DEVELOPING NOVEL METHODOLOGICAL APPROACHES TO UNDERSTAND THE HARVEST AND CONSERVATION OF NEOTROPICAL WILDLIFE

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DECLARATION

I hereby declare that this work has been originally produced by myself for this thesis and it has not been submitted for the award of a higher degree to any other institution. Inputs from co-authors are acknowledged throughout.

Natalie Swan,

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It seems to me that the natural world is the greatest source of excitement; the greatest source of visual beauty; the greatest source of intellectual interest. It is the greatest source of so much in life that makes life worth living. The truth is: the natural world is changing. It is the most precious thing we have and we need to defend it.

Sir David Attenborough

Abstract

Human impact on the natural world is pervasive. The effects of historical and contemporary industrialisation, agricultural expansion and globalisation can be felt even in remote environments. Addressing anthropogenic threats to biodiversity is becoming ever more urgent, and ever more challenging. Conservationists must navigate increasingly complex problems that consider not only natural processes, but also the inextricable social dimensions of environmental change, and must do so with limited human and financial resources. The challenge is particularly great in tropical regions. These are home to the majority of terrestrial biodiversity and are facing unprecedented pressures due to expanding and impoverished human populations, urbanisation and exploitation of natural resources. Conservation strategies in the tropics increasingly recognise the need to embrace social-ecological approaches, often designed around initiatives that aim to safeguard biodiversity and the benefits provided to humans by ecosystems, and promote social progress. Yet development of monitoring techniques to better inform these strategies has lagged behind. Despite recent growth in the presence of social science theory and methods in conservation, research to characterise threats and identify conservation priorities rely heavily on traditional ecological methods. These methods have limitations, including restricted replication capacity, small spatial scales and sampling error. Perhaps more importantly, they fail to elicit the social context of human activities and behaviours.

The main objective of this thesis was to critically examine and develop methods to address complex conservation problems in tropical forest contexts strongly influenced by human actions. The research is based in Brazil, a mega-diverse country experiencing turbulent economic and political times.

The thesis begins in the Brazilian Amazon, where recent evidence indicates that urban consumption and commercial trade of wild-meat may be widespread, presenting an important threat to Neotropical biodiversity. Yet adequate regional data is scarce. Subsequently the first two data chapters of this thesis examine two approaches that could provide important insights into the extent and characteristics of wildlife harvest and trade across large spatial scales: expert knowledge and federal enforcement reports. First, using caiman as a model taxon, I surveyed experts across the Brazilian Amazon using a Likert-style questionnaire (Chapter 2). The results of expert responses revealed novel evidence of common and geographically widespread caiman hunting, driven in part by urban demand for meat and resulting in long-distance trade networks. Chapter 3 examines the potential of federal enforcement data to provide valuable regional information on illegal harvest and trade activities, utilising reports of enforcement events in 549 Amazonian municipalities. I also examine spatial and temporal patterns of institutional capacity of Brazil's environmental agency to understand the efficacy of governance in tropical forests against this cryptic and hard-to-detect activity; and in turn how these realities impact our interpretations of the species data contained within enforcement reports. The analyses revealed evidence of inadequate institutional capacity and low enforcement of wildlife crime, particularly in smaller towns far from deforestation frontiers. Nonetheless, the approach yielded vital conservation information on spatial patterns and dynamics of species-level harvest and trade, including evidence of large-scale commercial trade in larger cities, and local-level harvest of vulnerable terrestrial vertebrates. The study also highlighted a potential Amazonian enforcement vacuum resulting from decentralization and institutional reforms.

From here, we move to the Atlantic forest, a severely modified biome and conservation hotspot, to explore the use of Local Ecological Knowledge (LEK) to inform on local-scale occupancy and population trends of large-bodied mammals and birds. I conducted interviews with rural people to assess their knowledge of selected native species, and also to elicit their perceptions of social, environmental and economic processes of change. The results demonstrate that LEK can provide valuable information on species responses within severely modified tropical landscapes. Perhaps more importantly, qualitative insights from respondent interviews illustrated the inter-linked social, economic and political drivers of changing landscapes and livelihoods that have shaped contemporary species patterns.

The findings of this thesis demonstrate the value of alternative and innovative research methods for eliciting important conservation-relevant information in tropical forest contexts. The research presented highlights the importance of critical and robust development and application of methods, recognizing the challenges that stem from integrating social-ecological knowledge systems and approaching complex problems at different spatial scales.

Keywords: tropical forests; wildlife harvest; mixed-methods; spatial scales; human-modified landscapes; conservation.

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GENERAL INTRODUCTION

Chapter 1 General Introduction

1.1 Tropical forests in the Anthropocene

Human impact on the natural world is now ubiquitous. The term “Anthropocene” appears increasingly in scientific and popular rhetoric (Caro et al. 2012; Schulte 2016; Young 2016), and defines a new geological epoch in which humans have become a dominant force in altering the earth’s dynamics, perhaps even beyond reversible thresholds (Lennon 2015; Malhi et al. 2014). With the industrial era and accelerating globalization responsible for widespread land conversion, atmospheric changes, shifting nutrient cycles, and mass species extinctions, it is clear that the world is changing, rapidly, globally, and directionally (Chapin and Fernandez 2013), and the consequences for natural environments and biodiversity are pervasive and diverse in both scale and impact (Gardner et al. 2010).

Home to between one-half and two-thirds of the world’s terrestrial biodiversity, tropical forests are arguably the most significant biomes on the planet (Gardner et al. 2010), whilst also playing a disproportionate role in global carbon and energy cycles (Dixon et al. 1994; Wright 2005). Congruent with global patterns of human development, they are also subject to increasingly severe anthropogenic pressures, with deforestation in the tropics occurring at an unprecedented rate of up to 130,000 km^2 a year (Laurance 2010). Occurring predominantly in developing nations, the destruction of tropical forests is driven largely by rapidly expanding and impoverished populations, agricultural intensification, infrastructure and extractive activities (Geist and Lambin 2002; Wright 2010). These activities result in habitat destruction, fragmentation, loss of ecosystem function, and species declines (Gaveau et al. 2009; Geist and Lambin 2002; Wright and Muller-landau 2006). Moreover, it has been estimated that tropical deforestation accounts for 14-20% of global greenhouse gas emissions (Pan et al. 2011); consequently, mounting international focus has been placed on addressing the most conspicuous and tangible threat to tropical forests, that of forest clearance (Arima et al. 2014; Boucher et al. 2014; Nepstad et al. 2014; Tucker and Townshend 2000).

However, despite the understandable prominence of deforestation, anthropogenic threats to tropical forests manifest in different ways and at different scales (Peres, Barlow, and Laurance 2006). Many non-structural anthropogenic perturbations are less obviously apparent, yet are nonetheless insidious in their impact on forest biota, including surface fires (Barlow and Peres 2004; Laurance 2003), alien species invasions (Asner and Vitousek 2005), proliferation of pathogens (Daszak, Cunningham, and Hyatt 2003) and extraction of non-timber forest products (NFTPs) (Peres and Lake 2003).

1.2 Conservation in a human-modified world

Against the ostensibly unrelenting backdrop of human population growth and consumption of natural resources, the challenge of safeguarding the future of tropical forest species is daunting. Conservation is fundamentally a crisis-driven, problem-solving field (Meine, Soulé, and Noss 2006), and as such is inherently dynamic, invariably involving new interdisciplinary connections and practices (Reyers et al. 2010). For much of the last century conservation in the tropics relied heavily on protectionist strategies that aimed to remove humans from supposedly "pristine" natural environments through the designation and enforcement of strict Protected Areas (Geldmann et al. 2013; Kilbane Gockel and Gray 2009). However, in recent decades, this exclusionary paradigm has been widely viewed as a failure, not only in terms of achieving biodiversity conservation goals, but also for potentially being ethically problematic and misanthropic (Kilbane Gockel and Gray 2009; Redford, Robinson, and Adams 2006). Consequently there has been a shift towards more holistic conservation approaches, focusing on strategies that seek to account for the needs of local communities within conservation, including community based conservation initiatives and Integrated Conservation and Development Projects [ICDPs]. These practices seek to advocate the much-heralded objective of delivering 'win-win' outcomes that both conserve biodiversity and enhance socio-economic development (Berkes 2007; Reed et al. 2016, 2017). Concurrently, the conservation discourse has become widely accepting of the fact that, although necessary, protected area networks (even with successful participatory management) are grossly inadequate to safeguard tropical biota, and conservationists are increasingly focusing on effective management of human-modified systems (Chazdon et al. 2009; Koh et al. 2010; Sodhi et al. 2010).

1.3 Methods in conservation

Regardless of the driving underlying paradigms, conservation as an applied discipline must operate within substantial practical constraints. The most pressing of which is perhaps time – we need to act fast. The moniker "crisis-driven" does not necessarily refer only to the urgency of the conservation issue, but also that during crises there is rarely time to fully assess a situation carefully, obtaining all the necessary information required, before one must act (Doak and Mills 1994). The situation is compounded by finite and often inadequate financial and human resources, particularly within developing tropical nations. Consequently, the identification of conservation priorities that maximise the benefit of investment, whether at species, landscape or regional level, has been a key strategy (Mills, Soule, and Doak 1993; Myers et al. 2000).

The traditional methods used to inform conservation strategy and identify priorities have been based primarily within the natural sciences, aiming to elicit relevant biological and ecological information on species, communities and ecosystems (Chiarello 2000; Durant et al. 2011; Hansen et al. 2001; Jorge et al. 2013; Myers et al. 2000). Such data are no doubt critical; however, acquiring such information in the context of tropical forests frequently requires carefully planned and resource-heavy fieldwork, using techniques such as terrestrial line transects (Chiarello 2000; Michalski and Peres 2007;

Singh and Milner-Gulland 2011). These techniques suffer from a limited replication capacity to detect temporal patterns, are often restricted to small spatial scales and can result in significant sampling error (Fragoso et al. 2016; Munari, Keller, and Venticinque 2011; Parry and Peres 2015). As such, conservation has been greatly hindered by a paucity of empirical data, especially in the vastly speciose and heterogeneous tropics (Elith et al. 2006; Gardner et al. 2007). In response to this lack of data, researchers have developed various ways of providing meaningful information through predictive analyses and scenario building, making use of species distribution models and Bayesian methods (Elith et al. 2006; Gardner et al. 2007; Wade 2000; Wright and Muller-landau 2006).

But as Balmford and Cowling (2006) have pointedly noted "conservation is primarily not about biology but about people and the choices they make". In this increasingly anthropogenic world it is accepted that social and environmental problems are inextricably linked, and in order to identify solutions we must obtain a comprehensive understanding of the intricacies of the problem (Berkes 2004; Fox et al. 2006; Poe, Norman, and Levin 2014) – and the methods we utilise must reflect this. Techniques and methods taken from the social sciences are being increasingly used to understand the human dimensions of conservation problems (Newing 2010), including the use of participant observation (Baral, Stern, and Heinen 2007), qualitative interviews and focus groups (Kahler and Gore 2014), and elicitation of local ecological knowledge (LEK) (Brook and McLachlan 2008; Parry and Peres 2015). However, although the paradigm shift towards a more socially conscious conservation and an increase in social method application demonstrates good intentions, robust method development and effective integration has lagged behind. The muted success of ICDPs and community-based conservation initiatives suggests that their design and implementation often fail to effectively account for the complex realities of people and natural systems (Baral et al. 2007; Berkes 2007; Brooks, Waylen, and Mulder 2013; McShane and Wells 2004).

1.3.1 In search of validation

Researchers have highlighted the methodological challenges of integrating natural sciences with social disciplines that stem from differing perspectives, terminologies and concepts, and even from disciplinary prejudices (Fox et al. 2006; St. John et al. 2014). In particular, natural scientists are often sceptical of the value of qualitative data to provide an empirical basis to reliably inform management decisions (St. John et al. 2014). However, given the serious consequences of failed conservation projects – for the environment as well as for future initiatives – conservationists cannot afford to disregard methods and data that do not subscribe to "traditional" empirical paradigms (Drury, Homewood, and Randall 2011; Sodhi and Ehrlich 2010). Instead we must make moves towards validating alternative methods and data sources and learn to appreciate what they can and cannot tell us (St. John et al. 2014; Keane 2013). This thesis aims to contribute to this important objective by exploring and critically appraising different methods and data sources in the context of tropical forest conservation, and by so doing highlight their value, potential biases and reduce the risk of their misinterpretation.

1.3.2 Developing methods for complex problems at multiple scales

The natural environment is a complex adaptive system, hierarchically organized with nested subsystems and understanding it requires diverse concepts and principles (Berkes 2007; Doak, Marino, and Kareiva 1992). Similarly, the social systems pertinent to conservation are complicated and polyvalent, and can be conceptualized over a multiplicity of local, landscape, regional and international scales (Berkes 2007; Cumming, Cumming, and Redman 2006).

Conservation has thus to operate across two much complicated systems and must negotiate these systems with a high degree of pragmatism, selecting from the various scales those which best address any given problem (Ostrom, Janssen, and Anderies 2007; Reed et al. 2016; Sayer et al. 2013). This pragmatism must extend to method selection, including the spatial scale at which methods are employed and developed. Eliciting large-scale patterns can be extremely useful, helping to characterize the extent of a problem and identify spatial priorities for conservation action. However, we must remain cautious of broad generalizations and panacea solutions and recognize that the significance of the local context can greatly influence the efficacy of implemented strategies (Nyaki et al. 2014; Ostrom et al. 2007; Sayer et al. 2013). Understanding and interpreting the value and pitfalls of data elicited at different scales is therefore critical for successful mitigation interventions.

1.4 Study region

The research presented in this thesis is based in two ecological biomes in Brazil. As the fifth largest country in the world and the first of the megadiversity countries (Mittermeier and Mittermeier 1997), Brazil offers an ideal context to explore diverse and complex socio-ecological conservation problems. The first two data chapters of this work explore methods to understand the harvest of wild-meat in the Amazon rainforest, while the final data chapter assesses the value of Local Ecological Knowledge (LEK) within the context of the severely degraded Atlantic forest. The following will offer a brief summary of these important biomes and their relevance to tropical conservation.

1.4.1 The Brazilian Amazon

The Amazon rainforest is the largest and most biodiverse expanse of tropical forest on Earth, spreading across nine South American countries. Seventy percent of the Amazon basin is within Brazil's borders, making Brazil the steward of the largest tract of tropical forest of any nation. The Amazon's local and global importance in providing a myriad of ecosystem services (including biodiversity conservation and climate regulation) has often pushed Brazil to the forefront of global concerns over deforestation, biodiversity loss and climate change (Ferreira et al. 2014).

Deforestation and environmental policy

Deforestation has been intense in the Brazilian Amazon since the early 1970s, driven notably by land conversion for cattle-raising and more recently, soy production (Hargrave and Kis-Katos 2012; Nepstad et al. 2014; Soares-Filho et al. 2006). Efforts to quantify and monitor the spatial extent of Amazonian forest loss (particularly through the use of remote sensing techniques) (Laurance, Albernaz, and Costa 2002; Mas 1999; Walker 2012), alongside better understanding of the drivers and dynamics of deforestation activities (Freitas, Hawbaker, and Metzger 2010; Soares-Filho et al. 2006), enabled the design and implementation of new forest protection strategies. These included improved frontier governance, collaborative interventions in soy and beef supply chains and the expansion of protected area networks (Nepstad et al. 2014). Such changes effectively brought about a 70% reduction in deforestation rates in the latter part of the last decade, earning Brazil a reputation as an environmental leader (Arima et al. 2014; Nepstad et al. 2014). More recently, however, political changes and a shift in government priorities toward the expansion of mining and extractive activities has led to legislative weakening of Brazil's Forest Code and environmental policies (Ferreira et al. 2014). Such actions jeopardise Brazil's past conservation successes and demonstrate the fragility of tropical conservation when pitted against economic and social development (Ferreira et al. 2014).

An increasingly urbanised wilderness

Despite retaining vast tracts of intact primary forest and a relatively low population density (e.g. average population density the Brazilian Amazon is around 3-4 inhabitants/km^2; FAO, 2016), the growth of urban areas is profoundly altering socio-ecological dynamics within Amazonia (Aide and Grau 2004; Parry, Barlow, and Pereira 2014). Three quarters of the population in the Brazilian Amazon now live in towns and cities (IBGE 2010), and the urban population in the Amazonian prefrontier is growing at a rate of 4.5% *per annum* (Parry et al. 2014). It has been suggested that a rural exodus and increasing urbanisation could inadvertently save tropical biodiversity by reducing dependency on natural forest resources and allowing for the recovery of secondary forests (Aide and Grau 2004; Wright and Muller-landau 2006). However, recent research has indicated that such assertions may be overly optimistic and deceptively simplistic by highlighting gaps in our knowledge about the conservation potential of forest regrowth and uncovering significant urban demand for forest resources (Chazdon et al. 2009; Parry et al. 2014; Parry, Barlow, and Peres 2009; Van Vliet et al. 2015a). The latter – urban demand for natural forest resources – is a major theme in this thesis, specifically with regard to commercial trade and urban consumption of Amazonian wildlife.

Wild-meat in the Amazon – the archetypal tropical conservation issue

The harvest of wildlife for human consumption is a pan-tropical activity that has gained prominence in recent decades as an important driver of species declines (Fa et al. 2006; Milner-Gulland and Bennett 2003). Recent studies have shown that the cascading effects of defaunation through overhunting also have the potential to significantly erode carbon storage (Bello et al. 2015; Osuri et al. 2016; Peres et al. 2015), offering another stark reminder of the interconnectedness of our planet's natural systems and the global reach that can stem from local or regional level human activity.

The hunting of wild terrestrial vertebrates for meat (often termed "bushmeat") is an archetype of contemporary tropical conservation issues: a complex multi-scale activity that sits firmly within both biodiversity concerns and social development. Alongside defaunation, unsustainable harvesting jeopardises the food and livelihood security of some of the poorest people on Earth, as it is frequently characterized as a subsistence activity for the marginalized rural poor (Bennett et al. 2007; de Merode, Homewood, and Cowlishaw 2004). However, consumption of wild-meat – which can include terrestrial and aquatic wildlife – occurs in both rural communities and urban areas, and commercial supply chains can be hundreds of kilometres long, supplying even international markets (Chaber et al. 2010; Milner-Gulland et al. 2002). Studies on the complexities of wild-meat commodity chains in Africa and Asia have highlighted the multi-dimensionality of the trade (Drury 2009; Mendelson, Cowlishaw, and Marcus 2003); consumption patterns can be influenced by topography, available infrastructure, market access and social and cultural factors (East et al. 2005; Fa, Peres, and Meeuwig 2002).

Conservation biologists contend that hunting levels in the tropics have increased dramatically in recent decades as tropical forests become more accessible to hunters, human population densities increase, traditional hunting practices are replaced with efficient modern technologies, and trade becomes more commercial with an increased urban demand for wild-meat (Milner-Gulland and Bennett 2003; Robinson and Bodmer 1999). Fa, Peres, and Meeuwig (2002) noted that bushmeat hunting is the single most geographically widespread form of resource extraction in tropical forests, and can affect even the core of some of the largest and least accessible nature reserves. Understanding the key drivers and dynamics of the bushmeat trade is vital; the complex interactions between ecological, socio-economic and cultural dimensions have hampered conservationists and policy-makers at both local and national level (Nasi, Taber, and van Vliet 2011; van Vliet et al. 2015b).

Much research on wild-meat harvest and consumption has been carried out in West and Central Africa (Bennett et al. 2007; Brashares et al. 2011; Nielsen, Meilby, and Smith-Hall 2014), where it has been asserted that the scale of commercial consumption and trade has reached a crisis point for African wildlife conservation (Cowlishaw, Mendelson, and Rowcliffe 2005; Nasi et al. 2008). In the Neotropics, however, our knowledge of the extent and impact of hunting activities on fauna had until recently been limited to the rural subsistence paradigm (Bodmer, Eisenberg, and Redford 1997; Franzen 2006), following the (untested) assumption that urban Amazonians were not consumers of wild-meat (Nasi et al. 2011; Rushton et al. 2005). Given more contemporary evidence of high levels of commercialization and consumption of wild-meat in Amazonian cities, this assumption is being re-examined (Baía Jr, Guimarães, and Le Pendu 2010; Parry et al. 2014; Van Vliet et al. 2015a), yet comprehensive regional data is notable by its absence.

1.4.2 The Atlantic forest

The Atlantic Forest is the second largest rainforest of the American continent. It originally stretched almost continuously along the Brazilian coast, extending into eastern Paraguay and northeastern Argentina, covering more than 1.5 million km^2 (Tabarelli et al. 2005). Extremely heterogeneous in composition, the Atlantic forest covers a wide range of elevations, climatic belts and vegetation formations, from tropical to subtropical. This heterogeneity fostered high levels of species richness and endemism; however, centuries of human occupation have destroyed much of the original habitat, with estimates of primary forest cover reduced to a mere 11.6% (Scarano and Ceotto 2015). Consequently the Atlantic forest is considered one of the most threatened tropical biomes and the "hottest" of conservation hotspots (Gardner et al. 2010; Laurance 2009; Myers et al. 2000).

The long history of human occupation and consequent deforestation in the Atlantic forest is closely related to the major cycles of Brazilian economic growth over the past five centuries (de Rezende, et al. 2015; Teixeira et al. 2009). Contemporary deforestation in this biome has been mostly driven by commercial eucalyptus plantations and sprawling urbanisation (Lira et al. 2012; Metzger 2009; Ribeiro et al. 2011). Moreover, the historical limits of this biome houses more than 60% (c.a. 120 million) of the Brazilian population and is responsible for nearly 80% of all Brazilian GDP (concentrated particularly within the highly developed south-eastern states of São Paulo, Rio de Janeiro and Minas Gerais) (Pinto et al. 2014). As a consequence of this relentless degradation and intense anthropogenic pressures, the remaining Atlantic forest exists now in a highly fragmented state, defined by human modified landscapes which are typically agro-mosaics dotted with scattered forest remnants (Gardner et al. 2010; Ribeiro et al. 2009, 2011). As a severely modified biome, the Atlantic forest provides a critical setting for studying the long term consequences of human disturbances, allowing better predictions for other regions that are only recently experiencing intense levels of human occupation (Gardner et al. 2010).

Many studies have documented the detrimental impacts of fragmentation and habitat loss on biodiversity in the Atlantic Forest, including defaunation and increased extinction risk of many species (Canale et al. 2012; Tabarelli et al. 2010). It has been suggested that the greatest threats to biodiversity in the Atlantic forest are less attributable to current land use pressures, but are driven principally by the time-delayed effects of past land use change, citing the detrimental impacts of edge effects and the gradual reduction of regional connectivity produced by stochastic local extinction processes (Gardner et al. 2010; Tabarelli et al. 2005). Recent evidence of forest regeneration (Baptista and Rudel 2006; Ferreira, Alves, and Shimabukuro 2015; da Silva et al. 2016), concurrent with Forest Transition theory (Rudel et al. 2005; Rudel, Schneider, and Uriarte 2010) has led to tentative optimism for recovering biodiversity prospects in the biome (Lira et al. 2012; de Rezende et al. 2015). However, it would be naïve to disregard current anthropogenic pressures: deforestation continues (Metzger 2009) and even within "developed" parts of southeast Brazil there have been reports of continuing natural resource extraction by rural dwellers (Bello et al. 2015; Cullen, Bodmer, and Valladares Pádua 2000). With conservationists recognizing the potential value in secondary forests and within human-modified rural landscapes, there is now a drive to better understand species responses under such dynamic conditions and identify the most effective management strategies (Gardner et al. 2009; Irwin et al. 2010; Tabarelli et al. 2005). Methods that can inform conservation and policy within such a heterogeneous and shifting context are much needed, and must do so by understanding current human-ecological interactions within these modified environments.

1.5 Objectives, methods and structure of this thesis

The main objective of this thesis was to develop and critically examine novel methods to address complex conservation problems in tropical forest contexts dominated by human actions. I use a diverse set of approaches to achieve this objective, including a remotely administered standardized questionnaire (Chapter 2), secondary data analysis of large crime databases and remote sensing data (Chapter 3), and fieldwork involving a structured survey and semi-structured interviews (Chapter 4). The research focuses on both terrestrial and aquatic vertebrates, encompassing a broad range of Neotropical wildlife.

Firstly, I assess the value of non-normative data sources to inform on the poorly known and spatially expansive issue of Amazonian wild-meat harvest and trade through the use of expert elicitation (Chapter 2) and regional law enforcement records (Chapter 3). Both of these approaches tackle the knowledge gap of these activities on a large spatial scale, but differ somewhat in scope.

