# Integrating local knowledge into wildlife population monitoring

Stephanie Marie Brittain

Jesus College

University of Oxford

A thesis submitted for the degree of

Doctor of Philosophy

Trinity term 2019

# Abstract

Gathering species population data from local people is growing in popularity in conservation, as a cost-effective way to rapidly collect information over spatiotemporal scales that are not always feasible using conventional surveys. However, few studies exist that assess the bias when incorporating local ecological knowledge (LEK) into wildlife population monitoring. In this thesis, I use a mixed-method approach to gain a better understanding of the uncertainty that can affect observational data and explore how local ecological knowledge can be better incorporated into wildlife population monitoring, using two wild meat hunting villages adjacent to the Dja Faunal Reserve in Cameroon as a case study.

I explore patterns of hunting offtake and intensity within a socioecological system framework, to identify the drivers of hunting and threats to hunted species in both hunting systems. I find clear village level differences in species depletion, as a result of differing geographical characteristics and hunter-level methods and motivations to hunt. I then triangulate estimates of species occupancy obtained from daily hunter diaries, seasonal interviews and camera traps, to better understands the sources of uncertainty when using each monitoring method. Estimates from LEK-informed methods were broadly comparable with camera trap data at the village level, but with species level differences. I explore the use of modern expert elicitation methods as a tool to better understand uncertainty around estimates of species occupancy and density from local people. Gender plays an important predictive role in the type of knowledge held. Further, those predicted to be the most knowledgeable by their peers (i.e. local experts) do not always provide the most robust estimates. Finally, power analysis that account for species detectability reveal that the cost

and effort required to ensure power to detect trends in species occupancy over time is often prohibitive.

The thesis highlights the value that LEK-informed methods have for providing monitoring data on species hunted for wild meat, and how an improved understanding of LEK-informed data and integration into wildlife population monitoring can result in more ethical and just conservation efforts. However, conservation actors must account for species detectability when designing monitoring programmes and consider whether they have the logistical and financial resources required for effective and sustainable monitoring of trends over time.

Word count: 51969

Acknowledgements

I'm so lucky to have a great number of brilliant and supportive people around me, without whom I would not have been able to produce this thesis. I am so eternally grateful for the unflappable support of my supervisors, EJ Milner-Gulland and Marcus Rowcliffe. I've had your backing since my MSc and I've felt wholeheartedly supported by your guidance and encouragement. EJ, you have the miraculous ability to calm even your most anxious students. The few times I've entered your office feeling like I can't cope, I've left feeling reassured. I cannot thank you enough for this.

I want to thank the fantastic support provided by ZSL Cameroon and the Conservation Programmes team in London. Paul DeOrnellas, thank you for your ever-friendly support, your advice and input was always really valued. David Olson and Andrew Fowler, without your logistical help my fieldwork would not happened at all. I want to thank the amazing team of drivers at ZSL Cameroon, especially Petit Jean. Due to your pretty impressive off-road driving skills, the research team were always in safe hands. I miss our long drives listening to your eclectic music collection. Mado, we've worked together at ZSL Cameroon since I first started working there in 2012. We've had some crazy fieldwork experiences together and I'm so happy to count you as one of my friends. Tom, my Cameroon fieldwork brother. Despite your dubious company, thrashing you at pool on a regular basis never failed to cheer me up. Cameroon isn't the same without you.

I never imagined when starting this work that I would be made to feel so welcome in the villages where I worked. To the village chiefs; you not only allowed me to work in your villages, but also hosted us so generously with your families. I was so humbled by the incredible warmth and generosity of the inhabitants of both study villages. Your enthusiasm for the research and willingness to share your great wealth of knowledge astounded me. I want to give particular mention to Archile and Venant, the key contacts in both villages. Without your support and friendship, great chunks of this work would not have materialised. Mama Celine, Gui, Mama Rachel, Roger, Papa Roger and Esther, Archile and Stephanie, I can't wait to come back to say hello and spend long evenings talking about everything and nothing.

Of course, this would never have happened without the support from the most wonderful research dream team, Fabrice, Tibo, Joel and Irene. Fabrice and Tibo we have had many bonding experiences in Cameroon, including that "short-cut" we took while setting camera traps in the forest, a 2-hour bike ride that became a 9.5 hour bike ride in the rain, as well as passing many hours in the villages playing board games. You started out as colleagues and finished up as friends. Without your humour and companionship, fieldwork would not have been the same.

While back in the UK, I'm privileged to work with a diverse, kind, inspiring and brilliant group of people at ICCS, who are both fantastic companions and friends. Having a group like you to share in the highs and lows of research really made this a lot easier. Harriett Ibbett, Leejiah Dorward, I'm so lucky to be able to work with friends like you who I can share my (sometimes ridiculous) thoughts with and feel totally comfortable in doing so. Victoria Griffiths and Sofia Castello Y Tickell, thank you for your emotional support at difficult times, and Victoria again for organising my social life (and everyone else in ICCS).

To my parents, you've always been fully supportive of me, not just during my PhD, but in life. Thank you for proof-reading pages of my PhD and making sure I'm eating and generally looking after myself. To my family-in-law, thank you for your patience and understanding during the hours I spent locked away working in the summerhouse. Andrew, I wish you were here to see me hand this in, but I know you'd be proud of me and that makes me so happy.

To Roxy, Rosie, Lydia, Emily, Marie, Hannah and Laura, the best friends I could ask for. Thank you for your support over the years, you don't have to hear about this anymore now!!

To Paul, my husband (I still can't get used to calling you that)! I owe you everything. Over the past 4 years you have supported me, listen to my concerns, talked through ideas and generally been a rock. Thank god you are also doing a PhD and understand what it can be like, but I'm certain I still bored you to death at times. Now let's go for a drink and try to remember what life without a PhD to finish is like.

## Funders

This work was funded by the Natural Environment Research Council CASE (NERC CASE), under the project "Robust wildlife population monitoring under challenging conditions". Additional support was awarded by the Royal Geographical Society Slawson Award, the Jana Robyst Fund and the Rufford Grant.

# Author Contributions

**Chapter 1:** Introduction

This chapter is entirely my own work and has been reviewed by E.J. Milner-Gulland, and Marcus Rowcliffe.

**Chapter 2:** Theoretical background and case study

This chapter is entirely my own work and has been reviewed by E.J. Milner-Gulland, and Marcus Rowcliffe.

**Chapter 3:** Using socio-ecological systems thinking to identify wild meat hunting motivations and intensity in Cameroon

The ideas and research questions were conceived by myself, E.J. Milner-Gulland, and Marcus Rowcliffe. I carried out the data collection and analyses and wrote the chapter. The chapter was reviewed by E.J. Milner-Gulland and Marcus Rowcliffe.

Author list for subsequent publications: Brittain S., Kamonge-Tagne T., Kentatchime F., Rowcliffe M., Milner-Gulland E.J

**Chapter 4:** Comparing interview methods with camera trap data to inform occupancy models of hunted mammals in forest habitats

The ideas and research questions were conceived by myself, E.J. Milner-Gulland, and Marcus Rowcliffe. I carried out the data collection and analyses and wrote the chapter. The chapter was reviewed by E.J. Milner-Gulland and Marcus Rowcliffe.

Author list for subsequent publications: Brittain S., Rowcliffe M., Kentatchime F., Kamonge-Tagne T., Tudge S., Tsambang J., Mbouwe I., Milner-Gulland E.J

**Chapter 5:** Eliciting expert knowledge to inform mammal monitoring in Cameroon

The ideas and research questions were conceived by myself, E.J. Milner-Gulland, and Marcus Rowcliffe. I carried out the data collection and analyses and wrote the chapter. The chapter was reviewed by E.J. Milner-Gulland and Marcus Rowcliffe.

Author list for subsequent publications: Brittain S., Rowcliffe M., Kentatchime F., Kamonge-Tagne T., Milner-Gulland E.J

**Chapter 6:** Power to the people: analysis of occupancy models informed by observational data from local knowledge

The ideas and research questions were conceived by myself, E.J. Milner-Gulland, and Marcus Rowcliffe. I carried out the data collection and analyses and wrote the chapter. The chapter was reviewed by E.J. Milner-Gulland and Marcus Rowcliffe.

Author list for subsequent publications: Brittain S., Earle, S., Rowcliffe M., Milner-Gulland E.J

**Chapter 7:** Discussion

This chapter is entirely my own work and has been reviewed by E.J. Milner-Gulland, and Marcus Rowcliffe.

# Table of Contents

# List of figures

# List of tables

# Acronyms and abbreviations

| | |
|---|---|
| AIC | Akaike Information Criterion |
| CBD | Convention on Biological Diversity |
| CUREC | Central University Research Ethics Committee |
| DFR | Dja Faunal Reserve |
| FAO | Food and Agriculture Association |
| GLM | Generalised Linear Model |
| IPBES | Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services |
| IUCN | International Union for Conservation of Nature |
| LEK | LEK Local Ecological Knowledge |
| MINFOF | Ministere des Forets et de la Faune |
| NBSAP | National Biodiversity Strategy and Action Plan |
| NGO | Non-governmental organisation |
| NIAP | National Ivory Action Plan |
| NTFP | Non-timber forest product |
| PA | Protected area |
| PC | Principal Component |
| PCA | Principal Component Analysis |
| PLEO | Pooling local expert opinions |
| REDD+ | Reduce Emissions from Deforestation and |
| SES | Socioecological system |
| U.N. | United Nations |

# Chapter 1

## Introduction

(Waiting to cross the Dja river by to start fieldwork)

## 1.1 Background

Growing threats to biodiversity in the tropics mean there is an increasing need for effective monitoring that balances scientific rigor with practical feasibility (Rist et al. 2010). Protein from forest wildlife is crucial to rural food security and livelihoods across the tropics (Nasi & van Vliet, 2011) and local extirpation of hunted species is widespread, with West and Central Africa particularly hard hit (Milner-Gulland & Bennett, 2003). In the Congo Basin, rural wild meat consumption was estimated at between 14.6 to 97.6 kg/capita/year in 2004 (Starkey, 2004), and total offtake in 2010 was estimated at 4.5 million tonnes (Nasi & van Vliet, 2011). Aside from the serious threat to local livelihoods and food security that overhunting can present (van Vliet, 2011) the subsequent shift in mammal community composition can alter ecological processes, forest structure and composition (Rosin & Poulsen, 2016), driving long-term ecological damage to tropical forests (e.g. Poulsen et al. 2009; Dirzo et al. 2014).

Wildlife population monitoring is defined as the process of gathering information about variables that characterise a system state, for the purpose of assessing system state and tracking changes over time (Yoccoz et al. 2001). It is an essential tool that allows researchers to identify trends in species populations (Parry & Peres, 2015) to assess a species' status (Witmer, 2005; Joseph et al. 2006; IUCN, 2012) the impact of threats (Kumpel et al. 2009) or measure success in conservation programmes (Lindenmayer et al. 2012).

While robust wildlife population monitoring is essential for effective conservation, monitoring faces substantial challenges. Conventional monitoring methods can be expensive, time consuming and require specialised training or technology which may render monitoring over large spatiotemporal scales a serious challenge, especially for budget restricted projects. These issues are worsened in remote or difficult habitats such as dense forest, where detectability is low and terrain is difficult to traverse (Moller et al. 2004).

Tension exists between monitoring methods that prioritize long-term practicality, and those that prioritize precision. One potential solution is to integrate local ecological knowledge (LEK) (Huntingdon, 2000) into wildlife population monitoring. Gathering species population data from LEK is growing in popularity in conservation, as a cost-effective way to rapidly collect information over spatiotemporal scales that are not always feasible using conventional surveys. Further, increased participation of local people results in more rapid decision making and sustainable decisions which have some local ownership (Danielsen et al. 2010), as interviews help to develop dialogue between local people and conservation actors (Mohd-Azlan et al. 2013). Nonetheless, there is concern about the accuracy (the degree to which the data agrees with the 'truth') and precision (the degree of uncertainty around the estimates) of LEK as a method of collecting biodiversity monitoring information.

Concerns about the accuracy and precision of wildlife population estimates are not confined to LEK-informed methods; all monitoring methods suffer from a degree of bias and imprecision, and in particular suffer from imperfect detection. Failure to account for imperfect detectability in wildlife population monitoring is indicative of a broader lack of focus on uncertainty in ecology and conservation, which, if ignored, can result in misleading conclusions being drawn about the suitable actions required for species conservation. Although the need to understand the determinants of precision and accuracy of wildlife population estimates is increasingly recognised (Elphick, 2008; Singh & Milner-Gulland,

2011) there remains a lack of understanding of the effect of observational error on population monitoring (Nuno et al. 2013). In particular, and given the increasing use of LEK-informed data for conservation, studies are needed to investigate the effect of observational error on the effectiveness of interview data from local people (Danielsen et al. 2005), and to better understand if there are certain observer characteristics that permit someone to be a more robust participant than others (Mohd-Azlan et al. 2013).

Recently, interview-based surveys have been combined with occupancy analysis, which potentially provides an unbiased estimation of species distribution and relative abundance through models that account for imperfect detection (MacKenzie et al. 2002). However, very few studies exist that assess the bias of LEK when combined with occupancy analysis, and none so in the context of wild meat hunting systems, where the method has broad potential applications. Further, as observational data (defined here as data collected from people, based on observations they make during their daily lives) is a widely used data type in ecology and conservation, a better understanding of the biases that affect observational data and how to address them is also beneficial for ecology and conservation more broadly.

## 1.2 Aims and objectives

The overall aim of this research is to explore how local ecological knowledge (LEK) can be better incorporated into wildlife population monitoring, with a strong focus on understanding the multifarious sources of uncertainty that can affect observational data obtained from local people. The results provide insights into the robustness of data obtained from local people compared to data obtained from camera traps and an understanding of how knowledge elicitation protocols can be applied to enable more robust estimates of species occupancy and density from local people, before identifying the optimal survey design and the costs associated with monitoring with each method. The context for this thesis is developing methods to monitor mammals hunted for wild meat in forest habitats, and therefore I also explore the threats to wild mammals and the sustainability of the hunting system from the perspective of local people. The study was undertaken in two forest dependent villages adjacent to the Dja Faunal Reserve in Cameroon. The main objectives are to:

1. Identify the individual and village level drivers of hunting, and the threats to hunted species in the Dja Region
2. Investigate how observational and camera trap methods, when combined with occupancy analysis, are affected by different types of uncertainty within the case study
3. Explore the trade-offs between cost effectiveness and power to detect change that affect observational and camera trap methods when combined with occupancy analysis
4. Identify barriers to and the potential for the successful integration of local ecological knowledge into wildlife population monitoring in the Dja region and more broadly.

## 1.3 Thesis outline

This thesis is divided into five parts: i) background information; ii) identifying threats to biodiversity in the study villages; iii) assessment of the trade-offs and biases when incorporating local ecological knowledge into wildlife population monitoring; iv) exploring the practical application and sustainability of monitoring methods informed by local ecological knowledge; and v) synthesis and application of the research (figure 1-1).

