therefore, this greater density of agoutis demonstrates that hunting is not threatening their populations in the studied protected area.

The Tinamidae, encompassing both *Tinamus* spp. and *Crypturellus* spp., was notably influenced by anthropogenic pressure, presenting higher density estimates in transects far from roads. Although hunters prefer eating and therefore hunting large mammals (Constantino et al. 2008), large birds are also important items especially in heavily hunted forest sites (Thiollay 2005). In fact, local residents of RESEX-TA reported that tinamids can be hunted, but they are not the animals most frequently killed (personal communication from YMS Reis). Furthermore, large forest-dwelling birds such as the large tinamids, exhibit life-history characteristics associated with greater sensitivity to anthropogenic disturbances, such as a frugivore-granivore diet and forest dependence (Vetter et al. 2011). Therefore, two hypotheses might explain the obtained result. Firstly, the easier accessibility of hunters nearby roads might facilitate the capture of these animals, therefore reducing their population density. In fact, there is robust evidence in the literature revealing the great sensitivity of large birds to hunting pressure (Peres 2001, Urquiza-Haas et al. 2009, 2011). Secondly, as tinamid species are highly dependent on forest-interior environments, they might consequently avoid transiting in areas close to open areas. Indeed, no individuals were seen in two transects close to roads in some monitoring years. Our results also revealed a low density of tinamids in the RESEX-TA (mean ± SD = 2.1 ± 1.19), contrasting to other Amazonian protected areas (Endo et al. 2010, Peres & Nascimento 2006). Although our time-series analyses revealed that tinamid density has not declined over the years across all surveyed transects, ensuring the continuity of monitoring in the same transects will enable us to identify whether hunting is causing a reduction in their population sizes in those transects near roads. The long-term monitoring will also help determine if implementing hunting regulations in these transects is necessary.

Our findings also revealed that the density of Primates was positively related to both human intensity and monitored years over the 6 years of the monitoring program. In particular, medium-sized primates are relatively tolerant to subsistence hunting (Peres 2000, Peres & Palacios 2007). In fact, the high number of sightings of capuchins was mainly responsible for this result. They are widely known for their ability to adapt to disturbed environments and exploit novel resources, such as plantations, which are common near the transects (Hill 2000). In addition, this species can replace a diet composed of naturally distributed resources for anthropogenic food products (see Liebsch & Mikich 2015, Mikich & Liebsch 2009). Therefore, it is possible that the high behavioral flexibility (Fragaszy et al. 2004) and the proximity to cultivation areas might explain the positive influence of human intensity on primate density. Moreover, it is possible that primates are benefiting from resources that were previously exploited by larger and more vulnerable species (Peres & Dolman 2000), such as spider monkey, which were not analyzed in this study. This may be leading to an increase in their populations, while larger species may be declining in number. The spider monkey has probably been driven to local extinction in many parts of the Arapiuns region due to overhunting, as previously mentioned (Peres et al. 2003). Furthermore, only a single record was documented by experienced monitors within the Tapajós region through the Monitora program. Finally, the increase of Primate density along the monitoring years might be associated with the reduction in hunting pressure in response to awareness and environmental education activities carried out by the Monitora. In particular, since the establishment of the monitoring program in the reserve, several educational programs were carried out, intending to demonstrate the importance of forest species to both ecosystem functionality and delivery of ecosystem services. Indeed, we have noticed a growing awareness of local residents on the importance of wild fauna along the monitoring years. Considering that primates were not favorite targets in the RESEX-TA, it is possible that hunting pressure on this group is indeed declining along the years, and therefore reflecting in greater densities. The continuity of the monitoring in the reserve, together with interviews to local hunters, can better elucidate this observed pattern.

Our temporal analyses revealed the stability trend in the densities of game populations during the 6-years of the monitoring program in the RESEX-TA, for overall medium and large-sized groups and for each taxon investigated, suggesting that at least within a short-time period, hunting is not threatening diurnal, forest game species. Likewise, subsistence hunting was not considered a severe threat for game species in two other Amazonian sustainable-use extractive reserves, even for less resilient and low-fertility ones (Paula et al. 2022). This pattern is consistent with other studies across the Amazon basin, where terrestrial wildlife species showed resilience to subsistence hunting due to access to large tracts of continuous forests that are virtually inaccessible to hunters (Iwamura et al. 2014, Antunes et al. 2016). It seems that the environmental and social configuration of the RESEX-TA can be favoring a source–sink dynamic, whereby large expanses of lightly populated, undisturbed forest may serve as population "sources" that can refuel "sink" areas (i.e., those where hunting take place due to accessibility and proximity to local communities) (Novaro et al. 2000). It seems that the high forest coverage within the reserve, comprising approximately 70% of the territory, has played a crucial role in supporting the persistence of game populations. In addition, 20% of the reserve is designated as a preservation zone, where anthropogenic activities do not occur. Therefore, we presume that large tracts of forests, coupled with limited human pressure, may be providing the necessary conditions for game populations to thrive (Naranjo & Bodmer 2007, Ohl-Schacherer et al 2007).

Study limitations

Our data is limited to a set of game species liable to be recorded through line-transect diurnal surveys, which therefore exclude other important game species such as lowland paca (Cuniculus paca) and armadillo (Dasypodidae). Additionally, some species that could have been analyzed had low records (e.g., *Tapirus terrestris* and *Tayassu pecari*). This may be an indication of past population losses, reflecting a semi-defaunated current scenario. Furthermore, we recognize that analyses conducted at the Order or Family level may obscure the actual patterns of population densities. We therefore suggest that future studies include camera-trapping data to better assess the responses of the complete vertebrate game assemblage to anthropogenic stressors along the time. In addition, our outcomes related to the temporal analysis included a short-term period (i.e., six years), and considering the longevity and reproduction rates of vertebrate species, long-term monitoring is essential to assess density patterns of both overall and species-specific along generations.

Conclusion

Our study brings new information on the status of game populations within a sustainable-use protected area in the Amazon basin for both local people and reserve managers. Moreover, although the analyses did not indicate an influence of the anthropogenic evaluated variables on the populations of medium and large vertebrates considering the evaluated time series, we recognize the importance of considering the RESEX-TA system within a broader context. Previous studies carried out in this same area have documented the influence of wildfires and hunting pressure on forest structure and wildlife game populations (Peres et al. 2003, Barlow & Peres 2005). The history of wildfires and hunting pressure in the region, combined with the high human population density (~3,55 persons/km² for the entire region; Silva et al. 2022, Instituto Socioambiental 2022) may be contributing to a semi-defaunated scenario - i.e., instead of a pervasive loss of fauna, the RESEX-TA might has experienced loss of certain species previously to our study, reflecting in a low density of game species. Therefore, the stability of densities over the six analyzed years may be an indicator that the system is currently in a state of fragile equilibrium, potentially maintaining stability with an altered ecological structure. Further investigation, including continuous monitoring and interviews with hunters, can provide deeper insights to clarify these reflections.

We finally emphasize the importance of community-based long-term biodiversity monitoring in tropical protected areas, as it comprises a valuable tool to evaluate species status and therefore subsidize effective management actions.

Supplementary material

For supplementary material accompanying this paper, visit www.cambridge.org/core/journals/environmental-conservation

Acknowledgements

We are grateful to the community members and professionals involved in the Monitora Program and the local management of RESEX-TA. We are also grateful to the post-graduate program in Ecology and Biodiversity Conservation of the Universidade Estadual de Santa Cruz – UESC for providing the course "Análise de Dados e Redação de Manuscritos Científicos", which contributed to the development of the manuscript. In particular, we are grateful to C. R. Cassano for valuable comments on the previous versions of this manuscript. Finally, we also thank Hani El Bizri, Milton de Paula, Elildo Carvalho who shared analytical knowledge.

Financial support

This study was supported by the Coordination for the Improvement of Higher Education Personnel (CAPES) - Brazil (CAPES - Financing code 001) and CNPq (productivity grant 304189/2022-7 to M. Benchimol).

Ethical standards

The study complies with the current laws of Brazil, with the use of database from the biodiversity monitoring program (Monitora) authorized of the Biodiversity Authorization and Information System (SISBIO) of the Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio).

Conflict of interest

The authors declare no conflict of interest.

References

Abrahams MI, Peres CA, Costa HC (2018) Manioc losses by terrestrial vertebrates in western Brazilian Amazonia. The Journal of Wildlife Management, 82(4), pp.734-746.

Antunes AP, Fewster RM, Venticinque EM, Peres CA, Levi T, Rohe F, Shepard GH (2016) Empty forest or empty rivers? A century of commercial hunting in Amazonia. *Science Advances* 2: e1600936.

Beirne C, Meier AC., Mbele AE, Menie GM, Froese G, Okouyi J, Poulsen JR (2019) Participatory monitoring reveals village-centered gradients of mammalian defaunation in central Africa. *Biological conservation* 233: 228-238.

Benchimol M (2014) *Protocolo Complementar do Efeito da Caça de Subsistência sobre Espécies Cinegéticas na RESEX Tapajós-Arapiuns*. Programa de Monitoramento da Biodiversidade com relevância para o Clima. Brasília: ICMBio.

Benchimol M (2016) *Medium and Large-sized Mammals*. In: Core Standardized Methods for Rapid Biological Field Assessment, ed. Trond H.Larsen, pp. 37-48. Conservation International. USA: Arlington, VA.

Benítez-López A, Santini L, Schipper AM, Busana M, Huijbregts MA (2019) Intact but empty forests? Patterns of hunting-induced mammal defaunation in the tropics. *PLoS biology* 17(5): e3000247.

Bernardo CSS, Galetti M (2004) Densidade e tamanho populacional de primatas em um fragmento florestal no sudeste do brasil. *Revista Brasileira de Zoologia* 21: 827– 832.

Braga TMP, Silva JT, Rebêlo GH, Jensen L, Fugimura MMS (2022) Manejo dos recursos pesqueiros pelos ribeirinhos do baixo tapajós (pesca artesanal e piscicultura). In: *O Rio que nos une,* Andrade DFC, Spínola JN, pp 205-216. Brasília. Brazil: ICMBio.

Brooks M, Bolker B, Kristensen K, Maechler M (2022) Package 'glmmTMB' [www document]. URL http://cran.uni-muenster.de/web/packages/glmmTMB/glmmTMB.pdf

Burnham KP, Anderson DR (2002) Information and likelihood theory: a basis for model selection and inference. *Model selection and multimodel inference: a practical information-theoretic approach*, pp 49-97.

Calouro AM; Marinho-Filho JS (2006) A caça e a pesca de subsistência entre seringueiros ribeirinhos e não-ribeirinhos da Floresta Estadual do Antimary (AC). In: *Fauna do Acre*, ed EDUFAC, Drumond, Rio Branco, AC.

Cardillo M, Mace GM, Jones KE, Bielby J, Bininda-Emonds ORP, Sechrest W, Orme CDL, Purvis A (2005) Multiple causes of high extinction risk in large mammal species. *Science* 309: 1239–1241.

Carvalho Junior, EAR (2008) *Caçando onças na Reserva Extrativista Tapajós-Arapiuns. Relatório de atividades*. Santarém. Brazil: ICMBio.

Carvalho E, Pezzuti J (2010) Hunting of jaguars and pumas in the Tapajós–Arapiuns Extractive Reserve, Brazilian Amazonia. *Oryx* 44(4): 610-612.

Caughley G, Sinclair AR (1994) *Wildlife Ecology and Management*. Cambridge, UK: Blackwell Science.

Constantino PL, Fortini LB, Kaxinawa FRS, Kaxinawa AM, Kaxinawa ES, Kaxinawa AP, Kaxinawa, LS et al. (2008) Indigenous collaborative research for wildlife management in Amazonia: The case of the Kaxinawá, Acre, Brazil. *Biological Conservation* 141: 2718-2719.

Constantino P (2016) Deforestation and hunting effects on wildlife across Amazonian indigenous lands. *Ecology and Society* 21(2).

Danielsen F, Eicken H, Funder M, Johnson N, Lee O, Theilade I, Argyriou D et al. (2022) Community monitoring of natural resource systems and the environment. *Annual Review of Environment and Resources* 47: 637-670.

Danielsen F, Enghoff M, Poulsen MK, Funder M, Jensen PM, Burgess ND (2021) The concept, practice, application, and results of locally based monitoring of the environment. *BioScience* 71(5): 484-502.

de Paula MJ, Carvalho EA, Lopes CKM, de Alencar Sousa R, Maciel ELP, Wariss M, Barboza RSL et al. (2022) Hunting sustainability within two eastern Amazon Extractive Reserves. *Environmental Conservation* 49(2):90-98.

Dubost G, Henry O (2006) Comparison of diets of the acouchy, agouti and paca, the three largest terrestrial rodents of French Guianana forests. *Journal of Tropical Ecology* 22(6): 641-651.

Endo W, Peres CA, Salas E, Mori S, Sanchez-Vega JL, Shepard GH, ... & Yu, D. W. (2010). Game vertebrate densities in hunted and nonhunted forest sites in Manu National Park, Peru. Biotropica, 42(2), 251-261.

Fragaszy DM, Fedigan L, Visalberghi E (2004) *The complete capuchin: The biology of the genus Cebus*. Cambridge: Cambridge University Press.

Galetti M, Bovendorp RS, Guevara R. (2015) Defaunation of large mammals leads to an increase in seed predation in the Atlantic forests. *Global Ecology and Conservation* (3): 824-830.

Gelman A, Shirley K (2011) Inference from simulations and monitoring convergence. In: *Handbook of Markov Chain Monte Carlo*, eds S Brooks, A Gelman, GL Jones, XL Meng (pp. 163–174). Boca Raton, FL, USA: Chapman & Hall/CRC Press.

Henry O (1999) Frugivory and the importance of seeds in the diet of the orange-rumped agouti (*Dasyprocta leporina*) in French Guiana. *Journal of Tropical Ecology* 15(3): 291-300.

Hill CM (2000) Conflict of interest between people and baboons: crop raiding in Uganda. *International Journal of Primatology* 21(2): 299–315.

Instituto Brasileiro de Geografia e Estatística (1992) *Manual técnico da vegetação brasileira*. Rio de Janeiro. Brazil: IBGE.

Instituto Chico Mendes de Conservação da Biodiversidade (2011) *Mapeamento participativo do uso dos recursos naturais na RESEX Tapajós-Arapiuns*. Conservação Internacional-CI, Associações da Reserva Extrativista Tapajós-Arapiuns-Tapajoara. Brazil: ICMBio.

Instituto Chico Mendes de Conservação da Biodiversidade (2014). *Plano de Manejo da Reserva Extrativista Tapajós-Arapiuns*. Brasília. Brazil: ICMBio.

Instituto Nacional de Pesquisas Espaciais, Centro de Previsão de Tempo e Estudos Climáticos (2023) Precipitação diária 2023 [www document]. URL http://clima1.cptec.inpe.br/estacaochuvosa/pt

Instituto Socioambiental (2023) Unidades de Conservação no Brasil [www document]. URL https://uc.socioambiental.org/pt-br/arp/1353

Iwamura T, Lambin EF, Silvius KM, Luzar JB, Fragoso JM (2014) Agent-based modeling of hunting and subsistence agriculture on indigenous lands: Understanding interactions between social and ecological systems. *Environmental Modelling & Software* 58:109-127.

Jerozolimski A, Peres CA (2003) Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. *Biological Conservation* 111(3): 415-425.

Jorge MLS (2008) Effects of forest fragmentation on two sister genera of Amazonian rodents (*Myoprocta acouchy* and *Dasyprocta leporina*). *Biological Conservation* 141(3): 617-623.

Liebsch D, Mikich SB (2015) First record of Eucalyptus spp. bark-stripping by brown-capuchin monkeys (*Sapajus nigritus*, primates: cebidae). *Ciência Florestal* 25(2): 501–505.

Lima D, Pozzobon J (2005) Amazônia socioambiental: sustentabilidade ecológica e diversidade social. *Estud. Avançados* 19: 45–76.

Kéry M (2010) *Introduction to WinBUGS for Ecologists: A Bayesian Approach to Regression, ANOVA, Mixed Models and Related Analyses*. Amsterdan, The Netherlands: Elsevier.

Koerner SE, Poulsen JR, Blanchard EJ, Okouyi J, Clark CJ (2017) Vertebrate community composition and diversity declines along a defaunation gradient radiating from rural villages in Gabon. J. *Appl. Ecol.* 54: 805–814.

MacArthur RH, Diamond JM, Karr JR (1972) Density compensation in island faunas. *Ecology* 53: 330–342.

Mayor P, Pérez-Peña P, Bowler M, Puertas PE, Kirkland M, Bodmer R (2015) Effects of selective logging on large mammal populations in a remote indigenous territory in the northern Peruvian Amazon. *Ecology and Society* 20(4): 36.

Mikich SB, Liebsch D (2009) O macaco-prego e os plantios de *Pinus* spp. Comunicado técnico. *Embrapa Florestas* 234: 1–5.

Montag FFA, Prudente BS, Ferreira CP, Dutra GM, Benone NL, Barbosa TAP, Ruffeil TOB (2012) *Ictiofauna na Reserva Extrativista Tapajós-Arapiuns, Estado do Pará, Brasil*. Pará. Brazil: UFPA.

Naranjo EJ, Bodmer RE (2007) Source–sink systems and conservation of hunted ungulates in the Lacandon Forest, Mexico. *Biological Conservation* 138: 412–420.