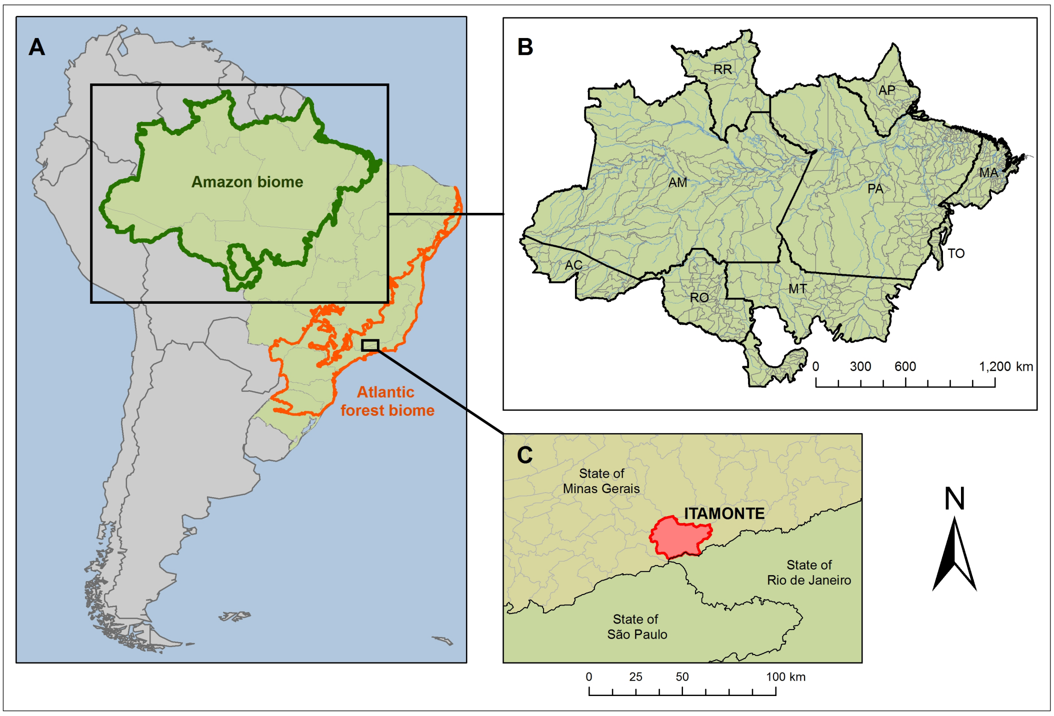

Figure 1 Map of study sites. (A) The location of the Amazon and Atlantic forest biomes in Brazil; (B) The nine states and municipal boundaries of the Amazon biome: Chapter 2 reports on the two largest states of Amazonas (AM) and Pará (PA), while Chapter 3 covered the entire biome, including Roraima (RR), Amapá (AP), Maranhão (MA), Tocantins (TO), Mato Grosso (MT), Acre (AC), Rondônia (RO); (C) The municipality of Itamonte, located within the state of Minas Gerais (Chapter 4).

Chapter 2 uses caiman as a model taxon to examine expert knowledge on specific harvest, trade and consumption patterns. Caiman offered an ideal focal taxon, following indications of significant trade for their meat (Da Silveira and Thorbjarnarson 1999), but limited empirical data across most of Amazonia. On a personal note, it allowed me to indulge my long-term research interest and affection for these animals after spending many years monitoring their populations in an Amazonian reserve in Peru. The method used in this chapter was a Likert-based questionnaire sent via email to relevant researchers and reserve managers in the two largest Brazilian Amazonian states, Amazonas and Pará.

The use of law enforcement records in Chapter 3 offers broader insights into the variable dynamics of multi-species harvest and trade systems, while also highlighting the important challenges of governance in vast tropical forests. I analysed staff allocation and reports of over 1300 wildlife crime enforcement events from Brazil's federal environmental agency, examining spatial patterns in relation to municipal geographic variables obtained from remote sensing data. The database I assembled covered the period January 2006 to October 2014, enabling me to examine temporal trends far beyond what would have been possible if restricted to the time available for fieldwork during a PhD.

Chapter 4 takes us to the Atlantic forest biome, exploring the value of Local Ecological Knowledge (LEK) to inform on species occurrence and trends within the context of a severely modified rural landscape. This research included *in situ* fieldwork, spending a month in the rural landscape of Itamonte, in the southeastern state of Minas Gerais. I conducted 63 interviews with local inhabitants using both a quantitative questionnaire on species status and trends, and an open semi-structured interview to elicit a rich social-cultural backdrop to landscape change and drivers of species responses.

All three data chapters of this thesis have been written for publication. At the time of submission Chapter 2 has been published in *Oryx: The international journal of conservation*, while Chapters 3 and 4 are in preparation for submission.

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EXPERT ELICITATION AS A METHOD FOR EXPLORING ILLEGAL HARVEST AND TRADE OF WILD-MEAT OVER LARGE SPATIAL SCALES

Photos: Caiman meat being salted and dried by rural Amazonians

Chapter 2 Expert elicitation as a method for exploring illegal harvest and trade of wild-meat over large spatial scales

2.1 Abstract

New evidence of commercialization and consumption of wild-meat in Amazonian cities has exposed an alarming yet poorly understood threat to Neotropical biodiversity. In response to the limitations of field sampling for large-scale surveys, I sought to develop a method of rapidly assessing wildlife harvest and trade in multiple areas using expert knowledge. Using caiman as a model taxon, I surveyed experts across the Brazilian Amazon. Expert responses to a Likert-style questionnaire suggest that caiman hunting, generally considered a localized rural activity, is in fact common and geographically widespread. Contrary to previous assumptions I found evidence that urban demand is partly driving the harvest, including via interstate trafficking. I highlight the need for further field validation of wild-meat trade and urban consumption patterns in Amazonia. I conclude that expert elicitation is a simple, cost-effective technique that can be a valuable precursor to inform and direct applied conservation research, especially where there are significant knowledge gaps and at large spatial scales.

Keywords: Amazonia, Brazil, caiman harvest, expert elicitation, wild-meat consumption, wild-meat trade

2.2. Introduction

The harvesting of wild-meat for human consumption is among the greatest threats to tropical wildlife and is a primary cause of population declines (Bennett et al. 2007). Aside from conservation concerns, unsustainable harvesting of wild-meat can also jeopardize livelihoods and food security (Nasi, Taber, & van Vliet, 2011). Characterising the dynamics of wild-meat harvesting and consumption is challenging because of interactions among biophysical variables (e.g. urban remoteness, species ecology) and social factors (e.g. poverty, cultural preferences; Brashares et al. 2011; Dupain et al. 2012). Despite its inherent complexity, harvesting of wild-meat is frequently dichotomized as a subsistence activity for the marginalized rural poor (de Merode, Homewood, & Cowlishaw, 2004) or a lucrative commercial endeavour supplying urban markets with a luxury heritage item (East et al. 2005). Legislation often reinforces this polarization by allowing the former but not the latter. The commercialization of wild-meat for urban markets is considered to be the primary driver of the bushmeat crisis in West and Central Africa (Bennett et al. 2007).

In the Neotropics, examination of wild-meat harvesting and consumption has centred on the rural subsistence paradigm (Peres, 2000). In Amazonia research has focused mainly on the determinants and ecological impacts of rural hunting practices (Peres & Nascimento, 2006). Regional estimates of bushmeat consumption have been extrapolated from rural village-level offtake studies (Fa, Peres, & Meeuwig, 2002; Peres, 2000) but these estimates ignore commercial trade and urban consumption, instead assuming that urban Amazonians do not eat forest wildlife (Nasi, Taber, & van Vliet, 2011). This assumption must now be re-examined, given evidence of high levels of commercialization and consumption of wild-meat in Amazonian cities (Parry, Barlow, & Pereira, 2014; van Vliet et al. 2014). With rapid growth of cities, aggregate urban consumption of wild-meat in Amazonia could be vast and increasing, yet the interacting dynamics of rural harvest, trade networks and urban markets remain under-studied.

Urban consumption of wildlife often requires long-distance trade networks connecting distal forests with metropolitan consumers. In Asia, for example, turtle meat is traded across international borders (Nijman, 2009) and significant quantities of African bushmeat ends up in illegal meat markets in Europe (Chaber et al. 2010). Even within national boundaries networks can span vast distances, and harvesters will travel further to meet demand as wildlife populations decline (Lindsey et al. 2013). Examining wildlife harvest and trade over large spatial scales is problematic for researchers because of the time and cost of fieldwork, and the difficulty of studying an often illegal and cryptic activity (Razafimanahaka et al. 2012). In Amazonia, where so little is known about the scale and nature of urban consumption of wild-meat, an important step is to synthesize present understanding and define key knowledge gaps to inform applied research and policy interventions. I use an expert elicitation approach to gauge knowledge, using caiman as a model taxon for understanding large-scale patterns of wildlife harvest, trade and consumption.

2.2.1. Caiman: a model taxon

There is a significant lack of knowledge about current levels of harvesting, trade and consumption of caiman in Amazonia. Harvest studies tend to focus on terrestrial mammals, as they are considered to be preferred by hunters and have long been studied by tropical ecologists (Bodmer, 1995; Peres & Nascimento, 2006). In contrast, caimans are considered to be a localized, secondary source of protein (Thorbjarnarson, 2010), and as such their extent and role in Amazonian diets is comparatively unknown. Two caiman species in particular are harvested and consumed in Amazonia: the black caiman *Melanosuchus niger* and the spectacled caiman *Caiman crocodilus*. They are the two largest crocodilian species in the region and both have a long history of human exploitation, having been commercially hunted for their skins to supply international demand for exotic leather throughout much of the 20th century. Sanctions on international trade facilitated the recovery of many populations, notably those of the black caiman, which were all but decimated by overharvesting, and both species are categorized as Least Concern on the IUCN Red List (Crocodile Specialist Group, 1996; Ross, 2000).

Based on a few localized rural harvest studies, consumption of caiman has been recorded in various localities across Amazonia (Ojasti, 1996); using extrapolated data Peres (2000) estimated annual harvest rates of caiman meat by the rural poor in the Brazilian Amazon to be 240–589 t (by 25,000–62,000 individuals). There is now some evidence that caiman meat is traded commercially and consumed in urban areas (Baía Jr, Guimarães, & Le Pendu, 2010; Parry et al., 2014); however, much of our knowledge remains anecdotal (Thorbjarnarson, 2010). Additionally, caiman meat may be sold fraudulently as a high-value prestige fish (Peres & Carkeek, 1993). Fraudulent meat substitution is a global concern (e.g. Europe's horsemeat scandal; Di Giuseppe et al. 2015) and an important conservation issue (von der Heyden et al. 2010). The use of caiman (together with river dolphins *Inia geoffrensis*) as fish bait is also a concern following increased international demand for the catfish *Calophysus macropterus* (Mintzer et al. 2013). My aim here was to enhance understanding of what is a potentially multifaceted and spatially extensive harvesting and trade system.

2.3 Methods

I used an expert survey approach to obtain information on patterns and drivers of harvesting and trade of caiman across the Brazilian Amazon. The increasing use of expert elicitation in conservation research and planning has been driven by the need to characterize dynamic systems, with limited resources (Martin et al., 2012). The method is an expedient approach for obtaining a regional synthesis of a politically invisible issue about which local experts may be aware (van Vliet et al. 2013).

I approached 122 experts to participate in the survey, based on relevant professional experience and/or current employment. I targeted people working in situ on caiman harvesting or management in the Brazilian Amazon, as well as other locally based individuals with current expert knowledge and professional experience of conservation and natural resource management. I identified potential respondents as follows: (1)

authors identified in relevant literature, (2) reserve managers (including of federal and state protected areas), and (3) researchers and analysts from academic institutions or federal government environmental institutions, such as the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA, responsible for enforcing environmental law outside protected areas) and the Chico Mendes Institute of Biodiversity Conservation (ICMBio, with a mandate for enforcement within federal protected areas). Contact was made via e-mail during September–November, 2013; e-mail addresses were obtained through known contacts, via author information provided with published articles, or from institutional websites. An initial e-mail explained the objectives of the research, sought consent to participate and included a questionnaire as an attachment. Respondents were asked to indicate other potential participants, thus expanding our contact database. In total I received 52 responses (whether accepting or declining the invitation to participate), with 24 completed questionnaires (20% of the 122 approached). This response rate is consistent with other expert surveys (23%, Lyytimäki & Hildén, 2011; 23%, Quijas et al. 2012).

The questionnaire (Appendix 1) consisted of 11 statements, which participants were asked to rate on a Likert scale (Likert, 1932), from 1 (completely disagree) to 5 (completely agree). Statements were related to relevant topics and knowledge gaps identified in a literature review, including the following: occurrence and drivers of caiman harvesting and trade, characteristics of harvesters, fraudulent meat substitution, and use of caiman meat as fish bait. I included a 2-year qualifier for questions that related to temporal trends (such as harvesting levels and caiman populations). This relatively short qualifier was chosen to focus on current harvesting (as opposed to historical trends) and also accounts for the rotation (change of location) of staff in government agency positions. Respondents also provided information on their employment role and the area or municipality from which their experience was derived. At the end of the questionnaire respondents were invited to explain the rationale for their responses and offer any other relevant information on hunting and trade of caiman in their region.

The 24 responses covered a wide area of the Brazilian Amazon, from the eastern city of Belém to the western border town of Tabatinga. The majority of responses were from experts based in Amazonas State (17 questionnaires, 71%). Of the remaining seven questionnaires five were from experts based in Pará State (21%). I also received one response each from the states of Mato Grosso and Mato Grosso do Sul; however, as these responses were from outside the Amazon biome I focus on Amazonas and Pará, which together cover 56% of the Brazilian Legal Amazon and are home to 48% of the region's human population (IBGE, 2010).

2.4 Results

The majority of experts (76%; (Figure 2) agreed (partially or completely) that caiman hunting was a common occurrence, extending across the Brazilian Amazon, from Santarém to Tabatinga on the Brazil–Colombia border (Figure 3). Whether caiman hunting was premeditated rather than opportunistic (e.g. by fishers) generated a more mixed response (Figure 2), even between proximate localities. Temporal trends in harvesting pressure were also unclear, with 68% of experts unable to affirm or refute that the occurrence of caiman hunting had increased in the previous 2 years. There was no apparent indication of a contemporary, widespread decline of caiman populations. Only one respondent considered local caiman populations to be decreasing, whereas 43% disagreed with this assertion and 52% were uncertain (Figure 2).

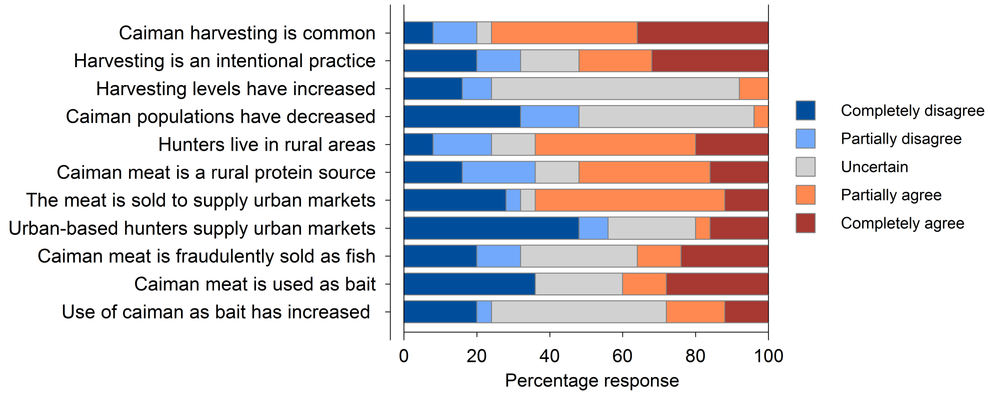

Figure 2 Expert responses on the scale and drivers of caiman harvesting and trade in the Brazilian Amazon. Respondents rated each statement on a five-point scale, based on their perceptions and experience within their locale.

Our results indicated that demand for caiman meat came from both urban and rural consumers. Urban demand for caiman meat was recognized by 64% of the experts (Figure 2), and harvesting caiman to supply urban markets was reportedly concentrated along the River Solimões–Amazonas and proximate sections of its tributaries (Figure 3d). Suggestive of distal source areas, a respondent in Belém [R20] stated that caiman meat was sold in urban markets (Figure 3d) and consumed in local rural areas (Figure 3c) but was not commonly harvested locally (Figure 3a). A respondent based in Amazonas [R1] elaborated on this potential long-distance trade:

> *"Ribeirinho [river-dwellers] hunters kill caiman indiscriminate of species or sex… 100% of the meat is salted and dried, to be sold to traders from the state of Pará. From there it is sold in urban markets around the capital of Belém."*

Another respondent based in Amazonas [R18] attributed this to different regional food preferences:

> *"I always hear talk that people who like caiman meat are from Pará, and that people from Amazonas do not like caiman meat."*

Whether hunters supplying the urban market were from rural or urban areas was unclear. More than half the experts disagreed that urban-based hunters were operating. However, 24% of respondents were uncertain and 16% completely agreed that urban hunters contribute significantly to the caiman meat market. Overall, there was stronger evidence that caiman were predominantly hunted by rural people (64%), and that harvesting was to supply rural people with animal protein (52%). The clearest evidence (i.e. strong agreement) of rural consumption came from around the major urban centres (Figure 3). In summary, expert opinion indicates that caiman are hunted mainly by rural people, providing a source of protein to rural communities close to larger cities, and a source of income when sold to urban markets, particularly along the main River Solimões–Amazonas.

There was uncertainty regarding whether caiman were hunted for fish bait or whether caiman meat was sold fraudulently as fish (Figure 2). The strongest evidence of fishers using caiman meat as bait was from along the River Solimões–Amazonas, from Santarém through to Tabatinga. There was disagreement or uncertainty in most other locations (Figure 3c). It was unclear whether this practice was increasing, as half of the respondents were uncertain of any temporal trend. Fraudulent selling of caiman meat as high-value fish appeared to be less widespread than using caiman as fish bait, with approximately equal numbers of experts expressing uncertainty (32%), agreeing that this happens in their area (36%), and disagreeing (32%). Strong agreement that such fraud occurs came from around the major cities of Manaus, Santarém and Belém (Figure 3e).

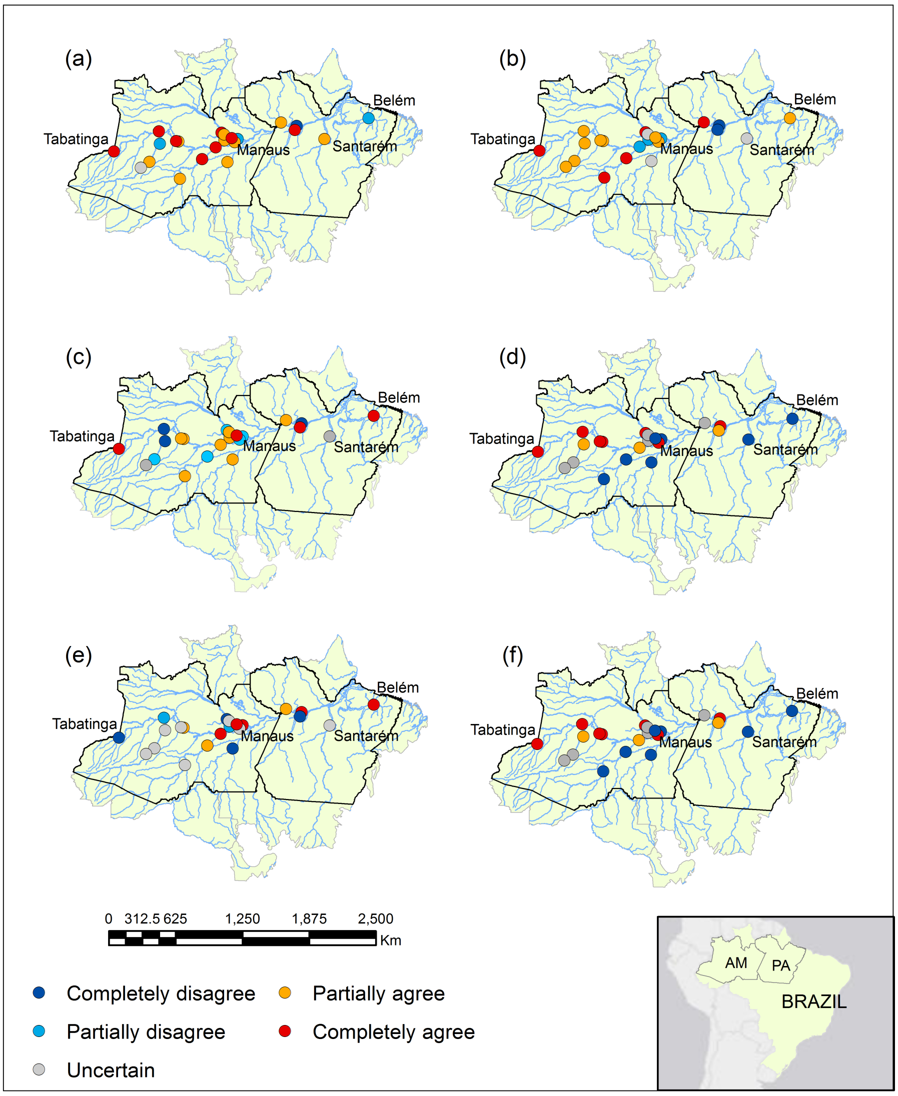

Figure 3 Spatial distribution of expert responses, on a Likert scale, to the following statements regarding the harvesting and trade of caiman in the Brazilian states of Amazonas (AM) and Pará (PA): (a) Caiman hunting is a common occurrence. (b) Caiman hunters live in rural areas. (c) Caiman meat is a source of protein in rural areas. (d) Caiman are hunted to supply urban markets. (e) Caiman meat is sold fraudulently as high-value fish. (f) Caiman are hunted for bait to capture piracatinga *Calophysus macropterus*.

2.5 Discussion

Our findings demonstrate that expert elicitation is an effective method for evaluating wildlife harvest over large spatial scales. The respondents offered novel insights into a complex harvest system in the Brazilian Amazon, including widespread harvesting, urban demand and long-distance trade, and the extent of lesser-known secondary drivers. I acknowledge the limitations of the use of expert surveys for generating knowledge from individual judgement, given potential bias and varying levels of expertise (Kynn, 2008). Nonetheless I believe expert elicitation is an appropriate, cost-effective approach to perform a rapid regional synthesis of a poorly known and complex issue. Viewed pragmatically, the information I obtained here is a valuable precursor to empirical data collection and I highlight the need for validating the expert knowledge underlying our findings (Keane, 2013; Kynn, 2008).

2.5.1 Key insights and interpretations

It was assumed that Amazonia had avoided a bushmeat crisis on the basis of urbanization and assertions that the only wild-meat eaten by city-dwellers was fish (Nasi et al. 2011). However, expert opinion indicates that killing caiman for meat, previously considered a localized rural activity (Peres, 2000), is in fact common and widespread across the Brazilian Amazon. I also found that urban demand is a significant driver of the harvest, with trade of caiman meat concentrated around large urban centres. Spatial patterns of expert responses and direct quotes support previous anecdotal evidence of long-distance trade in caiman meat from the state of Amazonas to Pará (Da Silveira & Thorbjarnarson, 1999). Long-distance trade suggests substantial demand that offsets higher costs for processing and transport, together with the presence of commodity chains and trade networks (Lindsey et al. 2013). It also implies that demand for caiman meat cannot be fulfilled by local sources and could therefore be a symptom of local overharvesting and depletion (Lindsey et al. 2013).

Market price data from a study in Pará showed that caiman meat is cheaper than fresh or dried beef, with prices comparable to chicken, shrimp and fish (Baía Jr et al. 2010). This implies that caiman meat in the Amazon does not fit the luxury good paradigm often documented in urban wild-meat consumption (Bennett, 2002). Instead, I infer that caiman meat provides a cheap alternative protein option for the urban poor, congruent with evidence of poverty-linked caiman consumption in two provincial cities (Parry et al. 2014). Regardless of access to domestic protein sources, poverty will drive demand for cheaper alternatives. With 70% of the rapidly growing population in the Brazilian Amazon living in cities, including millions living in urban poverty (IBGE, 2010), I predict demand for caiman meat will remain high (or increase). Furthermore, rural–urban migrants often remain dependent on rural livelihoods (Parry et al. 2014) and many Amazonian house- holds can be described as multi-sited, participating in rural–urban networks (Padoch et al. 2008). Such patterns have important implications for wild-meat consumption, including the persistence of rural food preferences in Amazonian cities (Padoch et al. 2008). Parry et al. (2014*)* found that urban households with strong rural linkages are more likely to eat not just caiman but a range of wildlife.

