Universidade do Estado do Rio de Janeiro

Centro Biomédico

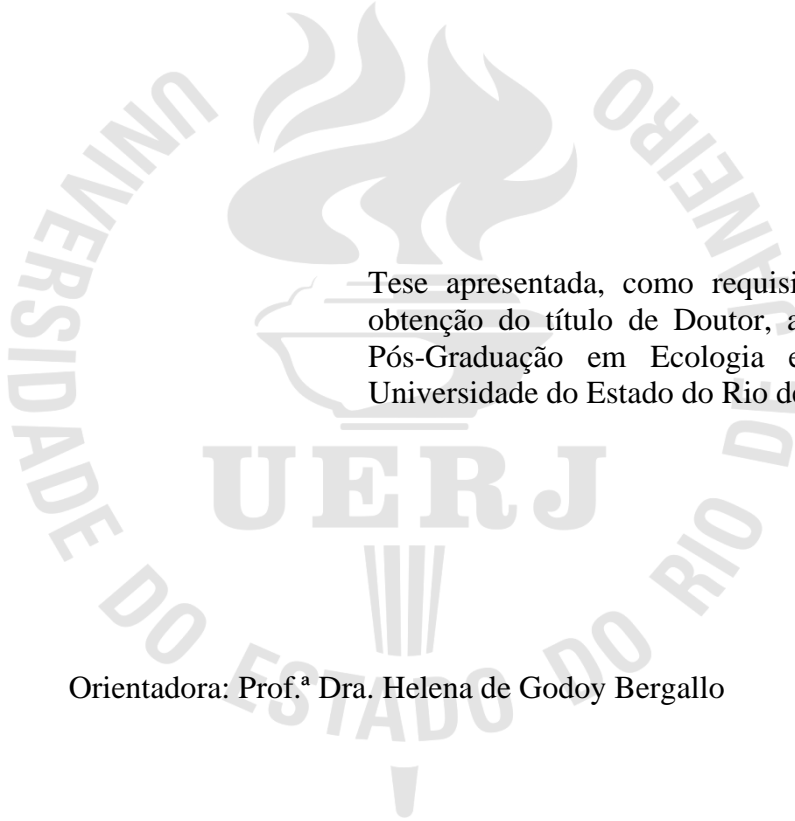Instituto de Biologia Roberto Alcântara Gomes

Átilla Colombo Ferreguetti

Don't shoot:poaching impact on the non-volant mammals community in a protected area in the Atlantic Forest, Southeastern Brazil

Rio de Janeiro

2020

Átilla Colombo Ferreguetti

Don't shoot: poaching impact on the non-volant mammals community in a protected area in the Atlantic Forest, Southeastern Brazil

Tese apresentada, como requisito parcial para obtenção do título de Doutor, ao Programa de Pós-Graduação em Ecologia e Evolução da Universidade do Estado do Rio de Janeiro.

Orientadora: Prof.ª Dra. Helena de Godoy Bergallo

Rio de Janeiro

2020

Autorizo, apenas para fins acadêmicos e científicos, a reprodução total ou parcial desta tese, desde que citada a fonte.

_____   _____

Assinatura   Data

Átilla Colombo Ferreguetti

Don't shoot: poaching impact on the community of non-volant mammals in a protected area in the Atlantic Forest, Southeastern Brazil

<div align="right">

Tese apresentada, como requisito parcial para obtenção do título de Doutor, ao Programa de Pós-Graduação em Ecologia e Evolução da Universidade do Estado do Rio de Janeiro.

</div>

Aprovado em 18 de Fevereiro de 2020.

Banca Examinadora:

_____

Prof.ª Dra. Helena De Godoy Bergallo (Orientadora)

Instituto de Biologia Roberto Alcantara Gomes

_____

Prof. Dr. Jayme Augusto Prevedello

Instituto de Biologia Roberto Alcantara Gomes

_____

Prof. Dr. Marcus Vinícius Vieira

Universidade Federal do Rio de Janeiro

_____

Prof. Dr. Adriano Pereira Paglia

Universidade Federal de Minas Gerais

Rio de Janeiro

2020

DEDICATION

Á minha mãe amada, Maria
Zeleonir Colombo Ferreguetti,
que me deu força, incentivo e
sonhou um dia ver seu filho
Doutor.

ACKNOWLEDGMENTS

Nenhum trabalho é possível sozinho! Essa tese e toda minha formação acadêmica não seriam possíveis sem a participação fundamental de vários amigos, professores, família e todos que de alguma forma contribuíram para que mais essa etapa da minha vida fosse realizada. Gostaria de agradecer então a todos que comigo construíram este trabalho.

Primeiramente gostaria de agradecer a minha família, que desde quando eu comecei a minha formação como biólogo, esteve me apoiando das demais formas possíveis. Sem essa família guerreira que eu tenho, eu jamais teria chegado até aqui. Então ao meu Pai (Átilio Câmara Ferreguetti), pela ajuda na preparação do material de campo, com sua boa e velha arte de construir "bugigangas" e também é claro por sempre acreditar em mim e falar de mim com tanto orgulho. À minha mãe (Maria Zeleonir Colombo Ferreguetti "zizona"), por ser minha melhor amiga e estar sempre lá me dando àquela palavra de "coragem, não desista" e por apoiar todo esse sonho de seguir essa profissão, que mesmo quando não tinha uma palavra, me consolava com aquele colo de mãe. Minha mãe que na reta final do doutorado enfretou tantas provações, e que mesmo em uma maca de hospital continou me dando forças. À minha querida e sensacional irmã (Cecyane Colombo Ferreguetti "cecy") por sempre acreditar em mim, me dar forças quando mais precisei e ser essa menina mulher, doce e sensível que tanto cresceu nesses últimos 2 anos. Também por aguentar o tranco lá de casa enquanto estive no Rio estudando. Muito obrigado mesmo, minha querida irmã! É claro que não poderia deixar de agradecer a Ju Souza (minha cunhada) por entrar em nossas vidas e trazer tanta luz!

À minha querida Orientadora Dra. Helena de Godoy Bergallo (Nena), por tanto me ensinar, me orientar e confiar no meu trabalho. Agradeço tremendamente por essa oportunidade e esse voto de confiança! Agradeço também por todos os ensinamentos, tanto os profissionais quanto os pessoais. Obrigado por me dar a chance de desenvolver algumas ideias megalomaníacas e sempre me permitir trabalhar com liberdade. Obrigado por todo esse processo, que além de orientadora você é uma mãezona científica, exemplo de profissional.

Ao meu querido companheiro Pablo Bonfante Batista, que abraçou essa empreitada comigo do início ao fim, seja dando forças, conforto e até mesmo ajudando no campo quando todos furavam com a gente. Esse economista colocou muitos biólogos no chinelo! Excelente pessoa e profissional, obrigado por tudo Tico!

Á minha amiga/parceira/dupla da equipe Rocket Juliane Pereira Ribeiro, que enfrentou comigo cada segundo nessa busca atrás do picachu. Foram muitos perrengues, desafios e risadas divididas em campo. Todo campo a gente chegava com mil planos e sempre decolávamos na

velocidade da luz. Com certeza nossas lombares aposentaram na Reserva Natural Vale haha, disso não tenho dúvidas. Obrigado por todo apoio e companhia Ju, você foi fundamental e indispensável.

Ao professor Dr. Carlos Frederico Duarte da Rocha por sempre ajudar e pelas ricas contribuições e ideias para melhorar o trabalho.

Á todos que nos ajudaram em campo: Stênnio, Gleika, Rayane, André, Nicole, Fabi, Carol, Karoline, Laura, Jane, Fink, Gabriel, Priscila, Marina, Pablo, Léo, Vinicius, Verônica, Paula, Luísa e Clara.

Aos meus queridos amigos do Espírito Santo, Lenon, Julia, Juliana, Ricardo, Yann, e Lia por todos os momentos maravilhosos, que me deixaram super feliz durante esse processo!! Amo vocês!!

Aos companheiros do Laboratorio de Ecologia de Mamíferos – LEMA. Aos professores da UERJ.

À toda a equipe da Reserva Natural Vale, pela oportunidade de realizar meu projeto de doutorado e agradeço principalmente à Jackeceli, Rayany, Manu e Márcio por me ajudarem durante o processo de realização da pesquisa na RNV. Agradeço também é claro a todos os funcionários que tornaram esse trabalho mais agradável com um sorriso no rosto para me receber. A pesquisa foi conduzida sob a licença do SISBIO nº: 47141.

Aos funcionários do PPGEE, em especial a Soninha eVeruska por sempre me ajudarem nas dúvidas que tive. Vocês são nota 1000!

Aos queridos membros da comissão avaliadora da presente tese, em especial ao Prof. Dr. Jayme Augusto Prevedello que acompanhou auxiliando desde a finalização do mestrado até esse etapa final do meu doutoramento.

Ao professor Dr. David Miller, da Penn StateUniversity por me receber durante 6 meses em seu departamento e me ensinar as técnicas de modelagem de ocupação. Obrigado pelos encontros semanais e também pelas oportunidades que tive de participar das atividades do seu laboratório e aprender bastante.

À todos os animais, que me concederam a honra de sua presença durante todos os 5 anos de esforço para a realização desse projeto. Sem eles, tudo teria sido tão chato!!

Meu muito obrigado!

Se você pudesse me dizer
Se você soubesse o que fazer
O que você faria
Aonde iria chegar?
Se você soubesse quem você é
Até onde vai a sua fé
O que você faria?
Pagaria pra ver
Se pudesse escolher
Entre o bem e o mal, ser ou não
ser?
Se querer é poder
Tem que ir até o final
Se quiser vencer
Se pudesse eu te levaria
Até onde você quer chegar
Brilho das estrelas
O primeiro lugar
Se pudesse escolher
Entre o bem e o mal, ser ou não
ser?
Se querer é poder
Tem que ir até o final
Se quiser vencer
O mundo é perigoso
E cheio de armadilhas
De mistério e gozo
Verdades e mentiras
Viver é quase um jogo
Um mergulho no infinito
Se souber brincar com fogo
Não há nada mais bonito
Se pudesse escolher
Entre o bem e o mal, ser ou não
ser?
Se querer é poder
Tem que ir até o final
Se quiser vencer

Vida Real - Paulo Ricardo

ABSTRACT

FERREGUETTI, A. C. *Don't shoot:* poaching impact on the non-volant mammals community in a protected area in the Atlantic Forest, Southeastern Brazil. 2020. 99f. Tese (Doutorado em Ecologia e Evolução) - Instituto de Biologia Roberto Alcantra Gomes, Universidade do Estado do Rio de Janeiro, Rio de Janeiro, 2020.

Hunting can be considered as a cultural trait strongly rooted in the Neotropics and considered in several aspects, depending on the human community in question and the region considered. Poachingmay affect mammalian populations, and change communities, although it tends to be underestimated, due to lack of standardization and difficulties for detection. Therefore, we aimed to quantify the effect of poaching on non-volant mammal species through 5-year monitoring in the Vale Natural Reserve (VNR), northern Espírito Santo state, Brazil. We hoped that the information generated here can contribute to the increase in knowledge of the ecology of neotropical mammals by knowing how species respond to anthropic actions (i.e. poaching) as well as providing information from this group to support environmental policies and actions aimed at conserving biodiversity. This thesis was divided into three chapters. We proposed a quantitative metric assessment, which considers the spatial distribution of hunting evidence per km2 and allows monitoring the trends of this impact over time. This metric will collect the evidence of illegal hunting in a standardized way over the time. The metric can be generated by considering each poaching event separately (date, reserve where the event was recorded, location/region of the event, geographic coordinates and type of evidence collected). Any evidence of hunting can be georeferenced over time. In addition, our results provide subsidies for increasing knowledge about an illegal but still widespread and common practice. We presented here an approach to identify poaching hotspots that is cost-effective and much cheaper than years of extensive monitoring, and therefore may be applied in other protected and non-protected areas throughout the world. We recommend that researchers and Reserve managers work together to apply the approach presented here in other Reserves, whenever possible. Camera-trapping has been increasingly used throughout the world, and it is likely that the necessary information to replicate our analyses has already been obtained in many Reserves. Lastly, our results bring alarming results to one of Brazil's largest Atlantic Forest remnants. Poaching is an illegal activity constantly practiced not only in the Vale Natural Reserve but throughout the Atlantic Forest and other Neotropical regions. We presented the first study in the Atlantic Forest that has estimated the long-term poaching impact on mammalian populations, demonstrating a potential negatively effect on the density, biomass and occupancy of species that are important to ecosystem functioning. In addition to the direct negative effects estimated in this study, we draw attention to potential indirect impacts of poaching on other mammalian species, such as large felids (*Panthera onca* and *Puma concolor*). It is extremely necessary to initiate and strengthen programs for poaching mitigation in our protected areas. It is necessary to sensitize the local community by establishing a sense of belonging with the protected area. A continuing environmental education program is recommended, where actions must be worked on in different age groups of the population.

Keywords: Biomass. Density. Conservation of biodiversity. Distance sampling. Imperfect detectability. Anthropogenic impact. Occupancy.

RESUMO

FERREGUETTI, A. C. *Não atire:* impacto da caça furtiva na comunidade de mamíferos não-voadores em uma área protegida na Mata Atlântica, sudeste do Brasil. 2020. 99f. Tese (Doutorado em Ecologia e Evolução) - Instituto de Biologia Roberto Alcantra Gomes, Universidade do Estado do Rio de Janeiro, Rio de Janeiro, 2020.

A caça pode ser considerada uma característica cultural fortemente enraizada nos neotrópicos e considerada em vários aspectos, dependendo da comunidade humana em questão e da região considerada. A caça pode afetar populações de mamíferos e alterar comunidades, embora tenda a ser subestimada, devido à falta de padronização e dificuldades de detecção. Portanto, objetivamos quantificar o efeito da caça furtiva em espécies de mamíferos não-voadores por meio de monitoramento de cinco anos na Reserva Natural do Vale (VNR), norte do Espírito Santo, Brasil. Esperamos que as informações geradas aqui possam contribuir para o aumento do conhecimento da ecologia dos mamíferos neotropicais, entendendo como as espécies respondem a ações antrópicas (ou seja, caça furtiva), bem como fornecendo informações desse grupo para apoiar políticas e ações ambientais voltadas à conservação da biodiversidade. Esta tese foi dividida em três capítulos. Propusemos uma avaliação métrica quantitativa, que considera a distribuição espacial das evidências de caça por km^2 e permite monitorar as tendências desse impacto ao longo do tempo. Essa métrica coletará as evidências de caça ilegal de maneira padronizada ao longo do tempo. A métrica pode ser gerada considerando-se cada evento de caça furtiva separadamente (data, reserva onde o evento foi registrado, local / região do evento, coordenadas geográficas e tipo de evidência coletada). Qualquer evidência de caça pode ser georreferenciada ao longo do tempo. Além disso, nossos resultados fornecem subsídios para aumentar o conhecimento sobre uma prática ilegal, mas ainda generalizada e comum. Apresentamos aqui uma abordagem para identificar hotspots de caça furtiva com boa relação custo-benefício e muito mais baratos que anos de monitoramento extensivo e, portanto, podem ser aplicados em outras áreas protegidas e não protegidas em todo o mundo. Recomendamos que pesquisadores e gerentes de reservas trabalhem juntos para aplicar a abordagem apresentada aqui em outras reservas, sempre que possível. A captura de câmeras tem sido cada vez mais usada em todo o mundo e é provável que as informações necessárias para replicar nossas análises já tenham sido obtidas em muitas Reservas. Por fim, nossos resultados trazem resultados alarmantes para um dos maiores remanescentes da Mata Atlântica do Brasil. A caça furtiva é uma atividade ilegal constantemente praticada não apenas na RNV, mas em toda a Mata Atlântica e outras regiões neotropicais. Apresentamos o primeiro estudo na Mata Atlântica que estimou o impacto da caça furtiva a longo prazo nas populações de mamíferos, demonstrando um potencial efeito negativo na densidade, biomassa e ocupação de espécies importantes para o funcionamento do ecossistema. Além dos efeitos negativos diretos estimados neste estudo, chamamos atenção para os possíveis impactos indiretos da caça furtiva em outras espécies de mamíferos, como os felídeos grandes (*Pantheraonca* e *Puma concolor*). É extremamente necessário iniciar e fortalecer programas para mitigar a caça furtiva em nossas áreas protegidas. É necessário sensibilizar a comunidade local, estabelecendo um sentimento de pertencimento à área protegida. Recomenda-se um programa contínuo de educação ambiental, onde ações devem ser trabalhadas em diferentes faixas etárias da população.

Palavras-chave: Biomassa. Densidade. Conservação da Biodiversidade. Distance sampling. Detectabilidade imperfeita. Impactoantropogênico. Ocupação.

LIST OF FIGURES

LIST OF TABLES

CONTENTS

GENERAL INTRODUCTION

Tropical ecosystems are the richest in species on the planet, while at the same time the most endangered (Myers, 1988). The anthropic influence on the environment has been increasingly intense, resulting in what some researchers call the "crisis" of biodiversity, a reference to extinctions driven by various factors related to human activities (Di Minin et al., 2016). Current extinction rates resulting from anthropogenic activities are unprecedented and highly unusual in Earth's history (Ceballos et al., 2015). Habitat fragmentation, the overexploitation of natural resources, and the introduction of invasive species are inter-related impacts associated with the human occupation of natural landscapes, and comprise the main threats to the planet's biodiversity (Hoffmann et al. 2011). The need for new areas for agriculture, livestock and logging is the main reason for the elimination or alteration of native vegetation cover, which results in the loss of other life forms, animals, fungi and microorganisms (Whitmore, 1997; Foley et al., 2005). It is estimated that 85% of forested areas cleared from 1981 to 1990 aretropical rainforest (Whitmore, 1997; Hansen et al., 2013).

Biodiversity hotspots are designated priority conservation regions that exhibit simultaneously high rates of species diversity, endemism and degree of threat (Myers et al. 2000). Altogether there are 35 recognized hotspots on the planet that cover only 2.3% of the earth's surface, but are endemic to 50% of all vascular plant species and 42% of all known vertebrates (Mittermeier et al., 2004; Mittermeier et al., 2011). The Atlantic Forest is considered as one of the five most diverse hotspots (Mittermeier et al., 2004; Mittermeier et al., 2011), being a complex set of ecosystems that houses a significant portion of Brazil's biological diversity. With an original area of approximately 1,300,000 km² in Brazil, which corresponds to 15% of the national territory, the Atlantic Forest biome encompasses 15 Brazilian states, extending from Rio Grande do Sul to Piauí, and houses about 61% of the Brazilian population (SOS Mata Atlântica, 2014). The Atlantic Forest is currently restricted to less than 26% of its original forest cover (SOS Mata Atlântica, 2014; Rezende et al., 2018) and aboutof80% is represented by fragments smaller than 50 ha (Ribeiro et al., 2009).

Mammals are among the most studied vertebrate taxa and yet new species are still being discovered and described (Mackinnon, 2000). Although the draw of mammal research may stem from their charismatic status (Clucas et al., 2008) or their taxonomic affinity, the ecological importance of the ~4,600 species is still often poorly understood. Mammals, particularly large carnivores and herbivores, are often regarded as keystone species, occurring

at relatively low densities but serving in a critical role to maintain balanced community structure (Mills et al., 1993). With an increasing human population, the encroachment on natural habitats has caused an increase in human-wildlife conflict and in many cases large carnivores and herbivores have been persecuted. Medium and large mammals may also serve as good indicator species of forest integrity because of their large home range and territory requirements. Some mammals prefer disturbed or open habitat, while other species are vulnerable to such disturbance. The community structure of forest patches may serve as an index of forest disturbance and degradation. It is important to examine all aspects of mammalian community structure, particularly in fragmented Neotropical landscapes where large carnivores and herbivores may be extirpated due to habitat loss, fragmentation, hunting and poaching. The primary limitations to studying medium and large Neotropical mammals are detectability issues associated with their low population densities, elusive nature, and dense vegetation in forests (Long et al., 2008; O'Connell et al., 2011). Recent noninvasive survey techniques, such as camera traps and line-transects, are being used to better detect medium and large mammals (Buckland et al., 2001; Gompper et al., 2006; Long et al., 2008; O'Connell et al., 2006; 2011; Tobler et al, 2008).

Hunting can be considered as a cultural trait strongly rooted in the Neotropics in several aspects. Hunting may affect mammalian populations (Chiarello, 2000; Peres, 2000b; Crawshaw et al., 2004), and change communities (Peres, 1990, 2001; Naughton-Treves et al., 2003), although it tends to be underestimated (Redford, 1992), due to lack of standardization and difficulties for detection (Peres et al., 2006). This occurs both in areas where there is anthropogenic habitat disturbance (Daily et al., 2003; Naughton-Treves et al., 2003), and in areas with little or no forest change (Redford, 1992; Peres, 1996; Peres and Lake, 2003), including within protected areas (Chiarello, 2000a; Altrichter and Almeida, 2002; Olmos et al., 2004). Most of the hunted species are frugivorous and/or herbivorous (Peres, 2000a, 2000b; Townsend, 2000), and they play an ecological role in the dynamics of natural environments (Dirzo and Miranda, 1991; Wright et al., 2000; Stoner et al., 2007).Mammals can compromise important ecological processes for the maintenance of forest structure and species composition (Dirzo and Miranda, 1991; Wright et al., 2000; Dirzo et al., 2014), reducing the biodiversity in the long term (Terborgh, 1992, 2000). Extirpation of species tends to compromise the ecosystem functionally and may result in the depletion of forest environments (Harrison, 2011). Population reduction of top-predators (e.g. *Panthera onca* and *Puma concolor*), due to systematic killing by hunting (Crawshaw et al., 2004), may result in increased prey species density, promoting alteration of community structure and overexploitation of resources by

herbivores that previously had their populations controlled by these predators (Terborgh et al., 2001).

Abundance and density estimates are crucial for population biology studies and wildlife monitoring. Knowledge about species abundance and density allows comparison between different regions where estimates for these parameters already exist. With long-term monitoring of populations in a given area, it is possible to assess whether there has been a population increase, decline or stability (Soulé, 1986; Glanz, 1996). In this context, the study of species abundance has a special meaning because it allows us to evaluate a basic characteristic of the populations and, mainly, due to conservation issues related to rare, threatened and low density species, as is the case with some species of medium and large mammals that are supposed to be more susceptible to extinction (Krebs, 1978; Arita et al., 1990).

It has been suggested that distribution area size and density are important factors in determining which management and conservation strategies should be adopted (Arita et al., 1990).Variation in species abundance and density at different sites is mainly attributed to differences in habitat composition and structure (August, 1983; Tews et al., 2004; Goulart et al., 2009; Ferreguetti et al., 2017a) , resource availability (Gautier-Hion et al., 1985; Paredes et al., 2017), predator occurrence (Cody, 1981), dietary plasticity of some species (Peres, 1996), competition for resources (Cody, 1981) and the degree of anthropogenic disturbance such as poaching and deforestation (Peres, 1996;Chiarello, 1999; 2000a; 2000b; Wright, 2003; Ferreguetti et al., 2015; 2016a; 2017a; 2017b).This variation is also attributed to the phenomenon of density compensation, that is, some species are favored in a given environment, because there is an absence of other species that would be potentially competing for food and space (MacArthur et al., 1972; Peres and Dolman, 2000).

In addition to abundance surveys, presence / absence data or abundance indices (i.e. statistical counting that assumes to be related to actual abundance) are generally used in monitoring programs to infer the conservation status of the species of interest. However, when such surveys implicitly assume constant detectability, the statistical count may be biased (Mackenzie et al., 2002). Detectability may vary between species, being affected by various factors besides abundance, such as spatial (e.g. sampling location) and temporal (e.g. seasonality, weather) variables present in a given sampling scale, and between detection methods (Mackenzie et al., 2006). Ignoring imperfection in detectability has generated criticism because the results may be biased and lead to misinterpretations of the parameters of interest, compromising the reliability of monitoring programs in assessing changes in populations over time and space (Yoccoz et al., 2001; Mackenzie et al., 2002; O'Connell et al., 2006; Noon et

al., 2012). Recently, occupancy models have been used to infer the status and dynamics of species distribution (e.g. proportion of occupied sites), incorporating the premise of imperfect detectability (Mackenzie et al., 2002; Long et al., 2011;Karanth et al., 2011). Moreover, such models allow not only to evaluate the spatial and temporal variables that may affect the detectability and occupancy of the species, but also the relative efficiency of different methodological survey procedures to record the occurrence of the species of interest (O'Connell et al., 2006; Nichols et al., 2008).

Therefore, we aimed to quantify the effect of poaching on non-volant mammal species through 5-year monitoring in the Vale Natural Reserve (VNR), northern Espírito Santo state, Brazil. We hoped that the information generated here can contribute to the increase in knowledge of the ecology of neotropical mammals by knowing how species respond to anthropic actions (i.e. poaching) as well as providing information from this group to support environmental policies and actions aimed at conserving biodiversity.

This thesis was divided into three chapters.The first chapter aimed to review published studies up to December 2017 that evaluated the impact of poaching in the Neotropical region to propose a quantitative metric to better evaluate the impact. The second chapter is a case study that evaluated which factors influence the occupancy and detectability of poachers in VNR to assist in the surveillance of protected areas. The first two chapters are published in the international journals, Animal Biodiversity and Conservation and Biological Conservation, respectively.Lastly, the third chapter evaluated the poaching effect on the mammalian community in VNR, specifically on species richness, biomass and occupancy.

STUDY AREA

This study was conducted in the Vale Nature Reserve (VNR), an area of 23,500 ha which belongs to Vale company, located between the municipalities of Linhares and Jaguaré (19º06 '-19º18' S and 39º45 '- 40º19' W), north of the Doceriver and northeast of the state of Espírito Santo (Figure 1).The VNR is adjacent to the Sooretama Biological Reserve (RBS; 24,250 ha) and three other protected areas (Black Mutum; Antas Corner and Cupid Farm and Refuge). Together these protected areas form a continuous block of native vegetation (Linhares / Sooretama block), intercepted by the road BR-101, and represent almost 10% of the remaining forested area in Espírito Santo (SOS Mata Atlântica, 2014).

VNR was formed from a gradual land acquisition process, which began in 1955, when Vale began the process of acquiring the first properties in the region (Figure 1). The company's initial objective was to produce beams to supply the Vitória-Minas Railroad (Jesus and Rolim, 2005).By the end of the 1950s, approximately 60% of the current area was already owned by the company, and subsequent acquisitions were made, which continued the VNR expansion process. Fortunately, initial results from the management of a small area in the 1960s revealed that the project for using timber resources would not be economically viable and the forest was kept as a reserve by Vale (Jesus and Rolim, 2005).The VNR reached its current limits in 1973, consisting of a main block (approximately 98.1% of the total area) and a small adjacent area (complementary fragment located southwest of the main block, called the Ibiribas Reserve) with non-regular contour, resulting from the fusion of 103 properties. In 1978, actions were taken to protect the area (maintenance of fences and firebreaks, prevention and fire fighting, combating logging, hunting, etc.), called Ecosystem Protection, and VNR was officially established as a designated area. biodiversity conservation. In 1998, following the occurrence of a major fire that struck RBS, the protection activities carried out at VNR were also carried out at the neighboring reserve.

