

DISENTANGLING THE EFFECTS OF DEFORESTATION AND INDIGENOUS
HUNTING ON WILDLIFE IN THE AMAZON

By

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1

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To my father

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TABLE OF CONTENTS

	<u>page</u>
ACKNOWLEDGMENTS.....	4
LIST OF TABLES.....	8
LIST OF FIGURES.....	9
LIST OF ABBREVIATIONS	10
ABSTRACT.....	11
CHAPTER	
1 INTRODUCTION	13
The Bushmeat Crisis in the Amazon.....	13
Wildlife Conservation in Amazonian Protected Areas	14
Pano People and Wildlife in Acre.....	15
Assessing Wildlife Abundance Using Hunting Data	15
Objectives	18
Research Questions	18
2 METHODOLOGY	19
Study Area	19
Acre Ecological-Economic Zoning	20
Pano Hunting	20
Data Collection and Sources.....	21
Hunting Data	21
Socio-economic Data.....	22
Spatial Data.....	22
Wildlife Status Indicators	23
Statistical analysis	25
Selecting Appropriate Indicator of Distance of Preferred Hunted Animals to the Village	25
Drivers of Wildlife Depletion across Pano Communities	25
Linear multiple regression.....	25
Spatial dependency.....	26
Capacity of indigenous lands to protect wildlife against external threats....	27
3 RESULTS.....	30
Pano Hunter's Prey Profiles	30
Information Provided by Indicators of Wildlife Status.....	30
Drivers of Wildlife Depletion across Pano Communities	31

Spatial Dependence	34
Capacity of Indigenous Lands to Protect Wildlife against External Threats	34
4 DISCUSSION.....	51
Wildlife Status nearby Pano Communities.....	51
Drivers of Wildlife Depletion	52
Effects of Indigenous Hunting Pressure on Game.....	53
Effects of Deforestation around Villages on Game.....	54
Effects of Surrounding Disturbance: the Acre State Zoning.....	56
Effects of Roads on Wildlife and Hunters.....	58
Source-sink Dynamics of Game Populations	60
Indigenous Lands as Protected Areas for Wildlife.....	61
Management Implications and State Conservation Policy.....	62
Considerations on Scale and the Absence of Hunting Effect	64
5 CONCLUSIONS.....	70
APPENDIX	
A KERNEL DENSITY ESTIMATES OF DISTANCE OF PREFERRED ANIMALS HUNTED FROM THE VILLAGE	71
B WILDLIFE STATUS INDICATORS AND ITS RELATED ECOLOGICAL MODELS AND THEORIES	73
C PEARSON'S CORRELATION MATRIX OF VARIABLES POSSIBLY INFLUENCING HUNTING IN INDIGENOUS VILLAGES, CONSIDERING 29 VILLAGES.....	75
LIST OF REFERENCES	78
BIOGRAPHICAL SKETCH.....	95

LIST OF TABLES

<u>Table</u>	<u>page</u>
2-1 Defensibility of Pano Indigenous Lands against external threats.....	29
3-1 Wildlife species hunted by Pano communities in Acre, Brazilian Amazon, during this study	37
3-2 Relative importance of prey taxa to Pano communities.....	39
3-3 Characteristics of Pano villages in the State of Acre, Brazilian Amazon, and absent species.....	39
3-4 Pearson correlation matrix between simple indicator variables of wildlife depletion in Pano villages	41
3-5 Principal Component Analysis axis of wildlife abundance simple indicators	41
3-6 Backward stepwise regression models for all response variables at the village level	42
3-7 Explanatory variables included in the final regression models after stepwise and supervised selection	43
3-8 Spatial dependence of response variables and residuals of final regression models	43
3-9 Student's t tests on wildlife indicators between villages protected and vulnerable to external threat	44
3-10 Pearson's correlations between indicators of wildlife status that were driven by deforestation and the total deforestation within 5km radius area	44
3-11 Pano Indigenous Lands with total area that would support the average local people harvesting of preferred species in the Amazon.....	45

LIST OF FIGURES

<u>Figure</u>		<u>page</u>
2-1	Kaxinawá and Katukina Indigenous Lands studied in Acre State, Brazilian Amazon.....	28
3-1	Relative contribution of wildlife species to Pano diet	46
3-2	Significant Pearson's correlation between wildlife status indicators and the proportion of individual species, major taxa, and preference group, after Bonferroni correction	47
3-3	Minimal linear regression models after backward stepwise selection.....	48
3-4	Examples of non significant Pearson's correlation between hunting pressure, deforestation and indicators of wildlife status.....	49
3-5	Box-plot of wildlife indicators among Indigenous Lands of different levels of defensibility against external threats to wildlife	50

LIST OF ABBREVIATIONS

AMAAI-AC	Associação do Movimento dos Agentes Agroflorestais Indígenas do Acre
CPI-AC	Comissão Pró-Índio do Acre
CPUE	Capture-per-unit-effort
IAA	Indigenous Agroforestry Agents
IL	Indigenous Land
PRODES	Projeto Prodes - Monitoramento da Floresta Amazônica Brasileira por Satélite

Abstract of Thesis Presented to the Graduate School
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DISENTANGLING THE EFFECTS OF DEFORESTATION AND INDIGENOUS
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The decline in wildlife populations in the Amazon threatens game species and forest dwellers subsisting from hunting. Deforestation and overhunting are the most commonly reported causes of depletion. Few studies assessed game drivers at regional scale, or they did not consider the effects of hunting and habitat loss. Indigenous people in the Brazilian State of Acre largely rely on wildmeat, although many villages experience game depletion. I worked with the Pano people of Acre to understand the drivers of game depletion across villages and Indigenous Lands. I derived four simple indicators and two multivariate indicators of wildlife status from ecological models to analyze 9109 hunting records from 35 Pano indigenous villages of eight Indigenous Lands between 2005 and 2009 in the Alto Juruá and Purus valleys. I conducted multiple regression analyses to identify the drivers of variation in animal dispersion, game populations, richness of sensitive species, and hunting success across villages. I supplemented these regressions with ANOVA of wildlife status in relation to village vulnerability and Indigenous Lands defensibility to external threats. Indigenous hunting pressure was not associated to forest loss in Pano hunting territory. Deforestation explained between 30% and 56% of variation in mean prey weight, richness of sensitive

species and capture-per-unit-effort. Indigenous density and road presence explained 30% of variation in the distance of animals from villages. Deforestation and hunting may have affected game species at two different spatial scales, the former at a large scale whereas the latter at a local scale. Deforestation and its related disturbances were the probable drivers of game depletion, possibly leading to the extirpation of sensitive species. Pano hunting pressure might only affect wildlife close to villages by changing population dispersion, although, not necessarily depleting them. Vulnerable villages and Indigenous Lands with low defensibility against external threats had more depleted wildlife than protected ones. Non-indigenous hunting and deforestation outside Indigenous Lands probably negatively affected wildlife hunted by Pano people. Other studies indicate hunting pressure as the main cause of depletion. The regional scale of a heterogeneous landscape intensively hunted for decades might help to explain the difference found in the drivers of game depletion across Pano villages. Deforestation can determine game conservation even in a little altered Amazon region. Indigenous Lands may be able to inhibit forest loss but not game depletion. The future scenario of reduced deforestation foreseen by the Acre State Government, depending on the execution of recent state policies, may avoid the current trend of regional wildlife depletion as a result of habitat loss.

CHAPTER 1 INTRODUCTION

The Bushmeat Crisis in the Amazon

The recent decline in populations of large-bodied game species in tropical forests worldwide led to the recognition of a global “bushmeat crisis”, which threatens forest ecosystems and its people (Milner-Gulland et al. 2003; Nasi et al. 2008). Large bodied mammals provide most of the wildmeat consumed in the tropics (Jerozolimski and Peres 2003; Fa et al. 2005) and their survival is threatened mainly by habitat loss occurring at unprecedented rates (Brook et al. 2008; Davidson et al. 2009) and overhunting to supply rural and urban populations (Isaac and Cowlshaw 2004; Robinson and Bennett 2004; Cardillo et al. 2005; Peres and Palacios 2007). In terms of conservation policy, however, the important debate over the relative threats posed by deforestation and hunting pressure remains open, even though these factors may interact (Brook et al. 2008; Laurance and Useche 2009). Hunters in unaltered forests represent the single anthropogenic disturbance to game populations. Even indigenous people hunting for subsistence can locally deplete wildlife (Souza-Mazurek et al. 2000; Peres and Nascimento 2006). It is controversial, though, if their impact alone depletes game species at regional scales, given their centenary widespread use of wildlife (Robinson and Bodmer 1999; Smith 2005; Milner-Gulland and Rowcliffe 2007; Adeney et al. 2009; Levi et al. 2009; Rist et al. 2009).

Several Neotropical indigenous groups increasingly hunt in regions that encompass both undisturbed and anthropogenically altered habitats (Smith 2005, 2008; Gavin 2007), and are potentially subject to multiple drivers of wildlife depletion. Yet, many studies did not properly differentiate between these factors (Peres 2001; Daily et

al. 2003; Smith 2005; Fa 2007; Fa and Brown 2009) or were conducted at scales inappropriate to policy formulation, given that the effects of these drivers change depending on the scale of study (Palmer and White 1994; Fragoso 1999; Ferrier 2002; Fisher and Owens 2004; Pautasso 2007; Cardillo et al. 2008; Fritz et al. 2009).

Wildlife Conservation in Amazonian Protected Areas

Protected Areas in the Amazon are mostly established to prevent and combat threats to biodiversity (Margules and Presley 2000; Brooks et al. 2009). Although titled for different purposes, Indigenous Lands are fundamental for conservation (Schwartzman and Zimmerman 2005; Malhi et al. 2008) because they cover approximately 21% of the Brazilian Amazon (Carneiro Filho and Souza 2009). They figure among the most effective Amazonian Protected Areas in maintaining biodiversity, and inhibiting deforestation and forest degradation (Asner et al. 2005; Oliveira et al. 2007; Adeney et al. 2009) possibly better than other Conservation Units in Brazil (Nepstad et al. 2006). Nonetheless, the consequences of indigenous subsistence hunting have been used to argue against the sustainable use policy inside Protected Areas (Redford and Sanderson 2000; Schwartzman et al. 2000; Peres and Zimmerman 2001; Schwartzman and Zimmerman 2005; Sunderland et al. 2008) in a debate that questions the conservation outcome of Protected Areas of sustainable use (Peres and Zimmerman 2001; Terborgh 2004; Brockington et al. 2006; Sirén 2006; Buck et al. 2007; Hames 2007). The degree to which Indigenous Lands are able to conserve wildlife remains unclear because of the absence of available data at regional scale (Barreto et al. 2006; Nepstad et al. 2006; Adeney et al. 2009).

Pano People and Wildlife in Acre

The debate and issues presented above is exemplified by the Pano Indigenous Lands in the Brazilian State of Acre, southwestern Amazon, a region characterized by increasing deforestation (yet low in comparison to other Brazilian states; Acre 2009; Hayashi et al. 2009), elevated hunting pressure (Peres and Palacios 2007) but where most large game species are still hunted (Calouro 2005; Ramos 2005; Constantino et al. 2008). Indigenous Lands are an important component of Acre environmental policy and conservation strategy, covering 14% of the state (Iglesias and Aquino 2006; Acre 2008). The Kaxinawá and Katukina are peoples of the Pano linguistic family, the most populous group in the region, for whom large game species are the preferred source of meat (Kensinger 1975, 1983; Deshayes 1986; Aquino and Iglesias 1994; Cunha and Almeida 2002; Amaral 2005). The reduced abundance of preferred game species in some Indigenous Lands, however, shifts reliance to alternative meats, purchased in city markets or raised in villages (Lima 2001; Calouro 2007). It has been suggested that the Kaxinawá hunting pressure had extirpated several species (Peres and Zimmerman 2001), and these data were used to understand hunting impacts on wildlife in the Amazon basin (Peres 2000; Peres and Palacios 2007). Other evidence suggests, however, that some Kaxinawá Indigenous Lands have allowed recovery of several depleted game species (Constantino et al. 2008).

Assessing Wildlife Abundance Using Hunting Data

Human-mediated disturbances can induce three main negative responses of game assemblages: 1) reduction in population abundance, 2) species extirpation, and 3) dispersion of individuals and populations. However, this is not exhaustive (e.g. some

species can increase abundance, or animals can differ in stress levels, body conditions, etc.).

Surveying vertebrates by direct observation is hard and expensive in tropical forests (Naughton-Treves et al. 2003; Fa et al. 2005; Norris et al. 2008), especially in harvested areas where animals are wary of people (Fitzgibbon 1998), imposing limitations on this method (Carrillo et al. 2000). Consequently, research on hunter societies and wildlife conservation, often uses hunting data and a variety of indicators to assess population abundance as well as identify its drivers (Appendix A; Juste et al. 1995; Souza-Mazurek et al. 2000; Nagaoka 2002; Rowcliffe et al. 2003; Fa et al. 2004; Crookes et al. 2005; Franzen 2006; Albrechtsen et al. 2007; Ohl-Schacherer et al. 2007; Norris et al. 2008; Levi et al. 2009).

The central place foraging model links resource depletion to local harvesting intensity. This model assumes that hunting pressure within a limited territory departing from a fixed settlement first depletes wildlife close to the settlement, subsequently increasing the distance where species sensitive to hunting are caught from the settlement (Sirén et al. 2004; Levi et al. 2009). Therefore, the distance that desirable species are hunted from settlements indicates the wildlife status in response specifically to central place hunting pressure (Alvard et al. 1997; Hill et al. 1997; Cannon 2003; Ohl-Schacherer et al. 2007; Smith 2008; Levi et al. 2009).

The status of wildlife population can also be assessed by indicators derived from Optimal Foraging Theory. Crudely, this theory predicts that hunters hunt more animals of lower ranked species, as the relative cost of hunting them declines compared to higher ranked species (Hames and Vickers, 1982; Smith 1983; Winterhalder and Lu

1997; Rowcliffe et al. 2003). The addition of new species to, and increasing frequencies of less preferred species in hunter's prey profile, is a response to the decline in abundance of preferred species (Smith 1983; Rowcliffe et al. 2005; Parry et al. 2009a). This framework has been applied to assess the effects of indigenous and non-indigenous Amazonian hunting (Alvard 1993, 1995; Hames and Vickers 1982, Jerozolinski and Peres 2003). The premises of Optimal Foraging Theory applied to hunters that prefer large-bodied animals allowed researchers to use the mean weight of hunted animals to assess the variation of wildlife status (Jerozolinski and Peres 2003; Fa et al. 2005; Franzen 2006; Ohi-Schacherer et al. 2007; Constantino et al. 2008; Parry et al. 2009a).