2.5.2 Knowledge gaps and research priorities

Areas of uncertainty in expert opinion highlighted important knowledge gaps, which are crucial to identifying research priorities. Quantifying the role of urban demand on caiman harvesting and trade is a key knowledge gap. Data on trade routes, market structure and the drivers of hunter and consumer behaviour are needed, following a commodity-chain approach (Allebone-Webb et al. 2011; Cowlishaw, Mendelson, & Rowcliffe, 2005). The African bushmeat crisis, defined as a critical conservation and development issue, has been studied extensively (Allebone-Webb et al. 2011; Lindsey et al., 2013; Milner-Gulland & Bennett, 2003), and consequently we should reflect on this body of research to identify appropriate strategies and anticipate challenges. What is clear is the inherent variability and complexity of wild-meat trade and consumption patterns (Brashares et al. 2011), which limits broad generalizations and often requires case-specific information from both social development and ecological perspectives.

The fraudulent substitution of dried caiman meat for the sought-after fish *Arapaima gigas* is perhaps the biggest unknown factor in this study. I found that this fraud occurs around the large urban centres where food demand is highest, and recommend that this practice be quantitatively assessed to draw conclusions on its extent and impact. As this is a global issue in food production systems, there has been significant advancement in food authentication techniques (Mafra, Ferreira, & Oliveira, 2008). DNA barcoding offers a relatively quick and inexpensive means of species identification and has been successfully used in bushmeat trade studies to confirm species misidentification (Minhós et al., 2013); it is a viable option for assessing caiman–fish substitution.

The use of caiman (and river dolphins) as catfish bait has already been recognized as a significant concern, and the implementation of a ***5***-year moratorium on *Calophysus macropterus* fishing in Brazil is intended to curtail this practice (MMA, 2014). However, environmental governance in Amazonia is often poor because of limited resources and low enforcement over such a large area (Parry et al. 2014). Continued assessment and monitoring is recommended.

2.6 Conclusion

The multidisciplinary nature of conservation means researchers must often utilise a diverse array of data sources and methods (Keane, 2013). There is increasing recognition of the role of alternative approaches in wildlife conservation and management; for example, the use of recall data for species consumption rates (Golden, Wrangham, & Brashares, 2013) and of local ecological knowledge for wildlife abundance and distribution trends (Anadón et al. 2009). Such methods are particularly apposite in resource-limited and spatially extensive tropical contexts (Parry et al. 2014). I used a simple expert elicitation method to gain insight into a complex and multifaceted harvesting system, and in doing so identified critical focal areas for further study, both thematically and spatially. Nonetheless I reiterate the importance of rigorous implementation of methods with regards to questionnaire design and interpretation of results (Martin et al. 2012). Expert opinion is not a like-for-like substitute for empirical research but is a complementary, cost-effective tool that can inform and direct more intensive data collection, especially when confronting complex dynamic systems across large spatial scales.

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***Appendix 1** Questionnaire on caiman harvest and trade activities sent to selected expert respondents (translated into English from the original Portuguese)*

Please email your responses to natalieswn@hotmail.co.uk

Confidentiality: Your responses to the following questions are absolutely confidential

Personal information:

Name		Region/municipality where you work	
Institution		Type of Conservation Unit (UC) (if applicable)	
Position		Time working in this location (Months and years)	

Based on your experience and knowledge about caiman hunting in this location, please mark the most appropriate response to the following statements (with X) :

	Completely disagree	Partially disagree	Uncertain	Partially agree	Completely agree
Illegal caiman hunting is a common occurrence in this municipality/UC.					
Hunters leave with a specific intention to hunt caiman.					
Over the last two years the occurrence of caiman hunting has increased.					
Over the last two years caiman populations have decreased in the region.					
Caiman hunters live primarily in rural areas.					
Hunted caiman are used as a source of protein in rural communities.					
Caiman are hunted in order to sell the meat in urban markets.					
Hunters residing in urban areas provide the majority of caiman meat in urban markets.					
Caiman meat is fraudulently sold as pirarucu (*Arapaima gigas*, or other fish species) in urban markets.					
Caiman are hunted to use as bait to capture piracatinga (*Calophysus macropterus*) in this location/UC.					

Please email your responses to natalieswn@hotmail.co.uk

Confidentiality: Your responses to the following questions are absolutely confidential

	Completely disagree	Partially disagree	Uncertain	Partially agree	Completely agree
The use of caiman as bait to capture piracatinga has increased in the last two years.					

General observations: Please provide below any further information or details about hunting and trade of caiman that you deem relevant.

(For example, your reasons for why you partially agree/disagree with any of the statements, an estimate of the number of caiman hunted per year in this location, a percentage of rural/urban households that have consumed caiman in the last 12 months, any plans to legalize caiman hunting in the region).

Thank you very much for your co-operation.

CHAPTER 3

POLICING THE AMAZON: UTILISING LAW ENFORCEMENT REPORTS TO CHARACTERISE AND MONITOR REGIONAL WILD-MEAT HARVEST AND TRADE IN TROPICAL FORESTS

Photo credits:
Top right: caiman meat arriving in Belem (g1.globo.com)
Bottom right: IBAMA agents with seized Amazonian turtles (Cristinalira.com)

Chapter 3 Policing the Amazon: Utilising law enforcement reports to characterise and monitor regional-scale wild-meat harvest and trade

3.1 Abstract

Characterising and monitoring illegal commercial trade and urban consumption of wild-meat is critical to implementing effective conservation policy. Methods for assessing wild-meat harvest (and its ecological impacts) tend to be site-specific and costly, thus adequate regional data is scarce in the forested tropics. I explore the potential utility of federal enforcement data to provide valuable regional insights into wild-meat harvest and trade in the Brazilian Amazon. I examined spatial and temporal patterns of institutional capacity and enforcement events against illegal wildlife harvest and trade in 549 municipalities. Using staff allocation as an indicator, I found evidence of overall limited institutional capacity that prioritizes municipalities with on-going deforestation, at the apparent cost of tackling wildlife crime in forest-rich areas. This results in very low detection of wildlife harvest and trade activities (averaging less than one enforcement event per year in 80% of municipalities), particularly in smaller towns far from deforestation frontiers. The results suggest that the effect of enforcement effort on crime-detection will vary depending on the scale and dynamics of the activity, with increased detection of large-scale commercial trade in larger cities. The data offers important conservation information on species-level harvest and trade. Over an eight-year period, environmental agents confiscated 46,801 game animals during enforcement events, equivalent to an estimated 269,627 kg in biomass. There was evidence of a substantial basin-wide commercial trade in turtles (*Podocnemis* spp), and agents apprehended 14,017 individuals with an estimated total value of $228,000 USD. I

conclude that, where and when available, the analysis of public environmental enforcement reports offers important new insights into wild-meat harvest and trade patterns over large spatial and temporal scales. However, such data must be used cautiously and pragmatically, considering the inherent biases that stem from the challenging reality of governance in tropical forest contexts.

Keywords: Amazonia, Brazil, environmental enforcement, tropical forests; illegal wildlife harvest and trade; enforcement reports

3.2 Introduction

Protecting tropical forests and their biodiversity is a global conservation priority (Gardner et al. 2009). Successful conservation strategies rely on characterising and understanding trends and extent of threats and disturbance through detection and monitoring (Assunção, Gandour, and Rocha 2013; Gavin, Solomon, and Blank 2010; Peres, Barlow, and Laurance 2006). Deforestation and forest degradation have generally taken centre stage as the primary threat to tropical forests, driven by agricultural expansion, urbanization and natural resource extraction (Malhi et al. 2014; Nepstad et al. 2014). Together with huge losses in biodiversity and ecosystem integrity, international concern has intensified within the last quarter of a century following greater understanding of the linkages between forest disturbance and global and regional climatic change (Shukla, Nobre, and Sellers 1990). Accurate detection and monitoring of deforestation has allowed scientists and policy-makers to quantify the scale of the issue, model potential impacts and consequently devise effective mitigation strategies (Hargrave and Kis-Katos 2012; Soares-Filho et al. 2006). Recent advancements in the availability and analysis of remotely sensed data have proved indispensable for monitoring rates of deforestation (Hansen et al. 2013), and have also been utilised to improve the efficacy of enforcement efforts, providing real-time detection and targeted enforcement of illegal forest clearing activities (Assunção, Gandour, and Rocha 2013).

However, anthropogenic threats to tropical forests are diverse and multifaceted, manifesting in different ways and at different scales (Peres, Barlow, and Laurance 2006). Harvesting of wildlife for consumption is considered one of the greatest and pervasive threats to tropical wildlife, being described as a pan-tropical wild-meat or "bushmeat" crisis (Milner-Gulland and Bennett 2003). Addressing wildlife overharvesting is not only a question of conservation; recent studies now show that the cascading effects of defaunation through overhunting also have the potential to erode globally-relevant ecosystem services such as a forest's ability to store carbon (Bello et al. 2015; Osuri et al. 2016; Peres et al. 2015). Moreover, hunting itself is strongly rooted in socio-economics and development (Bennett et al. 2007), and the unsustainable harvesting of wildlife jeopardizes livelihoods and food security, and any management or policy must consider the needs of people who depend on wildlife as a source of income or protein (Milner-Gulland and Bennett 2003).

Wildlife harvest is often dichotomized as either a subsistence activity for the marginalized rural poor (de Merode, Homewood, and Cowlishaw 2004) or a lucrative commercial endeavour supplying urban markets (East et al. 2005), generally reinforced by legislation that allows the former and criminalises the latter. Illegal commercial trade of wild-meat in West and Central Africa is thought to be the main driver of the bushmeat crisis in the region (Fa et al. 2014). Research examining these activities has highlighted the complex dynamics of harvest and trade, from small-scale local markets to complex commercial networks that span vast distances and even cross national borders, driven by social, cultural and economic factors (Brashares et al. 2011; Dupain et al. 2012). This inherent complexity and clandestine nature means that characterising and monitoring illegal wild-meat activities is particularly challenging, and furthermore must do so without the use of satellite technology that fails to detect these more subtle forest disturbances occurring beneath the canopy (Peres, Barlow, and Laurance 2006). Consequently, effective policy frameworks for sustainable management and mitigation of wild-meat activities have remained somewhat elusive and lack baseline data, especially at national and regional scales (van Vliet et al. 2015b).

In Amazonia, the world's largest tropical forest, the nature and scale of wildlife harvest for urban consumption and commercial trade has received less attention than in African and Asian forests. Nonetheless, recent studies suggest that illegal commercial trade is significant (Baía Jr, Guimarães, and Le Pendu 2010; van Vliet et al. 2014, 2017), and there is evidence of high levels of consumption of wild-meat in urban towns (Parry, Barlow, and Pereira 2014). However, serious knowledge gaps remain. Research that has shaped our current understanding has utilised various approaches, focusing on different aspects of the issue. For example, traditional *in situ* ecological studies have used long-term monitoring to examine trends and abundance of target species, providing insights into the ecological impacts of hunting pressure (Peres and Lake 2003). Yet such methods are often costly in terms of time and resources (Sutherland 2006), and more importantly they lack information on trade and consumer dynamics and the profiles of actors. This information is more difficult to obtain due to the sensitive and clandestine nature of illegal activities (Conteh, Gavin, and Solomon 2014; Nuno and St. John 2015; Regueira and Bernard 2012) and consequently methods can be susceptible to large or unknown standard errors (Gavin, Solomon, and Blank 2010).

A few studies in Amazonia have utilised methods such as direct questioning of hunters, traders or consumers (Parry, Barlow, and Pereira 2014; van Vliet et al. 2014; van Vliet et al. 2015a), or market studies (Baía Jr, Guimarães, and Le Pendu 2010; van Vliet et al. 2017) in order to obtain important information on species targeted and/or quantities of take or consumption. Most of these methods are restricted to local-scale case study research and, in the case of direct questioning regarding sensitive illegal activities, often subject to underreporting of levels (Gavin, Solomon, and Blank 2010; Nuno and St. John 2015). Regional baseline data of wild-meat activities in the Neotropics – outside of the rural subsistence paradigm – remains extremely limited. Extrapolating case-study data to regional scale can be misleading due to the heterogeneous nature of wild-meat dynamics (Brashares et al. 2011; Morsello et al. 2015), risking under- or overestimating the extent and impacts on biodiversity. Considering the many potential social and ecological issues related to wild-meat harvest in Amazonia, we urgently need an approach that can rapidly assess patterns of wild-meat over time and across broad spatial scales.

Up to now, estimates of faunal depletion and overexploitation over large spatial scales have been reliant on spatial interpolation based on high-replication of market surveys (e.g. Fa et al. 2015 in West Africa). However, this approach is limited in its ability to track changes in hunting offtake. In the Amazon a recent study inferred the impacts of urban demand for wild-meat by assessing relative wildlife depletion in relation to geographic factors, utilising local ecological knowledge (LEK) of rural hunters (Parry and Peres 2015). Yet this snapshot study does not provide information on change over time or link harvest patterns to urban trade. In this study I evaluate the extent to which the use of federal law enforcement records in the Brazilian Amazon provides a cost-effective method to obtain valuable large-scale insights into wild-meat harvest and trade patterns. Data on enforcement activities, such as park patrol encounters with poachers, have been used as a monitoring tool to measure the effectiveness of conservation strategies (Jachmann 2008), explore spatial patterns of illegal activity (Holmern, Muya, and Røskaft 2007), and identify targeted resources and estimate quantities of take (Davis et al. 2004). However, the method is often used at local-scales and has not been critically examined for its potential to inform on wild-meat harvest and trade on a regional scale as large as the Brazilian Amazon, which covers an area of more than 4.5 million km^2.

Brazil has often been at the forefront of global concerns over deforestation, biodiversity loss and climate change. Responding to international pressure, Brazil has been described as an environmental leader after expanding its protected area network and successfully reducing deforestation rates by 70% in the late 2000s (Ferreira et al. 2014; Nepstad et al. 2014). Although this unprecedented success in curbing deforestation has been attributed to a myriad of factors, including collaborative interventions in soy and beef supply chains and the expansion of protected area networks (Nepstad et al. 2014), research has demonstrated that accurate detection of forest-clearance and effective enforcement by the federal environmental protection agency (*Instituto Nacional do Meio Ambiente e dos Recursos Naturais Renováveis*, herein IBAMA) played a critical role (Assunção, Gandour, and Rocha 2013; Börner et al. 2015). More recently however, a political and economic crisis has led to increased development pressures and shifts in legislation that allow for mining and hydropower activities in the Amazon (Ferreira et al. 2014), and rates of forest loss have increased by nearly 36% since 2012 (Tollefson 2016). Until 2011[1] IBAMA was also responsible for enforcing all wildlife laws, including any illegal harvest and trade of wildlife for consumption that occurs outside of the rural subsistence paradigm (Presidência da República do Brasil, 1998). IBAMA's centralized enforcement records could therefore prove an important data source to shed further light on regional patterns of urban consumption and commercial trade and, further, offer a way of monitoring trends over time. Nonetheless, although IBAMA's successful enforcement of forest law is well documented, it remains unknown how or if the agency

[1] In 2011 responsibility for enforcing wildlife crime laws was transferred to state-level by Federal Law 140 (Presidência da República do Brasil, 2011). IBAMA continues to assist state police in arrests and seizures of wildlife crime, yet it is unclear to what extent this is of an official

is responding to this type of wildlife crime and, thus, whether the use of enforcement records is a reliable method to obtain representative information of wild-meat patterns.

I address these knowledge gaps related to understanding wildlife crime in large tropical regions by carrying out a critical assessment of institutional capacity and enforcement patterns. I do this in order to help identify bias in enforcement records and to facilitate interpretation of the data therein. The following outlines my three research questions and corresponding hypotheses:

1. *What is the institutional capacity (offices and agents) of the IBAMA agency to enforce wildlife law across the Brazilian Amazon?*

Using IBAMA staff allocation as an indicator I hypothesize that enforcement capacity is higher in: (a) areas of high forest cover; (b) areas with high deforestation rates; (c) larger towns and cities; (d) municipalities with international frontiers; (e) municipalities with high riverine water coverage (see Table 1 for justifications of selected variables).

2. *What can the spatial and temporal patterns of detected wild-meat crime reveal about the federal government's enforcement strategy?*

I used IBAMA's records of violations that relate to harvest and trade of game species to explore rates of detection and spatial and temporal variation. Here I hypothesized that, due to the difficulties in detecting subtle and cryptic activities in a vast forested area (Peres, Barlow, and Laurance 2006), overall enforcement of wild-meat crime will be low relative to the rates of harvest and consumption activities indicated in recent studies. I also hypothesized that fine-scale spatial patterns of enforcement events will reveal differences in detection due to accessibility and inadequate enforcement capacity (Contreras-Hermosilla 2002). Moreover, I expected that decentralization of enforcement responsibility and IBAMA office closures has led to a decline in enforcement efforts.

3. *What is the conservation value of enforcement records for revealing novel insights on wildlife harvest and trade patterns in tropical forest regions?*

I used data on the species and quantities of each to reveal important conservation-relevant information on the scale and dynamics of species harvest and trade. I then reflected on how both institutional capacity (Q1) and on-the-ground enforcement (Q2) likely impact any interpretations of this data, and on the value of enforcement records as a monitoring tool for wild-meat activities in the Brazilian Amazon.

3.3 Methods

The analyses in the following sections cover the Amazon Biome within Brazil, which includes 549 municipalities across nine states. Under Brazil's law on access to information (Presidência da República do Brasil, 2011) I obtained data on IBAMA staff allocation via the federal government's Transparency Portal. This is an open-access web portal, mainly designed to provide access to data on federal expenditures (www.transparencia.gov.br). I use the number and distribution of IBAMA staff as an indicator of institutional capacity and priorities (UNCTAD secretariat 2013).

Second, I submitted a request via IBAMA's citizen information system for all notifications of violations against wildlife recorded in the nine states that fall within the Legal Amazon. The citizen information system returned a total of 4617 records of violations against wildlife between January 2006 and October 2014 within the Legal Brazilian Amazon. The database was then filtered and processed to include only municipalities within the Amazon biome, which meant removing municipalities in the states of Mato Grosso and Tocantins that extend south into the *cerrado* biome. I considered only violations that involved illegal harvesting, possession, transportation or selling of known game species. I reviewed and cleaned the database, identifying and correcting (where possible) discrepancies including duplicate entries, spelling errors and inconsistency in the reporting of species names, from local name to genus. Our final sample consisted of 1,327 game species-specific enforcement events, 406 of which included geographic coordinates taken during the enforcement operation. I extracted, where possible, the number of individuals of each species recorded in the enforcement event, and estimated total biomass per event based on relevant literature (see Appendix 2).

Those records with spatial coordinates were used as a sub-sample to further explore the point in the commodity chain (e.g. extraction, transit or market) at which environmental agents detect and enforce wildlife crime. I used ArcGIS 10.2.2 (ArcMapTM, version 10.0) to map enforcement events over land cover data obtained from the Brazilian government's TerraClass (EMBRAPA & INPE 2011). The TerraClass project further classifies land mapped by the PRODES deforestation monitoring system into twelve land-use categories, including primary and secondary forest, pasture and urban areas (de Almeida et al. 2016). I also included river and road layers derived from Landsat images (IBGE, 2010; Imazon 2010, respectively).

Statistical Analyses

Using a negative-binomial distribution GLM I modelled the number of IBAMA staff allocated per municipality (as reported in June 2013[2]) against municipal-level parameters relevant for testing our hypotheses (Table 1). State capitals were excluded from analyses due to centralization of staff within state offices in order to focus on variation in non-metropolitan enforcement capacity across the region. From the violation records I estimated the trend in rate of recorded enforcement events over time

[2] The information provided by the Transparency Portal on staff allocation within the federal government is offered only as monthly reports from December 2012 onwards. I selected June 2013 as an example of recent staff numbers, mid-way through the year to minimize changes in staff turnover.

based on a cubic spline within a Poisson regression model. All statistical and descriptive analyses were carried out using R version 3.2.2 (R Development Core Team 2013).

Table 1: Municipal-scale data sources for factors used to assess allocation of federal environmental enforcement staff (IBAMA)

Data/ variable	**Metric/ resolution**	**Source**	**Hypothesis and justification**
IBAMA staff allocation	No. of staff	Transparency portal	Response variable: Human resources as an indicator of institutional capacity and priority setting
Forest cover	Km^2	INPE-PRODES (2013)	Increased institutional capacity in areas with higher forest cover due to conservation and biodiversity value
Deforestation rate	Km^2	INPE-PRODES (rate calculated from 2005-2013 forest loss data)	Increased institutional capacity in areas at risk of deforestation due to the conservation priority of curbing forest loss
Urban population	Municipal level	IBGE census 2010	Higher staff presence in larger towns and cities as they are more accessible than more remote provincial urban centres
Water coverage	Km^2	INPE-PRODES (2013)	Increased institutional capacity in municipalities with high water coverage due to the economic and livelihoods importance of fisheries in the Amazon region (Almeida et al. 2004)
International border?	Municipal Binomial (Y/N)		Increased staff presence due to perceived risk of international trade and border control

3.4 Results

What is the institutional capacity of the IBAMA agency to enforce environmental law across the Brazilian Amazon?

According to public records, in July 2013 a total of 1238 IBAMA employees were based across nine states within the Amazon biome. Employees were based within only 62 of the 549 municipalities, showing that 89% of Amazonian municipalities do not have a permanent IBAMA presence. Unsurprisingly the majority of staff (60%) worked in the seven state capitals that fall within the biome.

IBAMA staff allocation was higher in municipalities with higher absolute forest cover ($z = 2.23$, $p =< 0.05$) and deforestation rate ($z = 2.43$, $p =< 0.05$), implying that enforcement capacity is geared towards prioritising areas with higher risk of illegal forest clearance. Larger urban population was also a significant positive predictor of staff presence ($z = 7.19$, $p = < 0.05$), and there was little permanent enforcement presence in small towns. Municipalities with international borders had marginally significant increased staff presence ($z = 2.82$, $p = < 0.05$) (Figure 4).

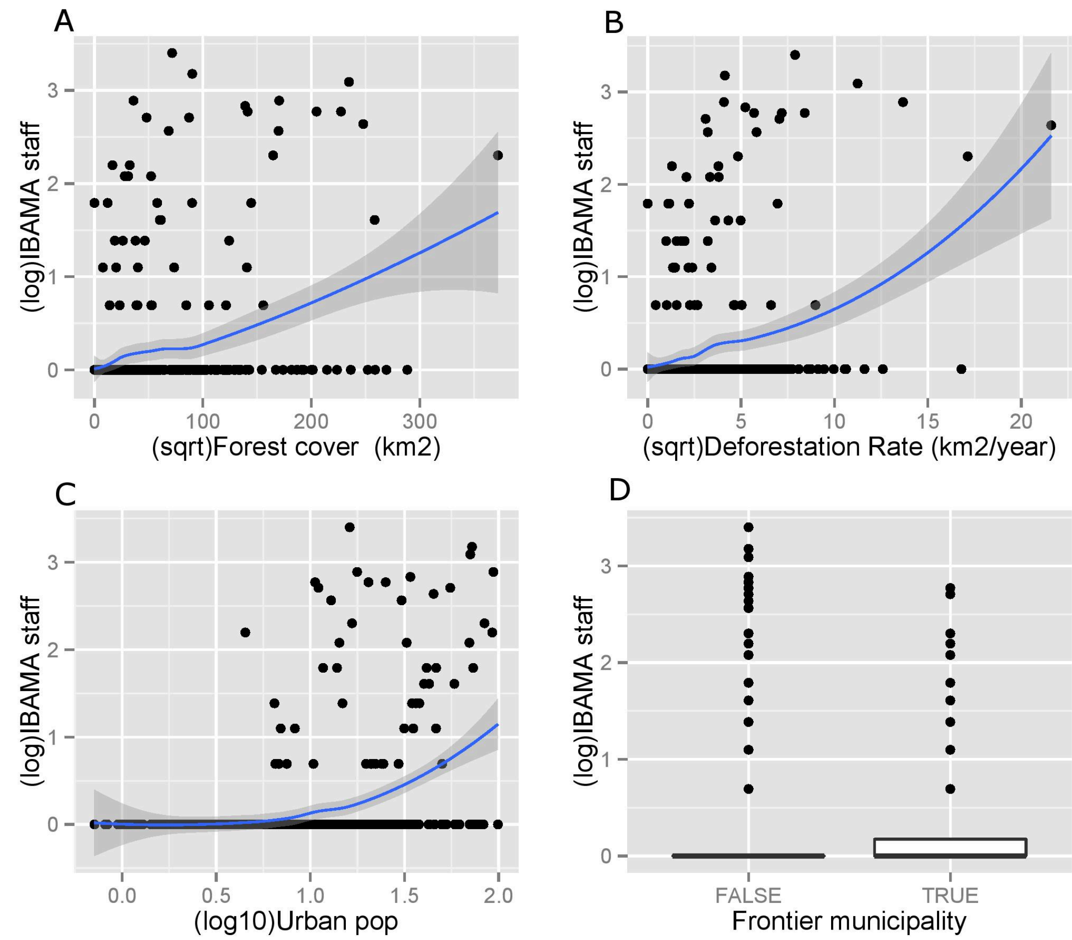

Figure 4: Significant predictors of IBAMA staff numbers within a negative-binomial regression model. A) Forest cover (km2) (PRODES); B) Rate of deforestation based on the previous eight years of forest loss (PRODES); C) Urban population (IGBE census 2010); D) Municipalities with international borders. Data points represent one municipality (n= 549). The regression line for continuous variables (A-C) is shown in blue, with shaded area showing standard error.