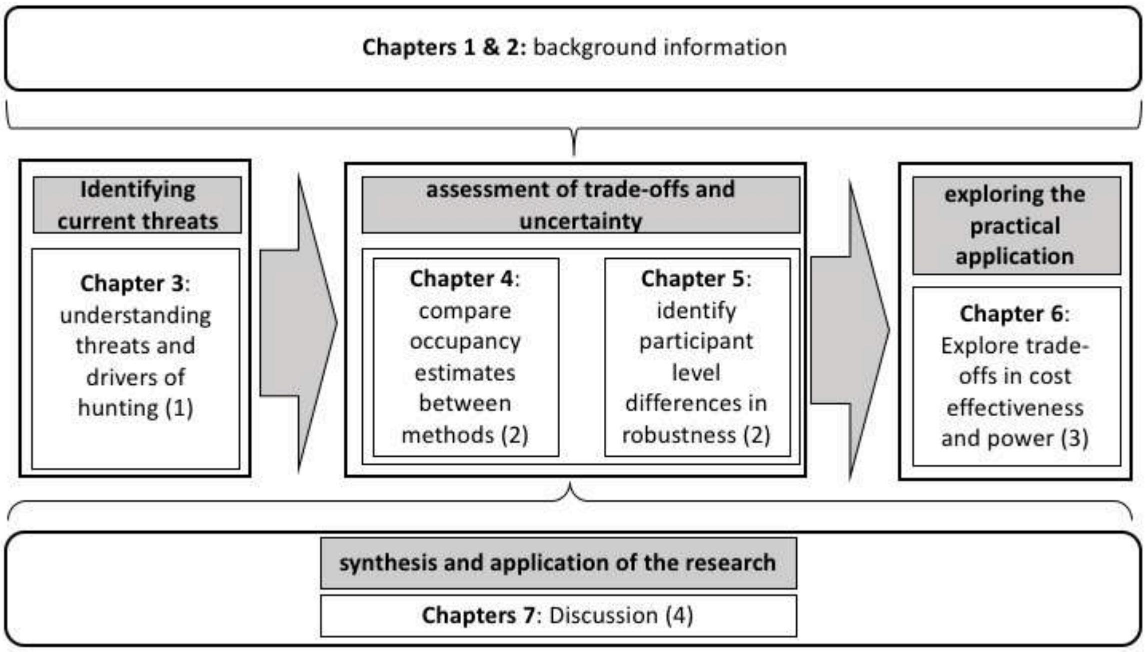



Figure 1-1: Conceptual framework of this thesis, highlighting how each chapter fits in with the separate thesis parts (grey boxes). Numbers in brackets relate to the research objectives addressed in each chapter.

In addition to this first introductory chapter, the thesis comprises a further six chapters and is structured as follows:

**Chapter 2:**

Here I set out the contextual background in which my research takes place. The chapter introduces Cameroon and the selected case studies villages used throughout the thesis. The chapter concludes with an explanation of why this case study area was selected and how my research contributes to ongoing research on wildlife population monitoring and local engagement in Cameroon.

Chapter 3:

There is a lack of data on wild meat hunting and offtake in my study site, a little-studied yet highly biodiverse area of Cameroon. Further, current offtake estimation methods do not take spatial variation in offtake into account. Using a mixed-methods approach, I use hunting offtake data obtained from daily diaries with hunters to understand which species are hunted and where, and explore links between hunter effort and the fate of the wild meat (e.g. eaten or sold, in the village or taken to market). I complement this with qualitative data on the drivers of wild meat hunting, changing hunting methods and livelihood pressures which may affect the observed patterns of wild meat hunting.

Chapter 4:
Here I contribute to the currently scant literature that directly compares wildlife population estimates obtained from local people with camera trap data, to better understand the differences in precision and accuracy of observation data, and what the sources of potential bias are. The results of this mixed-method approach are valuable for future researchers wishing to apply observational data from local people for wildlife population monitoring.

Chapter 5:
Often, experts are called upon to give their opinion or provide estimates, especially where primary data is missing or expensive to obtain. There is much research on knowledge elicitation in the context of technical experts in richer countries, but none in the context of local people in less developed countries, of wild meat hunting systems, or of wildlife population monitoring. Here, I explore what makes someone an expert in the context of wildlife population monitoring in Cameroon. Male hunters in Cameroon are often assumed to be the most knowledgeable about wildlife populations, yet little work aims to understand whether this stands true when it comes to making judgements. This work is the first to apply the expert knowledge elicitation methods set out by Burgman et

al. (2011) and Mcbride et al. (2012) with local people in the context of a forest resource dependent community.

Chapter 6:
The cost-effectiveness and relative speed of interview-based occupancy analysis could enable rapid identification of threats and changes in population distribution and occupancy. However, it is important that those designing population surveys ensure power to detect a given level of change is achieved, while optimising the use of conservation resources. A formula to assess the power of occupancy models exists (Guillera-Arroita & Lahoz-Monfort, 2012), yet, there are few real-life applications of this method. This chapter applies the power analysis formula to interview, diary and camera trap-informed occupancy analysis models. The results provide guidance on when each method works best, for what species and how to best design monitoring surveys to ensure adequate power.

Chapter 7:
This final chapter provides a synthesis of my research findings. It highlights my key conclusions, reflects on challenges, and explores the implications for wildlife population monitoring informed by local ecological knowledge. In particular, it looks at the potential for this approach as a robust monitoring method for mammal species hunted for wild meat in forest habitats and the moral and policy implications of employing this approach. The chapter concludes by suggesting directions for future research.

## 1.4 My research approach

Increasingly, researchers in applied sciences such as conservation are asked to integrate self-reflections in their writing, which commonly include self-ethnography and statements about the choices they make when conducting practical research. The following section aims to outline my personal approach to my research and to identify potential biases in my application of scientific expertise (Pasgaard et al. 2017).

Prior to my thesis I undertook an undergraduate degree in Geography, where I took an equal balance of human and physical geography modules throughout the course. I did so, as I felt that the interactions between the human and natural world were the most interesting, so I did what I could to get a grasp of both. Little did I know at the time that Conservation Science was the perfect field for me. After working in a grassroots NGO in London to personally train and support young offenders back into employment in the environmental sector for 3 years, I was accepted onto the Conservation Science MSc at Imperial College London. It was during this course that I started my work in Cameroon, employing interview-based occupancy analysis to identify the status, threats and distribution of forest elephant (*loxodonta cyclotis*) in timber concessions across 30,000 $km^2$ for my MSc thesis.

After my MSc, I worked for an advocacy group based at the Centre for Environmental Policy, Imperial College London, which lobbied the European Union to allocate more funding for sustainable agricultural practices in sub-Saharan Africa. I visited farming communities and cooperatives in Ethiopia and Mozambique, where I conducted interviews with community members and developed an understanding of the challenges subsistence farming communities encounter on a daily basis from their perspective. While of course in a different context, this engagement, I feel, provided me with a basic understanding of some of the realities facing the subsistence farmers I worked with during my PhD in Cameroon. Recognising that I knew a little bit about agricultural practices, people would talk to me about their farms, which in turn provided a means for me to strike up conversation about topics outside of conservation, and start to build friendships during my PhD.

Having employed interview-based occupancy analysis for forest elephants at a large-scale during my MSc, my supervisors and I identified the need to assess the strengths and limitations of the approach for wildlife population monitoring in a more in-depth study, which became the initial focus of my PhD. As my previous research and work had also primarily focussed on employing social science methods, I saw my PhD as an opportunity to develop biological research and monitoring skills and knowledge, to support the work of the Zoological Society of London (ZSL), and to contribute to conservation and the better integration of local knowledge in Cameroon and further afield.

Despite my prior experience of carrying out social surveys in different cultural contexts, I am not, and do not claim to be, a social scientist. I have done my best, with the training and experience I have, to employ social science methods in a robust manner, in order to answer complicated questions. The importance of conservationists using social science methods 'well', for the benefit of research participants, the communities we work with, and for the sustainability and effectiveness of conservation is close to my heart. In particular, the ethical challenges that both face and are created by conservation researchers working in unfamiliar cultural contexts, is something that I have spent a lot of time considering over the course of my PhD (see Ibbett & Brittain, 2019; Brittain et al. in review; see appendix A-1). As such, I have made no attempts to apply anthropological methods such as ethnography, which would require a whole new training, but which if executed well by correctly trained researchers, would add a lot of value and understanding to the challenges facing the robust incorporation of LEK in a more practical sense (e.g. ethnography of eco-guards, local non-governmental organisations, local people and their interactions in the Dja Reserve).

Integration of the natural and social sciences generates significant challenges (Fox et al. 2006; Evely et al. 2008). One such challenge is dealing with the wide range of underlying philosophies (a system of values that a person adheres to) and epistemologies (understandings of what can constitute knowledge, or what can be known) of the different people involved (e.g. Huntington 2000; Fazey et al. 2005). I did not consciously enter this PhD under a specific philosophy of science or worldview. However in hindsight, I used a pragmatic, post-positivist approach throughout my thesis, with the belief that multiple methods are necessary to identify a valid belief because all methods are imperfect (Moon & Blackman, 2014). However, I recognise that a positivist approach does not always provide the means to adequately examine human feelings, emotions or values. As such, I also took on a structural realist approach, which recognises that the social world changes through time, and I incorporated qualitative methods in my research to more fully understand this process of change (Evely et al. 2008).

I was aware of my position as a Western researcher and the biases this introduces. I did my best to recognise the importance of power dynamics as a researcher entering a village and how using extractive methods to gather data from participants could reinforce these dynamics. I also recognised that the priorities of conservation often clash with those of research participants. As such, I must acknowledge that the framing and research questions for this thesis were developed by me, who believes that the conservation of species in the Dja Faunal Reserve and surrounding community forests is important both for conservation, and for local livelihoods and food security. Further, I also recognize that when discussing the need for conservation and the potential applicability of local knowledge to help us reach this objective, that I am not impartial.

Throughout my data collection, I made a concerted effort to recognise and account for the biases that my presence in the study villages may have on the data collected. For example, I based myself in the villages I was working with, I was housed with families and embedded myself in their lives, adjusting to their routine. I am fluent in French, which allowed me to communicate with everyone in the village well, except, perhaps, the most elderly members of the village, whose French was more limited. The ability to communicate, combined with the relatively extensive period of time spent in the villages (7-8 months in the actual villages, just under a year in Cameroon in total), helped me to develop an understanding and appreciation for the local culture, humour, warmth and sensibility of the people I worked with. Finally, working with a team of Cameroonian MSc students who had an even stronger presence in the villages over the course of the data collection period, helped to minimise the impression of 'parachute researchers', which has been so negative to conservation in the region. In some cases, students from the study country may still be viewed as elite outsiders. However, where this study was conducted, local students are generally nurtured and welcomed, as people in the villages want to help them get a good education. Therefore, the research team were not viewed as elites at all here. Furthermore, this PhD research has paved the way for my postdoctoral research now taking place in the same villages. I am currently developing ideas based on conversation with people from the villages about potential future projects. As such, I hope to continue with action-based research in these villages in future that is beneficial to both conservation and to local people.

# Chapter 2

## Background

(View from my hut in one of the study villages)

## 2.1 Introduction

This chapter provides a theoretical and geographical background to my thesis. I first review the methods commonly used for wildlife population monitoring. Then, I discuss the common sources of bias when carrying out wildlife population monitoring, before examining how Local Ecological Knowledge (LEK) has been incorporated into monitoring to date, and where further knowledge is required to more robustly integrate LEK into monitoring. The chapter concludes with a description of the national and regional contexts in Cameroon, and the case study sites used throughout this thesis.

## 2.2 Wildlife population monitoring

Wildlife population monitoring is "the process of gathering information about system state variables to assess system state and draw inferences about changes in state over time" (Yoccoz et al. 2001). It is a fundamental aspect of conservation, if executed properly, providing us with the information required to identify trends in population size (Parry & Peres, 2015) to assess a species' status (Witmer, 2005; IUCN, 2012), the sustainability of certain threats (Kumpel et al. 2009), or measure success in conservation programmes (Lindenmayer et al. 2012).

Wildlife population monitoring programmes, whose activities usually span over several years, may be used to assess changes in system state variables, such as species density, richness, total abundance or simply the presence or absence of species at a given site. State variables can be monitored using a variety of direct or indirect monitoring methods. Each state variable and monitoring method has its own strengths and weaknesses (see table 2-1), and the choice of variable should depend upon the monitoring programme's objectives and target species. For example, whether a total count of the animal numbers in a given area, an estimate of distribution, or an understanding of species habitat preferences and threats is needed, should influence whether researchers select to use surveys of abundance, or assess distribution using presence–absence data (Joseph et al. 2006). As such, the variable selected will also determine the design and expense of the programme (Williams et al. 2002). Yet, many programmes are created without adequate attention to defining why monitoring is necessary, what needs to be monitored, and how the monitoring should be conducted (Yoccoz et al. 2001). Neglecting to ask consider these important questions can result in either poorly structured monitoring programmes that do not provide robust data, or excessive spending on monitoring that renders the programme unsustainable (Danielsen, 2003).

Table 2-1: Summary of the benefits and limitations of common system state variables for monitoring wildlife populations and their hierarchy.

| State variable | Definition | Data types | Monitoring benefits | Monitoring limitations |
|---|---|---|---|---|
| Presence only | Confirmed presences of each species are recorded. | Databases where people contribute observations or from museum/herbarium collections. | Very straightforward data collection process that can also help to identify environmental conditions experienced by a species (Guillera-Arroita, 2017). | Presence-only cannot distinguish between absence and non-detection. Therefore, species habitat preferences and habitat availability in the landscape can be confounded. |
| Presence background | Presence only data are analysed in conjunction with information about the characteristics of the environment in the wider landscape (Koshkina et al 2017). | This may be informed from biological records databases, including citizen science records, and analysed using MaxEnt (Elith, 2011; Guillera-Arroita, 2017). | Presence-background provides a more accurate portrayal of species' habitat preferences by comparing the types of environmental conditions where the species was detected to the frequency of these conditions in the landscape (Elith et al. 2011). | Data do not contain information about sampling effort, so they are very susceptible to sampling bias (Phillips et al. 2009; Guillera-Arroita, 2017).<br><br>Further, one cannot robustly quantify prevalence or probabilities of occurrence (Hastie & Fithian, 2013; Phillips & Elith, 2013) from such data, or distinguish between whether few species records are due to species rarity, or due to small survey effort (Guillera-Arroita, 2017). |
| Presence-absence | Apparent presence or absence of each species is recorded. | Presence-absence data are traditionally used to estimate site occupancy rates, or predict the | Data sets that also include species absence records are informative about sampling effort, hence their inference is more robust to | As with all state variables, presence-absence may suffer from imperfect detection, in that a species may be declared absent because it is not |

| | | | | |
|---|---|---|---|---|
| | | distribution of a species over large areas. Datasets can be produced from planned surveys or volunteer contributions (Guillera-Arroita, 2017). | sampling biases and they can estimate species occurrence probabilities. The proportion of sites where a species is present can be used as a surrogate for population size or abundance (MacKenzie, 2005).<br><br>Fast, easy and doesn't require much training compared to abundance or density. Data can be collected at more sites than abundance as the surveyor is not obliged to spend a fixed amount of time at each site (Joseph et al. 2006). | detected, resulting in biased data (Mackenzie, 2005). Model based corrections such as occupancy analysis can help to account for imperfect detection, although they are not a panacea (Mackenzie et al. 2002). |
| Density | The number of individuals per unit area | Distance sampling and mark-capture-recapture within a spatially explicit area are used to estimate density. | Given the close relationship between density and abundance, density can be scaled up to provide an estimate of abundance in a given area. | Imperfect detection: even where presence is detected, counts may be biased low by impaired detectability if the impairment isn't accounted for. As such, the accurate estimation of wildlife population density is considered by some to be difficult and require considerable investment of resources and time (Witmer, 2005). |
| Abundance | The number of individuals present | Distance sampling and mark-capture-recapture methods used to estimate | Provides a total population count or estimate in a given area which is | Imperfect detection: even where presence is detected, counts may be biased low by impaired detectability, |

| | | | |
|---|---|---|---|
| | density are extrapolated across the total study area (Buckland et al. 2015; McCrea & Morgan, 2015). | ideal data for monitoring population trends over time. | resulting in biased density if the impairment isn’t accounted for. Hard to do over large spatial scales (Royle & Nichols, 2003). Can be expensive in both time and effort (Yoccoz et al. 2001; Blanc et al. 2014). |

### 2.2.1 Detection, non-detection and imperfect detection

All system state variables described in table 2-1 rely on a person, camera trap, or drone detecting the animal or their signs (Stephens et al. 2015). However, as touched upon in table 2-1, detectability is rarely perfect (MacKenzie et al. 2002). When dealing with direct monitoring methods, detectability refers to the extent to which all individuals of a species present within a given area are detected (MacKenzie et al. 2002; Bailey et al. 2004). For indirect methods which rely on sign surveys as opposed to direct observations, detectability represents the disparity between the counts of species signs and the true number of animals present (Hedges, 2012). Imperfect detectability that remains un-corrected can result in major error types. Using occupancy estimates as an example, animals may go undetected when they are in fact present, so occupied sites may be classified as unoccupied (Royle & Nichols, 2003; Royle, 2006) (known as type II error). Conversely, false-positives (detecting an animal when it is not present) can occur through species misidentification, known as a type I error (Royle & Link, 2006; Miller et al. 2011), which results in data that overestimates occupancy. False-positive errors have been documented for many types of occurrence data, including auditory call surveys ( McClintock et al. 2010; Miller et al. 2012), surveys based on animal sign (Molinari-Jobin et al. 2012), and surveys for cryptic species (Shea et al. 2011).