The capture-per-unit-effort (CPUE) of game species is often used to evaluate variation in populations' abundance, given that hunters have to spend more time hunting in depleted sites to have similar return rates compared to non depleted sites (Souza-Mazurek et al. 2000; Hill et al. 2003; Sirén et al. 2004; Puertas and Bodmer 2005). While mean prey weight indicates the status of a group of high ranked species, CPUE allows the assessment of the abundance of individual species (Sirén et al. 2004; Parry et al. 2009a, b). Nevertheless, CPUE is a measurement of effort and should be carefully analyzed in order to provide estimates of wildlife status (Rist et al. 2008).

In the extreme cases of extirpation or drastic reduction in density, the species is no longer present in the hunter's prey assemblage. Certain large primates, birds and ungulates in the Amazon are less resilient to habitat disturbance and hunting pressure, and thus, more prone to extirpation (Robinson 1996; Peres 2000; Peres et al. 2003; Peres and Nascimento 2006; Michalski and Peres 2007; Barlow et al. 2007; Peres and

Palacios 2007; Boyle 2008; Takahashi 2008; Parry et al. 2009b). These species are often used as indicators of wildlife community status in the Neotropics (Daily et al. 2003; Parry et al. 2007). Given that these species are ranked high among Amazonian hunters, their absence in prey profiles may be a proxy of wildlife depletion close to hunter settlements instead of a consequence of narrowing diet breadth expected by Optimum Foraging Theory, in cases of high abundances of preferred prey (Smith 1983).

Objectives

In this study I analyzed the regional scale effects of deforestation and hunting on wildlife consumed in Indigenous Lands based on hunting data from Pano villages in Acre. I also related wildlife status to the level of defensibility of Indigenous Lands against non-indigenous disturbances. I developed indicators based on ecological models to assess different perspectives on wildlife responses to disturbances, and used multivariate indicators to combine these responses. Findings are discussed regarding the recent environmental and development political agenda of Acre State.

Research Questions

1. What are the drivers of variation of wildlife hunted by Pano people at the village level?
2. Are Pano Indigenous Lands able to conserve wildlife against surrounding threats?

CHAPTER 2 METHODOLOGY

Study Area

The eight Pano Indigenous Lands studied in the Brazilian State of Acre, southwestern Amazon, were titled between the 1980s and 2002. They are located in the Alto Juruá and Alto Purus valleys in Acre State, specifically in two tributaries of the Juruá river, the Breu and Tarauacá, and two tributaries of the Tarauacá river, the Jordão and Humaitá rivers, as well as along the Purus river (Figure 2-1). Approximately 4500 people live in these Indigenous Lands, comprising about 50% of titled areas for Pano people in Acre. Villages vary in their degree of linkage to markets in the most populous cities of Cruzeiro do Sul, Tarauacá and Feijó. The small urban centers of Jordão, Santa Rosa do Purus and Marechal Thaumaturgo have some of the lowest HDI in Brazil (PNUD 2003). Villages in Campinas and Igarapé do Caucho Indigenous Lands have road access to cities, whilst other villages access market by boat (Figure 2-1).

The region is embedded in the Southwestern Amazon Moist Forest ecoregion (TNC 2009), one of the priority ecoregions for Neotropical terrestrial vertebrate conservation, in particular for endemic species (Loyola et al. 2009). Predominant vegetation can be characterized as typical Terra Firme forest, dominated by open canopy with palm trees and natural patches of bamboo in clay soils (Silveira et al. 2002; Acre 2006). The exception is the Alto Purus Indigenous Land that has portions of the Iquitos Várzea ecoregion. Precipitation ranges from 1600 to 2000 mm/year (Sombroek 2001), causing little variation in ungulate abundances (Robinson and Bennett 2004; Mandujano and Naranjo 2010).

Acre Ecological-Economic Zoning

The Pano Indigenous Lands are under low threat compared to other Indigenous Lands in the Amazon (Carneiro Filho and Souza 2009). Within the state of Acre, however, Indigenous Lands are located in zones of different land use and development planning. The State Ecological-Economic Zoning indicated four major zones of planning, three of which are relevant to this study. The area surrounding the highways composes Zone 1, characterized as the development frontier of the state. Zone 2 encompasses all Protected Areas, including Indigenous Lands. Zone 4 comprises urban areas and its surroundings, sometimes including Indigenous Lands (Acre 2000). Beyond this zoning, the studied region can be spatialized as: 1) the development frontier alongside the BR-364, comprising the largest cities and the Campinas, Igarapé do Caucho and Praia do Carapanã Indigenous Lands, and 2) the international frontier with Peru, comprising most Protected Areas and the other Pano Indigenous Lands (Figure 2-1). The mosaic of Protected Areas along the border with Peru composes the Western Amazon Ecological Corridor (Brackelaire 2005).

Pano Hunting

The Kaxinawá and Katukina share cultural similarities, general to most Pano people. Hunting is a prestigious male activity, and wildlife is fundamental to structure communities through meat sharing, providing meat, and is strongly present in medicine and cosmology (Kensinger 1975, 1995; Almeida et al. 2002; Lima 2002; Aquino and Iglesias 1994; Lagrou 2004). Hunters prefer larger animals (Table 2-1) such as ungulates, large primates, reptiles, and understory birds, with few exceptions, that are hunted whenever encountered. Smaller species of secondary preference are hunted incidentally when hunters do not succeed in hunting preferred animals. The most

common hunting strategy is the single man one-day search for animals following defined tracks and its surroundings (Deshayes 1986; Navarro 2004; Constantino et al. 2008). The shotgun has substituted traditional bow-and-arrows for hunting. The ideal hunting territory of an isolated Pano village has a circular shape smaller than 6km from the center of the village, which is exclusively used by village members (Kensinger 1975; Navarro 2004). Although the actual Pano hunting territory is modified by the presence of nearby villages or Indigenous Land's boundary limits to reduce overlap between adjacent villages, the maximum distance of one-day hunts does not go beyond 6km (Bant and Pessoa 2008). One way Pano hunters evaluate wildlife abundance is estimating the straight distance, in minutes walking, to where the highest number of preferred animals is hunted (Kensinger 1975; Constantino et al. 2008).

Data Collection and Sources

Hunting Data

Kaxinawá and Katukina representatives participate in a long-term capacity-building program for environmental management led by the local NGO Comissão Pró-Índio do Acre (CPI-AC). These representatives are paid by the State Government to provide extension work as Indigenous Agroforestry Agents (IAAs) in their communities. The wildlife monitoring program was added to the capacity-building program in 2004 and is facilitated by the IAA, who engages other community members (Constantino et al. 2008). In Acre, 45 IAAs collect hunting data in spreadsheets containing hunting effort (date, number of hunters, and time spent hunting) and success (species, weight, sex, approximate age, and straight distance where caught from the village). Between 2004 and 2009 the total number of hunted animals recorded by these villages was 13,540. Out of this population, I selected 35 villages (33 Kaxinawá and two Katukina) from eight

Indigenous Lands, corresponding to approximately 33% of all Kaxinawá and 28% of all Katukina villages in Acre excluding villages according to the following criteria: 1) hunting records only to the first year of monitoring training, 2) less than six months of hunting records and less than 50 hunted animals recorded, 3) only records on preferred species hunted. The 35 selected villages recorded hunting 9109 animals between 2005 and 2009. I surveyed the georeferenced locations and recorded the IAA distance estimation in minutes of 65 hunted animals and landscape features that orient the indigenous hunts in 2006 and 2009 using GPS Garmin Csx60, and used the relationship to convert the distance of hunted animals from the village from minutes to meters.

Socio-economic Data

I used socio-economic data (e.g. population, number of hunters and employees, permanent goods, livestock etc.) collected from villages in Acre by CPI-AC since 2004. In addition, I updated the information for 17 communities during workshops held in villages between April and July 2009.

Spatial Data

Deforestation maps were obtained from the Program for Monitoring Deforestation in the Brazilian Amazon (PRODES; <http://www.dpi.inpe.br/prodesdigital/prodes.php>). PRODES information estimates a cumulative increase in deforested area, and hence, does not account for forest regrowth (INPE 2005). Nevertheless, PRODES provides the best dataset available in a regional scale for the area (DeFries et al. 2005) and has been widely used in land use research (Asner et al. 2005; Morton et al. 2006; Aragão et al. 2008; Broadbent et al. 2008; Broich et al. 2009) and planning (Acre 2009). The analyses of wildlife status are, thus, related to original forest cover loss in the period compatible to hunting data, and not necessarily to the current non-forest area. I

estimated zero deforestation in the Peruvian border near the four villages located in the Breu Indigenous Land (Figure 2-1), because no geographic data on deforestation was available. This is reasonable since this is one of the most remote and unchanged forest in Peru (CDC-UNALM 2006; Oliveira et al. 2007; Constantino P.A.L, personal observation). Spatial analyses were performed using ArcMap 9.3.

I used a 5km radius buffer to calculate the deforestation and density of indigenous people near each Pano village. This area corresponds to the current ideal Pano hunting territory (Constantino P.A.L, unpublished data). All villages hunt most of their preys within these limits (Appendix A). Other studies estimate the same hunting area for one-day hunts of indigenous and non-indigenous Amazonian groups (Stearman 1990; Bonaudo et al. 2005).

Wildlife Status Indicators

I assessed wildlife status in response to anthropogenic disturbance across the 35 villages using three simple indicators supported by ecological theories presented above: 1) mean prey weight of hunted animals, 2) richness of human-sensitive species, and 3) mode of straight distance at which preferred animals were caught from the village. In addition, I used the capture-per-unit-effort (CPUE) of hunted animals of preferred species, in terms of biomass, as another indication of overall abundance variation for 21 villages (Appendix B). The distance of hunted animals to the village is the only indicator assuming that hunting pressure is the main driver of wildlife variation (Levi et al. 2009). Although the other indicators attempt to reflect variation in wildlife status through species availability to hunters, they do not assume a priori the underlying cause of wildlife variation (Rist et al. 2009). Hence, the four indicators would reflect the same trend if hunting pressure was the main factor driving wildlife population variation. All

indicators were normally distributed. I conducted pairwise Pearson's correlations between the four indicators of wildlife status to assess the difference in information provided. None of these indicators were correlated with monitoring effort (all $-0.1 < r < 0.2$, $p > 0.1$), except for the number of sensitive species. This correlation was significant because of two villages located in an exceptionally conserved forested that recorded more than 30 months of data (Japinim and Vida Nova). There was no significant correlation if those villages are excluded from correlations ($r = 0.3$, $p = 0.09$). Other villages in the same Indigenous Land (i.e. Breu) had lower effort but the same number of species. Therefore, I assume no effect of sampling on observed variation. Furthermore, excluding these villages did not change multiple regression results. Likewise, monitoring effort had no effect if included in multiple regression models.

The ecological models from which these indicators derive have the following main assumptions: 1) equally random distribution of animals throughout landscape, both at the hunting territory and across the region, 2) hunters always seek large animals, although Amazonian hunters sometimes search small animals in their gardens for a single meal, 3) the return from other types of hunting (i.e. sit-and-wait) is minimal. In addition, there are some possible problems with hunting data in estimating wildlife status: 1) this study do not include animals killed after escaping from hunters, usually larger animals and primates, underestimating the effect of hunting on these populations, 2) hunters did not record effort time of unsuccessful hunts overestimating values of CPUE, 3) other researches indicate that hunting data can be biased due to unrecorded small animals, however, this bias should be the same to all villages (Jerozolimski and

Peres 2003). Nonetheless, indicators of wildlife status based on ecological models using hunting data are widely reported in the literature.

In addition to the simple indicators, I created multivariate indicators that capture the joint response of wildlife assemblage. These indicators resulted from a Principal Component Analysis that considered the mean prey weight, number of sensitive species and the mode distance of animals of the 35 villages. Principal components with eigenvectors corresponding to more than 80% cumulative variation in the data were selected. CPUE was not included because of missing data.

Statistical analysis

Selecting Appropriate Indicator of Distance of Preferred Hunted Animals to the Village

The estimation of wildlife distance to the village has been assessed elsewhere using the mean distance of hunted animals (Ohl-Schacherer et al. 2007). Instead, I selected the mode of the distance where animals of preferred species were hunted from the village), because data were not normally distributed (Appendix A). This distribution is likely to be a result of hunting pressure that is higher close to the village center (Levi et al. 2009). I estimated mode distance of hunted animals using the Gaussian kernel density estimator, a statistical technique often applied to study animals' use of space (Seaman and Powell 1996; Gitzen et al. 2006).

Drivers of Wildlife Depletion across Pano Communities

Linear multiple regression

I used multiple linear regressions to identify the drivers of wildlife status across Pano villages. I regressed the six wildlife status indicators (i.e. four simple indicators and two multiple indicators) against explanatory variables related to hunting pressure,

land use change and socio-economic condition of Pano communities (Appendix C). The explanatory variables were transformed (i.e. log10 or square root) when needed to achieve normal distribution. In order to avoid multicollinearity, I conducted pairwise Pearson's correlations between the 12 continuous explanatory variables before running the regression models, selecting for variables with fewer correlations and more accurately measured. The remaining variables used to build the full models were: the density of indigenous people in a 5km radius; village age; actual indigenous population in the village (hunting pressure); animal unit (socio-economic condition); and log10 deforested area (land use). The nominal variable "presence of the road" was also used in model building. The former three variables were used as proxy to hunting pressure and suggested to drive wildlife to decline elsewhere (Jerozolimski and Peres 2003; Brashares et al. 2001).

I selected the minimal model for each wildlife indicator excluding unrelated explanatory variables from the full model ($p > 0.1$) through backward stepwise elimination. I supplemented the backward selection with the analysis of all-possible-subset models (Neter et al. 1985) checking for AICc (considering a variation in 2.0 as significant), the consistency in variables entrance in models, and the signals of estimates. I selected all the possible final models for each response variables. This additional procedure reduces the selection of spurious patterns that can be present in multiple regressions with low ratio of response/explanatory variables.