What are the spatial and temporal patterns of detection and enforcement of wild-meat activities in the region?

221 municipalities had at least one record of an enforcement event, whereas 328 municipalities (60%) had no record of any wild-meat harvest or trade detected by federal enforcement officials over an almost nine-year period. In terms of area, the municipalities where illegal wild-meat activities were detected at least once cover over 70% of the region (Figure 5). Nonetheless, in general it appears that detection rates by IBAMA are low; 90 municipalities (40% of those with enforcement records) were represented by a single reported enforcement event, and more than 80% of the 221

municipalities where illegal wild-meat activities were detected averaged less than one enforcement report per year. I estimated the trend in the rate of enforcement records over time and found a significant declining trend, decreasing by over 50% after peaking in 2008 (z = -6.362, P <0.001) (Figure 6).

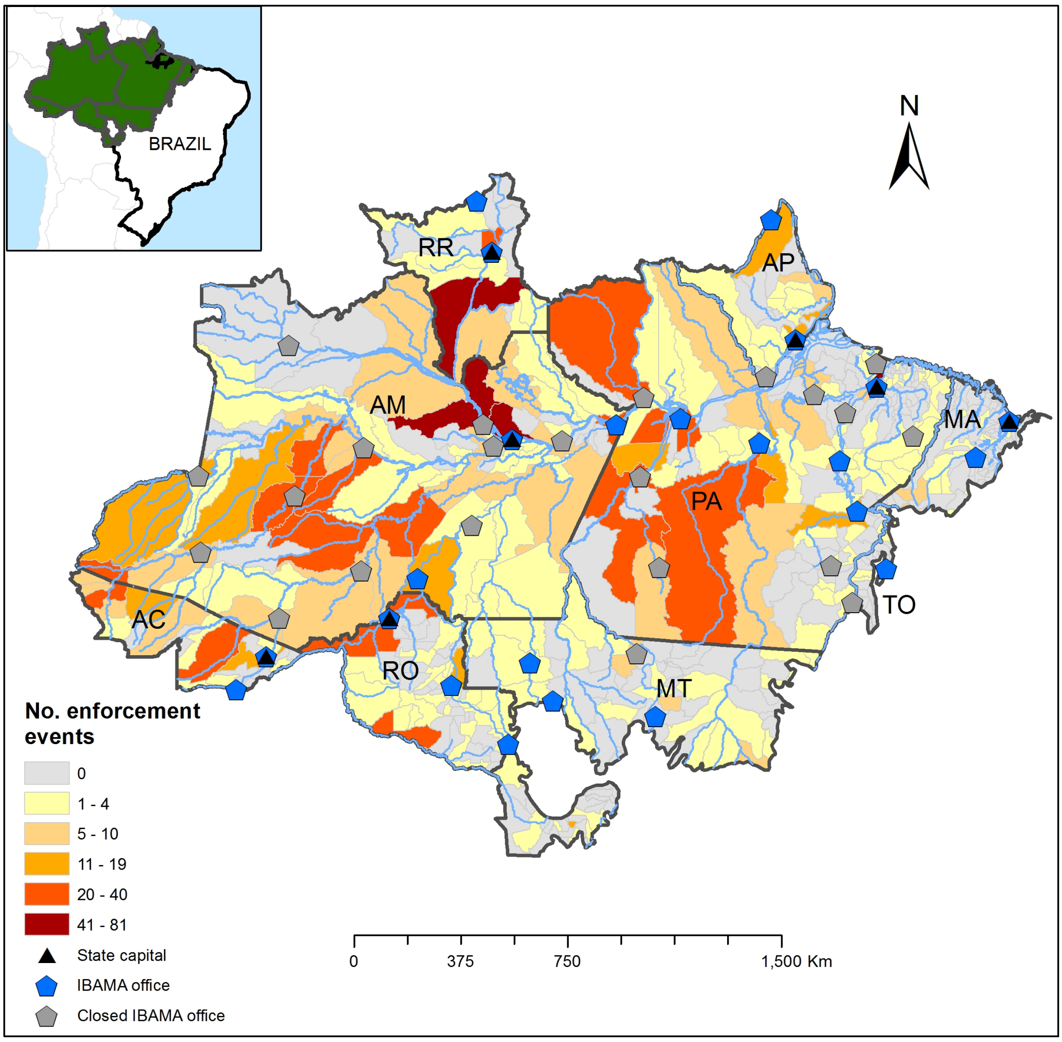

Figure 5: Total number of game-based crime enforcement events recorded in each municipality across the Brazilian Amazon biome over an almost nine year period between 2006 and 2014. The closed IBAMA offices were shut down between 2007 and 2012, following administrative reforms that were decided in early 2007.

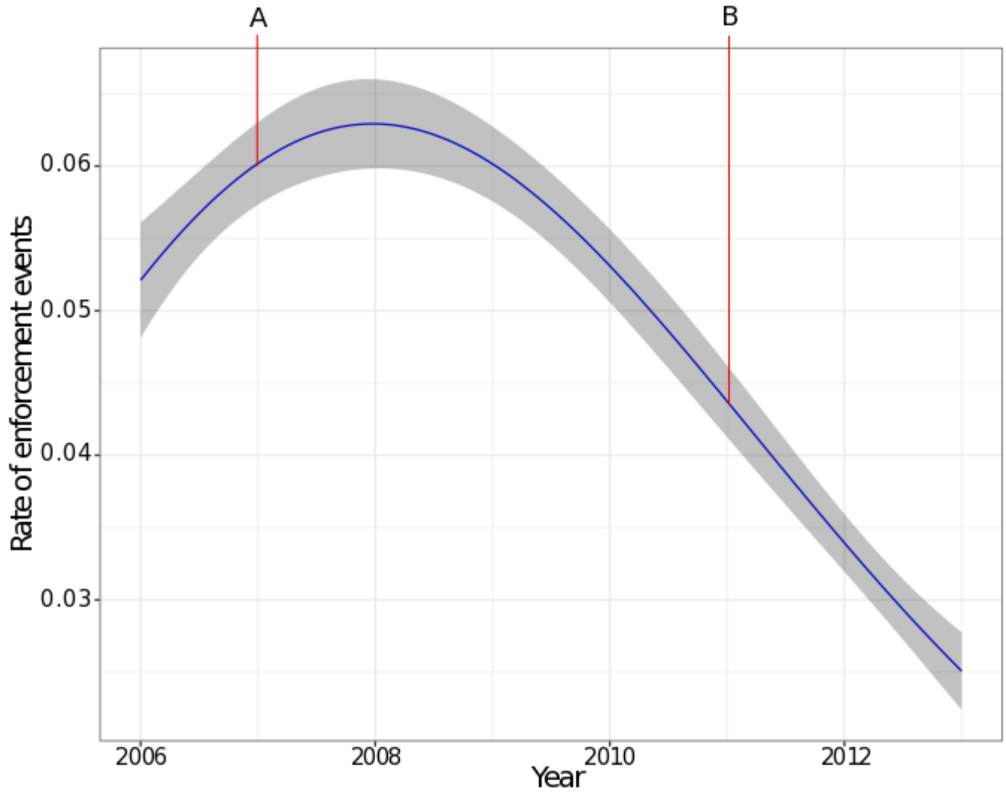

Figure 6: The mean annual rate of enforcement events (blue line) recorded between 2006 and 2013 across all municipalities (n= 549). Shaded grey area shows 95% confidence intervals. Red line indicates the following key events: (A) the commencement of IBAMA administrative restructuring and the closure of selected IBAMA offices, and (B) decentralization of responsibility for enforcing wildlife crime from federal to state and municipal levels.

Where do enforcement agents detect wild-meat activities? Fine-scale spatial patterns

Mapping the enforcement records with spatial coordinates I found that nearly half of events were recorded along potential transport routes, either on rivers (39%) or on roads (8%) (Figure 7). A significant proportion (31%) of game-based enforcement events were recorded in urban areas, indicative of the prevalence of urban consumption and trade, and further, evidence that enforcement officials are aware and active in enforcing against such activities (Figure 7). Comparatively fewer enforcement events were recorded within forested areas, indicating that enforcement takes place when the harvested game is in transit rather than at the point of extraction (at least for terrestrial species).

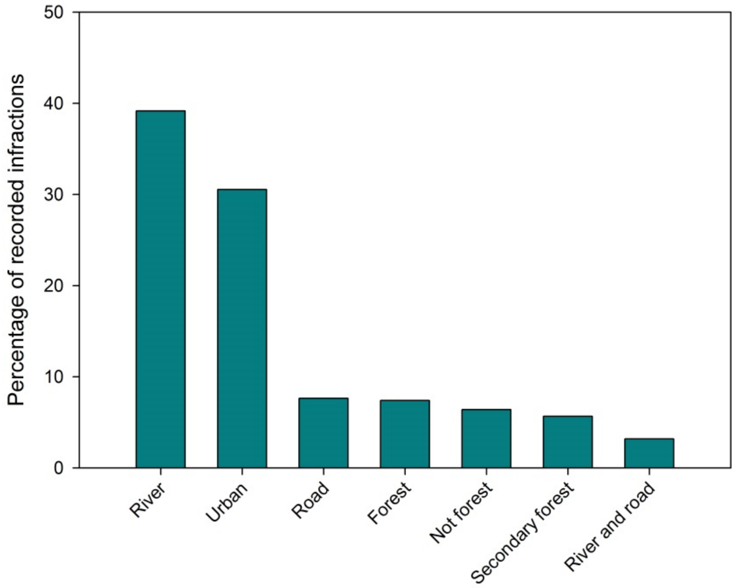

Figure 7: Locations of game-based arrests and seizures recorded by IBAMA with GPS information (n=406) based on TerraClass land cover data. The category "Not forest" includes locations categorized as "Pasture" or " Mosaic of occupations" (which is defined as undefined or heterogeneous land uses) (INPE, 2010).

Species composition of IBAMA's enforcement reports

I found records of 63 named vertebrate genera from 16 families within IBAMA's enforcement reports. Of the 1327 records, 975 involved reptiles (74%), 737 reported mammals (56 %), and 109 involved birds (8%). Seventeen species are currently found on Brazil's Red List, and a further four are listed as Near Threatened (ICMBio 2014). Three of the four Near Threatened species belonged to the river turtle genus *Podocnemis*, which were by far the most prevalent species recorded in the data; whether in terms of number of enforcement events (877; 66%), number of individuals recorded (15,195; 85% of total) or total estimated biomass (212,990 kg; 80% of total) (Table 2).

Terrestrial game mammals were also frequently recorded; notably *Cuniculus paca* (157 records) and the two peccary species *Tayassu pecari* and *Pecari tajacu* (139 records). High numbers of individuals seized in single events offer evidence of commercial trade; this occurred not only with *Podocnemis* species (for example, in one event 2,426 *P. expansa* were seized), but also caiman (total of 4,600 kg of *Melanosuchus niger* meat) and the whistling duck *Dendrocygna spp.* (579 individuals seized in just two enforcement events) (Table 2).

Table 2: The species composition documented in IBAMA's enforcement records between January 2006 and October 2014. I present the number of events for each species, the total number of individuals (ind.) seized during events, their corresponding estimated biomass, and the calculated average per event.

Taxa/Species	Number of recorded events	Total number ind.	Est. Biomass (kg)	Mean ind. per event	Mean biomass per event (kg)
Testudines	**921**	**43830**	**214635**	**48**	**233**
*Podocnemis unifilis**	239	2566	17705	11	74
P. unifilis - eggs	57	21236	478	373	8
*Podocnemis expansa**	279	7237	177307	26	636
P. expansa - eggs	21	3114	125	148	6
*Podocnemis sextuberculata**	82	3982	9159	49	112
P. sextuberculata - eggs	6	198	4	33	1
Podecnemis erythrocephala	23	232	264	10	11
P. erythrocephala - eggs	8	2171	36	271	5
Chelonoidis spp.	220	1130	6498	5	30
Other chelonians	63	280	2451	4	39
Unspecified chelonian	15	70	568	5	38
Unspecified chelonian - eggs	9	1614	41	179	5
Caimaninae	**99**	**782**	**23678**	**8**	**239**
Melanosuchus niger	12	229	9366	19	781
Caiman crocodilus	25	86	2021	3	81

Caiman yacare	4	308	7238	77	1810
Caiman latirostris	5	6	141	1	28
Paleosuchus trigonatus	1	1	17	1	17
Other/unspecified caiman	52	152	4894	3	94
Ungulates	**273**	**415**	**21757**	**2**	**80**
*Tapirus spp.***	53	42	10017	1	189
Mazama spp	13	14	483	1	37
*Ozotoceros bezoarticus***	3	5	175	2	58
Unspecified deer	65	82	2342	1	36
Tayassu pecari	58	163	5705	3	98
Pecari tajacu	32	61	1586	2	50
Unspecified peccary	49	48	1449	1	30
Rodents	**228**	**468**	**6310**	**2**	**28**
Cuniculus paca	157	377	3582	2	23
Dasyprocta spp	29	40	178	1	6
Hydrochoerus hydrochaeris	42	51	2550	1	61
Edentates	**100**	**149**	**1517**	**1**	**15**
*Myrmecophaga tridactyla***	1	1	31	1	31
Tamandua tetradactyla	1	2	12	2	12
*Bradypodidae/Megalonychidae***	7	9	35	1	5
Dasypus spp.	14	21	152	2	11
*Priodontes maximus***	3	3	90	1	30
Cabassous unicinctus	3	5	16	2	5
Euphractus sexcinctus	5	5	24	1	5
*Tolypeutes tricinctus***	1	1	1	1	1
Unspecified armadillo	65	102	1155	2	18
Primates	**127**	**164**	**806**	**1**	**6**
*Alouatta spp***	24	33	243	1	10
*Ateles spp***	14	25	204	2	15
*Lagothrix spp***	17	20	138	1	8
*Cebus spp***	41	51	158	1	4

*Saimiri spp***	7	10	9	1	1
*Saguinus spp***	7	7	3	1	0
Aotus spp	3	4	4	1	1
Pithecia spp	3	4	9	1	3
Unspecified primate	11	10	38	1	3
Game birds	**109**	**993**	**924**	**9**	**8**
*Crax spp***	2	2	5	1	3
Mitu tuberosum	3	3	8	1	3
Mitu mitu	2	3	9	2	4
Unspecified currasow	29	42	115	1	4
*Aburria jacutinga***	3	6	8	2	3
*Penelope superciliaris***	1	3	3	3	3
Unspecified guan	8	19	20	2	3
Crypturellus spp.	4	10	5	3	1
Tinamus solitarius	2	3	4	2	2
Tinamus guttatus	1	4	3	4	3
Ramphastidae spp	8	8	4	1	1
Jabiru mycteria	1	2	12	2	12
Cairina moschata	12	50	123	4	10
Dendrocygna spp	7	579	435	83	62
Callonetta leucophrys	2	27	10	14	5
Anas spp	10	125	54	13	5
Netta peposaca	1	26	26	26	26
*Mergus octosetaceus***	3	29	29	10	10
Unspecified duck	10	52	52	5	5
TOTALS	**1327**	**46801**	**269627**		

* *Near Threatened*
** *Vulnerable* (ICMBio 2014)

3.5 Discussion

By examining patterns of institutional capacity and enforcement events this study sheds important light on illegal harvest and trade of wildlife in Amazonia, whilst also revealing some of the challenges of governance of a hard-to-detect and spatially expansive activity. I find that the data on species and quantities of seized individuals contained within enforcement reports offers invaluable conservation-relevant information on species harvest and trade dynamics; information that may be difficult and/or costly to obtain through other methods. I found strong evidence of commercial and small-scale trade of multiple species, confirming that urban consumption is spatially expansive in Amazonia (Parry, Barlow, and Pereira 2014; Parry and Peres 2015). Yet the analyses on institutional capacity and enforcement rates indicate that IBAMA has struggled to effectively enforce against illegal harvest and trade activities (when compared to available studies on commercial trade and urban consumption rates in the region, e.g. Baía Jr, Guimarães, and Le Pendu 2010; Parry, Barlow, and Pereira 2014; van Vliet et al. 2015a), and the situation may be worsening with administrative reforms and the decentralization of governance. Without knowledge of the extent and drivers of illegal activities, it is not possible to design incentive programs to reward compliance or effectively punish violations (Yu, Levi, and Shepard 2010). I therefore outline the important novel insights on wildlife harvest and trade, interpreting these findings within the context of IBAMA's limited enforcement capacity and spatial patterns of enforcement events. I discuss future projections of governance in Amazonia and the implications for appropriate conservation strategies for wildlife harvest and trade in tropical regions.

3.5.1 Challenges of law enforcement in tropical forests and implications for data interpretations

The limited spatial coverage of IBAMA offices and staff outside of state capitals and large urban areas is symptomatic of the pervasive obstacles many countries face when protecting tropical forests (Bennett et al. 2007; Sánchez-Mercado et al. 2016). Weak institutional capacity due to limited resources is consistently cited as a barrier to effective environmental regulation, particularly in developing nations where poverty alleviation and development are often prioritised (Lo, Fryxell, and Wong 2006; Robinson, Kumar, and Albers 2010). The number and distribution of IBAMA staff suggest that inadequate human resources greatly restrict enforcement capacity in the region, as the majority of municipalities are left without the presence of any permanent staff. There is evidence of agent mobility that facilitates some enforcement within these municipalities; however, with only 1238 staff responsible for approximately 4.5 million km^2 of tropical forest there will inevitably be gaps in enforcement coverage. Governing institutions have no choice but to set priorities and allocate resources to best achieve their directives, targeting areas where illegal activities are more common or most damaging (Börner et al. 2011; UNCTAD secretariat 2013). The results of the spatial allocation of staff suggest that illegal harvest of wildlife is overshadowed by deforestation as an enforcement priority. This is unsurprising given the international pressure to reduce forest loss in recent decades (Arima et al. 2014), and Brazil's subsequent, and much publicized commitment to reducing deforestation (Nepstad et al. 2002; Soares-Filho et al. 2006).

What are the implications of a limited enforcement capacity and an institutional bias towards deforestation on how we interpret the data obtained on wild-meat activities? The most obvious answer is that the overall detection of these activities will be low, and consequently IBAMA's figures reported on wild-meat harvest and trade appear to be a considerable underestimate. Estimates in criminology commonly assume a 10% detection rate of contraband by officials (Sánchez-Mercado et al. 2016; Wasser et al. 2007); however, considering the limited capacity of IBAMA and the vast area of the Amazon, the detection rate of wild-meat harvest and trade is likely much lower. Indeed, a study on wildlife consumption in two Amazonas municipalities (Borba, 2959 households, and Novo Aripuana, 2672 households) revealed 80-90% of urban households reportedly consume wild-meat (based on respondents admitting consumption in the last 12 months) (Parry et al. 2014). Using these consumption rates we can estimate that 4798 households illegally consume wild meat at least once a year. Yet the detected crimes reported in the same municipalities in this study reveal an enforcement rate of less than one event per year, implying a detection rate of <0.1% of this type of activity by enforcement agents. As the detection of offences is a critical component for compliance within crime and enforcement theory (Robinson, Kumar, and Albers 2010), the current enforcement levels are likely rendering wildlife laws ineffective.

The spatial patterns of enforcement events and the large numbers of seized individuals of some species offer strong evidence of commercial trade and urban consumption, particularly in large cities, where staff presence is higher. Less enforcement (and lower staff presence) was reported in smaller, more remote provincial towns. This finding highlights another enforcement gap, as these small remote towns have potentially high rates of wild-meat consumption and trade (Baía Jr, Guimarães, and Le Pendu 2010; Parry, Barlow, and Pereira 2014; van Vliet et al. 2015a), driven by higher levels of poverty, increased participation in rural-urban networks, and persisting rural preferences (Padoch et al. 2008; Parry, Barlow, and Pereira 2014). Consequently, the combination of limited resources and institutional priority setting hinders the detection of illegal wild-meat activities in remote provincial towns.

3.5.2 What insights can be gained from enforcement record data on species-level wild-meat patterns in Amazonia?

Characterising and understanding species-level trade dynamics is vital to identify priorities for conservation and the potential for sustainable management, as different species have varying resilience to hunting pressure (Milner-Gulland and Bennett 2003;

Stork et al. 2009). I identify two different types of trade evidenced in the enforcement records: mass harvest of species, denoting high commercial demand and trade; and small-scale forest hunting with the likely intent to sell some or all of the meat.

First, I found evidence of mass commercial harvest of turtles (*Podocnemis* spp.), caiman (*Melanosuchus* and *Caiman* spp.) and whistling duck (*Dendrocygna* spp.), which all had high numbers of individuals or biomass recorded in single enforcement events. Commercial trade is known to be a driving force of overharvesting and population decline (Antunes et al. 2016; Milner-Gulland and Bennett 2003) so its detection for these species is of conservation concern. The detection rate of turtles and the high quantities involved are of a significant magnitude greater than other species, and warrant particular attention. In Tapauá, a municipality in the state of Amazonas, a study reported that 100% of the 72 urban households surveyed admitted consuming turtle meat (Pantoja-lima et al. 2014). In the same state, Parry et al. (2014) found that 52% of urban households had eaten turtle in the last year. Our broad-scale results indicate that such high consumption rates are likely to be widespread across the region, and an organized and lucrative commercial trade is supplying large urban centres. The three most commonly harvested *Podocnemis* species are listed as Near Threatened on Brazil's national redlist (ICMBio, 2014); such wide scale intensive pressure on turtle populations is unlikely to be sustainable, impacting populations and increasing extinction risk.

Terrestrial species, particularly ungulates and rodents, were also prevalent in the records, but the low average number of individuals seized (1-2 individuals) suggests a second type of trade pattern: a relatively high frequency of harvest but on a small local scale. Other terrestrial species present in the reports (albeit at a lower frequency) are particularly sensitive to overhunting due to low intrinsic rates of increase (such as tapir, large-bodied primates and large game birds), and are reportedly often preferred by hunters (Bodmer 1995, Peres, 2000). Consequently, even the apparent small-scale harvesting of more vulnerable terrestrial species is likely impacting populations, congruent with species depletion detected even in remote rainforest towns with low human population density (Parry and Peres 2015).

I assert that the detection bias due to the spatial patterns of institutional capacity and the realities of enforcing regulation over vast, inaccessible forest must be considered when assessing these species-level trade patterns. The enforcement data will likely be most biased against the detection of small-scale harvest events of terrestrial forest species. Indeed, the spatial analysis of XY coordinates of enforcement events showed that little enforcement occurs in primary or secondary forest where the original extraction (at least of terrestrial species) is assumed to take place (Parry, Barlow, and Peres 2009), instead occurring mostly along rivers (during transportation to the destined market point or consumer). These small-scale activities have a reduced "detection window", i.e. much less time and distance between extraction and point of sale or consumption. Furthermore, small harvests of only one or two individuals would likely be easier to conceal from officials, compared to larger commercial scale harvest activities. These same practical realities also have implications on the high frequency of turtle enforcement events; as aquatic species, turtles are easily accessible and can be harvested in large numbers during breeding seasons (when they predictably congregate on beaches to lay eggs) (Klemens and Thorbjarnarson 1995). Turtles can also be transported live for long periods of time, allowing them to supply large urban markets far from their original point of extraction (Pantoja-lima et al. 2014); thus increasing the time spent in the transportation phase, and also arriving in bigger cities with a higher enforcement presence. The same principles apply to caiman: harvesting occurs in rivers and lakes, and the meat is often dried and salted (Da Silveira and Thorbjarnarson 1999), allowing for much wider trade networks and, thus, increasing the chance of detection. To summarise: the spatial patterns of enforcement capacity and detection provide important insights into trade dynamics, but should not be used as indicators of hunting offtake. Small-scale harvests of terrestrial vertebrates are likely to be particularly under-estimated in IBAMA's database, and the use of enforcement reports may be better for highlighting large-scale commercial trade patterns.