Imperfect detection can lead to biased inference about species distributions and overstated precision about estimated parameters (Tyre et al. 2003, Gu & Swihart 2004, Kéry et al. 2010, Lahoz-Monfort et al. 2014), resulting in inaccurate estimation of extinction probability (Kery et al. 2006), biased inference on habitat preferences (Kéry & Schmid, 2004), or biased population trajectories (Shefferson et al. 2001). As such, type I error has led to long-term and irreversible effects on biodiversity, such as the collapse of fish stocks (Dayton, 2001) or at worst the extinction of threatened species (Taylor & Gerrodette, 1993). In contrast, type II error may cause unnecessary restrictions on natural resource use (Danielsen et al. 2003; Nuno et al. 2014). While type I error is often considered to be more serious than type II error in conservation, type II error can also result in serious implications for livelihoods or food security, if local people's usual access to species hunted for wild meat, for example, is limited unnecessarily.

### 2.2.2 Sources of bias and uncertainty

All monitoring methods are subject to bias. For wildlife population monitoring to be robust, conservation actors must aim to remove heterogeneity in detectability across the landscape through solid survey design. Any remaining heterogeneity in detectability that cannot be designed out must be accounted for analytically, to prevent actual trends in species populations being confounded with variation in species detectability (Yoccoz et al. 2001; Milner-Gulland & Rowcliffe, 2007). Yet, many programmes ignore or deal ineffectively with spatial variation and detectability. Here, I will introduce the most common sources of bias in wildlife population monitoring.

Biophysical sources of bias

Species detectability depends upon a variety of biophysical variables (Williams et al. 2002), such as heterogeneity in the landscape (Sutherland, 2006; Mackenzie et al. 2011), seasonality and variations in animal abundance (Royle & Nichols, 2003) and species ecology (Burton, 2012). Species detectability decreases in areas of dense vegetation such as forest, because they are harder to see, rendering wildlife population monitoring more challenging (Tracey et al.

2005). In these conditions, the use of many conventional survey methods are more limited (Hedges, 2012), and covering a large area of difficult ground is both expensive and time consuming.

Observer sources of bias

Observer variables can also influence the ability to detect a species. Sociodemographic variables such as an observers level of experience, or the survey effort undertaken are likely sources of bias for all population monitoring methods (see Buckland et al. 2001; Sethi et al. 2005; Nuno et al. 2013). Despite a growing recognition of the need to understand the impact that different bias have on the precision and accuracy of wildlife population estimates (Elphick, 2008) there remains a limited understanding of the effect of observational error on population monitoring, with little attention to the multiple potential sources of error involved (Nuno et al. 2013).

Survey design induced bias

Uncertainty can be built into a programme through poor survey and sample design (Blanchard et al. 2007), or a lack of consideration for the programme aims and objectives. For example, studies should be designed to ensure independence between sites (Nur et al. 1999) as animals may be double counted if the species range size and spatiotemporal scale of their movement is not accounted for, resulting in population overestimates (Sutherland et al. 2014). Further, non-random assignment of observers to conduct repeat visits (Miller et al. 2012), or variation in the distance from observer to the target species can render some species easier to detect than others (McClintock et al. 2010) which may subsequently result in heterogeneity in species detection.

### 2.2.3 Common trade-offs in wildlife population monitoring methods

Monitoring must be able to detect true trends over time, while accounting for cost (Kinahan & Bunnefeld, 2012). Therefore, monitoring effectiveness (the ability to detect trends) and efficiency (the ability to do so at a reasonable cost) are key considerations in the selection of appropriate monitoring methods (Nuno et al. 2014). However, there is a pervasive trade-off between monitoring methods that prioritise effectiveness and those that prioritise cost or sustainability (Gaidet-Drapier et al. 2006; Nuno et al. 2013). Some studies question whether the trade-off can ever be addressed successfully at all (Danielsen et al. 2003). As such, and in particular where monitoring budgets are restricted, the reliable and repeated data required to identify the drivers and trends of population change are often lacking due to the substantial resources required for ongoing population monitoring (Brashares & Sam, 2005; Lindenmayer & Likens, 2009). The effectiveness of monitoring is generally inhibited by budgets and varies with the manager's willingness to accept different error types (Lindenmayer et al. 2012; Nuno et al. 2014). In some cases, time, budget and observational constraints may mean that managers are better off allocating resources to other interventions, rather than monitoring (McDonald-Madden et al. 2010).

Due to the common trade-offs between cost and effectiveness, population monitoring at a large-scale remains challenging. For example, line-transects are considered to be an accurate tool for population monitoring (Buckland et al. 2001). Yet, the intensive sampling effort and relatively high cost and time investment required, means surveying using this approach is often restricted to a few sites (Ceballos & Ehrlich, 2002; De Thoisy et al. 2008). Table 2-2 summarises the key trade-offs between effectiveness and efficiency for conventional wildlife population monitoring methods, and serves to highlight how all methods are subject to bias and trade-offs.

Table 2-2: Commonly used population monitoring methods and their common trade-offs

| Direct monitoring methods | Precision/accuracy | Cost | Technical skills |
|---|---|---|---|
| Linear transects | Can be an accurate and powerful approach to describe populations through estimates of either relative abundance or absolute density (Sutherland, 2006). Direct observations of individuals can be recorded for abundant or bold species (Silveira et al. 2003). However, it is often imprecise. When using this method in difficult visibility or with cryptic species, the data can be quite unreliable as poor encounter rates can lead to sample sizes not large enough for data analysis (Bennun et al. 2004; Refsnider et al. 2011). | Costly & time consuming, limiting their use in challenging habitats (Moller et al. 2004). The restriction on following a precise path can make surveying in difficult terrain such as forest habitats problematic (Hiby & Krishna, 2001). | Depend on suitable field conditions and trained personnel (Burnham et al. 1980; Smallwood & Fitzhugh, 1995). |
| Camera trapping | Can provide accurate species density and abundance estimates (Rowcliffe et al. 2008; Howe et al. 2017; Nakashima et al. 2018). Robust to variation in ground conditions and climate, and can be used to gain information on cryptic species and in difficult terrain where other field methods are likely to fail (Rowcliffe et al. 2008). Camera traps can reduce observer bias and has a similar efficiency in collecting data on both diurnal and nocturnal species (Silveira et al. 2003; Rovero & Marshall, 2009), but may result in bias in species trapping rates towards trap-curious species, compelled to return to the camera locations more frequently (Wegge et al. 2004). | Although labour costs are relatively low (Rowcliffe et al. 2008), the cameras themselves are expensive and therefore often restricted to larger budgets (Lyra-Jorge et al. 2008). Cameras can malfunction or be stolen (Larrucea et al. 2007;Burton, 2012). | Camera traps do not require extensive training (Karlin & De La Paz, 2015). However, unbiased data relies upon accurate placement of cameras, which requires some training and experience. |

| | | | |
|---|---|---|---|
| Aerial surveys | Long established standards permit rapid counts of target species in open areas (Craig & Reynolds, 2004). However, aerial surveys are not suitable for forested habitats as visibility and therefore detectability will not be sufficient for an unbiased count. Estimates obtained may be affected by animal detectability, observer performance, variation in aircraft height and deviations from the transect (Jachmann, 2002). | Expensive and therefore cannot be conducted frequently enough for meaningful data in many cases (Msoffe et al. 2009). Recent 'unmanned' aerial surveys reduce costs and bias. | Need access to a plane or unmanned plane, plus training in how to use this equipment. Nuno et al. (2013) find aerial surveys are highly affected by observer effects, although the same can be said for all monitoring methods that rely on observational data |
| Mark-recapture | Population estimates can be very accurate, provided assumptions are met and the sampling design is robust. The approach is very useful for geographically well-defined populations of animals with restricted ranges (e.g. islands or discrete habitats). Capture–recapture approaches are prone to problems with heterogeneity in capture probability caused by unequal access to traps (Seber, 1992). | Expensive and time consuming. Not suitable for large study area, in dense habitats or for rapid assessments. Requires greater sampling effort and resources than other methods. | The animal usually needs to be captured to be marked which can require extensive training to avoid causing injury. |
| Indirect monitoring methods | Precision/accuracy | Cost | Technical skills |
| Indirect sign transects | Indirect signs such as tracks, dung and nests, are often more appropriate for elusive, rare and nocturnal species (Boddicker et al. 2002). However they have been the focus of much concern (Pollock et al. 2002; Saska et al. 2013); while it is often assumed that the relationship between the index and abundance is direct (Pollock et al. 2002), this is generally unverified (Witmer, 2005). | Cheaper than direct sighting methods. | The observer needs to be able to identify the signs correctly. |

Although common trade-offs between cost and accuracy are well documented, they are not always considered by conservation practitioners when designing monitoring programmes, which can reduce the programmes precision and efficiency. Conservation actors must assess the trade-offs between a monitoring programme's robustness and feasibility, given the limited resources available (Singh & Milner-Gulland, 2011). Power analyses are a common tool in conservation that can help conservation actors investigate such trade-offs.

Power analysis for wildlife population monitoring has focused mainly on assessing the power to detect changes in abundance (Rhodes et al. 2006) or occupancy (Steenweg et al. 2016, Latif et al. 2018). Jones et al. (2008) provide one of the few examples where the power of interviews with local informants to detect changes in catch and harvest effort was assessed. The authors carried out rapid assessment interviews with villagers in Madagascar regarding the quantity, timing and spatial patterns of natural resource collection (Crayfish and firewood), finding that interviews with local informants in Madagascar could detect a 20% change in crayfish and firewood harvesting with 90% power. Relatively recent developments now allow for the effect of imperfect detection to be incorporated into power analyses (Guillera-Arroita & Lahoz-Monfort 2012). In some cases, the cost of visiting and sampling sites has been integrated with power analysis to explore trade-offs between the number of sites, and the frequency and duration of monitoring possible given fixed budgets and objectives (Field et al. 2005). However, studies which combine power analysis with cost-effectiveness assessments are few and far between in conservation.

## 2.3 Incorporating local ecological knowledge into wildlife population monitoring

### 2.3.1 Local ecological knowledge

Local ecological knowledge (LEK) was defined by Huntington (2000) as "the knowledge and insights acquired through extensive observation of an area or species". The ecological knowledge held by people who live in the area of interest differs from the ecological knowledge that an external researcher may hold, which, due to the nature of conservation research, often represents a snapshot in time as opposed to a culmination of knowledge through in-depth and long-term interactions. LEK also differs from traditional ecological knowledge (TEK) (Turvey et al. 2014), the cumulative body of ecological knowledge and belief passed down between generations by cultural transmission (Berkes et al. 2000).

Interview-based surveys allow for data to be cost-effectively and rapidly collected across large spatial scales that cannot be achieved using conventional methods (Turvey et al. 2013, 2015; Mohd-Azlan et al. 2013; Service et al. 2014). Rist et al. (2010) also found locally-based monitoring approaches to be cost-effective, but went further to provide an assessment of their power to monitor the status of natural resources in wild meat hunting systems. The authors gathered data on wild meat hunting, catch and effort using a professional technique (accompanying hunters on hunting trips) and two locally-based methods in which data were collected by hunters (hunting camp diaries and weekly hunter interviews) in a 15-month study in Equatorial Guinea. Using power simulations of catch and effort data, they showed that locally-based methods can reliably detect meaningful levels of change (20% change with 80% power at significance level [$\alpha$]=0.05) in multispecies catch per unit effort.

As well as monitoring species populations, local ecological knowledge can be draw upon to better understand threats to wildlife populations. Abram et al. (2015) used local knowledge of threats to orangutan populations collected via questionnaires, integrating this data with environmental and socio-economic variables to predict threat levels and population trends. Jones et al. (2008) compared the results obtained from seasonal interviews with information from daily interviews with the same informants, using mixed-models to investigate how accurately people reported their activities in the rapid assessment interviews, and estimated the probability of detecting a change in harvesting rate from two such interviews. The authors found that the interviews provided reliable and acceptably precise information on quantities, effort, and the spatial pattern of harvesting.

### 2.3.2 Uncertainty when incorporating local knowledge

Several biases and potential solutions have been well documented in the literature, and are summarised in table 2-3:

Table 2-3: Common biases and potential solutions to consider when gathering data from participants

| Source of bias | Type of bias | Description/cause | Solution | Literature |
|---|---|---|---|---|
| Interview bias | Self-esteem effect | Participants give responses that they feel reflect well on them, or that the interviewer is looking for. | Remaining neutral in the phrasing of questions and stating that there are no correct answers. | Newing et al. (2011)<br>Meijaard et al. (2011) |
| | Order effect | Participants are influenced by that has previously been discussed. | Ensure interviews and questionnaires follow the 'interview funnel', where questions begin broadly and narrow down. | Newing et al. (2011) |
| | Audience effect | Participants may change what information they provide depending on who is listening to them. | Interviews should be conducted individually where possible to reduce this bias, in particular when discussing topics of an illegal or sensitive nature. | Newing et al. (2011) |
| | Participant may misunderstand the interviewer | Subtle differences in the phrasing of questions. | Ensuring that the phrasing of questions remains consistent and clear. | Turvey et al. (2015) |
| | Interviewer may misunderstand/misreport the participant | Can be compounded by errors in note-taking, during enthusiastic conversations as highlighted by filmed interviews. | Record interviews on a dictaphone where permission is granted by the participant. Ensure rigorous note-taking and work as a pair to ensure that all details are well-noted. | Mohd-Azlan et al. (2013)<br><br>Danielsen et al. (2005) |

| | | | | |
|---|---|---|---|---|
| | Intentional deception | Can be compounded by interview fatigue, by the participant not knowing the answer to a question or wanting to mislead the interviewer | Interviews were arranged at a time and place that suits participants and informed consent was gained from all participants before interviews. the option to say 'no answer' or 'I don't know' was always made clear to participants. | |
| Spatial design | Species may appear more abundant in areas which local people frequent more regularly | Local people are rarely evenly distributed across a landscape resulting in uneven survey effort. | Either formalize data collection, by creating a systematic monitoring programme, turning local people into monitors which brings its own challenges. Or, account for bias in survey design, such as with occupancy analysis. | Moller et al. (2004) |
| Observer bias | Over- or under-estimation | Social norms and pressure can incite deception as well as distortions. Further if local people are adept at finding a species, they may overestimate its population size as it is thought of as common. | Expert elicitation methods aimed to get participants to think more about estimates and reduce anchoring which can result in overestimation. | Lunn & Dearden (2006)<br>Moller et al. (2004)<br>Burgman (2011)<br>McBride et al. (2012)<br>Hemming (2017) |
| | | | Pooling expert opinions may result in estimates that more closely resemble reality | Van der Hoeven et al. (2004) |
| | Recall bias | Asking participants to recall information over a long period of time. | Some debate over the best time of year to ask for data (e.g. low or high season), or over what time scale (e.g. monthly or annually). | Mohd-Azlan et al. (2013).<br>Golden, Wrangham & Brashares (2013) |

| | | | | |
|---|---|---|---|---|
| | Shifting baseline | Perceptions of the state of biodiversity in the system in which they live are updated, so the new perception of reality is believed to be the same state that has always existed (shifting baseline). | It may be possible to target people who are less susceptible (e.g. older people), however this bias should be considered when interpreting the results. | Papworth et al. (2009) |
| | Change blindness | The opposite of shifting baseline, when participants do not update their sense of reality, known as change blindness. | It may be possible to target people who have more frequent, direct experience to avoid change blindness, although the bias should be considered when interpreting the results. | Papworth et al. (2009)<br>Simons et al (2005) |
| | Overconfidence | When a person's confidence in their judgements is not proportional to their expertise. | Expert elicitation methods aimed to get participants to think more about estimates and reduce anchoring which can result in overestimation. | Burgman et al. (2011)<br>McBride et al. (2012)<br>Martin et al. (2011) |

Meijaard et al. (2011) responded to the need for a more robust approach to interview data and provided useful guidance to address methodological weaknesses in social surveys, such as sampling and questionnaire design, respondent bias and statistical analysis, based on the findings of over 7000 interviews with local people on the distribution and threats to the Bornean orangutan. To reduce the effect of self-esteem' or 'yes' biases in their study, the authors did not state what the target species was. They also took measures to assess the participants' reliability by asking them to identify a series of animals from photographs prior to conducting the interviews. Information provided by participants was triangulated against other data sources and between participants.