Spatial dependency

The home range of some important game species (e.g. white-lipped peccary) can be as large as the 5km radius area used to estimate deforestation and density of indigenous people (Fragoso 1998; Reina-Hurtado et al. 2009). As a consequence, the

spatial distribution of villages could give rise to spatial dependence of wildlife status indicators and induce erroneous conclusions regarding drivers of wildlife variation. I used the Moran's I test for spatial dependence to analyze of wildlife indicators and residuals of minimal regression models using ArcMap 9.3.

Capacity of indigenous lands to protect wildlife against external threats

I analyzed the effects of surrounding disturbances on wildlife hunted in Pano villages in two levels: the village level and the Indigenous Land level. At the village level, those villages with a portion of the 5km radius outside of Protected Areas were considered vulnerable to external threats, whilst others were considered protected. Student t-tests were conducted to evaluate differences of wildlife indicators between these two groups.

Nonetheless, village vulnerability to external threats may be a result of the poor design and location of Indigenous Lands. Therefore, at the Indigenous Land level, I followed Peres and Terborgh's (1995) categorization of reserve defensibility to label Indigenous Lands according to their design and location in the state (Table 2-1). I grouped villages pertaining to Indigenous Lands as low, medium, or high defensibility against external threats. I tested the differences of wildlife status between these groups using ANOVA and Tukey-Kramer HSD tests. All statistical analyses were performed in the software JMP8. In addition, I compared the size of Pano Indigenous Lands to the minimum area required to maintain sustainable harvest (Peres 2001) of eight preferred species, considering the annual per capita harvest rates of local people throughout the Amazon.

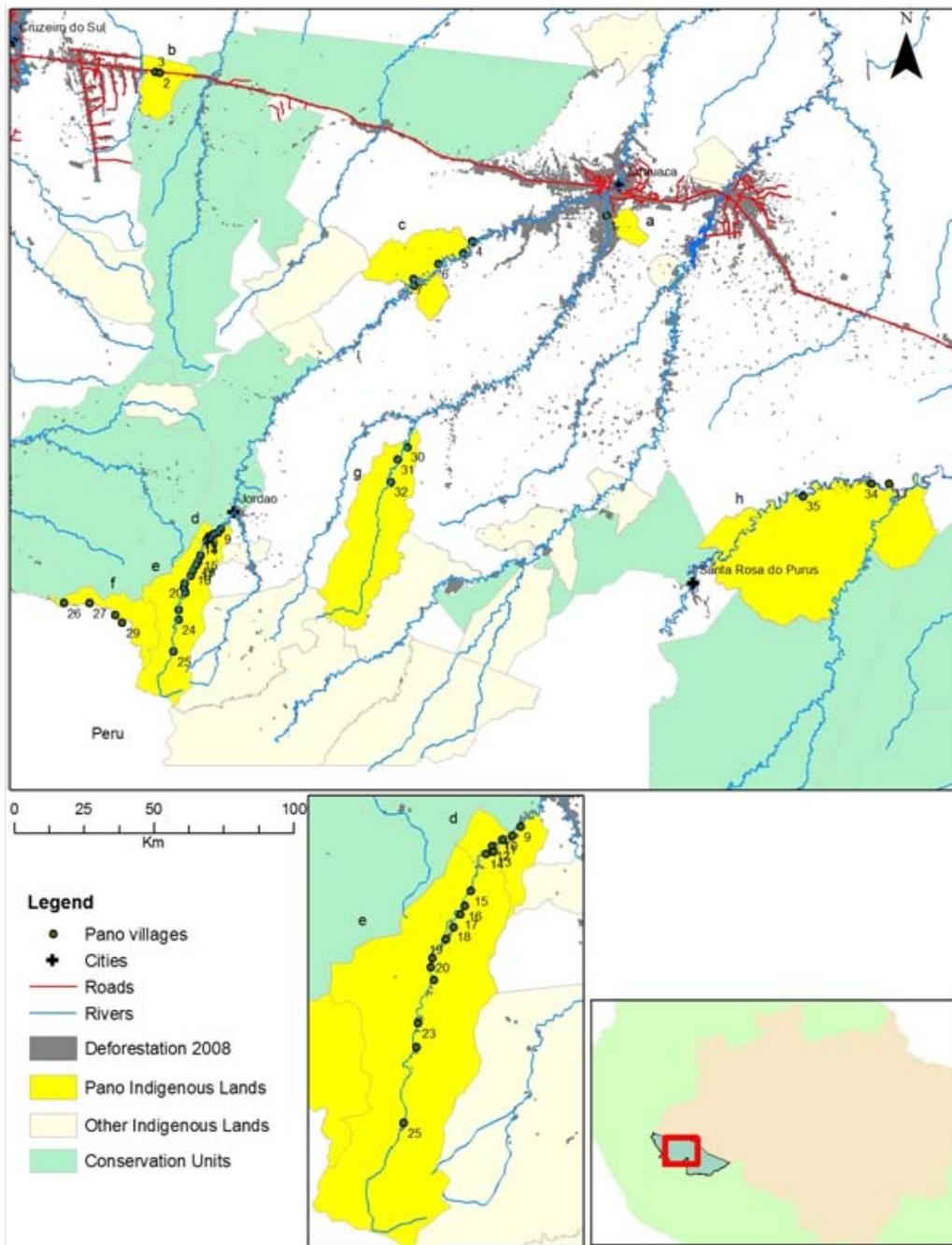


Figure 2-1. Kaxinawá and Katukina Indigenous Lands studied in Acre State, Brazilian Amazon. Label numbers correspond to identification in Table 3-4. Indigenous Lands studied (yellow), other Indigenous Lands (beige), other Conservation Units (green), cities (crosses). Indigenous Lands' labels: a) Kaxinawá do Igarapé do Cauho, b) Katukina do Campinas, c) Kaxinawá da Praia do Carapanã, d) Kaxinawá do Baixo Rio Jordão, e) Kaxinawá do Rio Jordão, f) Kaxinawá/Ashaninka do Rio Breu, g) Kaxinawá do Humaitá, and h) Alto Purus. Close up illustrates village locations in the Kaxinawá do Baixo Rio Jordão and Rio Jordão Indigenous Lands.

Table 2-1. Defensibility of Pano Indigenous Lands against external threats.

Indigenous Land	Size (km ²)	Defensibility	Design features to include in the category	Location in State
Igarapé do Caucho	123.18	Low	small size, road presence, unprotected surround, near major city	development frontier
Campinas	326.24	Low	small size, road presence, two access points, partially unprotected surround, near major city	development frontier
Praia do Carapanã	606.98	Low	two access point, intense river traffic, unprotected surround	development frontier
Baixo Jordão	87.26	Medium	small size, partially protected surround	western Amazon ecological corridor
Humaitá	1273.83	Medium	large size, unprotected surround, watershed headwaters	western Amazon ecological corridor
Alto Purus	2631.3	Medium	large size, partially protected surround, two access points, intense river traffic	western Amazon ecological corridor
Jordão	872.93	High	medium size, totally protected surround, watershed headwaters	western Amazon ecological corridor
Breu	312.77	High	totally protected surround, watershed headwaters, remoteness	western Amazon ecological corridor

CHAPTER 3 RESULTS

Pano Hunter's Prey Profiles

The Kaxinawá and Katukina communities hunted 54 wildlife species or taxa (e.g. Dasypodidae). Large bodied preferred vertebrates were the most hunted and provided the largest amount of meat (Table 3-1). The 3375 hunted ungulates provided more meat than other taxa to the whole Pano population (Table 3-2). Ungulates were most hunted in almost all villages, with white-lipped peccary being the most hunted among them, and provided up to 60% of wildmeat in some villages (Figure 3-1). Rodents and primates were second in terms of animals hunted and meat contribution (Table 3-2), especially because of the presence of large-bodied species. Large birds were frequently hunted but less important in providing meat (Table 3-2). Species of secondary preference that were frequently hunted include paca, agouti, armadillos, capuchin monkey, squirrels and coati (Table 3-1) but only provided 13% of total consumed meat. Human-sensitive species were absent in hunter's prey profiles of several villages, particularly in those located in the development frontier (Table 3-3). Villages alongside roads did not hunt large monkeys or birds. On the other hand, white-lipped peccary and howler monkey were hunted in almost all villages.

Information Provided by Indicators of Wildlife Status

Three indicators of wildlife status (i.e. mean prey weight, richness of sensitive species, and CPUE of preferred species) were significantly correlated among each other, representing a pattern in which the average weight of preys decreased associated to the loss of sensitive species, reflecting in the decreased CPUE of preferred animals. In turn, the mode distance of preferred animals to the village was not

correlated with other indicators, suggesting a different process causing its variation (Table 3-4).

As suggested by the Optimal Foraging Theory, the variation in mean prey weight represented the positive variation in the proportion of preferred species, specifically ungulates, simultaneously to the negative variation in the proportion of species of intermediate preference and low ranked species, specially primates (Figure 3-2a). Furthermore, the variation in CPUE of preferred species was positively associated to the proportion of hunted animals, in particular the tapir and all ungulate species, and negatively to intermediate and low ranked species, specially rodents (Figure 3-2b). Piping guan, razor-billed curassow, woolly and spider monkey, and tapir were frequently absent from Pano catchments. Villages that hunted four or less sensitive species never hunted the piping guan, razor-billed curassow, and tapir, whilst only one hunted the spider monkey and four the woolly monkey.

The two Principal Components (PCs), created from the multivariate analysis, correspond to 86.55% of the variation present in simple indicators. PC1 reflects the simultaneous variation in mean prey weight and sensitive species richness, with little contribution of mode distance. PC2 reflects the simultaneous variation of mode distance in opposition to sensitive species richness, where villages that have hunted animals farther away and hunted a small number of sensitive species (Table 3-5).

Drivers of Wildlife Depletion across Pano Communities

Only hunting pressure, represented by the density of indigenous people, and road presence composed the single significant minimal regression model, explaining 30% of variation in mode distance of preferred animals hunted away from villages (Table 3-6; Table 3-7). According to this model, an addition of 100 Pano people in 5km represented

an increase of 8.4 minutes distance (app. 230m) to the sites where preferred animals are more frequently hunted. Hunters in roadside villages hunted animals one hour (app. 1715m) farther away from village centers than riverside villages (Figure 3-3).

Deforestation had no association with mode distance of preferred animals from village center (Figure 3-4).

Only deforestation and presence of roads composed the single significant minimal model, explaining 50% of mean prey weight variation (Table 3-6; Table 3-7). Clearing the first 10 hectares of forest near villages resulted in a drop of app. 3kg in prey size. The rate of reducing prey size decreased as deforestation increased, probably because smaller species are more resilient to habitat change and can continuously provide meat. Hunters in roadside villages harvested animals an average of 6kg larger than riverside villages (Figure 3-3), a wildlife response contrary to the expected. Roads usually cause reduction in wildlife abundances and richness (Laurance et al. 2006, 2008). A possible explanation is that the mean weight of prey hunted in Pano communities alongside roads does not reflect game abundance but rather the access hunters have to alternative meat purchased in city markets. Hunting pressure, in turn, had no association with mean prey weight (Figure 3-4).

The minimal models for CPUE of preferred species and number of sensitive species hunted were composed only by deforestation that explained 30% and 56% of variation, respectively (Table 3-6; Table 3-7). Losing the first 10 hectares resulted in the absence of more than two sensitive species in prey profiles. The rate of species absence decreased as deforestation increased (Figure 3-3). In terms of hunting success, hunters caught 0.65kg/hour less after the loss of the first 10 hectares of forest

near villages, a rate that decreased as deforestation increased (Figure 3-3). Hunting pressure had no relationship with CPUE or sensitive species richness (Figure 3-4).

The minimal model explaining 55% of variation in PC1 (Table 3-6) was composed by deforestation, presence of road, and the density of indigenous people although the latter was not significant at $\alpha < 0.05$ (Table 3-7). According to this model, indigenous hunting and deforestation around villages have additive effects on wildlife, that responded by decreasing the abundance of large animals and richness of sensitive species, while increasing the distance of preferred animals from the village. Nevertheless, indigenous hunting seems to have less effect on this multiple response. In turn, PC2 was driven by road presence and indigenous density in a model explaining 50% of variation. Thus, indigenous hunting is negatively related to the dispersion of animals far away from villages and, to a smaller extent, the reduction of sensitive species richness, a process exacerbated by the presence of roads (Figure 3-3). Nevertheless, PC2 represents a smaller portion of wildlife multiple response.

Because the variables describing the two primary drivers of game depletion, deforestation and hunting pressure (i.e. village age, density of indigenous people, and actual village population) were not correlated, it was possible to assess the importance of the effect of each factor on wildlife status. The regression results suggested that deforestation and indigenous hunting induced different responses of wildlife species, and the road had a significant impact exacerbating effects of both drivers. Hunting pressure affected populations of preferred game species by displacing them away from village. Deforestation is the only factor that significantly led to an increase in the effort needed to obtain meat of preferred species, induced shifts in large prey abundances,

and was the main factor responsible for the decline in the richness of human-sensitive species.

Spatial Dependence

The actual values of mean prey weight, sensitive species richness and PC1 were autocorrelated, but not the residuals of the minimal regression models (Table 3-8). Hence, neighbor villages have wildlife statuses more similar than those farther apart. Deforestation across villages explained spatial dependency because this was the only variable included in regression models, which residuals were not autocorrelated. Given that the actual values and the residuals of regression model of mode distance of hunted animals were not spatially dependent, indigenous hunting that affects wildlife distribution is apparently a localized process within 5km around villages. It was not possible to test spatial dependence of CPUE due to the reduced data points.

Capacity of Indigenous Lands to Protect Wildlife against External Threats

The results from regression models clearly showed a difference of wildlife status between protected villages and those vulnerable to threats from outside Indigenous Lands (Figure 3-3). Hunters in vulnerable villages hunted animals on average 2kg smaller, one sensitive species less, and had three times less hunting success than those in protected villages. The distance where most of the animals were hunted was not significantly different between the two groups (Table 3-9). Deforestation was the only explanatory variable used in regression models significantly higher in vulnerable villages than in protected (Table 3-9), suggesting that deforestation is possibly the factor creating differences of wildlife status between protected villages and those vulnerable to external threats. The absence of difference in mode distance between the groups of villages provides additional evidence that indigenous hunting pressure has only

localized consequences to wildlife. Part of the deforested area affecting game hunted in vulnerable villages is outside the boundaries of Indigenous Lands. Across these villages, deforestation inside Indigenous Lands was not correlated to the variation of any wildlife status indicator, whereas total deforestation and deforestation near villages but outside Indigenous Lands were (Table 3-10).