3.5.3 Decreasing detection rate: Limitations for monitoring temporal trends and implications of decentralizing enforcement

Is the overall rate of enforcement against wild-meat crime decreasing? The analyses in this study essentially show a halving in the rate of enforcement events over the time period analysed. There is little evidence to suggest that the decrease in enforcement reports reflect a decrease in wild-meat offtake; indeed, increasing poverty and urbanization (IGBE, 2010) suggest that demand and consumption of wild-meat is likely to have increased over the time period examined (Parry, Barlow, and Pereira 2014). Instead it seems much more likely that the decline in wild-meat crime detection rates from 2007/2008 onwards reflects reduced detection rates, rather than reduced activity patterns. There are two key political reasons that likely underpin this reduction in detection rates.

Firstly, the reduction in the policing of wildlife crime may be the outcome of increased enforcement effort against deforestation (Nepstad et al. 2014) and is again indicative of the prioritisation of IBAMA's resources, which, additionally, have been increasingly constrained by budget cuts (Monteiro da Silva and Bernard 2015). This highlights the unstable and variable nature of an enforcement effort that is vulnerable to the political framework of a developing country with conflicting national priorities (Ferreira et al. 2014; Tollefson 2016). Secondly, and possibly most importantly, the reductions in enforcement events are likely to reflect the outcome of decentralization, whereby responsibility for enforcing wildlife crime laws was transferred to state and municipal-level by Federal Law 140 (Presidência da República do Brasil, 2011). This occurred in 2011, so although IBAMA continues to assist state police in arrests and seizures (hence the continued records in IBAMA's database, post 2011), the observed decrease in detected game-based crime is also likely a product of IBAMA's reduced role in enforcement efforts. This also pertains to the closing of many of IBAMA's offices from 2007, when it was decided that administrative restructuring would include the closure of 134 municipal IBAMA units in interior cities throughout the country (Rocha 2007).

Decentralization and local governance has been a key strategy in development policy over the last quarter-century (Rees and Hossain 2010). This strategy, in theory, aims to increase democratic participation, efficiently cater to varying local-level demands and reduce corruption (Olum 2014). Brazil's decentralizing of responsibility for wildlife crime enforcement to state and municipal level could improve governance and facilitate apposite policy, empowering local authorities to protect their natural resources and account for the spatial heterogeneity of trade and consumption patterns (Monteiro da Silva and Bernard 2015). However, the expansive literature on decentralization has highlighted its potential failings when implemented without careful planning, especially in developing countries (Olum 2014; Ribot, Agrawal, and Larson 2006). Unfortunately, many states and municipalities are ill prepared for their new responsibility (Monteiro da Silva and Bernard 2015) and consequently inconsistency in enforcement effort and strategy will likely increase as a result of decentralization (Ribot, Agrawal, and Larson 2006). The effects will likely be felt most strongly in the vast Amazon interior, creating an enforcement vacuum in the very areas where levels of wild-meat activities may be highest (Baía Jr, Guimarães, and Le Pendu 2010; Parry, Barlow, and Pereira 2014; van Vliet et al. 2015a).

3.5.4 Policy implications

A key objective of assessing and monitoring wildlife harvest and trade is to enable scientists and policy-makers to design appropriate and realistic mitigation strategies (Assunção, Gandour, and Rocha 2013; Gavin, Solomon, and Blank 2010). This study demonstrates that the use of enforcement reports can provide unique insights into broad-scale species-level trade patterns to help identify conservation priorities and potential sustainable management initiatives. Perhaps more importantly, the value of critically assessing enforcement capacity has significant pragmatic implications, highlighting the challenging realities of governance in vast tropical forest systems and a spatial enforcement gap that renders legislation essentially ineffective (Lo, Fryxell, and Wong 2006; Robinson, Kumar, and Albers 2010). Although improving the effectiveness of legislation and enforcement practices remains critical for protecting tropical forests (Börner et al. 2014; Börner et al. 2015), I impress the need for innovative and alternative approaches to top-down governance, especially in the context of wild-meat harvest (Bennett et al. 2007). Many have highlighted the ethical issues of criminalizing an activity that many poor and marginalized people rely on for food security and livelihoods (Cawthorn and Hoffman 2015). Consequently, I recommend more cooperative or conciliatory styles of regulation (McAllister 2008), taking into account the variable vulnerability of species to hunting pressure (Franzen 2006; Kümpel et al. 2010). Research suggests that external bodies of support, including pro-environment societal groups and NGOs, can be important allies for environmental agencies, particularly at the local level, boosting enforcement presence and also enacting social change through awareness campaigns (Lo, Fryxell, and Wong 2006).

3.5.5 Enhancing the value of enforcement record data: a worthwhile strategy for agencies

I have demonstrated that enforcement records represent an important source of information that, if utilised pragmatically, could highlight enforcement gaps and inform policy. However, the potential value of such data could be greatly improved. In the public health sphere, electronically available administrative data is used as a valuable tool for planning and surveillance, and its strength and weaknesses considered and evaluated to establish validity (Jutte, Roos, and Brownell 2011; Virnig and Mcbean 2001). In this study, data processing was hampered by a lack of standardization in data entry, such as use of local common species names (with various spellings), differing categorization of offences, and inconsistency in recording georeferenced location data. Implementing improved standardized protocol for data collection by enforcement officials would be a cheap and effective way of increasing the value of such databases (Gavin, Solomon, and Blank 2010; Jutte, Roos, and Brownell 2011; Virnig and Mcbean 2001), especially for monitoring purposes. Furthermore, expanding the types of information recorded to include socio-demographic information alongside each violation would offer a unique opportunity to examine characteristics of typical actors on a large scale (Gavin, Solomon, and Blank 2010). This study therefore highlights the need to improve data quality and record keeping at state and municipal level to maximize consistency. This is going to be even more challenging with further decentralization.

3.6 Conclusion

Conservation scientists are increasingly recognising the need for innovative approaches and the value of alternative data sources (Keane 2013; Lund et al. 2014; Parry and Peres 2015). Designing effective policy and management strategies for the multi-faceted and complex issue of wild-meat harvest and consumption in tropical forests is an immense, and urgent conservation challenge (Bennett et al. 2007; Cawthorn and Hoffman 2015; van Vliet, Gomez, et al. 2015). The first step towards achieving this goal is to characterise the extent and dynamics wild-meat harvest and trade, to enable baseline monitoring and to design appropriate mitigation strategies (Munari, Keller, and Venticinque 2011; Strandby and Olsen 2008; Yu, Levi, and Shepard 2010). This study demonstrates the value of enforcement records as an easy and cost-effective method to rapidly assess wild-meat activities on a large spatial scale, finding evidence of varying scales and dynamics of species-level trade. The biases inherent in such datasets mean we must be cautious in our interpretations, particularly when assessing temporal trends. Nonetheless, the data provides a rich set of information that complements intensive local-scale research and helps identify high priority species, patterns of trade, enforcement gaps and important insights into the challenging realities of governance in vast tropical forests (Assunção, Gandour, and Rocha 2013; Contreras-Hermosilla 2002; Kelman 2013). I believe that these approaches can therefore assist policy-makers to develop progressive and realistic strategies that can balance both social development and conservation goals in the context of resource-limited and spatially extensive tropical contexts.

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Appendix 2: *Sources used to estimate species biomass*

SPECIES	INDIVIDUAL BIOMASS (KG)	SOURCE/REFERENCE
Mammals		
Cuniculus paca	9.50	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Dasyprocta spp	4.45	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Hydrochoerus hydrochaeris	50.0	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Alouatta spp	7.35	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Ateles spp	8.15	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Lagothrix spp	6.90	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Cebus spp	3.10	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Saimiri spp	0.94	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Aotus spp	1.01	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Pithecia spp	2.35	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Saguinus spp	0.39	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Mazama americana	36.0	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Mazama nemorivaga	14.7	Rossi, R.V. and Duarte, J.M.B. (2015). *Mazama nemorivaga*. The IUCN Red List of Threatened Species 2015

Ozotoceros bezoarticus	35.0	Ungerfeld, R. et al. (2008). Reproductive biology of the pampas deer (Ozotoceros bezoarticus): a review. *Acta Veterinaria Scandinavica*, *50*(1), 16.
Dasypus spp	7.25	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Priodontes maximus	30.0	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Cabassous unicinctus	3.20	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Euphractus sexcinctus	4.85	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Tolypeutes tricinctus	1.47	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Tayassu pecari	35.0	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Pecari tajacu	26.0	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Tapirus terrestris	238.5	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Nasua nasua	5.10	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Bradypus spp	3.90	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Myrmecophaga tridactyla	31.0	Emmons, L. H., & Feer, F. (1997). *Neotropical rainforest mammals. A field guide (2)*. University of Chicago Press
Reptiles		
Podocnemis unifilis (eggs)	6.90 (0.026)	Ojasti, J. (1996). Wildlife utilization in Latin America: Current situation and prospects for sustainable management. *FAO Conservation Guide*, 25

Podocnemis expansa (eggs)	24.50 (0.040)	Ojasti, J. (1996). Wildlife utilization in Latin America: Current situation and prospects for sustainable management. *FAO Conservation Guide*, 25
Podocnemis sextuberculata (eggs)	2.30 (0.019)	Haller, E. & Rodrigues, M. (2006). Reproductive Biology of the Six-Tubercled Amazon River Turtle Podocnemis sextuberculata (Testudines: Podocnemididae), in the Biological Reserve of Rio Trombetas, Pará, Brazil. *Chelonian Conservation and Biology*, 2006, 5(2)
Podocnemis erythrocephala (eggs)	1.14 (0.017)	Mittermeier R. et al. (2015). *Podocnemis erythrocephala* (Spix 1824) – Red-headed Amazon River Turtle, Irapuca. In *Conservation Biology of Freshwater Turtles and Tortoises: A Compilation Project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group.*
Chelonoidis spp.	5.75	Ojasti, J. (1996). Wildlife utilization in Latin America: Current situation and prospects for sustainable management. *FAO Conservation Guide*, 25
Melanosuchus niger	40.90	Kluczkovski Junior, A. et al. (2015) Carcass yield and proximate composition of black caiman (Melanosuchus niger) meat. *International Journal of Fisheries and Aquaculture.* 7(4)
Caiman crocodilus	23.50	Ojasti, J. (1996). Wildlife utilization in Latin America: Current situation and prospects for sustainable management. *FAO Conservation Guide*, 25
Birds		
Crax spp,	2.55	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Mitu tuberosum	2.81	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Mitu mitu	2.85	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Tinamus solitarius	1.39	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
C. atrocapillus, C. cinereus	0.49	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Tinamus guttatus	0.69	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press

Aburria jacutinga	1.25	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Penelope superciliaris	0.90	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Jabiru mycteria	6.05	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Cairina moschata	2.45	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Dendrocygna spp	0.75	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Callonetta leucophrys	0.37	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Anas spp	0.43	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Netta peposaca	1.00	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press
Mergus octosetaceus	0.98	Dunning Jr, J. B. (1992) *CRC Handbook of Avian Body Masses* CRS press

USING LOCAL ECOLOGICAL KNOWLEDGE TO UNDERSTAND VERTEBRATE RESPONSES TO LANDSCAPE CHANGE

Rural Itamonte. Local resident feeding a wild guan *Penelope obscura*

Chapter 4 Using Local Ecological Knowledge to understand vertebrate responses to changing tropical landscapes

4.1 Abstract

Success in safeguarding much of the world's terrestrial biodiversity rests on our understanding of species presence/absence and population trends within tropical human-modified landscapes. However, adequate data on species' responses are sparse, hindered by the limitations of ecological methods and the complex heterogeneity of historic and current anthropogenic pressures. I critically examined the utility of Local Ecological Knowledge (LEK) to elucidate occupancy and population trends of vertebrate species in the Brazilian Atlantic forest, a severely fragmented landscape with a long history of human modification. I conducted interviews with rural people to assess their knowledge of 16 native species, and also elicit their perceptions of social, environmental and economic processes of change. Using species recognition as an indicator of knowledge, our results show that an individual's LEK is influenced by local forest cover and varies by species. I demonstrate that LEK can provide valuable information on species responses within transitioning severely-modified tropical landscapes, but the method is more effective for identifying positive responses for more resilient species still commonly seen by local people. A lack of knowledge and apparently incongruent responses for less-known or more cryptic species cannot necessarily be interpreted as evidence of species absence. Instead this highlights the need for more refined selection of local experts, better understanding of the process of knowledge acquisition and empirical validation of LEK in this landscape. Qualitative insights complimented the LEK-data, illustrating the social, economic and political drivers of landscape change that shape contemporary species patterns. I conclude that whilst LEK-based approaches in heavily-modified tropical landscapes require further field validation, they provide a

pathway for integrating socio-cultural dynamics into conservation management strategies.

Keywords: Atlantic forest, Brazil, species trends, human modified landscapes, Local Ecological Knowledge (LEK), social-ecological change

4.2 Introduction

As we enter the Anthropocene (Corlett 2015) it is now widely accepted that the fight to preserve terrestrial biodiversity hinges on the management of tropical and sub-tropical human-modified landscapes (Koh and Gardner 2010; Malhi et al. 2014; Newbold et al. 2014). The traditional conservationist perspective is that rural population growth, agricultural expansion and resource extraction are dramatically and rapidly altering these biodiverse landscapes, resulting in forest loss, habitat fragmentation, changes to species composition and loss of ecosystem function (Galetti and Dirzo 2013; Newbold et al. 2014).

However, following the land use transition trajectories of more developed regions, ongoing urbanization and industrialization in these developing tropical landscapes can lead to forest regeneration. As rural populations decline, former agricultural and pastureland becomes abandoned, allowing for native vegetation regrowth. This process is known as forest transition (Molin et al. 2017; Rudel et al. 2005), and it has been purported that increasing rates of forest regeneration could circumvent the anticipated mass extinction of tropical forest species (Wright and Muller-landau 2006). Cases of forest transition in heavily modified tropical landscapes have been documented in parts of Asia (Meyfroidt 2013; Meyfroidt and Lambin 2011) and in Brazil's severely degraded Atlantic forest (Baptista and Rudel 2006; Walker 2012). Yet prospects for conserving biodiversity in secondary forests are variable, depending on spatial and temporal landscape dynamics and continuing anthropogenic pressures (Chazdon, Peres, et al. 2009; Fahrig et al. 2013). Consequently, in order to identify effective management strategies, conservationists must now focus on the social-ecological dynamics of modified lands, navigating an increasingly complex reality that no longer separates humans from nature (Berkes, Colding, and Folke 2003; Gardner et al. 2013; Milner-Gulland 2011).

A critical step towards designing appropriate conservation actions for heavily-modified tropical landscapes is to understand and monitor species responses (Hansen et al. 2001). This is hugely challenging; the dynamic and interminable combinations of historical and contemporary land-use and land-cover change across heterogeneous regions greatly inhibit broad predictions (Gardner et al. 2010; Newbold et al. 2014; Sekercioglu et al. 2007), and this is further compounded by incongruent cross-taxon response patterns (Gardner et al. 2009). Furthermore, non-structural anthropogenic disturbance through extractive activities such as hunting often continue to impact species populations in degraded and secondary forests (Cullen, Bodmer, and Valladares Pádua 2000).

For larger-bodied invertebrates (often used as indicators of habitat disturbance and direct anthropogenic pressures on fauna) our current understanding of species responses to landscape change is based predominantly on ecological methods, such as diurnal line transects and sign or track surveys (Chiarello 2000; Munari, Keller, and Venticinque 2011; Peres 2001; De Thoisy, Brosse, and Dubois 2008). Along with spatial and temporal constraints, these methods require intensive sampling effort and can suffer from high levels of uncertainty, especially when detecting rare or cryptic species (Munari, Keller, and Venticinque 2011; De Thoisy, Brosse, and Dubois 2008). Relative abundance measures based on line transect census sightings tend to have large error coefficients, restricting their capacity to capture temporal changes in species abundance and consequently limiting their use as a monitoring tool (Munari, Keller, and Venticinque 2011; De Thoisy, Brosse, and Dubois 2008). Perhaps most importantly, ecological methods often fail to integrate their findings with information on the social or economic factors that are critical to understanding species' responses within rapidly changing landscapes and ecosystems (Meijaard et al. 2011).

The use of Local Ecological Knowledge (defined as experiential knowledge derived from lived interactions with local environments (Turvey 2016), hereafter LEK) has emerged as a method to monitor species distribution and trends, and provide much needed social context to drivers of change (Anadón et al. 2009; Turvey et al. 2014). The application of LEK in conservation science is appealing as it can enable the integration of local people into the design and implementation of community-based management and monitoring initiatives, which can contribute to conservation effectiveness (Kilbane Gockel and Gray 2009; du Toit, Walker, and Campbell 2004). Nonetheless, the robustness and application of LEK methods is still being explored, since data can be susceptible to multiple biases (Gilchrist, Mallory, and Merkel 2005), depending on levels of knowledge (Davis and Wagner 2003) and the sensitivity of the subject in question (Nuno and St. John 2015). Overall, using LEK to understand species responses has been largely restricted to relatively remote and traditional communities, albeit in a range of biomes. LEK methodologies originated in collaborative research with indigenous Arctic Inuit communities (Ferguson, Williamson, and Messier 1998; Gilchrist, Mallory, and Merkel 2005; Mallory et al. 2003). Efforts have also focused on assessing trends in commercial fish populations (Damalas et al. 2015; Frezza and Clem 2015) and, more recently on assessing status and threats to tropical forest species (Meijaard et al. 2011; Turvey 2016). For example, Parry and Peres (2015) drew on the local knowledge of Amazonian hunters for understanding large-scale spatial variation in the depletion of hunted Amazonian species. Others have utilised LEK to inform on particular highly threatened and cryptic forest species based in communities that still practice traditional extractive activities (Nash, Wong, and Turvey 2016; Turvey et al. 2014).

The potential of LEK to inform on species status and trends within heavily modified landscapes, beyond a highly specialized group of target resource users, remains largely unknown. This knowledge gap is problematic due to the conservation importance of these landscapes, combined with the limitations of ecological methods. With development and urbanization, studies have reported changes and losses in various types of local knowledge within traditional communities, as they become more integrated into national societies and the market economy (Gómez-Baggethun and Reyes-García 2013; Iniesta-Arandia et al. 2014). Nonetheless, Iniesta-Arandia et al. (2014), documented the persistence of LEK in rural southern Spain, explained by the time respondents spent living in the area and the social relationships among farmers that fostered information exchanges. Consequently there is scope for LEK to provide a cost-effective means to gain novel and important insights on species responses to environmental change in modified tropical landscapes.

In this study I aspired to refine the use of LEK by assessing its adequacy for assessing species occupancy and trends in the heavily modified sub-tropical landscape of Brazil's Atlantic forest. With exceptionally high levels of biodiversity and endemism, the Atlantic forest represents arguably the most threatened tropical biome (Gardner et al. 2010; Laurance 2009), following centuries of deforestation that has reduced forest cover by 92% (Ribeiro et al. 2009, 2011; Scarano and Ceotto 2015). The biome now exists in a highly fragmented state of forest remnants embedded within an expansive agricultural matrix (Ribeiro et al. 2009). Although the Atlantic forest could be considered a "post-disturbance" landscape in the context of its severe historical forest loss, the dynamic state of this heterogeneous biome persists (Metzger et al. 2009; Ribeiro et al. 2009); deforestation of native forest is ongoing (albeit at much lower rates than in previous centuries), coexisting with evidence of forest regeneration (Baptista and Rudel 2006; de Rezende et al. 2015; Walker 2012). Moreover, there is evidence of continued subsistence and commercial hunting of wildlife (Cullen, Bodmer, and Valladares Pádua 2000; Souza and Nóbrega 2014). Despite clear evidence of the detrimental impacts of fragmentation and habitat loss on biodiversity (Canale et al. 2012; Tabarelli et al. 2010), there is also evidence of various taxa exhibiting resilience and adaptive strategies within edge and disturbed habitats, and studies have even documented incongruent species responses (Martensen, Pimentel, and Metzger 2008; Metzger et al. 2009; Tabarelli et al. 2010).

The severely modified and dynamic nature of the Atlantic Forest therefore makes it an ideal study site to assess the long term consequences of human disturbance, thus providing valuable lessons for other regions that unfortunately will face similar consequences in the absence of effective management interventions (Gardner et al. 2010). I assess this by combining a quantitative analysis of LEK in rural dwellers with qualitative data to explore the drivers of variation in both local knowledge levels and species responses. Through the use of participants' perceptions of social, environmental and economic changes I expect to provide a deeper local context for species responses, and explore how this context may impact the applicability of LEK in these landscapes. To do this I ask the following research questions:

1. *Based on the recognition of charismatic or culturally important native species, what are the levels of LEK in this landscape?*
2. *How do levels of LEK vary among (i) respondents and (ii) species?*
3. *How effective is LEK for elucidating species occupancy and population trends in these landscapes?*
4. *How do perceived social, economic and environmental changes identified by respondents (i) impact levels of LEK and (ii) help explain species trends?*

4.3 Methods

4.3.1 Study site

This study was conducted in the municipality of Itamonte (22°17’02” S; 44°52’12” W), located within the state of Minas Gerais, in southeastern Brazil (Figure 8). The territory covers an area of 432 km^2, the vast majority of which is rural, composed of forest remnants, dispersed farmland, abandoned pastures, and small-scale commercial plantations. The municipality includes part of the Mantiquiera Mountain range and elevation reaches up to 2300m above sea level. Large areas of Itamonte are preserved by state and federal Protected Areas of the Brazilian Atlantic Forest (Figure 8). The total amount of primary forest cover is estimated at around 157.4 km^2, or 36% of the municipal area (SOS Mata Atlântica 2015), which has decreased from 257.4 km^2 (60%) estimated in 2003 (Scolforo and Carvalho 2006; SOS Mata Atlântica 2015).

The total population of Itamonte is estimated to be around 15,267 (projected population from previous census data; IBGE, 2010). A small city (of the same name) is located close to the western border of the municipality, where over two-thirds of the population reside (IBGE 2010). Data from the last two population censuses show a decreasing rural population trend, falling from 5,512 inhabitants in 2000 to 4,391 in 2010 (IBGE 2000, 2010). The rural population is scattered across the municipality in 35 rural “districts” or neighbourhoods (translated from the Portuguese “bairros”), which consist of clusters of houses and farms. The last population census registered 40% of the total population as working in the agricultural sector (29.9% of men and 9.8% of women; IBGE 2010).