Data collected from local people may often be opportunistic or non-systematic, yet opportunistic data are often hard to use in statistical analysis as the distribution of the effort is unknown. Giraud et al. (2015) developed a statistical framework to combine opportunistic data on species abundance with data collected using a known sampling effort, finding that combined estimates were more precise than those obtained with a known sampling effort alone. When the opportunistic data were abundant, the gain in precision was significant, especially for rare species. Athreya et al. (2015) used a novel approach of collecting data on leopard related incidents from media reports combined with habitat occupancy modelling methods to map their distribution across the State of Karnataka, Southern India. Combining opportunistic data from local people within an occupancy framework that accounted for spatial uncertainty reduced spatial bias and resulted in cost-effective yet robust estimates of leopard occupancy, with estimates that concurred with previous nationwide surveys.

There is some disagreement in the literature over the ability of local people to correctly identify rare or elusive species. Bryman et al. (2014) reported that observers were more likely to report rare sightings, but in contrast, other papers found that anecdotal evidence from local people was unreliable for rare species (McKelvey et al. 2008; Miller et al. 2011; Mohd-Azlan et al. 2013). It must be noted from the literature that the ability of local people to correctly identify species is extremely context specific. Further, species misidentification is not a problem confined to the use of LEK for monitoring, but afflicts many different types of expert.

Locally-based monitoring is particularly relevant in developing countries, where it can lead to rapid decisions to solve the key threats, empower local communities to better manage their resources, and can refine sustainable-use strategies to improve local livelihoods. Gouwakinnou et al. (2011) and Houehanou et al. (2011) investigated the extent to which ethnicity influences the LEK held by participants about food trees and multipurpose trees, respectively, in Benin, West Africa. Gouwakinnou et al. (2011) found that ethnic groups were informed differently on tree population decline, as the tree species did not have the same importance to all groups, while Houehanou et al. (2011) found that gender, local availability, ethnicity and community location interacted to influence the use value of the species. Despite the growing support for the integration of LEK for conservation, the accuracy and precision of monitoring by local communities needs further study, and field protocols need to be developed to get the best from the unrealized potential of this approach (Danielsen et al. 2008).

## 2.4 Interview-based occupancy analysis

Presence-absence interview surveys used in an occupancy modelling framework (interview-based occupancy analysis) are growing in popularity and provide potential for rapid conservation status assessments of multiple species across large-spatial scales over time, that also allow for heterogeneity in occupancy and detection to be accounted for. The approach has been used to monitor individual across large-spatial scales, including the distribution of sloth bears across 38,500 $km^2$ in the Western Ghats, India (Puri et al. 2015); and to identify Jaguar corridors in Nicaragua (Zeller et al. 2011) and Mexico (Petracca et al. 2013).

To improve the cost-effectiveness of species population monitoring programmes, researchers often try to monitor several species of conservation interest at the same time. For example, Pillay et al. (2011) used key informant surveys over large spatial scales to generate detection histories for 18 species of large mammals (body mass > 2 kg) using recall at two points in time (present and 30 years ago) in the Southern sub-region of the Western Ghats, India. Their results showed significant declines in distribution for large carnivores, the Asian elephant and endemic ungulates and primates. The authors concluded that presence-absence surveys elicited from key informants used in an occupancy modelling framework provided potential for rapid conservation status assessments of multiple species across large spatiotemporal scales.

While previous studies highlight the potential value of interview data to inform occupancy models, interview data from local people can also provide rich context and understanding of the drivers of change, and perceived threats to the target species. As such, some studies have incorporated open-ended questions into their interviews to provide data which can help to identify the threats to the target species and potential drivers of population change. Martinez-Marti (2011) used semi-structured interviews with key informants to assess the current status of the leopard (Panthera pardus) and golden cat (Caracal aurata) in Equatorial Guinea and provide baseline information for their conservation. Presence-absence data were used with occupancy modelling to describe the geographical range of cats, great apes and forest elephants and to identify the principal factors explaining their distributional range and threats, which included direct and incidental killing and trade, and habitat fragmentation for transportation and agriculture. Brittain et al. (2018) were inspired by the methodology set out by Martinez-Marti (2011) and applied the approach to collect data on forest elephant (Loxodonta cyclotis) trends, relative abundance and perceived threats using interviews with timber concession workers across in the eastern region of Cameroon. The authors found that elephant detectability had decreased over 6 years, consistent with declining perceived abundance in occupied sites as reported by participants, and identified perceived threats to the species across the region that were concurrent with data from other reports. They concluded that interview-based occupancy analysis was a suitable method for a rapid assessment of forest elephant occupancy and threats across a large-scale, as a compliment or first-stage in a monitoring process.

Pillay et al. (2014) fitted occupancy models that simultaneously account for false-positives and negatives (see Miller et al. 2011), to data collected from a large-scale key informant interview survey for 30 large vertebrate species. They tested the false positive model performance against standard occupancy models that account only for false-negatives and found standard occupancy models tended to overestimate species occupancy due to false-positive errors. Petracca et al. (2017) also applied occupancy models that account for false-positive detections, but noted that, where sites have only one detection, possibly due to one particularly good observer for example, false-positive models may confound sites with few detections with false-detection, which may lead to an underestimation of occupancy as a result. As such, in datasets where some sites have fewer detections due to uneven observer ability, or uneven distribution of observers across the landscape, such models must be used with caution.

### 2.4.1 Comparative studies

Comparative studies are a useful approach to assess the precision, accuracy and cost of different monitoring methods or assess their applicability to different species. Yet, research that adequately compares results obtained from different monitoring methods is lacking. Comparison studies in the context of forest wildlife tend to focus on comparing camera trap data to other monitoring methods (Silveira et al. 2003), often to better understand the effect of different camera trap placements on the resulting estimates (Foster & Harmsen, 2012). Therefore, the biases associated with camera traps in forest habitats are relatively well understood compared to other monitoring methods.

In recognition of the vital role that detection plays in dictating the efficiency and effectiveness of monitoring methods, several comparison studies have focussed on assessing detection efficiency. Munari et al. (2011) compared the detection efficiency of three commonly used monitoring methods: camera trapping, diurnal and nocturnal line surveys. All techniques failed to detect the most common species, agouti, in at least two sites. As such, they recommended gathering data from a combination of sources to maximize the efficiency of medium and large mammals monitoring programmes. Gaidet-Drapier et al.

(2003) compared the sampling and detection efficiency of six direct count monitoring methods in Zimbabwe to estimates derived from aerial surveys. Not only did simple community-based monitoring methods such as bicycle and on foot counts result in better detection rates than the aerial surveys, they were also less arduous and expensive.

Comparison studies have also been helpful to show not only the cost-effectiveness of data obtained by local people compared to conventional methods, but also to assess the biases associated with different monitoring methods and how they may affect the robustness of estimates. Anadon et al. (2009) illustrated how interviews with shepherds were effectively used to estimate the abundance of tortoises (Testudo graeca) in south-eastern Spain. This study made comparisons between the abundance estimates from interviews with the estimates from distance sampling protocols and concluded that estimates informed by local knowledge not only showed a close relationship with the distance sampling estimate, but interviews were 100 times cheaper to conduct. Results from Danielsen et al. (2014) also concurred with Anadon et al. (2009). The authors compared how the indigenous and local knowledge of natural resources, obtained from focus groups in two communities in Nicaragua, matched information collected during line-transects by trained scientists. The authors found that both approaches provided robust data on natural resource abundance, but focus groups allowed data to be collected for eight times less money than line-transects.

## 2.5 Cameroon: national context

### 2.5.1 Cameroon

Cameroon is a lower-middle-income country with a population of close to 24 million (ODI, 2017). Located along the Atlantic Ocean, it shares its borders with Chad, the Central African Republic (CAR), Equatorial Guinea, Gabon, and Nigeria. Two of its border regions with Nigeria (northwest and southwest) are Anglophone, while the rest of the country is Francophone. Cameroon is endowed with rich natural resources, including oil and gas, minerals, high-value species of timber, and agricultural products, such as coffee, cotton, cocoa, maize, and cassava.

Cameroon is striving to become an upper-middle income country by 2035. In order to do this, the World Bank (2019) have stated that Cameroon need to increase productivity and place greater emphasis on developing their private sector. As such, the government are prioritising developments in infrastructure, often assisted by foreign investment and aid from China and other international countries. However, the benefits of development and extraction of natural resources do not always reach local communities.

In November 2018, disputed election results returned President Paul Biya to office. At 85 years old, Paul Biya, who has held power since 1982, is now serving his seventh term as the country's president. He is the longest ruling non-royal national leader in the world. Civil war has raged in the Anglophone regions of North-West and South-West Cameroon since 2015, when separatists declared an independent state of Ambazonia as a result of tensions between the Francophone and Anglophone legal systems, and frustration at the lack of financial and logistical development in the Anglephone regions, despite being regions rich in natural resources. Tensions quickly escalated into violence between the separatists and the Rapid Intervention Battalion (BIR), an elite army unit trained and financed in part by the USA and Israel. As a result of this conflict, the International Crisis Group (ICG) estimated that about 246,000 people have fled the South West region alone with about 25,000 believed to have fled to Nigeria. The Cameroonian Government's human rights record remains flawed, with continued reported abuses, including beatings of detainees, arbitrary arrests, and illegal searches. The judiciary is frequently corrupt, inefficient, and subject to political influence (Global Integrity Report, 2008).

### 2.5.2 Cameroon environmental law

In 1992, during the United Nations (U.N.) Conference on Environment and Development, Cameroon ratified the Convention on Biological Diversity (CBD), which signified its first step away from a relatively piecemeal approach to environmental policy, and towards a more holistic approach. After the CBD, Cameroon created law No. 94/01 in 1994, to lay down forestry, wildlife, and fisheries regulations. This law included an objective to involve communities in the management and protection of forest resources, recognising traditional custodians of wildlife resources as partners in natural resource management (Egbe et al. 2001).

Cameroon is often known as "Africa in miniature" because of its geographical and cultural diversity. Cameroon’s vision for biodiversity is to be a country which a) exploits or rationally uses its natural biological resources sustainably to meet the development needs and well-being of her population; b) preserves its ecosystem balance; and c) hands down the richness of its biodiversity to future generations (Republic of Cameroon, 2012). To reach this vision, Cameroon's conservation efforts include the establishment of a National Biodiversity Strategy and Action Plan (NBSAP), the National Ivory Action Plan (NIAP), and the development of a National Action Plan for the Conservation of Great Apes in March 2003. It also approved a National REDD+ Strategy in 2013 (World Bank, 2013).

### 2.5.3 Wild meat: local significance and monitoring challenges

Wild meat is important for livelihoods and food security in Cameroon. Yet wild meat supplies are steadily declining, due primarily to over-harvesting, in-turn encouraged by the commercialization of the wild meat trade (Wright and Priston, 2010; Kamgaing et al. 2019) and improved access via logging roads (Yasuoka et al. 2015; Kleinschroth et al. 2019), which facilitates illegal hunting by increasing the access points to the forest for hunters. The distinction between subsistence and commercial use of wildlife for food is often blurred, as meat from the forest supplements both diets and incomes. Income alternatives to hunting are scarce in rural villages across the Congo Basin (De Merode et al. 2004). When they do exist, they can be short-term and unpredictable, leading young men to continue hunting rather than engaging in potentially more profitable activities (Solly, 2001).

Harvested species populations should not be reduced to densities so low that they can no longer fulfil their economic and social role in contributing to livelihoods and food security for dependent populations (van Vliet et al. 2011). To ensure that densities do not diminish, robust monitoring of species populations is required. Several recent studies have sought to quantify the extent of hunting pressure on mammals globally (Nielsen et al. 2017) synthesise and collate knowledge on the wild meat trade and consumption across West and Central Africa (Taylor et al. 2015), map hunting pressure across Central Africa (Ziegler et al. 2015) and use wild meat market data and correlates of human activity to identify areas in need of conservation action in Nigeria and Cameroon (Fa et al. 2015).

However, monitoring of species hunted for wild meat and the hunting pressure these species are exposed to presents a unique set of challenges. Monitoring duiker populations is difficult due to their cryptic nature and occurrence in often densely vegetated habitats (Bowkett et al. 2006). The occupancy or density of species in hunting systems that are under intense hunting pressure may change rapidly, and as such monitoring efforts must be frequent enough to detect rapid changes in species populations. Market surveys are commonly used to investigate hunting pressure; such surveys may allow researchers to identify shifts in the composition of traded species and the loss of slower-growing species (Cowlishaw et al. 2005) or a reduction in the proportion of trade represented by vulnerable taxonomic groups, such as the primate: rodent ratio (Wilkie & Carpenter 1999). However, it is challenging to disentangle evidence for depletion of wild meat from market surveys, as the dynamics observed may result from changes in hunter opportunity costs, or the price of alternative goods (Crookes et al. 2005; Ling & Milner-Gulland 2006), rather than providing an indication of species depletion.

In order to gain inference on species depletion, the spatial dynamics of the trade need to be understood. For example, static species compositions in the trade may be caused by expansion of trade into previously less-exploited areas rather than sustainable use of a constant catchment area (Allebone-Webb et al.

2011). Therefore, methods that allow for spatially explicit data on hunting offtake are invaluable for the development of monitoring methods for wild meat. It is also important to use methods that involve detailed local knowledge on hunter behaviour, to disentangle processes that may result in changes in traded wild meat volumes and compositions (Crooks et al. 2005).

## 2.6 Study site: The Dja Faunal Reserve (DFR)

To identify the ideal study area for this research, I considered the site characteristics required to fully explore my research objectives and identified a set of essential and desirable criteria. A number of different site options were considered, including the Pendjari-WAP complex in Benin, and sites adjacent to other protected areas across Cameroon. Finally, the Dja Faunal Reserve (DFR) in Cameroon was selected, because the area fulfils all the essential and most of the desirable criteria for this study, as outlined in table 2-4.