Another clear pattern is the spatial distribution of vulnerable and protected villages within Indigenous Lands of different levels of defensibility against external threats. Vulnerable villages were all located in Indigenous Lands with low or medium defensibility whereas protected villages were located in reserves with high defensibility, except for one protected village in a reserve with medium defensibility (Table 2-1; Table 3-3). Hunters in Indigenous Lands of low defensibility hunted prey twice as small ($F=3.47$, $p<0.05$), and twice as many less sensitive species ($F=15.59$, $p<0.01$), than hunters in reserves of medium or high defensibility. Those hunters also had almost four times lower hunting success than hunters in Indigenous Lands of high defensibility ($F=5.68$, $p<0.05$; Figure 3-5). Hence, Indigenous Lands better protected against external threats have wildlife in a better status. The distance where animals are caught was not different among Indigenous Lands of different levels of defensibility ($F=0.22$, $p>0.1$; Figure 3-5), suggesting again that the process driving this wildlife response occurs at a smaller spatial scale.

In addition, the size of the reserve seems to be of less importance to Indigenous Lands, at least, with partially protected surroundings and located out of the development frontier. According to Peres (2001), hunting sustainability is associated to the size of protected areas because these should be able to maintain viable populations of game

species at the harvest level of subsistence hunting of Amazonian people. Analyzing Pano Indigenous Lands according to this rationale, Alto Purus is the only Indigenous Land large enough to guarantee alone sustainably harvested populations of all preferred species. If reserve size were the most important factor determining the presence of species, Jordão Indigenous Land would not be able to maintain the harvest of tapirs, while, Breu Indigenous Land would support harvested populations of tapir, howler and woolly monkey and curassow (Table 3-11). However, these species are still hunted in these reserves (Table 3-3). Although Baixo Jordão Indigenous Land would only harvest collared peccary according to its small size (Table 3-11), hunters heavily relied on other preferred species (Figure 3-1). Indigenous Lands size, year of title, and Pano population were not correlated to any of the wildlife status indicators (all $-1.5 < r < 2.5$, $p > 0.15$). Nonetheless, the size of the reserve might be important to Igarapé do Caucho and Campinas that just hunted preferred species that do not depend on the surrounding area to have viable population (Table 3-11).

Table 3-1. Wildlife species hunted by Pano communities in Acre, Brazilian Amazon, during this study. Preferred species (1), secondary species (2), low ranked species (3) indicated according to Cunha and Almeida (2002) and Constantino et al. (2008). For some analysis in this study we collapsed groups 2 and 3 in the group of species of secondary preference. Taxa referenced with superscript letters were not always recorded at the species level or identification was not certain.

Scientific name	Common name	# of animals	Weight of animals	Preference	Sensitive species ref.
<i>Tayassu pecari</i>	white-lipped peccary	1335	24455.6	1	1
<i>Pecari tajacu</i>	collared peccary	1267	16049.0	1	
<i>Mazama americana</i>	red brocket deer	695	15381.8	1	
<i>Tapirus terrestris</i>	lowland tapir	68	6786.0	1	2
<i>Alouatta seniculus</i>	howler monkey	539	3343.5	1	3
<i>Geochelonia denticulata</i>	tortoise	536	3161.3	1	4
<i>Ateles chamek</i>	spider monkey	258	1937.5	1	5
<i>Caiman</i> sp.	cayman	177	1614.5	1	
<i>Lagothrix lagotricha</i>	woolly monkey	72	536.2	1	6
<i>Mitu tuberosum</i>	razor-billed currasow	96	457.1	1	7
<i>Mazama gouazabira</i>	gray brocket deer	10	136.0	1	
<i>Pipile cujubi</i>	piping guan	18	37.5	1	8
<i>Agouti paca</i>	paca	524	3933.1	2	
Dasypodidae ^a	armadillo	427	2368.5	2	
<i>Dasiprocta fuliginosa</i>	agouti	411	1812.2	2	
<i>Nasua nasua</i>	coati	313	1450.8	2	
<i>Cebus apela</i>	brown capuchin	266	1031.1	2	
<i>Sciurus</i> sp. ^b	squirrel	636	657.5	2	
<i>Penelope jacquacu</i>	Spix's guan	228	421.7	2	
<i>Tinamus guttatus</i>	large tinamous	200	326.1	2	
<i>Hydrochaeris hydrochaeris</i>	capybara	13	291.0	3	
<i>Psophia leucoptera</i>	pale-winged trumpeter	132	251.3	2	
<i>Dinomys branickii</i>	pacarana	31	236.0	3	
<i>Pithecia</i> sp. ^c	saki monkey	68	203.0	3	
<i>Myoprocta pratii</i>	acouchi	162	183.8	3	
<i>Cebus albifrons</i>	white-fronted capuchin	45	162.5	3	
<i>Aotus nigriceps</i>	night monkey	88	119.8	3	

Table 3-1. Continued

<i>Scientific name</i>	Common name	# of animals	Weight of animals	Preference	Sensitive species ref.
<i>Priodontes maximus</i>	giant armadillo	4	95.0	3	
<i>Myrmecophaga tridactyla</i>	giant anteater	4	87.0	3	
<i>Ara</i> sp. ^e	macaw	53	74.9	3	
<i>Saimiri sciureus</i>	squirrel monkey	53	60.7	3	
<i>Tinamus tao</i>	grey tinamous	24	55.8	2	
<i>Callicebus moloch</i>	titi monkey	44	53.1	3	
<i>Saguinus</i> sp. ^d	tamarin	53	44.2	3	
<i>Amazona</i> sp. ^e	parrot	34	31.8	3	
<i>Crypturellus cinereus</i>	small tinamous	30	21.6	3	
<i>Ramphastos</i> sp. ^e	toucan	26	20.5	3	
<i>Ortalis guttata</i>	Speckled chachalaca	23	16.8	3	
<i>Crypturellus soui</i>	little tinamous	25	16.3	3	
Rallidae	rail	23	15.0	3	
<i>Coendou</i> sp.	quandu	3	15.0	3	
<i>Tinamous major</i>	small tinamous	15	12.5	2	
<i>Crypturellus</i> sp.3	small tinamous	15	11.9	3	
<i>Crypturellus</i> sp.2	small tinamous	18	11.6	3	
<i>Odontophorus</i> sp. ^e	wood quail	11	6.9	3	
<i>Jabiru mycteria</i>	jabiru	1	6.0	3	
<i>Harpia harpyja</i>	harpy eagle	1	5.0	3	
Columbidae	dove	14	4.3	3	
Anatidae	duck	3	4.0	3	
Accipitridae	kite	3	2.5	3	
<i>Tupinambis teguixin</i>	common tegu	2	2.0	3	
<i>Psarocolius</i> sp.	oropendola	5	1.9	3	
Psittacidae	macaw	5	1.9	3	
<i>Pteroglossus</i> sp.	aracari	2	0.9	3	
Total		9109	88023.3		

^a*Dasyopus novemcinctus*, *Dasyopus kappleri*, *Cabassous unicinctus*

^b*S. ignitus* and *S. spadiceus*.

^c*P. irrorata* or *P. monachus*.

^d*S. imperator*, *S. fuscicolor*, *S. melanoleucus*, and possibly *S. mystax*.

^etaxa containing more than one species but with uncertain identification.

Table 3-2. Relative importance of prey taxa to Pano communities.

Taxa	Number of animals hunted	% of offtake	Amount of meat	% of meat contribution
ungulates	3375	37.05	62808.4	71.35
primates	1486	16.31	7491.6	8.51
rodents	1777	19.51	7113.6	8.08
reptiles	715	7.85	4777.8	5.43
birds	1005	11.03	1815.8	2.06
other	751	8.24	4016.3	4.56
total	9109	100	88023.5	100

Table 3-3. Characteristics of Pano villages in the State of Acre, Brazilian Amazon, and absent species.

Map ref.†	Village	External threat††	# months monitored	# animals hunted	# species hunted	Absent sensitive species†††
Igarapé do Caucho						
1	18 Praia*	1	8	50	8	2, 3, 4, 5, 6, 7, 8
Campinas						
2	Samauma**a	1	11	158	18	2, 4, 5, 6, 7, 8
3	Varinawa**a	1	11	103	9	2, 3, 5, 6, 7, 8
Praia do Carapanã						
Segredo do						
4	Artesão ^a	1	10	60	19	2, 5, 6, 7, 8
5	Água Viva	1	10	115	26	1, 2, 5, 6, 7, 8
6	Goiania ^a	1	9	359	33	2, 4, 5, 6, 7, 8
7	Nova Vida	1	10	210	33	1, 2, 5, 6, 7, 8
8	Mibayã	1	16	288	35	2, 5, 6, 7, 8
Baixo Jordão						
9	Nova Extrema ^a	1	22	373	27	2, 7, 8
10	Nova Cachoeira	1	14	219	23	2, 4, 5, 7, 8
11	Nova Empresa ^a	1	14	196	20	7, 8
12	Nova Mina	1	16	158	19	2, 4, 5, 6, 7, 8
Jordão						
13	Morada Nova	0	10	78	20	2, 5, 7, 8
14	Novo Astro	0	13	167	26	5, 7, 8
15	Sacado	0	10	104	22	2, 4, 5, 6, 7, 8
16	Boa Vista ^a	0	16	244	29	2, 7, 8
17	Nova Fortaleza	0	10	155	23	6, 7, 8
18	Nova Aliança	0	11	83	19	2, 5, 7, 8

Table 3-3. Continued.

Map ref.†	Village	External threat††	# months monitored	# animals hunted	# species hunted	Absent sensitive species†††
19	Novo Natal	0	7	56	18	2, 7, 8
20	Chico Curumim	0	17	279	28	2, 7, 8
21	Bom Jesus	0	21	307	28	8
22	Verde Floresta ^a	0	22	278	17	6, 8
23	Belo Monte	0	28	529	30	6, 8
24	Paz do Senhor	0	8	126	26	2, 7, 8
25	Novo segredo	0	8	125	21	6
Breu						
26	Vida Nova	0	30	927	41	none
27	Cruzeirinho	0	18	321	29	none
28	Japirim	0	45	1423	38	none
29	Jacobina	0	8	492	28	none
Humaitá						
30	Boa Sorte ^a	1	10	99	18	2, 6, 8
31	Boa Vista ^a	1	9	133	12	2, 6
32	Porto Brasil ^a	0	16	126	17	6, 7
Alto Purus						
33	Nova Fronteira ^a	1	18	373	35	6, 7, 8
34	Dois Irmãos ^a	1	9	149	25	6, 8
35	Porto Rico ^a	1	9	124	20	6

†Numbers refer to village location in Figure 2-1.

††Villages subject to external threat are

†††Numbers refer to species numbers in column "Sensitive species" of Table 2-1.

*villages accessing city market by roads.

^avillages excluded from CPUE analysis.

Table 3-4. Pearson correlation matrix between simple indicator variables of wildlife depletion in Pano villages (upper) and excluding roadside villages (bottom) in Acre, Brazilian Amazon.

Simple indicators	Mean prey weight (kg)	CPUE preferred species (kg/hour)	Number of sensitive species
Mean prey weight (kg) n=35	1		
CPUE of preferred species (kg/hour) n=21	0.83**	1	
Number of sensitive species n=35	0.44*	0.62*	1
Mode distance of preferred species hunted from the village (min.) n=35	0.73**	0.59*	
	0.33	0.00	-0.09
	0.19	0.14	0.18

*p<0.01; **p<0.001

Table 3-5. Principal Component Analysis axis of wildlife abundance simple indicators.

	PC1	PC2	PC3
Axis Percent (%)	50.45	36.01	13.46
Components			
Mean prey weight	0.73	0.05	-0.68
Richness of sensitive species	0.57	-0.59	0.57
Mode distance	0.37	0.81	0.46

Table 3-6. Backward stepwise regression models for all response variables at the village level.

Response variable	R^2_{adj}	DF	F ratio	p	SSE final	AICc full model	AICc minimal model	n
Simple indicators								
Mean prey weight (kg)	0.50	2	18.26	<0.001	119.17	51.79	48.88	35
CPUE preferred species (kg/hour)	0.30	1	9.67	0.006	7.63	-11.99	-17.27	21
Richness of sensitive species	0.56	1	44.8	<0.001	60.3	25.53	23.04	35
Mode distance of preferred species hunted from the village (min.)	0.30	2	8.42	0.001	39135.16	251.09	251.68	35
Multivariate indicators								
PC1	0.55	3	14.64	<0.001	21.29	-6.67	-9.39	35
PC2	0.50	2	18.08	<0.001	17.28	-15.19	-18.7	35

Table 3-7. Explanatory variables included in the final regression models after stepwise and supervised selection.

Response variable	Parameters	Estimate	SE	t Ratio	Prob>t
Mean prey weight					
	Intercept	18.93	1.68	11.29	<0.001
	Log deforested area	-3.36	0.64	14.67	<0.001
	Road presence	3.00	0.62	5.44	<0.001
CPUE of preferred species					
	Intercept	2.84	0.46	6.13	<0.001
	Log deforested area	-0.65	0.21	-3.11	0.006
Richness of sensitive species					
	Intercept	10.36	0.89	11.69	<0.001
	Log deforested area	-2.63	0.39	-6.69	<0.001
Mode distance of preferred species to village center					
	Intercept	72.76	16.71	4.35	<0.001
	Road presence	31.50	10.71	2.94	<0.01
	Density of indigenous people	6.60	2.83	2.33	0.03
PC1					
	Intercept	4.18	0.73	5.72	<0.001
	Road presence	0.97	0.28	3.53	0.001
	Density of indigenous people	0.14	0.07	2.00	0.055
	Log deforested area	-1.80	0.28	-6.53	<0.001
PC2					
	Intercept	0.21	0.35	0.60	0.55
	Road presence	1.07	0.22	4.78	<0.001
	Density of indigenous people	0.17	0.06	2.81	0.008

Table 3-8. Spatial dependence of response variables and residuals of final regression models. Moran's I index reported and its significance given by Z-scores in parenthesis.