Figure1 - Map of the fragmentation status of the state of Espírito Santo, showing the Vale Natural Reserve connected to the Sooretama Biological Reserve.
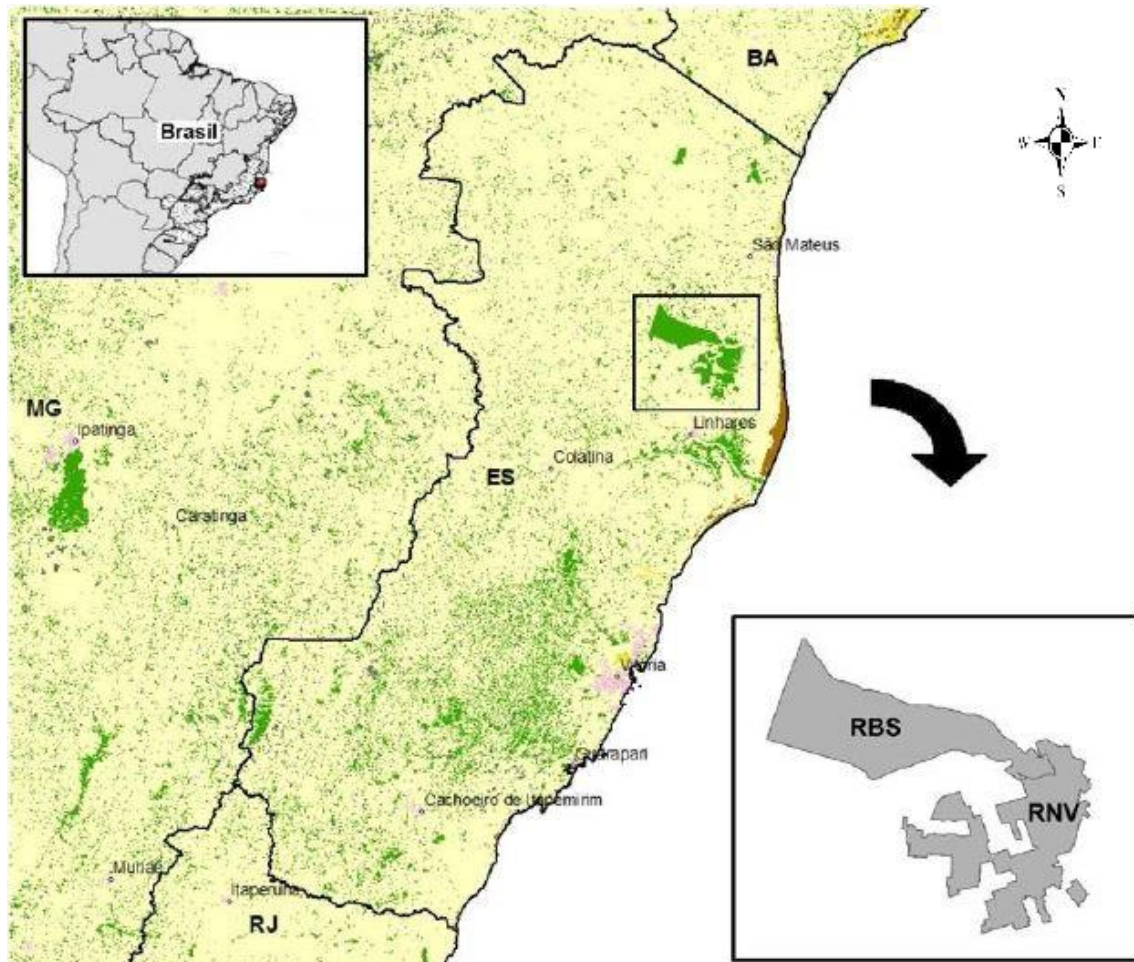
Source: Author, 2014.

The VNR climate is tropical hot and humid (Aw) according to the Köppen classification, with rainy season in summer and dry in winter (Jesus and Rolim, 2005). The average annual temperature is 23°C, with annual rainfall around 1200 mm (Kierulff et al., 2014). However, this rainfall over the last 3 years (i.e. since 2014) remains less than half below the normal average, with values reaching 510 mm per year (Figure 2).

Figure 2 - Historicalclimogramof Vale Nature Reserve (1975-2017), located in Linhares, Espírito Santo.
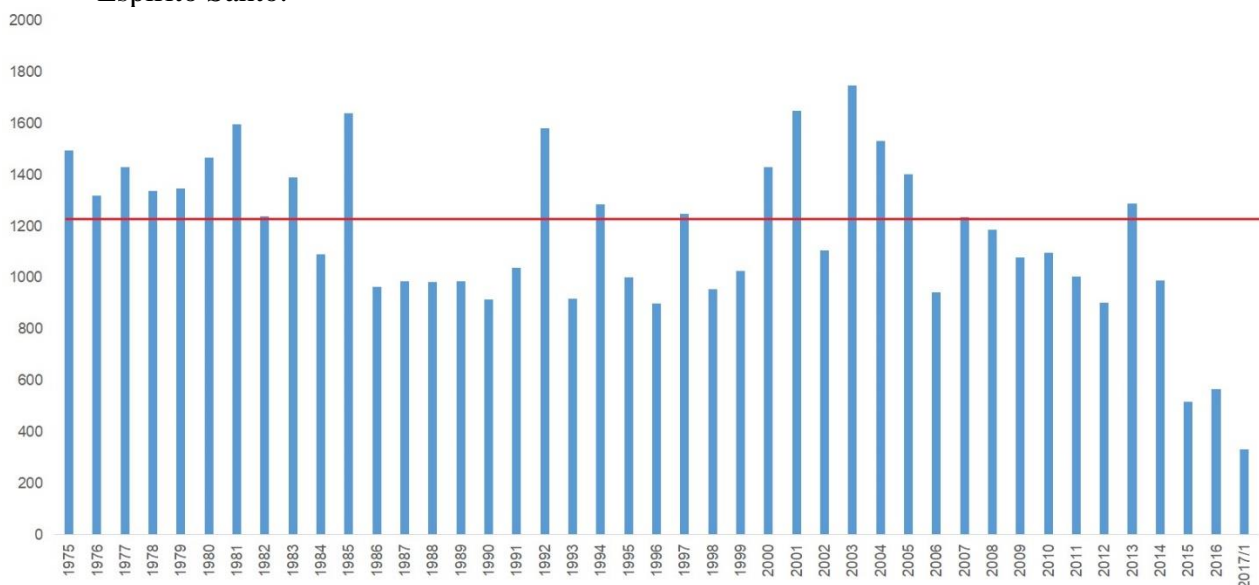
Source: Author, 2017.
Legend: red line highlighting the historical average.

The topography presents a flat relief, with altitudes between 28 and 65 m (Jesus and Rolim, 2005). The hydrography in the reserve consists of a drainage network of tributary streams of the Barra Seca and Pau Atravessado rivers, where the main one for the reserve is the João Pedro stream (Jesus and Rolim, 2005). However, this drainage network has been strongly affected by the construction of illegal dams, in addition to the abnormal drought of the last 3 years.

According to the Brazilian Vegetation Map (IBGE, 1993), VNR is located in the Dense Ombrophylous Forest domains, but according to Jesus and Rolim (2005) it would be classified as Evergreen Seasonal Forest. The reserve is composed of a habitat mosaic formed by four main vegetation types (adapted from Jesus, 1987 and Peixoto and Gentry, 1990): Coastal Plain Forest(always green forest, with two or more upper strata and high occurrence of lianas and epiphytes - approximately 68% of the total area), Riparian Forest (vegetation associated with bodies of water, with mixed appearance, showing sparse trees and predominance of palm trees - about 4% of the Reserve area), Sandy soil forest (tree vegetation over sandy soils, being physiognomically similar to an area at an early or middle stage of regeneration - approximately 8% of VNR) and Natural Grasslands (open sandy fields with herbaceous vegetation, emerging as forest enclaves in areas that constituted lagoons in earlier geological periods - about 6% of the area). In addition to these formations, about 8% of the total VNR area is occupied by

wetlands (swamps) and water bodies (Figure 3). Each of the four vegetation types are exemplified in Figure 4.

Figure 3 - Map of the vegetation types present in the Vale Nature Reserve, and roads that cross the reserve.
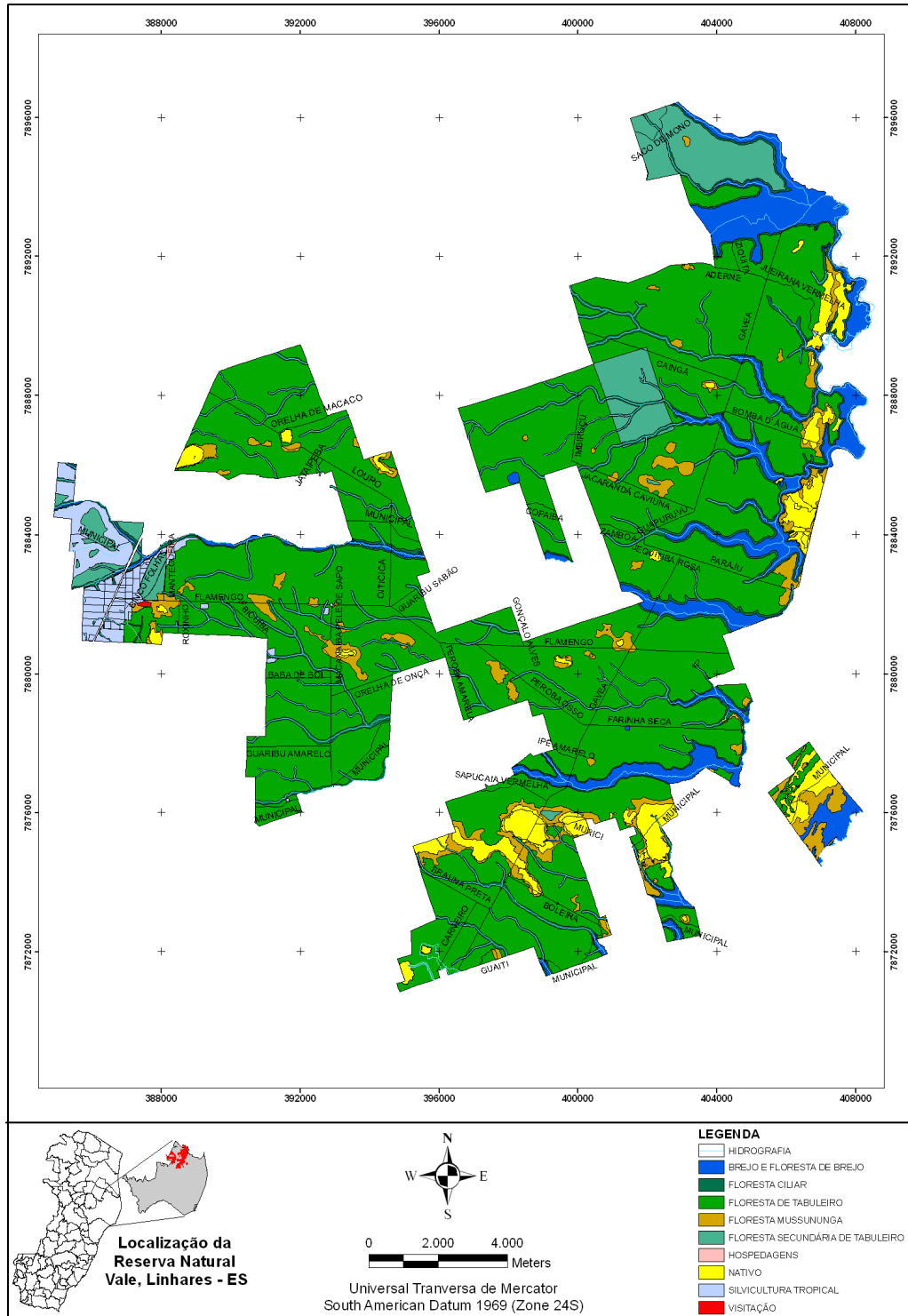
Note: Location of Vale Nature Reserve in Espírito Santo State below.

Source: Author, 2017.

Figure 4 – Vegatation types present in Vale Nature Reserve.

Legend:A – Coastal Plain Forest; B – Sandy Soil Forest; C – Transition between Sandy Soil Forest – Natural Grasslands; and D – Natural Grasslands.
Source: Author, 2015.

1 POACHING IN NON-VOLANT MAMMALS IN THE NEOTROPICAL REGION: THE IMPORTANCE OF A METRIC TO ASSESS ITS IMPACTS

Human activities have deeply changed most of the ecosystems in different regions of the world (Steffen et al., 2015), causing widespread loss of biodiversity (Vellend et al., 2007; Arroyo-Rodriguez et al., 2013; Newbold et al., 2015), changes in community structure (Dornelas et al., 2014), and loss of ecosystem functions and services (Mitchell et al., 2015). Tropical forests are one of the most threatened biomes by human activities, and each year about 13 million hectares of these forests have been devastated in the world (Myers et al., 2000). Exploitation of plant and animal resources in a non-sustainable approach in the natural landscape have led to different impacts such as biodiversity loss, pollution, invasion of exotic species, local extinction of native species (Cardinale et al., 2012), deforestation and habitat fragmentation (Laurance and Biereegaard Jr., 1997; Laurance, 1999). Tourism, hunting, agriculture and livestock practices also affect biodiversity and the survival of species (Cullen et al., 2000). Loss of habitats and hunting of species have been considered as the main threats to the maintenance of non-volant mammal populations (Redford, 1992; Peres, 2001; Milner-Gulland and Bennet, 2003).

Excessive removal of specimens from nature is considered one of the most important threats to the world fauna (Robinson and Redford, 1991; Bennett and Robinson, 2000a; Alves et al., 2012) and different studies show that, in general, the hunting activities in the Neotropics are carried out in an uncontrolled way and therefore impactful, making populations unviable and leading natural resources unsustainable for ecosystem function (Hill and Padwe, 2000; Bodmer and Robinson, 2006; Fernandes-Ferreira et al., 2012). Much of the information on the hunting of mammals in natural environments is generally focused on one or few species. In addition, this information is not standardized through a general protocol, being dispersed by different locations or regions that have different environmental structures. This lack of standardization prevents to detect trends or patterns concerning biomes that would be under higher pressure, and even quantify what are the rates and level of the hunting impact.

We performed a review based on the set of information published in scientific journals on hunting in non-volant mammals in the Neotropical region. The literature review performed

in this analysis had aimed to evaluate the use of metric assessment and the impact of hunting in different study sites of the Neotropical region. Based on this aim, we seek to answer the following questions: i) Which are the biome that have more studies evaluating the hunting impact on mammals? (ii) How many studies evaluate the events and classified on illegal or subsistence hunting? iii) Which metric assessment was used to evaluate the hunting impact in each study? iv) Has the metric assessment used to test the impact of hunting produced a statistically significant result? And (v) Can hunting records help to build a metric assessment to monitor impact of hunting?

Three electronic databases were used for the search of scientific literature: ISI Web of Science, Google Scholar and Scielo. The search terms used were entered the categories "title, abstract and keywords" and "Topic" (TS). The search was based on seven sets of keywords, equally applied to the three databases. The main set referred to variations in hunting terms (impact studied) and included "Hunt *" OR "poach *" OR "bushmeat". The main set was crossed separately with five other sets referring to the object of study (mammals) and locality (Neotropical Region) through the Boolean operator AND: ("mammal*") AND ("Neotropic*"). We restrict our search to articles published in three languages: English, Portuguese and Spanish. We consider only the published literature since 1920 from 20 XII 2017, the date of the search execution.

1.1 **Human and -hunting: contextualizing this interaction**

Since 6 million years ago, wildlife has been a resource to primitive humans (Stanford and Bunn, 2001). Humans have interacted with wild mammal species of many different forms throughout history (Happold, 1995). This has promoted relationships that vary according to different human cultures and which are reflected in negative or positive effects on the wild mammals involved (Leopold, 1959; Bodmer et al., 1997; Alves et al., 2009). In addition, meat has been used as an important protein source, wild mammals have been widely used in the creation of artifacts such as props, or as a source of energy to move or transport, and used as sources of beauty and inspiration, and symbols of gods in religious rituals (Ripple and Perrine, 1999; Alves et al., 2012).

On the other hand, some species are persecuted and killed because they represent risks to human health or domestic livestock such as felines species or by causing losses to farmers such as rodents and some species of medium-sized mammals that feed on product plantations (Treves et al., 2006; Mendonça et al., 2011; Macedo et al., 2015). This ambiguity in the

interaction between human and animals is common in different cultures and depends on the species that are involved (Antonites and Odendaal, 2004; Alves et al., 2012; Alves and Souto, 2015). Indeed, in agricultural societies, hunting involves a dual relationship of familiarity and friendship with domestic animals, and hostility and aggression with the wild and mysterious world (Macedo et al., 2015). Hunting, especially in rural areas, tends to promote a rapprochement or rejection relationship with wild organisms, which tends to be transmitted through generations of human settlement in natural environments.

1.2 Hunting in the Neotropical region

The Neotropical region is the biogeographic region that extends from Central America (including Tropical Mexico) to southern South America and is characterized by significant biotic and climatic diversity (Morrone, 2014). The Neotropical region comprises 78 ecoregions, formed predominantly by tropical and subtropical forests and open formations interrupted by rivers (Morrone, 2014).

Hunting of wild animals occurs throughout the Neotropical region, being carried out by indigenous, rural or urban populations (Becker, 1981; Cullen et al., 2000; Fernandes-Ferreira et al., 2012). Hunting can be considered as a cultural trait strongly rooted in the Neotropics and considered in several aspects, depending on the human community in question and the region considered. The considerable progress in living conditions recorded in the second half of the twentieth century resulted in unprecedented urbanization, as well as an improvement and dynamization of the productive processes of animal protein (meat) and its derivatives. Together with the advancement of the perception of values of the importance of the preservation of natural resources, hunting started to be discussed more intensely in society. Many groups have advocated an unrestricted ban on hunting, especially sport hunting (Leopold, 1959; Collazos et al., 1960; Pierret and Dourojeanni, 1966).

In the Neotropical region, hunting began to be studied in the beginning of the 20th century in order to characterize the activity with a cultural focus (Leopold 1959; Collazos et al., 1960; Pierret and Dourojeanni, 1966). However, it was not until the end of this century that studies began to focus on the hunting impact on wildlife (Bodmer et al., 1988; Paz y Miño, 1988; Peres, 1990). In fact, of the 112 scientific articles reviewed, more than half (66/112, 59%) had the main objective to characterize the hunting activity without any evaluation of the potential impact. Only 58 articles found had as main objective to evaluate the hunting through a metric, be it qualitative or quantitative. Of this total, 38 studies published were carried out in

the Amazon (about 70%), followed by 10 studies in Neotropical Forest in general (17.3 %), eight studies in the Atlantic Forest (13.7%), and the Bolivian Chaco and Brazilian Semi-Arid were represented by only one study each (1.7%) (Figure 5).

Figure5 - The number of scientific publications that evaluated the impact of hunting on non-volant Neotropical mammals.
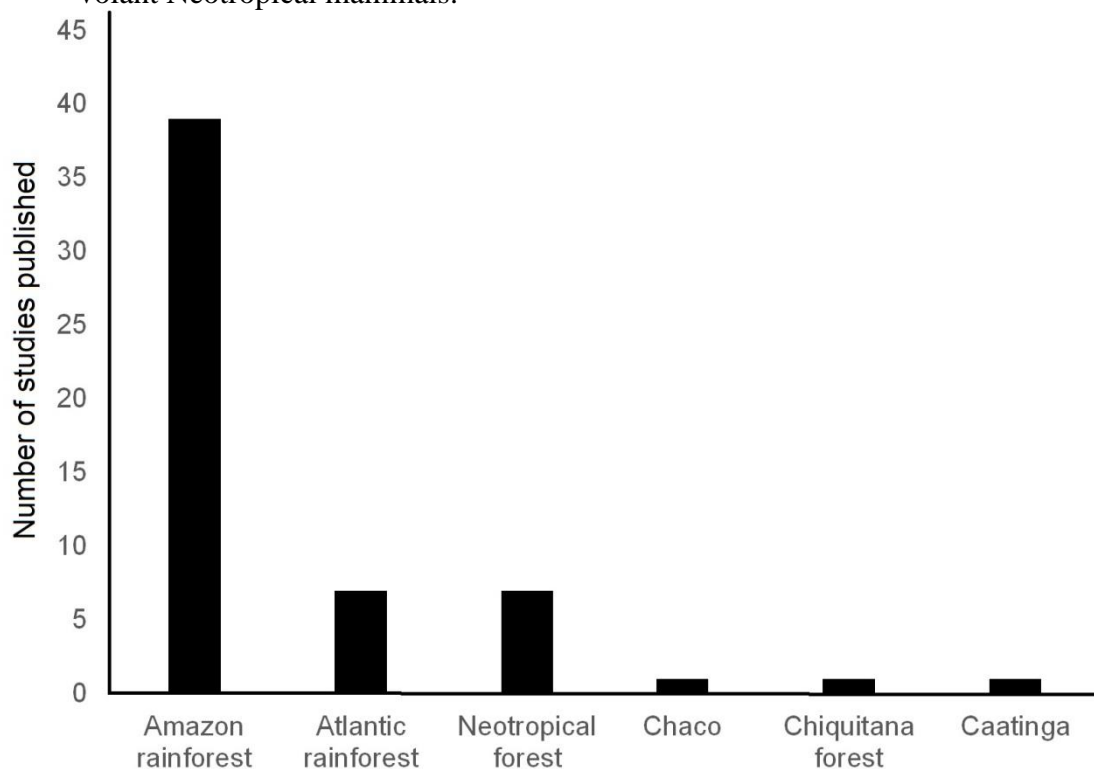
Note: classified by types of environment. This figure was generated based on the 58 scientific publications evaluated.
Source: Author, 2018.

The importance of hunting as a source of animal protein was evidenced in the first reports about the Amazon. In 1864, naturalist Henry Bates described hunts and the habit of local populations along the Amazon River to consume wild animals (Bates, 1864). In fact, many hunting studies with mestizo and indigenous populations have been carried out in the Amazon, especially since 1970. In that decade the availability of protein foods was already discussed as a limiting factor for human groups (Gross, 1975) and the importance of hunting as a source of protein and fat for the Amazon populations (Ayres and Ayres, 1979). The hunting practiced by mestizo and indigenous populations of the Amazon was compared at the end of the 1980s, where biological factors such as density and abundance of species, and cultural factors, such as food and technical restrictions of hunting, were crucial to differentiate between these human

groups (Redford and Robinson, 1987). In the 1990s, some authors suggested that human population growth and settlement age (a supposed index of time to familiarize with the local environment and fauna) could be associated with the negative effects of hunting on vertebrate fauna (Vickers ,1991; Redford, 1992). Since 2000, some aspects related to the sustainability of hunting in tropical forests have been studied (e.g. Bennett and Robinson, 2000a), although most of the studies have addressed subsistence hunting and few have addressed poaching (illegal hunting). Data from the available hunting studies classified the events as subsistence (46, 78%), while 12, 20% classified hunting as illegal and only 1, 2% as legal.

Currently, Amazonian rural communities continue frequently to carry out hunting, although the commercial exploitation of wildlife has become an illegal activity in Brazil since 1967 with the Wildlife Protection Act. (Law No. 5,197, of February 3, 1967). According to this law, hunting was prohibited even for human populations that depended on wildlife for food. Only in 1998, with the advent of the Environmental Crimes Law (Law No. 9,605, of February 12, 1998), subsistence hunting was recognized as a non-criminal activity, provided that "in a state of need to quench hunger agent or his family ". However, this law does not fit the reality in the Amazon region, where the barter of hunting products by primary necessities is characterized as commercial hunting and is therefore considered illegal (Caughley and Gunn, 1996).

1.3 **Mammals and hunting: impacts**

Loss of habitat and species overhunting have been considered the main threats to the survival of species of large forest vertebrates (Redford, 1992; Milner-Gulland and Bennett, 2003; Dirzo et al., 2014). Increased human density (Brook et al. 2006), the growth of access to new technologies (Vickers, 1991; Mena et al., 2000; Stearman, 2000), and the loss of traditional hunting practices (Leeuwenberg and Robinson, 2000; Mena et al., 2000; Stearman, 2000) have promoted the overhunting of populations of Neotropical mammals (Bennett and Robinson, 2000a, 2000b; Silvius et al., 2004). The overhunting of tropical forest vertebrates has led many species population decline (Bennett and Robinson, 2000b), causing local and global species extinctions (Peres, 1990; Ulloa et al., 2004).

Hunting may affect mammalian populations (Chiarello, 2000a; Peres, 2000b; Crawshaw et al., 2004), and change communities (Peres, 1990, 2001; Naughton-Treves et al., 2003), although it tends to be underestimated (Redford, 1992), due to lack of standardization and difficulties for detection (Peres et al., 2006). This occurs both in areas where there is

anthropogenic habitat disturbance (Daily et al., 2003; Naughton-Treves et al., 2003), and in areas with little or no forest change (Redford, 1992; Peres, 1996; Peres and Lake, 2003), including within protected areas (Chiarello, 2000a; Altrichter and Almeida, 2002; Olmos et al., 2004). Most of the hunted species are frugivorous and / or herbivorous (Peres, 2000a, 2000b; Townsend, 2000), whose play an ecological role in the dynamics of natural environments (Dirzo and Miranda, 1991; Wright et al., 2000; Stoner et al., 2007). The overhunting of large forest vertebrates can compromise important ecological processes for the maintenance of forest structure and species composition (Dirzo and Miranda, 1991; Wright et al., 2000; Dirzo et al., 2014), reducing the biodiversity in the long term (Terborgh, 1992, 2000).

Extirpation of species tends to compromise the ecosystem functionally and may result in the depletion of forest environments (Harrison, 2011). Population reduction of top-predators (e.g. *Panthera onca* and *Puma concolor*), due to systematic killing by hunting (Crawshaw et al., 2004), may result in increased prey species density, promoting alteration of community structure and overexploitation of resources by herbivores that previously had their populations controlled by these predators (Terborgh et al., 2001). In addition, often human hunters tend to select for hunting those species that top predators select as prey such as ungulates and rodents species (Leite and Galvão, 2002), and may reduce the capacity of habitat to sustain populations of large carnivores. In the Neotropical region, primates, tapirs and carnivores are particularly vulnerable to overhunting due to their low intrinsic rates of natural growth, high longevity, long generation time, and low population densities (Bodmer et al., 1997; Cardillo et al., 2004). Populations of ungulates and large primates decline as soon as hunting becomes a chronic process (Peres, 2000b).