Table 2-4: Essential and desirable criteria for the selection of a study site and how the DFR meets these criteria.

| Essential Criteria | Justification | How does the DFR meet these criteria? |
|---|---|---|
| A selection of species with different sizes, ranges and density. | To test the impact of species characteristics on species detectability and the ability of participants to make robust judgements. | Yes: The reserve and surrounding area is home to over 100 mammal species. |
| Participants are legally permitted to access the main area of interest in the study for livelihood activities and to hunt for subsistence. | It was important for me that I did not require that participants compromise themselves in illegal activity in order to meet my research objectives. If participants did share information on entering a protected area, or hunting a protected species, it was entirely up to them and provided additional information as opposed to data central to the research. | Yes: Many villages around the protected area are situated within their own community forest, which they are able to access to hunt for subsistence. |
| A variety of willing respondent groups that regularly access the study site. | Respondents need to be willing to take part and they need to access areas that are occupied by the target species if they are to be detected. | Yes: The pilot study identified a number of different villages who were willing to take part in the study. Participants regularly access the community forest that surrounds their village. |
| Ability to obtain robust wildlife population monitoring data from conventional methods (e.g. primary data from camera traps, or data from previous studies). | Required to compare the data obtained from my research with data obtained from more conventional methods that would usually be employed to gather the same information that this study will gather from interview methods. | Yes: ZSL Cameroon work within the DFR and kindly allowed me to use their camera traps to create my own comparative data sets. |

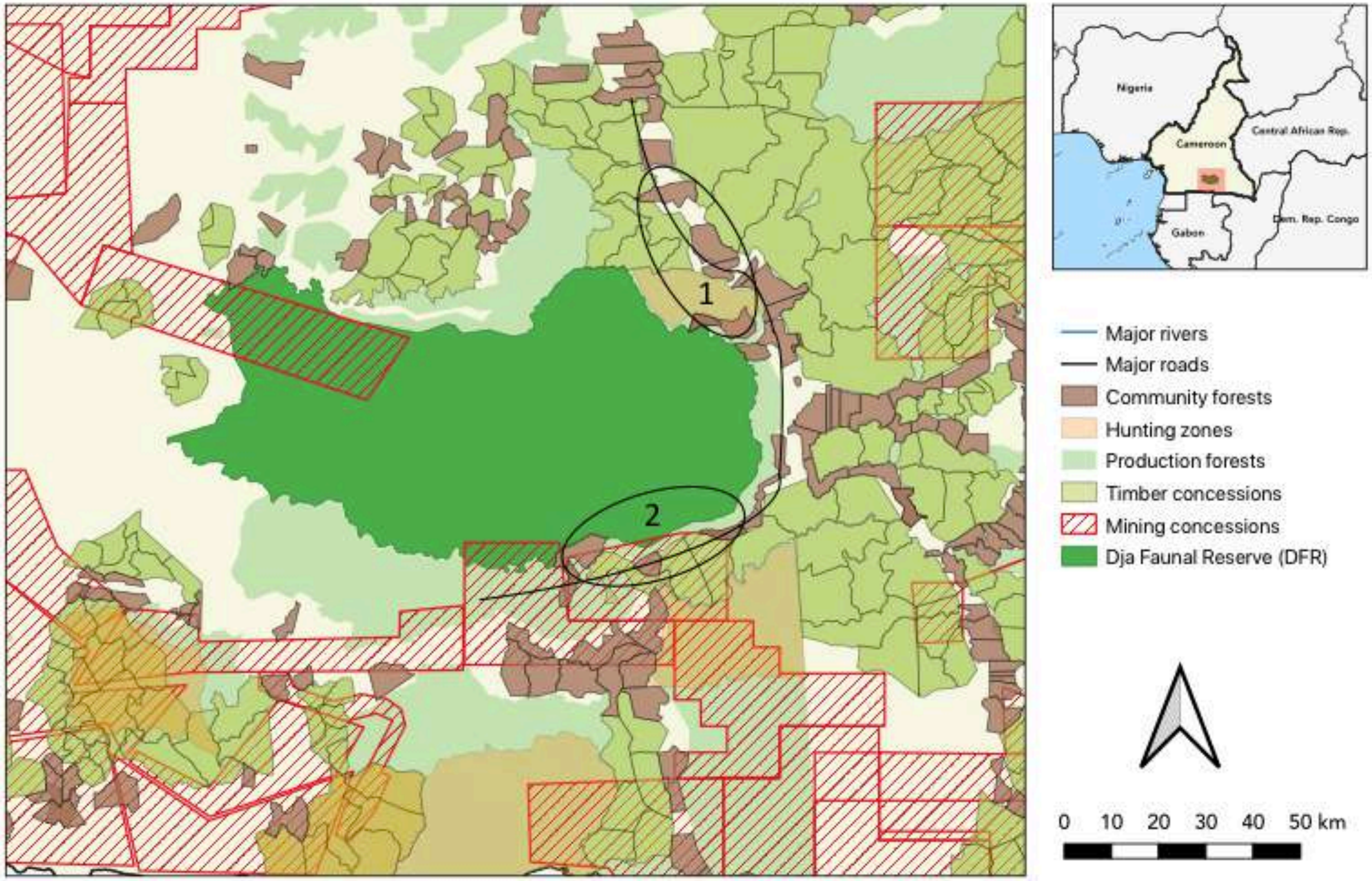
| | | |
|---|---|---|
| An interesting socioenvironmental context. | To use observational methods such as daily diaries combined with occupancy analysis within an SES framework, which allows me to investigate the use of this approach aa tool for identifying threats to biodiversity and building positive relationships between local people and conservationists. | Yes: Many people in the area hunt for wild meat for livelihood and subsistence purposes. Additionally, the area is under rapid change and potential threat from logging and mining companies, whose road networks extend rapidly into the forest. |
| Desirable criteria | Justification | How does the DFR meet these criteria? |
| A diversity of demographics within each responder group (age, gender, ethnicity, job). | To examine the impact of sociodemographic variables on detectability and robustness of estimates from participants. | Yes: Straddling the east and southern regions, the villages around the DFR are ethnically diverse. Although subsistence farming is the main livelihood, people often have a diverse array of secondary livelihoods, including collecting non-timber forest product (NTFPs) and hunting. |
| A high density of respondents who achieve an even coverage of the study site. | This would give the data the highest chance of being precise and would increase the likelihood of detection through an increased and relatively systematic survey effort. | No: This area does not have a high population density (see section 3.1.10) and local people do not evenly access their community forests. However, medium to large villages will have sufficient participants for this study. |
| A variety of different land uses/access. | In order to test the impact of different land uses on species occupancy. E.g. protected area, vs community forest, vs timber concession. | Yes: The DFR is a protected area, surrounded by a mosaic of community forests, timber concessions, mining concessions and safaris (see figure 3.1). |

### 2.6.1 Site description

The Dja Faunal Reserve (DFR) is a lowland rainforest, situated between the eastern and southern regions of Cameroon. The eastern region is the most sparsely populated region (population density of 7.35/km$^2$, although this can increase to between to 20-60/km$^2$ in villages along main roads (World Bank, 2013). This habitation pattern is a result of deliberate government policy during and after the colonial period to move people from the forest to the roads (Neba, 1999). As the poorest of Cameroon's regions, many of the DFR's inhabitants rely on subsistence farming for food security and livelihoods (World Bank, 2013) and on wild meat hunting as an additional livelihood and as a source of protein (Bobo et al. 2015).

Surrounded on three sides by the Dja River, the DFR is one of the largest rainforests in Africa, covering 5,260km$^2$ (2,030 sq. mi). It is home to 107 mammal species, five of which are threatened (UNESCO, 2015), including the endangered forest elephant (*Loxodonta cyclotis)* and the critically endangered western lowland gorilla (*Gorilla gorilla*). The reserve was designated a World Heritage Site in 1987 (figure 2-1). The IUCN states that globally, 25% of mammals ae at risk of extinction globally, so at first glance it may appear that the DFR is performing well comparatively. However, recent camera trap monitoring efforts in the  in the reserve revealed that, while still present in the reserve in the face of ongoing threats, larger terrestrial vertebrate fauna of the Dja Biosphere Reserve falls, most likely in the Diminished Fauna category (the third-level of a faunal intactness continuum that ranges from intact, relatively intact, diminished, depleted, to an ‘empty forest’ status) (Bruce et al 2017).

Figure 2-1: Map of Cameroon and the Dja Faunal Reserve (DFR),Cameroon, Central Africa. Map shows the location of the DFR and the different land uses that surround the reserve. Oval shading show general area of study sites 1 & 2



## 2.6.2 Threats to biodiversity in the Dja region

At the time of World Heritage listing in 1987, 90% of the Dja Faunal Reserve was considered intact and human pressure was low. UNESCO describes the Dja reserve as "one of the largest and best protected humid forests in Africa". Yet, issues with the protection and governance of the protected area have long been known to be acute.

Threats to biodiversity in the reserve have been on the increase over the past decade, including from hunting, the construction of the Mékin hydroelectric dam north of its boundaries, the prospect of a nickel-cobalt mining project to the East, and the development of the Sud-Cameroun Hevea rubber plantation a few hundred meters from its western border (UNESCO, 2018). Land adjacent to the reserve is increasingly allocated to private, often internationally owned timber concessions, hunting zones (whereby hunters can sport-hunt large game with a permit), community forests (whereby local people are able to access natural resources for subsistence purposes) and mining concessions, which overlap with the boundaries of the reserve (see figure 3-1). Combined with growing human pressure from adjacent communities, the pressures on resources within the reserve are augmenting (Allen et al. 2016).

All access and hunting by local people is prohibited within the DFR. Where access is permitted in the surrounding buffer zone, hunting is permitted only if: a) the target species are not in the list of protected animals (class A) which contains some commonly hunted and preferred species such as giant and tree pangolin; b) it relies solely on "traditional" implements, which excludes the most commonly used gears; steel wire snares and guns and; c) it is only used for household consumption, which is also rarely the case as much wild meat is both consumed and sold. Consequently, studies in Cameroon have found that most of the animals hunted are protected, and the hunting method used is generally illegal (Yasouka et al. 2015).

The DFR faces a number of challenges to its management. Despite growing threats to biodiversity, the DFR has no validated management plan in place; plan drafting and validations efforts have stalled because of the high costs and time required. The DFR is understaffed, and the eco-guards' (MINFOF park rangers) salaries are small (Bernad et al. 2001). Further, eco-guards have sometimes go unpaid for months. Culverwell et al. (1998) found that less than 10% of the required amount had been spent on the DFR since 1986. As such, staff are also under-resourced, often lacking vehicles or camping equipment to facilitate their work. Additionally, Bernad et al. (2001) found that just one eco-guard was responsible for 11680 hectares of forest, greater than double the IUCN recommendation of one ecoguard per 5000 hectares.

Institutional corruption at all levels of Cameroon's political system means that some officials ranking highly within government and military institutions have been found to facilitate, rather than prevent, the illegal hunting and trafficking of protected species such as gorillas and forest elephants. The knowledge that individuals responsible for enforcing the law are also those breaking it hinders the motivation of local people to protect biodiversity. A combination of growing threats to biodiversity, a lack of funding and poor management means that the DFR will likely soon be listed as a world heritage site in danger (UNESCO, 2018).

### 2.6.3 Conservation efforts in the Dja region

Conservation efforts inside the DFR currently focus on wildlife population monitoring and capacity building for park staff. ZSL run a long-term camera trapping effort within the reserve (see Bruce et al. 2017; 2018) and the African Wildlife Foundation (AWF) also worked, until very recently, to support park rangers and improve patrol effectiveness. ZSL have provided training in basic and tactical law enforcement for over 40 guards since 2018 (ZSL, 2018). Several other local and international NGOs and research institutions work in the communities around the periphery of the DFR, including Living Earth Foundation (LEL), the Royal Zoological Society of Antwerp (RZSA) the Association de la Protection de Grands Singes(APGS), Tropical Forest and Rural Development (TF-RD) and the Fondation Camerounaise de la Terre Vivante (FCTV).

Some of these actors work to provide livelihood and protein alternative projects to diminish hunting pressure for wild meat (see Darwin Initiative project references 24005 & 25015), while a new community hunting zone to encourage more formal and legal hunting practices is being established to the east of the reserve by FCTV. The Dja Actors Forum represents a yearly opportunity for NGOs and other conservation actors operating around the Dja to collaborate and share their progress and learnings.

Despite capacity building efforts from NGOs, a general lack of species identification training prevents some eco-guards from confidently distinguishing protected from unprotected species, a problem further complicated by the practice of smoking meat once hunted. As such, some eco-guards reportedly confiscate all wild meat they encounter from local communities, as they a) cannot be certain of the species, b) cannot be certain that it came from the community forests where communities are permitted to hunt and c) cannot be sure that the hunt method used to hunt the species was legal (see section 3.1.11). This conflict has reportedly aggravated relations between community members and park officials.

### 2.6.4 Local wild meat hunting

Wild meat has been shown to be an important source of income and subsistence for local people living close to the DFR; in 1996, it was estimated that individual hunters near the DFR could generate as much as $650 per year from selling wild meat (Ngnegueu & Fotso, 1996). Koppert et al. (1996) found that wild meat provides 30 to 80% of the overall protein intake of rural households in Cameroon. Studies on wild meat consumption indicate which species are being exploited and allow for generalisations regarding the likely impact of hunting on wildlife populations. In another old study, Dethier (1995) found that ungulates made up 88% of the wild meat captured in the DFR, followed by other (6%), rodents (5%), and primates (4%). However, without a detailed understanding of the quantity of wild meat extracted and produced over a given time period from a known area of forest, these studies provide only anecdotal assertions about the sustainability of wild meat hunting in any given area. They are also very out-of-date, given the fast-moving situation in the DFR and the likely major increase in wild meat hunting and other natural resource extraction over the last two decades.

Studies that estimate effort are even rarer, as they generally require an intensive survey of village hunters. During hunting studies in the village of Ekom, adjacent to the DFR, Jeanmart (1998) attempted to quantify trapping effort, and draw community maps of the distribution of trap lines. However, only 24% of hunters cooperated with the study, meaning that the densities of traps given in these reports are greatly underestimated.

More recent studies on wild meat around the DFR have focussed on characterising primate hunters and identifying the commodity chain that allows movement of primate meat from rural to urban areas (Tagg et al. 2018). Studies on the impact and importance of wild meat consumption and hunting exist elsewhere in Cameroon, but not around the DFR (see Bobo et al. 2015; Dounias, 2016). Considering the significant biological wealth in the DFR and its boundaries, and the clear importance of wild meat for communities, it is surprising how little recent literature there is that attempts to quantify impacts of wild meat hunting, identify emerging threats to biodiversity, and assess the consequent risk to local livelihoods and food security. Challenging forest habitats, low budgets and changing threats to biodiversity further complicate the ability to gather robust data. As such, there is a real need in this region for rapid and cost-effective methods of monitoring and assessing threats and wildlife species populations, that can also better integrate the knowledge of local people and provide robust wildlife species population estimates for species that are commonly hunted for wild meat and may be experiencing high pressures at the moment.

### 2.6.5 Case study villages

Further to the site criteria identified in my research proposal, two study villages were identified using the following additional criteria set out in table 2-5:

Table 2-5: Criteria for the selection of study villages adjacent to the DFR

| Criteria | Justification |
|---|---|
| The villages and their hunting areas are accessible. | Myself and the research team need to be able to access the site in line with the University risk assessment. |
| Participants are willing to host us who regularly use the forest resources and are willing to discuss hunting and species they encounter during their daily lives. | Respondents need to be willing to take part and willing to discuss potentially sensitive information. |
| Study villages that potentially represent two different hunting typologies or where species are exposed to different threats. | To be able to draw comparisons between estimates obtained from participants in one site compared to another, so as to have more generalisability in the assessment of the usefulness of LEK under different circumstances, and so as to be able to draw conclusions about the threat levels and sources of threat in different village types around the DFR. |

Due to the potentially illegal nature of hunting at these study sites and to maintain anonymity at the community level, I will not name the villages in this study, referring to them as village 1 and village 2 throughout this thesis instead.

Village 1 in the South East and Village 2 to the North East of the DFR were selected as in-depth study sites for this research. Both villages are surrounded by a mix of different land managements and rely heavily upon forest resources. During my scoping trip in February 2016, people in both villages were welcoming towards me and the team, and welcomed the objectives of the research, showing willingness and trust to discuss wild meat hunting and the animals encountered in the forest. The location of both villages within community forests, may make them feel more at ease when discussing wild meat hunting and other activities, as they have the right to hunt.