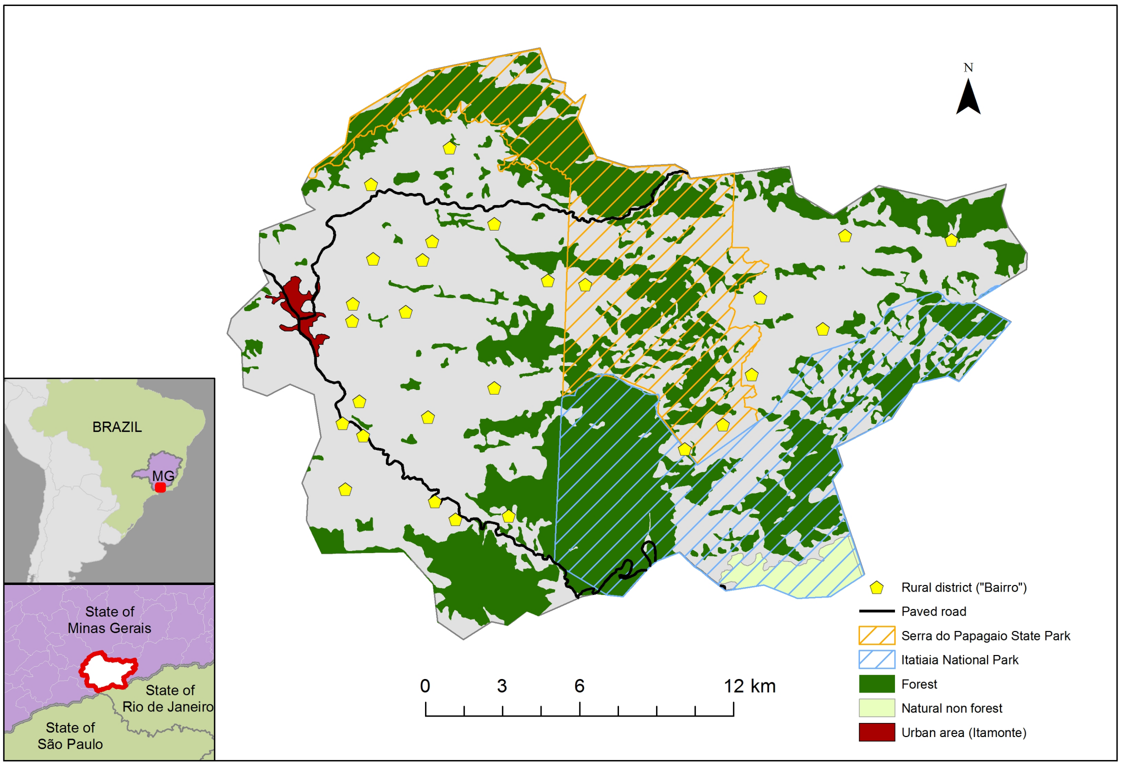

Figure 8: The municipality of Itamonte, including land cover, paved roads and boundaries of protected areas. District points refer to the approximate centre point of the rural districts where households were surveyed, recorded with a GPS.

4.3.2 Respondent selection

Respondent selection is particularly important when eliciting LEK due to the inherent variability of people's lived experiences and therefore knowledge of the natural environment (Chalmers and Fabricius 2007). My aim here was to gauge overall levels of LEK for large vertebrates in this landscape and consequently our sampling protocol was to select heads of households with a minimum of 10 years residency in Itamonte. Participants were identified via targeted "snowball sampling" (Newing 2011), initially using local contacts that indicated potential long-term residents who would be willing to participate. We (the author and a research assistant) also opportunistically requested interviews with people we met when we entered a rural district (Nash, Wong, and Turvey 2016). Children and teenagers were not interviewed, but we included respondents of both genders and of any occupation. Although we hoped to visit all 35 districts, access to some was restricted due to the poor condition of Itamonte's mountainous and unpaved roads. We successfully visited 27 of the 35 rural districts, obtaining the necessary number of interviews to capture the sufficient variation of responses in compliance with predicted response saturation levels (Guest, Bunce, and Johnson 2006; Nash, Wong, and Turvey 2016).

Verbal consent was obtained before starting interviews, and prior to being asked questions respondents were informed that they could stop the interview at any time, and that their responses would remain anonymous. I conducted all interviews in Portuguese together with a female research assistant, who was Brazilian.

Table 3: The 16 native vertebrate species selected as relevant in terms of their ecological importance in modified landscapes and their increased potential for recognition by local respondents due to ecological characteristics and perceived cultural or social importance. Also included is the selected negative control species.

SPECIES	COMMON NAME	CRITERIA FOR SELECTION
Mammals		
Cuniculus paca	Paca	Highly prized game species
Hydrochoerus hydrochaeris	Capybara	Game species; Hunted due to damages to agriculture
Pecari tajacu	Collared peccary	Game species; Hunted due to damages to agriculture; Important seed-disperser
Tayassu pecari	White-lipped peccary	Game species; Hunted due to damages to agriculture; Important seed-disperser
Mazama americana	Red brocket deer	Highly prized game species
Nasua nasua	Coati	Game species; Seed disperser
Puma concolor	Puma	Apex predator; Hunted due to damages to animal husbandry and fear
Leopardus pardalis	Ocelot	Hunted due to damages to animal husbandry

Leopardus tigrinus	Margay	Hunted due to damages to animal husbandry
Panthera onca	Jaguar	Apex predator; Hunted due to damages to animal husbandry and fear
Alouatta guariba	Red howler monkey	Game species; Hunted for pet trade; Important seed disperser
Cebus apella	Brown capuchin monkey	Game species; Hunted for pet trade; Important seed disperser
Birds		
Penelope obscura	Dusky-legged guan	Game species; Notable population declines
Ramphastos toco	Toco toucan	Game species; Hunted for pet trade; Important seed disperser
Ramphastos dicolorus	Green-billed toucan	Important seed disperser
Reptiles		
Tupinambis spp	Tegu	Game species; Hunted due to damages to animal husbandry
Negative Control Species		
Lemur catta	Ring-tailed lemur	A distinct non-native species (endemic to Madagascar) that does not occur in the region used as a negative control to check response accuracy (Nash, Wong, and Turvey 2016)

4.3.3 Interviews

Interviews comprised a standard questionnaire and a semi-structured interview (see Appendix 3) and generally took around 1 h to complete. The initial, quantitative part of the interview began with showing respondents printed cards with individual colour images of selected species. Sixteen native species were included in this study and were chosen based on relevant criteria presented in Table 3. The species are mostly large-bodied or otherwise distinctive vertebrates that are ecologically important (i.e. apex predator or seed disperser) and/or culturally significant in the region (i.e. previously or currently known to be hunted). Consequently, untrained observers were more likely to provide meaningful information (Turvey et al. 2014). The ring-tailed lemur (*Lemur catta*), endemic to Madagascar, was included as a negative control species in order to check response accuracy and identify unreliable informants (Nash, Wong, and Turvey 2016). Data from interviews in which respondents reportedly recognized lemurs as occurring locally were discarded, although this only occurred once.

Upon presenting an image of a species, respondents were asked if they recognised it, the local name, and whether it is known to currently exist in the municipality. If they could not identify the species, names or further morphological and ecological details (such as calls or size) were used to prompt recall (Nash, Wong, and Turvey 2016). Those that affirmed species presence were asked about the source of this knowledge, whether through direct observations (in which case last sighting and sighting frequency would be recorded), or by other means such as word of mouth, photographs or media.

To elicit perceptions of species trends, respondents were asked whether they considered the species to be more or less abundant now compared to 10 years ago (as per the minimum time as resident parameter of participants) (Abram et al. 2015). If respondents affirmed a population trend we asked why they thought this change had occurred. The use of open-ended questions and elaboration on these quantitative species questions provided qualitative information on drivers of trends (Kahler and Gore 2014).

The second part of the interview was semi-structured and explored respondent's perceptions of local environmental and social changes. The interviewer steered the conversation around a list of topics including land use, presence of environmental law enforcement, quality of life, human diet and cultural customs (including hunting), livelihoods and infrastructure. However the discussion remained flexible and participants were encouraged to elaborate on these topics in any way they felt comfortable (Brooks, Roberton, and Bell 2010). This part of the interview aimed to provide the qualitative backdrop of our quantitative results of part one in two interrelated ways: firstly, how the lives and experiences of rural people have changed and how these changes likely impact levels of LEK, and secondly, to identify drivers of landscape change in order to underpin potential species responses. Although notes were taken during the interviews, the majority of the respondents (82%) gave permission to be recorded, allowing us to revisit the conversation and extract more detailed accounts of responses. For those preferring not to be recorded, the two researchers reviewed the notes as soon as the interview was over to confirm that as much information had been logged as possible. The responses were then carefully checked and coded into categories further defined from the general topics discussed, in order to facilitate a qualitative analysis of the information obtained.

4.3.4 Quantitative analysis

Biases that impact a person's local ecological knowledge on species can be described as internal – based on an individual's personal experiences and circumstances, or external – stemming from ecological differences in the target species themselves (Turvey et al. 2014). To explore the internal bias (i.e. variation across respondents) I used respondents' cumulative species recognition as an indicator of their LEK as a response variable in a Linear Model using environmental and demographic predictors (Table 4). Because of the strong correlation between the age of individuals and the time spent in the area (Spearman's $\rho = 0.913$, $p<0.001$), age was not included as an explanatory variable as to avoid colinearity. Similarly, to explore external bias (i.e. the variation in recognition across species) I modelled recognition (0/1) of individual species against the same predictors (Table 4), using Generalized Linear Models with binomial error distributions. I used the 'dredge' function from the *MuMIn* package to test all possible models from all combinations of predictor variables and ranked by Akaike's Information Criterion corrected for small sample sizes (AICc). Model averaging was used for all models with delta AICc (Δ) < 5 because this balances the need to capture all plausible models whilst avoiding inclusion of spurious models (Burnham and Anderson 2002; Symonds and Moussalli 2011). All analyses were carried out using R version 3.2.2 (R Development Core Team 2013).

Table 4: Potential explanatory variables that predict levels of LEK in rural Itamonte

Explanatory variable	Hypothesis/justification
Proximate forest cover (% cover in 2km buffer of rural district centre point)	Proximity to forest increases people's awareness/knowledge of native species that may occur more in intact landscapes (Silva-Andrade et al. 2016)
Sex of respondent (Male/Female)	Traditional gender roles in rural agricultural lifestyles tend to dictate that men spend more time interacting with natural environments and will have higher LEK (Nash, Wong, and Turvey 2016)
Time in municipality (Number. of years as a resident)	Longer-term residents (closely correlated with age of respondent) will have higher LEK due to accumulated experience and knowledge of the local area (Iniesta-Arandia et al. 2014)

4.4 Results

A total of 63 interviews were carried out during the survey, though one interview was discarded after not passing the negative control. More men than women were interviewed (74%, n = 46, versus 26%, n = 16). The vast majority of respondents had been born in Itamonte (85%, n = 53), spending most, if not all of their lives living in the municipality (mean residence time = 52 years, SD = 18, n = 53). Of the 9 "non-native" respondents, two were women from neighbouring municipalities within Minas Gerais state (Baepende and Alagoa) who had moved to Itamonte after marriage. The other seven respondents had moved to Itamonte from other states: Sao Paulo (n = 4), Rio de Janeiro (n = 1), Pernambuco (n = 1) and Brasilia (n = 1). The mean time spent as a resident of Itamonte for those born elsewhere was 23 years (SD = 12, n = 9).

4.4.1 What are the levels of LEK in this human modified landscape?

Using species recognition as an indicator I found that levels of LEK are highly variable in rural Itamonte. However, there is an apparent baseline of knowledge with the minimum number of species a respondent recognised recorded as four (n = 1). Only two respondents were able to correctly identify all 16 species included in the study, suggesting that very high levels of LEK are uncommon. Nonetheless, the average number of species recognised was 12 (SE = 0.3) (Figure 9), showing that the majority of respondents have acquired some knowledge of the majority of the species in this study.

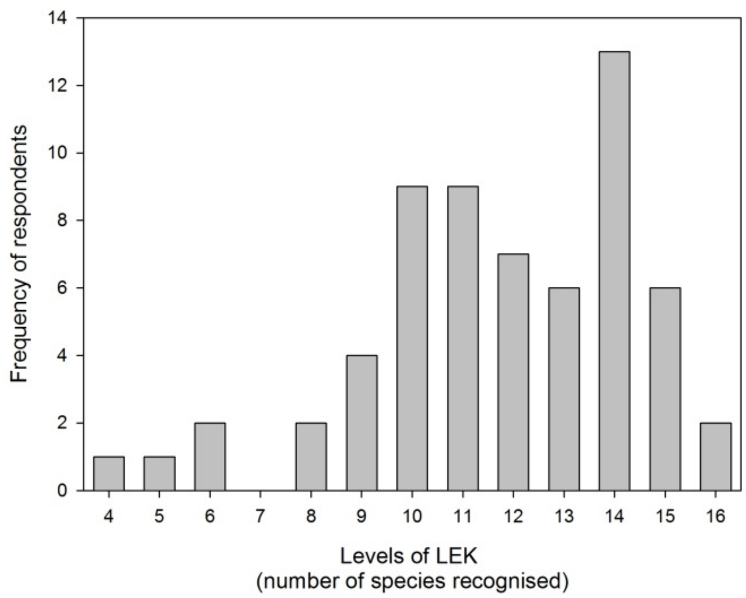

Figure 9: Variation in LEK using the total number of species recognised by respondents

4.4.2 What affects LEK-levels across respondents in this human modified landscape?

Model selection and multi-model inference showed strong support for the effect of proximate forest cover and sex on levels of LEK (Model 1; Table 5). Our results indicate that men (averaged coefficient: 2.3 ± 0.7SE) and those respondents that live close to more intact forest habitat (averaged coefficient: 0.04 ± 0.01SE) recognised more species (Figure 10). The second model (including forest cover, sex and time as resident as variables) was also highly plausible ($\Delta < 2$; (Burnham & Anderson, 2002) (Table 5), although the model weight was c. 50% that of the first model (indicating it was half as likely). Hence our results provide only limited support for the expectation that time spent in the municipality would be an important predictor of LEK. Finally, our predictors explained only 27% of the variation detected in our results (averaged Adj R_2 of models with $\Delta <2 = 0.27$), suggesting that the acquisition of LEK in these landscapes is dependent on a more complex set of factors and experiences.

Table 5: AICc –based model selection for respondent variation in LEK (cumulative species recognition) against proximate forest cover measured as percentage forest within a 2 km buffer zone around each rural district (FOREST), sex of respondent (SEX) and time spent as a resident in the municipality of Itamonte (TIME). I show AICc differences (Δ) and Akaike weights (Wi). Models are shown up to nearest 95% of cumulative Akaike weights (Cumulative Wi).

Model ranks	Model	AICc	Δ	W_i	Cumulative W_i
Outcome variable: LEK (number of species recognised)					
1	FOREST + SEX	284.6	0.0	0.65	0.65
2	FOREST + SEX + TIME	286.0	1.4	0.33	0.98

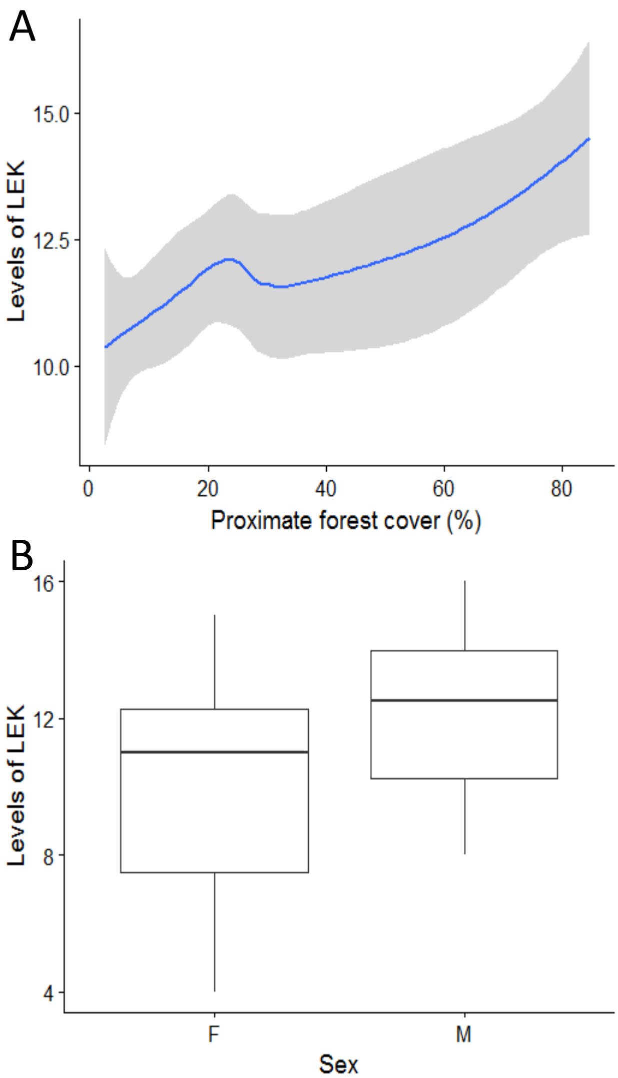

Figure 10: Effect of the influential predictors on levels of LEK supported by multimodel inference (A) proximate forest cover (percentage of forest within 2 km buffer zone from the centre point of a respondent's rural district) and (B) sex of respondent.

4.4.3 How did recognition ability (LEK) vary across species?

I detected variation in respondents' species recognition, with extremely high recognition (over 95%) of *H. hydrochaeris*, *P. obscura*, *Tupinambis sp.* and *M. americana*, yet notably low recognition of the two primate species *A. guariba* and *C. apella* (<40%) (Figure 11). I also found evidence of uncertain and erroneous recognition, including instances in which a respondent claimed to recognise a species but, due to different usages of local names, it was not possible to definitively confirm the validity of their recognition. For example, this occurred particularly regarding the two peccary species, which are both sometimes locally referred to as "porco do mato" (forest pig), and can be further confused with the invasive Eurasian wild boar (*Sus scrofa*, known locally as *javali,* but also referred to as "porco do mato").

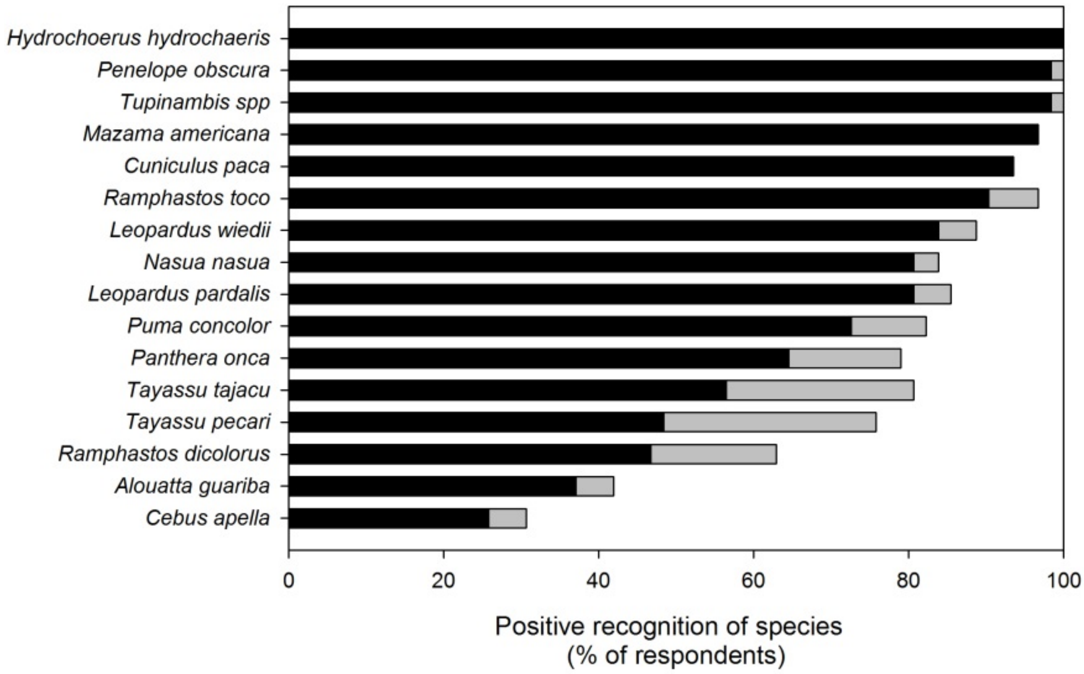

Figure 11: Percentage of respondents able to positively identify different species (black bars). The grey bars indicates doubt noted by the researcher, which resulted from potential mis-identification, often due to the use of general local names that do not explicitly refer to species, but rather groups of similar species.

I was unable to model species-specific recognition (0/1) for all species due to the extremely high recognition rates of some species. A relatively small sample size also hindered my ability to fit robust models, and I present only those species where model averaging indicated effects within 95% confidence intervals (Burnham and Anderson 2002): *Tayassu pecari, Pecari tajacu, Puma concolor* and *Panthera onca.*

For the white-lipped peccary (*Tayassu pecari*) model selection and inference revealed strong support for proximate forest (averaged coefficient: 0.06 ± 0.02SE) as a predictor of species recognition (Model 1: Table 6). The low delta AICc for Model 2 ($\Delta < 2$) also indicated strong evidence for including sex as a recognition-predictor; however, the standard error for this variable fell outside of the 95% confidence intervals (averaged coefficient: 0.4 ± 0.7 SE), suggesting that higher proximate forest cover was the most important predictor for *T. pecari* recognition. For the collared peccary (*Pecari tajacu*) only one model had strong support from the data, and included both proximate forest cover and sex (Model 1; Table 6). Again, men (averaged coefficient: 2.05 ± 0.8SE) and respondents who lived close to more forest (averaged coefficient: 0.06 ± 0.02SE) were more likely to recognise *P. tajacu*. For the two large felids (*Puma concolor* and *Panthera onca*) sex featured in all of the top ranked models (Table 6), revealing strong support that men are more likely to recognise these species (*P. concolor*: average coefficient: 1.9 ± 0.7SE; *P. onca*: 1.8 ± 0.7SE). However, proximate forest cover featured in the strongest model for *P. concolor* (Model 1: Table 6), suggesting that living close to intact forest is also an important predictor for recognising this species (averaged coefficient: 0.02 ± 0.02SE). Interestingly, model selection and inference for recognition of *P. onca* revealed that alongside sex, time spent living in the municipality was an important predictor, present in the two top ranked models (Table 6). However, contrary to our hypothesis the effect of this factor was negative (-0.1 ± 0.02SE), indicating that younger respondents or migrants who have spent less time in the municipality are more likely to recognise *P. onca*.

Table 6: AICc –based model selection for those species where multimodel inference included averaged coefficients [±SE] that fell within 95% confidence intervals (i) Tayassu pecari (ii) Pecari tajacu (iii) Puma concolor (iv) Panthera onca. I used General Linear Models with binomial distribution using percentage of proximate forest cover within the 2 km buffer zone (FOREST) and time spent in municipality (TIME) as random factors, and gender (SEX) as a fixed factor. I show AICc differences (Δ) and Akaike weights (Wi). Models are shown up to nearest to 95% of cumulative Akaike weights (Cumulative Wi).

Model ranks	Model	AICc	Δ	W_i	Cumulative W_i
Recognition of Tayassu pecari					
1	FOREST	73.8	0.0	0.42	0.42
2	FOREST + SEX	74.2	0.5	0.33	0.75
3	FOREST + TIME	76.0	2.2	0.14	0.87
4	FOREST + SEX + TIME	76.4	2.6	0.11	0.99
Recognition of Pecari tajacu					
1	FOREST + SEX	70.5	0.0	0.71	0.71
2	FOREST + SEX + TIME	72.8	2.3	0.23	0.94
Recognition of Puma concolor					
1	FOREST + SEX	67.1	0.0	0.50	0.50
2	SEX	68.6	1.6	0.23	0.73
3	FOREST + SEX + TIME	69.3	2.3	0.16	0.89
4	SEX + TIME	70.7	3.7	0.08	0.97
Recognition of Panthera onca					
1	SEX + TIME	74.5	0.0	0.61	0.61
2	FOREST + SEX + TIME	76.3	1.7	0.26	0.87

4.4.4 What is the potential for LEK to detect species occupancy and population trends?

The most highly recognised species correspond to congruent responses on species' local occupancy and population trends (Table 7). Identifying population trends proved more difficult for respondents than species occupancy, with a consensus of population trends only emerging for four species: *H. hydrochaeris*, *P. obscura*, *Tupinambis sp* and *R. toco* (Table 7).

Table 7: Species recognition, occupancy and population trends elicited from respondents during interviews with rural dwellers of Itamonte (n = 62). Current occupancy refers to the percentage of respondents that affirmed species presence within the municipality of Itamonte. Occupancy and trend data calculated from those respondents who positively identified the species.

Species	IUCN (2017)		LEK (Recognition)	Current occupancy	Population trend (Majority consensus?)
	Category	Trend			
Hydrochoerus hydrochaeris	LC	Stable	100%	98%	Increasing (52%)
Penelope obscura	LC	Decreasing	98%	100%	Increasing (95%)
Tupinambis sp.	LC	Stable	98%	97%	Stable (56%)
Mazama americana	DD	Unknown	97%	98%	Unclear
Cuniculus paca	LC	Stable	94%	97%	Unclear
Ramphastos toco	LC	Decreasing	90%	100%	Increasing (66%)
Leopardus wiedii	NT	Decreasing	84%	100%	Unclear
Nasua nasua	LC	Decreasing	81%	96%	Unclear
Leopardus pardalis	LC	Decreasing	81%	100%	Unclear
Puma concolor	LC	Decreasing	73%	93%	Unclear

Panthera onca	NT	Decreasing	65%	43%	Unclear
Pecari tajacu	LC	Stable	56%	90%	Unclear
Tayassu pecari	V	Decreasing	48%	94%	Unclear
Ramphastos dicolorus	LC	Decreasing	48%	86%	Unclear
Alouatta guariba	LC	Decreasing	37%	78%	Unclear
Sapajus apella	LC	Decreasing	26%	63%	Unclear

4.4.5 According to respondents, what are the drivers of environmental change and species trends?

The qualitative insights elicited from respondents during the semi-structured interviews reveal that a range of interconnected social, economic and environmental changes have occurred in our study area. These factors have transformed the household level dynamics of rural inhabitants of Itamonte and driven landscape change, which in turn has implications on species trends and expected levels of LEK (Figure 12).