1.4 **How has the hunting impact been assessed in the Neotropical region?**

One of the most cited hypotheses in the field of Conservation Biology is undoubtedly Kent Redford's 'Empty Forest' (1992). It has been temptingly proposed that extensive forest areas seemingly intact as a result of hunting and a supposed 'defaunation' would present a series of extinct ecological processes. Large species, especially mammals, would have their populations so small that vital functions for the maintenance of ecosystems would be highly affected; thus, in the long term, the preservation of tropical forest vegetation would not be possible if the fauna were not also preserved (Redford, 1992). The question of empty forest has also been evaluated considering the effects of hunting, showing the potential association between hunting and the negative effects that tend to result in an "empty forest" (Harrison,

2011). The most species appreciated by subsistence hunters are generally responsible for ecological interactions that directly influence plant regeneration (Dirzo, 2001; Wright et al., 2007; Terborgh et al., 2008). These interactions include predation of seeds before and after dispersion, primary and secondary seed dispersal, leaf and grass herbivory (Wright et al., 2007). The consequences of deforestation from fauna hunting in forest dynamics include reductions in predation and dispersal of seeds, which may lead to changes in total recruitment of seedlings, composition, and decreasing of the diversity of flora (Dirzo and Miranda, 1991; Terborgh et al., 2008; Dirzo et al., 2014), or even altering carbon stocks in tropical forests (Bello et al., 2015; Kurten et al., 2015).

Many studies that have succeeded the 'empty forest' hypothesis qualitatively compared the impact of hunting on wildlife in areas without hunting or hunting or classified at different hunting intensities. Of these 58 studies evaluated, 39 used a qualitative way to characterize hunting and assess the impact on mammals (Figure 6A). The methods used to characterize the impact of hunting used in 95% of the studies were: 20 studies used hunting intensity classes (low, medium and high) by locality and 18 relied on presence/absence data (i.e. with and without hunting) (Figure 6A). However, by using a qualitative way to evaluate the impact of hunting almost half of the studies did not find statistically significant results (Figure 6B).

Concomitantly, models were developed for measuring quantitatively the sustainability of hunting in tropical areas representing about 33% (19/58) of the studies as shown in Figure 6C (Robinson and Redford, 1991; Robinson and Bennett, 1999; Bodmer and Robinson, 2004). Of those 19 studies that assessed the impact of hunting quantitatively, 11 were for subsistence hunting in the Amazon Forest using the number of slaughtered animals as metric. For subsistence hunting this is possible, because the communities that practice hunting inform the number of individuals that are extracted from nature. On the other hand, to measure illegal hunting this metric can not be applied. Therefore, the eight studies that evaluated poaching used evidence of hunting as an indicator but continued ranking the intensity of hunting. Some studies assume that the density of huntable species in in non-hunting areas would represent a precise estimate of the support capacity in a region, thus concluding the number of individuals an area could harbor (Caughley, 1977; Caughley and Sinclair, 1994). All 19 studies using a quantitative metric found a statistically significant result on the impact of hunting on mammals (Figure 6D).

Figure 6 - Classification of scientific publications evaluating the impact of hunting in non-volant Neotropical mammals.
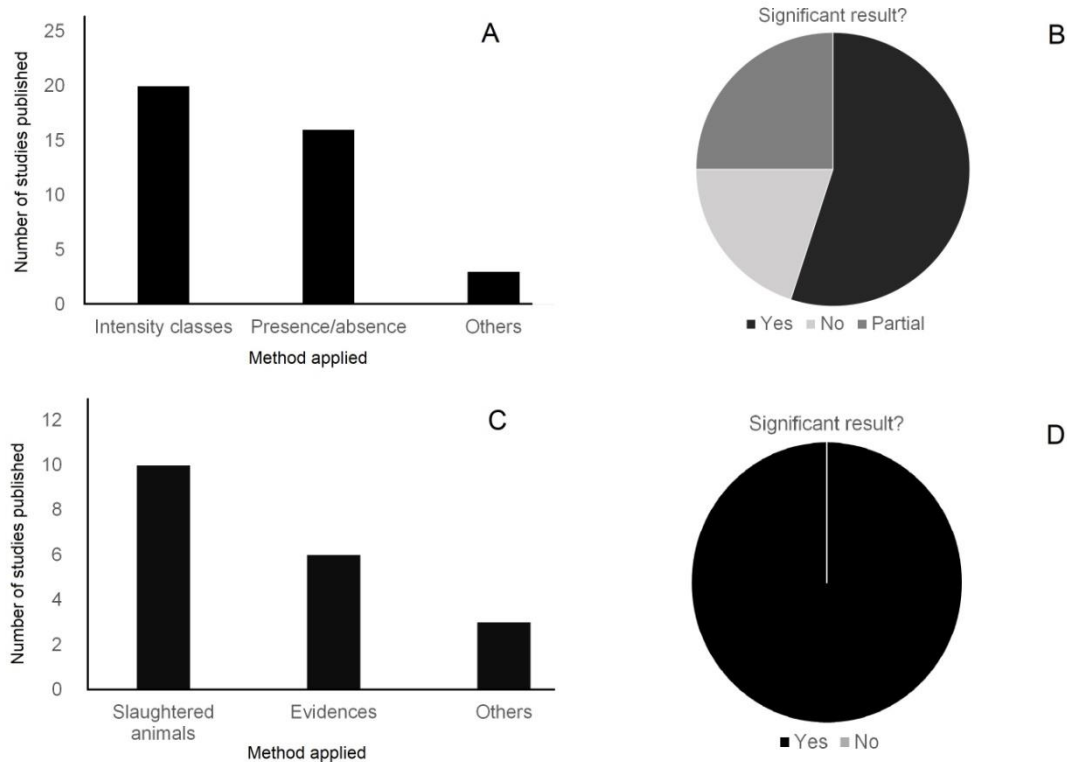
Legend: A) metric estimated by the qualitative method; B) significance of the result found in each study that used the qualitative method; C) metric estimated by the quantitative method and D) significance of the result found in each study that used the quantitative method.
Source: Author, 2018.

1.5 **The importance of a quantitative metric method to detect the poaching impact and the long-term standardized monitoring**

As previously reported, most of the studies that evaluated the impact of hunting were conducted considering subsistence hunting. To quantify the impact of hunting, the number of animals slaughtered (Figure 6C) was used as a metric assessment in most studies. For subsistence hunting this metric may indicate an estimate of how species are being affected (Aquino and Calle, 2003; Peres and Nascimento, 2006; Parry et al., 2009), but for poaching it would not be possible to quantify, since there is no access to the actual number of animals slaughtered. Therefore, there is the challenge of quantifying the impact of illegal hunting. Few studies have used hunting evidence as a metric to quantify impact (Chiarello, 2000a; Wright et al., 2000), but these have used this evidence as a general value, not considering that such evidence was not spatially distributed uniformly. The temporal distribution of this evidence was also not considered. The studies found, although using a quantitative metric, are not done in a

standardized way that allows comparison between different Neotropical regions. This highlights the importance that in addition to a quantitative metric, it is necessary to have a minimum of possible standardization that allows to have a bigger picture of the impact on the mammals.

In this context, a quantitative metric assessment is proposed, which considers the spatial distribution of hunting evidence per km^2 and allows monitoring the trends of this impact over time (Ferreguetti et al., 2015, 2016b, 2017b). This metric will collect the evidence of illegal hunting in a standardized way over the time. The metric can be generated by considering each poaching event separately (date, reserve where the event was recorded, location/region of the event, geographic coordinates and type of evidence collected). Any evidence of hunting can be georeferenced over time. Examples that can be considered as evidence of hunting to georeferenced:

- hunting elements found such as traps or baited sites: leg-hold traps, snare traps, crushing or weight traps, fall-and-apprising traps ("arapucas"), cage traps, cartridges and archery traps, corral, pitfall, among others kind of traps.

- Direct evidence of hunters such as encounters, slaughtered animals, camps, among others. Together with this georeferenced database, it is recommended to use the poacher's records by using camera traps to calculate the metric.

Based on the construction of this database of georeferenced hunting events it is possible to calculate a quantitative metric that consists of dividing the study area into 1-km^2 grids by positioning on a digital map of the target Reserve and identifying sample sites by each area size. For example, if a Reserve has 100 km^2, will result in 100 grids with an intensity of hunting events per km^2.

Moreover, it is important to avoid counting the same record twice by removing the evidence found. Monitoring should be done on a regular basis, not exceeding a period of three months without monitoring. The metric proposed can be carry out in protected areas and can still rely on the population for a monitoring performance through training on how to gauge this georeferenced data and how to pursue conservation actions to mitigate the impact of hunting in mammal species.

2 ONE STEP AHEAD TO PREDICT POTENTIAL POACHING HOTSPOTS: MODELING OCCUPANCY AND DETECTABILITY OF POACHERS IN A NEOTROPICAL RAINFOREST

Wild animal hunting, widely spread in tropical forests, has great nutritional, economic, social and cultural importance for local communities (Bennett and Robinson, 2000b). However, when there is no adequate management of the exploited set of species, it can become a consistent predatory activity (Redford, 1992). Illegal hunting, hereafter referred to as poaching, also feeds the international traffic of wild animals (RENCTAS, 2001). Each year, 5-20 billion dollars are moved through the illegal trade of fauna and flora (Rosen and Smith, 2010). Importantly, in addition to direct impacts caused by the removal of specimens from natural environments, poaching has caused the emergence and spread of diseases and facilitated biological invasions, threatening the health of humans, native species, and agricultural activities (Karesh et al., 2005; Rosen and Smith, 2010). Because it is a widely disseminated and highly relevant activity for tropical forest conservation, poaching has attracted the attention of researchers in recent decades (Redford, 1992; Bodmer et al., 1997; Peres, 2000b; Fa et al., 2002; Brashares et al., 2004; Corlett, 2007; Parry et al., 2009; Parry and Peres, 2015; Sousa and Srbek-Araujo, 2017).

Populations of animals can be affected by poaching in different ways, including population size reduction (Peres, 1996; Peres and Nascimento, 2006; de Souza and Alves, 2014) and changes in behavior (Ferreguetti et al., 2017b). Local extinctions of vertebrates caused by both subsistence hunting and habitat loss and fragmentation have already been reported in the literature (Bodmer et al., 1997). Mammals, which have a relatively longer life span, lower population growth rate, and long generation time, tend to be highly susceptible to local extinction by poaching (Bodmer et al., 1997). Species such as the lowland tapir (*Tapirusterrestris*) and the white-lipped peccary (*Tayassupecari*), for example, were locally extinct in areas of Mexico, Panama, and Brazil, both in the Amazon region and in the Atlantic Forest (Peres, 2001). In 101 areas of the Brazilian Amazon, populations of vertebrates moderately or intensely exploited by hunting had their densities reduced by about 90% (Peres, 2000b; Peres and Palacios, 2007; Wilkie et al., 2011). In the Brazilian Atlantic Forest, the

abundance of medium and large vertebrates was on average 37% lower in intensely hunted areas compared to lightly hunted areas (Cullen et al., 2000). Vertebrate defaunation caused by hunting, poaching and illegal trade in tropical forests has even been considered comparatively more harmful than deforestation (Fa et al., 2002). Through cascading effects, the abrupt decline or even extinction of populations probably have had pervasive impacts on ecosystems structure and dynamics (Dirzo et al., 2014).

The rapid and continuous increase in human population density nearby forest ecosystems, especially in tropical regions, have increased access and facilitated harvesting, hunting and poaching activities (Peres and Lake, 2003; Laurance et al., 2009). In Congo's forests, for example, human population growth as a result of the setting up of logging companies and the expansion of highways, have further boosted the extraction and sale of wild animal meat in the region's markets (Wilkie et al., 2000). Due to its long history of occupation and destruction after European colonization (Dean, 1996), the remaining fragments and the few Reserves in the Atlantic Forest biodiversity hotspot become easily accessible by poachers. Indeed, 73% of the remaining Atlantic forests are located less than 250 m from non-forest areas (Ribeiro et al., 2009).

Unfortunately, poaching is still widespread even within protected areas (Cullen et al., 2000; Sousa and Srbek-Araujo, 2017), partly due to the high cost of monitoring and combating poaching over large areas. To better focus conservation efforts within protected areas, it is still necessary a better understanding of the spatial distribution of poaching activities. The scarceness of quantitative information on poaching distribution is partly explained by the fact that poaching activities occur under the forest canopy, which hampers its monitoring by conventional remote sensing techniques (Peres et al., 2006; Ahmed et al., 2014). Instead, investigation of poaching patterns often requires *in situ* techniques that can compromise both the spatial extent and the number of survey repetitions (Parry and Peres, 2015), as well as result in an increased cost of the research activity. Therefore, studies are urgently needed to predict potential poaching areas and to identify the set of conditions that favor this activity, to maximize the efficiency of surveillance efforts and to better support the decision-making process aiming at wildlife protection.

In this study, we show how occupancy modeling may be applied to produce a spatially-explicit diagnostic of poaching in protected areas. Occupancy modelling provides an approach for estimating the probability of occupancy of a site by an organism of interest (in our case, poachers) while accounting for imperfect detection, based on data collected at repeated visits to multiple sites (Bailey et al., 2014; Mackenzie et al., 2002). Given the difficulties of obtaining

abundance estimates of a given species from count- or detection/non-detection data, researchers often avoid using density estimation altogether, adopting occupancy (i.e. the probability that a site is occupied by a species), in combination with detectability (i.e. the probability that a species is detected given that it is present), as alternative parameters of interest (Burton et al., 2015). Occupancy modeling is considered to be a more cost-effective alternative for the monitoring of populations in comparison with other methods (Gerber et al., 2014; Gray et al., 2010; 2014) but has not been applied yet to estimate poacher's occupancy. Here, we show that occupancy modeling may be useful to estimate poaching hotspots in protected areas, and to determine how different covariates could affect poacher's occupancy and detectability, which reflect poacher's occurrence and abundance, respectively. Detectability is treated as a proxy for intensity of poachers (i.e. poachers will be more easily detected in places they go more often). We tested the general hypothesis that poaching occurrence and intensity is higher in sites with higher accessibility by poachers. We predicted that poacher occupancy and detectability would be higher in sites closer to Reserve edges, human settlements, trails, roads and water resources. We also predicted that poacher detectability would be higher in nights with higher lunar light intensity, which would facilitate poacher's movement, and in sites with higher frequency of prey. The innovative methodological approach used here, as well as the findings of our empirical study, may advance understanding and combating poaching activities in different protected areas.

2.1 **Material and Methods**

2.1.1 Camera trapping.

To model occupancy and detectability of poachers and their distribution in the VNR, we conducted a camera-trapping study along a 13-month period (April 2013 to May 2014). The camera-traps operated for a total of 180 days during this period. We placed a grid over a digital map of the Reserve and identified the sampling sites by selecting, based on this grid, a number of trap stations proportional to the area occupied by a given vegetation type(Figure 7). We placed a grid over a digital map of the Reserve and identified the sampling sites by randomly selecting grid cells. We deployed the camera-trap in the center of each selected grid cell. The sites were separated from one another by more than 1 km. At each site, we installed one passive infrared Bushnell® camera trap, approximately 40–50 cm above the ground. All cameras were checked every 20-25 days to replace depleted batteries. Traps were programmed to operate for

24 hours/day. We classified as poachers all individuals photographed carrying a weapon and/or some slaughtered animals.

Figure7- Habitat mosaicinsidethe Vale Natural Reserve, Espírito Santo, Brazil.
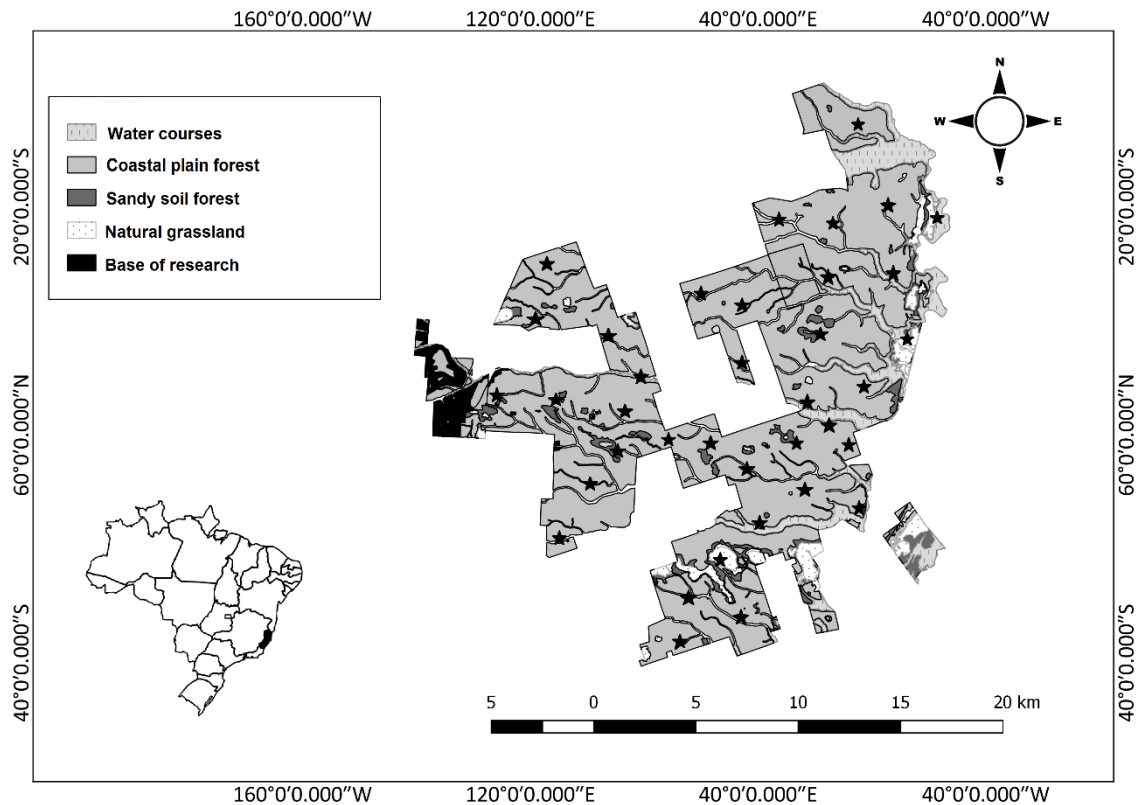
Legend: Black stars represent the location of the 39 camera traps.
Source: Author, 2018.

2.1.2 Historical Poaching records

The VNR has an environmental surveillance system (locally called Ecosystem Protection) responsible for the protection of the area against threats (e.g., poaching, palm harvesting, logging, and forest fires). Poaching data were generated by the Ecosystem Protection team in the entire VNR considering each poaching event separately (date, location/region of the event, geographic coordinates and type of evidence collected). We used an index of poaching intensity (records/ 2-km^2) within the study area to compare our estimates of poaching hotspots. This index was calculated using the georeferenced database of 14 years of poaching records collected

by the Reserve's surveillance system. The index was calculated for each grid square to compare with occupancy estimates.

2.1.3 <u>Model Covariates</u>

We selected covariates based on previous studies on both poaching and habitat selection by more frequently poached species in the region. A total of seven covariates were quantified to assess their influence on poaching site selection. We used five covariates of accessibility to model both occupancy and detectability, namely the distance from the nearest: Reserve edge (edge), water resource (water), road (road), human settlement (human), and trail (trail). Two other covariates, lunar light intensity (moon) and the frequency of records of game species (game_sp), were also used to model detectability, as they could affect the activity of poached species and poachers.

All distance covariates (edge, water, road, human and trail) were calculated as a straight line (Euclidean distances) using QGIS software (QGIS Development Team, 2017). To access lunar light intensity, we used Moonphase SH software (version 3) to classify moon phase of records, according to the percentage of visible moon surface, as follows: 0-25% (New Moon), 25-75% (Waxing and Waning Moons) and 75-100% (Full Moon). For the frequency of record of game species, we summed the total records obtained for each camera-trap of the following species: lowland paca (*Cuniculus paca*), three armadillos' species (*Cabassoustatouay*, *Dasypusnovemcinctus*, and *Euphractussexcinctus*), red and grey brocket deer (*Mazama americana* and *M. gouazoubira*), lowland tapir (*Tapirusterrestris*), and collared and white-lipped peccaries (*Pecaritajacu* and *Tayassupecari*). These species were selected according to previous studies that detected a negative effect of poaching in the VNR (Kierulff et al., 2014; Ferreguetti et al., 2015; 2016b; 2017b; Sousa and Srbek-Araujo, 2017). Correlations between all covariates were relatively low (Pearson correlation coefficients < 0.6; Table 1).

Table 1 - Pearson correlation matrix of the covariates used in the occupancy modelling of the Poachers in the Vale Natural Reserve, Espírito Santo, Brazil.

	edge	water	road	human	trail	moon	game_sp
edge	1						
water	0.12	1					
road	0.35	0.15	1				
human	0.31	-0.13	0.34	1			
trail	0.09	0.31	0.09	-0.09	1		
moon	0.02	-0.14	0.23	0.11	-0.12	1	
game_sp	-0.17	0.56	-0.24	-0.18	0.22	0.34	1

Covariates include: distance from the nearest Reserve edge (edge), distance from the nearest water resource (water), distance from nearest road (road), distance from the nearest human settlement (human), and distance from nearest trail (trail); lunar light intensity (moon), and the frequency of records of game species (game_sp).

2.1.4 Data analysis

Occupancy modeling requires temporally- and spatially-replicated data to account for imperfect detection, thus allowing to estimate the probability of detecting a species given that it was present at a site during sampling (MacKenzie et al., 2006). We discretized our camera-trapping data into 26 sampling intervals of seven consecutive days to build a reliable detection history (MacKenzie et al., 2006). We estimated both site occupancy (Ψ) and detectability (p), modeling our observations with three possible outcomes: (1) the site was occupied, and a poacher was detected ($\Psi \times p$); (2) the site was occupied but no poacher was detected ($\Psi \times [1-p]$), and (3) the site was unoccupied ($1-\Psi$), and therefore no poacher was detected.

Using a single-species, single-season occupancy model, we evaluated seven covariates that might influence occupancy and detectability, to identify and compare potential sites selected by poachers. Given the large number of covariates we were interested in, we split the analysis into two components: 1) determining the 'best-fit' model for detectability while holding occupancy constant, and 2) determining the 'best-fit' model for occupancy while modeling detectability as determined by the 'best-fit' model in component 1, above. This allowed us to evaluate differences in potential sites selected by poachers as determined by a single covariate or a set of covariates, which would contribute to an improvement in the model's performance.

We built our single-species, single-season occupancy models using the UNMARKED package in R environment (Fiske and Chandler, 2011; R Development Core Team, 2016). Top-ranked models were selected using Akaike's Information Criterion adjusted for small sample sizes (AICc), and all models with a ΔAICc value < 2 were equivalent. We also used the model weight (AICcw), which corresponds to the amount of evidence in favor of a given model, to choose our 'best-fit' model, which we subsequently used to test our specific hypotheses.

For the best (top-ranked) occupancy model, we calculated probability of occupancy for each 2-km^2 grid cell in the study area using the model-specific coefficient estimates and covariate information for each grid cell. The occupancy estimates predicted by the top-ranked model resulted in a map of predicted poacher occurrence in the Reserve. We then applied the heat map technique for hotspot identification (Rosenblatt, 1956; Parzen, 1962). The heat map allows for the rapid identification of hotspots by weighting the value of an interest variable within a moving window that moves across the entire study area. We built this map using QGIS (QGIS Development Team, 2017) with the Heat Map plug-in using the centroids of the regular grid as input. We then compared our predictions to the historical poaching records in the Reserve by using two correlation tests: (1) between only the 39 sites where we had a camera-trap installed, and (2) between all the 118 2-km^2 grid cells. Pearson's correlation coefficients were calculated as a measure of spatial congruence between historically- and occupancy-estimated poaching.

2.2 Results

A total of 7,020 trap-days was conducted during the study, with poachers being observed in 16 of the 39 sites. Considering the 20 most plausible occupancy models (Table 2), the best covariates to predict occupancy by poachers were distance from water resources and distance from forest edges. To predict detectability, the covariates that most influenced poachers were: distances from water resources, from Reserve edges, and from human settlements; the lunar light intensity; and the frequency of record of game species.

Table 2 - Single-season occupancy and detectability models for Poachersinthe Vale Natural Reserve, Brazil, estimated using camera trap data between May 2013 and June 2014, grouped in sampling intervals of 7 consecutive days (to be continued).

Model	AICc	ΔAICc	AICcw	nºparameters
Ψ(water);p(moon;edge;water;human_settle;game_sp)	693.45	0	0.34	8
Ψ(edge);p(moon;edge;water;human_settle;game_sp)	693.86	0.41	0.28	8
Ψ(water;edge);p(moon;edge;water;human_settle;game_sp)	695.12	1.67	0.25	9
Ψ(water;trail);p(moon;edge;water;human_settle;game_sp)	697.32	3.87	0.06	9
Ψ(water;human_settle);p(moon;edge;water;human_settle;game_sp)	698.89	5.44	0.03	9
Ψ(water;edge;trail);p(moon;edge;water;human_settle;game_sp)	698.95	5.5	0.02	10
Ψ(edge;trail);p(moon;edge;water;human_settle;game_sp)	700.12	6.67	0.01	9
Ψ(.);p(moon;edge;water;human_settle;game_sp)	701.34	7.89	0.001	7
Ψ(water;edge;human_settle);p(moon;edge;water;human_settle;game_sp)	702.89	9.44	0.001	10
Ψ(water;game_sp);p(moon;edge;water;human_settle;game_sp)	703.82	10.37	0.001	9
Ψ(water;edge;game_sp);p(moon;edge;water;human_settle;game_sp)	703.84	10.39	0.001	10
Ψ(water;trail;game_sp);p(moon;edge;water;human_settle;game_sp)	704.05	10.6	0.001	10
Ψ(water;edge;trail;game_sp);p(moon;edge;water;human_settle;game_sp)	705.32	11.87	0.001	11
Ψ(edge;game_sp);p(moon;edge;water;human_settle;game_sp)	705.79	12.34	0.001	9
Ψ(edge;human_settle);p(moon;edge;water;human_settle;game_sp)	707.04	13.59	0.001	9
Ψ(edge;trail;game_sp);p(moon;edge;water;human_settle;game_sp)	707.89	14.44	0.001	10
Ψ(water;road);p(moon;edge;water;human_settle;game_sp)	708.31	14.86	0.001	9

Table 2 - Single-season occupancy and detectability models for Poachers in the Vale Natural Reserve, Brazil, estimated using camera trap data between May 2013 and June 2014, grouped in sampling intervals of 7 consecutive days (continuation).