Nobre RDA, Kinouchi MR, Constantino PAL, Costa-Pereira R, Uehara-Prado M (2014) *Monitoramento da biodiversidade: roteiro metodológico de aplicação*. Brasília. Brazil: ICMBio.

Novaro AJ, Redford KH, Bodmer RE (2000) Effect of hunting in source–sink systems in the Neotropics. *Conservation Biology* 14: 713–721.

Ohl-Schacherer J, Shepard Jr GH, Kaplan H, Peres CA, Levi T, Yu DW (2007) The sustainability of subsistence hunting by Matsigenka native communities in Manu National Park, Peru. *Conservation Biology* 21(5): 1174-1185.

Oliveira ACM, Junior OC, Chaves R (2005) Gestão participativa e a atividade de caça na Reserva Extrativista do Tapajós – Arapiuns, Santarém, PA. *Raízes* 23: 42–51.

Peres CA (1999) General guidelines for standardizing line-transect surveys of tropical forest primates. *Neotropical primates* 7(1): 11-16.

Peres CA (2000) Effects of subsistence hunting on vertebrate community structure in Amazonian forests. *Conservation Biology* 14: 240-253.

Peres CA (2001) Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian forest vertebrates. *Conservation Biology* 15: 1490–1505.

Peres CA, Dolman P M (2000) Density compensation in neotropical primate communities: evidence from 56 hunted and nonhunted Amazonian forests of varying productivity. *Oecologia* 122: 175-189.

Peres C, Barlow J, Haugaasen T (2003) Vertebrate responses to surface wildfires in a central Amazonian forest. *Oryx* 37(1): 97-109.

Peres CA, Nascimento HS (2006) Impact of game hunting by the Kayapó of south-eastern Amazonia: implications for wildlife conservation in tropical forest indigenous reserves. *Human exploitation and biodiversity conservation* 15:2627–2653.

Peres CA, Palacios E (2007) Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: implications for animal-mediated seed dispersal. *Biotropica* 39: 304–315.

Plummer M (2015) JAGS Version 4.0.0 user manual 0–41 [www document]. URL https://www.uvm.edu/~bbeckage/Teaching/PBIO_294/Manuals/manual.jags.pdf

QGIS Development Team (2017) QGIS Geographic Information System.

Reis YMS, Benchimol M (2023) Effectiveness of community-based monitoring projects of terrestrial game fauna in the tropics: a global review. *Perspectives in Ecology and Conservation*. https://doi.org/10.1016/j.pecon.2023.03.005.

Rosas GKC (2006) Pressão de caça, abundância, densidade e riqueza de mamíferos em duas áreas de coleta de castanha-do-brasil situadas no sudoeste do estado do Acre, Brasil. Brazil: Universidade Federal do Acre.

Royle JA, Kéry M (2007) A Bayesian state-space formulation of dynamic occupancy models. *Ecology* 88: 1813–1823.

Sampaio R, Morato RG, Royle A, Abrahams MI, Peres CA, Chiarello AG (2023) Vertebrate population changes induced by hunting in Amazonian sustainable-use protected areas. *Biological Conservation* 284: 110206.

Schmeller DS, Böhm M, Arvanitidis C, Barber-Meyer S, Brummitt N, Chandler M, Chatzinikolaou E, Costello MJ, Ding H, García-Moreno J, Gill M (2017) Building capacity in biodiversity monitoring at the global scale. *Biodiversity and conservation* 26: 2765-90.

Silva MJS, Nascimento BDC, Magalhães LFR, Spínola JN (2022) Histórico de criação e implementação da Floresta Nacional do Tapajós e da Reserva Extrativista Tapajós-Arapiuns. In: *O Rio que nos une*, Andrade DFC, Spínola JN, pp 205-216. Brasília. Brazil: ICMBio.

Silvius KM, Fragoso JV (2003) Red-rumped agouti (*Dasyprocta leporina*) home range use in an Amazonian Forest: implications for the aggregated distribution of forest trees. *Biotropica* 35: 74-83.

Thomas L, Buckland ST, Rexstad EA, Laake JL, Strindberg S, Hedley SL, Bishop JRB, Marques TA, Burnham KP (2010) Distance software: design and analysis of distance sampling surveys for estimating population size. *J. Appl. Ecol.* 47: 5–14.

Team R C (2022) R: A language and environment for statistical computing. URL: https://www.r-project.org/

Reis YS, Valsecchi J, Queiroz H (2018) Caracterização do uso da fauna silvestre para subsistência em uma unidade de conservação no Oeste do Pará. *Biodiversidade Brasileira* 9(2): 187-202.

Reis YS, Soares CCM, Maduro R, Spinola JN, da Rocha BD (2022) O padrão da caça de subsistência em uma reserva extrativista na Amazônia oriental, Brasil. *Etnobiología* 20(1): 18-26.

Ribeiro KT (2018) *Estratégia do Programa Nacional de Monitoramento da Biodiversidade– Programa Monitora: estrutura, articulações, perspectivas*. Brasília. Brazil: ICMBio.

Saúde & Alegria (2012) *Mapeamentos Participativos e elaboração de Planos de Uso: uma experiência nas comunidades de Anã, Arimum, Atodi e Vila Amazonas nas margens do Rio Arapiuns*. Saúde & Alegria, Brazil: Santarém, Pará.

Scabin AB, Peres CA (2021) Hunting pressure modulates the composition and size structure of terrestrial and arboreal vertebrates in Amazonian forests. *Biodivers Conserv* 30: 3613–3632.

Su YS, Yajima M (2012) R2jags: a package for running JAGS from R. R package version 0.03-08 [www document]. URL https://cran.r-project.org/web/packages/R2jags/index.html

Thiollay JM (2005) Effects of hunting on Guianan forest game birds. *Biodiversity and Conservation* 14: 1121–1135.

Urquiza-Haas T, Peres CA, Dolman PM (2009) Regional scale effects of human density and forest disturbance on large-bodied vertebrates throughout the Yucatán Peninsula, Mexico. *Biological Conservation* 142: 134–148.

Urquiza-Haas T, Peres CA, Dolman PM (2011) Large vertebrate responses to forest cover and hunting pressure in communal landholdings and protected areas of the Yucatan Peninsula, Mexico. *Animal Conservation* 14:271-282.

Veiga JE, Ehlers E (2003) Diversidade biológica e dinamismo econômico no meio rural. *Economia do meio ambiente: teoria e prática*, eds Campus (pp 271-290). Rio de Janeiro. Brazil: Ed. Campus.

Veloso HP, Rangel-Filho ALR, Lima JCA (1991) Classificação da vegetação brasileira, adaptada a um sistema universal. Brazil: IBGE.

Vetter D, Hansbauer M, Vegvari Z, Storch I (2011) Predictors of forest fragmentation sensitivity in Neotropical vertebrates: a quantitative review. *Ecography* 34: 1–8.

Supplementary data

Table S1. Taxa and species targeted for hunting in the Tapajós-Arapiuns Extractive Reserve, corresponding to the focal species of the Monitora program and recorded in hunting events. This information includes the common name of each species, their average weight, the order of consumption preference among local residents (unpublished data from the hunting activity census, carried out in 2019), the number of sightings per year and in total, as well as the number of transects in which they were recorded between 2015 and 2020.

Taxa	Target species	Popular name	Average size (kg)	Preference	N° sightings/Year						N° sightings	N° transects
Mammals					1	2	3	4	5	6		
Atelidae	*Alouatta nigerrima*	howler monkeys	6.19	secondary	53	31	124	62	54	103	427	8
Primates	*Callicebus hoffmannsi*	titi monkeys	0.92	tertiary	4	24	22	30	17	6	103	8
	Cebus unicolor	Spix's white-fronted capuchin	2.25	tertiary	-	18	51	64	114	64	311	6
	Sapajus paella	capuchin monkeys	2.91	secondary	58	169	471	364	598	302	1962	8
	Pithecia irrorata	Vanzolini's Bald-faced Saki	2.20	tertiary	-	7	27	41	56	6	137	7
Tayassuidae	*Dicotyles tajacu*	collared peccary	20.00	primary	1	17	37	32	51	27	165	8
	Tayassu pecari	white-lipped peccary	25.00	primary	-	12	1	24	-	-	37	3
Cervidae	*Mazama Americana*	red brocket deer	30.00	primary	1	10	17	17	38	35	118	8
	Mazama nemorivaga	brown brocket deer	25.00	primary	-	2	12	14	13	9	50	6
	M. americana or M. nemorivaga		27.50		8	17	17	3	-	-	45	8
Dasypodidae	*Dasypus* spp.	armadillo	6.00	primary	1	-	-	3	1	-	5	3
	Priodontes maximus	giant armadillo	39.40	tertiary	-	-	-	-	-	-	-	-
Tapiridae	*Tapirus terrestris*	lowland tapir	160.00	primary	-	-	3	1	-	-	4	3

Cuniculidae	*Cuniculus paca*	lowland paca	7.82	primary	-	-	-	2	-	-	2	1
Dasyproctidae	*Dasyprocta croconota*	agouti	2.50	primary	75	208	242	191	229	225	1170	8
Felidae	*Panthera onca*	jaguar	68.75	tertiary	1	-	-	1	1	1	4	3
	Puma concolor	puma	34.50	tertiary	-	-	3	2	-	-	5	2
Procyonidae	*Nasua nasua*	coati	3.88	secondary	5	5	95	155	176	76	512	7
Birds												
Cracidae	*Crax* spp., *Pauxi tuberosa*	curassows	1.88	secondary	6	7	18	7	11	18	67	7
	Penelope spp.	common guan	1.88	primary	26	30	82	69	57	83	347	8
	Crax spp., *Pauxi tuberosa* or *Penelope spp.*		1.88		-	5	3	-	18	-	26	2
Psittacidae	*Ara chloropterus, Anodorhynchus hyacinthinus, Amazona spp.,*	macaw and parrot	1.13	secondary	-	-	-	-	-	-	-	-
Psophiidae	*Psophia viridis*	dark-winged trumpeter	0.99	secondary	41	28	113	87	54	44	367	7
Ramphastidae	*Ramphastos* spp.	toucan	0.54	tertiary	-	-	-	-	-	-	-	-
Tinamidae	*Crypturellus* spp.	tinamous	0.60	primary	11	8	29	25	42	13	128	8
	Tinamus spp.	tinamous	0.80	primary	-	7	28	18	22	13	88	7
	Tinamus spp. or *Crypturellus* spp.		0.70		-	10	3	-	-	8	21	4
Reptiles												
Testudinidae	*Chelonoidis carbonaria*, C. *denticulate*	forest tortoise	7.79	primary	-	-	-	-	-	-	-	-
Podocnemididae	*Podocnemis unifilis*	yellow-spotted river turtle	6.08	tertiary	-	-	-	-	-	-	-	-

Text summary:

The table includes the taxa and species targeted for hunting in the Tapajós-Arapiuns Extractive Reserve, corresponding to the focal species of the Monitora program and recorded in hunting events. This information includes the common name of each species, their average weight, the order of consumption preference among local residents (unpublished data from the hunting activity census, carried out in 2019), the number of sightings per year and in total, as well as the number of transects in which they were recorded between 2015 and 2020.

Table S2. Annual effort (in km) per linear-transect performed by the Monitora Program in the Tapajós-Arapiuns Extractive Reserve, between 2015 and 2020.

Transect	2015	2016	2017	2018	2019	2020	Total	Mean
Anã	0	0	60	50	45	40	195	32.5
Boim	15	45	65	45	50	50	270	45
Cametá	45	45	60	45	50	50	295	49.17
Escrivão	10	40	30	45	48	35	208	34.67
Pascoal	25	45	60	50	50	40	270	45
Prainha	0	0	60	27	50	30	167	27.83
São Pedro	5	30	50	50	35	50	220	36.67
São Tomé	45	35	65	45	50	50	290	48.33
Total	145	240	450	357	378	345	1915	319.17
Mean	18.13	30	56.25	44.63	47.25	43.13	239.38	

Text summary:

The table includes the average and total annual effort values (in km) of line-transect surveys, for sampling of medium-sized and large-sized game forest species, performed by trained local monitors of the biodiversity monitoring program (Monitora) in eight transects within the Tapajós-Arapiuns Extractive Reserve between 2015 and 2020.

Table S3. Best models for effective width (EW), based on values of the Akaike information criterion (AIC), coefficient of variation (DCV) and the value of the GOF Chi-p.

	Model	Function	EW (m)	AIC	DCV	GOF Chi-p
Cracidae	hazard-rate	simple polynomial	38	632.47	0.16	0.98
Dasyproctidae	half-normal	hermite polynomial	20	1986.21	0.1	0.74
Procyonidae	half-normal	hermite polynomial	27	226.92	0.32	0.93
Primates	hazard-rate	simple polynomial	50	1092.65	0.13	0.72
Psophidae	hazard-rate	simple polynomial	25	208.21	0.22	0.98
Tinamidae	half-normal	cosine	30	432.75	0.15	0.65
Atelidae	half-normal	cosine	50	222.7	0.21	0.57
Cervidae	half-normal	cosine	30	553.11	0.16	0.99
Tayassuidae	half-normal	hermite polynomial	25	94.86	0.26	0.55

Text summary:

The table includes the best fit models for effective width (EW) of the transect for nine analyzed taxa, selected with base on the lowest values of the Akaike information criterion (AIC), coefficient of variation (DCV) and the value of the GOF Chi-p, using the *Distance* 4.1 software.

Table S4. Mean, standard deviation, minimum and maximum values of the proxy of human intensity and distance to nearest road (anthropogenic variables).

	Mean	Standard deviation	Minimum	Maximum
Proxy of human intensity	306.78	273.31	0	680.60
Distance to nearest road (km)	10.72	10.33	2.11	30.99

Text summary:

The table includes the mean, standard deviation, minimum and maximum values of the proxy of human intensity and distance to nearest road, anthropogenic variables used to assess their influence on overall and individual patterns of mammal and bird forest game species in eight transects within the Tapajós-Arapiuns Extractive Reserve.

Table S5. Average density and biomass of nine game taxa or group (medium-sized and large-sized game species) in eight linear-transects in the Tapajós-Arapiuns Extractive Reserve.

	Medium-sized taxa							Large-sized taxa			
Transect	Cracidae	Dasyproctidae	Procyonidae	Primates	Psophidae	Tinamidae	**Total**	Atelidae	Cervidae	Tayassuidae	**Total**
Density (ind./km²)											
Anã	0.2	22.2	39.4	23.8	0.0	0.9	**86.5**	1.3	2.7	6.2	**10.3**
Boim	1.0	14.1	0.9	4.7	2.7	0.8	**24.3**	0.3	1.8	0.5	**2.7**
Cametá	2.6	5.3	1.4	29.2	3.0	4.4	**45.9**	5.3	2.1	1.3	**8.7**
Escrivão	3.6	8.4	0.5	9.3	2.4	3.2	**27.4**	4.0	1.3	3.3	**8.6**
Pascoal	4.0	29.0	0.9	7.6	6.5	2.6	**50.5**	0.8	1.8	0.9	**3.4**
Prainha	5.7	12.9	2.3	12.3	9.3	1.9	**44.3**	1.0	2.6	6.3	**10.0**
São Pedro	5.7	24.2	2.6	9.3	3.8	2.0	**47.7**	1.2	2.8	0.9	**5.0**
São Tomé	2.7	11.6	0.0	6.0	4.0	1.0	**25.3**	2.7	0.3	0.3	**3.3**
Mean	**3.2**	**15.9**	**6.0**	**12.8**	**4.0**	**2.1**	**44.0**	**2.1**	**1.9**	**2.5**	**6.5**
Biomass (kg/km²)											
Anã	0.3	55.4	152.9	49.3	0.0	0.6	**258.6**	8.1	75.5	140.3	**223.9**
Boim	1.9	35.3	3.5	9.8	2.7	0.6	**53.7**	2.1	50.5	10.8	**63.5**
Cametá	4.9	13.3	5.5	60.4	2.9	3.1	**90.1**	32.6	57.9	29.6	**120.0**
Escrivão	6.8	20.9	2.0	19.2	2.4	2.2	**53.6**	24.5	37.0	74.6	**136.1**
Pascoal	7.4	72.4	3.5	15.8	6.4	1.8	**107.4**	5.0	48.8	19.4	**73.2**
Prainha	10.6	32.1	8.8	25.5	9.2	1.3	**87.6**	6.3	72.4	141.8	**220.5**
São Pedro	10.7	60.6	10.0	19.3	3.7	1.4	**105.7**	7.7	77.1	20.4	**105.2**
São Tomé	5.0	29.0	0.0	12.4	4.0	0.7	**51.0**	16.6	9.5	5.8	**31.9**
Mean	**6.0**	**39.9**	**23.3**	**26.5**	**3.9**	**1.5**	**101.0**	**12.9**	**53.6**	**55.3**	**121.8**

Text summary:

The table includes the average density (ind./km²) and biomass (kg/km²) estimates of nine game taxa (Cracidae, Dasyproctidae, Procyonidae, Primates, Psophidae, Tinamidae, Atelidae, Cervidae and Tayassuidae) or group (medium-sized and large-sized game species) in eight linear-transects within Tapajós-Arapiuns Extractive Reserve. Additionally, it provides the total density and biomass estimates per transect.