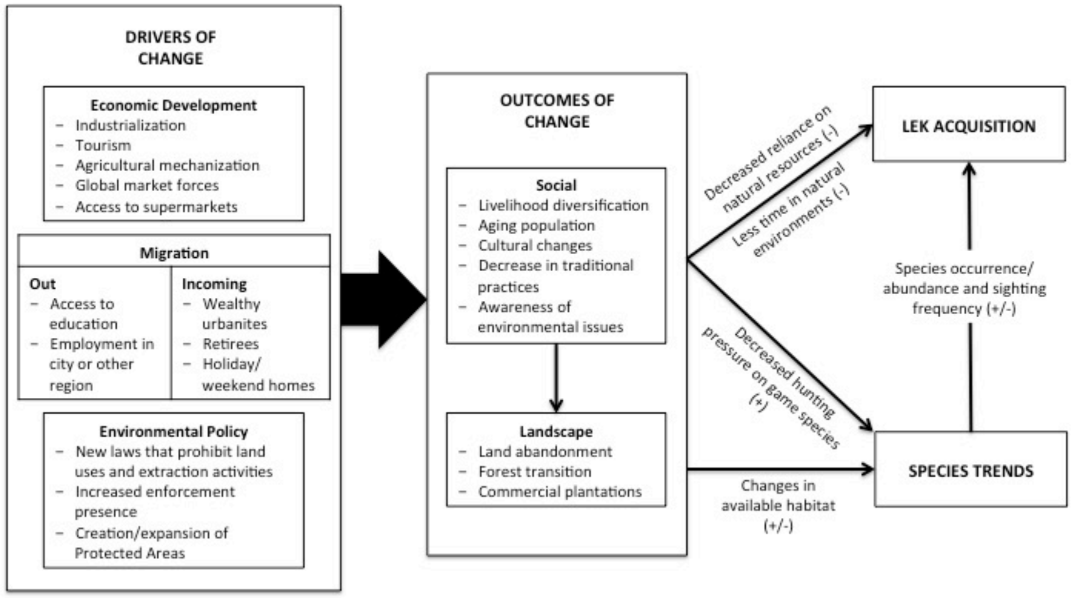

Figure 12: Conceptual framework of social, economic and environmental drivers of species trends and LEK acquisition in the Brazilian Atlantic forest. Plus (+) or minus (-) signs denote the hypothesized effect of outcomes.

Customs, behaviours and activities with direct consequences on native fauna:

(1) Hunting

When discussing direct drivers of species trends the majority of respondents commented on hunting activities (n = 51; 82%), stating that it had been more common in past decades. The motives for historical hunting were identified as a protein source (notably of guans *P. obscura*), human-animal conflict (e.g. felid species attacking chickens and livestock), and also a cultural traditional practice (i.e. as a leisure or "sport" activity). Twenty-five respondents were explicit in linking the change in environmental law and enforcement presence with a clear decrease in hunting activities, with one respondent commenting: "*if you want to be imprisoned, just kill an animal*." However, a few respondents asserted that covert illegal leisure hunting activities persist, particularly of red brocket deer (*Mazama americana*) and paca (*Cuniculus paca*).

(2) Bird feeding

Thirteen respondents commented that they or others would feed wild birds, including guans (n = 11). This action was often brought up by respondents describing a decline in the previously common custom of keeping caged wild songbirds, and a subsequent increase in their populations. This apparent shift in cultural custom was again brought about through the fear of enforcement (with one respondent previously having "*30 caged birds, but 15 years ago [I] felt increased pressure to release them for fear of fines*"), but also through education and awareness around issues of animal welfare: "*when a bird is in a cage it doesn't sing, it cries.*"

(3) The invasive javali

The invasive wild boar *Sus scofra*, locally known as "javali", was an important topic of discussion for many respondents (n=46), with most noting its arrival and increasing presence across the municipality within the last 10 years. Perceptions were highly negative of the species, with human-javali conflict commonly cited due to the destruction of gardens and plantations, and even the maiming or killing of domestic dogs. Five respondents believed that the recent legalization of javali hunting has fostered increased interest in sport hunting, including of native species: "*most hunters will say they will hunt javali, but will hunt any creature that appears.*"

Changes to the rural lifestyle

All participants spoke of changes to the rural lifestyle within their lived experience in the municipality. A decrease in the previously ubiquitous nature of agricultural practices (i.e. planting crops and raising livestock) was commonly affirmed (n=51) with statements such as "*everybody used to plant,*" whether as a primary income or for their own consumption. Often the principal driver given for people's decision to stop planting was the implementation of environmental laws that prohibited certain land use, particularly slash-and-burn activities: "*the environmental agency arrived – and everything changed.*" The perceived increased presence of enforcement and the risk of fines or arrests were a key deterrent to these activities: "*we can't do anything, the environment agency won't let us*", "*if you take a flower, they* [the environmental officers] *will fine you.*" However, other drivers of change to household livelihoods were identified during interviews, including increased production costs that made farming less financially viable, compounded by advancements in mechanized farming that reduced labour opportunities. Some participants mentioned the increased access to supermarkets, facilitated by improved roads and infrastructure, meant many people preferred not to plant. Alternative *in situ* livelihood opportunities arose with urbanization and industrialization, including factory jobs, and more recently through eco-tourism; three respondents currently work in the tourism sector. External urban employment opportunities have led to increased rural out-migration, particularly by the younger generation; many respondents had children or grandchildren working or studying in the city or further afield. Indeed, one participant noted that the decrease in household scale farming is the result of an aging population that are becoming physically less able to farm, and a lack of young people to take the reins. Migration into rural Itamonte was also commonly commented upon and perceived generally as wealthy "outsiders" coming from other states either to purchase holiday homes or retire.

Intimately related to the societal changes of inhabitants, is the changing landscape: many respondents noting increased forest regeneration and natural habitats (n = 30), driven by increased environmental enforcement and land abandonment as people desist with traditional agriculture. However, an increase in eucalyptus plantations was also noted by some (n = 4) and was referred to by one respondent as "dead forest."

4.5 Discussion

Monitoring of species distributions and population trends within human modified dynamic landscapes is a key component of conservation planning (Elith et al. 2006). Methods employed must be robust and provide reliable information, but also be time and cost-effective considering the urgency of the problem and the general resource constraints in conservation (Meijaard et al. 2011). This study examined the potential for LEK to inform on species occupancy and trends in a severely modified tropical landscape in Brazil's Atlantic Forest. Contrary to previous assumptions, our results demonstrate that useful local knowledge is variable but persists in these landscapes. I show social research methods can offer important insights into species responses and, importantly, an understanding of the social and environmental changes affecting biodiversity. Nonetheless, I discuss the levels and sources of variation in knowledge, between respondents and across species, and critically examine limitations of LEK for understanding biodiversity responses to severely modified landscapes in the tropics and sub-tropics.

4.5.1 Who knows? Identifying the relevant experts in severely-modified landscapes

Careful selection of participants is considered fundamental for LEK to provide meaningful and reliable information (Anadón et al. 2009; Davis and Wagner 2003). In general, studies that utilise LEK in order to monitor wildlife focus on people that have spent the most time interacting with natural environments, i.e. older individuals within a targeted group, such as traditional herders, fishers, farmers or hunters (Damalas et al. 2015; St. John, Mai, and Pei 2014; Parry and Peres 2015). However, in severely modified landscapes such as the Atlantic forest, identifying potential "experts" is more challenging. As rural lifestyles and the experiences of individuals continue to evolve, there tends to be decreasing interaction with natural environments and less reliance on natural resources (Iniesta-Arandia et al. 2014; Reyes-garcía et al. 2014). Indeed, despite a sample skewed towards older, long-term rural dwellers, our analyses of respondent-level knowledge variation indicate an important gap in our understanding of how and when LEK is acquired in these landscapes. Our finding that men generally have higher LEK-levels than females is congruent with traditional concepts of gender roles and time spent interacting within the landscape. Often, men in rural areas spend more time outdoors carrying out the more physical farming practices and participate in hunting activities (Nash, Wong, and Turvey 2016). However, the gender difference was modest, and I assert that women should not be discounted as a valuable source of information (Nash, Wong, and Turvey 2016), particularly when the drivers of LEK acquisition in these landscapes remain unclear. Close proximity to intact forest appears to foster increased LEK among the inhabitants of the Atlantic Forest, indicating that knowledge is still in part obtained through a direct spatial connection with proximate natural environments (Nash, Wong, and Turvey 2016). This suggests that even on a relatively small spatial scale, areas of low forest cover in degraded landscapes potentially limit the efficacy of LEK to inform on species.

4.5.2 How do levels of LEK vary among species?

The utility of LEK for eliciting information is heavily dependent on species recognition, which is highly responsive to species' appearance, behavioural ecology and to local cultural attitudes to a given species (Gilchrist, Mallory, and Merkel 2005; Turvey et al. 2014). The handful of studies using LEK to monitor rare or cryptic species (Turvey 2016; Turvey et al. 2015) acknowledge that awareness of these species will be low compared to more common and easily detected species. In our study, low recognition generally corresponded with lower perceived site-occupancy, particularly for the two primates *C. apella* and *A. guariba*. Primates have often been selected for LEK studies because they are charismatic and often targeted by hunters (Meijaard et al. 2011; Parry and Peres 2015; Turvey 2016), and have important ecological roles as seed dispersers (Catenacci, De Vleeschouwer, and Nogueira-Filho 2009; Pizo 2009; Vespa, Zurita, and Bellocq 2014).

Both *C. apella* and *A. guariba* have shown resilience to habitat degradation, persisting in fragmented modified landscapes (Arroyo-Rodríguez and Dias 2010; Michalski and Peres 2005). The low recognition and apparent low occupancy of these species in this study could therefore indicate extremely low abundance or even local absence. However, I assert that this assumption of "pseudo-absence" is open to considerable bias and error and the low reported occupancy rates of these species could be attributed to decreased LEK for rarer forest specialist species. Although this is a clear weakness of the method, it is not one that is easily resolved by other survey techniques, as the rare and most forest-dependent species are often the hardest to incorporate in ecological models (Banks-Leite et al. 2014).

4.5.3 The potential of LEK for detecting species trends

Detecting a clear consensus in species population trends was possible only for a few species. It appears that the rural people I interviewed often struggled to provide reliable trend information, even for easily recognizable, locally present species. To acquire knowledge on population trends requires a measured sense of abundance over time, such as changes in the number of encounters (McPherson and Myers 2009; Turvey et al. 2015), which ultimately requires time spent in nature. For example, paca and red brocket deer were highly recognised and generally locally present, yet there was no consensus on whether or how their populations had changed over the last 10 years. Both species were repeatedly cited as a prime hunting target in the past, indicating that the knowledge of these species stems partly from their cultural prominence during the lifetimes of the respondents. Yet the trend uncertainty suggests that changes in human behaviour and activities (i.e. decreased hunting) have resulted in people no longer encountering these species with any frequency. This highlights the intrinsic variation in detectability of species, an issue that also hinders traditional sampling methods such as point counts or line transects, which can struggle to obtain accurate estimates of shy (e.g. brocket deer) or nocturnal animals (e.g. paca) (Singh and Milner-Gulland 2011). Consequently, the potential of LEK to elucidate species trends is effective for species that are both easily recognized and encountered reasonably often. Less time spent in forested habitats and the increasing peri-urban nature of the rural experience in developing modified landscapes (Restall and Conrad 2015) will limit encounter opportunities, especially for forest specialist species (Newbold et al. 2014). Conversely, habitat generalists and species that demonstrate adaptive capacity within novel anthropogenic ecosystems and landscapes will likely be encountered by people with more frequency, increasing the efficacy of LEK to inform on population trends (McPherson and Myers 2009). This was the case for *Penelope obscura*, which received the highest consensus for an increasing population trend, and is known to adapt its feeding strategies to take advantage of anthropogenic food sources, (Ramos and Pinto 2015). As such I note that overall population trends for such species may be over-estimated as their adaptive feeding strategies lead to increased densities close to human occupation (Ramos and Pinto 2015).

Furthermore, there was no consensus on declining species, and I postulate that declining population trends may be more difficult to detect in post-transition modified landscapes. With the long history of human occupation and environmental degradation in the Atlantic forest, species declines and defaunation have been occurring over timescales beyond the lifetimes of current residents (Canale et al. 2012; Corlett 2013). Large-bodied vulnerable vertebrate species are often the first to disappear from impacted areas, leading to trophic cascades (Galetti and Dirzo 2013; Jorge et al. 2013; Paviolo et al. 2016). Jorge et al. (2013) estimated that jaguar and white-lipped peccary are missing from over 90% of the remaining Atlantic forest. As such, the reported low levels of LEK for such species, and the inability of residents to detect decreasing trends could be evidence of a "shifting baseline syndrome", whereby a lack of communication between generations results in a loss of perspective about past ecological conditions and local species (Nash, Wong, and Turvey 2016; Turvey et al. 2010).

In summary, our results show that the limitations of using LEK for monitoring biodiversity are amplified when applied to dynamic severely modified landscapes. Consequently, it is very important that LEK methodologies are used cautiously and pragmatically within these contexts. An important next step is understanding the acquisition process of LEK in these landscapes in order to better identify (and perhaps 'foster' through training) knowledgeable experts. Careful choice of experts and filtering of appropriate target species offers valuable potential for LEK as a monitoring tool. In this study, evidence of species resilience and adaptive strategies of *Penelope obscura* and *Ramphastos toco* underpin the perceived increases in their populations, and make them more suitable as target species within LEK studies.

4.5.4 Qualitative local context: where LEK shines in characterising dynamic modified landscapes

By utilising LEK I obtained rich social and cultural insights of behaviours that have direct impacts on species populations, such as evolving hunting practices, an emerging custom of feeding wild birds and a decrease in the custom of keeping wild songbirds. Respondents also reported on the dynamic nature of the landscape, including forest transition, through both natural regeneration and increasing eucalyptus plantations. Available data supports these assertions; eucalyptus plantations have been recorded in the municipality since 2013 (IBGE 2016), and forest regeneration over areas of degraded pastures has been documented in the nearby Paraíba Valley (c. 100 km away) (da Silva et al. 2016). Against a backdrop of rapid global change, the safeguarding of tropical biodiversity rests on our understanding of human modified rural landscapes; the degree to which tropical forest biota can persist in them, and which management strategies will be most effective at enhancing persistence (Chazdon, Harvey, et al. 2009;

Gardner et al. 2009). To do this there has been a move towards integrated landscape approaches in conservation that recognize the coupled social-ecological dynamics that characterize modified lands (Reed et al. 2016, 2017; Sayer et al. 2013). The results of our qualitative interviews with rural dwellers demonstrate the value of LEK to identify drivers of social and environmental change and their impacts on biodiversity. Many of the drivers I identified mirror the common global forces cited in the rural transformation and development literature, including the diversification of rural livelihoods away from agriculture and the suburbanization of rural areas (Berdegué, Rosada, and Bebbington 2014; Wang, Yang, and Zhang 2010). These drivers can be described through a "push-pull" paradigm (Lee 1966; Wang, Yang, and Zhang 2010); i.e. factors that push people away from traditional farming activities (including restrictive laws and enforcement), and those that draw people towards alternative opportunities (e.g. factory work and ecotourism). Nonetheless, research shows that local context is critical, and the outcomes of changes on the landscape and biodiversity are dependent on the interplay of both global and local factors (Berdegué, Rosada, and Bebbington 2014; Poe, Norman, and Levin 2014; Reed et al. 2016). Indeed, it is human perception, choice and action that often drive local-level political, economic or cultural decisions that lead to or respond to change in ecological systems (Grimm et al. 2008).

Although I encountered difficulties in eliciting temporal species trends from respondents, there is clear potential for LEK as a species monitoring tool (Jones et al. 2008; Moller et al. 2004; Stem et al. 2005). By repeating surveys that elicit current knowledge the method could detect relative differences in perceived abundances for certain species, increasing recall accuracy by comparing current experiences of respondents over time (Golden, Wrangham, and Brashares 2013). Incorporating LEK into management and monitoring initiatives has also been shown to increase the effectiveness of conservation actions, enhancing community participation and fostering an increased connectedness of local people to natural environments (Danielsen et al.

2010; Torres et al. 2016). Initiatives that aim to promote the maintenance of LEK could prove an important strategy for policy-makers in these landscapes.

4.6 Conclusion

This study demonstrates that valuable information on species status and trends can be found utilising LEK within severely modified tropical landscapes. Considering the urgent need for effective conservation strategies for tropical fauna and the scarcity of long-term, systematic data on species distribution and abundance (Gardner et al. 2007; McPherson and Myers 2009), these findings are particularly apposite, supporting the use of LEK as a rapid and cost-effective monitoring tool (Beaudreau and Levin 2014; Parry and Peres 2015). Nonetheless, our results also confirm assertions that careful and robust method design and development is critical (St. John et al. 2014). Indeed, the known limitations of LEK, such as respondent bias and varying levels of knowledge of different types of species (Anadón et al. 2009; Turvey et al. 2014) are likely more prominent within dynamic modified landscapes. It appears that reduced abundance of species within a highly fragmented Atlantic forest landscape make it more difficult for local respondents to perceive trends (Chiarello 2000; Turvey 2016). Furthermore, the changing rural lifestyles, revealed using qualitative methods, make identifying local experts more challenging. I conclude that better understanding of the processes of LEK-acquisition in these landscapes is needed in order to increase the potential for this method to provide robust data on species occupancy and population trends. Nonetheless, our findings strongly support the value of LEK for providing the critical contextual backdrop of local socio-economic drivers of landscape change and species patterns. This information is becoming ever-more relevant for tropical forest conservation in an increasingly anthropogenic world (St. John et al. 2014; Koh and Gardner 2010).

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Appendix 3.1: *Questionnaire administered to local respondents in rural Itamonte (in Portuguese)*

UNIVERSIDADE FEDERAL DE LAVRAS	PARTE 1: INFORMAÇÕES DE PARTICIPANTE NÚMERO DE INDENTIFICAÇÃO/REGISTRO: **R___**	
Data: ___/___/____ Horário: *Início:* : hs *Término:* : hs	Entrevistador: ______________	UTM X |__|__|__|__|__|__|__|\ _____ Y |__|__|__|__|__|__|__| Lugar: ____________________

Antes da Entrevista: ☐ Explicou objetivos? ☐ Obteve permissão? ☐ GPS? ☐ Gravação?

1. Informações básicas

1.1 **Nome do entrevistado:** ______________________________

1.2 **Idade:** __________

1.3 **Sexo:** ☐ Homen ☐ Mulher

1.4 **Ocupação** ______________________________

2. Origem e tempo na região

2.1 **Voce sempre morou aqui na região?** ☐ Não ☐ Sim

2.1.1 **(Se não) Há quanto tempo senhor(a) mora neste município?** _______________

2.2 **Há quanto tempo o senhor(a) mora nesta propriedade?** ____________________

Comentários *(Utilize este espaço para obter mais informações relevantes sobre residência ou ocupações anterior)*   **1**

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PARTE 2: RECONHECIMENTO DE ESPÉCIES E PERCEPÇÕES DO ESTADO DO POULAÇÃO PRESENTE E DO PASSADO (A utilização de placas)

Registro: R ____

Ref.	**Nome comun**	**Reconhec-imento de especies?**		**Como se chama?**	V = VIU; OC = OUVIU CANTOS; OF = OUVIU DE FALAR; R = RASTROS (SINAIS); AF = ARMADILHA FOTOGRAFICA	Frequencia: D = Diariamente; S = Semanalmente; M = Mensal; 6 = Semestral; A = Annual; 10 = Decadas; U = UMA VEZ	Quando foi a ultima vez?	**Percepções de mudança no status da população**		
		S/ N	Na região					Mais comun/menos comun/igual NO PASSADO	**Foi quando?** com qual idade, quantos anos atras?	Em caso afirmativo – você tem uma idéia porque voce viu mais/menos? (Refere-se a temas na parte 3)
E1	Jacuaçu									
E2	Jacupemba									
E3	Paca									
E4	Capivara									

Comentarios:

2

UNIVERSIDADE FEDERAL DE LAVRAS

PARTE 2: RECONHECIMENTO DE ESPÉCIES E PERCEPÇÕES DO ESTADO DO POULAÇÃO PRESENTE E DO PASSADO (A utilização de placas)

Registro: R ____

Ref.	**Nome comun**	**Reconhec-imento de especies?**		**Como se chama?**	V = VIU; OC = OUVIU CANTOS; OF = OUVIU DE FALAR; R = RASTROS (SINAIS); AF = ARMADILHA FOTOGRAFICA	Frequencia: D = Diariamente; S = Semanalmente; M = Mensal; 6 = Semestral; A = Annual; 10 = Decadas; U = UMA VEZ	Quando foi a ultima vez?	**Percepções de mudança no status da população**		
		S/ N	Na região					Mais comun/menos comun/igual NO PASSADO	**Foi quando?** com qual idade, quantos anos atras?	Em caso afirmativo – você tem uma idéia porque voce viu mais/menos? (Refere-se a temas na parte 3)
E5	Queixada									
E6	Caititu									
E7	Tucano – bicho verde									
E8	Tucanuçu									

Comentarios:

3

UNIVERSIDADE FEDERAL DE LAVRAS	**PARTE 2: RECONHECIMENTO DE ESPÉCIES E PERCEPÇÕES DO ESTADO DO POULAÇÃO PRESENTE E DO PASSADO (A utilização de placas)**	
		Registro: R ____

Ref.	**Nome comun**	**Reconhecimento de especies?**		**Como se chama?**	V = VIU; OC = OUVIU CANTOS; OF = OUVIU DE FALAR; R = RASTROS (SINAIS); AF = ARMADILHA FOTOGRAFICA	Frequencia: D = Diariamente; S = Semanalmente; M = Mensal; 6 = Semestral; A = Annual; 10 = Decadas; U = UMA VEZ	Quando foi a ultima vez?	**Percepções de mudança no status da população**		
		S/N	Na região					Mais comun/menos comun/igua l NO PASSADO	**Foi quando?** com qual idade, quantos anos atras?	Em caso afirmativo – você tem uma idéia porque voce viu mais/menos? (Refere-se a temas na parte 3)
E9	Quati									
E18	Lemure									
E10	Veado									
E17	Teiu									

Comentarios:

4

UNIVERSIDADE FEDERAL DE LAVRAS	**PARTE 2: RECONHECIMENTO DE ESPÉCIES E PERCEPÇÕES DO ESTADO DO POULAÇÃO PRESENTE E DO PASSADO (A utilização de placas)**	
		Registro: R ____

Ref.	**Nome comun**	**Reconhecimento de especies?**		**Como se chama?**	V = VIU; OC = OUVIU CANTOS; OF = OUVIU DE FALAR; R = RASTROS (SINAIS); AF = ARMADILHA FOTOGRAFICA	Frequencia: D = Diariamente; S = Semanalmente; M = Mensal; 6 = Semestral; A = Annual; 10 = Decadas; U = UMA VEZ	Quando foi a ultima vez?	**Percepções de mudança no status da população**		
		S/N	Na região					Mais comun/menos comun/igua l NO PASSADO	**Foi quando?** com qual idade, quantos anos atras?	Em caso afirmativo – você tem uma idéia porque voce viu mais/menos? (Refere-se a temas na parte 3)
E11	Maracajá									
E12	Juguatirica									
E13	Onça-pintada									
E14	Puma/onça-parda									

Comentarios:

5

UNIVERSIDADE FEDERAL DE LAVRAS	PARTE 2: RECONHECIMENTO DE ESPÉCIES E PERCEPÇÕES DO ESTADO DO POULAÇÃO PRESENTE E DO PASSADO (A utilização de placas)
	Registro: R ____

Ref.	Nome comun	Reconhec-imento de especies?		Como se chama?	V = VIU; OC = OUVIU CANTOS; OF = OUVIU DE FALAR; R = RASTROS (SINAIS); AF = ARMADILHA FOTOGRAFICA	Frequencia: D = Diariamente; S = Semanalmente; M = Mensal; 6 = Semestral; A = Annual; 10 = Decadas; U = UMA VEZ	Quando foi a ultima vez?	Percepções de mudança no status da população		
		S/N	Na região					Mais comun/menos comun/igual NO PASSADO	Foi quando? com qual idade, quantos anos atras?	Em caso afirmativo – você tem uma idéia porque voce viu mais/menos? (Refere-se a temas na parte 3)
E15	Bugio									
E16	Macaco-prego									

Comentarios:

6

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PARTE 3: ENTREVISTA ABERTA - EXPLORANDO MUDANÇAS

Registro: R___

Essa seção é para anotar informações importantes e citações relevantes durante uma discussão aberta.