The villages have different characteristics and as such provide an interesting contrast to one another. Village 1 is a small village (c. 90 households), relatively isolated, far from the main road and very close to the boundary of the DFR (c. 7km). In 2015, a new logging road arrived past the village, linking the village to the main road network and wild meat market towns. In contrast, village 2 is medium sized (c. 150 households) and located on the main axis road that links the east of Cameroon to Yaoundé, c. 15km from the reserve. The village is located adjacent to timber concessions with long-term contracts (25+ years), meaning that noise and habitat disturbance in these areas has occurred over prolonged periods of time with implications for surrounding wildlife. However, residents of village 2 reported that the village was first established in that location around the time of WWI, in part, due to the abundance of wild meat that could be found in the surrounding forests, and that hunting was once an easy and productive livelihood activity. Both villages have their own community forest, granting hunting rights to of class B and non-classified species to community members for subsistence purposes. Throughout this thesis, I expect village 1 to result in higher detection and occupancy rates than village 2, due to their more remote location and greater proximity to the reserve, and a higher abundance of biodiversity within their community forest as a result of their location and fewer drivers to hunt for commercial purposes.

During the scoping trip, villagers in village 1 reported that finding food and resources were quite easy while in contrast village 2 reported having to walk further and further and some now hunt inside the DFR. No studies on wild meat have been conducted in village 1 and no NGOs are active there, while in contrast village 2 has been subject to past research and efforts to tackle the problems associated with hunting in the region.

### 2.6.6 Species selected for this study

Table 2-6 shows the species selected for inclusion in this study. During the scoping trip to both villages in February 2016, I had informal conversations with people about the animals that are important to them, both for food and for their culture. I wanted to understand what species they saw regularly, which species were seen to be a pest, and which ones were considered to be rare, but still present within the community forests. To do this, I used photographs of over 30 mammal species that I knew to both be present in the region, plus some that were not present, to assess how truthful and knowledgeable people would be about whether these species could be found in their forests. When selecting the final species, I focussed on mammal species as they represent the group of animals most important for wild meat hunting. Table 2-6 shows the species that were selected and the justification for their inclusion.

Table 2-6: Table of the species included in this study and why they were chosen. IUCN categories: CR=Critically endangered, EN= Endangered, NT= Near threatened, LC= Least concern. Hunting class: A= illegal to hunt; B=species can be hunted for subsistence only in community forests. No species can be hunted from within the reserve.

| Species | Hunting class | IUCN Status | Reason for inclusion in study |
|---|---|---|---|
| Western lowland gorilla *(Gorilla gorilla)* | A | CR | Rare, bold diurnal species. Social and sleep in distinct nests, mostly on the floor but occasionally in trees. They have experienced an 18.75% decline in species from 2003-2013 (IUCN, 2016) driven by hunting for wild meat, habitat loss and disease. It is illegal to kill or trade gorilla for their parts |
| Chimpanzee *(Pan troglodytes)* | A | EN | Bold diurnal species. Most abundant and widespread, but declines satisfy endangered criteria, due to hunting for wild meat, habitat loss and disease (IUCN, 2016). Cameroon represents a stronghold for the species. The IUCN (2016) calls for 'Better understanding of the interactions between people and chimpanzees, and involving local stakeholders in participative management, especially outside or at the periphery of protected areas.' |
| Putty-nosed monkey *(Cercopithecus nictitans)* | - | LC | Common bold, vocal, diurnal, arboreal species, although their population trend is decreasing. They commonly live in groups of between 12 and 30 animals and are locally threatened in some areas due to deforestation and wild meat hunting. Commonly hunted in the villages for wild meat and are easily seen in the forests around village 1. |
| Yellow-backed duiker *(Cephalophus silvicultor)* | B | NT | Shy, diurnal terrestrial species, they are the most widespread of duikers and reasonably common, but intensive hunting and snaring has resulted in decline of 20-25% over 3 generations. They occur in moist lowland and montane forests (primary and secondary), forest-savanna mosaics, gallery forests and are hunted for wild meat in both villages |
| Bongo *(Tragelaphus eurycerus)* | A | NT | Bongo are a flagship and trophy species. Their occurrence in dense forest habitat, patchy distributions, wide-ranging patterns, retiring behaviour and nocturnal activity patterns hinder any reliable density estimation (IUCN, 2016). Habitat loss and wild meat hunting led to 20% decline over 3 generations. |

| | | | |
|---|---|---|---|
| Blue duiker *(Philantomba monticola)* | - | LC | Widespread and abundant with total population numbers estimated to be greater than 7 million. The species can withstand hunting pressure and habitat degradation, adapt to increasing human colonization, and can persist in small patches of modified or degraded forest and thicket, even on the edge of urban centres. They are heavily hunted for wild meat. |
| Sitatunga *(Tragelaphus spekii)* | B | LC | Shy, drab, most active at dawn and dusk, males have a loud bark. They live alone or in small all female groups. They occur in tall and dense vegetation of perennial and seasonal swamps, marshy clearings within forests, riverine thickets, and mangrove swamps (IUCN, 2016). Their cryptic nature and the relative inaccessibility of their habitat makes reliable estimates of abundance difficult. Habitat degradation and intensive meat hunting may cause it to disappear from many areas. |
| Red river hog *(Potamochoerus porcus)* | B | LC | Very vulnerable to deforestation, less so to hunting but this is dependent on hunting intensity. Although considered a common and abundant species (Oliver 1995), the Red River Hog is one of the preferred species for subsistence hunters across its range in Africa, a primary prey species harvested for commercial purposes within the wild meat trade in Cameroon (Ayeni et al. 2001, Fa et al. 2006, Wilcox and Nambu 2007, Wright and Priston 2010, Macdonald et al. 2012). Red River Hogs are reported to damage maize crops in the study villages, especially during the rainy season, and for this reason the species is persecuted by farmers. |
| African brush-tailed porcupine *(Atherurus africanus)* | - | LC | Porcupines can cause damage to crops and agricultural fields. It is subject to extensive exploitation for human consumption in much of its range (being a ground-dwelling, large-sized rodent, capable of producing up to 2 kg of meat). This is a favoured species in Cameroonian wild meat markets. |
| African golden cat *(Caracal aurata)* | A | VU | Rare, cryptic, nocturnal species. Golden cats are often not a primary target species, but are frequently killed by wire-snares probably owing to similarities in body size and trail use to target species such as duikers. In an area of moderate wild meat hunting, Golden Cats were recorded at less than a quarter of the population densities that they are found at in pristine areas (Bahaa-el-din et al. in prep). African golden cat has recently been recorded by Bruce et al (2018) for the first time ever in the Dja Faunal Reserve. Where more intense hunting occurs, such as in village hunting areas, camera trap and wild meat studies have not record the species despite the presence of suitable habitat contiguous with the main forest of the Congo Basin. |

| | | | |
|---|---|---|---|
| Servaline genet *(Genetta servalina)* | - | | This species is present in primary and secondary lowland, submontane and montane forests and gallery forests. However, the species is also considered a pest in village 1, reportedly eating farmers crops (pers comms.) Servaline genets are common in wild meat markets and the skins are used among both Mbuti and Ba'aka (Van Rompaey and Colyn 2013). However, hunting of this species is not currently considered to pose a threat to their populations. |
| Forest elephant *(Loxodonta cyclotis)* | A | VU | Although forest elephants are taxonomically and functionally unique, the IUCN recognizes only one species of African elephant (Loxodonta Africana), which is categorized as Vulnerable on the IUCN Red List (Blanc set al. 2007). However, there is significant geographical variation in the level of threat, and based on a regional assessment the Central African forest elephant is Endangered (Blanc et al. 2007). Illegal hunting remains a significant factor in some areas, particularly in Central Africa. |
| Giant pangolin *(Smutsia gigantean)* | A | VU | Giant pangolin occurs in lowland tropical moist and swamp forest, and in forest-savanna-cultivation mosaic habitats. As with other pangolins, S. gigantea is subject to widespread exploitation for wild meat and traditional medicine and is regularly recorded in wild meat markets. |
| Tree pangolin *(Phataginus tricuspis)* | A | VU | This species occurs predominantly in moist tropical lowland forests and secondary growth, but also occurs in dense woodlands, especially along water courses (Kingdon 1971, Gaubert 2011) although the species can adapt to at least some degree of habitat modification. White-bellied pangolins are predominantly nocturnal and equally at home in trees and on the ground (Pagès 1975). The species feeds exclusively on ants and termites. White-bellied pangolins are subject to widespread and often intensive exploitation for wild meat and traditional medicine, and are by far the most common of the pangolins found in African wild meat markets.<br><br>Note: While participants were able to describe the difference between white and black bellied pangolin in conversation, they were not able to confidently recall what species of pangolin they saw in the seasonal interviews. As such, 'tree pangolin' refers to both the white and black bellied pangolin, although detections are more likely to be the white-bellied pangolin (Phataginus tricuspis) as opposed to the less common black-bellied pangolin (Phataginus tetradactyla). |

### 2.6.7 Research ethics

For the semi-structured seasonal interviews, all people over the age of 18 in the village were invited to participate. In all cases, the research team explained the objectives and how the data would be used. Free informed consent was verbally obtained, and participants knew they could stop participating at any time. To ensure personal anonymity, identification numbers were allocated to each hunter and used on all datasheets. Village locations are not recorded to ensure anonymity at the community level, nor are easily identifiable village characteristics presented here (e.g. geographical features).

For the daily hunter diary work, hunters were selected based on their willingness to participate in the study over 6-12 months, and on the quality and openness of their responses during previous surveys within this study. Hunters completing the daily diaries were given a small compensation for their participation when their data sheets were checked by the key contact on a weekly basis (equivalent to £0.10p a day). However, the compensation was such that it did not incentivize hunters to falsify and add data (i.e. hunters who recorded that they had seen or hunted nothing were compensated the same as those how had recorded many sightings). The research team worked in both villages for over 1 year prior to this research, and had built relationships with the participants.

The research was approved by Oxford University's Central University Research Ethics Committee (CUREC) (R45771/RE001). See appendix A-1 for the ethical scripts used to explain this work to all participants.

# Chapter 3

## Using social ecological systems thinking to identify wild meat hunting motivations and intensity in Cameroon

*(Smoked duiker and chili sauce prepared by one of the women in the villages)*

## 3.1 Introduction

Wild meat is a valuable non-timber forest product across tropical Africa, Asia and the Neotropics (Robinson & Bennett, 2004), helping local people to meet their food and livelihood needs (Nasi, 2011). Yet, the local extirpation of hunted species is widespread. A meta-analysis demonstrated declines of up to 83% in mammal abundances across the tropics as a result of hunting pressure (Benítez-López et al. 2017); West and Central Africa are particularly hard hit (Milner-Gulland et al. 2003). The cost of wildlife loss falls most heavily on rural populations who most depend on wild meat. Therefore, overhunting represents a problem both for food security and conservation (de Merode et al. 2004; Bennet et al. 2006; Wilkie et al. 2016).

The conservation of animals hunted for wild meat focusses on sustainable use, in recognition of the importance of wild meat for food security in rural communities (CBD, 1993; Milner-Gulland & Akcakaya, 2001). To assess the sustainability of hunting, species populations must be monitored. Conventionally, biological indicators, such as the Robinson & Redford, index (1991) or the Bodmer index (Robinson & Bodmer, 1999), are used to provide a benchmark of sustainable population production. However, these measures of sustainability provide a snapshot in space and time which, in turn, encourages a static assessment of hunting sustainability. Hunting systems can quickly be knocked out of equilibrium by environmental, economic or political shocks (Ling & Milner-Gulland, 2006). Purely biological assessments of sustainability do not account for the dynamism of hunting systems and may result in unhelpful assessments of unsustainability, further promoting the need for “fortress conservation", with implications for the legacy of conservation and impacts on human well-being.

Socioecological systems are defined by Folke (2004) as 'dynamic, interacting associations of social and environmental components'. Socioecological systems frameworks have clear applications for assessing wild meat hunting systems (Nasi et al. 2011; Coad et al. 2013; González-Marín et al. 2017), acknowledging that environmental and social circumstances may greatly affect how a wild meat hunting system operates. In their review of the limits of traditional methods currently used to investigate sustainability of wild meat hunting, van Vliet et al. (2015) found that that trends in wild meat hunting result from environmental, sociodemographic and temporal trends, at both the hunter and the village scales. Compared to purely biological approaches, SES frameworks provide a more nuanced understanding of the heterogeneity and uncertainty surrounding the effect of overhunting on different species and between locations (Milner-Gulland, 2012), allowing for recommendations to move the hunting system in question closer to sustainability if required (Zurlini et al. 2006; van Vliet et al 2015).

Milner-Gulland, 2006) classifies wild meat hunting in Africa under three broad scenarios (table 3-1). Each typology has its own characteristics and drivers, resulting in hunting systems that vary in sustainability and conservation priority.

Table 3-1: Hunting system typologies , informed by descriptions from Milner-Gulland (2006)

| Typology | Conservation priority | Threat | Primary wild meat source | Species | Development implications |
|---|---|---|---|---|---|
| Frontier bonanza | High | Opening up of primary forest | Forest | Rapid extirpation of vulnerable and large mammals | Need to safeguard the livelihoods of local people already there |
| Declining source-sink | Unknown | Varied – bioeconomic interactions, external pressures, hunting | Mixed, caused by heterogeneity in hunter behaviour | Build-up of hunter effort reduces species populations over wide area | Unknown/varied. |
| Mature wild meat market | Low | Long-term exploitation | Farm-bush matrix | All resilient, fast growing. Vulnerable species extirpated | Need to understand the potential of these areas to produce a sustainable wild meat supply |

The key components that may shift the system from its current state, to one in which hunting is more or less sustainable, depend upon the typology of the hunting system (Gibson, 2006). These components should be placed within a holistic framework that provides a theoretical representation within which to explore their interactions and describe the drivers of change (figure 3-1).

At the hunter scale, individual decisions concerning where and how hunters hunt may alter village-level distribution of prey availability, productivity and composition (Wilkie & Carpenter, 1999; Rowcliffe et al. 2005). Hunter age (Fonkwo et al. 2017) or hunting method may alter their wild meat preferences and effectiveness at hunting (van Vliet & Nasi, 2008), which determine whether sensitive species are at risk of becoming more intensively hunted and therefore depleted (Peres, 2000; Newing, 2001). Village scale characteristics such as roads may facilitate access across the system (Sirén et al. 2012; Ichikawa et al. 2016), while rivers may hinder access (Barber et al. 2015). Habitat type can affect the availability and type of prey (Gavin, 2007; van Vliet & Nasi, 2008; Martínez-Cruz, Juárez-Torres & Guerrero, 2017). In turn, such characteristics may influence individual hunter decision-making about where to hunt.

Central-based foraging theory dictates that hunters travel only as far as required, therefore local depletion may be evident close to the village. As animals become depleted, hunters move further away to hunt. However this theory does not recognise that external pressures may also motivate hunters to travel further than is required to meet simple quantity demands, for example to hunt larger animals further afield to sell (MacKenzie et al. 2012). The development of wild meat markets (Ling & Milner Gulland, 2008; Brashares et al. 2011), access to infrastructure (Franzen, 2006), or crop price fluctuations (FAO, 2013), can influence spatial trends in offtake. Further, cultural events or changes in livelihood opportunities may influence hunting intensity throughout the year (van Vliet & Nasi, 2008; Alexander et al. 2015). Interactions between the variables at each scale in the system drive both the rate and trends in wild meat hunting (van Vliet et al. 2015).

In this chapter, I use a social-ecological systems approach to combine social and biological data at the village and hunter scale and make inferences about the status and role of wild meat hunting in two contrasting village systems (figure 3-1).