Table S6. Coefficients (Coef), standard error (SE), z-value and correspondent significance [Pr (>|z|)] and 95% confidence intervals (CI 95%) of the variables included in the best models explaining the variability in density of Dasyproctidae, Tinamidae and Primates. The variable in bold has CI 95% that do not cross 0.

Predictor	Coef	SE	z-value	Pr(>|z|)	CI 95%	
Dasyproctidae						
Model 1						
Intercept	4.62	0.33	14.06	<0.001	4.08	5.16
distance to nearest road	**-0.08**	**0.02**	**-3.38**	**<0.001**	**-0.11**	**-0.04**
Model 2						
Intercept	4.91	0.36	13.73	<0.00	4.32	5.50
distance to nearest road	**-0.07**	**0.02**	**-3.59**	**0.00**	**-0.11**	**-0.04**
human intensity	0.00	0.00	-1.42	0.16	0.00	0.00
Model 3						
Intercept	4.67	0.36	12.91	<0.001	4.07	5.26
distance to nearest road	-0.04	0.03	-1.46	0.14	-0.09	0.01
human intensity	0.00	0.00	0.01	1.00	0.00	0.00
distance to nearest road:human intensity	0.00	0.00	-1.44	0.15	0.00	0.00
Model 4						
Intercept	98.40	120.70	0.82	0.41	-100.13	296.93
Year	-0.05	0.06	-0.78	0.44	-0.14	0.05
distance to nearest road	**-0.08**	**0.02**	**-3.38**	**0.00**	**-0.11**	**-0.04**
Tinamidae						
Intercept	-63.58	119	-0.53	0.59	-259.28	132.11
Distance to nearest road	**0.03**	**0.01**	**2.95**	**0.00**	**0.01**	**0.05**
Primates						
Model 1						
Intercept	-709.00	210.60	-3.37	0.00	-1055.38	-362.63
Year	**0.35**	**0.10**	**3.38**	**0.00**	**0.18**	**0.52**
human intensity	**0.00**	**0.00**	**2.46**	**0.01**	**0.00**	**0.00**
Model 2						
Intercept	-720.50	210.90	3.42	0.00	-1067.46	-373.62
Year	**0.36**	**0.10**	**3.43**	**0.00**	**0.19**	**0.53**
human intensity	**0.00**	**0.00**	**2.49**	**0.01**	**0.00**	**0.00**
distance to nearest road	0.03	0.03	1.20	0.23	-0.01	0.07
Model 3						

Intercept	-701.73	211.78	-3.31	0.00	-1050.08	-353.38
Year	**0.35**	**0.11**	**3.33**	**0.00**	**0.18**	**0.52**

Text summary:

The table includes coefficients (Coef), standard error (SE), z-value and correspondent significance [Pr (>|z|)] and 95% confidence intervals (CI 95%) of the variables included in the best models explaining the variability in density of Dasyproctidae, Tinamidae and Primates. The variable in bold has CI 95% that do not cross 0.

Table S7. State-space models examining temporal trends in terms of densities (ind./km²) of the medium-sized and large-sized game groups and nine individual taxa (Tinamidae, Psophiidae, Cracidae, Dasyproctidae, Primates, Procyonidae, Atelidae, Cervidae and Tayassuidae) in eight transects surveyed across the Tapajós-Arapiuns Extractive Reserve from 2015 to 2020.

Density	Parameter	MTS	SD	2.5% BCI	97.5% BCI	Rhat
Medium-sized group	*R*	0.03	0.23	-0.46	0.53	1.001
Large-sized group	*R*	0.05	0.14	-0.25	0.35	1.001
Tinamidae	*R*	0.02	0.26	-0.52	0.56	1.001
Psophiidae	*R*	-0.01	0.33	-0.69	0.68	1.001
Cracidae	*R*	0.10	0.18	-0.31	0.50	1.001
Dasyproctidae	*R*	-0.06	0.25	-0.60	0.48	1.001
Primates	*R*	0.09	0.31	-0.56	0.72	1.001
Procyonidae	*R*	0.62	0.41	-0.22	1.43	1.001
Atelidae	*R*	0.15	0.34	-0.52	0.86	1.001
Cervidae	*R*	0.03	0.25	-0.50	0.54	1.001
Tayassuidae	*R*	-0.04	0.30	-0.66	0.58	1.001

Text summary:

The table includes the state-space models examining temporal trends in terms of densities (ind./km²) of the medium-sized and large-sized game groups and nine individual taxa (Tinamidae, Psophiidae, Cracidae, Dasyproctidae, Primates, Procyonidae, Atelidae, Cervidae and Tayassuidae) in eight transects surveyed across the Tapajós-Arapiuns Extractive Reserve from 2015 to 2020.

CAPÍTULO 3

Game harvest in the most populated Amazonian sustainable-use protected area: insights from 5-years of community-based monitoring

Article prepared for submission to *Conservation and Society* (A3, IF 2.49).

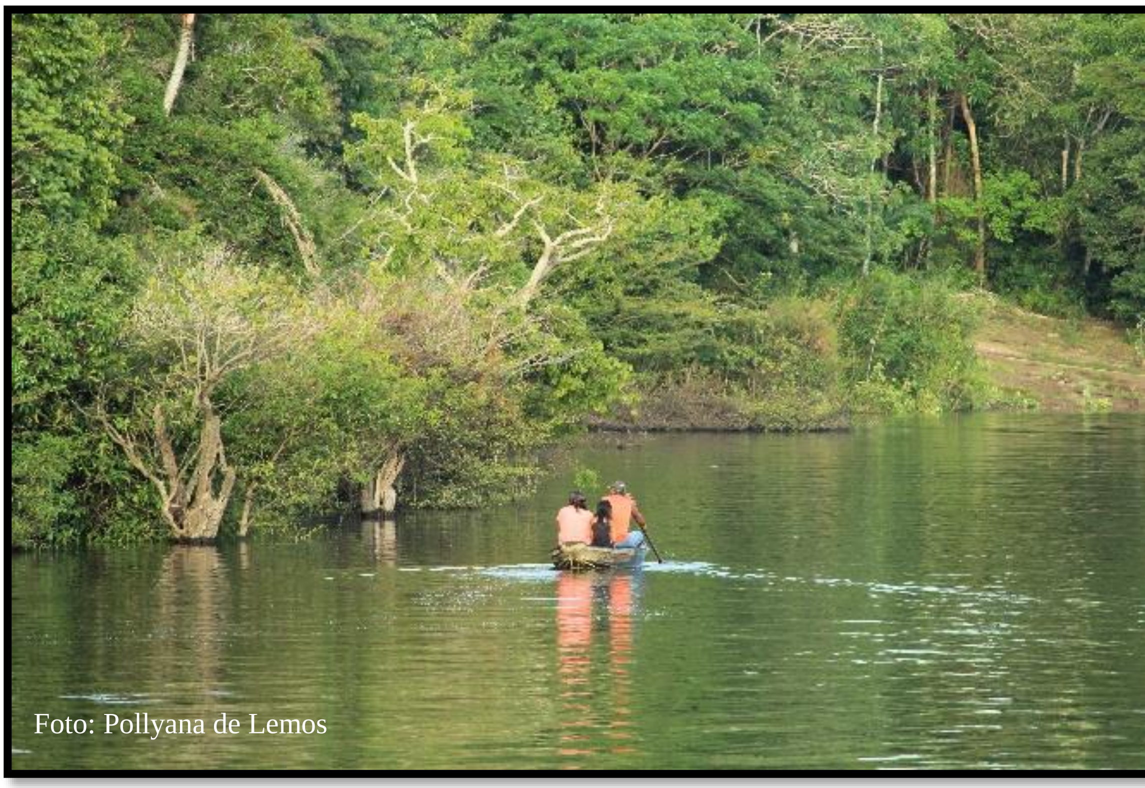


Foto: Pollyana de Lemos

Game harvest in the most populated Amazonian sustainable-use protected area: insights from 5-years of community-based monitoring

Running head: Sustainable Use in Amazon: Game Harvest Insights from 5 Years

Yasmin Maria Sampaio dos Reis[1*], Maíra Benchimol[2]

[1,2]Applied Ecology & Conservation Lab, Programa de Pós-graduação em Ecologia e Conservação da Biodiversidade, Universidade Estadual de Santa Cruz, Rodovia Ilhéus-Itabuna, km 16, Salobrinho, 45662-000 Ilhéus, Bahia, Brazil

E-mail addresses: yasmin.sampaioreis@gmail.com (Y. Reis)*, mairabs02@gmail.com (M. Benchimol)

*Corresponding author.

Abstract

Although wildlife long been crucial for food provision and traditions, wildlife exploitation can pervasively affect populations. Thus, game harvest monitoring initiatives became a key tool across tropical forests. We used a database from a 5-years community-based monitoring conducted in the Tapajós-Arapiuns Extractive Reserve (Brazilian Amazonia), to examine the hunting profile across different villages and assess the influence of anthropogenic stressors and monitored year on patterns of both community composition (considering relative frequency and relative biomass) and hunting productivity through CPUE (catch per unit of effort). We further assessed trends in CPUE of all game species and the six most hunted species along the time series. A total of 5,760 hunting events were performed by 391 families from 13 villages, resulting in 6,436 hunted animals from 24 taxa and a harvest of 65,488 kg. Medium to large-sized mammals comprised the main targets of hunters, yet villages exhibited different hunting profiles, thereby reflecting different hunting strategies and species abundance *in situ*. We showed that anthropogenic variables did not affect the community composition and CPUE, and CPUE remained stable through time. Our findings suggest that hunting is likely sustainable for most species, with the exception of those large and threatened species like *Tapirus terrestris* and *Tayassu pecari*. We finally encourage that other Amazonian sustainable-use reserves monitor subsistence hunting through our hunting protocol.

Key words: Biodiversity monitoring, participatory monitoring, game fauna, mammals, birds, hunting pressure.

1. **INTRODUCTION**

Throughout human history, wildlife has always been crucial for food provision, and still to date, countless people continue to rely on animals for their diets, traditions, and/or livelihoods (Ingram et al. 2021, Fa et al. 2022). In tropical regions, wild meat has great importance to traditional communities, as it provides protein, fat, and micronutrients essential to their subsistence (Sirén, 2008, Van Vliet et al. 2017). However, overhunting combined with unprecedented habitat loss can severely threat wildlife, especially those large animals that are more vulnerable to anthropogenic pressures (Davidson et al. 2009, Ripple et al. 2016). In fact, the decline of game vertebrate populations due to overhunting is widespread, particularly in Africa and Asia (Benítez-López et al. 2019), likely affecting the long-term persistence of game species (Wilkie et al. 2011). Therefore, monitoring game harvest in the long-term becomes crucial to improve our understanding on the effects of hunting on wildlife populations.

Over the past two decades, community-based monitoring projects have been significantly expanding. Indeed, these initiatives play a vital role in monitoring game fauna and harvest in tropical countries, particularly within sustainable-use protected areas (Reis & Benchimol 2023). Characterized by high biodiversity and inhabited by indigenous or traditional communities (Lima & Pozzobon 2005), such protected areas aim to conciliate biodiversity conservation with the sustainable use of natural resources. However, limited information is available on the effectiveness of sustainable-use protected areas in achieving this goal. Nevertheless, community-based monitoring projects focused on biodiversity or extraction of natural resources can contribute on this evaluation. For instance, game offtake data collected by local people within the Manu National Park in Peru showed minimal or no depletion of the main hunted species (Ohl-Schacherer et al. 2007), suggesting that hunting is likely sustainable and the park has been successfully safeguarding the game fauna while providing food security for local communities.

Across the Amazon basin, several community-based hunting monitoring initiatives have so far been implemented (Zapata-Ríos et al. 2009, Luzar et al. 2011, Constantino et al. 2012, Mayor et al. 2017, El Bizri et al. 2020, Oliveira & Calouro 2020). Considering that medium to large-sized mammals and birds are the main targets of hunters in Amazonian forests, most existing monitoring programs in terrestrial ecosystems focus on this group. In particular, large-sized mammals are favored by hunters due to their meat flavor and higher hunting returns (Constantino et al. 2008). However, these species are more vulnerable to extirpation or population reduction given their lower reproductive rates and long generation times (Bodmer et al. 1997, Cardillo et al. 2005). Once large mammals are vanished or rarely found, hunters can capture medium-sized species or start foraging farther away from their own villages (Stafford et al. 2017), demanding greater effort and resulting in lower hunting productivity (i.e., a lower rate of meat obtained per hunter per hour hunted; Weinbaum et al. 2013, Iwamura et al. 2014).

Assessing the sustainability of hunting is not an easy task, as is often require the contribution of hunters in providing their foraging information to enable the monitoring of hunting productivity. One of the most useful procedures is recording the catch per unit of effort (CPUE), i.e., the quantity (in kg, for instance) of specific animals caught per hour during a hunting trip. This indicator reflects changes in prey population abundances, which is supported by the behavior of hunters to intensify their efforts in regions where preferred large-sized species are depleted to achieve the desired meat return rates (Souza-Mazurek et al. 2000, Sirén et al. 2004, Parry et al. 2009). Consequently, factors contributing to a decline in CPUE of these preferred species (e.g. anthropogenic stressors) ultimately lead to a reduction in their overall abundance. Furthermore, estimating the CPUE over time allows for the quantification of population trends. For instance, in the Likouala region of the Republic of the Congo, researchers monitored the CPUE of game terrestrial fauna over 10 consecutive years and provided evidence that bushmeat is hunted at unsustainable rates throughout much of the Congo basin (Riddell et al. 2022).

Several anthropogenic factors have been halted as highly associated to hunting pressure for forest game species, including hunter access points (e.g. villages), human population density (Beirne et al. 2019, Benítez-López et al. 2019) and village age (Constantino 2015). For instance, Jerozolimski & Peres (2003) demonstrated that older communities exerted greater hunting pressure on large game vertebrates across Neotropical forest settlements, in which hunters needed to shift their target species from large-sized to numerous small-sized as the villages aged. Therefore, assessing the influence of anthropogenic variables on hunting patterns becomes very useful specially in sustainable-use protected areas, given their potential in leading vertebrates to population depletion.

Here, we used a database from 5-years of a community-based hunting monitoring conducted within a sustainable-use protected area in the Brazilian Amazonia to describe the subsistence hunting profile across eight hub villages from 2015 to 2019. In addition, we examine how (i) anthropogenic stressors and monitored year modulate patterns of community composition (considering species frequency and biomass) and hunting productivity of game species through CPUE; and (ii) hunting productivity fluctuates along the time series, i.e., if the CPUE of all game taxa and the six most hunted taxa were stable, increased or decreased through the surveyed months. Specifically, we measured two variables related to anthropogenic stressors (i.e., a proxy of human intensity and the village age) and estimated the quantitative composition of at least 17 hunted species, in addition to overall CPUE of hunting events recorded by villages in the Tapajós-Arapiuns Extractive Reserve (hereafter, RESEX-TA). Given that higher levels of hunting pressure exert pervasive effects on the killing profile in hunting events (Souza-Mazurek et al. 2000, Peres & Nascimento 2006, Constantino 2016, Nunes et al 2020), we hypothesize that villages will present different hunting profiles due to their varied intensities of hunting pressures. As a consequence, we expect that villages under higher anthropogenic influence (i.e., older and under higher human influence) will hunt more medium-sized species due to reduced availability of the preferred large-size species, whereas villages under lower anthropogenic influence will harvest more large-sized species. We also hypothesize that the interaction between our two anthropogenic stressors would best explain the overall patterns of composition and CPUE. As a result, we expect that villages under higher anthropogenic influence would exhibit greater dissimilarity in species composition. For hunting productivity, we expect that CPUE will decrease as anthropogenic influence increases, as hunters would presumably need to increase their effort to achieve the required meat return rates (Sirén et al. 2004). Finally, we expect that CPUE of medium-sized taxa (e.g., *Cuniculus paca* and *Dasypus* spp.) would increase over time, while the intensive hunting would be leading to a decrease in CPUE of the large-sized taxa (e.g. *Mazama* spp., *Dycoteles tajacu*, *Tayassu pecari* and *Tapirus terrestris*).

2. **MATERIALS AND METHODS**

2.1 **Study area**

This research was conducted within the RESEX-TA, the most populous Brazilian terrestrial extractive protected area, situated in western Pará state, Amazon (02º 20' to 03º 40' S, 55º 00' to 56º 00' W; see Fig. 1). Covering an extensive area of nearly 650,000 hectares, the RESEX-TA houses approximately 23,000 residents distributed across 78 villages located along the banks of the Tapajós and Arapiuns rivers (Silva et al. 2022). Additionally, few families reside near small streams (Oliveira et al. 2004, ICMBio 2014; see Fig. 1).