Discussão aberta:

1. **Estilo de vida: tempo passado fora / em ambientes naturais / na floresta**
2. **Meios de vida: práticas agrícolas/trabalho/opções**
3. **Uso da terra: uso familiar histórico de floresta, restauração / desmatamento, poluição**
4. **O alimento: preferências, disponibilidade**
5. **Caça dos animais selvagens**
6. **Presença da polícia ambiental / IBAMA , conhecimento da legislação**

7

Appendix 3.2: ***Questionnaire administered to local respondents in rural Itamonte (English translation)***

UNIVERSIDADE FEDERAL DE LAVRAS	PART 1: PARTICPANT INFORMATION IDENTIFICATION NUMBER: **R**___	
Date: ___/___/____ Time: *Start:* : hrs *End*: : hrs	Interviewer: ________________	UTM X |__|__|__|__|__|__|__|__| _____ Y |__|__|__|__|__|__|__| Location: ____________________

Before the interview: ☐ Explained objectives? ☐ Obtained permission? ☐ GPS? ☐ Recording?

1. Basic Information

1.1 **Name of interviewee:** ____

1.2 **Age:** __________

1.3 **Sex:** ☐ Male ☐ Female

1.4 **Occupation** ____

2. Origin and time in region

2.1 **Have you always lived in Itamonte?** ☐ No ☐ Yes

2.1.1 **(If no) How long have you lived here in this municipality?** ________________

2.2 **How long have you lived in this property?** ________________________

Comments

(Use this space to record further information on previous locations and occupations)

1

UNIVERSIDADE FEDERAL DE LAVRAS

PART 2: SPECIES RECOGNITION AND PERCEPTIONS OF PRESENT AND PAST POPULATION (Using picture cards)

Register: R ____

Ref.	**Common Name**	**Species recognition?**		**Local name given?**	V = seen it; OC = heard calls; OF = heard others speak of it; R = physical signs; AF = camera trap	Frequency: D = Daily; S = Weekly; M = Monthly; 6 = 6-months; A = Annual; 10 = Decades; U = once	When was the last time you saw it?	**Perceptions of changes in species population status**		
		Y/N	In the region					More or less common now than in the past?	If yes - When did you perceive a change? (at what age? How many years ago?)	In the case of perceived population changes – do you have any thoughts or ideas as to why the species population has increased/decreased (Can refer to the themes in part 3)
E1										
E2										
E3										
E4										

Comments:

2

UNIVERSIDADE FEDERAL DE LAVRAS

PART 3: OPEN INTERVIEW – EXPLORING CHANGES

Register: R___

This section is to note important information and relevant quotes during an open discussion based around some key topics/ themes.

Open discussion themes:

1. **Lifestyle: time spent outdoors/in natural environments/in forests**
2. **Livelihoods: agricultural practices/work opportunities/options**
3. **Land use: historical forest use/restoration/deforestation/pollution**
4. **Diet: preferences, availability**
5. **Hunting**
6. **Environmental police/IBAMA presence/ knowledge/awareness of environmental laws and legislation**

7

CONCLUDING REMARKS

Chapter 5 Concluding Remarks

This thesis embraced a diverse set of themes in different environments in Brazil (Amazon and Atlantic forests) and at different spatial scales from millions of square kilometres to a single municipality (analogous to a US County). Nonetheless an overarching common thread runs throughout: the value of alternative and innovative research methods for eliciting important conservation-relevant information in tropical forest contexts. The findings presented here demonstrate that valuable and novel insights and data exist in places we may not expect; information that would remain untapped and unknown if we relied solely on traditional data sources and methods. The following outlines the key findings and conservation messages obtained through this work, and the broader implications for the continued advancement of conservation ideologies and method application.

5.1 Key findings

5.1.1 Novel finding: Large-scale harvest and trade of wild-meat in Amazonia

The evidence of extensive illegal harvest and trade of Amazonian wildlife for human consumption is of huge conservation concern. Insights into wide-scale caiman hunting and trade (Chapter 2) and the scale of harvest and trade of multiple species detected in enforcement records (Chapter 3) are congruent with recent localized studies reporting high levels of consumption of wild-meat in urban Amazonia (Parry, Barlow, and Pereira 2014; van Vliet et al. 2014); yet, to my knowledge, this research is the first to explore the spatial extent and dynamics of harvest activities outside of the rural subsistence paradigm at a regional scale in the Neotropics.

The deleterious impacts of commercial harvest on wildlife populations have been well documented (Abernethy et al., 2013; Brashares et al., 2004; Da Silveira & Thorbjarnarson, 1999; Strandby & Olsen, 2008; van Vliet et al., 2014), particularly in

Africa where it has been asserted that demand for wild-meat in urban markets (as opposed to rural subsistence) is the key driver of the African 'bushmeat crisis' (Lindsey et al. 2013; Nasi et al. 2008; Wilkie and Carpenter 1999; Wright et al. 2007). In the Neotropics the impacts of rural subsistence hunting are better known (Bizri, 2014; Peres, 2001; Peres & Nascimento, 2006), with reports of significant basin-wide declines in species populations in intensely hunted sites (Peres & Palacios, 2007). However the study by Parry & Peres (2015) estimated severe species depletion shadows extending well over 100km from urban centres, suggesting that commercial trade and urban consumption of wild-meat in the Neotropics is having far-reaching impacts on forest wildlife.

My thesis revealed new lines of evidence to support these estimates: the species data in federal environmental agency enforcement records revealed extremely high levels of commercial trade in various species, notably river turtles (*Podocnemis* spp.) and caiman (*Melanosuchus* and *Caiman* spp.), with indications of long-distance trade (further supported by expert knowledge on caimans in Chapter 2). Long-distance trade networks can be indicative of local species depletion, as people travel further (despite increasing costs) to meet demand (Macdonald et al. 2012), suggesting that species populations are likely already being impacted. Continuing urbanisation and rapidly increasing populations in Amazonia suggest that pressure on wildlife populations is projected to increase, and that the Neotropics could be heading towards its own 'bushmeat crisis' (Parry et al., 2014).

5.1.2 Novel finding: A developing enforcement vacuum

This thesis also showed how the Neotropical bushmeat crisis could be being exacerbated by changes in enforcement. By assessing enforcement capacity and wildlife crime detection rates (Chapter 3) there is robust evidence of inadequate federal governance in Brazilian Amazonia, which has significant implications on the activities themselves (i.e.

the dynamics and rate of illegal wild-meat trade and harvest) (Leader-Williams and Milner-Gulland 1993), and any conservation strategy that seeks to mitigate their impacts (Rowcliffe, de Merode, and Cowlishaw 2004). The challenges of effective enforcement in developing tropical nations have been discussed in the conservation and governance literature (Keane et al. 2008; Smith et al. 2003; Yu, Levi, and Shepard 2010). Limited financial and human resources, inaccessible vast forested areas and corruption are cited as common barriers to effective enforcement across the tropics (Ameyaw, Arts, and Wals 2016; Contreras-Hermosilla 2002; Gore, Ratsimbazafy, and Lute 2013). However empirical research on the efficacy of enforcement against illegal wildlife harvest tends to be restricted to within protected areas (Hilborn et al. 2006; Jachmann 2008; Nolte et al. 2013).

The findings of inadequate institutional capacity and low detection rates in this thesis (Chapter 3) offer important new perspectives of governance outside of protected areas. The results indicate that the common barriers in tropical federal governance manifest in different ways depending on the type of illegal forest activity; they are far more prohibitive when confronted with ostensibly subtler and more cryptic extraction activities such as wildlife harvest, than for the easier-to-detect issue of deforestation (Bruner et al., 2001; Peres, Barlow, & Laurance, 2006). Effective enforcement (which includes detection *and* punishment) is fundamental for compliance of laws and regulations (Hansen 2011; Heyes 2000; Aidan Keane et al. 2008), and without it even the most well-designed legislation becomes ineffective, particularly when the activities are driven by poverty or ingrained in cultural/societal norms (Morsello et al. 2015; Parry, Barlow, and Pereira 2014).

Is the level of environmental governance in Brazil getting worse? Perhaps counter intuitively, one of the most concerning findings is the marked drop in wildlife crime reporting rates in Amazonia as this is most parsimoniously attributed to the decentralization of enforcement responsibility to local and state level rather than a reduction *per se* in wildlife crimes. At the time of the study, there was little evidence to support that the worryingly low detection and enforcement of Amazonian wildlife harvest and trade will improve under the new regulatory framework. In fact these changes may promote something of an enforcement vacuum, with local and state agencies ill-prepared for their new responsibility (Monteiro da Silva and Bernard 2015). Research shows that the success of decentralization is dependent on appropriate design and implementation; the following conditions are considered essential: (i) a functioning local democracy, (ii) adequate fiscal autonomy for local governments, and (iii) technical expertise among local and national government officials (Brixiova 2008). As these conditions are rarely met in developing countries, the outcome of decentralization could easily result in lower levels of enforcement than in an inefficient central bureaucracy (Brixiova 2008).

More importantly, we must consider the wider political context in Brazil within which this thesis was researched and written. We find ourselves in politically tumultuous times; a surge in right-wing conservatism is occurring across the globe and this may have serious consequences for the natural environment. Hard-fought legislation and international climate agreements (particularly the recent United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement) may be under threat; the current head of the US Environmental Protection Agency recently claimed that carbon dioxide emissions are not a "primary contributor" to global warming (Johnston 2017), contributing to fears that the US will pull out of climate change agreements. Brazil, one of the world' mega-diversity nations, is no exception, and perhaps offers one of the most dramatic political changes in recent years. Amidst a pervasive government corruption scandal, 2016 saw the impeachment of the worker's party leader President Dilma Rouseff, allowing their previous coalition partner, the Brazilian Democratic Movement Party (PMDB) to seize power. The new president Temer has been quick to usher in a host of privatization and austerity measures, including a 20-year freeze on spending on social and welfare services; a move that has been described by a United Nations official as the most socially regressive austerity package in the world, and has been widely condemned as an attack on poor people (Watts 2016). In November 2016 over 20 legislative proposals were in circulation in the Brazilian Congress to weaken regulations governing activities such as building roads and hydroelectric dams or expanding agricultural businesses – in the Amazon and beyond (Tollefson 2016). This follows on from a near 36% increase in Amazonian forest loss since 2012, as Brazil struggled with the worst recession in decades and began prioritising economic development over environmental stewardship (Bernard, Penna, and Araújo 2014; Ferreira et al. 2014). In this case (and many others) economic development is given even greater priority over the natural environment and social justice for the most vulnerable people within society.

Against this backdrop of social, economic and political turmoil, the inadequate governance for hard-to-detect, yet widespread wildlife harvest is extremely vulnerable to further deterioration; indeed, Brazil's environmental agency has already experienced budget cuts (Monteiro da Silva and Bernard 2015). Perhaps more importantly the implications of social and welfare austerity measures may lead to increase in illegal harvesting rates, driven by increasing poverty and food insecurity (Brashares et al. 2011; Duffy et al. 2016; Parry, Barlow, and Pereira 2014).

5.2 Alternatives to top-down governance in tropical conservation

Where does this leave us as conservation scientists or the conservation movement? The vulnerability of environmental policy and governance to the changing political and economic landscape weakens traditional top-down approaches, and instead places more emphasis on the need for local-level strategies and pragmatic low-cost monitoring methods to address conservation problems. This thesis tested the use of two such potential methods: expert elicitation and Local Ecological Knowledge (LEK).

5.2.1 Reserve managers: an underused knowledge resource?

The insights gained on the complex dynamics of caiman harvest and trade from reserve managers and practitioners in Chapter 2 supports the potential for expert knowledge elicitation as a low-cost monitoring framework in conservation practice and planning (Laurance et al. 2012). Monitoring and evaluation has been pushed to the forefront of conservation priorities in recent years (Ervin 2003; Hockings 2003), with growing recognition among practitioners and scholars that good project management is intrinsically linked to well-designed monitoring and evaluation systems (Stem et al. 2005). The use of on-the-ground expertise has been increasingly recognised as a valuable method for identifying priorities and judgements on pragmatic implementation of conservation plans, and appraising the success of reserve management (Cowling et al. 2003; Laurance et al. 2012; Chapter 2). Such insights may often be otherwise unobtainable if relying solely on traditional ecological site data (Cowling et al. 2003).

The drawbacks of using on-the-ground knowledge are generally stipulated to be the inevitable inherent biases associated with experts' uneven knowledge of regions and taxa, as well as biases attributed to their personal experiences with regards to management practices (Cowling et al. 2003; Davis and Wagner 2003; Hagerman and Satterfield 2013). With this in mind I impress the need for rigorous method design and implementation when using expert knowledge to inform conservation strategy. Acknowledging and understanding likely sources of bias is critical to ensure pragmatic and effective interpretation of the resulting information; such considerations must be incorporated in initial study design and carefully examined before decisions are taken on management strategy and conservation policy.

It is also important to note the distinction between scientists or researchers versus local reserve managers. Often scientists may be somewhat transient visitors, perhaps foreigners – like myself, with distinct experiences and perspectives formed outside of the local context in which they now work. Local reserve managers on the other hand, perhaps more likely be native to the local area, may have unique understanding of more complex social and cultural dynamics to inform their judgements.

Another important consideration of expert approaches as reliable monitoring tools is the apparent high turnover of staff in reserve managers and environmental agents, especially in remote and/or dangerous frontier areas. This issue is occasionally mentioned in reports of park management effectiveness and discussions of governance (Ames and Keck 1998; Lindsey et al. 2013), yet is not empirically measured. As such it is difficult to assert to what extent high staff turnover may impact the potential for eliciting meaningful long-term trends and monitoring data. Low salaries (perhaps further constrained by limited budgets) along with the often inherent isolation of working in remote conservation areas are likely drivers of personnel turnover; but there is also a far more ominous factor – that of personal safety. Following last year's high-profile murder of Berta Cáceres, a human rights activist and prominent environmentalist in Honduras, the dangers of protecting the environment from powerful political and business interests have been sharply exposed. In 2015, Brazil recorded more killings of land and environmental defenders than any previous year (Global Witness 2016), the vast majority of which occurred at the frontiers of agricultural lands and tropical forests in Amazonia. Not just activists have been targeted; Luiz Alberto Araijo, a government official working as the secretary for the environment in Altamira, Pará was shot and killed outside his home last year (Sandy 2016). Threats and violence towards reserve managers is a sad reality; something I learned while visiting a federal Ecological Station in Jarí, in the state of Pará, where amidst local protests against logging encroachment, the reserve manager told disturbing accounts of threats to himself and his family.

5.2.2 Persistence of Local Ecological Knowledge in transitioning landscapes – novel insights and valuable monitoring tool

LEK has been gaining recognition among conservationists as a valuable and cost-effective source of information for species monitoring and conservation planning (Anadón et al. 2009; Brook and McLachlan 2008; Gandiwa 2012). The information obtained can offer much-needed social context and nuance to landscape change dynamics and species responses (Chapter 4). This thesis offers important novel perspectives for the application of LEK, demonstrating that the approach can bear valuable insights even in a dynamic severely-modified landscape (Chapter 4), where it is assumed that the loss of traditional lifestyles leads to a loss of meaningful local knowledge (Iniesta-Arandia et al. 2014). The local knowledge of inhabitants of Itamonte showed that LEK can still be useful for detecting species trends and potential species recovery in transitioning landscapes, notably for adaptive generalist species such as a guan, *Penelope obscura.* Indeed, for this species our data suggest the Birdlife species factsheet may need to be revised, as it is listed as having a "*population trend [that] appears to be decreasing*" (Birdlife International 2017). Thus, although hugely dependent on careful and apposite species selection, the method could prove a valuable and much-needed tool to rapidly assess and monitor spatial recovery patterns within the novel ecosystems and dynamic environments of transitioning landscapes (Ewers and Didham 2006; Gardner et al. 2009).

The LEK elicited in the Brazilian Atlantic Forest areas of Itamonte also revealed that despite dramatic social and cultural changes to the rural lifestyle and decreased reliance on natural resources, illegal hunting still exists, even with perceptions of a tangible enforcement presence. Although likely to be occurring at much lower rates than in Amazonia, and defined by different drivers (i.e. sport hunting, cultural pastime), the existence of hunting in such a fragmented and threatened biome could still pose a significant threat to local species that are already vulnerable to habitat loss and edge effects (Arroyo-Rodríguez and Dias 2010; Cavada et al. 2016; Cullen et al. 2000). Management strategies in these landscapes cannot afford to overlook hunting activities, and must make efforts to understand their dynamics and drivers and incorporate these realities into management design and implementation (Gama et al. 2016).

5.3 LEK as a strategy for capacity building in transitioning landscapes

A significant advantage of utilising LEK is that it offers a means of engaging local actors and communities within conservation research and planning, which can increase success rates of implemented management strategies (Campos-Silva and Peres 2016; Walters et al. 2015). Such engagement is even more important in transitioning landscapes, where decreased contact with natural habitats erodes forest values, and can facilitate further landscape degradation (Torres et al. 2016). The findings of meaningful LEK persisting in these dynamic landscapes offer a potential positive feedback loop for conservationists, in that they can utilise LEK to elicit pertinent information, while also building capacity for local engagement in conservation planning. Capacity building is a key component for generating positive outcomes in conservation initiatives and can significantly influence attitudes of local people (Brooks, Waylen, and Mulder 2013), which may in turn foster increased LEK acquisition in local communities.

5.4 The Atlantic forest: a laboratory for examining the future of tropical forests

The persistence of LEK in severely modified transitioning landscapes such as the Atlantic forest validates the potential of utilising such methods in comparative research with other tropical contexts, particularly those have until recently experienced low levels of human occupation, thus helping to predict future scenarios and challenges (Gardner et al. 2010). The rapid social and environmental changes occurring in the Amazon (Aide and Grau 2004) and throughout the tropics means that conservationists will increasingly need to understand the long-term consequences of human disturbance patterns (Koh and Gardner 2010; Peres et al. 2010). Identifying useful methods and understanding their limitations within these severely modified landscapes will become ever more important as the world is increasingly shaped by human activities.

5.5 Further research priorities

Inevitably, this work has certain limitations that need to be acknowledged. The broad spatial scale of the study and the limited time period available means that the conclusions drawn are based on correlations. Although their interpretation was informed by and builds upon the body of contemporary knowledge and research, they would be supported by experimental or additional validation. In regards to Chapter 2 site-level field validation of caiman harvest and consumption rates is clearly needed, and as I recommended in that chapter, an important priority should be to characterise trade drivers and dynamics through a commodity-chain approach (Bowen-Jones, Brown, and Robinson 2003; Cowlishaw, Mendelson, and Rowcliffe 2005; Mendelson, Cowlishaw, and Marcus 2003). In Chapter 3 I note the absence of an institutional perspective on the realities of governance (i.e. IBAMA agents/state environmental police). Indeed, practical insights from environmental agents on decision-making and priorities would facilitate pragmatic discussions on solutions and improvements for governance. Similarly, I accept the clear need to validate the species occupancy data elicited in Chapter 4, however I nonetheless emphasise the value of methods such as LEK as one of the only means to explore and detect past trends, when faced with a reality of scant long-term, systematic data on species distribution and abundance (Gardner et al. 2007; McPherson and Myers 2009).

5.6 The future of conservation: Towards pragmatism, adaptive strategies and transdiciplinarity

It is clear that the challenge of preserving tropical biodiversity in this anthropogenic world is a daunting and difficult task. Even more so if attempting to conserve nature in a way that respects and indeed seeks to promote the rights, dignity and welfare of local people. This is an especially important consideration for conservation research and interventions affecting the historically disadvantaged and marginalized communities living in and around tropical forests. The complexity induced by interactions between human and natural systems requires conservationists to simultaneously navigate multiple world views, stakeholder values, cross-scale interactions and uncertainty (Cundill, Fabricius, and Marti 2005), and must do so with limited resources. Despite the inherent inter-disciplinarity within conservation ideologies, compartmentalization of disciplines continues to be an impediment to effective conservation (Reyers et al. 2010), and there remains a shortage of information and consensus on integrating concepts, methodologies, and techniques (Cundill, Fabricius, and Marti 2005). Moving towards effective integration requires trans-disciplinarity, not only forging linkages between scientific disciplines, but also acknowledging complexity and the value of different knowledge spheres (St. John et al. 2014; Reyers et al. 2010). Methods and approaches must be pragmatic and adaptive depending on the context of the problem being addressed, not on the basis of ideological preference (Robinson 2011). In other words, ideally the problem, not the discipline should define the tools of study (Newing 2010).

This need for context-dependent pragmatism will result in inevitable trade-offs, as evidenced by the varying techniques and approaches utilised in this thesis. Understanding the value, and perhaps more importantly, the limitations of different methods is critical for conservationists to select the most appropriate approach and sensibly interpret the resulting data, and in turn devise effective management strategies. Methods developed within certain disciplines have inherent strengths and weaknesses. Typical quantitative methods can reveal correlations and cause-effect relationships, identifying statistically significant differences within very focused questions (Chiarello 2000; Michalski and Peres 2007; Starr et al. 2011), while qualitative data can offer an overview of a situation, disentangling complexities and providing in-depth understanding of different perspectives (Ball and Brancalion 2016; Drury, Homewood, and Randall 2011; Newing 2010). Similarly the scale at which we approach a problem will impact the implications and application of the resulting data within conservation strategy design. Broad scale approaches, as in those used in Chapters 2 and 3 to explore wild-meat in Amazonia, offer valuable means of identifying coarse regional patterns, revealing spatial priorities and the general extent of an issue. However, focusing on smaller scales, at a local or landscape level, allows for better understanding of the processes behind the patterns, revealing important nuance and potential heterogeneity, as demonstrated by the use of LEK in Chapter 4.

The tensions between research approaches and the selected spatial and temporal scale at which they are employed are not irreconcilable; on the contrary they are both necessary and can be complimentary to each other. The rise in mixed-method approaches, that aim to benefit from the differing contributions of quantitative and qualitative methods, shows that conservationists continue to adapt to the increasing complexity of contemporary environmental and social challenges (Ameyaw, Arts, and Wals 2016; Coad et al. 2013; Van Vliet et al. 2015). However, we must be rigorous in our method development, validation and application (St. John et al. 2014). Hence we must seek to understand how and at what stage complementarity is best achieved within the practical limitations of research, while ensuring strong conceptual and empirical foundations for our decisions (Hattam et al. 2015; St. John et al. 2014; A Keane 2013).

For researchers, the different conceptual and practical approaches needed to address dynamic complex problems can be confusing and overwhelming. From my personal experiences through the writing of this thesis I can attest to this. Like the majority of conservationists, I had been trained predominantly in the natural sciences, and grappling with different epistemologies (particularly qualitative social data) was an eye-opening experience. The process of formulating and building this work has been a sometimes arduous, yet transformative journey, challenging my own preconceptions and disciplinary biases. Yet ultimately it has been an incredibly rewarding experience that I feel has made me a better researcher.

5.7 Conclusion

This thesis has attempted to improve understanding of the tools available to researchers in the context of complex socio-environmental conservation issues in the tropics. This has been achieved by demonstrating the potential and limitations of alternative methods and data sources outside of traditional ecological approaches. In doing so, I have uncovered important conservation issues in Brazil, including the spatial extent of wild-meat harvest and trade and the difficulties in governance in Amazonia, and the value of local knowledge to inform conservation in transitioning landscapes in the Atlantic Forest. I have highlighted the need to be critical in our development and application of methods and cautious in our interpretations, recognizing the challenges that come from integrating different ways of thinking about and approaching a complex problem at different scales. The immense challenge of safeguarding tropical biodiversity in the Anthropocene requires us to be adaptive and pragmatic, remaining open to seize novel and seemingly unconventional opportunities to inform our decision-making and design effective and ethical conservation strategies.

5.8 References

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