Model	AICc	ΔAICc	AICcw	n°parameters
Ψ(edge;game_sp_human_settle);p(moon;edge;water;human_settle;game_sp)	709.11	15.66	0	10
Ψ(water;edge;road);p(moon;edge;water;human_settle;game_sp)	710.23	16.78	0	10
Ψ(edge;trail;game_sp);p(moon;edge;water;human_settle;game_sp)	710.27	16.82	0	10

Legend: Covariates include distance from the nearest Reserve edge (edge), distance from the nearest water resource (water), distance from nearest road (road), distance from the nearest human settlement (human), and distance from nearest trail (trail); lunar light intensity (moon), and the frequency of records of game species (game_sp).

The distance from water resources had a negative relationship with occupancy and detectability, with lower occupancy and detectability at longer distances (Figure8A; Figure9A). Similarly, the distance from Reserve edges also had a negative relationship with occupancy and detectability (Figure8B; Figure9B). The frequency of record of game species had a positive relationship with poacher detectability (Figure9C), and the distance from the nearest human settlement was negatively related to detectability of poachers (Figure9D). The lunar light intensity also had a positive relationship with detectability of poachers (Figure9E).

Figure8 - Estimated effect of covariates and occupancy of poachers in the Vale Natural Reserve.
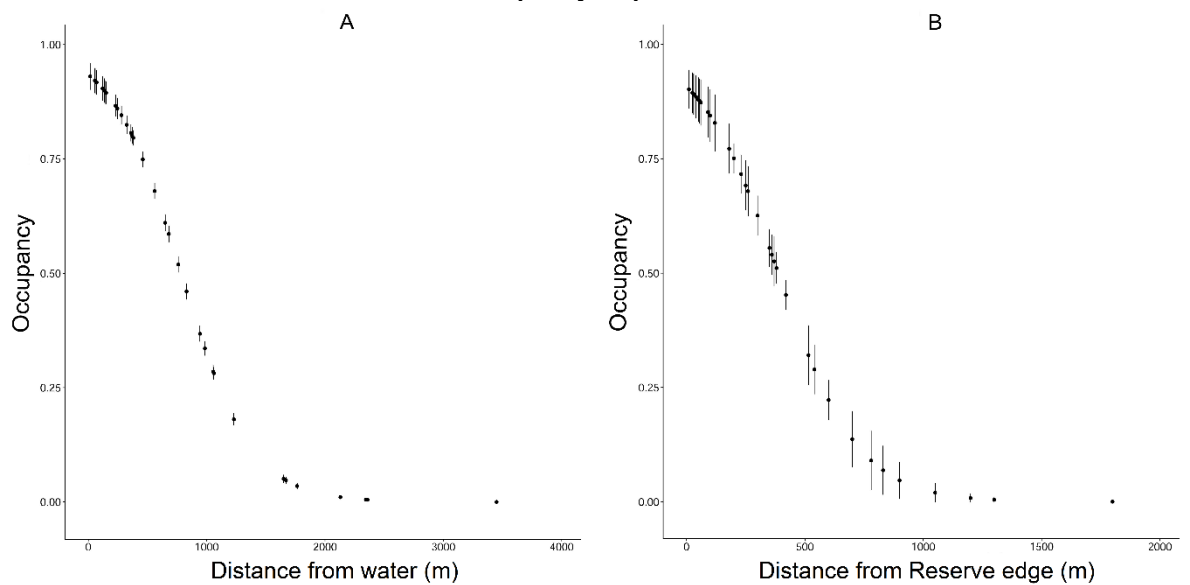
Legend: (A) distance to water resources and (B) distance to forest edges
Source: Author, 2018.

Figure 9 - Estimated effect of covariates on the probability of detecting a poacher at a site.
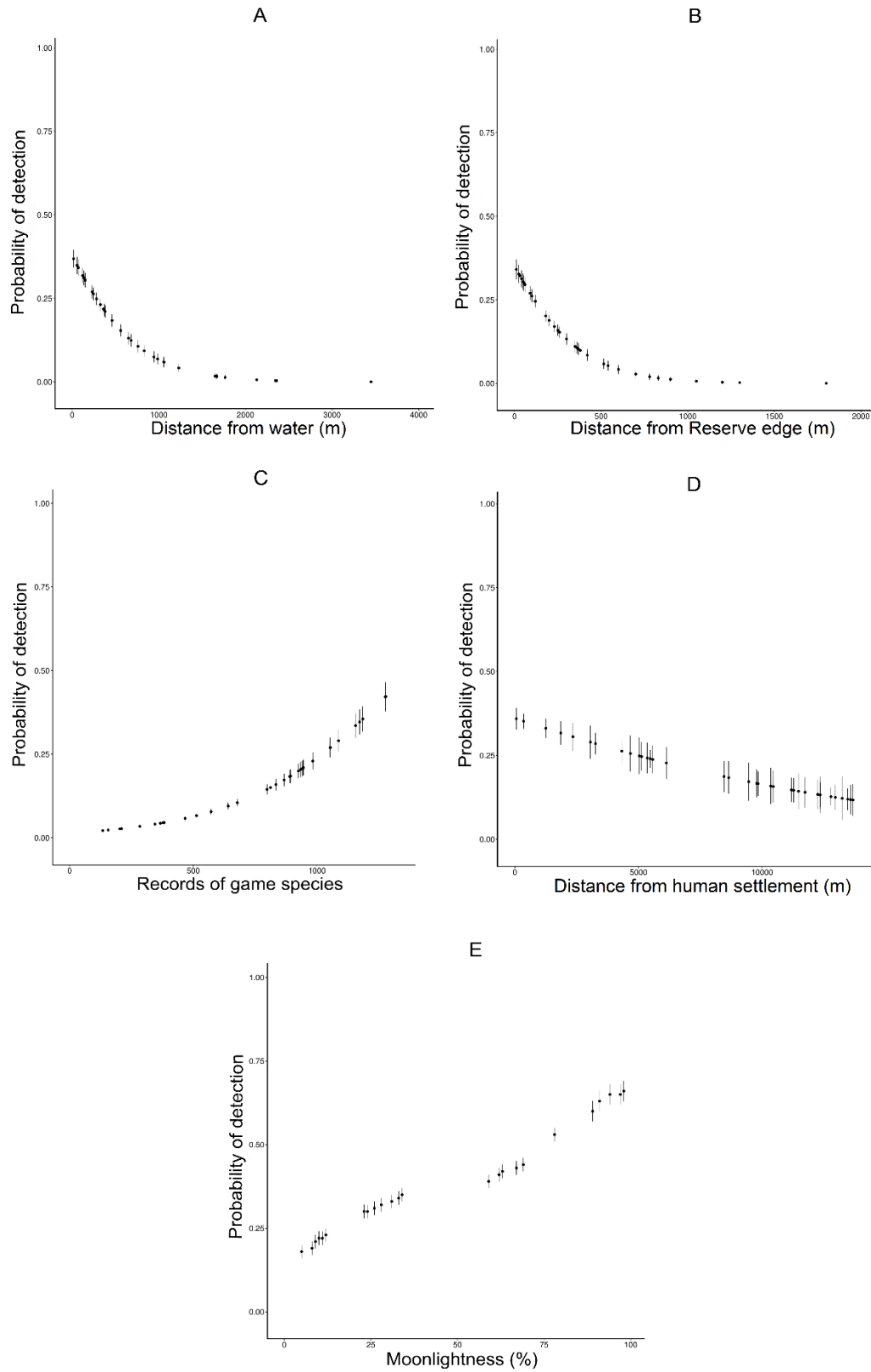
Legend: (A) distance to water resources, (B) distance to forest edges, (C) frequency of record of game species, (D) distance to human settlements, and (E) moonlightness (%).
Source: Author, 2018.

The resulting map of these estimates showed that the intensity of poaching varied consistently among areas of the Reserve, being higher in areas with higher accessibility by poachers (Figure10). The overall patterns of distribution estimated from models were similar to the distribution of historical poaching surveillance data for the VNR (Figure11). Accordingly, there was a significant correlation between historically- and occupancy-estimated poaching, considering either only the 39 camera-trap sites (r=0.81; P≥0.01) or all 118 grid cells (0.76; P≤0.01).

Figure10 - Map with estimates of top-ranked occupancy model (Ψ(water);p(moon;edge;water;human_settle;game_sp)) that predicted poaching spatial distribution for the Vale Natural Reserve.
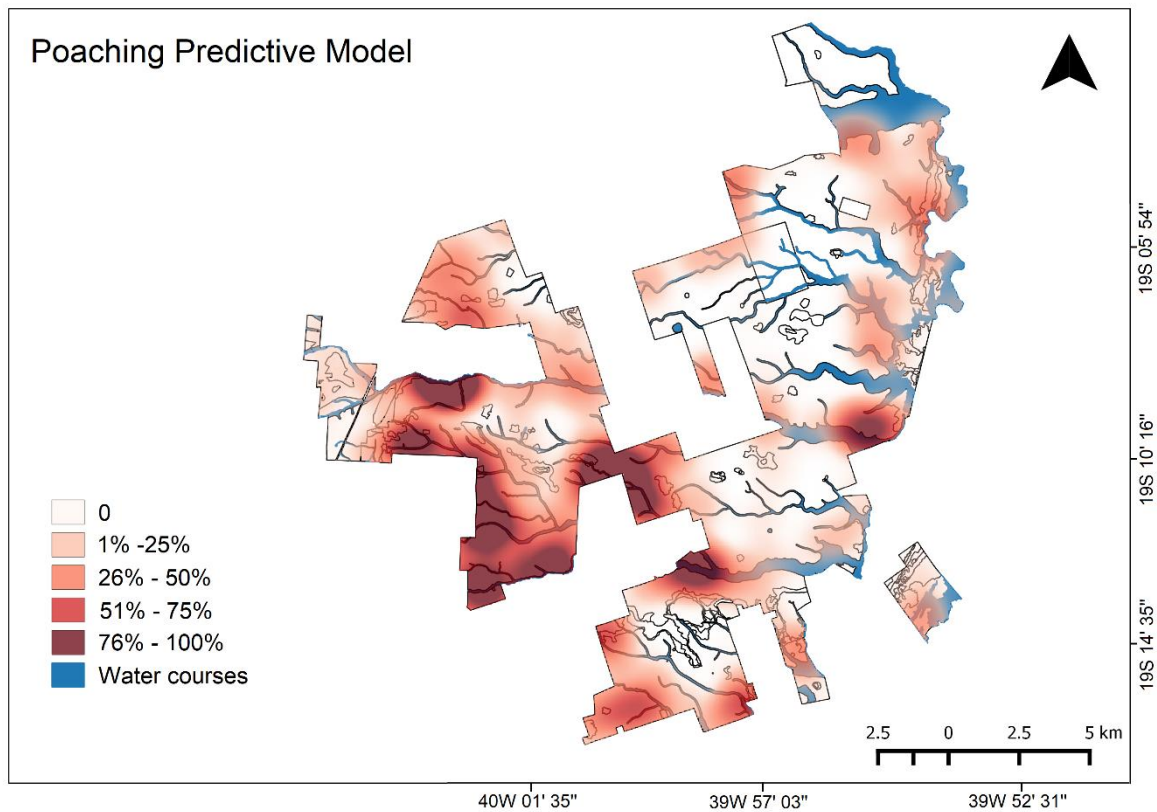
Source: Author, 2018.

Figure11 - Map showing the historical poaching records for the period of 2000-2014 in the Vale Natural Reserve.
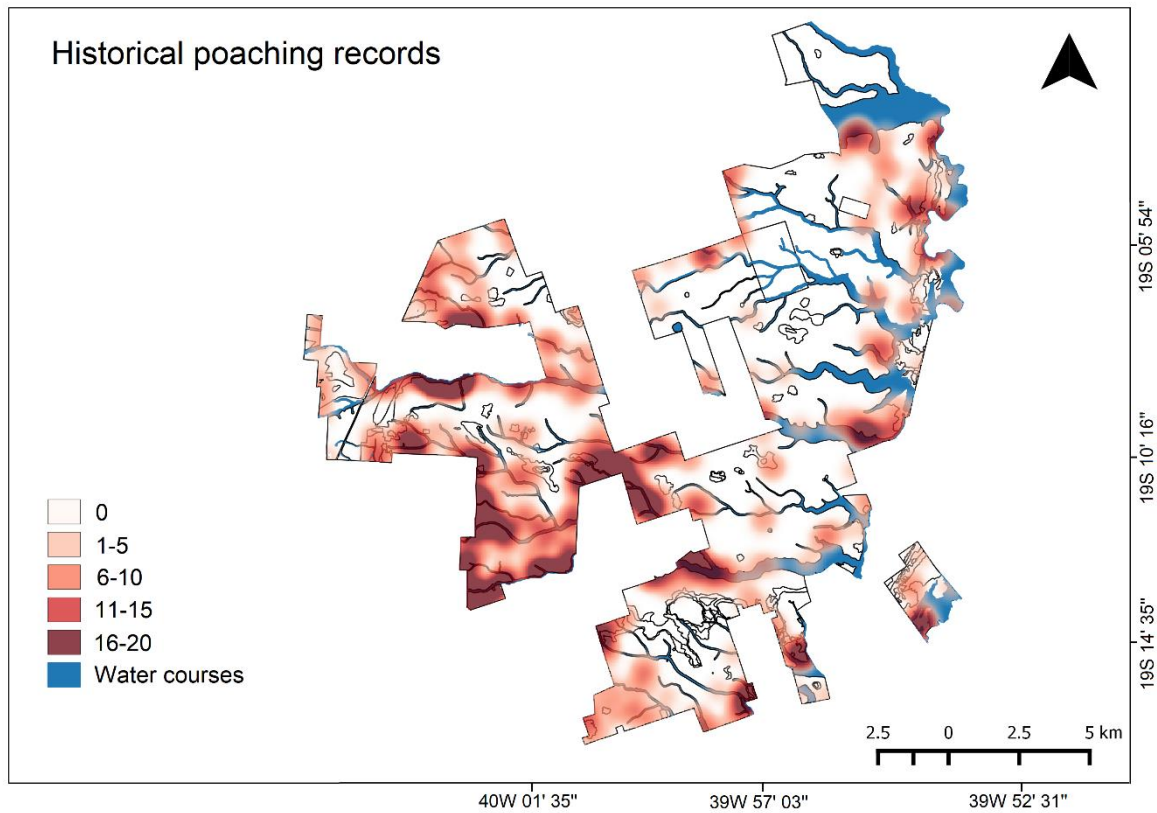
Source: Author, 2018.

2.3 **Discussion**

Our trapping-data indicated that the best covariates to predict poachers' occupancy for VNR were the distances from water resources and Reserve edge, while three of the best covariates for detectability were distances from water resources, Reserve edge, and from human settlements. These covariates can all be considered as proxies for the accessibility of different Reserve areas by poachers. Thus, the distribution of poaching occurrence and intensity was highly correlated with accessibility to the VNR. Indeed, it has been suggested that human disturbances on many species and ecosystems are stronger in areas with higher physical accessibility (Peres and Lake, 2003). The ease of access to natural areas favors not only poaching, but also other illegal activities such as smuggling, fishing, logging, encroachment for farming and grazing, and mining and extraction of natural products for trade (Kramer et al., 1997; Tabarelli et al., 2005). It is important to note that although the distribution of poachers (i.e. occupancy) was related to the distances from water and Reserve edge, the intensity of

poachers (i.e., detectability) besides affected by these two covariates was also affected by the distance from human settlements. This may be related to the fact that areas with higher human densities, such as human settlements, increase the intensity of poachers, but this does not rule out the fact that poachers' distribution can go beyond Reserve areas close to human settlements. Accordingly, poaching pressure intensity within protected areas tends to increase mainly due to human population growth and distance from human settlements, allowing higher intensity of access to preserved areas (Robinson et al., 1999).

Both historical poaching records and those estimated in the present study indicate that poaching pressure is not homogeneous across the Reserve, with the presence of hotspots of higher poaching intensity. First, the positive association of poaching pressure with accessibility suggest that poachers prefer areas of easy access. Second, the positive effect of frequency of game species on detectability suggests that poaching intensity is higher in areas of higher prey abundance. Take these two findings together, our results suggest that poachers prefer not only areas of easier access, but also areas where they know that some of the preferred mammalian species would be found more often, thus increasing the cost-benefit balance of the poaching expeditions.

In addition to showing a heterogeneous spatial distribution of poaching in VNR, our results also indicate that environmental and biological factors, such as lunar light intensity and frequency of records of potential game species, also affect poachers' detectability. Poachers are good connoisseurs in common sense questions of animal behavior, and are probably able to identify and track patterns of distribution of game species. Poaching strategies and site selection usually demonstrate the fact that poaching implies the need for a detailed traditional knowledge of the ecology of the exploited animals as well as to local ecological aspects (Hertz and Kapel, 1986; Takekawa, 2000; Henfrey, 2002). This traditional knowledge may also have influenced the higher poacher detectability in nights with higher lunar light intensity (i.e. full moon nights). This higher detectability of poachers on full moon nights is an opposite result to the behavior of some of the game species that tend to be less active on full moon nights (Julien-Laferrière, 1997; Kotler et al., 2002; Mori et al., 2014). However, this behavior is similar to wild primate species that use full moon nights to improve their foraging performance, because they are highly-dependent on vision for predator detection, orientation and feeding (Nash, 1986; Fernández-Duque and Gursky, 2003; Schwitzer et al., 2007). The choice for hunting in clear nights can also be related to predator avoidance or orientation (increasing the chances of survival and taking the food back home).

It is important to consider that species such as the red brocket deer (*Mazama americana*), the nine-banded and the six-banded armadillos (*Dasypusnovemcintus* and *Euphractussexcinctus*, respectively), and the lowland tapir (*Tapirusterrestris*), respond to poaching intensity, as suggested by our previous studies in VNR (Ferreguetti et al., 2015; 2016b; 2017b). In addition, an average of 173 events of poaching/year were recorded for the same region, over a period of 4 years (2010-2013; Sousa and Srbek-Araujo, 2017). The same authors highlighted a 32.1% increase in poaching events compared to previous years based on historical data since 1998 (Kierulff et al., 2014). This is an alarming situation that calls for urgent measures such as increasing Reserve security and monitoring. To this aim, the results of our study can be helpful, as they allow focusing surveillance actions to the areas with the highest poaching potential (Figure 4).

Our results provide subsidies for increasing knowledge about an illegal but still widespread and common practice. The unique historical poaching data validated the model presented here, highlighting the usefulness of the approach used (i.e. occupancy modeling of camera-trapping data) to predict spatial distribution of poaching. The approach presented here to identify poaching hotspots is cost-effective and much cheaper than 14 years of extensive monitoring, and therefore may be applied in other protected and non-protected areas throughout the world. We recommend that researchers and Reserve managers work together to apply the approach presented here in other Reserves, whenever possible. Camera-trapping has been increasingly used throughout the world, and it is likely that the necessary information to replicate our analyses has already been obtained in many Reserves. By replicating our analysis in other Reserves, it will be possible to detect patterns and generalizations that will be useful to inform management in different areas. However, we acknowledge that replication may be prohibitive in many areas in developing countries, due to constraints in both human and financial resources. If this is the case, we recommend as a first general guideline to focus surveillance / monitoring mainly near forest edges and water resources.

3 LIVING ON THE EDGE: HOW POACHING AFFECTS RICHNESS, BIOMASS, AND OCCUPANCY OF NON-VOLANT MAMMALS IN A PROTECTED AREA IN THE ATLANTIC FOREST, SOUTHEASTERN BRAZIL

Ferreguetti, A.C.; Allemand, M.M.; Pereira-Ribeiro, J.; Tomás, W.M.; Rocha, C.F.D.;

Bergallo, H.G. & Miller, D.A.

In preparation

The anthropic influence on the environment has been increasingly intense, resulting in what some researchers call the "crisis of biodiversity", a reference to extinctions driven by various factors related to human activities (Di Minin et al., 2016). Current extinction rates resulting from anthropogenic activities are unprecedented and highly unusual in Earth's history (Ceballos et al., 2015). Studies emphasize habitat loss and fragmentation as major threats to tropical biodiversity and ecosystem services, as changes in land use have drastically reduced forest cover worldwide (Laurance et al., 2014). Activities such as poaching, extraction of timber and non-timber forest products, as well as extensive livestock, are examples of anthropogenic disturbances and important drivers of biodiversity loss (Rito et al., 2016).

Hunting in tropical forests provides nutrition and income for local communities and is deeply rooted in social and cultural traditions (Bennett and Robinson, 2000a). While hunting, on the one hand, plays a vital role for many human populations, on the other, this activity, by reducing or extinguishing populations of game species, may be responsible for the degradation and loss of key forest maintenance services (Redford, 1992; Peres et al., 2016). This imbalance occurs most often when the motivation for hunting is unregulated trade, driven by the demands of international and local black markets (Antunes et al., 2016). For example, Benítez-López et al. (2017) estimated that the extraction of animals from nature for consumption or marketing of their products is responsible for the 83% decline in mammalian populations across the globe.It is because of this negative role, coupled with the use of more efficient slaughter techniques, habitat fragmentation, increased numbers of large enterprises and the expansion of the black market economy, that hunting is considered one of the main triggering factors of the current "biodiversity crisis" (Bennett et al., 2002; Nasi et al., 2008). In Brazil, hunting is illegal, hereafter referred to as poaching (Federal Law 9605/98—Law of Environmental Crimes and Decree 6514/2008), and penalties are more severe for poaching inside protected areas and for the poaching of threatened species (Art. 29 Federal Law 9605/98). Populations of different

species can be affected by poaching at different magnitudes (Peres, 1996; Bodmer et al., 1997; Peres and Nascimento, 2006; Peres and Palacios, 2007; de Souza and Alves, 2014). However, mammals that have a comparatively longer life span, lower population growth rate and longer generation time tend to be more susceptible to local extinction (Bodmer et al., 1997).

Static models of species ranges assume species responses are in equilibrium within the environment and homogenous across widely varying habitats, species communities, and climatic conditions (Zurell et al., 2009). This may be a reasonable assumption in some systems or at a subset of areas within a species' range, but for many species there is population heterogeneity in resiliency, adaptability, dispersal dynamics and growth across space (Kokko and López-Sepulcre, 2006; Eckert et al., 2008; Sexton et al., 2009). This translates into increasing or decreasing population growth rates that can explain population persistence or local extinction patterns (Kirkpatrick and Barton, 1997) that help form range limits. Understanding this variation allows for more robust models of species distributions (e.g. Occupancy Modelling). Modeling species distributions previously relied heavily on occurrence-only data, limiting the understanding of what conditions exclude species and at what level they act (Phillips et al., 2009). However, more recent approaches consider not only species presences but absences, population growth rates, and mechanistic responses. Expansion of the types of models available to estimate these parameters has allowed researchers to create ecologically-grounded species distribution models and forecast range shifts and contractions (Guisan and Thuiller, 2005; Keith et al., 2008; Elith and Leathwick, 2009; Zurell et al., 2016). Occupancy modelling provides an approach for estimating the probability of occupancy of a site by an organism of interest while accounting for imperfect detection, based on data collected at repeated visits to multiple sites (Bailey et al., 2014; Mackenzie et al., 2002).

In this context, we aimed to evaluate, by incorporating imperfect detectability, the effects caused by poaching on non-volant mammal species populations in the Vale Natural Reserve (VNR). Specifically, we evaluated how poaching would affect species biomass, richness and occupancy (occupancy defined as species sensitivity to poaching pressures). We expected that poaching would have a negative effect on the general community (richness) and that biomass and sensitivity of game species to poaching would be negatively higher.

3.1 **Material and methods**

3.1.1 <u>Study Area</u>

Vale Natural Reserve (VNR) is a protected area covering 235 km^2. The reserve is located in the neighboring municipalities of Linhares and Jaguaré (19°18'23''S, 39°45'34"W), in northeastern Espírito Santo, Brazil (Figure 1), and belongs to a major Brazilian mining enterprise, the Vale Company.

The reserve is composed of one main block of rainforest (approximately 98.1% of the total area), and a much smaller fragment, known as Biribas Reserve, southeast of the main block (Figure 3). The VNR is covered by a mosaic of habitats with four main vegetation types (adapted from Jesus, 1987; Peixoto and Gentry, 1990): coastal plain, riparian, and sandy soil forests, and natural grasslands.

3.1.2 Data Collection

We collected data over a 5-year period (between 2013 to 2018) to verify poaching effect on the density, biomass, richness, and occupancy of mammalian species using two methods described below.

3.1.2.1 Line-transects

To estimate the density of mammalian species, we established four 5 km–long transects separated at least 4 km from each other divided by three regions of the VNR (Figure 12). Data were collected for 5-years between 2013 to 2018. In the first year, between April 2013 and May 2014, we collected data monthly through 14 field trips lasting eight days each. In subsequent years, between June 2014 and October 2018, we collected data bi-monthly through field trips lasting eight days each. Transects were surveyed using a distance sampling technique (Buckland et al. 2001). Morning surveys were conducted by a single observer beginning between 05:30 and 06:30 h. The observer waited three hours before starting the afternoon survey along the same transect, between 12:00 and 14:00 h. Surveys were conducted at a speed of approximately 1 km/h, and we alternated survey order every month. We recorded the perpendicular distance of every observed species from the transect line (with a measuring tape) and recorded the date and time.