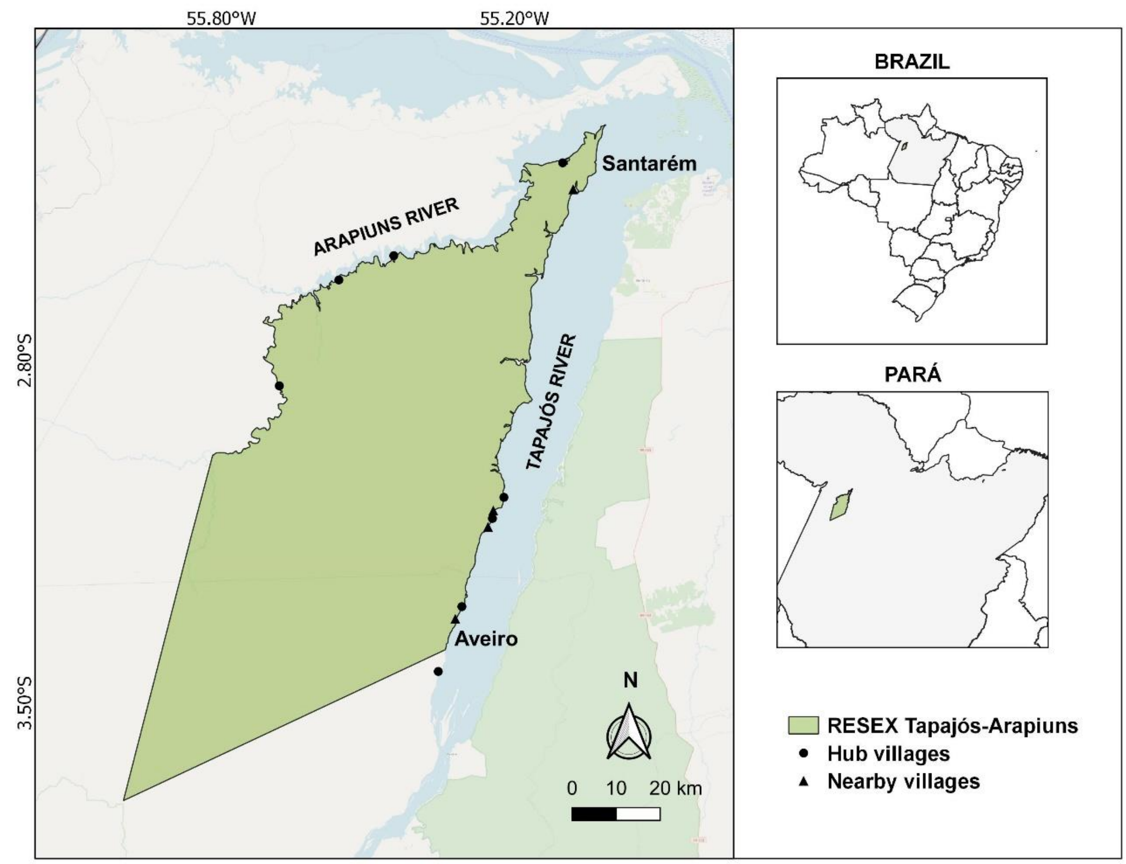


Fig.1. Location of the Tapajós-Arapiuns Extractive Reserve, situated in the Santarém and Aveiro cities (Pará), Brazil.

The dominant vegetation in the RESEX-TA is the ombrophilous dense forest, characterized by emergent trees and a rich abundance of woody plants, epiphytes, and lianas (Veloso et al. 1991, IBGE 1992), predominantly found in *terra firme* (i.e., non-flooded) forests. Other common plant typologies include savannah, secondary forests, and flooded forests (Carvalho-Junior 2008, Montag et al. 2012). The climate is classified as humid tropical (Ami), with a rainy season spanning from November to May, and a dry season from

August to October (INPE & CPTEC 2023).

All villages are established within *terra firme* forests, where residents rely directly on natural resource exploitation, including hunting, fishing, and the extraction of both timber and non-timber forest products. Additionally, some groups engage in subsistence agriculture and small-scale animal husbandry (Saúde & Alegria 2012). Notably, this protected area was the first sustainable-use protected area of this category in Brazil to create a management plan, which is a technical document that establishes the zoning and norms that guide the use of natural resources within the protected area. Hunting is permitted strictly for subsistence purposes, and its management relies on internal agreements between the villages and reserve managers.

2.2 *Participatory Biodiversity Monitoring Project (PBMP)*

In 2014, the Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio), a governmental organization in Brazil, implemented a large-scale systematic biodiversity monitoring initiative called Programa Monitora. Specifically, in the Amazon biome, Monitora developed monitoring projects adapted to local realities in a set of 16 protected areas, which started monitoring the natural resources used by local communities (Cronemberger et al. 2023). Protocols were built collaboratively between local villages, protected area managers and scientists, who suggested appropriate sampling techniques to achieve the proposed objectives (Souza et al. 2019). Within this initiative, the RESEX-TA stands out by prioritizing the monitoring of species hunted for subsistence. Such emphasis is motivated by concerns expressed by local populations in the reserve regarding the impact of subsistence hunting practices on terrestrial game fauna populations, i.e., encompassing medium to large-sized mammals and terrestrial birds. Accordingly, the monitoring efforts within the RESEX-TA focused specifically on species from these groups in addition to reptiles, totalizing 25 target species. The implementation of Monitora in the RESEX-TA followed the recommendations of the so-called *Complementary Protocol for the Effect of Subsistence Hunting on Game Species*, and was carried out via the Participatory Biodiversity Monitoring Project (PMPB), a partnership in between the Instituto de Pesquisas Ecológicas (IPÊ) and ICMBio (Ribeiro 2018). Following the protocol recommendation, RESEX-TA simultaneously adopted two monitoring sub-protocols: 1. *Occurrence and Abundance of* *forest game species*; and 2. *Hunting pressure on terrestrial game populations*. In both sub-protocols, monitoring occurred systematically by local residents that willed to participate in the monitoring (Benchimol, 2014).

Firstly, different meetings were carried out in 2014 across the RESEX-TA, aiming to identify different villages willing to participate in the monitoring program. In particular, hub villages were selected according to their interest and good receptivity. Moreover, these villages were previously recognized as exerting different intensities of hunting pressure on game fauna. For this, residents provided information on the key areas used for hunting activity, and the level of difficulty in capturing the target species (ICMBio, 2011). A total of nine villages were subsequently selected to participate in the monitoring at the RESEX-TA. However, after beginning data collections in 2014, one village left the program, and eight villages remained (Fig. 1). In addition, five nearby villages were included in the monitoring program based on their expressed interest, but for analyses were coupled together with its respective nearby hub village (Table S1).

For each hub village, a group of 3 to 4 monitors was chosen, with a minimum age requirement of 18 years, based on their expressed interest and extensive knowledge of game fauna. These monitors were then provided with training courses conducted by ICMBio and collaborating partners, which focused on instruct the monitors in (i) conduct game fauna surveys (sub-protocol 1) and (ii) select, guide and collect hunting forms related to hunting events of families from their own and nearby villages (sub-protocol 2). In this study, we use data from sub-protocol 2.

2.2.1 *Sub-protocol of hunting pressure on terrestrial game populations - hunting forms*

After having identified families that hunt within each village, monitors conducted individual conversations to both explain and invite them to collaborate with the monitoring, i.e., by providing information on their hunting events. The monitor further assisted in clarifying any doubts and ensured the monthly collection and distribution of these forms. The form, which was easy to understand and complete (see Fig. S1), needed to be filled out by any family member following an independent hunting event conducted by any person from its own family. In particular, the form contains blank fields for filling in information about the name and quantity of hunted species, the date and start and end time of each event, the number of hunters involved, the hunting success, and whether game pursuit was either intentional or opportunistic (i.e., hunter was able to kill an animal while involved in other activities, such as agriculture). Forms from nearby villages were grouped together with forms from hub villages. Participating families were identified by numbers instead of names to avoid any potential embarrassment associated with reporting hunting events. In order to maintain the confidentiality of the provided information, each family received a commitment agreement from the technical team of Monitora (Benchimol, 2014).

A total of 5,760 hunting events were recorded between 2015-2019 by 391 families (mean ± SD = 30.1 ± 21.9 families per village) belonging to the hub and nearby villages in the RESEX-TA (Table S1). The successful hunting events (i.e., events that yielded at least one animal killed) recorded in each village varied from 4 to 729 events per year (mean ± SD = 832.6 ± 519.4 events per year), totalizing 4,163 hunting events (mean ± SD = 520 ± 503 events per village; Table S2) considering the five years of monitoring. Given that the common name (i.e., vernacular name) of hunted species was mentioned in the forms in several situations, we needed to combine species belonging to the same genus or family in further analysis. For example, records that only mentioned "deer" without identifying the species, was classified as *Mazama* spp., which covers the two deer species found in the region, *M. americana* and *M. nemorivaga*, whereas tinamids were all grouped in the Tinamidae family. Therefore, based on all these events, 26 taxa (i.e., 26 families, genus or species of mammals, birds and reptiles) were recorded as being hunted in the RESEX-TA. Excluding feline kills (jaguar, puma and margay), which are not consumed for subsistence, a total of 4,137 hunting events were recorded. Out of this total, 2,845 hunting events were intentional and provided information on the duration and number of hunters involved. All hunting events were destined to subsistence.

2.3 **Composition**

Based on records from 4,137 hunting events, we estimated patterns of vertebrate community composition across hub villages per year, through Principal Coordinates Analysis (PCoA). We used the Bray–Curtis dissimilarity matrix to account for taxa identity on quantitative community composition (Legendre & Legendre 2012) based on the *pcoa* function in the *ape* package (Paradis et al. 2004) in R version 4.2.3 (R Development Core

Team 2022). For this, we considered the (i) relative frequency, and (ii) relative biomass of the main taxa hunted in the RESEX-TA (i.e., those that contributed by ≥1% of the biomass shot in at least one hub village) per village and per year. The measure of relative frequency of hunt per taxon was calculated by dividing the number of each taxon hunted by the total number of all species killed per village and per year. For example, the relative frequency of *Dasyprocta croconota* in Village Cametá in 2017 = (Number of *D. croconota* hunted) / (Total number of all species killed in Village Cametá in 2017) = 209 / 1041 = 0.20. Relative biomass was calculated as the product of the hunting frequency of each taxon multiplied by the mean body weight of the species (or genus or family) of each taxa and divided by the total biomass hunted (based on Robson & Redford 1986, Ayres & Ayres 1979, Valsecchi 2013, Cozzuol et al. 2013). We therefore used the scores obtained from Axis 1 of the PCoA, based on the relative capture frequency and relative biomass, for further analysis. The percentage of variance explained was 31.51% and 30.25%, respectively.

2.4 **Hunting productivity**

Based on records from 2845 intentional hunting events, we estimated patterns of hunting productivity. Hunting productivity is expressed by the CPUE, which was calculated as the aggregate prey biomass (kg) per hunter per hour spent hunting (kg hunter^{-1} h^{-1}) after hunt time was converted into decimal numbers (i.e., 7 h 30 min = 7.5 h CPUE). This procedure was calculated in two separate ways: (i) for the overall killed assemblage (23 taxa) per village and per year; and (ii) for the overall killed assemblage per month (42 months of data) and for the six most hunted taxa in terms of the harvested biomass (also per month): *Cuniculus paca* (37 months of data), *Dasypus* spp. (37 months), *Mazama* spp. (36 months), *Dycoteles tajacu* (34 months), *Tayassu pecari* (30 months) and *Tapirus terrestris* (16 months).

2.5 **Anthropogenic stressors**

We obtained two metrics related to anthropogenic pressure. First, we extracted a proxy of human intensity (HI) based on the distance to nearest village and its human population size (adapted from Scabin & Peres 2021), according to the equation:

$$HI = \sum \frac{S(vil)}{\sqrt{d(vil)}}$$

where S represents the human population size at each village (*vil*); and d represents the Euclidean distance from the center of each hub village to each nearby village. For this, we considered all villages distant up to 10 km from the center of the hub village. This distance was based on the evidence that hunters forage in areas located up to 10 km from their homes at the RESEX-TA (unpublished data from the participatory mapping of hunting activity, carried out in 2018). Distances were calculated in QGIS 2.18.9 (QGIS Development Team 2017) and human population size of each village was obtained from the local ICMBio database from 2022. We also calculated the age of each hub village using data from the local ICMBio database from 2023. The proxy of human intensity and hub village age (anthropogenic variables) can be assessed in Table S1.

2.6 **Data analyses**

We performed descriptive analyses to report subsistence hunting profiles across villages from 2015 to 2019. We used Generalized Linear Mixed Model (GLMMs) to relate variation in human intensity, village age and year of data collection (incorporating the term ‘village’ as random variable) to composition (considering species frequency and biomass) and CPUE, separately. We previously tested the correlation among our predictor variables (i.e., human intensity, village age and year) using a Sperman's correlation, and since none was strongly correlated at >0.50 to each other, we maintained all variables in the global model. For each response variable, we constructed different models: the null model, models presenting a single or two predictors, models with all possible additive combinations, and models including the interaction effects between the human intensity and village age. We selected the best model(s) based on the Akaike Information Criterion (AIC; Burnham & Anderson 2002) values, considering as parsimonious those presenting ΔAICc ≤ 2.0. When the null model appeared amongst the parsimonious models, we considered that no other model best explained the specific pattern than the chance. We log transformed all predictor variables to achieve normality and used the Gaussian distribution family (“identity” linkage function) in all models. The analyses were performed in the package *glmmTMB* (Brooks et al. 2022) in R 4.2.3 (R Development Core Team 2022).

To assess population trends over the months, we built state-space models that describe the stochastic and deterministic relationships between the observable and unobserved values using a Bayesian approach (Royle & Kéry 2007, Kéry 2010). We used as a response variable the monthly mean CPUE values for all game species and specifically for the six most hunted taxa, considering all villages (de Paula et al. 2022). We examined the data from all available months and applied a logarithmic transformation (log(x+1)) to the dataset. This evaluation involved estimating the exponential growth rate (*r*) of CPUE. Although the *r* value is commonly used when direct measurements of population size are accessible, it can also be applied to indirect estimates such as CPUE, as is occasionally done in studies of game harvest (Sirén et al. 2004, Parry et al. 2009). The *r* value serves as an indicator of changes in population size and can take positive, negative, or zero values to represent increasing, declining, or stable populations, respectively. The *r* is determined using the equation:

$$r = \frac{log(N_{\mathrm{t}}/N_0)}{t}$$

where N_0 represents the population size at the start of the period, while N_t represents the population size after a certain period of time (in this instance, months; Caughley & Sinclair 1994). If the average *r* value for a time series is equal or greater than 0, it may indicate stable or growing populations. Conversely, values below 0 can be interpreted as evidence of a decline in the assessed parameter (de Paula et al. 2022). Consequently, the *r* value served as an indicator of CPUE behavior over the months. Thus, if the average *r* values for CPUE across the time series are either 0 or positive for all game taxa and the six most targeted taxa, we can infer that hunting was sustainable. Conversely, declining *r* values indicates that hunting is unsustainable. We used the *R2jags* package (Su & Yajima 2012) to assess the state-space model in *JAGS* (Plummer 2015). Non-informative priors were for the CPUE, running 200 000 Markov chain Monte Carlo iterations in two independent chains with a 100 000 burn-in and a thinning factor of 0.06. Further, the Gelman-Rubin (*Rhat*) diagnosis was used to evaluate parameter convergence (Gelman & Shirley 2011). *Rhat* is a measure of convergence where values of 1.001 indicate a satisfactory model (the closer to 1, the better the model) and where 1.1 is the acceptable limit. If the 95% Bayesian confidence interval did not include 0, the *r* value was deemed substantially different from 0. The results are expressed as means ± standard deviations, unless otherwise specified.

3. **RESULTS**

3.1 **Hunting profile**

A total of 6,436 animals were killed in 4,163 hunting events performed by the eight hub villages monitored in the RESEX-TA, with an average of 1.54 (SD = 0.998) individuals killed per hunting event, totaling 65,49 kg (Table S3). Out of all the biomass harvested, 92.73% (60,73 kg, n = 5,542 individuals) correspond to mammals, 6.62% (4,33 kg, n = 559) to reptiles, and 0.65% (424.52 kg, n = 335) to birds. The taxa killed are predominantly forest-dwelling, but the hunters also pursued taxa that inhabit areas along the riverbanks and rivers, such as capybaras (*Hydrochaeris hydrochaeris*) and the yellow-spotted river turtle (*Podocnemis unifilis*). The most captured taxa in terms of biomass were *Mazama* spp. (i.e., including *M. americana* and *M. nemorivaga*) and *D. tajacu*, contributing to 34.22% of the total harvested weight (22,41 kg). Medium-sized taxa such as *D. croconota* (n = 1,549 individuals), *C. paca* (n = 1,240), *Dasypus* spp. (n = 1,051), and *Chelonoidis* spp. (n = 528) were the most frequently hunted taxa. Among birds, the Cracidae (n = 117 individuals), Psittacidae (n = 98), Tinamidae (n = 52), and Psophiidae (n = 46) were the most hunted taxa (Table S3).

Hunting events recorded in the RESEX-TA involved, on average, 1.5 hunters per event (SD = 0,695). The most used technique was active search (n=3,171 events, 55.05%), which consists of walking in search of target species and could occur with or without dogs. Most hunting events with active search involved the assistance of dogs (n=2,063, 65.06%), and resulted in the slaughter of 42.59% of the animals hunted in RESEX-TA (n=2,741 individuals), of which 38.78% (n=1,058) were agoutis. In addition, hunting activities with dogs enable the capture of more elusive and nocturnal species; for instance, six jaguars, two pumas, two giant armadillos and five tapirs were dog hunted with dogs. Active search without dogs occurred in 1,108 hunting events (34.94%). This practice is generally carried out on previously cleared trails or inside the forest where animals are detected and killed in any location by direct sighting or by looking for traces. The strategy resulted in the slaughter of the 19.95% of the animals (n=1,284 individuals), with 20.09% (n=258) encompassing armadillos, 18.07% (n=232) agoutis and 13.47% (n=173) pacas.