Wildlife indicators	Actual values	Residuals of minimal models	n
Mean prey weight	0.23 (2.57)*	0.04 (0.66)	35
Number of sensitive species	0.2 (2.3)*	-0.04 (-0.1)	35
Mode distance of preferred species to village center	0.06 (0.85)	0.08 (1.11)	35
PC1	0.24 (2.7)**	-0.06 (-0.35)	35
PC2	0.08 (1.07)	-0.02 (0.11)	35

*significant at $\alpha < 0.05$

Table 3-9. Student's t tests on wildlife indicators between villages protected and vulnerable to external threat.

Variables	Vulnerable		Protected		t	p
	mean (SD)	n	mean (SD)	n		
response variable						
mean prey weight (kg)	8.11 (3.18)	17	10.05 (1.88)	18	2.17	<0.05
mean prey weight (kg)*	7.30 (2.46)	14	10.05 (1.88)	19	3.46	<0.001
richness of sensitive species	3.53 (1.80)	17	5.67 (1.71)	18	3.58	<0.001
CPUE of preferred species (kg/hour)	0.83 (0.63)	6	1.72 (0.66)	18	2.89	<0.01
mode distance of animals hunted from village center (min.)	80.12 (44.51)	17	67.13 (39.51)	18	-0.91	n.s.
PC1	-0.51 (1.32)	17	0.48 (0.94)	18	2.57	<0.005
PC1*	-0.73 (1.25)	14	0.48 (0.94)	18	3.03	<0.005
PC2	0.43 (1.04)	17	-0.40 (0.89)	18	-2.53	n.s.
	mean (SD)	n	mean (SD)	n	t	p
explanatory variables						
Deforested area (ha)	2.57 (0.49)	17	1.81 (0.41)	18	-4.87	<0.001
Density of indigenous people (in 5km radius)	4.18 (2.30)	17	3.94 (2.06)	18	-0.31	n.s.
Village age (year)	11.47 (8.31)	17	14.61 (8.50)	18	1.10	n.s.
Village population	83.53 (54.71)	17	79.17 (29.14)	18	-0.29	n.s.

*comparisons excluding the roadside villages of indicators subjected to road effects on access to alternative meat. Values of mean prey weight in these villages do not reflect wildlife abundance.

Table 3-10. Pearson's correlations between indicators of wildlife status that were driven by deforestation and the total deforestation within 5km radius area, then inside and outside Indigenous Lands (IL) for the group of villages vulnerable to external threats. Roadside villages were excluded.

	Total	Deforestation		N
		Inside IL	Outside IL	
Mean prey weight	-0.63 (0.02)*	-0.48 (0.09)	-0.60 (0.04)*	13
Sensitive species richness	0.38 (0.19)	-0.4 (0.20)	-0.12 (0.70)	13
PC1	-0.58 (0.04)*	-0.58 (0.05)	-0.43 (0.16)	13

*significant at $\alpha < 0.05$

Table 3-11. Pano Indigenous Lands with total area that would support the average local people harvesting of preferred species in the Amazon.

Species	Minimum required area (km ²) ^a	Indigenous Lands
tapir	2003±475	Purus
howler monkey	864±303	Purus, Humaitá, Jordão
woolly monkey	443±132	Purus, Humaitá, Jordão, Praia do Carapanã
currasow	369±67	Purus, Humaitá, Jordão, Praia do Carapanã
white-lipped peccary	298±99	all but Igarapé do Caucho and Baixo Jordão
spider monkey	258±64	all but Igarapé do Caucho and Baixo Jordão
red brocket deer	112±44	all but Baixo Jordão
collared peccary	69±19	all

^aEstimated mean±SD minimum sustainable area for hunted species according to Peres (2001), based on estimated average annual harvest in the Amazon applied in Robinson and Redford (1991) model of hunting sustainability.

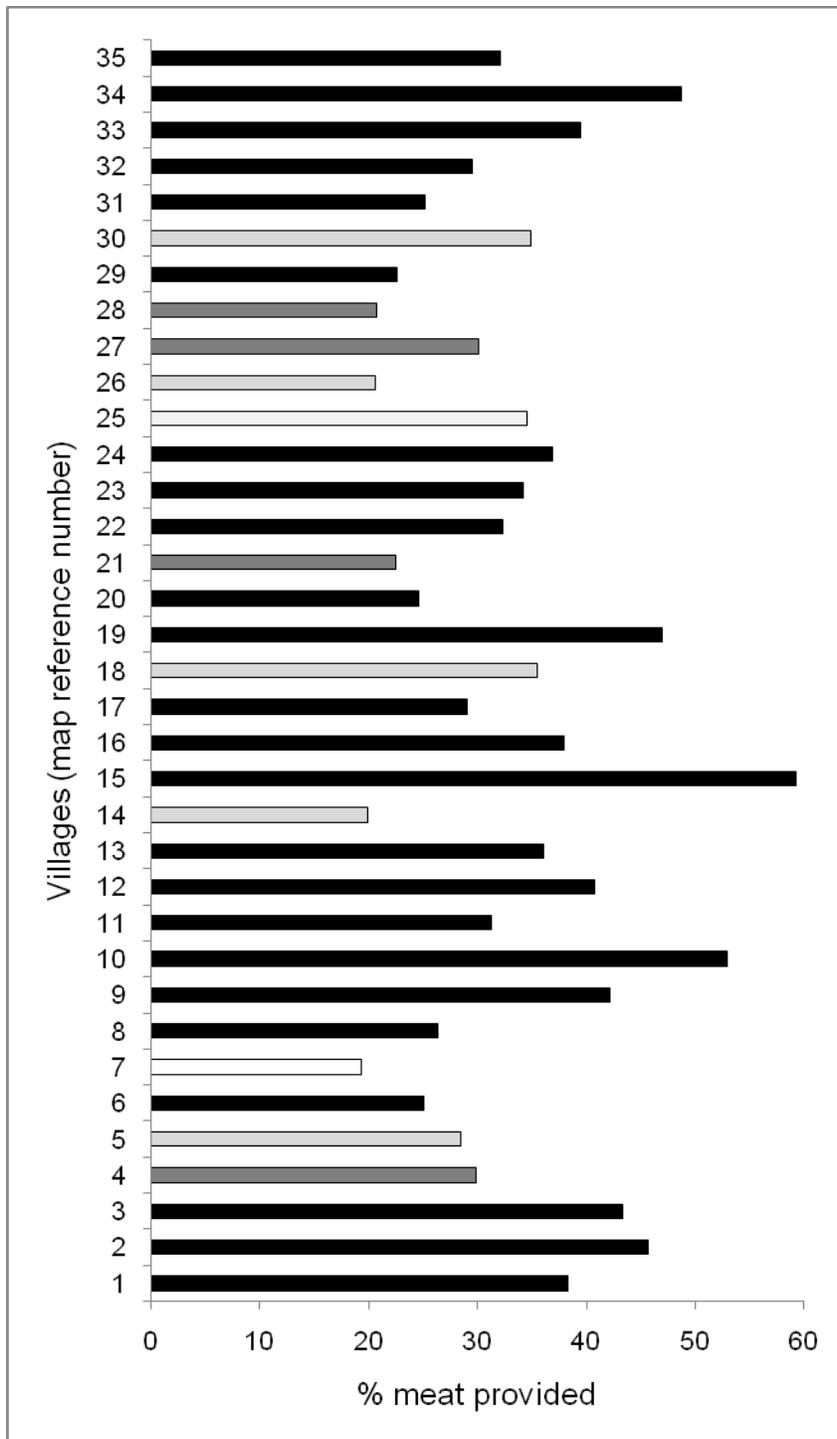


Figure 3-1. Relative contribution of wildlife species to Pano diet. Black: white-lipped peccary; dark grey: brocket deer; medium grey: collared peccary; light grey: tapir; white: agouti. Column numbers refer to village numbers in Figure 2-1.

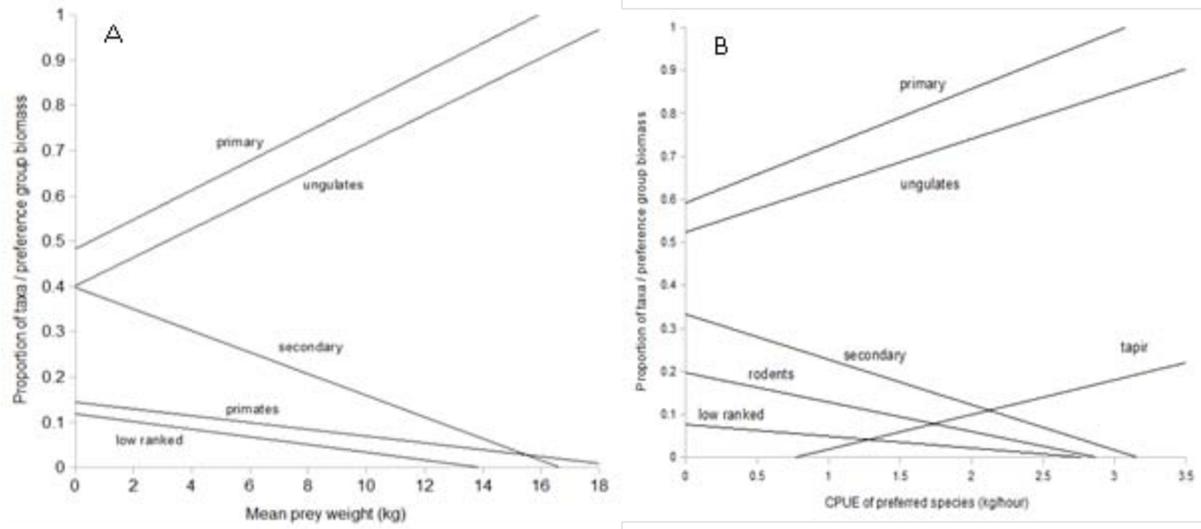


Figure 3-2. Significant Pearson's correlation between wildlife status indicators and the proportion of individual species, major taxa, and preference group, after Bonferroni correction. A) Mean prey weight. Only the proportion of ungulates and preferred species were positively correlated ($r=0.69$, $p<0.000$; $r=0.67$, $p<0.000$, respectively) and the proportion of primates, low ranked and secondary species ($r=-0.46$, $p=0.02$; $r=-0.59$, $p=0.001$; $r=-0.62$, $p<0.001$, respectively) were negatively correlated. B) CPUE of preferred species.

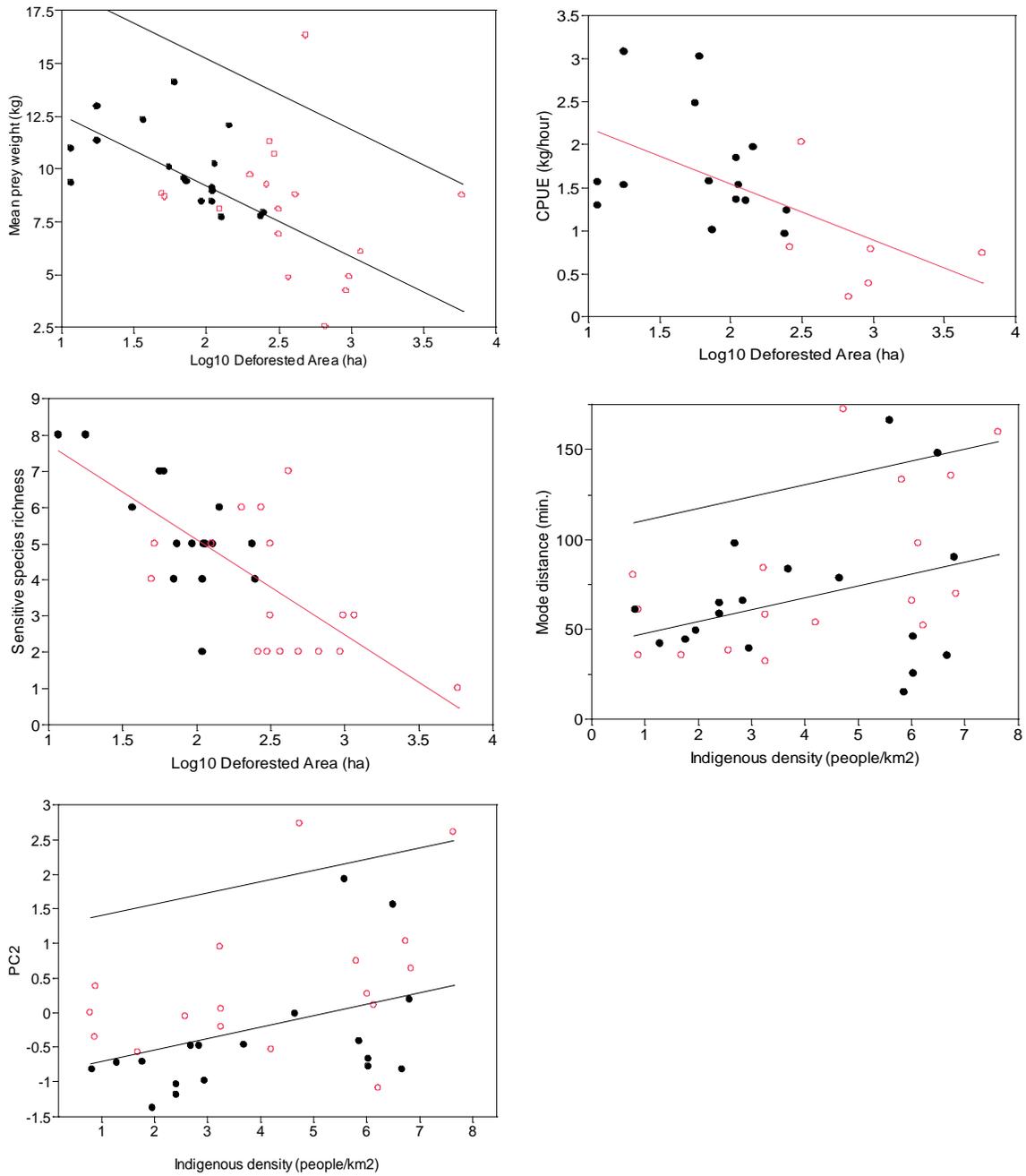


Figure 3-3. Best linear regression models after backward stepwise selection (based on AICc). Protected villages (solid dots), vulnerable villages (open dots), roadside villages regression line (bottom line), riverside villages regression line (upper line). PC1 could not be represented in two-dimensional graph.