Figure12 - Habitat mosaicandsampling site locationinsidethe Vale Natural Reserve, Espírito
    Santo, Brazil.
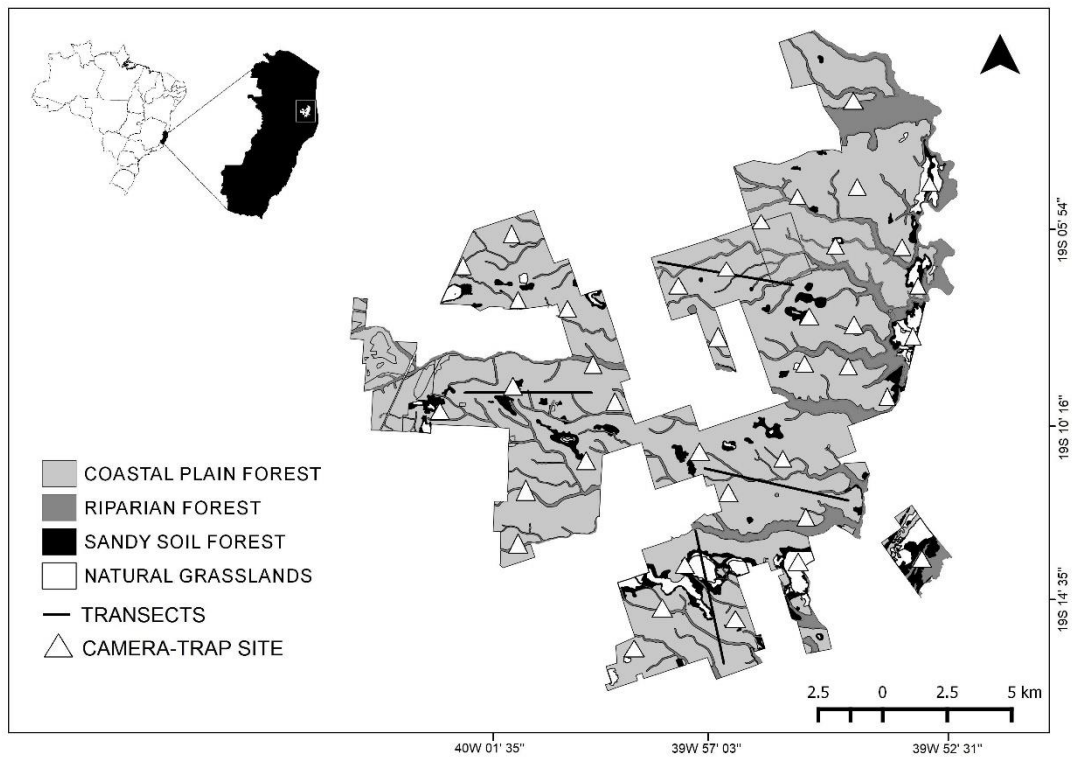
Source: Author, 2019.

3.1.2.2 Camera-trapping

We selected 39 sampling sites using a systematic design. We placed a 1 km^2 grid over a digital
map of the reserve and identified sample sites by selecting grid cells, separated by at least 1 km
from another cell. This separation of 1 km was delimited to avoid pseudoreplication of
individuals during sampling. This strategy resulted in a relatively even site distribution within
the VNR. Data were collected for approximately 3 months April-June each year from 2013 to
2018. We used this sampling scheme to model the species' richness, occupancy and
detectability over the 5-years of sampling (Figure 12). At each site, we installed one passive
infrared Bushnell® camera trap (i.e. with movement detection sensor) in the center of the
selected grid cell, 40–50 cm above the ground. All stations were checked every 20–25 days to
change batteries if needed. The traps were programmed to operate for 24 h/d and were
simultaneously sampled for 50 days of trapping effort each year. We did not use bait to attract
mammals. We separated the photographic records into sampling intervals of 5 consecutive days
(occasions), totaling 10 occasions per year. The pictures were examined to determine the
species detection/non-detection in every occasion to construct a species detection history.

3.1.2.3 Poaching Index

The VNR has an environmental surveillance system (locally called Ecosystem Protection) responsible for the protection of the area against threats (e.g., poaching, palm harvesting, logging, and forest fires). This surveillance system is performed in a standardized way to equally cover all areas of the reserve. Poaching data was generated by the Ecosystem Protection team across the entire VNR considering each poaching event separately (date, reserve where the event was recorded, location of the event, geographic coordinates, and type of evidence collected). Together with this database, we used the poachers that were photographed by camera traps during this study to calculate the poaching intensity within the study area. These records were used to calculate a poaching density (records per km^2) and added to each grid square in which a camera trap had been installed (For more details see Ferreguetti et al. 2018a; 2019).

3.1.3 Data Analysis

3.1.3.1 Density and Biomass

Although we recorded 19 species of medium and large sized mammals, we only obtained enough records (i.e. >40 records; Buckland et al., 2001) to estimate the density of 11 species through Distance Sampling. These species were grouped into four functional groups: armadillos (*Dasypusnovemcinctus*, *D. septemcinctus*, *Euphractussexcinctus*, and *Cabassoustatouay*), ungulates (*Mazama gouazoubira*, *M. americana*, *Tapirusterrestris*, *Pecaritajacu*, and *Tayassupecari*), carnivore (*Nasuanasua*), and rodent (*Dasyproctaleporina*). The densities of species were estimated using the Distance 7.3 Software (Thomas et al., 2010) for each region of the VNR (north, south, and west). We used Akaike information criterion (AIC; Akaike, 1973) for choosing the best model for effective strip width (ESW) definition. Using density values (D) multiplied by average body weight (BW), we calculated the biomass of species (D x BW = kg km^2). BW was based on Paglia et al. (2012).We tested the difference between years in the estimates using one-way analysis of variance (ANOVA).We tested the poaching effect on the mammalian biomass of each functional group through linear regressions. The analysis were performed in R software version 3.4.4(R Development Core Team, 2016).

3.1.3.2 Multi-species Occupancy Models

We investigated species-specific variation in occupancy at a site without poaching effect (intercept values) and the strength of the relationship between poaching and occupancy (slope values) for each site. We estimated occupancy intercept values to understand how frequently a species occurred at a site without the effect of poaching. Slope values instead estimates the responsiveness of a species to different values of poaching index at a site. We used all species detected at least once as our species pool and estimated relative species richness for each site (MacKenzie et al., 2006; Russell et al., 2009; Burton et al., 2012). We divided the species into two groups, poached and non-poached species following Kierulff et al. (2014) and Sousa and Srbek-Araujo (2017).

We started with a modified hierarchical multi-species occupancy model(Kéry and Royle, 2008; Zipkin et al., 2009; Broms et al., 2016).The model accounts for imperfect species detection probabilities through repeat surveys (t) while also allowing for simultaneous estimation of individual species (s) occupancy probabilities from a shared community distribution (Dorazio and Royle, 2005; Kéry and Royle, 2008). This allows multi-species occupancy models to borrow strength from more abundant species of the community to allow for better estimates of parameters for rare or harder to detect species.

Models were fit in JAGS (Plummer, 2003) via the jagsUI package (Kellner, 2018) in program R (R Development Core Team, 2016) using Markov chain Monte Carlo (MCMC) by generating 3 parallel chains with 25,000 iterations, a burn-in period of 5,000 and thinning of 10. We accounted for imperfect detection probability by using a 5-day period as the sampling occasion and recorded all detections of a particular species with a 1 or a 0 for no detections, for each sampling occasion. We used independent flat (uninformative) priors for the group level hyper-parameters. We assessed convergence using the Gelman–Rubin statistic with values <1.1 considered acceptable (Gelman and Rubin, 1992). We tested the difference between years in the detections using one-way analysis of variance (ANOVA).

3.2 **Results**

3.2.1 <u>Density and Biomass</u>

We obtained 4876 records of 11 mammalian species in a total effort of 3093 km walked between 2013 to 2018 (Table 3). All density estimates had a coefficient of variation (CV) of less than 20%. Among the four functional groups, ungulates and armadillos had their respective biomass negatively affected by different poaching values in the VNR regions with the west (i.e. highest poaching index) with lower biomass of species of these functional groups (armadillos - $R^2=0.954$; F=268.65; df=14; p<0.001; n=15 and ungulates - $R^2=0.908$; F=128.9; df=14; p<0.001; n=15; Figure 13). On the other hand, the carnivores and rodents group not have their respective biomass marginallyaffected by VNR poaching (carnivores - $R^2=0.44$; F=8.65; df=14; p=0.06; n=15 and rodents - $R^2=0.41$; F=7.78; df=14; p=0.07; n=15; Figure 13). We did not found a significant difference on species biomass estimated over the 5-years (one-way ANOVA; Biomass = F=0.12; df=14; p=0.85   Detailed estimates by species for each VNR region were compiled in Table 4 below.

Table 3–Species records per year and employed effort in the Vale Nature Reserve, Linhares Municipality, Espírito Santo State, Brazil (to be continued).

Species	Numberofrecords	Effort (km)
2013 – 2014		
Mazama americana	61	
Mazamagouazoubira	74	
Tapirusterrestris	63	
Tayassu pecari	58	
Pecarytajacu	75	
Dasypusnovemcinctus	49	908
Euphractussexcinctus	62	
Cabassoustatouay	35	
Dasypusseptemcinctus	41	
Dasyprocta leporina	413	
Nasuanasua	190	

Table 3 –Species records per year and employed effort in the Vale Nature Reserve, Linhares Municipality, Espírito Santo State, Brazil (to be continued).

Species	Numberofrecords	Effort (km)
2014-2015		
Mazama americana	55	
Mazamagouazoubira	71	
Tapirusterrestris	58	
Tayassu pecari	61	
Pecarytajacu	64	
Dasypusnovemcinctus	45	565
Euphractussexcinctus	52	
Cabassoustatouay	41	
Dasypusseptemcinctus	36	
Dasyprocta leporina	342	
Nasuanasua	115	
2015-2016		
Mazama americana	52	
Mazamagouazoubira	68	
Tapirusterrestris	62	
Tayassu pecari	55	
Pecarytajacu	67	
Dasypusnovemcinctus	52	545
Euphractussexcinctus	63	
Cabassoustatouay	38	
Dasypusseptemcinctus	33	
Dasyprocta leporina	371	
Nasuanasua	99	
2016-2017		
Mazama americana	61	
Mazamagouazoubira	64	
Tapirusterrestris	60	
Tayassu pecari	57	
Pecarytajacu	68	
Dasypusnovemcinctus	47	545
Euphractussexcinctus	60	
Cabassoustatouay	35	
Dasypusseptemcinctus	32	
Dasyprocta leporina	317	
Nasuanasua	106	

Table 3 –Species records per year and employed effort in the Vale Nature Reserve, Linhares
Municipality, Espírito Santo State, Brazil (continuation).

Species	Numberofrecords	Effort (km)
	2017-2018	
Mazama americana	58	
Mazamagouazoubira	69	
Tapirusterrestris	59	
Tayassu pecari	54	
Pecarytajacu	63	
Dasypusnovemcinctus	52	530
Euphractussexcinctus	57	
Cabassoustatouay	42	
Dasypusseptemcinctus	39	
Dasyprocta leporina	336	
Nasuanasua	119	

Figure13 - Relationship between estimated biomass and poaching index by mammalian functional group for the different regions in Vale Natural Reserve, Espírito Santo state, Brazil.
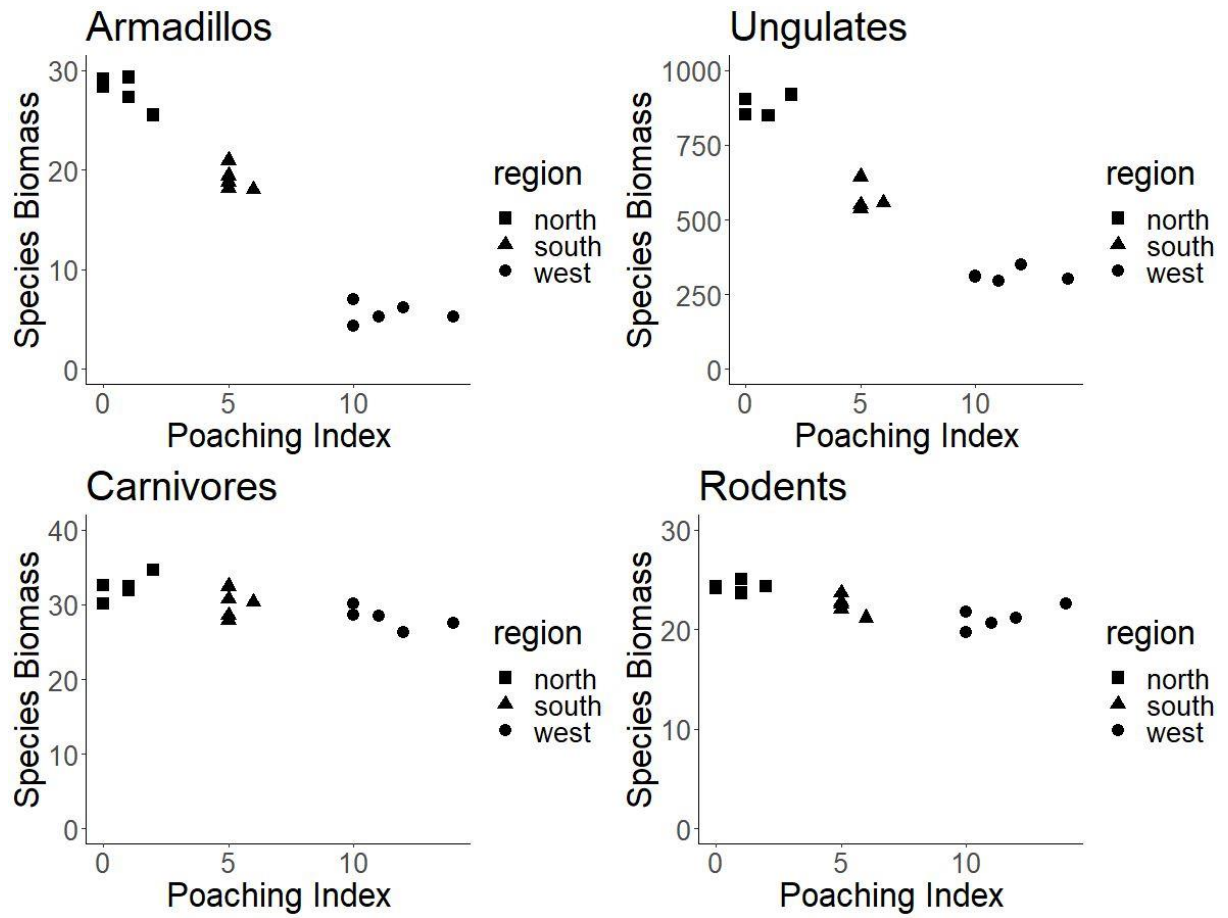
Source: Author, 2019.

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo State, Brazil (to be continued).

Species	Mass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Dasypusnove mcinctus	4	2013-2014	north	armadillo	285	2.12	0.18	8.48	0.72	2
		2014-2015	north	armadillo	165	2.56	0.15	10.24	0.6	1
		2015-2016	north	armadillo	155	2.68	0.16	10.72	0.64	0
		2016-2017	north	armadillo	165	2.75	0.18	11	0.72	0
		2017-2018	north	armadillo	150	2.69	0.18	10.76	0.72	1
		2013-2014	south	armadillo	365	2.15	0.12	8.6	0.48	5
		2014-2015	south	armadillo	245	2.05	0.14	8.2	0.56	5
		2015-2016	south	armadillo	240	2.08	0.08	8.32	0.32	5
		2016-2017	south	armadillo	235	1.98	0.14	7.92	0.56	6
		2017-2018	south	armadillo	235	2.04	0.13	8.16	0.52	5
		2013-2014	west	armadillo	255	0.78	0.07	3.12	0.28	10
		2014-2015	west	armadillo	155	0.64	0.09	2.56	0.36	12
		2015-2016	west	armadillo	150	0.61	0.11	2.44	0.44	11
		2016-2017	west	armadillo	145	0.52	0.09	2.08	0.36	14
		2017-2018	west	armadillo	145	0.32	0.07	1.28	0.28	10
Dasypussepte mcicntus	1	2013-2014	north	armadillo	285	1.25	0.12	1.25	0.12	2
		2014-2015	north	armadillo	165	1.35	0.18	1.35	0.18	1
		2015-2016	north	armadillo	155	1.28	0.14	1.28	0.14	0
		2016-2017	north	armadillo	165	1.24	0.12	1.24	0.12	0
		2017-2018	north	armadillo	150	1.32	0.15	1.32	0.15	1
		2013-2014	south	armadillo	365	1.05	0.11	1.05	0.11	5
		2014-2015	south	armadillo	245	1.08	0.09	1.08	0.09	5
		2015-2016	south	armadillo	240	1.07	0.12	1.07	0.12	5

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (to be continued).

Species	Mass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Dasypussepte mcicntus	1	2016-2017	south	armadillo	235	1.04	0.07	1.04	0.07	6
		2017-2018	south	armadillo	235	1.11	0.11	1.11	0.11	5
		2013-2014	west	armadillo	255	0.32	0.08	0.32	0.08	10
		2014-2015	west	armadillo	155	0.24	0.07	0.24	0.07	12
		2015-2016	west	armadillo	150	0.28	0.08	0.28	0.08	11
		2016-2017	west	armadillo	145	0.22	0.06	0.22	0.06	14
		2017-2018	west	armadillo	145	0.26	0.07	0.26	0.07	10
Euphractussex cinctus	4.5	2013-2014	north	armadillo	285	2.15	0.19	9.675	0.855	2
		2014-2015	north	armadillo	165	2.34	0.17	10.53	0.765	1
		2015-2016	north	armadillo	155	2.21	0.17	9.945	0.765	0
		2016-2017	north	armadillo	165	2.38	0.14	10.71	0.63	0
		2017-2018	north	armadillo	150	2.36	0.19	10.62	0.855	1
		2013-2014	south	armadillo	365	1.36	0.12	6.12	0.54	5
		2014-2015	south	armadillo	245	1.28	0.14	5.76	0.63	5
		2015-2016	south	armadillo	240	1.31	0.15	5.895	0.675	5
		2016-2017	south	armadillo	235	1.25	0.11	5.625	0.495	6
		2017-2018	south	armadillo	235	1.15	0.14	5.175	0.63	5
		2013-2014	west	armadillo	255	0.54	0.08	2.43	0.36	10
		2014-2015	west	armadillo	155	0.38	0.07	1.71	0.315	12
		2015-2016	west	armadillo	150	0.36	0.05	1.62	0.225	11
		2016-2017	west	armadillo	145	0.39	0.07	1.755	0.315	14
		2017-2018	west	armadillo	145	0.41	0.08	1.845	0.36	10
Cabassoustato uay	5.3	2013-2014	north	armadillo	285	1.15	0.12	6.095	0.636	2

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (to be continued).

Species	Averagemass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Cabassoustatouay	5.3	2014-2015	north	armadillo	165	0.98	0.14	5.194	0.742	1
		2015-2016	north	armadillo	155	1.21	0.13	6.413	0.689	0
		2016-2017	north	armadillo	165	1.17	0.15	6.201	0.795	0
		2017-2018	north	armadillo	150	1.25	0.11	6.625	0.583	1
		2013-2014	south	armadillo	365	0.98	0.09	5.194	0.477	5
		2014-2015	south	armadillo	245	0.58	0.11	3.074	0.583	5
		2015-2016	south	armadillo	240	0.78	0.14	4.134	0.742	5
		2016-2017	south	armadillo	235	0.65	0.08	3.445	0.424	6
		2017-2018	south	armadillo	235	0.82	0.06	4.346	0.318	5
		2013-2014	west	armadillo	255	0.21	0.08	1.113	0.424	10
		2014-2015	west	armadillo	155	0.32	0.04	1.696	0.212	12
		2015-2016	west	armadillo	150	0.18	0.06	0.954	0.318	11
		2016-2017	west	armadillo	145	0.22	0.07	1.166	0.371	14
		2017-2018	west	armadillo	145	0.19	0.05	1.007	0.265	10
Mazamagouazoubira	17	2013-2014	north	ungulate	285	8.25	2.06	140.25	35.02	2
		2014-2015	north	ungulate	165	8.59	2.12	146.03	36.04	1
		2015-2016	north	ungulate	155	7.48	1.98	127.16	33.66	0
		2016-2017	north	ungulate	165	7.25	1.89	123.25	32.13	0
		2017-2018	north	ungulate	150	7.65	2.02	130.05	34.34	1
		2013-2014	south	ungulate	365	5.98	1.86	101.66	31.62	5
		2014-2015	south	ungulate	245	6.02	1.54	102.34	26.18	5
		2015-2016	south	ungulate	240	4.58	1.21	77.86	20.57	5
		2016-2017	south	ungulate	235	5.64	1.32	95.88	22.44	6

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (to be continued).

Species	Averagemass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Mazamagouaz oubira	17	2017-2018	south	ungulate	235	5.12	1.28	87.04	21.76	5
		2013-2014	west	ungulate	255	3.12	1.12	53.04	19.04	10
		2014-2015	west	ungulate	155	2.95	0.89	50.15	15.13	12
		2015-2016	west	ungulate	150	2.48	0.92	42.16	15.64	11
		2016-2017	west	ungulate	145	2.65	0.82	45.05	13.94	14
		2017-2018	west	ungulate	145	2.72	0.93	46.24	15.81	10
Mazama americana	21	2013-2014	north	ungulate	285	6.25	1.12	131.25	23.52	2
		2014-2015	north	ungulate	165	6.12	1.56	128.52	32.76	1
		2015-2016	north	ungulate	155	5.98	1.48	125.58	31.08	0
		2016-2017	north	ungulate	165	6.02	1.32	126.42	27.72	0
		2017-2018	north	ungulate	150	5.89	1.54	123.69	32.34	1
		2013-2014	south	ungulate	365	4.56	0.98	95.76	20.58	5
		2014-2015	south	ungulate	245	4.25	1.12	89.25	23.52	5
		2015-2016	south	ungulate	240	4.12	0.89	86.52	18.69	5
		2016-2017	south	ungulate	235	4.02	1.25	84.42	26.25	6
		2017-2018	south	ungulate	235	4.09	1.12	85.89	23.52	5
		2013-2014	west	ungulate	255	2.35	0.89	49.35	18.69	10
		2014-2015	west	ungulate	155	2.56	0.92	53.76	19.32	12
		2015-2016	west	ungulate	150	1.89	0.78	39.69	16.38	11
		2016-2017	west	ungulate	145	2.05	0.82	43.05	17.22	14
		2017-2018	west	ungulate	145	2.32	0.78	48.72	16.38	10
Pecari tajacu	17	2013-2014	north	ungulate	285	7.85	1.35	133.45	22.95	2
		2014-2015	north	ungulate	165	7.65	1.65	130.05	28.05	1

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (to be continued).

Species	Average mass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Pecari tajacu	17	2015-2016	north	ungulate	155	6.98	1.42	118.66	24.14	0
		2016-2017	north	ungulate	165	7.05	1.38	119.85	23.46	0
		2017-2018	north	ungulate	150	6.89	1.12	117.13	19.04	1
		2013-2014	south	ungulate	365	5.25	1.32	89.25	22.44	5
		2014-2015	south	ungulate	245	6.02	1.25	102.34	21.25	5
		2015-2016	south	ungulate	240	5.89	1.12	100.13	19.04	5
		2016-2017	south	ungulate	235	6.12	1.14	104.04	19.38	6
		2017-2018	south	ungulate	235	5.45	1.18	92.65	20.06	5
		2013-2014	west	ungulate	255	3.56	1.05	60.52	17.85	10
		2014-2015	west	ungulate	155	4.02	0.98	68.34	16.66	12
		2015-2016	west	ungulate	150	3.65	1.12	62.05	19.04	11
		2016-2017	west	ungulate	145	4.65	0.98	79.05	16.66	14
		2017-2018	west	ungulate	145	3.58	1.08	60.86	18.36	10
Tayassu pecari	22	2013-2014	north	ungulate	285	5.98	1.05	131.56	23.1	2
		2014-2015	north	ungulate	165	5.55	0.98	122.1	21.56	1
		2015-2016	north	ungulate	155	6.02	1.12	132.44	24.64	0
		2016-2017	north	ungulate	165	6.12	1.25	134.64	27.5	0
		2017-2018	north	ungulate	150	6.45	1.15	141.9	25.3	1
		2013-2014	south	ungulate	365	4.25	1.23	93.5	27.06	5
		2014-2015	south	ungulate	245	4.12	0.98	90.64	21.56	5
		2015-2016	south	ungulate	240	4.36	1.05	95.92	23.1	5
		2016-2017	south	ungulate	235	4.85	1.08	106.7	23.76	6
		2017-2018	south	ungulate	235	4.02	1.32	88.44	29.04	5

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (to be continued).

Species	Averagemass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Tayassu pecari	22	2013-2014	west	ungulate	255	2.56	0.98	56.32	21.56	10
		2014-2015	west	ungulate	155	3.12	0.89	68.64	19.58	12
		2015-2016	west	ungulate	150	2.85	0.95	62.7	20.9	11
		2016-2017	west	ungulate	145	2.42	1.05	53.24	23.1	14
		2017-2018	west	ungulate	145	3.04	0.98	66.88	21.56	10
Tapirusterrestris	170	2013-2014	north	ungulate	285	2.25	0.82	382.5	139.4	2
		2014-2015	north	ungulate	165	1.89	0.89	321.3	151.3	1
		2015-2016	north	ungulate	155	2.35	0.98	399.5	166.6	0
		2016-2017	north	ungulate	165	2.05	0.92	348.5	156.4	0
		2017-2018	north	ungulate	150	1.98	0.78	336.6	132.6	1
		2013-2014	south	ungulate	365	1.55	0.81	263.5	137.7	5
		2014-2015	south	ungulate	245	1.05	0.39	178.5	66.3	5
		2015-2016	south	ungulate	240	1.12	0.42	190.4	71.4	5
		2016-2017	south	ungulate	235	0.98	0.32	166.6	54.4	6
		2017-2018	south	ungulate	235	1.15	0.29	195.5	49.3	5
		2013-2014	west	ungulate	255	0.55	0.15	93.5	25.5	10
		2014-2015	west	ungulate	155	0.65	0.12	110.5	20.4	12
		2015-2016	west	ungulate	150	0.52	0.18	88.4	30.6	11
		2016-2017	west	ungulate	145	0.48	0.15	81.6	25.5	14
		2017-2018	west	ungulate	145	0.51	0.12	86.7	20.4	10
Nasuanasua	4	2013-2014	north	carnivore	285	8.65	1.58	34.60	6.32	2
		2014-2015	north	carnivore	165	8.12	1.43	32.48	5.72	1
		2015-2016	north	carnivore	155	8.15	1.52	32.60	6.08	0

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (to be continued).

Species	Averagemass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Nasuanasua	4	2016-2017	north	carnivore	165	7.52	1.49	30.08	5.96	0
		2017-2018	north	carnivore	150	7.98	1.39	31.92	5.56	1
		2013-2014	south	carnivore	365	6.98	1.42	27.92	5.68	5
		2014-2015	south	carnivore	245	7.15	1.32	28.60	5.28	5
		2015-2016	south	carnivore	240	8.12	1.36	32.48	5.44	5
		2016-2017	south	carnivore	235	7.58	1.42	30.32	5.68	6
		2017-2018	south	carnivore	235	7.69	1.55	30.76	6.2	5
		2013-2014	west	carnivore	255	7.52	1.29	30.08	5.16	10
		2014-2015	west	carnivore	155	6.58	1.25	26.32	5.00	12
		2015-2016	west	carnivore	150	7.12	1.32	28.48	5.28	11
		2016-2017	west	carnivore	145	6.89	1.42	27.56	5.68	14
		2017-2018	west	carnivore	145	7.15	1.36	28.6	5.44	10
Dasyprocta leporina	3	2013-2014	north	rodent	285	8.12	1.28	24.36	3.84	2
		2014-2015	north	rodent	165	7.89	1.26	23.67	3.78	1
		2015-2016	north	rodent	155	8.05	1.56	24.15	4.68	0
		2016-2017	north	rodent	165	8.12	1.38	24.36	4.14	0
		2017-2018	north	rodent	150	8.36	1.52	25.08	4.56	1
		2013-2014	south	rodent	365	7.55	1.42	22.65	4.26	5
		2014-2015	south	rodent	245	7.35	1.36	22.05	4.08	5
		2015-2016	south	rodent	240	7.89	1.28	23.67	3.84	5
		2016-2017	south	rodent	235	7.05	1.12	21.15	3.36	6
		2017-2018	south	rodent	235	7.58	1.38	22.74	4.14	5

Table 4 - Estimates of density and biomass for each species in the Vale Natural Reserve regions between 2013 and 2018, Espírito Santo, Brazil (continuation).