The second most frequently used technique was "mutá" (n=1,291 events, 22.41%), in which the hunter waits for the prey in a hammock or raised platforms, in areas frequently used by target animals (e.g., near a fruit tree of after deploying a bait). This practice resulted in the slaughter of 19.48% of the animals (n=1,254 individuals), in which 37.24% (n=467) were lowland pacas and 26.56% (n=333) were armadillos. In addition, hunting with canoe occurred in 15.59% of the events (n=898 events) and was responsible for the slaughter of 11.54% of the animals (n=743 individuals), including 34.32% (n=255 individuals) armadillos and 27.46% (n=204) pacas. Finally, hunting with a trap was the least used, corresponding to 0.49% of the events (n=28 events) and 0.59% of the slaughtered animals (n=38 individuals).

3.2 **Harvested fauna composition**

Based on the PCoA ordination, we noticed that hunting pattern diverged across focal villages (Fig. 2). In particular, most villages exhibited similar profiles over the years, but some showed dissimilarities, such as *Anã*, *São Tomé* and *Escrivão*. In *Anã* and *São Tomé*, *Dasypus* spp. was the most frequently hunted taxon, which is considerably distinguished from other villages in terms of relative frequency (Fig. 3A). In addition, *C. paca* and *T. pecari* comprised the most hunted taxa in slaughtered weight in *Escrivão*, turning patterns of relative biomass consistently different to the other focal villages (Fig. 3B).

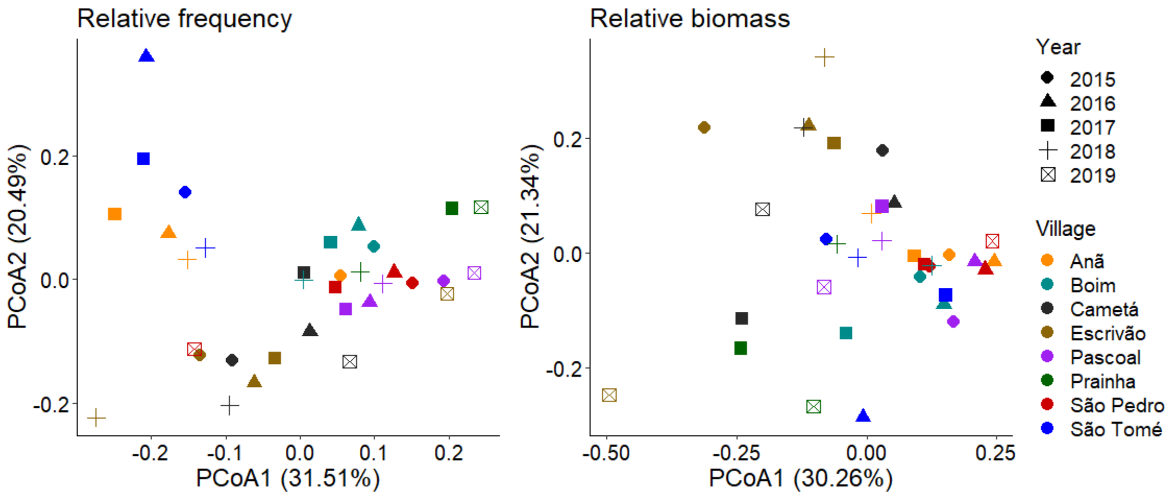


Fig. 2. Ordination based on Principal Coordinates Analysis (PCoA) using the Bray–Curtis dissimilarity matrix of the main game taxa harvested in eight hub villages of the Tapajós-Arapiuns Extractive Reserve along the 5-years of monitoring. Panels show ordination plots for relative frequency and biomass. The percentage of variance explained is reported (in brackets) for each PCoA axis.

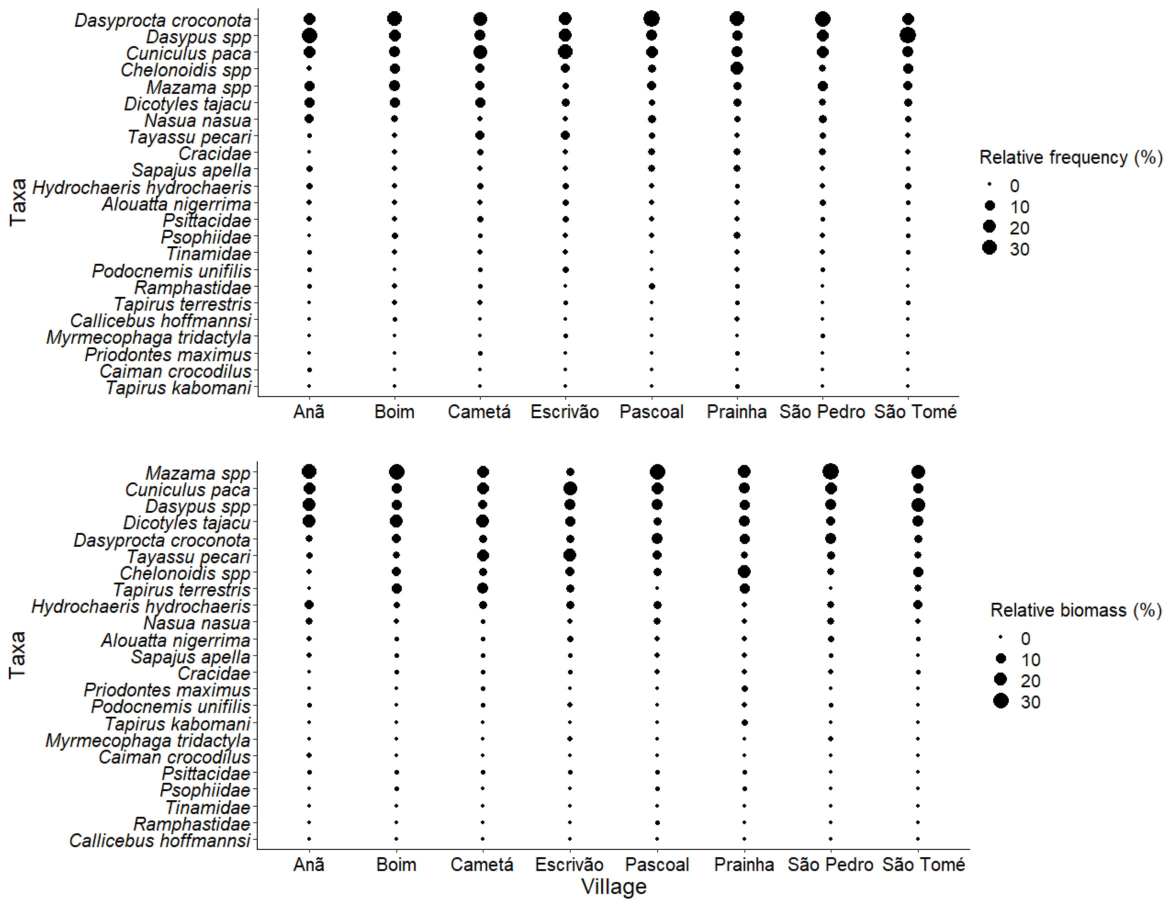


Fig. 3. Relative frequency (A) and relative biomass (B) of the game taxa hunted for consumption (expressed as percentage (%)) in eight hub villages of the Tapajós-Arapiuns Extractive Reserve, between 2015 and 2019. Taxa are ordered top to bottom by their overall numerical offtakes.

The mean CPUE of all game taxa was 2.6 (± 0.52) kg hunter^{-1} h^{-1}, with hunting events varying from 2.0 to 3.50 kg hunter^{-1} h^{-1} (Table S4). Individually, the taxa with the highest mean CPUE were *T. terrestris* with 20.2 kg hunter^{-1} h^{-1} (±14.74) and *Mazama* spp. with 6.2 kg hunter^{-1} h^{-1} (±1.46), whereas *Dasypus* spp. exhibited the lowest CPUE with 2.3 kg hunter^{-1} h^{-1} (±0.70) for (Table S4).

3.3 **Influence of anthropogenic variables on game harvest and temporal trends in hunting productivity**

The GLMMs revealed that patterns of composition (considering relative frequency and relative biomass) and CPUE of game fauna were neither influenced by human intensity nor village age, and did not change along the years. Indeed, the null model appeared among the parsimonious in the group analyses in all cases (Table 1).

Table 1. Model selection table based on a candidate set of the 'best' models (ΔAIC $\leq$ 2.00) predicting patterns of composition (considering relative frequency and relative biomass) and CPUE for all game taxa in eight hub villages at the Tapajós-Arapiuns Extractive Reserve. Models are ranked by the AICc and ΔAICc values.

Model	**Df**	**logLik**	**AICc**	**ΔAICc**	**weight**
Composition (relative frequency)					
~ null	3	21.93	-37.07	0.00	0.28
~ human intensity	4	22.90	-36.42	0.64	0.20
~ village age	4	22.65	-35.92	1.15	0.16
Composition (relative biomass)					
~ year	4	19.24	-29.09	0.00	0.34
~ null	3	17.40	-28.01	1.09	0.20
CPUE					
~ null	3	-33.84	74.72	0.00	0.48

The parameter convergence was satisfactory for all state-space models (Table S5) and the exponential growth rate *r* was not different from 0 for the mean CPUE considering overall (mean ± SD = 2.41 ± 1.86 kg hunter^{-1} h^{-1}), and individual taxa of *C. paca* (2.45 ± 2.94 kg hunter^{-1} h^{-1}), *Dasypus* spp. (1.84 ± 0.61 kg hunter^{-1} h^{-1}), *Mazama* spp. (6.11± 8.91 kg hunter^{-1} h^{-1}), *D. tajacu* (3.30 ± 0.79 kg hunter^{-1} h^{-1}), *T. pecari* (3.41± 1.37 kg hunter^{-1} h^{-1}) and *T. terrestris* (18.92 ± 12.64 kg hunter^{-1} h^{-1}). These temporal trends indicate that CPUE remained stable for all game taxa and for the six most hunted taxa considering all villages within the RESEX-TA (Fig. 4).

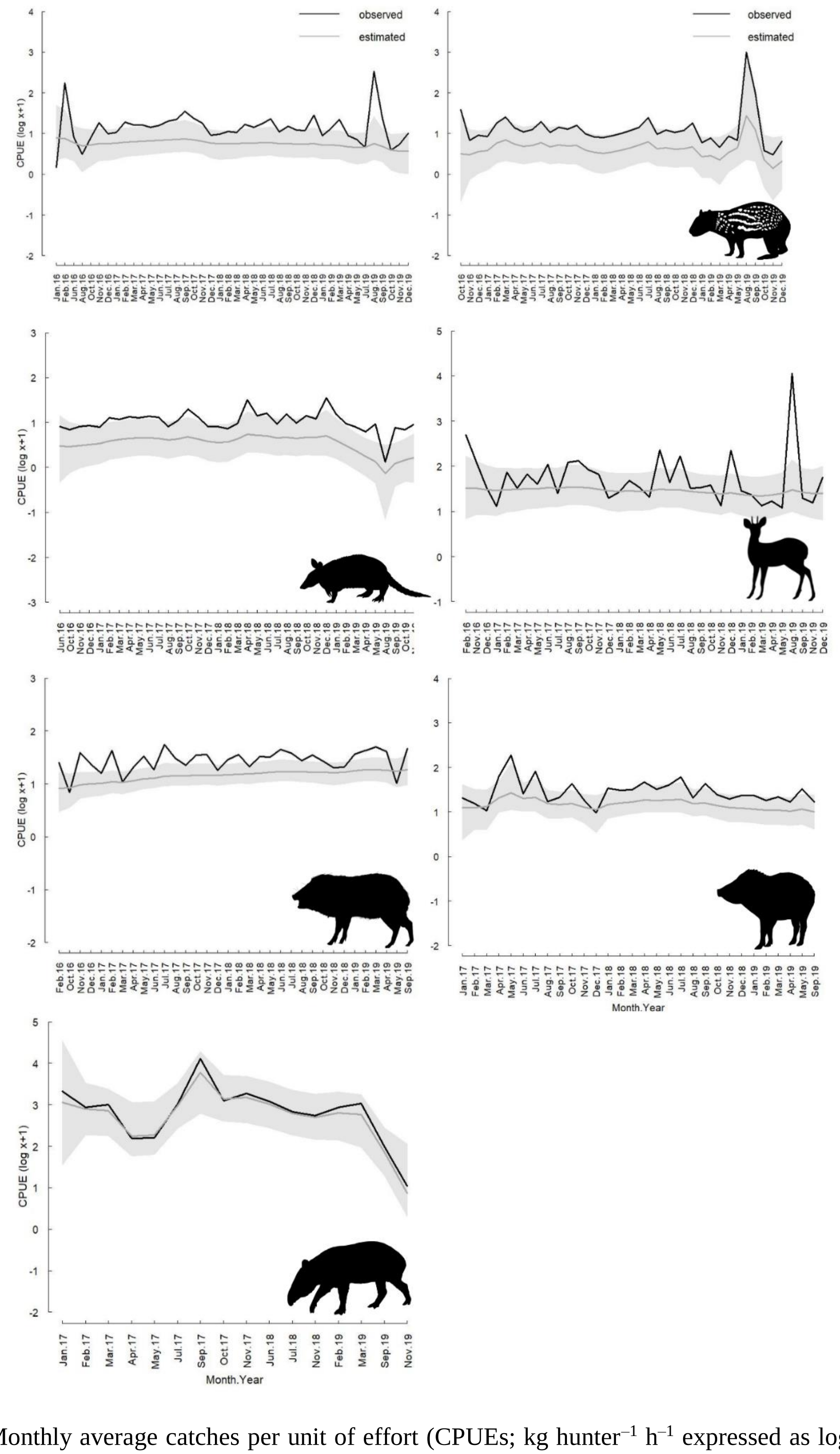


Fig. 4. Monthly average catches per unit of effort (CPUEs; kg hunter^{-1} h^{-1} expressed as log (x + 1)) in eight hub villages of the Tapajós-Arapiuns Extractive Reserve over 42 months (January 2016 - December 2019) for all game taxa (n = 42 months), *Cuniculus paca* (n = 37), *Dasypus* spp. (n = 37 months), *Mazama* spp. (n = 36), *Dycoteles tajacu* (n = 34), *Tayassu pecari* (n = 30) and *Tapirus terrestris* (n = 16). Black and grey lines represent, respectively, the observed and estimated offtakes.

4. **DISCUSSION**

This comprises the first study assessing the hunting profiles and patterns of hunting productivity through a time series based on the Complementary Protocol for the Effect of Subsistence Hunting on game species from the Brazilian Biodiversity Monitoring Program, the Monitora. As expected, medium to large-sized mammals comprised the main targets of hunters in the RESEX-TA, yet hub villages exhibited different hunting profiles, thereby reflecting different hunting strategies and species abundance *in situ*. Contrary to our predictions, anthropogenic variables and the monitored year failed in significantly explain the prey profile (considering taxa frequency and biomass) and the hunting productivity across the eight hub villages. In addition, our results revealed that the CPUE remained stable during the monitoring months. Our findings suggest that hunting is likely sustainable for most species, with the exception of those large and threatened species like tapirs and white-lipped peccaries. We therefore discuss the implications of our findings to support local management actions towards the conservation of wild game fauna. In addition, we encourage the use of the RESEX-TA protocol for monitoring subsistence hunting in other Amazonian sustainable-use protected areas.

4.1 **Hunting profile at the RESEX-TA**

The overall hunting patterns in the RESEX-TA are similar to those observed in other *terra firme* environments in the Amazon, given that medium and large mammal species continue to be the most hunted group by rural and indigenous populations across the Amazon basin (Bodmer & Lozano 2001, Constantino et al. 2008, Valsecchi & Amaral 2009). In particular, the ungulates brocket deer (*Mazama* spp.) and collared peccary (*D. tajacu*) were the taxa that most contributed in terms of weight to the consumption by hunters in the RESEX-TA. In fact, these animals comprise the preferred items by local hunters (YMS Reis, personal communication) and other Amazonian hunters (Redford 1992, Bodmer 1995, Stafford et al. 2017) due to the appreciated meat flavor and their large size. Although ungulates substantially contribute to the harvested biomass in Amazonian forests, other medium-sized taxa were more frequently hunted and also most abundant, including agouti (*D. croconota*), lowland paca (*C. paca*), and armadillo (*Dasypus* spp.) (Calouro 1995, Hill & Padwe 2000, Peres 2000). In addition, the slaughter profile within the RESEX includes threatened species, such as tapirs (*T. terrestris*) and white-lipped peccaries (*T. pecari*). However, it is important to highlight that the majority of captures of tapirs occurred in a single village and most captures of white-lipped peccaries took place in just two villages. Additionally, when comparing to two other sustainable-use protected areas in the Amazon (de Paula et al. 2022), we observe that these areas killed almost the triple number of tapirs and more than triple the number of peccaries in less than half of the recorded hunting events, compared to our study area. In heavily hunted areas, these species usually exhibit low population densities, become rare, or may even face local extinction, resulting in their exclusion from the hunting profiles (Peres & Palacios 2007, Parry et al. 2009). These results are an indicator that defaunation (i.e., the depletion of animal species or population, see Dirzo et al. 2014) may be occurring and therefore hunting practices of these species in the Tapajós-Arapiuns protected area seem to be unsustainable (see Chapter 02).