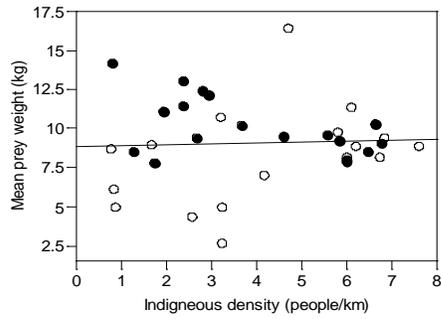


Figure 3-4. Examples of non significant Pearson's correlation between hunting pressure, deforestation and indicators of wildlife status. Vulnerable villages (open dots) and protected villages (solid dots). Mean prey weight and hunting pressure ($r_{\text{village age}}=0.14$, $p=0.42$; $r_{\text{indigenous density}}=0.04$, $p=0.83$), CPUE and hunting pressure ($r_{\text{village age}}=-0.15$, $p=0.53$, $r_{\text{indigenous density}}=-0.31$, $p=0.18$), sensitive species richness and hunting pressure ($r_{\text{village age}}=-0.05$, $p=0.79$, $r_{\text{indigenous density}}=-0.22$, $p=0.21$), and mode distance and deforestation ($r=0.09$, $p=0.89$).

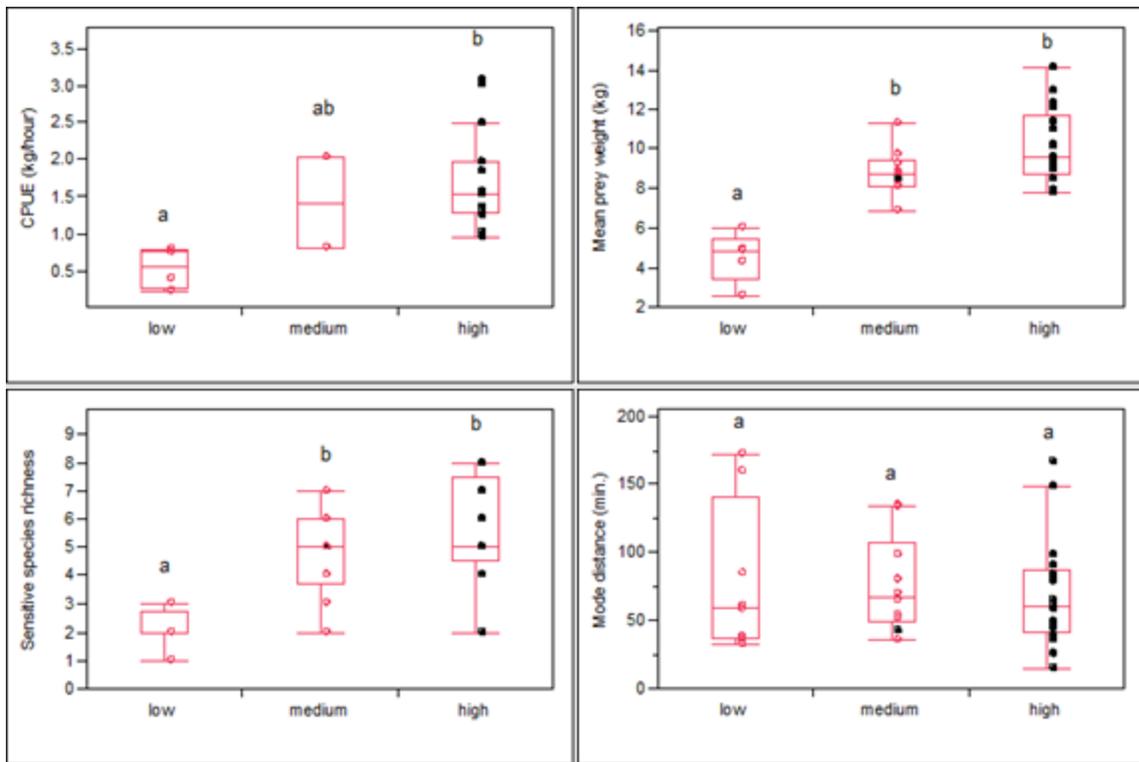


Figure 3-5. Box-plot of wildlife indicators among Indigenous Lands of different levels of defensibility against external threats to wildlife: low (n=8 villages), medium (n=10 villages), and high (n=17). Roadside villages were excluded from mean prey weight analysis. Vulnerable villages (open dots) and protected villages (dark dots). Letters indicate significantly different means ($p < 0.05$). Indigenous Lands within each category are listed in Table 2-1.

CHAPTER 4 DISCUSSION

Wildlife Status nearby Pano Communities

Most Pano communities still relied on preferential animals as a meat source, even after many decades of intense wildlife harvesting and high current hunting pressure in the state of Acre. The pattern of game use by Pano communities was similar to those observed in tropical forests at smaller and larger spatial scales. In the Alto Purus region of Peru, where wildlife is less depleted, ungulates predominate in the diet of Kaxinawá communities (Navarro 2004). Ungulates predominance is observed in other indigenous and non-indigenous unaltered forest settlements in the Amazon (Alvard 1993; Robinson and Bodmer 1999; Mena et al. 2000), as well as commonly at the large scale of the Amazon basin (Jerozolimski and Peres 2003) and other tropical forests (Fa et al. 2005). The transition from hunting principally ungulates to smaller and less preferred species observed across Pano villages agrees with the pattern found in single Amazonian settlements that had wildlife depleted by persistent hunting (Stearman 1990) or by factors other than hunting (Escamilla et al. 2000). The same gradual shift is also observed in the spatial variation of wildlife status across multiple sites in a single Extractive Reserve in the Amazon (Ramos 2005) as well as larger tropical spatial scales (Jerozolimski and Peres 2003; Fa and Brown 2009). Pano hunters' return rate of meat declined in response to decline in abundance of preferred game, particularly ungulates, as observed in local communities in the Peruvian Amazonian where peccaries and tapir declined over 10 years reflecting in hunters return rate (Bodmer et al. 1997). The depleted Pano villages disproportionately hunted smaller resilient species, while sensitive species like razor-billed curassow, piping guan, spider and woolly monkeys,

and tapir were usually absent from their prey profiles. In the Neotropics these species are the most sensitive to anthropogenic disturbances and likely to be the first ones to be extirpated locally (Daily et al. 2003; Parry et al. 2007) and across sites in the Amazon basin (Peres 2000). Hence, wildlife responses in Pano hunting territory refer to change in assemblage structure by reducing abundances of preferred species, particularly ungulates, and composition by extirpation of some specific sensitive species, which resulted in the decline of capture-per-unit-of-effort according to Optimum Foraging Theory.

The increase in the distance of animals from village is related to the reduction of abundance of tropical game (Hill et al. 1997; Ohi-Schacherer et al. 2007) and meat return rate in other indigenous groups in the Amazon (Alvard et al, 1997; Sirén et al. 2004). In these sites, hunting was indicated the driver of game variation. The pattern of animal distance to the village across Pano communities, in turn, differed from these Amazonian. Pano communities hunting preferred animals farther away were not those with their wildlife assemblage considerably depleted, but were those with higher hunting pressure, conforming to the premises of the central-place forager model. Therefore, this indicator might represent a different process driving wildlife response across Pano villages than the others.

Drivers of Wildlife Depletion

In this study it was possible to disentangle the effects of indigenous hunting and deforestation influencing game hunted by Pano people. Even in one of the less deforested regions in the Amazon, indicators of game depletion across Pano villages was mostly explained by increasing nearby deforestation. Indigenous hunting, in turn, was only associated to the dispersion of animals from the village. However, non-

indigenous hunting may be responsible for part of game depletion since hunting outside villages and Indigenous Lands was not estimated, and is possibly related to external deforestation. The most disturbed villages, in terms of wildlife and forest loss, are those more vulnerable to non-indigenous threats because of their location close to Indigenous Lands' boundaries. Moreover, these vulnerable villages are located in Indigenous Lands of low and medium defensibility against external threats, which suggests that factors operating at this level can affect wildlife close to villages. In fact, wildlife is more depleted in villages of Indigenous Lands poorly designed and located in the state development frontier.

Effects of Indigenous Hunting Pressure on Game

The dispersion of large game species was the only wildlife response possibly related to indigenous hunting pressure across Pano communities, represented by the density of indigenous people in neighbor villages. Hunting is responsible for game depletion at local scales (Robinson and Bodmer 1999) and across sites in the Amazon (Peres and Palacios 2007). High concentration of villages exacerbates the effects of hunting because the source of disturbance has many close origins (Levi et al. 2009). Given that the distance of animals hunted by Pano was not associated to other wildlife responses, I suggest two possible non-exclusive hypotheses. One is indigenous hunting caused game depletion only very close to the villages, which is supported by the absence of spatial dependence of distance values, the lack of difference between villages of different protection to external threat, and lack of difference among Indigenous Lands of varying levels of defensibility. In a similar scale to my study, game depletion was related only to local effects of human density (grain of 3.1km²) across forest fragment of Mexican tropical forests (Urquiza-Hass et al. 2009). Or, indigenous

hunting just displaced animal populations and did not reduce their abundances. Game species in hunted areas are more wary of people (Hill et al. 1997; Fitzgibbon 1998) and, given the mobile nature of this resource, may flee and avoid disturbance in the village area (Kensinger 1975; Urquiza-Hass et al. 2009). A previous study in a single Kaxinawá Indigenous Land suggested that the history of land use, not indigenous hunting, explained wildlife status (Constantino et al. 2008). The population of local people was not associated to depletion across settlements in Central/West Africa forests (Fa et al. 2005) and in the Amazon (Jerozolimski and Peres 2003). Although Jerozolimski and Peres (2003) found that the persistence of local hunting over time explained wildlife depletion in the Amazon, the age of Pano villages was not related to game variation. Therefore, there is no clear evidence in this study to support the idea that contemporary and past indigenous hunting severely depletes game species at the regional scale.

Effects of Deforestation around Villages on Game

Deforestation was the only factor significantly affecting wildlife across villages with drastic consequences for Pano hunters, who responded by spending more time hunting to supply their meat needs. At large spatial scales, deforestation is the most important factor depleting wildlife in the tropics, and has been used to project the reduction of populations' range and species extinction (Cuarón 2000; Grelle 2005; Kinnaird et al. 2003; Soares-Filho et al. 2006; Vale et al. 2008). At local scales, higher deforestation in hunting territory near a highway caused Amazonian hunters to catch smaller prey (Bonaudo et al. 2005). At regional scales, hunters caught more ungulates in areas of primary forests of heterogeneous landscape in eastern Amazon (Parry et al. 2009b). In the Alto Juruá Extractive Reserve, adjacent to the Pano Indigenous Lands of Jordão and Breu Indigenous Lands (Figure 2-1), ungulates hunted by local people declined in

response to increasing deforestation density (Ramos 2005). Deforestation, in spite of local hunting, caused the loss of many game species across fragments in a regional scale in Mexico (Urquiza-Hass et al. 2009) and the Brazilian Amazon (Michalski and Peres 2005, 2007). Even though deforestation estimates around Pano villages may include regrown forests creating mosaics of habitats, there is enough evidence that slight habitat disturbances in tropical forests (i.e. fallows and secondary forests) are related to decline in wildlife abundances and richness (Naughton-Treves et al. 2003; Parry et al. 2009a). Forest specialist species are expected to be affected by light habitat alteration but, depending on the quality of secondary forests, tolerant species are only present in primary forests (Estrada et al. 1994; Parry et al. 2009a, b; Urquiza-Hass et al. 2009).

Despite the independence of indigenous hunting, deforestation estimates may have other associated factors that deplete game species. Habitat loss and land use practices inside Indigenous Lands are likely to affect wildlife since most deforested patches correspond to slash-and burn agriculture and small scale pasture bordered by forests, which are prone to effects of fire and fragmentation (Norris et al 2008). Fragmentation, susceptibility to wildfire, and wildlife disease are related to deforestation and have direct (i.e. reducing population size) and indirect (i.e. reducing game resources) impacts on large game, operating independently or in synergy (Laurance et al. 2002; Michalski and Peres 2007; Laurance and Useche 2009).

These findings, thus, cause alarm for the negatives consequences of deforestation to game species even in tropical forest regions yet well conserved. The effects of deforestation and associated disturbances were described in highly fragmented

landscape (Cullen et al. 2000; Chiarello 1999; Michalski and Peres 2005, 2007; Urquiza-Hass et al. 2009) but less studied in more conserved regions (but see Ramos 2005).

Effects of Surrounding Disturbance: the Acre State Zoning

Habitat loss near vulnerable villages could not be disentangled from the effects of non-indigenous hunting pressure and land use around Indigenous Lands. In the absence of hunting pressure data, deforestation is used as proxy for both because clearing forest increases hunting intensity (Lewis et al. 2004) and hunters' accessibility to hunting sites (Peres 2001). Near vulnerable Pano villages, these factors are likely to be more important than disturbances originated in Indigenous Lands to affect wildlife. Nevertheless, the surrounding effect may operate at the Indigenous Land level reflecting on villages within them, since all vulnerable villages were located in reserves with lower defensibility against external threats. The effects of human disturbance (i.e. habitat loss, its associated disturbances, and hunting) outside Protected Areas impact game populations within their boundaries because of their mobile nature and population dynamics (Woodroffe and Ginsberg 1998; Hansen and DeFries 2007). Therefore, poorly designed reserves (i.e. small and without surrounding protection) may not support viable game populations even in the absence of encroachment, especially for those species requiring large tracts of forests, such as tapir, white-lipped peccary and large monkeys (Peres and Terborgh 1995; Peres 2005). Low defensibility and location of Pano Indigenous Lands is, nonetheless, related to non-indigenous invasion for hunting. Only Pano Indigenous Lands not bordered by Protected Areas report frequent non-indigenous encroachment using more impacting hunting techniques (Peres 1993; Lima 2001; CPI-AC 2005a,b; Bant and Pessoa 2008; Constantino et al. 2008), becoming a

major problem in those areas located in the development frontier (i.e. Praia do Carapanã (Bant and Pessoa 2008) and Campinas (Lima 2001; Calouro 2007)). Private lands around Indigenous Lands have a more intensive land use pattern than do Pano people (IBGE 2009), supporting the notion of external threats associated to land cover conversion affecting wildlife hunted in vulnerable villages.