Species	Average mass (kg)	Period (year)	Region	Group	Effort (km)	Density	Standard error (density)	Biomass	Standard error (biomass)	Poaching index
Dasyprocta leporina	3	2013-2014	west	rodent	255	6.58	1.25	19.74	3.75	10
		2014-2015	west	rodent	155	7.05	1.18	21.15	3.54	12
		2015-2016	west	rodent	150	6.89	1.25	20.67	3.75	11
		2016-2017	west	rodent	145	7.52	1.36	22.56	4.08	14
		2017-2018	west	rodent	145	7.26	1.25	21.78	3.75	10

3.2.2 Multi-species Occupancy Models

Our complete data set included a total survey effort of 32,175 camera days across five camera trap surveys (825 working days), with a total of 25,254 records of 27 non-volant mammals (Table 5; Table 6). The total numbers of detections per species were very heterogeneous and varied from 3 for the lesser grison*Galicitiscuja* to 823 for the lowland tapir *Tapirusterrestris* for the 5-day pooled data (Figure 14, Table 6).We did not found a significant difference on mammalian detections over the 5-years sampled (one-way ANOVA; Detections = F=0.77; df=131; p=0.56).

Table 5 – Total effort and success of capture from 39 camera traps over the years of monitoring at Vale Nature Reserve, Linhares, Espírito Santo, Brazil.

Surveyperiod	Workingdays	Total effort	Records frequency
April 2013 – March 2014	180	7020	5192
April 2014 – March 2015	150	5850	5077
April 2015 – March 2016	165	6435	5031
April 2016 – March 2017	170	6630	5012
April 2017 – March 2018	160	6240	4942
Total	825	32,175	25,254

Table 6 - List of recorded species and frequency per year of photographic records obtained from the 39 camera traps during monitoring at Vale Nature Reserve, Linhares, Espírito Santo, Brazil (to be continued).

| Species	2013-2014	2014-2015	2015-2016	2016-2017	2017-2018
Didelphimorphiaorder					
DidelphidaeFamily					
Didelphisaurita (Wied-Neuwied, 1826)	115	126	141	133	145
Rodentiaorder					
CuniculidaeFamily					
Cuniculus paca (Linnaeus, 1766)	356	340	325	333	312
SciuridaeFamily					
Guerlinguetusingrami(Thomas, 1901)	45	42	38	39	34
DasyproctidaeFamily					
Dasyprocta leporina (Linnaeus, 1758)	425	418	405	407	398
CaviidaeFamily					
Hydrochoerushydrochaeris(Linnaeus, 1766)	42	47	38	40	33
Lagomorphaorder					
LeporidaeFamily					
Sylvilagus brasiliensis (Linnaeus, 1758)	52	58	51	42	47
Cingulataorder					
DasypodidaeFamily					
Cabassoustatouay(Desmarest, 1804)	8	11	6	9	12
Dasypusnovemcinctus Linnaeus, 1758	203	195	192	189	178
*Dasypusseptemcinctus*Linnaeus, 1758	13	11	15	9	10
Euphractussexcinctus (Linnaeus, 1758)	205	198	183	185	178
Pilosa order					
MyrmecophagidaeFamily					
Tamanduatetradactyla (Linnaeus, 1758)	110	98	105	112	91

Table 6 -List of recorded species and frequency per year of photographic records obtained from the 39 camera traps during monitoring at Vale Nature Reserve, Linhares, Espírito Santo, Brazil (continuation).

Species	2013-2014	2014-2015	2015-2016	2016-2017	2017-2018
Cetartiodactylaorder					
CervidaeFamily					
Mazama americana (Erxleben, 1777)	328	315	308	305	299
Mazamagouazoubira (Fischer, 1814)	425	455	433	412	441
TayassuidaeFamily					
Tayassu pecari Link, 1814	402	399	391	385	382
Pecari tajacu Linnaeus, 1758	521	505	512	498	503
Perissodactylaorder					
TapiridaeFamily					
Tapirusterrestris Linnaeus, 1758	826	833	821	812	803
Carnivoraorder					
CanidaeFamily					
Cerdocyonthous (Linnaeus, 1766)	256	238	241	233	249
FelidaeFamily					
Leoparduspardalis (Linnaeus, 1758)	89	82	91	85	93
Leoparduswiedii (Schinz, 1821)	7	5	9	8	10
*Leopardusguttulus*Hensel, 1872	11	8	9	11	7
Pantheraonca(Linnaeus, 1758)	3	5	6	4	5
Puma concolor(Linnaeus, 1771)	6	7	5	7	6
Puma yagouaroundi (É. Geoffory, 1803)	55	62	52	58	57
MustelidaeFamily					
Eira barbara (Linnaeus, 1782)	112	98	104	91	107
Galictis cuja (Molina, 1782)	5	7	4	3	0
ProcyonidaeFamily					
Nasuanasua (Linnaeus, 1766)	520	442	491	541	485
Procyon cancrivorus(G.[Baron] Cuvier, 1798)	52	72	55	61	57

Figure14- Distribution of the total number of detections for 5-day pooled data (20 occasions), mean per-individual detection probabilities and mean naive occupancy for five camera trap surveys in the Vale Natural Reserve, Espírito Santo state, Brazil.
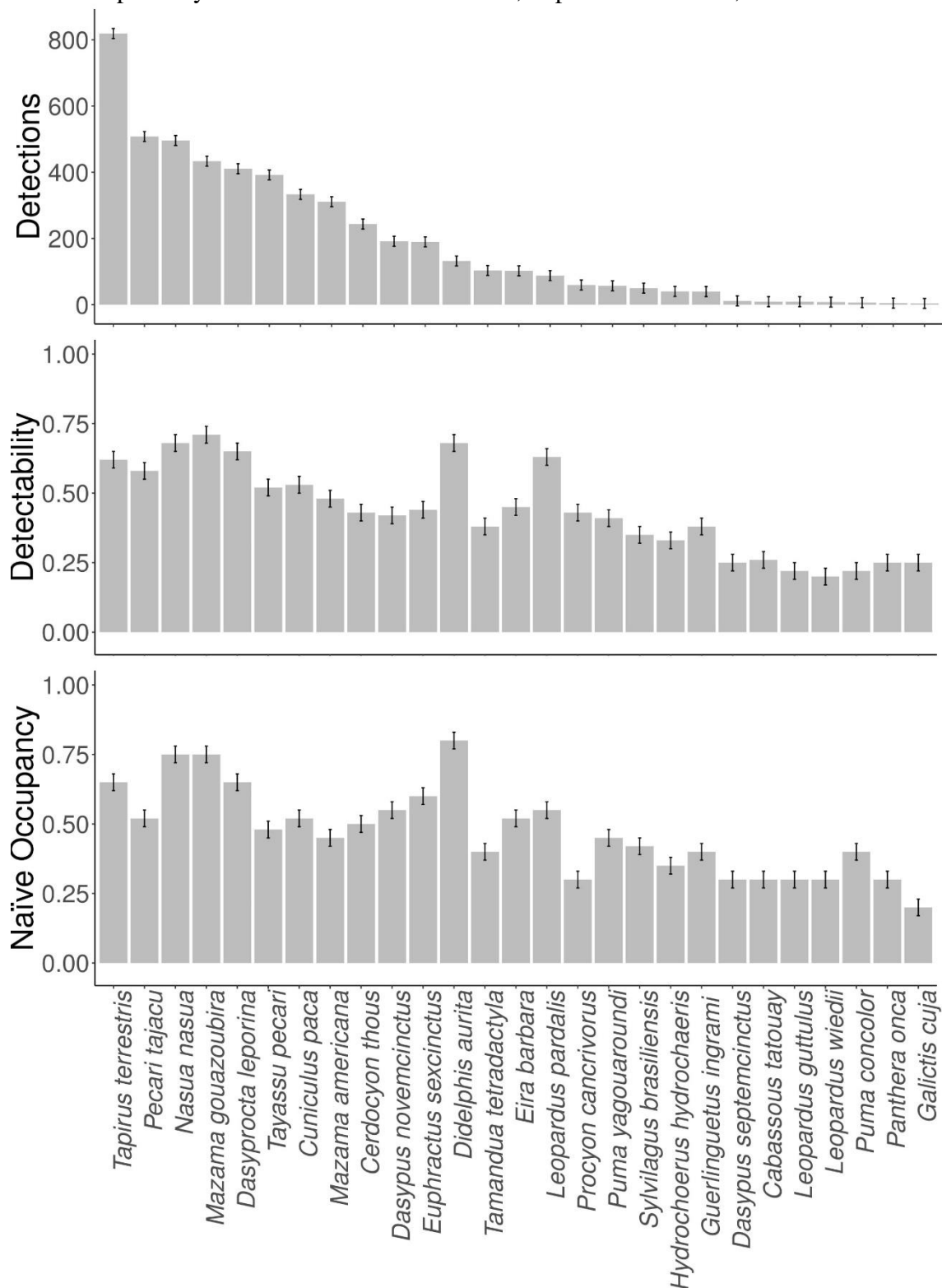

Note: Detection probabilities and occupancy were estimated under a multi-species occupancy model, and values shown are a means across all years.
Source: Author, 2019.

We estimated a negative effect of poaching index on all-species richness through the built multi-species occupancy model (Figure 15). However, when we separated the 27 recorded species into two distinct groups of poached and non-poached, only the richness of poached species was negatively affected by the increased VNR poaching index (Figure 15). While the richness of non-poached species was not affected by poaching.

Figure 15 – Relationship between estimated species richness and poaching index for 39 camera traps in the Vale Natural Reserve, Espírito Santo state, Brazil.
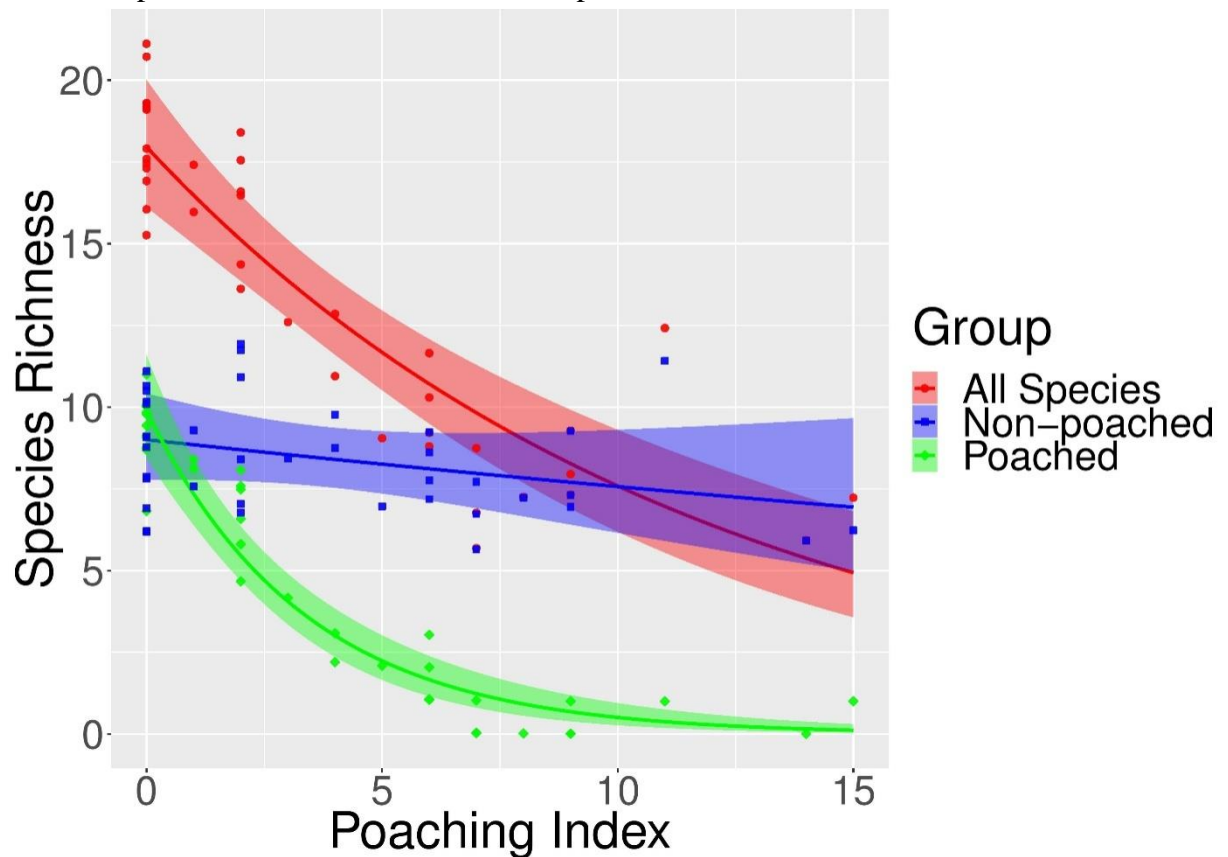
Note:Richness estimated with a multi-species occupancy model divided into three distint groups: all species richness, non-poached species richness (*Didelphis aurita*, *Tamandua tetradactyla*, *Hydrochoerushydrochaeris, Sylvilagusbrasiliensis*, *Guerlinguetusingrami*, *Cerdocyonthous*, *Galictiscuja*, *Eirabarbara*, *Procyon cancrivorus*, *Leoparduspardalis*, *Puma yagouaroundi*, *Leopardusguttulus*, *Leoparduswiedii*, *Puma concolor*, *Panthera onca*), and poached species richness (*Tapirusterrestris*, *Pecaritajacu*, *Tayassupecari*, *Mazama gouazoubira*, *Mazama americana*, *Dasyproctaleporina*, *Cuniculus paca*, *Nasuanasua*, *Dasypusnovemcinctus*, *Dasypusseptemcinctus*, *Cabassoustatouay*, *Euphractussexcinctus*).Species were classifed in poached and non-poached according to Kierulff et al. (2014)

Source: Author, 2019.

Poaching negatively influenced 11 of the 12 species classified as poached (Figure 16). These 11 species showed positive values when estimated occupancy at site without poaching effect (intercept values) showed high occupancy values (Figure 14A). However, the strength of the relationship between poaching and slope values for each site was higher for these species. The most affected species were *Pecaritajacu*, *Tayassupecari*, *Cuniculus paca*, and three species of armadillos (*Dasypusnovemcinctus*, *D. septemcinctus*, and *Euphractussexcinctus*). *Nasuanasua* was the only species with a poaching history in the VNR that was not affected by poaching (Figure 16). The 15 species that had no history of VNR poaching, classified as non-poached, were not affected by poaching (Figure 16).

Figure 16– Species-specific variation in occupancy in the Vale Natural Reserve, Espírito Santo state, Brazil.
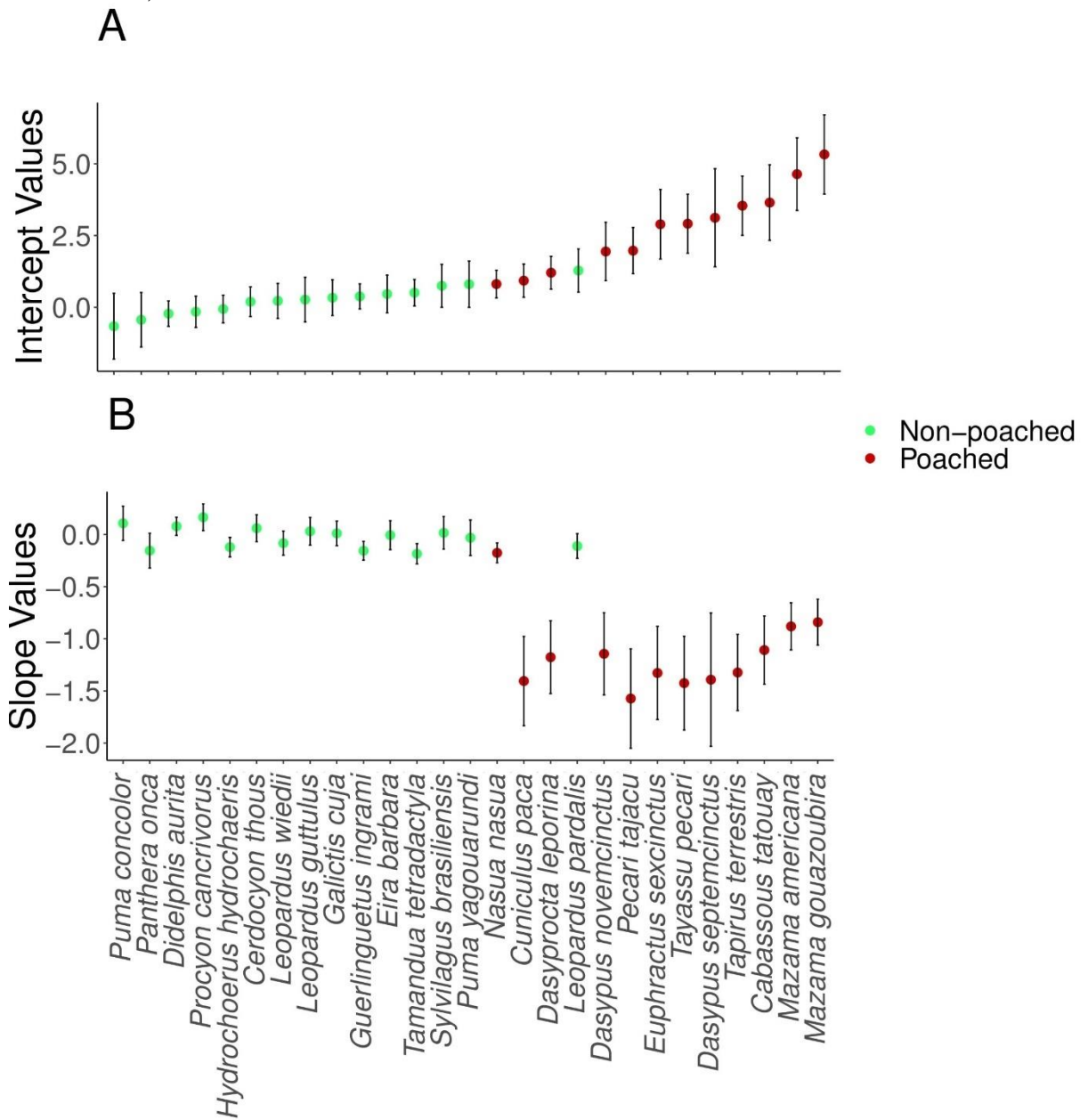
Legend: A) occupancy at a site without poaching effect (intercept values) and B) the strength of the relationship between poaching and occupancy (slope values) for each site.

Source: Author, 2019.

3.3 **Discussion**

Poaching negatively affected the non-volant mammals community in the VNR, reducing overall species richness and with significant effect on the richness of species with a poaching history in the literature (Kierulff et al., 2014; Sousa and Srbek-Araújo, 2017). We found a negative impact of poaching on the density and biomass in nine of the 11 mammal species evaluated in the VNR. These nine species are classified as ungulates and armadillos.Moreover, through the multi-species occupancy model, we found a strong negative effect of poaching on the occupancy of these same nine species belonging to these two groups. These two mammalian groups are often identified as the most sensitive to the effects of poaching on the netropical region.In the Atlantic Forest of Sao Paulo state, for example, Cullen et al. (2000) listed species of armadillos and ungulates as the most poached among the vertebrates evaluated by them. In addition, Cullen et al. (2000) mentions that colonist hunters in the region of Sao Paulo state consider ungulates and armadillos as subsistence wildlife species as a justification for poaching.In the semi-arid region of northeastern Brazil, Alves et al. (2009) also pointed the armadillos as the most poached species.In fact, in VNR, Kierulff et al. (2014) described that about 45% of the slaughtered animals were armadillos.In the Amazon, ungulates are among the most hunted species by local communities and these populations of ungulates decline as soon as hunting becomes a chronic process (Peres 2000b). In the Neotropical region, primates, ungulates and armadillos are particularly vulnerable to overhunting due to their low intrinsic rates of natural growth, high longevity, long generation time, and low population densities (Bodmer et al., 1997; Cardillo et al., 2004).The overhunting of large forest vertebrates can compromise important ecological processes for the maintenance of forest structure and species composition (Dirzo and Miranda, 1991; Wright et al., 2000; Dirzo et al., 2014), reducing the biodiversity in the long term (Terborgh, 1992, 2000). Most hunted species are frugivorous and / or herbivorous, (Peres, 2000a, 2000b; Townsend, 2000), which are the case of ungulate specieswhose play an ecological role in the dynamics of natural environments (Dirzo and Miranda, 1991; Wright et al., 2000; Stoner et al., 2007).

Although we provided here evidence that poaching has an impact on these species, we did not find a difference in density, biomass, and occupancy over the 5-years study in VNR.The fact that poaching did not affect species over time may be related to two main factors, the constancy of poaching in certain VNR localities and the time interval in which the study was conducted.In fact, poaching in VNR did not vary between the 5-years, only between the regions

of the reserve. Ferreguetti et al. (2018a) showed that the poaching incidence is higher in the western region of VNR and in the western region, we obtained the lowest density and biomass estimates.This also may be linked to the response time of species to impact. Several studies show a declining trend when poaching becomes a chronic activity over time (Terborgh, 1992, 2000; Peres 2000b; Wright et al., 2000; Dirzo et al., 2014). Large species, especially mammals, would have their populations so small that vital functions for the maintenance of ecosystems would be highly affected; thus, in the long term, the preservation of tropical forest vegetation would not be possible if the fauna were not also preserved (Redford, 1992).In Paraguay, Hill et al. (2003) analyzed game species encounter rates over 8 years and suggested long-term declines in crude encounter rates for all species, indicating that species in VNR may have a tendency to decline in the long term.

On the other hand, the two species classified as carnivores and rodents (i.e. *Nasuanasua* and *Dasyproctaleporina*, respectively) have their density and biomass affected marginally by poaching in the VNR.For *Dasyproctaleporina*, in particular, we did not expected to find this result, as the species is identified among the most poached in the VNR region, representing 10% of all slaughtered animals recorded(Kierulff et al., 2014).However, although we did not find a negative effect of poaching on *D. leporina* density and biomass, the estimated values are lower when compared to other studies with the species in the Neotropical region (Ferreguetti et al., 2018b).This low density estimated in the present study may be related to the poaching pressure (Wright et al., 2000; Ferreguetti et al., 2018b). In central Panama, Wright et al. (2000) found negative correlations between population density and poaching intensity for *Dasyproctapunctata*, with density estimates ranging from 100 ind / km^2 in areas considered without poaching to 20 ind / km^2 in areas with the highest incidence of poaching.However, *Nasuanasua* had no negative effect of poaching despite having records of poaching in the VNR (Kierulff et al., 2014). This may be due to *N. nasua* being a species that may have been opportunistically killed and not being a preferred prey of poachers in the VNR (Kierulff et al., 2014; Sousa and Srbek-Araújo, 2017).

However, poaching negatively affected the occupancy by *Dasyproctaleporina* in the VNR. This result indicates that, in fact, the species is being impacted by poaching in the reserve, although we have not detected a strong effect on its estimated density and biomass. As mentioned earlier, the estimated density values are low and therefore indicate a poaching pressure in the population of *D. leporina*. We also found a negative poaching effect for *Cuniculus paca*, one of the most hunted species in the Neotropical region (Emmons, 2016). *Cuniculus paca*is one of the three most poacher-slaughtered mammal species in VNR,

representing about 40% of all recorded slaughtered animals (Kierulff et al., 2014). Poaching represents one of the major impacts for *C. paca* in its entire area of occurrence (Emmons, 2016), this rodent species represents up to 8% of all wild meat consumed (Patton, 2015). For the other rodent species*Hydrochoerushydrochaeris*, on the other hand, we did not find a poaching effect on its occupancy in the VNR. This species, despite being a main target for poachers in other areas in the Neotropical region (Reid, 2016), is not commonly poached in VNR (Kierulff et al., 2014).

Poaching had no direct effect on occupancy by large felids (i.e. *Panthera onca* and *Puma concolor*) in the VNR. However, it is noteworthy that poaching could eventually negatively impact these species indirectly due to the extirpation of their main prey, which are the species directly impacted by poaching in the present study.This indirect impact of poaching can be predicted as large carnivores are especially vulnerable to low density and biomass of their preferred prey (Carbone et al., 2011) and prey depletion is one of their major threats worldwide (Wolf and Ripple, 2016). Overexploitation of wildlife for meat is a widespread phenomenon, which drives populations of many species toward extinction and may in turn affect large carnivores (Milner-Gulland et al., 2003; Ghoddousi et al., 2017). Indeed, the indirect impact of poaching on top-predator has benn observed in othe location such as *Panthera tigris*, in Asia, showed lower persistence in their populations according to high prey depletion rates (Chapron et al., 2008). In the Golestan National Park, located in Iran, the globally endangered Persian leopard (*Panthera pardussaxicolor*) showed a change in its prey preference because of the depletion of the usually preferred prey species (Ghoddousi et al., 2017). However, despite this change in the diet of the species, Ghoddousi et al. (2017) point out that this reduction in dietary niche breadth may threaten the long-term survival of the species.Therefore, it is extremely necessary to evaluate the diet of these top predators and whether poaching would be affecting these species in VNR.

Our results bring alarming results to one of Brazil's largest Atlantic Forest remnants. Poaching is an illegal activity constantly practiced not only in the Vale Natural Reserve (VNR) but throughout the Atlantic Forest and other Neotropical regions. We presented the first study in the Atlantic Forest that has estimated the long-term poaching impact on mammalian populations, demonstrating a potential negatively effect on the density, biomass and occupancy of species that are important to ecosystem functioning. In addition to the direct negative effects estimated in this study, we draw attention to potential indirect impacts of poaching on other mammalian species, such as large felids (*Panthera onca* and *Puma concolor*). We recommend that this study serve as a basis for start monitoring other populations of vertebrates target to

poaching, not only in the Atlantic Forest, but in other Neotropical ecosystems. We also hope that theses results could be the first step in drawing the attention of decision makers and enforcement officials to the importance in increasing penalties for those who engage in this illegal activity. It is important that more stringent measures be applied and that environmental police enforcement activities be intensified in all Atlantic Forest.