Reptiles were the second group with the highest number of individual kills and hunted weight. Their importance is related to the hunting of *Chelonoidis* spp., which accounted for 8.20% and 6.28% of the overall hunted taxa and harvest biomass, respectively. Despite local residents expressed a lack of preference for consuming this game meat, the pursuit of tortoises can serve as an alternative food source when hunters are unable to capture an animal of greater desirability. Indeed, as these animals do not require the use of capture instruments neither physical effort due to its slow behavior, an encounter after an unsuccessful hunting event is advantageous. Furthermore, chelonians, including terrestrial species, are frequently targeted by Amazonian hunters, as observed by Pezzuti et al. (2010) and Morcatty & Valsecchi (2015).

As in other protected areas (Valsecchi & Amaral 2009, Constantino 2015), most foraging events were performed by a single hunter, a strategy to prevent scaring the animals away. In addition, hunters were usually armed with a shotgun and applied one of the most common hunting techniques in the Amazon, including active search (with or without dogs), "mutá", hunting with a canoe or with a trap. The choice is related to their own knowledge of animal behavior and distribution, which is fundamental for detecting the species of interest (Reis et al. 2018), as well as the cost-benefit analysis of each approach (Braga-Pereira et al. 2020). Nevertheless, active search with dogs was the most common technique among local hunters across all focal villages. Well-trained dogs can significantly enhance the efficiency of hunting, resulting in an equivalent amount of meat with the use of less effort (Alves et al. 2018, Constantino et al. 2019, Santos et al. 2022). For instance, hunting with dogs led to encountering eight times more agoutis compared to hunting without the aid of dogs in Nicaragua (Koster 2008). However, hunting with dogs can exert negative impacts on wildlife, by scaring animals away from their natural habitats, facilitating the simultaneous killing of individuals from species that form groups and/or causing diurnal predation on nocturnal species, including the endangered ones (Ramos et al. 2008; Vieira et al. 2015). In fact, dogs were able to capture rare and threatened species such as jaguars (*Panthera onca*), pumas (*Puma concolor*), giant armadillos (*Priodontes maximus*), and tapirs (*T. terrestris*) during daylight hours.

Conflicts related to use of dogs in hunting activities at the RESEX-TA were reported in 2022, during a participatory mapping event. According to village members, during hunts, dogs not only chase animals away from hunting areas but also do not respect the limits of hunting territories resulting in chasing preys in hunting territories of neighboring families or villages, and consequently leading to disagreements among residents. Furthermore, the number of endangered animals killed during the five years of monitoring calls attentions for the negative impacts of use of dogs in hunting events. Indeed, Carvalho & Pezzuti (2010) recorded that most jaguars and puma killed in the RESEX-TA was a result of occasional encounters in the forest, and a large portion of these encounters were caused by the presence of domestic dogs. Given the negative consequences on wildlife, we recommend that villages discuss and potentially ban the use of dogs in hunting events.

4.2 **Harvested fauna composition**

Over the monitoring period, overall composition patterns of killed animals were not consistently dissimilar, demonstrating that hunters across the RESEX-TA are selective in terms of prey species pursued. Despite this, we observed some hunting specificities in a set of villages, likely reflecting the hunting strategies and species abundance *in situ*. In particular, both *Anã* and *São Tomé* villages killed a greater number of individuals of armadillo (*Dasypus* spp.) than other focal villages, which is potentially explained by the higher nocturnal hunting activities of both villages (Macdough & Loughry 2003). In particular, armadillo meat comprises the most preferred meat by local people in *São Tomé*, which may motivate hunters to focus more on this taxon during hunting activities (YMS Reis, personal communication). Likewise, a single village (*Escrivão*) presented higher killed biomass comprised by paca (*C. paca*) and white-lipped peccary (*T. pecari*) compared to other focal villages. In this village, hunters forage both during day and night, leading the capture of animals exhibiting different habits. Furthermore, the capture of white-lipped peccaries, greater than in other villages, underscores the significance of the forested area surrounding this village in maintaining their populations.

Contrary to our expectations, patterns of community composition of killed animals and CPUE were unaffected by anthropogenic pressure through years. Likewise, human pressure failed in explaining the density of medium and large-sized groups, as well as the four taxa most frequently recorded in transects nearby the same focal villages (see chapter 02). Furthermore, temporal trends of densities were also stable, indicating an equilibrium in the populations of common game species. For instance, medium- and large-sized species, including lowland paca (*C. paca*), brocket deer (*Mazama* spp.) and collared peccary (*D. tajacu*), which are widely the most preferable prey items among hunters in Amazonian forest sites (Mesquita & Barreto 2015), have been frequently recorded in field surveys (Chapter 02) and also comprised the most hunted species. This demonstrates that these species are more resilient to hunting pressure and reinforces the concept that undisturbed areas surrounding our study villages can act as a "source" of animals, facilitating the dispersal of individuals to hunting "sinks" areas (Novaro et al. 2000), and enabling the sustainability of hunting these species at the RESEX-TA.

Our results on the temporal trends based on CPUE indicate that hunters maintained a stable kill rate during the monitoring months in the RESEX-TA, for all game taxa and for each of the six most harvested taxa. However, hunting productivity showed variation over the months. At the taxon level, monthly CPUE values were less variable for armadillos (*Dasypus* spp.), probably because of the high demand for this taxon as an important source of game meat locally; while CPUE was more variable for tapir (*T. terrestris*), possibly because of the current semi-defaunation scenario in the RESEX, resulting in a lower probability of encounter during hunts. The harvested of paca (*C. paca*), armadillo (*Dasypus* spp.) and brocket deer (*Mazama* spp.) were well distributed throughout the monitoring months, further supporting the idea that these taxa are the most targeted by local hunters.

Variation in CPUE may be related to several factors including techniques and transport used (Santos et al. 2022), seasonal changes in rainfall patterns in the Amazon basin (Endo et al. 2016), period (Riddell et al. 2022) and even hunters' experience. However, CPUE spikes are also related to specific situations during hunting events, such as rare encounters with multiple animals in a single event. For example, we noticed a peak of CPUE in August 2019 for all game taxa, paca (*C. paca*) and brocket deer (*Mazama* spp.), resulted from five hunting events, where one deer (*Mazama* spp.), one tortoise (*Chelonoidis* spp.), one lowland paca (*C. paca*), one tapir (*T. terrestris*) and one white-lipped peccary (*T. pecari*) were killed.

Our study provides evidence of the sustainability of hunting in one of the most populous sustainable-use protected areas in the Amazon. In addition to the maintenance of the hunting profile within villages and stable hunting productivity during the monitoring months, *in situ* surveys also indicate that populations of common game species remain stable over the years (Chapter 02). Indeed, both changes in prey profile and CPUE were identified as the most accurate capture metrics and are therefore considered sustainability indicators (Kumpel et al. 2010). In addition, as these indices are easily obtainable through community-based monitoring and interviews with hunters, they have been proven to be accurate in obtaining information about the status and trends of species/resources. Thus, both indices significantly contribute to the effective community management of hunting (Rist et al. 2010, Kumpel et al. 2010), especially in sustainable-use protected areas. In a similar vein, de Paula et al. (2022) assessed quantitative prey species profiles and CPUE over 63 consecutive months in two sustainable-use protected areas within the eastern Brazilian Amazon and provided evidence that hunting was sustainable. Furthermore, a recent study demonstrated that sustainable-use forest reserves can mitigate species declines due to overhunting across nine Amazonian protected areas (Sampaio et al. 2023).

5. **CONCLUSION**

Although our data is limited to short-term monitoring period (i.e., five years), they suggest that subsistence hunting is not depleting the entire game species assemblage. Our data indicate that hunting may be sustainable for those more common and resilient species such as pacas, armadillos and brocket deers, while unsustainable hunting is likely occurring for large-bodied and threatened species (such as tapirs and white-lipped peccaries) across distinct villages in the RESEX-TA. Given the long-life spans and reproductive patterns of vertebrate species, long-term monitoring of the game fauna and harvest remains crucial to assess hunting sustainability across generations and subsidize sound management policies. We therefore recommend the continuity of the monitoring program over time, which includes maintaining sub-protocol 1 (i.e., game fauna surveys) and re-establish sub-protocol 2 (i.e., hunting events surveys) in the same villages. We also recommend the expansion of the monitoring program to other villages in the reserve, as this will enhance the sampling effort and therefore improve the inference power of data analyses and results.

In relation to management actions, we particularly recommend implementing rules to restrict hunting with dogs (e.g., establish penalties for this practice) to avoid the capture of endangered species, which regrettably occurs in the RESEX-TA. However, it is important to formalize management actions to avoid subjecting the hunting practice to arbitrary local inspection (Antunes et al. 2019) and, finally, ensure adequate supervision for compliance with the proposed actions. Hunting practices in the RESEX-TA is restrict to subsistence purposes for village members, with no evidence of commercial hunting to supply urban centers (YMS Reis, personal communication). Nevertheless, local trade in bushmeat is a frequent activity since not everyone hunts, and those who hunt have financial costs (e.g., gun bullet and fuel) and can be reimbursed by local sales (Oliveira et al. 2005, Maduro 2022). Game meat can also be exchanged for other products (e.g., cassava flour, rice, and beans), and used in traditional medicine (e.g., fat, hoof, foot, tooth, and tail) (Reis 2014), therefore demonstrating the still cultural connection with the original pre-Columbian (*Tupiniquim*) peoples of the region (Oliveira et al. 2005). We therefore highlight the socioeconomic value of wild meat extraction to ensure local food security and preserve customs and livelihoods of local villages but emphasize that commercial hunting should be continuously banned to avoid overhunting and therefore unsustainable hunting in the RESEX-TA.

At the wider-level, we consider that the established hunting monitoring protocol in the RESEX-TA can be used as a robust model for other Amazonian sustainable-use protected areas. In fact, 32.5% of established protected areas in the Amazonia are comprised by sustainable-use reserves (Pereira & Ferreira 2020). As overhunting can exert a strong negative impact on biodiversity maintenance (Benítez-López et al. 2019), the establishment of monitoring programs towards hunting profiles should be prioritized by governmental sectors. Nevertheless, we encourage that local people become actively engaged in the monitoring program, while also ensuring well-structured planning for its long-term execution to prevent interruptions. This holds particular significance, given that the discontinuation of numerous monitoring initiatives undermines the acquisition of enduring data concerning the monitored populations (Reis & Benchimol 2023).

Author Contribution Statement

Yasmin Maria Sampaio dos Reis conducted the data entry and analyzed the data; Yasmin Maria Sampaio dos Reis and Maíra Benchimol led the drafting of the manuscript.

Declaration of competing/Conflicting Interests

The authors declare no competing interests in the conduct of this research.

Financial Disclosures:

This study was supported by the Coordination for the Improvement of Higher Education Personnel (CAPES) - Brazil (CAPES - Financing code 001) and CNPq (productivity grant 304189/2022-7 to M. Benchimol).

Research Ethics Approval

The study complies with the current laws of Brazil, with the use of database from the biodiversity monitoring program (Monitora) authorized of the Biodiversity Authorization and Information System (SISBIO) of the Instituto Chico Mendes de Conservação da Biodiversidade (ICMBio).

Data Availability

Access and utilization of data and information from the Monitora Program must adhere to ICMBio's Data and Information Policy on Biodiversity and comply with Law No. 12,527 of November 18, 2011 - the Access to Information Law, and its subsequent amendments. Data requests should be directed to the Biodiversity Monitoring Coordination (COMOB).

Acknowledgements

We are grateful to the community members and professionals involved in the Monitora Program and the local management of RESEX-TA. We are also grateful to the post-graduate program in Ecology and Biodiversity Conservation of the Universidade Estadual de Santa Cruz – UESC for providing the courses which contributed to the development of the manuscript. Finally, we also thank Hani El Bizri, Milton de Paula, Elildo Carvalho who shared analytical knowledge.

6. **REFERENCES**

Alves, RRN., WMS. Souto, H. Fernandes-Ferreira, et al. 2018. The importance of hunting in human societies. In Ethnozoology (eds. Alves, R.R.N. and U.P. Albuquerque). Pp. 95-118. Academic Press.

Antunes, AP., GH. Rebêlo, JCB. Pezzuti, et al. 2019. A conspiracy of silence: Subsistence hunting rights in the Brazilian Amazon. *Land use policy* 84: 1-11.

Ayres, JM. and C. Ayres. 1979. Aspectos da caça no alto rio Aripuanã. *Acta Amazonica* 9 (2): 287-298.

Beirne, C., AC. Meier, AE. Mbele, et al. 2019. Participatory monitoring reveals village-centered gradients of mammalian defaunation in central Africa. *Biological conservation* 233: 228-238.

Benchimol, M. 2014. *Protocolo Complementar do Efeito da Caça de Subsistência sobre Espécies Cinegéticas na RESEX Tapajós-Arapiuns*. Programa de Monitoramento da Biodiversidade com relevância para o Clima. Brasília: ICMBio.

Benítez-López, A., R. Alkemade, AM. Schipper, et al. 2017. The impact of hunting on tropical mammal and bird populations. *Science* 356: 180-183.

Benítez-López, A., L. Santini, AM. Schipper, et al. 2019. Intact but empty forests? Patterns of hunting-induced mammal defaunation in the tropics. *PLoS biology* 17(5): e3000247.

Braga-Pereira, F., JA. Bogoni, and RR. Alves. 2020. From spears to automatic rifles: The shift in hunting techniques as a mammal depletion driver during the Angolan civil war. *Biological Conservation*. 249:108744.

Brooks M., B. Bolker, K. Kristensen, et al. 2022. Package ‘glmmTMB’ [www document]. Available at: http://cran.uni-muenster.de/web/packages/glmmTMB/glmmTMB.pdf.

Bodmer, RE. 1995. Managing Amazonian Wildlife: Biological Correlates of Game Choice by Detribalized Hunters. *Ecological Aplications* 15(4): 872-877.

Bodmer, RE. and EP. Lozano. 2001. Rural development and sustainable wildlife use in Peru. *Conservation Biology* 15: 1163–1170.

Bodmer, RE., JF. Eisenberg, and KH. Redford. 1997. Hunting and the likelihood of extinction of Amazonian mammals. *Conservation biology* 11(2): 460-466.

Burnham, KP. and DR. Anderson. 2002. Information and likelihood theory: a basis for model selection and inference. *Model selection and multimodel inference: a practical information-theoretic approach*, pp 49-97.

Calouro, AM. 1995. Caça de Subsistência: Sustentabilidade e Padões de Uso entre Seringueiros Ribeirinhos e Não-Ribeirinhos do Estado do Acre. MSc dissertation. Universidade de Brasília/Brasília/Brazil.

Cardillo, M, GM. Mace, KE. Jones, et al. 2005. Multiple causes of high extinction risk in large mammal species. *Science* 309: 1239–1241.

Carvalho, E., J. Pezzuti. 2010. Hunting of jaguars and pumas in the Tapajós–Arapiuns Extractive Reserve, Brazilian Amazonia. *Oryx* 44(4): 610-612.

Caughley, G. and AR. Sinclair. 1994. *Wildlife Ecology and Management*. Cambridge, UK: Blackwell Science.

Constantino, PL., LB. Fortini, FRS. Kaxinawa, et al. 2008. Indigenous collaborative research for wildlife management in Amazonia: The case of the Kaxinawá, Acre, Brazil. *Biological Conservation* 141: 2718-2719.

Constantino, PAL., HAS. Carlos, EE. Ramalho, et al. 2012. Empowering local people through community-based resource monitoring: a comparison of Brazil and Namibia. *Ecology and Society* 17(4).

Constantino, PDAL. 2015. Dynamics of hunting territories and prey distribution in Amazonian Indigenous Lands. *Applied Geography* 56: 222-231.

Constantino, PAL. 2016. Deforestation and hunting effects on wildlife across Amazonian indigenous lands. *Ecology and Society* 21(2): 3.

Constantino, PDAL. 2019. Subsistence hunting with mixed-breed dogs reduces hunting pressure on sensitive Amazonian game species in protected areas. *Environmental conservation* 46(1): 92-98.

Cozzuol, MA., CL. Clozato, EC. Holanda, et al. 2013. A new species of tapir from the Amazon. *Journal of Mammalogy* 94(6), 1331-1345.

Cronemberger, C., KT. Ribeiro, RK. Acosta, et al. 2023. Social Participation in the Brazilian National Biodiversity Monitoring Program Leads to Multiple Socioenvironmental Outcomes. *Citizen Science: Theory and Practice* 8(1): 32, 1–15. 2023.

de Paula, MJ., EA. Carvalho, CKM. Lopes, et al. 2022. Hunting sustainability within two eastern Amazon Extractive Reserves. Environmental Conservation 49(2):90-98.

Davidson, AD., MJ. Hamilton, AG. Boyer, et al. 2009. Multiple ecological pathways to extinction in mammals. *Proceedings of the National Academy of Sciences* 106(26): 10702-10705.

Dirzo, R., HS. Young, M. Galetti, et al. 2014. Defaunation in the Anthropocene. *Science* 345(6195): 401-406.

El Bizri, HR., JE. Fa, LP. Lemos, et al. 2020. Involving local communities for effective citizen science: Determining game species' reproductive status to assess hunting effects in tropical forests. *Journal of Applied Ecology*. 58: 224-235.

Endo, W., CA. Peres, and T. Haugaasen. 2016. Flood pulse dynamics affects exploitation of both aquatic and terrestrial prey by Amazonian floodplain settlements. *Biological Conservation* 201: 129–136.