Thus, the historical spatial occupancy of western Amazon and the current State Ecological-Economic Zoning create a heterogeneous regional landscape with polarized differences, including land tenure, infrastructure, urban development and incentives for deforestation, that have considerable influence on wildlife and indigenous societies using them. Highly defensible Indigenous Lands strategically compose the Western Amazon Ecological Corridor, protected by adjacent Conservation Units and covering the watershed's headwaters along the international frontier with Peru, hosting less modified wildlife assemblages. There also lie the Conservation Units of restricted use in the state. Development and forest exploitation in this region threaten little the Indigenous Lands (Carneiro Filho and Souza 2009), given their remoteness, tenure, and the fact that considerable deforestation started only after 1995, remaining limited (Franke 2008; Acre 2009). Hunting with shotguns, conversely, occurs since before 1920s everywhere in the state and is likely to have been more intensive along the international frontier in the past because of meat and pelt trade routes to Peru until the 1970s (Doughty and Myers 1971). Low defensible Indigenous Lands, in turn, have unprotected perimeters and small relative size and are scattered in the development frontier zone planned by the state government, which hosts the largest cities alongside the BR-364 (Acre 2000), probably resulting in depleted wildlife. People in these Indigenous Lands have received

compensation to mitigate social and environmental road impacts (Iglesias and Aquino 2006). Deforestation there was limited until the exponential growth in cattle production during the 1980s and intensification during the 1990s (Valentin et al. 2002; Arima et al. 2005, Oliveira and Bardales 2006; Franke 2008). In addition, commercial logging, regulated or not, takes place in this region including inside some Protected Areas (Acre 2009). In surroundings of the Antimary State Forest, located in the development frontier in Acre State, game species hunted by local people are likely to have been affected by encroachment and deforestation (Calouro 1995, 2005). In that Protected Area, the density of local people decreased 13% in the last 10 years, reducing their hunting pressure; however, the region went through a rapid development process and forest loss, leading to deforestation in the surroundings. Landscape transformation and encroachment, rather than local hunting, could explain the disappearance of some sensitive species even in the less intensive hunted areas. At a regional scale, Urquiza-Hass et al. (2009) associate non-local people's hunting to the loss of many species in fragments of Mexican forest.

Effects of Roads on Wildlife and Hunters

Roads had a dual effect on the indigenous hunting system. On the one hand they exacerbated the negative effect of indigenous hunting on animal distance from roadside villages, which is expected given the disturbances associated to roads that deplete wildlife in their surroundings (Laurance et al. 2006, 2008). Road effects are expected to negatively affect large mammals and birds in particular because of their biological traits (i.e. large movement range, low reproductive rates, and low natural densities; Fahrig and Rytwinski 2009). Furthermore, roads diminish defensibility of Pano Indigenous Lands since these are often used by poacher to access forest areas inside and close to

Campinas Indigenous Land (Lima 2001; Iglesias and Aquino 2006; Calouro 2007), a typical consequence of roads in tropical forests (Peres and Lake 2003; Laurance et al. 2008). On the other hand, it seems that, due to their easy access to cities, roadside Pano communities had different hunting behavior in comparison to communities alongside rivers, by excluding secondary species from prey profiles. Pano roadside communities shifted into a market dependent diet that includes farmed fish, livestock and industrialized meat (Lima E.C., Pessoa, M. and Tavares, R.A. Personal communication). Because of elevated ammunition prices and low return rates for small species hunting, hunters substitute them for easily accessed alternative meat sold in markets, and restrict their hunting to large bodied mammals, mainly ungulates, even in situations of depleted wildlife (Kensinger 1975; Redford 1993; Jerozolimski and Peres 2003). Independent of my study, a contemporary wildlife census and hunting survey in the Campinas Indigenous Land indicates that most sensitive species are absent, and ungulates occur in very low densities, although these contribute most to wildmeat consumed, while smaller species occur at higher densities but yet are little hunted (Calouro 2007). This substitution pattern was observed in communities that have livestock as an alternative meat source, and therefore avoided hunting certain previously acceptable species (Jerozolimski and Peres 2003) or after an upgrade in hunting weapons and transportation to market (Winterhalder 1981; Hill and Hawkes 1983). Increasing the profits of hunting (i.e. shifting from subsistence to commercial harvest) changes the composition of off-take assemblages by focusing on more profitable species (Damania et al. 2005). Other indigenous people intensified their game harvest in certain species with increased access to market as a consequence of the

shift from subsistence to commercial hunting systems (Sierra et al. 1999). Although this could explain the pattern in these Indigenous Lands, if Pano engage in illegal commercial hunting it is to a much lesser extent than subsistence hunting, even though illegal wildlife commercialization in the region is high. All other Pano communities that had restricted access to markets, because of reduced size of nearby cities (i.e. Jordão and Santa Rosa do Purus) or their distance to them (i.e. Tarauacá and Cruzeiro do Sul to Praia do Carapanã and Breu Indigenous Lands), relied on a much more diversified fauna, shifting to smaller animals when large game is depleted.

Hence, resulting off-takes and mean prey weight in these villages reflect human processes more than prey abundances (Ling and Miner-Gulland 2006) and should not be used to infer wildlife status. Nonetheless, these results are important to understand the effects of roads in indigenous communities' behavior and societies.

Source-sink Dynamics of Game Populations

The Western Amazon Ecological Corridor has very low deforestation and human density, concentrated around small settlements, being one of the few residences of isolated indigenous groups in the Amazon (Castillo 2002). In such a landscape, game populations in the large tracts of forests may source populations to the disturbed areas (Novaro et al. 2000). Source-sink systems might be responsible for the persistence of healthy populations of sensitive species hunted for decades at unsustainable rates by indigenous people in a nearby region of Peruvian Amazon (Ohl-Schacherer et al. 2007). Peres (2001) suggests that hunting persistence (age of village) does not deplete most of the targeted species near villages because of the constant animal immigration from adjacent large underharvested forested areas, that include source populations. This dynamic might also be important to continually provide animals from a same source

area for Kaxinawá villages in the Jordão Indigenous Land (Constantino et al. 2008) and extractivist settlements in the Alto Juruá Extractive Reserve (Ramos 2005). At a larger scale, source-sink dynamics may be important to maintain the high proportion of ungulates and the presence of tapirs and large monkeys in villages located in the headwaters even after many decades of intense hunting. Conversely, source-sink dynamics in Indigenous Lands of the development frontier, containing more forest and less hunting pressure than outside, would not function due to reduced habitat or connectivity (Peres 2001; Hansen and DeFries 2007) or, paradoxically, would invert the flow, sourcing game to surrounding disturbed sites (Woodroffe and Ginsberg 1998; Naranjo and Bodmer 2007; Hansen, in preparation). In this zone, this process would exacerbate the negative effect of surrounding disturbances on wildlife hunted by Pano communities. Therefore, even if all Pano villages overharvest game species, depletion would be hidden behind the differential effects of source-sink dynamics.

Indigenous Lands as Protected Areas for Wildlife

Indigenous Lands avoid forest impoverishment by deforestation, logging, and fire (Nepstad et al. 2006; Adeney et al. 2009; Asner et al. 2009). The same pattern is observed across Indigenous Lands at the regional scale in Acre, where these are responsible for only 1% of State deforestation (Acre 2009). The capacity of Indigenous Lands to avoid depletion of other natural resources, however, has not been assessed at large scales using empirical data (Nepstad et al. 2006; Adeney et al. 2009). Azevedo-Ramos and colleagues (2006) even suggest that Indigenous Lands are key to mammal conservation in the Brazilian Amazon, because they overlap the geographic range of more species than other Protected Area categories and their large size and low human

population density would effectively protect mammal species that easily recover from hunting loss.

Pano people's land use and forest conversion, but not hunting, affected wildlife, principally large mammals. More important, disturbances associated to deforestation outside Indigenous Lands depleted wildlife in vulnerable villages and low defensible reserves whereas their size alone was not associated to wildlife conservation. For instance, most of these areas were smaller than the minimum required to support the average Amazonian sustainable harvest of most preferred mammals and birds (Peres 2001). The mobility of wildlife also makes it more likely to respond to disturbances from surrounding areas (Hansen and DeFries 2007). The high performance of Indigenous Lands in protecting Amazon forests, related to their capacity to inhibit disturbance in regions of high anthropogenic impact (Nepstad et al. 2006; Azevedo-Ramos et al. 2006), may not function for wildlife protection, as evidenced by the fact that the most affected Pano villages are located in Indigenous Lands in the Acre development frontier, thus subject to extreme external influence. Therefore, unlike other sources of forest degradation (e.g. fire and deforestation) Pano Indigenous Lands alone are not able to guarantee game conservation and maintenance of indigenous culture.

Management Implications and State Conservation Policy

The findings of this study, although concerning the Kaxinawá and Katukina people, could be extended to other central place hunter indigenous peoples targeting large vertebrates with the same hunting techniques in similar heterogeneous environments. This includes most of the non-isolated indigenous people in Acre State. The majority of governmental economic and environmental policies related to Indigenous Lands in Acre are designed and implemented by the State government, which understands them as

strategically important for conservation (Acre 2008, 2009). Hunting restrictions have been adopted as management norms (i.e. quotas, sex and age selection) by many indigenous and local people to guarantee local sustainable wildlife use, including in Acre (Gavazzi 2007; Zapata-Ríos et al. 2009). Yet if these norms are to succeed, decision-making and compliance has to occur in groups of neighboring villages, not only within single villages, although the efficacy of these restrictions may just influence wildlife conservation at very local scales. At regional scale, nevertheless, these management regulations probably do not ensure wildlife conservation. Yet, landscape-scale is the most appropriate to natural resource management policies that involve local people and government interaction in the tropics (Frost et al. 2006). In this sense, state policies that attempt to maintain indigenous culture and wildlife conservation should target the deforestation within and in the surroundings of Indigenous Lands, their land use associated disturbances, and non-indigenous hunting in the vicinity of Indigenous Lands. Regarding Pano people, there is a recent increase in deforestation inside some Indigenous Lands (SIPAM 2008), and my data indicate that it is positively associated with village linkage to market but negatively associated with monthly income per family (Appendix C), explaining together 63% of deforestation. Although the mechanisms for these associations are not explicit, the government could explore them, given that most of Pano income comes from retirement and salaries paid by the government to control deforestation. Indigenous Lands surroundings, in turn, are projected to be severely deforested in the near future, principally in the development frontier, if the current forest loss persists (IPAM and Consórcio MABE 2009). In response, the Acre government recently released a new policy to prevent and control deforestation in the state including

within Indigenous Lands (Acre 2008, 2009). Indigenous hunting and game could be considerably protected as a consequence of deforestation reduction and reforestation increase according to the government plan (Acre 2009). There is, however, no link between Indigenous Lands and their vicinity, as well as no mention concerning integrated wildlife management. Moreover, the government recognizes that these goals will only be achieved in a scenario of high governance, which depends upon several factors, including fund raising not yet guaranteed, to strengthen environmental management capacity and economic sustainability of environmental policies (Acre 2009). At a local scale, the Ethnozoning and Territorial Management Plan of Indigenous Lands currently supported by the State government (Acre 2008) can be the instruments through which indigenous people negotiate the landscape approach for wildlife management.

Considerations on Scale and the Absence of Hunting Effect

The findings in this study differed from most of the others studying the drivers of game depletion, since Pano hunting had minor effects (Peres 2000; Peres and Palacios 2007). Here I briefly explore some possible explanations. Using hunting data, instead of direct observation, to infer wildlife status can be biased due to non random sampling and a possible inability to capture light variation in game populations, since hunters persist in harvesting rare preferred species as by-catch hunting products (Rowcliffe et al. 2003). However, this is less likely to be the reason for dissimilar results, since Pano hunting off-take followed the widespread pattern observed in the Amazon that can be modeled according to the ecological theories used to derive the indicators. Moreover, I restricted my analyses to more than 9000 records from 35 communities of Pano people in a regional scale with small natural environmental variation, with similar hunting

preferences and strategies, that had reported hunting on more 50 animals and over a period of more than six months between 2005 and 2009. By contrast, other large extent studies analyzing drivers of game variation using hunter-kill profile in multiple tropical forest sites (>20) were based on meta-analysis of studies with decades of difference, conducted at continental scale or at intercontinental scale (Robinson and Bennett 2004; Fa and Peres 2001). These studies used lower cutting values to include sites of hunters of different cultures in the sample (i.e. undetermined number of animals in Peres (2001); >30 animals in Jerozolimski and Peres (2003); >40 animals in Fa et al. (2005)). These differences of culture and hunting technology, ecosystem productivity and spatial and temporal variation on wildlife abundances may introduce noise in these results (Emmons 1984; Redford and Robinson 1987; Lupo and Schmitt 2005; Peres and Palacios 2007; Nasi et al. 2008). Nevertheless, I acknowledge that the ecological theories supporting the indicators are not expected to explain all the variation in hunters subsistence systems (Hawkes et al. 2008) and that neglected environmental differences (i.e. soil fertility and hydrological regime) within the scale of this study (i.e. grain and extent) may influence game populations non randomly.

Many other studies do not clearly define hunting pressure (Rist et al. 2008), since it is frequently hard to determine the actual number of hunters using a territory and the intensity of hunting pressure (Nasi et al. 2008). For example, the common use of number of people in a settlement to represent hunting pressure (Jerozolimski and Peres 2003; Fa et al. 2005) might not be appropriate, as many indigenous communities fraction in response to depleted wildlife, becoming a settlement with a small number of people in a depleted site (Ferguson 1989). Also, a local community hunting territory is

often used by neighbors or poachers or is adjacent to areas more intensively harvested, affecting wildlife hunted by locals (Ferguson 1989; Stearman 1990). Therefore, attributing the responsibility of game depletion to local people living within Protected Areas might underestimate the effect of encroachment, animal population dynamics, spread of wildfire, and edge effects: some of the most common external threats to biodiversity within Brazilian Conservation Units (Peres and Terborgh 1995; Carrillo et al. 2000; Bruner et al. 2001; Laurance et al. 2000; Nepstad et al. 2006). I was able to isolate indigenous hunting pressure from other factors due to extensive field surveys as well as identify the Pano territories exclusively used by them. Nonetheless, hunting from outside may still have some effect on wildlife since I could not disentangle the effects of deforestation from non-indigenous hunting on game hunted by the Pano. These factors may influence wildlife in synergy, which make it more difficult to distinguish between their effects (Nasi et al. 2008; Fa and Brown 2009; Parry et al. 2009b). In addition, many tropical forest species that are sensitive to hunting are also sensitive to habitat loss (Michalski and Peres 2007; Rist et al. 2009) and fire effects (Peres et al. 2003; Barlow and Peres 2006). Therefore, fluctuations in their populations, frequently assumed to be a consequence of hunting pressure may be related erroneously to one of the processes if the others are not properly acknowledged. Satellite based information only recently became available, yet for restricted regions, allowing the regular use of habitat disturbance variables in scales appropriate to game species, mostly neglected in previous studies (Redford and Robinson 1987; Peres 2001; Jerozolinski and Peres 2003; Fa et al. 2005).