FINAL CONSIDERATIONS

- It is important to use a standardized quantitative metric to assess the impact of poaching on populations not only limited to the mammalian group, but all animal species that may be impacted by this illegal activity. To this end, we have proposed a feasible metric with the current poaching scenario, an illegal activity where it is not possible to count how many animals are being slaughtered and extracted from nature. This metric is to quantify how much evidence of poaching, both directly and indirectly spatially, thus estimating a density of poaching occurrence for each location.

- Concurrent with the standardized quantitative metric, it is important to continue the systematic surveillance monitoring carried out at Vale Natural Reserve. Minimally in protected areas, frequent surveillance is required. It is recommended that such surveillance be carried out systematically both spatially and temporally. Ideally, it is necessary to reconcile the results obtained by surveillance monitoring in the construction of the metric described above. These two steps will assist in creating a map of hotspots and hotmoments, where poaching is most frequent in space and time. This map should be updated frequently and used in surveillance planning.

- We have provided alarming information on the negative impact of poaching on mammal species evaluated here. To better understand the dimensions of this impact, it is important to continue monitoring these species and make projections through population viability analyzes to understand how species would be impacted in different scenarios. These scenarios should be constructed to assess how species would respond if the incidence of poaching remained stable, reduced or increased over time. In addition, it is necessary to understand how the impact of this illegal activity would be affecting ecological interactions and processes.

- Last but not least, it is extremely necessary to initiate and strengthen programs for poaching mitigation in our protected areas. It is necessary to sensitize the local community by establishing a sense of belonging with the protected area. A continuing environmental education program is recommended, where actions must be worked on in different age groups of the population. Poaching is not a simple impact to solve or mitigate, for besides being a cultural activity extremely rooted in human civilization, it is often an activity that generates income for local populations. Thus, multidisciplinary

programs should be adopted, where not only the environmental issue is addressed, but social, economic and several other important axes to mitigate this illegal activity.

REFERENCES

AKAIKE, H. Maximum likelihood identification of Gaussian autoregressive moving average models. **Biometrika**, v. 60, n. 2, p. 255-265, 1973.

AHMED, S. E., LEES, A. C., MOURA, N. G., GARDNER. T. A., BARLOW, J., FERREIRA, J., EWERS, R. M. Road networks predict human influence on Amazonian bird communities. **Proceedings of the Royal Society Biological Sciences,** v. 281, 2014. DOI: 10.1098/rspb.2014.1742

ALTRICHTER, M. ; ALMEIDA, R. Exploitation of white-lipped peccaries Tayassupecari (Artiodactyla: Tayassuidae) on the Osa Peninsula, Costa Rica. **Oryx**, v. 36, n. 2, p. 126-132, 2002.

ALVES, R. R. N. ; SOUTO, W. M. S. Ethnozoology: a brief introduction. **Ethnobiology and Conservation**, v. 4, p. 1–13, 2015.

ALVES, R. R., MENDONÇA, L. E., CONFESSOR, M. V., VIEIRA, W. L., LOPEZ, L. C. 2009. Hunting strategies used in the semi-arid region of northeastern Brazil. **JournalofEthnobiologyandEthnomedicine**, v. 5, n. 1, p. 12, 2009.

ALVES, R. R. N., GONÇALVES, M. B. R., VIEIRA, W. L. S. Caça, uso e conservação de vertebrados no semiárido Brasileiro. **Tropical Conservation Science**, v. 5, n. 3, p. 394-416, 2012.

ANTONITES, A., ODENDAAL, J. S. J. Ethics in human-animal relationships. **Acta Veterinaria Brno**, v. 73, n. 4, p. 539-548, 2004.

ANTUNES, A. P. F. R. M.; VENTICINQUE, E. M.; PERES, C. A.; LEVI, T.; ROHE, F. ; SHEPARD JR., G. H. Empty forest or empty rivers? A century of commercial hunting in Amazonia. **Science Advances**, v. 2, p. 1-14, 2016.

AQUINO, R., CALLE, A. Evaluacióndel estado de conservación de los mamíferos de caza: Un modelo comparativo en comunidades de la Reserva Nacional PacayaSamiria (Loreto, Perú). **Revista Peruana de Biología**, v. 10, n. 2, p. 163-174, 2003.

ARITA, H.T., ROBINSON, J.G. ; REDFORD, K.H. Rarity in neotropical forest mammals and its ecological correlates. **Conservation Biology**, v. 4, n. 2, p. 181-192, 1990.

ARROYO-RODRÍGUEZ, V., GONZÁLEZ-PEREZ, I. M., GARMENDIA, A., SOLÀ, M., ESTRADA, A. The relative impact of forest patch and landscape attributes on black howler monkey populations in the fragmented Lacandona rainforest, Mexico. **Landscape Ecology**, v. 28, n. 9, p. 1717-1727, 2013.

AUGUST, P.V. The role of habitat complexity and heterogeneity in structuring tropical mammal communities. **Ecology,** v. 64, p. 1495-1507, 1983.

AYRES, J. M., AYRES, C. Aspectos da caça no alto rio Aripuanã. **Acta Amazonica**, v. 9, n. 2, p. 287-298, 1979.

BAILEY, L. L., MACKENZIE, D. I., NICHOLS, J. D. Advances and applications of occupancy models. **Methods in Ecology and Evolution**, v. 5, n. 12, p. 1269-1279, 2014.

BATES, H. W. **The Naturalist on the River Amazons: a record of adventures, habits of animals, sketches of Brazilian and Indian life, and aspects of nature under the equator during eleven years of travel**. London: John Murray, 1864, 466p.

BECKER, M. Aspectos da caca em algumas regioes do cerrado de Mato Grosso. **Brasil Florestal**, v. 11, n. 47, p. 51-63, 1981.

BELLO, C., GALETTI, M., PIZO, M. A., MAGNAGO, L. F. S., ROCHA, M. F., LIMA, R. A., PERES, C. A., OVASKAINEN, O.; JORDANO, P. Defaunationaffectscarbonstorage in tropical forests. **Science Advances**, v. 1, n. 11, p. 1-10, 2015.

BENÍTEZ-LÓPEZ, A. A. R.; SCHIPPER, A. M.; INGRAM, D. J.; VERWEIJ, P. A.; EIKELBOOM, J. A. J. ; HUIJBREGTS, M. A. J. The impact of hunting on tropical mammal and bird populations. **Science**, v. 356, p. 180-183, 2017.

BENNETT, E. L., ROBINSON, J. G. Hunting of wildlife in tropical forests: implications for biodiversity and forest peoples. World Bank, Environment Department, Universidade de Minnesota, 2000a, 42 p.

BENNETT, E.L., ROBINSON, J.G. Carrying capacity limits to sustainable hunting in tropical forests. Hunting for sustainability in tropical forests. **In:** John G. Robinson, Elizabeth L. Bennett (Eds.). **Hunting for Sustainability in Tropical Forests**, Publisher: Columbia University Presss, 2000b, p.13-30.

BENNETT, E. E. H.; ROBINSON, J. ; WILKIE, D. Why is eating bushmeat a biodiversity crisis? **Conservation Biology**, v. 3, n. 1, p. 1-2, 2002.

BODMER R. E., ROBINSON J. G. 2004. **Evaluating the sustainability of hunting in the Neotropics**. In: People in nature: wildlife conservation in South and Central America. Silvius K., Bodmer R. E. and Fragoso J. M. V. (Eds.), pp.299–323. Columbia University Press, New York, USA.

BODMER, R. E. ROBINSON, J. G. 2006. **Análise de sustentabilidade da caça**. In: Métodos de Estudos em Biologia da Conservação e Manejo da Vida Silvestre. Cullen-Junior L., Rudran, R. Valladares-Padua, C. (Eds.). Curitiba: Editora UFPR.

BODMER, R. E., FANG, T. G. MOYA I. L. Primates and ungulates: a comparison of susceptibility to hunting. **Primate Conservation**, v. 9, p. 79-83, 1988.

BODMER, R.E., EISENBERG, J.F.; REDFORD, K.H. Hunting and the likelihood of extinction of Amazonian mammals. **Conservation Biology**, v. 11, n. 2, p. 460-466, 1997.

BRASHARES, J.S., ARCESE, P., SAM, M.K., COPPOLILLO, P.B., SINCLAIR, A.R. ; BALMFORD, A. Bushmeat hunting, wildlife declines, and fish supply in West Africa. **Science**, v. 306, n. 5699, p. 1180-1183, 2004.

BROMS, K. M., HOOTEN, M. B., ; FITZPATRICK, R. M. Model selection and assessment for multi-species occupancy models. **Ecology**, v. 97, n. 7, p. 1759-1770, 2016.

BUCKLAND, S.T., ANDERSON, D.R., BURNHAM, K.P., LAAKE, J.L., BORCHERS, D.L. ; THOMAS, L. **Introduction to distance sampling: estimating abundance of biological population**. Oxford: Oxford University Press, 2001. 432p.

BURTON, A. C., SAM, M. K., BALANGTAA, C., ; BRASHARES, J. S. Hierarchical multi-species modeling of carnivore responses to hunting, habitat and prey in a West African protected area. **PloS One**, v. 7, n. 5, e38007, 2012.

BURTON, A. C., NEILSON, E., MOREIRA, D., LADLE, A., STEENWEG, R., FISHER, J. T., BAYNE, E.; BOUTIN, S. Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. **Journal of Applied Ecology**, v. 52, n. 3, p. 675-685, 2015.

CARBONE, C., PETTORELLI, N., ; STEPHENS, P. A. The bigger they come, the harder they fall: body size and prey abundance influence predator–prey ratios. Biology Letters, v. 7, n. 2, p. 312-315, 2011.

CARDILLO, M., PURVIS, A., SECHREST, W., GITTLEMAN, J. L., BIELBY, J., MACE, G. M. HUMAN population density and extinction risk in the world's carnivores. **PLoS biology**, v. 2, n. 7, p. e197, 2004.

CARDINALE, B. J., DUFFY, J. E., GONZALEZ, A., HOOPER, D. U., PERRINGS, C., VENAIL, P., NARWANI, A., MACE, G. M., TILMAN, D., WARDLE, D. A., KINZING, A. P., DAILY, G. C., LOREAU, M., GRACE, J. B., LARIGAUDERIE, A., SRIVASTAVA, D. S., NAEEM, S. Biodiversity loss and its impact on humanity. **Nature**, v. 486, p. 59-67, 2012.

CAUGHLEY, G. Sampling in aerial survey. **The Journal of Wildlife Management**, v. 41, p. 605-615, 1977.

CAUGHLEY, G., GUNN, A. **Conservation biology in theory and practice**. Cambridge, US: Blackwell Science, 1996.

CAUGHLEY, G. SINCLAIR, A. R. E. **Wildlife ecology and management**. Blackwell, Boston, 1994, 334p.

CEBALLOS, G., EHRLICH, P. P., BARNOSKY, A. D., GARCÍA, A., PRINGLE, R. M. ; PALMER, T. M. Accelerated modern human–induced species losses - Entering the sixth mass extinction. **Science Advances,** v. 1, p. e1400253, 2015.

CHAPRON, G., MIQUELLE, D. G., LAMBERT, A., GOODRICH, J. M., LEGENDRE, S., ; CLOBERT, J. The impact on tigers of poaching versus prey depletion. **Journal of Applied Ecology**, v. 45, n. 6, p. 1667-1674, 2008.

CHIARELLO, A.G. Effects of fragmentation of the Atlantic Rainforest on mammal communities in south-eastern Brazil. **BiologicalConservation**, v.87, p. 71-82. 1999.

CHIARELLO, A.G. Influência da caça ilegal sobre mamíferos e aves das matas de tabuleiro do norte do estado do Espírito Santo. **Boletim Museu de Biologia Mello Leitão**, v. 11, n. 12, p. 229-247, 2000a.

CHIARELLO, A. G. Density and population size of mammals in remnants of Brazilian Atlantic forest. **Conservation Biology**, v. 14, n. 6, p. 1649-1657, 2000b.

CLUCAS, B., K. MCHUGH, ; T. CARO. Flagship species on covers of US conservation and nature magazines. **Biodiversity and Conservation**, v. 17, p. 1517-1528, 2008.

CODY, M.L. Habitat selection in birds: the roles of vegetation structure, competitors, and productivity. **Bioscience**, v. 31, n. 2, p. 107-113, 1981.

COLLAZOS, C. C., MOSCOSO, F. I., BRAVO, D. R. Y., CASTELLANOS, A., CACERES, D. F. C., ROCA, A., BRADFIELD, R. B. Nutrition and the state of nutrition in Peru. Anales. Universidad Nacional Mayor de San Marcos. **Facultad de Medicina**, v. 43, p. 7, 1960.

CORLETT, R.T. The impact of hunting on the mammalian fauna of tropical Asian forests. **Biotropica**, v. 39, n. 3, p. 292-303, 2007.

CRAWSHAW, P. G., MAHLER, J. K., INDRUSIAK, C., CAVALCANTI, S. M. C., LEITE-PITMAN, M. R. P. SILVIUS K. M. 2004. **Ecology and conservation of the jaguar (*Panthera onca*) in Iguaçu National Park, Brazil**. Pp. 286–296 in People in nature: wildlife conservation in South and Central America. Silvius K. M., Bodmer R. E., Fragoso J. M. V. (Eds.). Columbia University Press, New York.

CULLEN, L., BODMER, R.E.; PÁDUA, C.V. Effects of hunting in habitat fragments of the Atlantic forests, Brazil. **Biological conservation**, v. 95, n. 1, p. 49-56, 2000.

DAILY, G. C., G. CEBALLOS, J. PACHECO, G. SUZAN, ; A. SANCHEZ-AZOFEIFA. Countryside biogeography of Neotropical mammals: Conservation opportunities in agricultural landscapes of Costa Rica. **Conservation Biology**, v. 17, p. 1814– 1826, 2003.

DE SOUZA, J.B.; ALVES, R.R.N. Hunting and wildlife use in an Atlantic Forest remnant of northeastern Brazil. **Tropical Conservation Science**, v. 7, n. 1, p. 145-160, 2014.

DEAN, W. With **Broadax and Firebrand: The Destruction of the Brazilian Atlantic Forest.** University of California Press, California, 1996.

DI MININ, E., SLOTOW, R., HUNTER, L. T. B., POUZOLS, F. M., TOIVONEN, T., VERBURG, P. H., LEADER-WILLIAMS, N., PETRACCA, L. ; MOILANEN, A. Global priorities for national carnivore conservation under land use change. **Science Reports**, v. 6, 23814, 2016.

DIRZO, R. **Plant-mammal interactions: lessons for our understanding of nature, and implications for biodiversity conservation**. In: Ecology: Achievement and Challenge. Press, M. C., Huntly, N. J., Levin, S. (Eds.), pp. 319–335. Blackwell Science, Oxford, 2001.

DIRZO, R. ; MIRANDA, A. **Altered patterns of herbivory and diversity in the forest understory: a case study of the possible consequences of contemporary defaunation**. In: Plant-Animal Interactions: Evolutionary Ecology in Tropical and Temperate Regions. Price, P. W., Lewinsohn, T. M., Fernandes, G. W. Benson, W. W. (Eds.). Pp. 273–287. Wiley, New York, 1991.

DIRZO, R., YOUNG, H.S., GALETTI, M., CEBALLOS, G., ISAAC, N.J.; COLLEN, B. Defaunation in the Anthropocene. **Science**, v. 345, n. 6195, p. 401-406, 2014.

DORAZIO, R. M., ; ROYLE, J. A. Estimating size and composition of biological communities by modeling the occurrence of species. **Journal of the American Statistical Association**, v. 100, n. 470, p. 389-398, 2005.

DORNELAS, M., GOTELLI, N. J., MCGILL, B., SHIMADZU, H., MOYES, F., SIEVERS, C., MAGURRAN, A. E. Assemblage time series reveal biodiversity change but not systematic loss. **Science**, v. 344, n. 6181, p. 296-299, 2014.

ECKERT, C. G., K. E. SAMIS, AND S. C. LOUGHEED. Genetic variation across species' geographical ranges: the central-margin hypothesis and beyond. **Molecular Ecology**, v. 17, p. 1170-1188, 2008.

ELITH, J.,; J. R. LEATHWICK. Species Distribution Models: Ecological Explanation and Prediction Across Space and Time. **Annual Review of Ecology, Evolution, and Systematics**, v. 40, p. 677-697, 2009.

EMMONS, L. 2016. **Cuniculus paca.** The IUCN Red List of Threatened Species 2016: e.T699A22197347. http://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T699A22197347.en. Downloaded on 08 November 2019.

FA, J.E., PERES, C.A.; MEEUWIG, J. Bushmeat exploitation in tropical forests: an intercontinental comparison. **Conservation Biology**, v. 16, n. 1, p. 232-237, 2002.

FERNANDES-FERREIRA, H., MENDONÇA, S. V., ALBANO, C., FERREIRA, F. S., ALVES, R. R. N. Hunting, use and conservation of birds in Northeast Brazil. **Biodiversity and Conservation**, v. 21, n. 1, p. 221-244, 2012.

FERNANDEZ-DUQUE, E., ROTUNDO, M. Field methods for capturing and marking Azarai night monkeys. **International Journal of Primatology**, v. 24, n. 5, p. 1113-1120, 2003.

FERREGUETTI, A.C., TOMAS, W.M.; BERGALLO, H.G. Density, occupancy, and activity pattern of two sympatric deer (Mazama) in the Atlantic Forest, Brazil. **Journal of Mammalogy**, v. 96, n. 6, p. 1245-1254, 2015.

FERREGUETTI, A.C., TOMAS, W.M.; BERGALLO, H.G. Density and niche segregation of two armadillo species (Xenarthra: Dasypodidae) in the Vale Natural Reserve, Brazil. **MammalianBiology**, v. 81, n. 2, p. 138-145, 2016a.

FERREGUETTI, A.C., TOMAS, W.M.; BERGALLO, H.G. Abundância e Densidade de Mamíferos de médio e grande porte na Reserva Natural Vale. **In:** Rolim, S.G., de Menezes, L.F.T.; Srbek-Araujo, A.C. **Floresta Atlântica de Tabuleiro: Diversidade e Endemismos na Reserva Natural Vale**. EditoraRupestre, Belo Horizonte, 2016b, p. 453-467.

FERREGUETTI, A. C., TOMAS, W. M., BERGALLO, H. G. Differences in the mammalian habitat use in a mosaic of vegetation types of an Atlantic rain-forest reserve, Brazil. **MastozoologiaNeotropica**, v. 24, p. 355-364, 2017a.

FERREGUETTI, A.C., TOMAS, W.M.; BERGALLO, H.G. Density, occupancy, and detectability of lowland tapirs, Tapirusterrestris, in Vale Natural Reserve, southeastern Brazil. **Journal of Mammalogy**, v. 98, n. 1, p. 114-123, 2017b.

FERREGUETTI, Á. C., PEREIRA-RIBEIRO, J., PREVEDELLO, J. A., TOMÁS, W. M., ROCHA, C. F. D., ; BERGALLO, H. G. One step ahead to predict potential poaching hotspots: Modeling occupancy and detectability of poachers in a neotropical rainforest. **Biological conservation**, v. 227, p. 133-140, 2018a.

FERREGUETTI, A. C., TOMAS, W. M., ; BERGALLO, H. G. Density, habitat use, and daily activity patterns of the Red-rumped Agouti (Dasyproctaleporina) in the Atlantic Forest, Brazil. **Studies on Neotropical Fauna and Environment**, v. 53, n. 2, p. 143-151, 2018b.

FERREGUETTI, A. C., ROCHA, C. D., ; BERGALLO, H. G. Poaching in non–volant mammals in the Neotropical region: the importance of a metric to assess its impacts. **Animal Biodiversity and Conservation**, v. 42, n. 2, p. 203-211, 2019.

FISKE, I. ; CHANDLER, R. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. **Journal of Statistical Software**, v. 43, n. 10, p. 1-23, 2011.

FOLEY, J. A., DEFRIES, R., ASNER, G. P., BARFORD, C., BONAN, G., CARPENTER, S. R., CHAPIN, S. F, COE, M. T., DAILY, G. C., GIBBS, H. K., HELKOWSKI, J. H., HOLLOWAY, T., HOWARD, E. A., KUCHARIK, C. J., MONFREDA, C., PATZ, J. A., PRENTICE, C. I., RAMANKUTTY. N.; SNYDER, P. K. Global consequences of land use. **Science**, v. 309, p. 570-574, 2005.

GELMAN, A., ; RUBIN, D. B. Inference from iterative simulation using multiple sequences. **Statistical Science**, v. 7, n. 4, p. 457-472, 1992.

GERBER, B. D., WILLIAMS, P. J., BAILEY, L. L. Primates and cameras. **International Journal of Primatology**, v. 35, n. 5, p. 841-858, 2014.

GHODDOUSI, A., SOOFI, M., HAMIDI, A. K., LUMETSBERGER, T., EGLI, L., ASHAYERI, S., ...; WALTERT, M. When pork is not on the menu: assessing trophic competition between large carnivores and poachers. **Biological Conservation**, v. 209, p. 223-229, 2017.

GLANZ, W.E. The terrestrial mammal fauna of Barro Colorado Island: censuses and long-term changes. **In:** LEIGH JR., E.G., RAND, A.S. ; WINDSOR, D.M. (Ed.). **The ecology of a**

tropical forest: seasonal rhythms and long-term changes. Washington, Smithsonian: Institution Press, 1996. p. 455-466.

GOMPPER, M. E., R. W. KAYS, J. C. RAY, S. D. LAPOINT, D. A. BOGAN, ; J. R. CRYAN. A comparison of noninvasive techniques to survey carnivore communities in northeastern North America. **Wildlife Society Bulletin**, v. 34, p. 1142-1151, 2006.

GOULART, F. V. B., CÁCERES, N. C., GRAIPEL, M. E., TORTATO, M. A., GHIZONI JR, I. R., ; OLIVEIRA-SANTOS, L. G. R. Habitat selection by large mammals in a southern Brazilian Atlantic Forest. **Mammalian Biology**, v. 74, n. 3, p. 182-190, 2009.

GRAY, T. N. E., PHAN, C., ; LONG, B. Modelling species distribution at multiple spatial scales: gibbon habitat preferences in a fragmented landscape. **Animal Conservation**, v. 13, n. 3, p. 324-332, 2010.

GRAY, T. N., QUANG, H. A. N., ; VAN, T. N. Bayesian occupancy monitoring for Annamite endemic biodiversity in central Vietnam. **Biodiversity and conservation**, v. 23, n. 6, p. 1541-1550, 2014.

GROSS, D. R. Protein capture and cultural development in the Amazon Basin. **American Anthropologist**, v. 77, n. 3, p. 526-549, 1975.

GUISAN, A., ; W. THUILLER. Predicting species distribution: offering more than simple habitat models. **Ecology Letters**, v. 8, p. 993-1009, 2005.

HANSEN, M. C., POTAPOV, P. V., MOORE, R., HANCHER, M., TURUBANOVA, S. A. A., TYUKAVINA, A., THAU, D., STEHMAN, S. V., GOETZ, S. J., LOVELAND, T. R., KOMMAREDDY, A., EGOROV, A., CHINI, L., JUSTICW, C. O.; TOWNSHEND, J. R. G. High-resolution global maps of 21st-century forest cover change. **Science**, v. 342, n. 6160, p. 850-853, 2013.

HAPPOLD, D. C. D. The interactions between humans and mammals in Africa in relation to conservation: a review. **Biodiversity and conservation**, v. 4, n. 4, p. 395-414, 1995.

HARRISON, R. D. Emptying the forest: hunting and the extirpation of wildlife from tropical nature reserves. **BioScience**, v. 61, n. 11, p. 919-924, 2011.

HENFREY, T. B. **Ethnoecology, resource use, conservation and development in a Wapishana community in the South Rupununi, Guyana** (Doctoral dissertation, University of Kent at Canterbury), 2002.

HERTZ, O., ; KAPEL, F. O. Commercial and subsistence hunting of marine mammals. **Ambio**, v. 15, n. 3, p. 144-151, 1986.

HILL, K.; PADWE, J. **Sustainability of Aché hunting in the Mbaracayu reserve, Paraguay**In: Hunting for sustainability in tropical forests, pp. 79-105, 2000.

HILL, K., MCMILLAN, G.; FARIÑA, R. Hunting-related changes in game encounter rates from 1994 to 2001 in the Mbaracayu Reserve, Paraguay. **Conservation Biology**, v. 17, n. 5, p. 1312-1323, 2003.

HOFFMANN, M., BELANT, J. L., CHANSON, J. S., COX, N. A., LAMOREUX, J., RODRIGUES, A. S., SCHIPPER, J. ; STUART, S. N. The changing fates of the world's mammals. Philosophical Transactions of the **Royal Society B: Biological Sciences**, v. 366, n. 1578, p. 2598-2610, 2011.

IBGE. Mapa de Vegetação do Brasil. Rio de Janeiro: Fundação Instituto Brasileiro de Geografia e Estatística. 1993.

JESUS, R.M. Mata Atlântica de Linhares: Aspectos Florestais. **In:** SEMA, IWRB ; CVRD. (Ed). **Desenvolvimento Econômico e Impacto Ambiental em Áreas de Trópico Úmido Brasileiro: a experiência da CVRD**. Rio de Janeiro: CVRD. 1987. p. 35-53.

JESUS, R.M. ; ROLIM, S.G. Fitossociologia da Mata Atlântica de Tabuleiro**. Boletim Técnico da Sociedade de InvestigaçõesFlorestais**, v. 19, p. 1-149, 2005.

JULIEN-LAFERRIÈRE, D. The influence of moonlight on activity of woolly opossums (*Caluromys philander*). **Journal of Mammalogy**, v. 78, n. 1, p. 251-255, 1997.

KARANTH, K.U., GOPALASWAMY, A.M., KUMAR, N.S., VAIDYANATHAN, S., NICHOLS, J.D.; MACKENZIE, D.I. Monitoring carnivore populations at the landscape scale: occupancy modelling of tigers from sign surveys. **Journal of Applied Ecology**, v. 48, n. 4, p. 1048-1056, 2011.

KARESH, W.B., COOK, R.A., BENNETT, E.L.; NEWCOMB, J. Wildlife trade and global disease emergence. **EmergingInfectiousDiseases**, v. 11, n. 7, p. 1000, 2005.