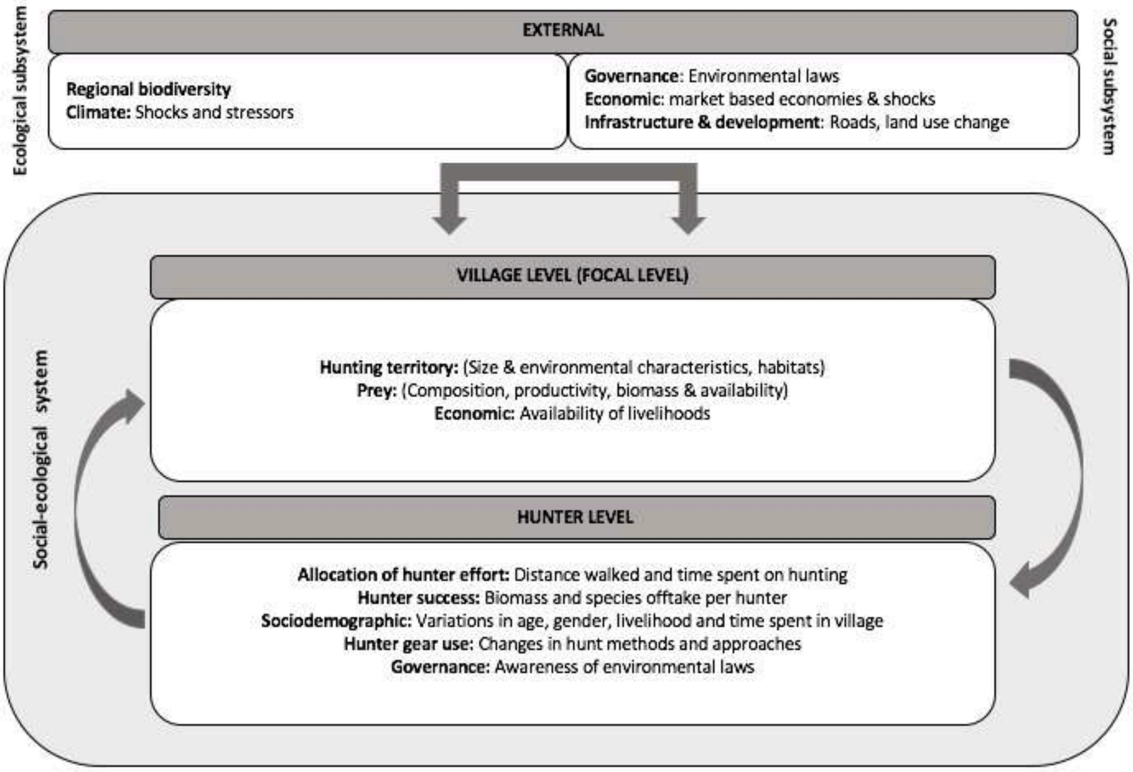



Figure 3-1: Integrated SES model framework for this study (adapted for use from the Resilience Alliance Handbook 2010).

I identify key external pressures and their relative influence on the spatial-temporal trends of hunting within these systems. I focus on mammal species, the most important class to tropical forest hunters (Robinson & Bennett, 2000). My objectives, following the key lessons outlined by van Vliet et al. (2015) are to: 1) identify the village level characteristics affecting spatial and temporal heterogeneity of wild meat hunting; 2) investigate the social drivers affecting hunter decision-making and 3) identify the wider drivers of change affecting the wild meat hunting systems at present, and into the future. Finally, I draw some preliminary inferences about the systems' position in the wild meat typology and their sustainability, and recommend how sustainability can be promoted to benefit conservation, local food security and livelihood. Table 3-2 displays the research questions and hypotheses relating to each research objective.

Table 3-2: Environmental and social drivers of change considered in my conceptual model (figure 3-1), their reasons for inclusion and how they will be assessed. Research objectives in this chapter fall under overall thesis objective 1: Identify the individual and village level drivers of hunting, and the threats to hunted species in the Dja Region.

| Research objective | Research question | From the literature | Hypotheses |
|---|---|---|---|
| Identify the spatial and temporal heterogeneity of wild meat hunting in both hunting systems (Village level). | Q1. What geographical variables define the hunting territories at both hunting systems? | Shape and size of a hunting area can indicate system resilience; unhunted sites may provide valuable 'source' areas, allowing hunted animal populations to regenerate (Gavin, 2007). | The hunting territory of village 1 will be smaller than village 2 because of its proximity to the source area of the DFR (see Q2). Both villages have similar habitat types. |
| | Q2. What are the environmental determinants of wild meat hunting? | Hunters have been found to stay close to roads, rivers and villages where possible, minimizing the cost of travelling to less accessible places (Siren et al. 2013; 2015). Hunters may also gravitate to certain areas because of variation in abundances (Fimbel and Curran, 2000). | Hunters in both villages will hunt close to roads and the village to minimize effort. |
| | | Proximity to source areas such as reserve affect prey availability; reserves may act as refuges for wildlife. Illegal hunting may occur within their boundaries and provide spillover effects (whereby hunters benefit from wildlife emigrating from the reserve into surrounding areas) outside them. Both processes may result in higher offtake of vulnerable species (such as ungulates) but not of more generalist species (such as rodents) in areas close to reserves (Fa et al 2006; McNamara et al 2014). | Wild meat offtake of preferred and/or vulnerable animals, such as primates and large mammals will be higher in village 1 due to proximity to the protected area. |

| | | | |
|---|---|---|---|
| | | Habitat type can affect the type (van Vliet & Nasi, 2008) and volume (Robinson & Bennett 2004; Gavin 2007) of meat hunted. | Hunters will target forest habitats, where more valuable animals can be found and abundance may be higher, optimizing their returns and providing meat for sale and to eat. |
| | Q3. What can the catch composition tell us about hunting intensity? | Large animals with low reproductive rates such as elephants and apes are most susceptible to over-exploitation (Duncan et al. 2002). In contrast, smaller more productive species, such as larger rodents and smaller antelopes tend to be more tolerant to relatively intensive hunting (Wilkie & Carpenter 1999). | A greater proportion of rodents in the village 2 catch will be observed, suggesting intensive hunting, while a greater proportion of ungulates and primates will be observed in village 1 catch, suggesting limited depletion in comparison. |
| | | Optimal foraging theory states that medium to large-sized animals are preferred prey and thus targeted first by hunters (Peres, 2000; Newing, 2001). | Hunters in both villages target large mammals where possible. Catch composition in village 1 will be composed of more large mammals. |
| Investigate the social drivers that affect hunter decision making (Hunter level). | Q4. What are the hunter level determinant of wild meat hunting? | Age (Fonkwo et al. 2017) and gender (Hofner et al. 2017) impact the prevalence of hunting, the method used, success rate, and motivation for hunting. | Young hunters are more successful than older hunters because they are better able to travel further and hunt for longer. Most hunters are male. However, it may be that more experienced hunters are more successful, therefore stay hunting. |
| | | Commercial hunters predominantly use guns and hunt larger mammals further from the village, destined for sale at market (Duda et al. 2017). | Hunters in village 2 use guns more than hunters in village 1 due to greater connectedness with markets and |

| | | | |
|---|---|---|---|
| | | | infrastructure required to buy ammunition and sell wild meat. |
| | Q5. What are the livelihood and food security motivations for hunting in both villages? | Wild meat provides an important source of protein and livelihood (Nasi et al 2011), especially where alternatives to hunting are scarce (de Merode et al. 2004). | Both villages rely on wild meat for subsistence and livelihoods, but to varying degrees. Hunters in village 1 hunt predominantly for subsistence due to lack of connection with wild meat markets. In contrast, alternative income activities mean hunters in village 2 can generate income elsewhere. However, they are better connected to external markets and infrastructure, facilitating their ability to sell wild meat. |
| | Q5. What is the temporal variation of hunting? | Hunting intensity varies with season and agricultural work (Alexander et al 2014; van Vliet & Nasi 2008). | Trends in offtake are linked to agricultural activities, such as during crop planting in the small rainy season, and the harvest of cash crops in December. |
| | | Hunting is used as an important coping strategy in times of financial stress (Alexander et al 2014; Coad et al 2010) and cultural events (van Vliet & Nasi 2008). | Hunting is more prevalent at times of environmental stress (e.g. end of the dry season in January/February) and before cultural events such as Christmas and international women's day. |
| | | Because guns can be used to target larger animals for sale, an increase in gun hunting in times of financial | Gun hunting will increase close to cultural events such as Christmas and international women's day, and to pay for school fees. |

| | | | |
|---|---|---|---|
| | | emergency has been observed in the Congo (de Merode 2003) and in Cameroon (Solly 2004). | |
| Identify the wider drivers of change and how they affect the wild meat hunting systems now and into the future. | Q6. Is there a link between distance walked, hunting method and the fate of wild meat? | External influences such as the development of wild meat markets (Ling & Milner Gulland, 2008; Brashares et al. 2011), access to infrastructure (Franzen 2006) or crop price fluctuations (FAO, 2013) influence hunter decision making and behaviour. | There will be evidence of commercialization by hunters in village 2, shown by an increase of large mammals hunted far from the village, with guns, which are then sold. There will be little/no evidence of commercialization in village 1; animals caught will range in size, using several hunt methods without dominance of guns. The proportion of wild meat sold will not increase with the distance from village a hunter travels. |

## 3.2 Methods

### 3.2.1 Study site

For study site description see section 2.1.10

### 3.2.2 Data collection

Using a mixed-method approach, I gathered biological and social data on wild meat hunting patterns and motivations. Ten trained hunters in each village kept icon-based daily diaries from May 2017 to April 2018, providing information on the species hunted, hunt location, habitat type, the hunting gear used and whether wild meat was sold or eaten. 10 hunters per village was deemed a manageable number of participants to ensure that all were fully trained and that the research team were able to assist them if needed, while also providing sufficient data points so that the effect of any sociodemographic differences in age or ethnicity, for example, could be detected. Daily diaries were complemented with participatory maps developed during two focus groups per village (Newing, 2011), which served to map animal detections and assess the spatial heterogeneity of hunting pressure (van Vliet & Nasi 2008). Ideally, data would have been collected over the same months, but due to logistical constraints, data collection in village 2 was delayed, which means that comparison during some months is not possible. However, data overlap during 6 of the 9 possible months, allowing comparisons to be drawn. Informal and semi-structured interviews were conducted with hunters and other community members to better understand perceived changes to species populations and hunting practices and add further contextual data to the findings. A key contact was employed within each village to provide assistance when needed and help to collate the datasheets. In both cases, he was a trusted community member and not associated with the village leader's family. They were involved in the creation of the community forests as a community representative, therefore familiar with reading maps. The accuracy of data collection was checked with opportunistic hunter follows at the start of the data collection (n=9) and the key contact was in weekly contact with me to feedback on progress. Each data collection method presents different strengths and potential biases; efforts made to account and control for these are outlined in table 3-3.

Indirect questioning methods such as the Randomised Response Technique or Unmatched Count Technique (Nuno et al 2013; Hinsley et al 2017) aim to provide respondents with greater levels of privacy and anonymity (Chaudhuri & Christofides, 2013). I contemplated using such methods in this study, and in other chapters, but did not do so for the following reasons:

1. Witchcraft is highly prevalent in the study region, and most people I spoke to, regardless of religion or age, both believed in and feared it. As such, I didn’t want to risk a situation occurring where I had used indirect questioning methods, then something bad happened to anyone involved and for me to be accused of witchcraft. The consequences of accusation are serious and often dangerous for the accused.
2. Related to point 1, I found early on that people much prefer you to be upfront and ask questions directly. When I was initially nervous about asking people about hunting, and tried to ask in a roundabout way, I was frequently met with an eye roll and “If you have something to ask me, just ask”. I found participants to be more wary when I didn’t just ask questions in a very matter of fact way as they then felt I myself had some reason to hide or wasn’t being honest with them.

Table 3-3: Table outlining the different datasets used to collect information at the village and hunter scales, and their strengths and potential biases

| Method | Method strengths | Potential bias | Actions taken |
|---|---|---|---|
| Daily diaries | Good for hunter-level analyses of predictors of hunting success and choice of hunting location | Relatively small sample size. The hunters selected could potentially be more or less active or more prone to hiding the truth than the general hunting population. | The respondents, while not reflective of the whole village, were reflective of the hunting community. Responders that hunt were identified through a sociodemographic survey earlier in the year. A range of ages, livelihoods ethnicities and demographics were included were possible, to ensure the sample is as representative as possible and the patterns of hunting reflect broader trends in hunting across the villages. Hunters were selected from different households and different parts of the village, to ensure spatial representation of offtake. Hunters that hunt together were not selected, to prevent duplication of records. |
| | | Potential errors in data collection | Thorough training was conducted with each participant at their households, to ensure that participants were comfortable with the daily diaries and how to locate and record sightings and hunting events from the participatory map (which all were already familiar with through previous surveys). |
| | | Fear of being reported may limit recording of sensitive data: | No names were recorded on the data sheets, ID codes ensured the respondents identity remained anonymous. Ensuring anonymity to respondents helped reduce reporter bias. Furthermore, hunting is a daily activity and combined |

| | | | |
|---|---|---|---|
| | | | with the lack of enforcement of hunting laws, means hunting is not a particularly sensitive topic in either village |
| Participatory mapping | Excellent tool to help define village territories, key landmarks and environmental characteristics that may define why hunters use certain areas. | The zone of use and therefore the focal study area is dynamic and can change over time.<br><br>Depiction of the study site from participatory mapping may not be 100% accurate | No static inferences about the sustainability of the current system are made, but rather this information is used to better understand how the system may change over time.<br><br>Data provided in the focus groups were ground truthed used GPS, so that the result incorporates local ecological knowledge with GIS data. The map was updated throughout with additional data/landmarks and the accuracy was checked by participants and the wider community throughout the study. |
| Informal interviews | Interviews with both hunters and other villagers to get a broader understanding of people's well-being, resilience to change. | Data may be inaccurate<br><br>Interviewees may not be representative | All open interviews were led by the respondent to reduce bias. Where a point of interest was raised, we asked similar questions to other respondents in order to triangulate and verify information from respondents. Anonymity was again assured to all interviewees.<br>The research team attempted to reach a balance of gender and ages to represent a diversity of views. |

### 3.2.3 Ethics

For a description of the ethical procedures followed see section 2.1.17

### 3.2.4 Data analysis

To explore the village level characteristics of the hunting systems, participatory maps were digitized and combined with shapefile data of roads, rivers, and surrounding land designations (i.e. protected areas, timber concessions) in QGIS. Using the environmental variables extracted from GIS and measured at sites where hunting occurred, principal component analysis (PCA) (Jolliffe & Cadima, 2016) was used to explore the relative importance of village level environmental characteristics on hunting and to present the geographical variables of the hunting systems in an anonymous way. The variation in environmental variables between locations do not show correlation, but partition the variation in hunting locations.

To investigate the catch composition and make inferences about the intensity of hunting in both systems (Wilkie & Carpenter 1999; Duncan et al. 2002; Rowcliffe et al. 2003), animal detections recorded as 'hunted' in the daily diaries were extracted and presented by species, species order and body mass group. Animals were classified as small (<3kg), medium (3-10kg) and large (>10kg) (Abernethy et al. 2010; Coad et al. 2013). Total biomass offtake with distance from village was calculated by summing the total body mass of all the animals hunted at a given distance from village.

To investigate the environmental and social variables that best predicted hunting success in both villages, I used generalized linear models (GLMs) with a binomial distribution and a logit-link function, using the 'glm' function in R.

Environmental and social explanatory variables were selected a-priori, based on the literature and my experience (appendix B-1). To avoid collinearity among variables in the model, Spearman's correlation coefficients were calculated for pairs of variables. None were highly correlated (r >0.8) therefore all were kept in the model (see appendix B-2 for correlation analysis output). The models with delta AIC <2 were shortlisted. An interaction term was included between variables for which an interaction was hypothesized a priori; these were village and distance from the reserve, because the effect of distance to the reserve on hunting success may differ between villages.