The scale of this study is a plausible explanation for the marked difference in results. Most wildlife and hunting research is conducted either at local or global scales in the Amazon, mainly restricted by methodological issues (Nepstad et al. 2006). Spatial scale alone can explain much of the variation in the response of wildlife to habitat loss (Hill and Hamer 2004; Dumbrell et al. 2008) and to human occupancy (Pautasso 2007). Robinson and Bodmer (1999) claim that indigenous hunting has been sustainable at the landscape level whereas, locally, many species must have been extirpated, since hunting effects operate only around settlements (Lewis et al. 2004). Pano hunting had only local effects on wildlife, which might have been suppressed by the landscape differences in deforestation. Likewise, most local studies report the hunting-induced depletion trend, although confounded by source-sink dynamics (Ohl-Schacherer et al. 2007), whilst recent studies at similar regional scale to mine have assessed the effects of hunting and habitat disturbance on game animals (Ramos 2005; Peres and Michalski 2005, 2007; Urquiza-Hass et al. 2009), with impressive indications of the importance of deforestation over local hunting. In this intermediate spatial scale (grains circa 10km diameter and extent circa 70,000km²), species richness is expected to have no relationship with human presence (Pautasso 2007). Although local and global studies are relevant they might not be appropriate to orient policies that are usually implemented at regional levels (Ferrier 2002; Fisher and Owens 2004; Frost et al. 2006; Fritz et al. 2009), the scale of decisions regarding Indigenous Lands in Acre, and particular by appropriate to study large mammals in the Amazon due to their metapopulation dynamics.

In addition, major drops in mammals densities due to hunting are more likely to occur in previously un hunted sites than those already exploited (Nasi et al. 2008; Fa and Brown 2009). Many studies suggest that hunting depletes wildlife, based on comparisons between un hunted and hunted sites (Carrillo et al. 2000; Peres 2000) or following game population changes in the first years after settlements establishment (Vickers 1980; Ayres et al. 1991). This is certainly not the case in Pano villages as the region is subject to centenary intense hunting but only recently has been affected by major land conversion, more prominent in the development frontier. A selective filter may operate after initial hunting pressure, where tolerant species and population abundances remain stable in sites (Balmford 1996; Cowlshaw et al. 2005) until other processes start adding to the negative effect of hunting (Cullen et al. 2000). Frontiers are more prone to witness extirpation of large bodied species (Davis et al. 2008; Fritz et al. 2009) since sensitive species are still present but unprotected from anthropogenic threats (Balmford 1996).

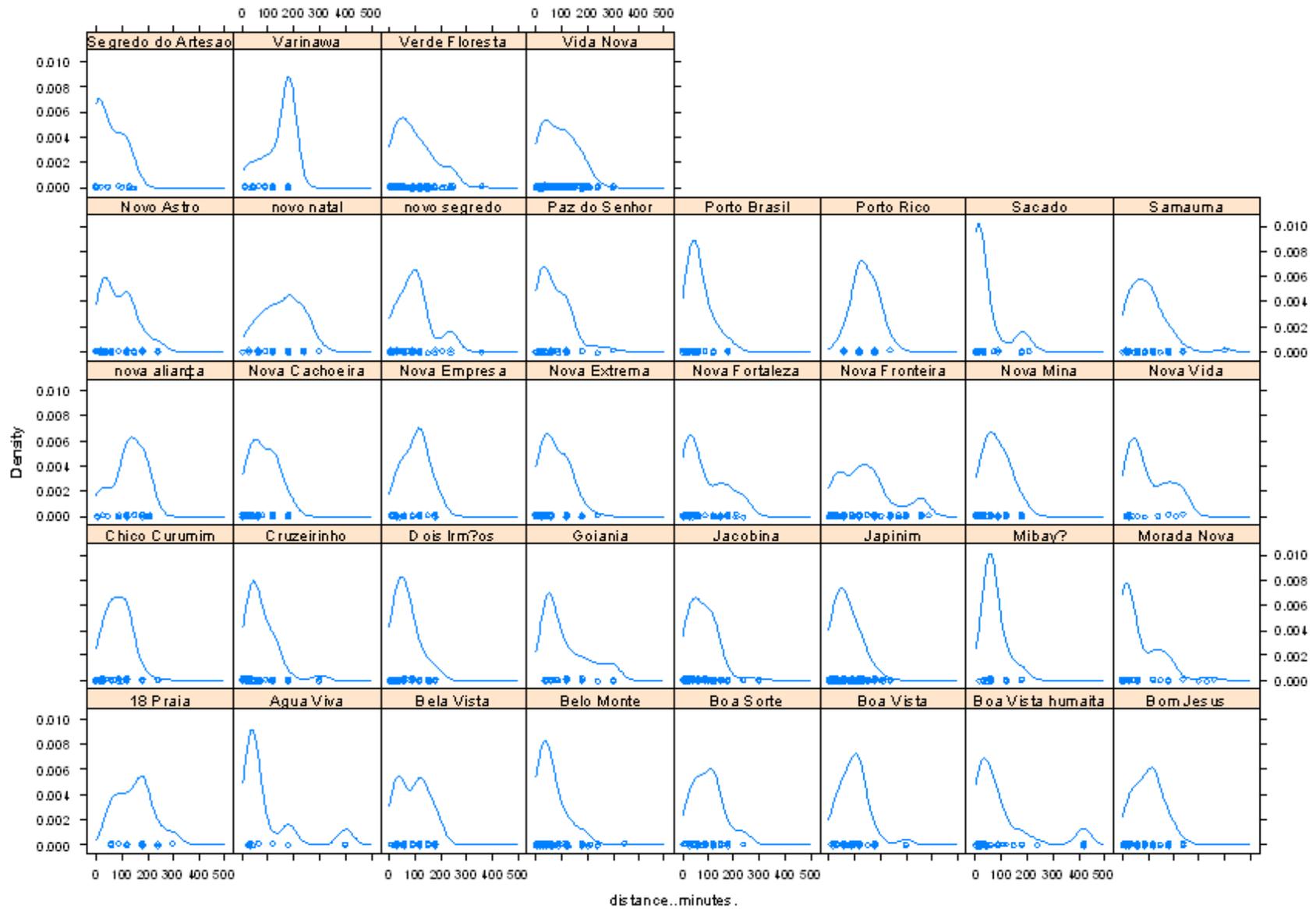
The temporal dimension of this study may also have implications in my findings. In 2005, the southwestern Amazon experienced an abnormally severe drought with widespread and intense associated fire (Aragão et al. 2007). Although villages all over the state were affected by uncontrolled fire, those located in the development frontier were more affected by external disturbance than those protected in the headwaters, with several recurrent burns (Carneiro Filho and Souza 2009). Hence, burning patterns overlaps deforestation distribution, making the effects of these threats hard to distinguish. Despite our vague understanding of long term responses of wildlife to forest fire, certain species seems to be absent from recurrent burned sites or have abundance

reduced (Barlow and Peres 2006). However, if game species in Acre are somehow responding to the extended wildfire that occurred in 2005, climate change must be a major threat to those species because the southwestern Amazon region is particularly affected in cases of North Atlantic warming, becoming more prone to forest fires, principally alongside roads (Aragão et al. 2007).

CHAPTER 5 CONCLUSIONS

Pano people's hunting does not cause major impacts in the game they hunt. Even in an Amazon region relatively conserved, deforestation is the main driver of gradual game depletion. Forest loss and probably other associated disturbances (i.e. fragmentation and fire) reduce game populations and extirpate most sensitive species inside Indigenous Lands. Indigenous Lands may avoid forest loss but do not alone guarantee wildlife conservation in Acre. Land use and hunting by non-indigenous neighbors, and the Indigenous Land's location and design, may contribute considerably to game depletion in Pano villages. Although hypothesized for regional scales, the null indigenous hunting effect as opposed to habitat loss impact on game had not been reported in the literature, due to the difficulty of disentangling factors, and data and scale restrictions. Roads impacted wildlife negatively by exacerbating hunting effects, while restricting hunters' dependence on large game, independence of its abundance, by allowing easier access to alternative meat. It is possible that the anticipated disaster of near future deforestation will not affect game species and Pano people, if the recent Acre State policy for deforestation and fire control and prevention is effectively implemented. However, policies that integrate Indigenous Lands and its vicinity are still required.

APPENDIX A
KERNEL DENSITY ESTIMATES OF DISTANCE OF PREFERRED ANIMALS HUNTED
FROM THE VILLAGE



APPENDIX B
WILDLIFE STATUS INDICATORS AND ITS RELATED ECOLOGICAL MODELS AND
THEORIES

Indicator	Model/Theory	Expected outcome	Comments
Mean prey weight	Optimum Foraging Theory	Average weight of hunted animals decline in depleted sites in response to decline in large bodied vertebrates	Provide a proxy for the assemblage of game animals regardless of species, likely to be related to variation in population abundances
Capture per Unit of Effort (CPUE) of preferred species	Optimum Foraging Theory/ Ecological Economics	Hunters spend more time to pursue meat of preferred animals in depleted sites in response to decline in large game abundance	Restricted to preferred species since return rates of secondary animals may not reflect abundance; Allows for indication of hunting success on single species or taxa
Sensitive species richness	Indicator Species	Species sensitive to human disturbance of different natures decline to level that are no longer hunted, ultimately becoming extirpated, with increasing wildlife depletion	Given non random extinction process certain species reflect overall status of wildlife assemblage status
Mode distance of preferred species from the village	Central Place Forager Model	Hunters catch preferred animals farther away from villages where central place hunting depletes game populations	Restricted to preferred animals since the distance of secondary animals might not reflect species abundances; given a non parametric distribution of distance values, mode was more appropriate to describe the most common distance

APPENDIX C
PEARSON'S CORRELATION MATRIX OF VARIABLES POSSIBLY INFLUENCING
HUNTING IN INDIGENOUS VILLAGES, CONSIDERING 29 VILLAGES

Explanatory variables	Hunting pressure				Land use				Socio-economy			
	Population size	Village age (years)	Square Root of number of hunters	Density of Indigenous people (people/5km radius buffer)	Log ₁₀ deforested area in 5km radius buffer (ha)	Square root of distance to entrance of Indigenous Land (km)	Log ₁₀ deforested area in 5km buffer but inside Indigenous Land (ha)	Log ₁₀ deforested area in 5km buffer but outside Indigenous Land (ha)	Log ₁₀ number of employees	Income per family (\$/family)	Distance to market (km)	Log ₁₀ Animal Unit ^a
Population size ^a	1	0.189 (0.33)	0.672 (0.00)***	0.181 (0.35)	-0.187 (0.33)	0.030 (0.88)	-0.14 (0.40)	0.04 (0.88)	0.733 (0.00)***	-0.143 (0.46)	-0.70 (0.72)	0.02 (0.9)
Village age ^a		1	0.132 (0.49)	-0.07 (0.7)	-0.076 (0.69)	0.156 (0.42)	0.03 (0.84)	-0.06 (0.81)	0.294 (0.12)	0.209 (0.28)	0.025 (0.90)	0.04 (0.83)
Square Root # Hunters			1	-0.127 (0.51)	-0.576 (0.00)**	0.353 (0.06)	-0.50 (0.00)**	0.37 (0.20)	0.598 (0.00)**	0.119 (0.54)	0.353 (0.06)	0.06 (0.76)
Density of Indigenous people ^a				1	0.348 (0.06)	-0.22 (0.25)	0.37 (0.3)*	-0.29 (0.27)	0.107 (0.58)	-0.029 (0.88)	-0.258 (0.18)	0.31 (0.1)
Log ₁₀ Deforested area (ha) ^a					1	-0.719 (0.00)***	0.94 (0.00)***	0.73 (0.00)***	-0.243 (0.20)	-0.568 (0.00)**	-0.748 (0.00)***	0.46 (0.07)*
Square Root of distance to entrance of Indigenous Land						1	-0.63 (0.00)***	-0.74 (0.00)**	-0.071 (0.71)	0.433 (0.02)*	0.887 (0.00)***	-0.37 (0.05)*
Log ₁₀ deforested area in 5km buffer but inside Indigenous Land (ha)							1	0.26 (0.32)	-0.27 (0.15)	-0.54 (0.00)**	-0.74 (0.00)***	0.30 (0.11)
Log ₁₀ deforested area in 5km buffer but outside Indigenous Land (ha)								1	0.00 (0.99)	-0.43 (0.12)	-0.20 (0.51)	0.26 (0.40)

APPENDIX C. Continued

Explanatory variables	Hunting pressure			Land use				Socio-economy				
	Population size	Village age (years)	Square Root of number of hunters	Density of Indigenous people (people/5km radius buffer)	Log ₁₀ deforested area in 5km radius buffer (ha)	Square root of distance to entrance of Indigenous Land (km)	Log ₁₀ deforested area in 5km buffer but inside Indigenous Land (ha)	Log ₁₀ deforested area in 5km buffer but outside Indigenous Land (ha)	Log ₁₀ number of employees	Income per family (\$/family)	Distance to market (km)	Log ₁₀ Animal Unit ^a
Log ₁₀ # Employees									1	0.273 (0.15)	-0.071 (0.71)	0.21 (0.28)
Income per family										1	0.368 (0.05)*	-0.13 (0.49)
Distance to market											1	-0.21 (0.26)

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BIOGRAPHICAL SKETCH

Pedro A.L. Constantino was born in Rio de Janeiro, Brazil, and received his Bachelor of Science degree in biological sciences at the Universidade Federal do Rio de Janeiro studying insect ecology. His research on gall midge development received Honor Mention from the Brazilian Botanical Society in 2004. Since then he works in collaboration with the local NGO Comissão Pró-Índio do Acre and many indigenous people of Acre. He helped the implementation of the natural resource use monitoring program in Indigenous Lands of Acre and currently collaborates in the Indigenous Agroforestry Agents Capacity-Building program of CPI-AC. In 2007, Pedro was an ACLI visiting scholar at the University of Florida. Meanwhile, he worked for the Rio de Janeiro Botanical Garden on the conservation of threatened Brazilian orchids.