KEITH, D.A., H.R. AKÇAKAYA, W. THUILLER, G.F. MIDGLEY, R.G. PEARSON, S.J. PHILLIPS, H.M. REGAN, M.B. ARAÚJO, ; T.G. REBELO. Predicting extinction risks under climate change: coupling stochastic population models with dynamic bioclimatic habitat models. **Biology Letters**, v. 4, p. 560-563, 2008.

KELLNER, K. 2018. **jagsUI: A Wrapper Around 'rjags' to Streamline 'JAGS' Analyses**. R package version 1.5.0. https://CRAN.R-project.org/package=jagsUI

KÉRY, M., ; ROYLE, J. A. Hierarchical Bayes estimation of species richness and occupancy in spatially replicated surveys. **JournalofAppliedEcology**, v. 45, n. 2, p. 589-598, 2008.

KIERULFF, M.C.M., AVELAR, L.H.S., FERREIRA, M.E.S., POVOA, K.F.; BÉRNILS, R.S. Reserva Natural Vale: História e Aspectos Físicos. **Ciência&Ambiente**, v. 49, p. 7-40, 2014.

KIRKPATRICK, M., ; BARTON, N.H. Evolution of a species' range. **The American Naturalist**, v. 150, p. 1-23, 1997.

KOKKO, H., ; LÓPEZ-SEPULCRE, A. From Individual Dispersal to Species Ranges: Perspectives for a Changing World. **Science**, v. 313, p. 789-790, 2006.

KOTLER, B. P., BROWN, J., MUKHERJEE, S., BERGER-TAL, O., ; BOUSKILA, A. Moonlight avoidance in gerbils reveals a sophisticated interplay among time allocation,

vigilance and state-dependent foraging. **Proceedings of the Royal Society of London B: Biological Sciences**, v. 277, n. 1687, p. 1469-1474, 2010.

KRAMER, R. A., RICHTER, D. D., PATTANAYAK, S., ; SHARMA, N. P. Ecological and economic analysis of watershed protection in Eastern Madagascar. **Journal of Environmental Management**, v. 49, n. 3, p. 277-295, 1997.

KREBS, C.J. A review of the Chitty hypothesis of population regulation. **Canadian Journal of Zoology**, v. 56, n. 12, p. 2463-2480, 1978.

KURTEN, E. L., WRIGHT, S. J., CARSON, W. P. Hunting alters seedling functional trait composition in a Neotropical forest. **Ecology**, v. 96, n. 7, p. 1923-1932, 2015.

LAURANCE, W. F. Reflections on the tropical deforestation crisis. **Biological conservation**, v. 91, n. 2, p. 109-117, 1999.

LAURANCE, W. F., BIERREGAARD JR., R. O. **Tropical forest remnants**. Chicago, University of Chicago Press, 1997.

LAURANCE, W. F., ; VASCONCELOS, H. L. Conseqüências ecológicas da fragmentação florestal na Amazônia. **OecologiaBrasiliensis**, v. 13, n. 3, p. 434-451, 2009.

LAURANCE, W. F., SAYER, J., ; CASSMAN, K. G. Agricultural expansion and its impacts on tropical nature. **Trends in Ecology & Evolution**, v. 29, n. 2, p. 107-116, 2014. Leeuwenberg and Robinson, 2000

LEITE, M. R. P., GALVÃO, F. **El jaguar, el puma y elhombreentres áreas protegidas del bosque atlánticocostero de Paraná, Brasil.**In: El jaguar en el nuevo milenio, R. Medellin, C. Equihua, C. Chetkiewicz, P. Crawshaw, A. Rabinowitz, A. Redford, J. Robinson, E. Sanderson y A. Taber (Eds.). Fondo de Cultura Económica, Universidad Autónoma de México, WildlifeConservation Society, pp. 237-250, 2002.

LEOPOLD, A. S. **Wildlife of Mexico: the game birds and mammals**. University of California Press, Los Angeles, 1959.

LONG, R. A., P. MACKAY, W. J. ZIELINSKI, ; J. C. RAY. **Noninvasive survey methods for carnivores**. Island Press, Washington, DC, 2008.

LONG, R.A., DONOVAN, T.M., MACKAY, P., ZIELINSKI, W.J.; BUZAS, J.S. Predicting carnivore occurrence with noninvasive surveys and occupancy modeling. **Landscape Ecology**, v. 26, n. 3, p. 327-340, 2011.

MACARTHUR, R.H., DIAMOND, J.M.; KARR, J. R. Density compensation in island faunas. **Ecology**, v. 53, n. 2, p. 330-342, 1972.

MACEDO, J. S., BRANQUINHO, F. T. B., DE GODOY BERGALLO, H. A rede sociotécnica na relação entre ribeirinhos e onças (*Pantheraonca* e *Puma concolor*) nas Reservas de Desenvolvimento Sustentável Amanã e Mamirauá no Amazonas. **Desenvolvimento e Meio Ambiente**, v. 35, 2015.

MACKENZIE, D.I., NICHOLS, J.D., LACHMAN, G.B., DROEGE, S., ROYLE, J.A. ; LANGTIMM, C.A. Estimating site occupancy rates when detection probabilities are less than one. **Ecology**, v. 83, p. 2248-2255, 2002.

MACKENZIE, D.I., NICHOLS, J., ROYLE, J., POLLOCK, K., BAILEY, L. ; HINES, J. **Occupancy estimation and modeling: Inferring patterns and dynamics of species occurrence**. [s. l]: Elsevier Publishing, 2006.

MACKINNON, J. New mammals in the 21st Century? **Annals of the Missouri Botanical Garden**, v. 87, p. 63-66, 2000.

MENA, V. P., STALLINGS, J. R., REGALADO, J. B., CUEVA, R. L. **The sustainability of current hunting practices by the Huaorani**. In: Hunting for sustainability in tropical forests, p. 57-78, 2000.

MENDONÇA, L. E. T., SOUTO, C. M., ANDRELINO, L. L., SOUTO, W. M. S., VIEIRA, W. L. S., ALVES, R. R. N. Conflitos entre pessoas e animais silvestres no semiárido paraibano e suas implicações para conservação. **Sitientibus Série Ciências Biológicas**, v. 11, p. 185-199, 2011.

MILLS, L. S., M. E. SOULÉ, ; D. F. Doak. The keystone-species concept in ecology and conservation. **Bioscience**, v. 43, p. 219-224, 1993.

MILNER-GULLAND, E. J., BENNETT, E. L. Wild meat: the bigger picture. **Trends in Ecology & Evolution**, v. 18, n. 7, p. 351-357, 2003.

MITCHELL, M. G., SUAREZ-CASTRO, A. F., MARTINEZ-HARMS, M., MARON, M., MCALPINE, C., GASTON, K. J., JOHANSEN, K., ; RHODES, J. R. Reframing landscape fragmentation's effects on ecosystem services. **Trends in Ecology & Evolution**, v. 30, n. 4, p. 190-198, 2015.

MITTERMEIER, R.A.; GIL, P.R.; HOFFMANN, M.; PILGRIM, J.; BROOKS, T.; MITTERMEIER, C.G.; LAMOREUX, J.; FONSECA, G.A.B. **Hotspots revisited**. México: CEMEX/Agrupación, 2004. 390p.

MITTERMEIER, R. A., TURNER, W. R., LARSEN, F. W., BROOKS, T. M., ; GASCON, C. (2011). **Global biodiversity conservation: the critical role of hotspots**. In Biodiversity hotspots (pp. 3-22). Springer, Berlin, Heidelberg.

MORI, E., NOURISSON, D. H., LOVARI, S., ROMEO, G., ; SFORZI, A. Self-defense may not be enough: moonlight avoidance in a large, spiny rodent. **Journal of Zoology**, v. 294, n. 1, p. 31-40, 2014.

MORRONE, J. J. Biogeographical regionalization of the Neotropical region. **Zootaxa**, v. 3782, n. 1, p. 1-110, 2014.

MYERS, N. Tropical forests and their species: Going, going...? In: WILSON, E.O. ; PETER, F.M. (Ed.). **Biodiversity**.Wasington, DC: National Academic Press, 1988. p. 28-35.

MYERS, N.; MITTERMEIER, R.A.; MITTERMEIER, C.G.; FONSECA, G.A.B. ; KENT, J. Biodiversity hotspots for conservation priorities. **Nature** v. 403, p. 853-858. 2000.

NASH, L. T. Dietary, behavioral, and morphological aspects of gummivory in primates. **American Journal of Physical Anthropology**, v. 29, n. 7, p. 113-137, 1986.

NASI, R.; BROWN, D.; WILKIE, D.; BENNETT, E.; TUTIN, C.; VAN TOL, G. ; CHRISTOPHERSEN, T. **Conservation and use of wildlife-based resources: the bushmeat crisis**. Technical Series. 51p, 2008.

NAUGHTON-TREVES, L., GROSSBERG, R., TREVES, A. Paying for tolerance: rural citizens' attitudes toward wolf depredation and compensation. **Conservation biology**, v. 17, n. 6, p. 1500-1511, 2003.

NEWBOLD, T. et al. Global effects of land use on local terrestrial biodiversity. **Nature**, v. 520, n. 7545, p. 45-50, 2015.

NICHOLS, J.D. et al. Multi-scale occupancy estimation and modelling using multiple detection methods. **Journal of Applied Ecology**, v. 45, n. 5, p. 1321-1329, 2008.

NOON, B.R., BAILEY, L.L., SISK, T.D.; MCKELVEY, K.S. Efficient species-level monitoring at the landscape scale. **Conservation Biology**, v. 26, n. 3, p. 432-441, 2012.

O'CONNELL, A.F., TALANCY, N.W., BAILEY, L.L., SAUER, J.R., COOK, R.; GILBERT, A.T. Estimating site occupancy and detection probability parameters for meso-and large mammals in a coastal ecosystem. **Journal of Wildlife Management**, v. 70, n. 6, p. 1625-1633, 2006.

O'CONNELL, A. F., J. D. NICHOLS, ; K. U. KARANTH. **Camera traps in animal ecology: methods and analyses**. Springer, New York, New York, 2011.

OLMOS, F., BERNARDO, C. S. S., GALETTI, M. **O impacto dos guarani sobre Unidades de Conservação em São Paulo**. In Terras Indígenas e Unidades de Conservação da Natureza–O desafio das sobreposições territoriais. Instituto Socioambiental, São Paulo, p. 246-261, 2004.

PARRY, L.; PERES, C. Evaluating the use of local ecological knowledge to monitor hunted tropicalforest wildlife over large spatial scales. **Ecology and Society**, v. 20, n. 3, 2015.

PARRY, L., BARLOW, J.; PERES, C.A. Allocation of hunting effort by Amazonian smallholders: implications for conserving wildlife in mixed-use landscapes. **Biological Conservation**, v. 142, n. 8, p. 1777-1786, 2009.

PARZEN, E. On estimation of a probability density function and mode. **The annals of mathematical statistics**, v. 33, n. 3, p. 1065-1076, 1962.

PATTON, J.L. 2015. **Family Cuniculidae G.S. Miller and Gidley, 1918**. In: Patton, J.L., Pardiñas, U.F.J. and D'Elía, G. (eds), Mammals of South America, pp. 726-733. University of Chicago Press, Chicago and London.

PAZ Y MIÑO, G. 1988. Notas sobre lacacería y laconservación de losfélidosenlaAmazoníaecuatoriana. Fundación Simón Bolívar. **BoletínCientífico**, v. 2, n. 3, p. 1-14.

PEIXOTO, A.L. ; GENTRY, A. Diversidade e composição florística da Mata de Tabuleiro na Reserva Florestal de Linhares (Espírito Santo, Brasil). **RevistaBrasileira de Botânica**, v. 13, p. 19-25, 1990.

PERES, C. A. 1990. Effects of hunting on western Amazonian primate communities. **Biological Conservation**, v. 54, n. 1, p. 47-59.

PERES, C.A. Population status of white-lipped Tayassupecari and collared peccaries T. tajacu in hunted and unhunted Amazonian forests. **Biological Conservation**, v. 77, p. 115-123, 1996.

PERES, C.A. Effects of subsistence hunting on vertebrate community structure in Amazonian Forests. **Conservation Biology**. v. 14, n. 1, p. 240-253, 2000a.

PERES, C. A. **Evaluating the impact and sustainability of subsistence hunting at multiple Amazonian forest sites**. In J. G. Robinson and E. L. Bennett, editors. Hunting for sustainability in Tropical Forests. Columbia University Press, New York, p. 83–115, 2000b.

PERES, C.A. Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian forest vertebrates. **Conservation Biology**, v. 15, n. 6, p. 1490-1505, 2001.

PERES, C.A.; DOLMAN, P.M. Density compensation in neotropical primate communities: evidence from 56 hunted and nonhunted Amazonian forests of varying productivity. **Oecologia**, v. 122, n. 2, p. 175-189, 2000.

PERES, C. A., LAKE, I. R. Extent of nontimber resource extraction in tropical forests: accessibility to game vertebrates by hunters in the Amazon basin. **Conservation Biology**, v. 17, n. 2, p. 521-535, 2003.

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

PERES, C.A.; PALACIOS, E. Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: implications for animal-mediated seed dispersal. **Biotropica**, v. 39, n. 3, p. 304-315, 2007.

PERES, C. A., BARLOW, J., ; LAURANCE, W. F. Detecting anthropogenic disturbance in tropical forests. **Trends in ecology & evolution**, v. 21, n. 5, p. 227-229, 2006.

PERES, C.A.; EMILIO, T.; SCHIETTI, J.; DESMOULIÈRE, S.J.M. ; LEVI, T. Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. **Proceedings of the National Academy of Sciences**, v. 113, n. 4, p. 892-897, 2016.

PHILLIPS, S.J., M. DUDIK, J. ELITH, C.H. GRAHAM, A. LEHMANN, J. LEATHWICK, ; S. FERRIER. Sample selection bias and presence-only distribution models: implications for background and pseudo-absence data. **Ecological Applications**, v. 19, p. 181-197, 2009.

PIERRET, P. V., DOUROJEANNI, M. J. La caza y laalimentacion humana enlasriberasdel rio Pachetea, Perua. **Turrialba**, v. 16, n. 3, p. 271-28, 1966.

PLUMMER, M. **JAGS: A Program for Analysis of Bayesian Graphical Models Using Gibbs Sampling**. In K Hornik, F Leisch, A Zeileis (eds.), Proceedings of the 3rd International Workshop on Distributed Statistical Computing, Vienna, Austria, V. 124, N. 125, p. 10, 2003.

QGIS DEVELOPMENT TEAM. **QGIS Geographic Information System**. Open Source Geospatial Foundation Project, 2017. Available:. http://qgis.osgeo.org.

R DEVELOPMENT CORE TEAM, 2016. R: **A Language and Environment for Statistical Computing**. R Foundation for StatisticalComputing, Vienna, Austria. disponível em <http:// www.R-project.org>. Acessado em 15 de agosto de 2019.

REDFORD, K.H. The empty forest. **Bioscience**, v. 42, p. 412-422, 1992.

REDFORD, K. H., ROBINSON, J. G. The game of choice: patterns of Indian and colonist hunting in the Neotropics. **American anthropologist**, v. 89, n. 3, p. 650-667, 1987.

REID, F. 2016. *Hydrochoerushydrochaeris*. The IUCN Red List of Threatened Species 2016: e.T10300A22190005. http://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T10300A22190005.en. Downloaded on 08 November 2019.

RENCTAS. **1º relatório nacional sobre o tráfico de fauna silvestre**. Brasília, Brazil, 2001.

REZENDE, C. L., SCARANO, F. R., ASSAD, E. D., JOLY, C. A., METZGER, J. P., STRASSBURG, B. B. N., TABARELLI, M., FONSECA, G. A., MITTERMEIER, R. A. From hotspot to hopespot: An opportunity for the Brazilian Atlantic Forest. **Perspectives in Ecology and Conservation**, v. 16, n. 4, p. 208-214, 2018.

RIBEIRO, M.C.; METZGER, J.P.; MARTENSEN, A.C.; PONZONI, F.J. ; HIROTA, M.M. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. **Biological Conservation**, v. 142, p.1141-1153. 2009.

RIPPLE, J. ; PERRINE, D. **Manatees and Dugongs of the World**. Voyageur Press. Stillwater, 1999.

RITO, K. F., ARROYO-RODRÍGUEZ, V., QUEIROZ, R. T., LEAL, I. R., ; TABARELLI, M. Precipitation mediates the effect of human disturbance on the Brazilian Caatinga vegetation. **Journal of Ecology**, v. 105, n. 3, p. 828-838, 2016.

ROBINSON, J. G., BENNETT, E. L. **Hunting for sustainability: the start of a synthesis**. In: Hunting for sustainability in tropical forest, p.499-519, 1999.

ROBINSON, J. G. REDFORD, K. H. **Neotropical wildlife use and conservation**. Chicago, University of Chicago Press, 1991.

ROBINSON, J. G., REDFORD, K. H., ; BENNETT, E. L. Wildlife harvest in logged tropical forests. **Science**, v. 284, n. 5414, p. 595-596, 1999.

ROSEN, G.E.; SMITH, K.F. Summarizing the evidence on the international trade in illegal wildlife. **EcoHealth**, v. 7, n. 1, p. 24-32, 2010.

ROSENBLATT, M. Remarks on some nonparametric estimates of a density function. **The Annals of Mathematical Statistics**, v. 27, n. 3, p. 832-837, 1956.

RUSSELL, R. E., ROYLE, J. A., SAAB, V. A., LEHMKUHL, J. F., BLOCK, W. M., ; SAUER, J. R. Modeling the effects of environmental disturbance on wildlife communities: avian responses to prescribed fire. **Ecological Applications**, v. 19, n. 5, p. 1253-1263, 2009.

SCHWITZER, N., RANDRIATAHINA, G. H., KAUMANNS, W., HOFFMEISTER, D., ; SCHWITZER, C. Habitat utilization of blue-eyed black lemurs, Eulemurmacacoflavifrons (Gray, 1867), in primary and altered forest fragments. **Primate Conservation**, v. 22, n. 1, p. 79-87, 2007.

SEXTON, J.P., P.J. MCINTYRE, A.L. ANGERT, ; K.J. RICE. Evolution and Ecology of Species Range Limits. **Annual Review of Ecology, Evolution, and Systematics**, v. 40, p. 415-436, 2009.

SILVIUS, K. M., BODMER, R. E., FRAGOSO, J. M. **People in nature: wildlife conservation in South and Central America**. Columbia University Press, 2004.

SOS MATA ATLÂNTICA. **Atlas dos Remanescentes Florestais da Mata Atlântica - Período 2008-2014**. Relatório Final. São Paulo: Fundação SOS Mata Atlântica ; Instituto Nacional de Pesquisas Espaciais, 2014. 356p.

SOULÉ, M.E. **Conservation Biology: The Science of Scarcity and Diversity**. Sunderland, MA: Sinauer, 1986.

SOUSA, J.A.C.; SRBEK-ARAUJO, A.C. Are we headed towards the defaunation of the last large Atlantic Forest remnants? Poaching activities in one of the largest remnants of the Tabuleiro forests in southeastern Brazil. **Environmental Monitoring and Assessment**, v. 189, n. 3, p. 129, 2017.

STANFORD, C. B., BUNN, H. T. **Meat-eating and human evolution**. Oxford University Press, 2001.

STEARMAN, A. M. **A pound of flesh: social change and modernization as factors in hunting sustainability among neotropical indigenous societies**. In: Hunting for sustainability in tropical forests, p. 233-250, 2000.

STEFFEN, W., RICHARDSON, K., ROCKSTRÖM, J., CORNELL, S. E., FETZER, I., BENNETT, E. M., BIGGS, R., CARPENTER, S. R., DE VRIES, W., DE WIT, C. A., FOLKE, C., GERTEN, D., HEINKE, J., MACE, G. M., PERSSON, L. M., RAMANATHAN,

V., REYERS, B; SORLIN, S. Planetary boundaries: Guiding human development on a changing planet. **Science**, v. 347, n. 6223, p. 1259855, 2015.

STONER, K. E., VULINEC, K., WRIGHT, S. J., PERES, C. A. Hunting and plant community dynamics in tropical forests: a synthesis and future directions. **Biotropica**, v. 39, n. 3, p. 385-392, 2007.

TABARELLI, M., PINTO, L. P., SILVA, J. M. C., HIROTA, M. M., ; BEDÊ, L. C. Desafios e oportunidades para a conservação da biodiversidade na Mata Atlântica brasileira. **Megadiversidade**, v. 1, n. 1, p. 132-138, 2005.

TAKEKAWA, D. Hunting method and the ecological knowledge of dolphins among the Fanalei villagers of Malaita, Solomon Islands. **SPC Traditional Marine Resource Management and Knowledge Information Bulletin**, v. 12, p. 3-11, 2000.

TERBORGH, J. Maintenance of diversity in tropical forests. **Biotropica**, v. 24, n. 2, p. 283-292, 1992.

TERBORGH, J. 2000. The fate of tropical forests: a matter of stewardship. **Conservation Biology**, v. 14, n. 5, p. 1358-1361, 2000.

TERBORGH, J., LOPEZ, L., NUÑEZ, P., RAO, M., SHAHABUDDIN, G., ORIHUELA, G., ... & BALBAS, L. Ecological meltdown in predator-free forest fragments. **Science**, v. 294, n. 5548, p. 1923-1926, 2001.

TERBORGH, J., NUÑEZ-ITURRI, G., PITMAN, N. C., VALVERDE, F. H. C., ALVAREZ, P., SWAMY, V., ... & PAINE, C. T. Tree recruitment in an empty forest. **Ecology**, v. 89, n. 6, p. 1757-1768, 2008.

TEWS, J., BROSE, U., GRIMM, V., TIELBÖRGER, K., WICHMANN, M. C., SCHWAGER, M., ; JELTSCH, F. Animal species diversity driven by habitat heterogeneity diversity: the importance of keystone structures. **Journal of Biogeography**, v. 31, n. 1, p. 79-92, 2004.

THOMAS, L., BUCKLAND, S. T., REXSTAD, E. A., LAAKE, J. L., STRINDBERG, S., HEDLEY, S. L., ... & BURNHAM, K. P. Distance software: design and analysis of distance sampling surveys for estimating population size. **Journal of Applied Ecology**, v. 47, n. 1, p. 5-14, 2010.

TOBLER, M. W., S. E. CARRILLO-PERCASTEGUI, R. LEITE PITMAN, R. MARES,; G. POWELL. An evaluation of camera traps for inventorying large andmedium sized terrestrial rainforest mammals. **Animal Conservation**, v. 11, p. 169-178, 2008.

TOWNSEND, W. R. **The sustainability of subsistence hunting by the Sirionó Indians of Bolivia**. In: Hunting for sustainability in tropical forests, p. 267-281, 2000.

TREVES, A., WALLACE, R. B., NAUGHTON-TREVES, L., MORALES, A. Co-managing human–wildlife conflicts: a review. **Human Dimensions of Wildlife**, v. 11, n. 6, p. 383-396, 2006.

ULLOA, A., RUBIO-TORGLER, H., CAMPOS-ROZO, C., SILVIUS, K. M., BODMER, R. E., FRAGOSO, J. M. V. **Conceptual basis for the selection of wildlife management strategies by the Embera people in Utria National Park, Choco, Colombia**. In: People in Nature: Wildlife Conservation in South and Central America. Columbia University Press, New York, p. 11-36, 2004.

VELLEND, M., VERHEYEN, K., FLINN, K. M., JACQUEMYN, H., KOLB, A., VAN CALSTER, H., ... ; BRUNET, J. Homogenization of forest plant communities and weakening of species–environment relationships via agricultural land use. **Journal of Ecology**, v. 95, n. 3, p. 565-573, 2007.

VICKERS, W. T. Hunting yields and game composition over ten years in an Amazon Indian territory. **Neotropical wildlife use and conservation**, v. 400, p. 53-81, 1991.

WHITMORE, T.C. Tropical forest disturbance, disappearance, and species loss. In: LAURENCE, W.F. ; BIERREGAARD JR., O. (Ed.). **Tropical Forest Remnants**. Chicago: The University of Chicago Press, 1997. p. 3-12

WILKIE, D., SHAW, E., ROTBERG, F., MORELLI, G., ; AUZEL, P. Roads, development, and conservation in the Congo Basin. **Conservation Biology**, v. 14, n. 6, p. 1614-1622, 2000.

WILKIE, D.S., BENNETT, E.L., PERES, C.A.; Cunningham, A.A. The empty forest revisited. **Annals of the New York Academy of Sciences**, v. 1223, n. 1, p. 120-128, 2011.

WOLF, C., ; RIPPLE, W. J. Prey depletion as a threat to the world's large carnivores. **Royal Society Open Science**, v. 3, n. 8, p. 160252, 2016.

WRIGHT, S.J. The myriad consequences of hunting for vertebrates and plants in tropical forests. **Perspectives in Plant Ecology, Evolution and Systematics**, v. 6, n. 1-2, p. 73-86, 2003.

WRIGHT, S. J., ZEBALLOS, H., DOMÍNGUEZ, I., GALLARDO, M. M., MORENO, M. C., IBÁÑEZ, R. Poachers alter mammal abundance, seed dispersal, and seed predation in a Neotropical forest. **Conservation Biology**, v. 14, n. 1, p. 227-239, 2000.

WRIGHT, S. J., HERNANDÉZ, A., CONDIT, R. The bushmeat harvest alters seedling banks by favoring lianas, large seeds, and seeds dispersed by bats, birds, and wind. **Biotropica**, v. 39, n. 3, p. 363-371, 2007.

YOCCOZ, N.G., NICHOLS, J.D.; BOULINIER, T. Monitoring of biological diversity in space and time. **Trends in Ecology & Evolution**, v. 16, n. 8, p. 446-453, 2001.

YOUNG, J.K., OLSON, K.A., READING, R.P., AMGALANBAATAR, S.; BERGER, J. Is wildlife going to the dogs? Impacts of feral and free-roaming dogs on wildlife populations. **BioScience**, v. 61, n. 2, p. 125-132, 2011.

ZIPKIN, E. F., DEWAN, A., ; ANDREW ROYLE, J. Impacts of forest fragmentation on species richness: a hierarchical approach to community modelling. **Journal of Applied Ecology**, v. 46, n. 4, p. 815-822, 2009.

ZURELL, D., F. JELTSCH, C.F. DORMANN, ; B. SCHRÖDER. Static species distribution models in dynamically changing systems: how good can predictions really be? **Ecography**, v. 32, p. 733-744, 2009.

ZURELL, D., W. THUILLER, J. PAGEL, J.S. CABRAL, T. MÜNKEMÜLLER, D. GRAVEL, S. DULLINGER, S. NORMAND, KH. SCHIFFERS, KA. MOORE, ; N.E. ZIMMERMANN. Benchmarking novel approaches for modelling species range dynamics. **Global Change Biology**, v. 22, p. 2651-2664, 2016.