Fa, JE., SM. Funk, and R. Nasi. 2022. *Hunting wildlife in the tropics and subtropics*. Cambridge University Press.

Gelman A. and K. Shirley. 2011. Inference from simulations and monitoring convergence. In: Handbook of Markov Chain Monte Carlo (eds. Brooks, S. et al.). Pp. 163–174. Boca Raton, FL, USA: Chapman & Hall/CRC Press.

Hill, K. and J. Padwe. 2000. Sustainability of Aché hunting in the Mbaracayu reserve, Paraguay. *Hunting for sustainability in tropical forests* 79-105.

Instituto Chico Mendes de Conservação da Biodiversidade. 2011. *Mapeamento participativo do uso dos recursos naturais na RESEX Tapajós-Arapiuns*. Conservação Internacional-CI, Associações da Reserva Extrativista Tapajós-Arapiuns-Tapajoara. Brazil: ICMBio.

Instituto Chico Mendes de Conservação da Biodiversidade. Plano de Manejo da Reserva Extrativista Tapajós-Arapiuns. 2014. Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio. Brasília, Brazil: ICMBio.

Ingram, DJ., L. Coad, EJ. Milner-Gulland, et al. 2021. Wild meat is still on the menu: Progress in wild meat research, policy, and practice from 2002 to 2020. *Annual Review of Environment and Resources* 46: 221-254.

Instituto Nacional de Pesquisas Espaciais, Centro de Previsão de Tempo e Estudos Climáticos. 2023. Precipitação diária 2023 [www document]. Available at: http://clima1.cptec.inpe.br/estacaochuvosa/pt.

Iwamura, T., EF. Lambin, KM. Silvius, et al. 2014. Agent-based modeling of hunting and subsistence agriculture on indigenous lands: Understanding interactions between social and ecological systems. *Environmental Modelling & Software* 58: 109-127.

Jerozolimski, A. and CA. Peres. 2003. Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. *Biological Conservation*, *111*(3), 415-425.

Kéry, M. 2010. Introduction to WinBUGS for Ecologists: A Bayesian Approach to Regression, ANOVA, Mixed Models and Related Analyses. Amsterdan, The Netherlands: Elsevier.

Koster, J. 2009. Hunting dogs in the lowland Neotropics. *Journal of Anthropological Research* 65(4): 575-610.

Koster, JM. 2008. Hunting with dogs in Nicaragua: an optimal foraging approach. *Current Anthropology* 49(5): 935-944.

Kümpel, NF., EJ. Milner-Gulland, GUY. Cowlishaw, et al. 2010. Assessing sustainability at multiple scales in a rotational bushmeat hunting system. *Conservation Biology* 24(3): 861-871.

Legendre, P. and L. Legendre. 2012. *Numerical Ecology*. 3rd English ed. Development in Environmental Modelling Vol. 24.

Lima, D. and J. Pozzobon. 2005. Amazônia socioambiental: sustentabilidade ecológica e diversidade social. *Estudos avançados*. 19, 45–76.

Luzar, JB., KM. Silvius, H. Overman, et al. 2011. Large-scale environmental monitoring by indigenous peoples. *BioScience* 61: 771-781.

Mayor, P., H. El Bizri, RE. Bodmer, et al. 2017. Assessment of mammal reproduction for hunting sustainability through community-based sampling of species in the wild. *Conservation Biology* 31(4): 912-923.

Montag FFA., BS. Prudente, CP. Ferreira, et al. 2012. Ictiofauna na Reserva Extrativista Tapajós-Arapiuns, Estado do Pará, Brasil. Pará. Brazil: UFPA.

Morcatty, TQ. And J. Valsecchi. 2015. Social, biological, and environmental drivers of the hunting and trade of the endangered yellow-footed tortoise in the Amazon. *Ecology and Society* 20(3).

Macdough, CM., WJ. Loughry. 2003. Armadillos (Dasypodidae). In: Grzimek's Animal Life Encyclopedia. V. 13. (Ed. Hutchins, M.). Famington Hills. Gale Group.

Mesquita, GP. and LN. Barreto. 2015. Evaluation of mammals hunting in indigenous and rural localities in Eastern Brazilian Amazon. *Ethnobiology and Conservation* 4:2.

Novaro AJ., KH. Redford, RE. Bodmer. 2000. Effect of hunting in source–sink systems in the Neotropics. *Conservation Biology* 14: 713–721.

Nunes, AV., LGR. Oliveira-Santos, BA. Santos, et al. 2020. Socioeconomic Drivers of Hunting Efficiency and Use of Space By Traditional Amazonians. *Human Ecology* 48, 307-315.

Paradis, E., J. Claude, and K. Strimmer. 2004. APE: an R package for analyses of phylogenetics and evolution. *Bioinformatics* 20:289–290.

Parry, L., J. Barlow, and CA. Peres. 2009. Allocation of hunting effort by Amazonian smallholders: implications for conserving wildlife in mixed-use landscape. *Biological Conservation* 142:1777-1786.

Peres, CA. 2000. Effects of subsistence hunting on vertebrate community structure in Amazonian forests. *Conservation Biology* 14: 240-253.

Peres, CA. and HS. Nascimento. 2006. Impact of game hunting by the Kayapó of south-eastern Amazonia: implications for wildlife conservation in tropical forest indigenous reserves. *Biodiversity & Conservation* 15(8): 2627-2653.

Peres, CA. and E. Palacios. 2007. Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: Implications for animal-mediated seed dispersal. *Biotropica* 39(3): 304-315.

Pezzuti, JC., JP. Lima, DF. da Silva, et al. 2010. Uses and taboos of turtles and tortoises along Rio Negro, Amazon Basin. *Journal of Ethnobiology* 30(1), 153-168.

Plummer. M. 2015 JAGS Version 4.0.0 user manual 0–41 [www document]. Available at: https://www.uvm.edu/~bbeckage/Teaching/PBIO_294/Manuals/manual.jags.pdf.

Ohl-Schacherer, J., GH. Shepard Jr., H. Kaplan, et al. 2007. The sustainability of subsistence hunting by Matsigenka native communities in Manu National Park, Peru. *Conservation Biology* 21(5): 1174-1185.

Oliveira, ACM., OC. Junior, R. Chaves. 2005. Gestão participativa e a atividade de caça na Reserva Extrativista do Tapajós – Arapiuns, Santarém, PA. *Raízes* 23: 42–51.

Oliveira, MA. and AM. Calouro. 2020. Medium-sized and large mammals of the Cazumbá-Iracema Extractivist Reserve, Acre, Brazil. *Check List* 16(1): 127-136.

Pereira, JLG. and LV. Ferreira. 2020. Suscetibilidade das áreas protegidas da Amazônia Legal ao desflorestamento. *Boletim do Museu Paraense Emílio Goeldi-Ciências Naturais* 15(2): 445-463.

Ramos, RM., NS. Carmo, and JCB. Pezzuti. 2008. Caça e uso da fauna, p. 224-232. In: Atlas socioambiental: municípios de Tomé-Açu, Aurora do Pará, Ipixuna do Pará, Paragominas e Ulianópolis (org. Monteiro, MA.). 464p. NAEA.

Redford, KH. 1992. The Empty Forest. *BioScience* 42(6): 412-422.

Reis, YMS. 2014. Caracterização da Atividade de Caça desenvolvida por Comunidades da Reserva Extrativista Tapajós-Arapiuns no estado do Pará 2014. Final Project of the Course (Bachelor's Degree in Biological Sciences). Universidade Federal do Oeste do Pará, Santarém/PA/Brazil.

Reis, YS., J. Valsecchi, and H. Queiroz. 2018. Caracterização do uso da fauna silvestre para subsistência em uma unidade de conservação no Oeste do Pará. *Biodiversidade Brasileira* 8(2), 187-202.

Reis, YMS., M. Benchimol. 2023. Effectiveness of community-based monitoring projects of terrestrial game fauna in the tropics: a global review. *Perspectives in Ecology and Conservation* 21: 172–179.

Ribeiro, KT. 2018. *Estratégia do Programa Nacional de Monitoramento da Biodiversidade* – Programa Monitora: estrutura, articulações, perspectivas. Brasília. Brazil: ICMBio.

Riddell, M., F. Maisels, A. Lawrence, et al. 2022. Combining offtake and participatory data to assess the sustainability of a hunting system in northern Congo. *African Journal of Ecology* 60(2): 250-267.

Ripple, WJ., K. Abernethy, MG. Betts, et al. 2016. Bushmeat hunting and extinction risk to the world's mammals. *Royal Society open science* 3 (10): 160498.

Rist, J., EJ. Milner-Gulland, GU. Cowlishaw, et al. 2010. Hunter reporting of catch per unit effort as a monitoring tool in a bushmeat-harvesting system. Conservation Biology, 24(2): 489-499.

Robinson, JG. and KH. Redford. 1986. Body size, diet and population density of neotropical forest mammals. *American Naturalist* 128:665-680.

Royle, JA. and M. Kéry. 2007. A Bayesian state-space formulation of dynamic occupancy models. Ecology 88: 1813–1823.

Sampaio, R., RG. Morato, A. Royle, et al. 2023. Vertebrate population changes induced by hunting in Amazonian sustainable-use protected areas. *Biological Conservation* 284:110206.

Santos, SL., MF. De la Fuente, and RRN. Alves. 2022. Patterns associated with hunting with dogs in a semiarid region of northeastern Brazil. *Journal of Ethnobiology and Ethnomedicine* 18(1): 1-13.

Saúde & Alegria. 2012. Mapeamentos Participativos e elaboração de Planos de Uso: uma experiência nas comunidades de Anã, Arimum, Atodi e Vila Amazonas nas margens do Rio Arapiuns. Saúde & Alegria, Brazil: Santarém, Pará.

Scabin, AB. and CA. Peres. 2021. Hunting pressure modulates the composition and size structure of terrestrial and arboreal vertebrates in Amazonian forests. *Biodiversity and Conservation* 30: 3613–3632.

Sirén, A. and J. Machoa. 2008. Fish, wildlife, and human nutrition in tropical forests: a fat gap? *Interciencia* 33:186–93.

SNUC - Sistema Nacional de Unidades de Conservação; Lei 9.985 de 18 de julho de 2000. Ministério do Meio Ambiente - Brazil. Available at: https://www.planalto.gov.br/ccivil_03/leis/l9985.htm.

Souza-Mazurek, RR., T. Pedrinho, X. Feliciano, et al. 2000. Subsistence hunting among the Waimiri Atroari indians in central Amazonia, Brazil. *Biodiversity & Conservation* 9(5): 579-596.

Souza, TT., KT. Ribeiro, CF. Tófoli, et al. 2019. Monitoramento da biodiversidade para as estratégias de conservação: experiência do Programa Monitora. In Tófoli, C. F.; et al. (Eds.). Monitoramento participativo da biodiversidade. Aprendizados em evolução. 2nd ed. São Paulo: IPÊ – Instituto de Pesquisas Ecológicas/MEMNON, 68–83. Available at: https://conteudo.ipe.org.br/livro-mpb.

Stafford, CA., RF. Preziosi, and WI. Sellers. 2017. A pan-neotropical analysis of hunting preferences. *Biodiversity and Conservation* 26: 1877-1897.

Su, YS., M. Yajima. 2012. R2jags: a package for running JAGS from R. R package version 0.03-08 [www document]. Available at: https://cran.r-project.org/web/packages/R2jags/index.html.

Valsecchi, J. and PV. Amaral. 2009. Perfil da caça e dos caçadores na Reserva de Desenvolvimento Sustentável Amanã, Amazonas–Brasil. *Scientific Magazine UAKARI* 5(2): 33-48.

Valsecchi, J. 2013. Caça de Animais Silvestres nas Reservas de Desenvolvimento Sustentável Mamirauá e Amanã. PhD thesis. Universidade Federal de Minas Gerais, Minas Gerais/BH/Brazil.

Van Vliet, N., J. Moreno, J. Gómez, et al. 2017. Bushmeat and human health: assessing the evidence in tropical and sub-tropical forests. *Ethnobiology and Conservation* 6(3):1–45.

Vieira, MARM., EM. von Muhlen, and GH. Shepard Jr. 2015. Participatory monitoring and management of subsistence hunting in the Piagaçu-Purus reserve, Brazil. *Conservation and Society* 13(3): 254-264.

Zapata-Rios, G., C. Urgiles, and E. Suarez. 2009. Mammal hunting by the Shuar of the Ecuadorian Amazon: is it sustainable?. *Oryx* 43(3): 375-385.

Weinbaum, KZ., JS. Brashares, CD. Golden, et al. 2013. Searching for sustainability: are assessments of wildlife harvests behind the times?. *Ecology letters* 16(1): 99-111.

Wilkie, DS., EL. Bennett, CA. Peres, et al. 2011. The empty forest revisited. *Annals of the New York Academy of Sciences* 1223(1): 120-128.

Supplementary data

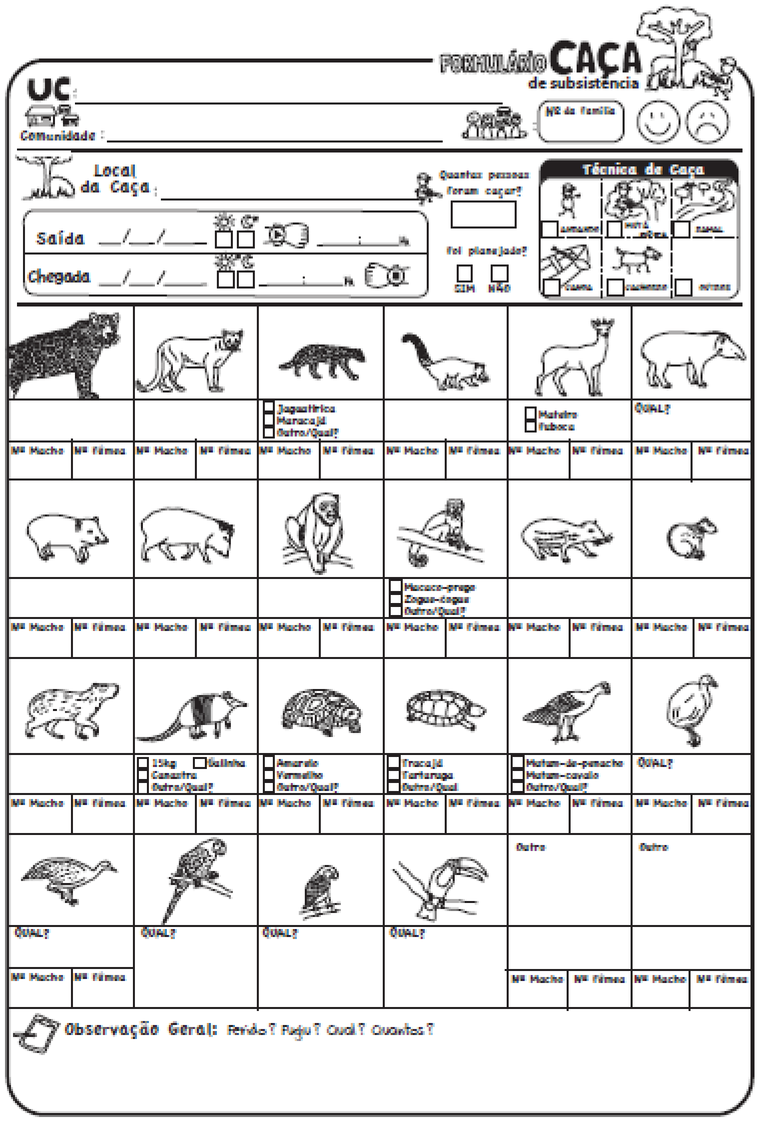


Fig. S1. Hunting form used by families that participated in sub-protocol 2 (*Hunting pressure on terrestrial game populations*) of the Monitora Program across eight hub villages in the Tapajós-Arapiuns Extractive Reserve.

Table S1. Values of the *proxy* of human intensity, hub village age and hunting events recorded per family in the hub and nearby villages that participated in sub-protocol 2 (*Hunting pressure on terrestrial game populations*) of the Monitora Program of the Tapajós-Arapiuns Extractive Reserve, between 2015 and 2019.

Hub village	Hub or nearby village	Nº families	Hub village age	Proxy of human intensity	Successful hunting events	Unsuccessful hunting events
Anã	Anã	46	69	615.64	215	302
Anã	Maripá	16	-	-	101	14
Boim	Boim	24	286	869.68	237	109
Boim	Rosário	6	-	-	57	16
Boim	Tucumatuba	8	-	-	54	4
Cametá	Cametá	87	73	1025.60	1383	503
Cametá	Pinhel	46	-	-	461	208
Escrivão	Escrivão	24	188	393.74	354	111
Pascoal	Pascoal	14	33	272.93	202	43
Pascoal	Mentai	14	-	-	29	5
Prainha	Prainha	27	44	318.67	294	35
São Pedro	São Pedro	54	82	738.00	390	81
São Tomé	São Tomé	25	112	670.60	386	166
Total		391			4163	1597

Table S2. Successful hunting events per hub village monitored through the sub-protocol 2 (*Hunting pressure on terrestrial game populations*) of the Monitora Program in the Tapajós-Arapiuns Extractive Reserve, between 2015 and 2019.

Village	2015	2016	2017	2018	2019	Total	Mean
Anã (Anã, Maripá)	47	22	166	81	-	316	79
Boim (Boim, Rosário, Tucumatuba)	43	41	183	81	-	348	87
Cametá (Cametá, Pinhel)	90	97	687	729	241	1844	368.8
Escrivão (Escrivão)	126	56	117	32	23	354	70.8
Pascoal (Pascoal, Mentai)	33	45	56	68	29	231	46.2
Prainha (Prainha)	-	-	72	144	78	294	98
São Pedro (São Pedro)	81	7	298	-	4	390	97.5
São Tomé (São Tomé)	178	51	73	84	-	386	96.5
Total	598	319	1652	1219	375	4163	832.6
Mean	85.4	45.6	206.5	174.1	75.0	520.4	

Table S3. Number of individuals (i) and overall biomass (Kg) harvested by game taxon in the villages that participated in sub-protocol 2 (*Hunting pressure on terrestrial game populations*) of the Monitora Program in the Tapajós-Arapiuns Extractive Reserve from 2015 to 2019.

			Ana		Boim		Cametá		Escrivão		Pascoal		Prainha		São Pedro		São Tomé		Total	
Taxa	Species	Popular name	i	kg	i	kg	i	kg	i	kg	i	kg	i	kg	i	kg	i	kg	i	kg
									Mammals											
Alouatta nigerrima	*Alouatta nigerrima*	howler monkeys	5	30.9	4	24.7	18	111.3	14	86.6	2	12.4	8	49.5	14	86.6	1	6.2	66	408.2
Callicebus hoffmannsi	*Callicebus hoffmannsi*	titi monkeys	-	-	1	0.9	-	-	-	-	-	-	4	3.7	-	-	-	-	5	4.6
Sapajus apella	*Sapajus apella*	capuchin monkeys	9	26.2	4	11.6	17	49.5	4	11.6	9	26.2	19	55.3	7	20.4	1	2.9	70	203.7
Dicotyles tajacu	*Dicotyles tajacu*	collared peccary	34	680.0	45	900.0	325	6500.0	26	520.0	6	120.0	28	560.0	17	340.0	25	500.0	506	10120.0
Mazama spp.	*Mazama americana, M. nemorivaga*	brown and red brocket deer	33	907.5	60	1650.0	189	5197.5	10	275.0	25	687.5	32	880.0	63	1732.5	35	962.5	447	12292.5
Tayassu pecari	*Tayassu pecari*	white-lipped peccary	2	50.0	7	175.0	197	4925.0	45	1125.0	6	150.0	5	125.0	9	225.0	6	150.0	277	6925.0
Dasypus spp.	*Dasypus* spp.	armadillo	118	708.0	74	444.0	372	2232.0	111	666.0	47	282.0	70	420.0	91	546.0	168	1008.0	1051	6306.0
Myrmecophaga tridactyla	*Myrmecophaga tridactyla*	giant anteater	-	-	-	-	-	-	2	54.0	-	-	-	-	2	54.0	-	-	4	108.0
Priodontes Maximus	*Priodontes Maximus*	giant armadillo	-	-	-	-	4	157.6	-	-	-	-	3	118.2	-	-	-	-	7	275.8
Tapirus kabomani	*Tapirus kabomani*	black tapir	-	-	-	-	-	-	-	-	-	-	1	110.0	-	-	-	-	1	110.0
Tapirus terrestris	*Tapirus terrestris*	lowland tapir	-	-	3	480.0	25	4000.0	2	320.0	-	-	3	480.0	-	-	1	160.0	34	5440.0
Cuniculus paca	*Cuniculus paca*	lowland paca	63	492.7	69	539.6	675	5278.5	167	1305.9	48	375.4	74	578.7	90	703.8	54	422.3	1240	9696.8
Dasyprocta croconota	*Dasyprocta croconota*	agouti	55	137.5	136	340.0	663	1657.5	118	295.0	123	307.5	186	465.0	200	500.0	68	170.0	1549	3872.5
Hydrochaeris hydrochaeris	*Hydrochaeris hydrochaeris*	capybara	8	252.0	3	94.5	53	1669.5	9	283.5	3	94.5	2	63.0	4	126.0	9	283.5	91	2866.5
Leopardus wiedii	*Leopardus wiedii*	margay	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	3.3	1	3.3
Panthera onca	*Panthera onca*	jaguar	-	-	-	-	6	412.5	1	68.8	1	68.8	1	68.8	7	481.3	1	68.8	17	1168.8
Puma concolor	*Puma concolor*	puma	-	-	2	69.0	5	172.5	-	-	-	-	-	-	1	34.5	-		8	276.0
Nasua nasua	*Nasua nasua*	coati	30	116.4	16	62.1	40	155.2	8	31.0	20	77.6	16	62.1	31	120.3	7	27.2	168	651.8
Total																			**5542**	**60729.5**
									Birds											
Cracidae	*Penelope* spp., *Crax* spp., *Pauxi tul*	curassows and common guan	-	-	4	7.5	50	94.0	7	13.2	12	22.6	23	43.2	18	33.8	3	5.6	117	220.0
Psittacidae	*Ara chloropterus, Anodorhynchus h*	macaw and parrot	5	5.7	5	5.7	65	73.5	9	10.2	3	3.4	7	7.9	2	2.3	2	2.3	98	110.7
Psophiidae	*Psophia viridis*	dark-winged trumpeter	-	-	10	9.9	5	5.0	4	4.0	4	4.0	18	17.8	4	4.0	1	1.0	46	45.5
Ramphastidae	*Ramphastos* spp.	toucan	1	0.5	6	3.2	7	3.8	-	-	5	2.7	3	1.6	-	-	-	-	22	11.9
Tinamidae	*Tinamus* spp., *Crypturellus* spp.	tinamous	1	0.7	3	2.1	32	22.4	4	2.8	-	-	6	4.2	5	3.5	1	0.7	52	36.4
Total																			**335**	**424.5**
									Reptiles											
Caiman crocodilus	*Caiman crocodilus*	caiman	1	38.6	-	-	-	-	-		-	-	-	-	-	-	-	-	1	38.6
Chelonoidis spp.	*Chelonoidis carbonaria*, C. *denticula*	forest tortoise	3	23.4	51	397.3	228	1776.1	52	405.1	15	116.9	118	919.2	15	116.9	46	358.3	528	4113.1
Podocnemis unifilis	*Podocnemis unifilis*	yellow-spotted river turtle	2	12.2	-	-	9	54.7	11	66.9	-	-	7	42.5	1	6.1	-	-	30	182.3
Total																			**559**	**4334.1**
Grand total																			**6436**	**65488.1**

Table S4. Mean and standard deviation of the CPUE of overall game taxa and the six most hunted taxa in eight hub villages that participated in sub-protocol 2 (*Hunting pressure on terrestrial game populations*) of the Monitora Program in the Tapajós-Arapiuns Extractive Reserve.

Hub village	**All game taxa**	***Cuniculus paca***	***Dasypus* spp.**	***Mazama* spp.**	***Dicotyles tajacu***	***Tayassu pecari***	***Tapirus terrestris***
Anã (Anã, Maripá)	3.2 ± 4.2	3.0 ± 4.6	2.4 ± 2.7	6.5 ± 6.5	5.6 ± 3.3	7.4 ± 0.0	-
Boim (Boim, Rosário, Tucumatuba)	3.5 ± 6.2	2.9 ± 1.9	2.3 ± 1.4	7.4 ± 4.6	4.0 ± 1.6	4.8 ± 2.4	43.0 ± 32.9
Cametá (Cametá, Pinhel)	2.0 ± 2.8	1.60 ± 1.7	1.5 ± 1.8	3.0 ± 1.7	3.1 ± 1.9	3.2 ± 1.6	18.2 ± 8.5
Escrivão (Escrivão)	2.2 ± 2.4	2.5 ± 2.6	1.9 ± 1.3	6.6 ± 4.0	4.8 ± 0.3	3.6 ± 1.2	1.8 ± 0.0
Pascoal (Pascoal, Mentai)	2.9 ± 4.0	3.7 ± 4.7	3.7 ± 3.7	8.0 ± 5.5	6.6 ± 0.8	9.9 ± 6.7	-
Prainha (Prainha)	2.3 ± 4.4	3.4 ± 7.4	2.7 ± 2.4	7.1 ± 11.6	3.4 ± 1.7	2.0 ± 0.6	17.6 ± 11.3
São Pedro (São Pedro)	2.3 ± 2.7	2.3 ± 2.4	2.3 ± 2.4	5.3 ± 3.3	4.9 ± 3.7	7.0 ± 2.8	-
São Tomé (São Tomé)	2.1 ± 2.2	2.1 ± 1.1	1.4 ± 0.8	5.7 ± 3.0	5.0 ± 1.9	8.3 ± 0.0	-
Mean ± SD	2.6 ± 0.5	2.7 ± 0.7	2.3 ± 0.7	6.2 ± 1.5	4.6 ± 1.1	5.8 ± 2.6	20.2 ± 14.7

Table S5. State-space models examining trends in hunting productivity in terms of catch per unit of effort (CPUE; kg hunter^{-1} h^{-1}) of overall game taxa and the six most hunted taxa in eight hub villages that participated in sub-protocol 2 (*Hunting pressure on terrestrial game populations*) of the Monitora Program in the Tapajós-Arapiuns Extractive Reserve from 2016 to 2019.

CPUE trend	Parameter	MTS	SD	2.5% BCI	97.5% BCI	Rhat
All game species	r	-0.007	0.026	-0.067	0.039	1.003
Cuniculus paca	r	-0.005	0.068	-0.157	0.15	1.001
Dasypus spp.	r	-0.007	0.039	-0.086	0.079	1.001
Mazama spp.	r	-0.003	0.03	-0.064	0.059	1.002
Dycoteles tajacu	r	0.011	0.016	-0.018	0.046	1.001
Tayassu pecari	r	-0.003	0.035	-0.07	0.074	1.002
Tapirus terrestris	r	-0.147	0.187	-0.529	0.222	1.001

CONCLUSÃO GERAL

Os resultados obtidos nesta tese oferecem uma contribuição significativa no entendimento dos desafios e potencialidades associados ao monitoramento comunitário de vertebrados cinegéticos terrestres *in situ*, bem como da fauna caçada em florestas tropicais. Adicionalmente, apresentamos novas evidências acerca dos impactos da pressão antropogênica nas populações de espécies terrestres florestais e nos padrões de caça ao longo do tempo, contribuindo, por fim, para a gestão da caça em uma Unidade de Conservação amazônica de uso sustentável – a RESEX Tapajós-Arapiuns. Destacamos igualmente a crescente importância dos projetos de monitoramento comunitário no fornecimento de dados essenciais para avaliar a eficácia das reservas de uso sustentável em compatibilizar a conservação da biodiversidade com o uso dos recursos naturais. Por último, propomos um protocolo robusto de monitoramento da caça de subsistência para outras áreas protegidas interessadas em monitorar esse recurso.

A partir da revisão sistemática da literatura realizada no primeiro capítulo, conduzimos uma análise abrangente de projetos de monitoramento comunitário da fauna cinegética terrestre em florestas tropicais. Nossas descobertas revelaram que esses projetos possuem um potencial significativo para gerar informações contínuas sobre os recursos monitorados, empoderar as comunidades locais e implementar ações de manejo. No entanto, identificamos alguns desafios, como interrupções frequentes devido à falta de financiamento e a necessidade de maior participação local para alcançar um empoderamento efetivo. Nós assim enfatizamos a importância de engajar ativamente as comunidades em todas as etapas do monitoramento, estabelecer parcerias sólidas para garantir financiamento a longo prazo e traduzir os resultados em práticas de manejo voltadas para a conservação da fauna monitorada.

No segundo capítulo, utilizamos um banco de dados de seis anos de monitoramento comunitário na Reserva Extrativista Tapajós-Arapiuns, e revelamos que, apesar das pressões antropogênicas, a densidade populacional e biomassa das espécies de fauna terrestre cinegética, de maneira geral, não foram afetadas. Especificamente, observamos que Tinamidae foi o único táxon influenciado negativamente por variáveis antrópicas, entretanto, as densidades de todos os táxons analisados se mantiveram estáveis ao longo da série temporal. Nossas descobertas sugerem que as populações de vertebrados cinegéticos foram mantidas durante o período avaliado, mas os baixos registros de algumas espécies podem indicar perdas populacionais passadas, o que se refletiria em um cenário atual de semi-defaunação. Como utilizamos apenas os dados de transecções lineares, sugerimos que futuros estudos analisem os dados das armadilhas fotográficas, que permitirá a inclusão de outras espécies tradicionalmente caçadas, mas pouco registradas pelo método de transecção linear como a anta (*Tapirus terrestris*), a paca (*Cuniculus paca*) e os tatus (*Dasypodidae*).

No terceiro e último capítulo, evidenciamos que o perfil de caça na Reserva Extrativista Tapajós-Arapiuns segue o padrão observado em outras regiões da Amazônia, e demonstramos que a pressão antropogênica não exerce impacto sobre o perfil e a produtividade de caça (medida pela CPUE) ao longo de 5 anos de monitoramento. Além disso, constatamos que a CPUE permaneceu estável ao longo do período analisado. Nossos resultados indicam que a caça é provavelmente sustentável para espécies abundantes e resilientes (incluindo pacas, tatus e veados), mas insustentável para espécies ameaçadas como a anta (*T. terrestres*) e o queixada (*Tayassu pecari*). O monitoramento realizado na Reserva Tapajós-Arapiuns fornece um protocolo sólido que pode ser aplicado em outras áreas protegidas de uso sustentável na Amazônia, que tenham interesse em monitorar a caça de subsistência.

Especificamente na RESEX Tapajós-Arapiuns, os dados coletados entre 2015 e 2020 sugerem que, embora as análises não tenham indicado influência das variáveis antropogênicas avaliadas nas populações de vertebrados de médio e grande porte e nos padrões de caça, é preciso reconhecer a importância de considerar o sistema RESEX-TA dentro de um contexto mais amplo. Estudos anteriores realizados nesta mesma área protegida documentaram a influência dos incêndios florestais e da pressão da caça na estrutura florestal e nas populações cinegéticas (PERES *et al*., 2003; BARLOW & PERES, 2005). O histórico de incêndios florestais e pressão de caça na região, combinado com a alta densidade populacional humana (~3,55 pessoas/km² para toda a região; SILVA *et al*., 2022; INSTITUTO SOCIOAMBIENTAL, 2022) pode estar contribuindo para um cenário de semi-defaunação. Portanto, a estabilidade das populações e da produtividade de caça ao longo da série temporal analisada pode ser um indicador de que o sistema se encontra atualmente num estado de equilíbrio "frágil", potencialmente mantendo a estabilidade com uma estrutura ecológica alterada. O monitoramento de longo prazo e entrevistas com caçadores serão fundamentais para fornecer informações mais profundas que esclareçam essas reflexões e assim avaliar a necessidade de implementar regulamentos de caça nessas áreas. Além disso, nossos resultados evidenciaram que a técnica de caça mais prevalente entre os caçadores, a busca ativa com cães, levou ao abate de espécies raras e ameaçadas durante o dia, incluindo onças (*Panthera* *onca*), onças-pardas (*Puma concolor*), tatus gigantes (*Priodontes maximus*) e antas (*T. terrestris*), demonstrando os impactos negativos do uso de cães em eventos de caça. Nesse caso, recomendamos especificamente a implementação de regras para restringir a caça com cães, como a imposição de penalidades, a fim de evitar a captura de espécies ameaçadas de extinção. Contudo, é crucial formalizar essas ações de manejo para evitar arbitrariedades na fiscalização local e garantir supervisão adequada para o cumprimento das medidas propostas. Por fim, em relação à indagação das comunidades da RESEX sobre a preservação futura da fauna, destacamos que é prematuro fornecer uma resposta conclusiva. Contudo, os dados de uma série temporal de curto prazo sugerem que a conservação e a caça sustentável estão possivelmente sendo praticadas na Tapajós-Arapiuns para as espécies mais abundantes *in situ* e mais resilientes. A persistência dessas espécies ocorre provavelmente devido a grandes extensões de floresta pouco povoada e não perturbada no interior da unidade de conservação que podem servir como "fontes" populacionais de animais, facilitando a dispersão para áreas de "sumidouros" (i.e., aquelas onde a caça ocorre devido à acessibilidade e proximidade com as comunidades locais) (NOVARO *et al.*, 2000). Por outro lado, para as espécies ameaçadas de grande porte, a caça provavelmente está ocorrendo de forma insustentável como resultado de perdas passadas em populações de vertebrados de grande porte.

Em uma escala mais ampla, ressaltamos a importância e encorajamos a inclusão de protocolos de monitoramento comunitário da caça como estratégia para gerar informações úteis destinadas à gestão desse recurso em unidades de conservação de uso sustentável, visando assegurar tanto a conservação da biodiversidade quanto a segurança alimentar das populações tradicionais. Adicionalmente, incentivamos tanto a população local quanto os gestores a se envolverem ativamente no programa de monitoramento, além de garantir um planejamento bem estruturado para sua execução a longo prazo. Isso compreende também o estabelecimento de parcerias que garantam financiamento contínuo, visando evitar interrupções indesejadas. Isso tem particular importância, uma vez que a interrupção de inúmeras iniciativas de monitoramento prejudica a aquisição de dados a longo prazo sobre as populações monitoradas (REIS & BENCHIMOL, 2023).