To explore the role of hunter decision making on hunting intensity, relationships between the distance the hunter walked to hunt, their species offtake, changes in the hunt gear used and the impact of cultural and farming events on the seasonal and temporal variation in offtake were assessed. Multinomial GLMs were run using the 'mlogit' package in R (Croissant, 2019) to draw inferences about the variables that best predict the fate (rotten, sold, lost or eaten) of the wild meat hunted. Finally, interviews were transcribed and coded in nVivo (NVivo, 2011), to identify key themes raised within the discussions. All statistical analyses were conducted using R 3.4.1 (R Core Development Team, 2017).

## 3.3 Results

A total of 6325 animals were recorded as harvested in the daily diaries during the study (table 3-4). Similar numbers of individual animals were hunted in both villages. However, hunters in village 2 recorded fewer detections of wildlife and fewer hunting trips than village 1.

Table 3-4: Summary of the descriptive results from 10 hunters in each village, obtained from the hunter diaries. 'Detections' means the total number of times individual animals and their signs were detected by hunters.

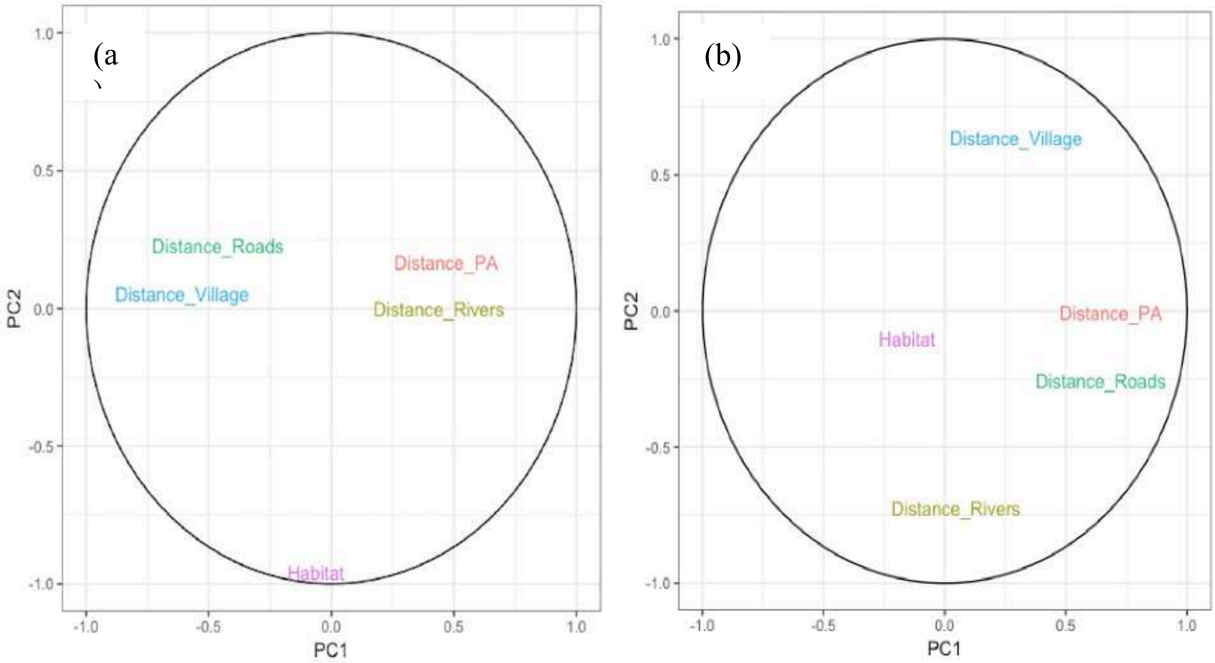
| Variable | Village 1 | Village 2 |
|---|---|---|
| Total days diary | 262 | 202 |
| Age | 18-25= 3<br>26-40=7<br>40-65=0<br>65+=0 | 18-25 =3<br>26-40 =0<br>40-65 =6<br>65+=1 |
| Gender | Men= 8<br>Women = 2 | Men= 8<br>Women = 2 |
| Time spent in forest on a given trip | 1 day= 96.4%<br>2-3 days= 2.5%<br>1 week= 0%<br>2 weeks= 0.4%<br>1 month= 0.2% | 1 day= 15%<br>2-3 days= 16%<br>1 week= 15%<br>2 weeks = 11%<br>1 month= 8%<br>No answer/uncertain = 35% |
| Detections | 6276 | 4724 |
| Detection time of day | Day = 85%<br>Night = 15% | Day = 59%<br>Night =41% |
| Total trips to forest | 2531 | 1356 |
| Number of trips resulting in successful hunt | 1860 (73%) | 877 (64%) |
| Total animals hunted | 3172 | 3153 |
| Methods | Gun= 39%<br>Snare= 54%<br>Dogs/Mixed = 7% | Gun= 56%<br>Snare= 18%<br>Dogs/Mixed=26% |
| Number of animals hunted by size | Small = 231 (7.3%)<br>Medium = 1550 (48.9%)<br>Large = 1391 (43.9%) | Small = 440 (14%)<br>Medium = 1575 (50%)<br>Large = 1138 (36%) |

### 3.3.1 Village-level variables affecting the spatial heterogeneity of wild meat hunting

To identify the spatial and temporal heterogeneity of wild meat hunting at the village scale, I first identified the geographical variables that define both hunting territories. The hunting territories in both villages are comprised of an agricultural zone containing a mosaic of agricultural land, surrounded by a community forest comprised of semi-deciduous and riparian forest. The territories are bordered by the Dja river and the Dja Faunal Reserve (DFR) in village 1, and by timber concessions in village 2. Both villages identified a hunting territory substantially larger than their community forest. Hunters in both villages reported entering the DFR to hunt (appendix B-3 & B-4).

PCA analysis was carried out to reflect the geography of the villages where hunting takes place. Environmental features have a strong effect on hunting presence in both villages (figure 3-2). Principal component 1 (PC1) shows that hunting in village 1 occurs either in a "road/village" area or a "river/PA" area. PC2 shows that hunting is determined by habitat, suggesting that hunters actively target animals from certain habitats, or perhaps that certain habitats are more productive. In village 2, habitat is not particularly important, but PC1 shows that hunting either occurs in a "distance from PA/roads" cluster, or not. PC2 shows that hunting either occurs close to the village or near rivers.

Figure 3-2: PCA plot of environmental determinants of hunting. PCA component loadings for each characteristic are plotted for (a) village 1 and (b) village 2, to allow insight on possible interpretations of the first two components identified in the PCA. For village 1, PC1 accounts for 41.5% of the variance, while PC2 accounts for 20%. For village 2, PC1 accounts for 44.7% while PC2 accounts for 26.4%




GLMs were used to identify the environmental and social variables that best predicted hunting success; individual hunting trip success is strongly affected by habitat type, village, distance from reserve and hunting method in both villages. The top model had strong support with a relative weight of 46%. The next-best model (weight = 33%) included an additional variable, so for parsimony, I chose to accept the top model, which includes 4 geographic variables (village, distance from reserve, distance from road and habitat type) and one hunter behaviour variable (method) (table 3-5). All other models had delta AIC>4 and weight <= 5%, so were not pursued.

Table 3-5: Binomial GLM top models, indicating the characteristics associated with whether hunting trips are successful (1) or not (0). Intercept: village=1, habitat=forest, method=gun. Relative variable importance: village 0.84, distance reserve 0.84, habitat 0.84, method 0.84, distance reserve*village 0.84, distance road 0.79, distance village 0.33. Table displays models with an AIC < 2.

| Rank | Intercept | Habitat (Degraded) | Habitat (Farm) | Habitat (clearing) | Method (Snare) | Method (Mix) | Village 2 | Dist reserve | Dist reserve*V2 | Dist Road | Dist Village | df | Delta AIC | Weight |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 1.41 | -1.28 | -0.87 | -1.35 | 0.71 | -1.69 | 0.23 | -0.07 | 0.12 | -0.06 | | 10 | 0 | 46% |
| 2 | 1.35 | -1.28 | -0.87 | -1.35 | 0.71 | -1.68 | 0.08 | -0.06 | 0.12 | -0.07 | 0.02 | 11 | 0.68 | 33% |
| 3 | 1.20 | -1.29 | -0.88 | -1.36 | 0.70 | -1.69 | 1.37 | -0.06 | 0.05 | | | 9 | 4.11 | 5% |

Table 3-6: Parameters of the best performing GLM. Dependent variable is hunt success (1) or failure (0).

| Reference | Variable | Estimate (SE) | Lower 95% CI | Odds ratio | Upper 95% CI | P value |
|---|---|---|---|---|---|---|
| | Intercept | 1.42 (0.12) | 3.23 | 4.12 | 5.27 | <0.001 |
| | Distance reserve | -0.07 (0.02) | 0.88 | 0.92 | 0.97 | 0.003 |
| | Distance reserve*village2 | 0.12 (0.04) | 1.04 | 1.13 | 1.22 | 0.002 |
| | Distance road | -0.06 (0.03) | 0.89 | 0.94 | 0.98 | 0.01 |
| Habitat | Farm | -0.87 (0.17) | 0.29 | 0.41 | 0.58 | <0.001 |
| (reference = forest) | Degraded | -1.28 (0.18) | 0.19 | 0.27 | 0.39 | <0.001 |
| | Clearing | -1.34 (0.15) | 0.19 | 0.26 | 0.35 | <0.001 |
| Method | Snares | 0.71 (0.1) | 1.67 | 2.03 | 2.48 | <0.001 |
| (reference=gun) | Dogs/mix | -0.69 (0.09) | 0.04 | 0.13 | 0.22 | <0.001 |
| Village (reference= village 1) | Village 2 | 0.23 (0.60) | 0.38 | 1.26 | 4.15 | 0.69 |

$R^2 =$ 0.17 (Hosmer-Lemeshow), 0.19 (Cox-Snell), 0. 27 (Nagerkerke). Model $x^2(1) = 811.63$, p=<0.001.

Habitat type, method, distance from reserve and distance from reserve interacting with village are all equally important predictors of hunt success, each with a Relative Variable Importance (RVI) of 0.84 (table 3-5), while village alone (0.33) and distance from road (0.79) are less important. Hunters are less likely to successfully hunt in farmland, degraded forest or clearings compared to undisturbed forest habitats (table 3-6). Hunters in village 1 are more successful if they hunt closer to the reserve, while hunters in village 2 are more successful further from the reserve, perhaps because this implies they are in forest concession land. A hunt in village 2 is 1.26 times more likely to be successful than a hunt in village 1. This may be because hunters in village 2 spend longer per hunting trip than in village 1 (see table 3-4), either because they are travelling further afield or because they need to spend longer hunting to be successful.

Hunters using snares are twice as successful as those using guns, although gun hunters are more successful than hunters using dogs or a mixture of methods. While hunters in village 2 hunt more with guns than in village 1 (table 3-4), hundreds of snares can be left over several days, increasing the chance of making a catch over time.

In order to tease apart whether the environmental predictors in the top model (habitat type, distance from road and distance from reserve) observed in the GLM are due to animals being more easily found in particular locations, or because of ease of access for hunters, we need contextual information from interviews. The road in village 1 runs around the outside of the reserve, and small paths lead off from the main road allowing relatively easy access to the reserve:

> *"We take the road down to the Dja, or the path up to the reserve for big hunting….It's ok, there are animals…. Everyone knows that the forest here is good. It's a problem…. our forest is too easy to hunt, animals are just by the track. I check my snares nearby and walk back to the village in half a day. It's ok for me."*
> *(Male, village 1)*

While hunting close to the reserve and along roads to facilitate access in village 1 remains easy, in contrast village 2 doesn't appear to have the luxury of choice. They are located further from the reserve and report that their community forest is depleted, so they hunt more in the timber concessions. Either way, they must travel far from the village (and therefore the road) to hunt

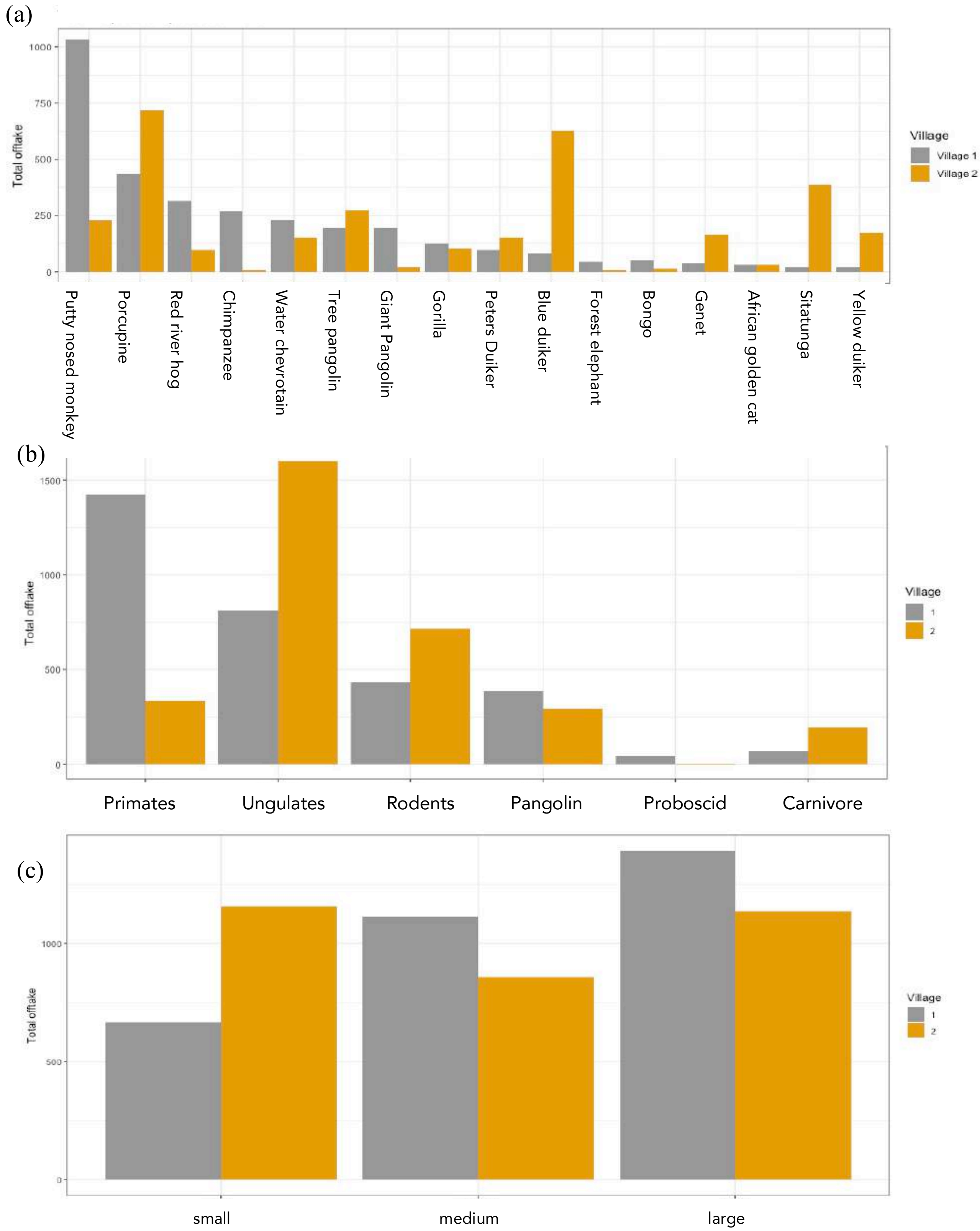
*"Do I even hunt in the community forest anymore? Now we suffer more to have the animals. Before the animals were closer to the village than now, they were just eating by the road. There are animals that are not even found here before going 30 km, for example giant pangolin. If you do not leave the community forest or if you are not in the UFA (timber concessions), you cannot meet it unless you are very lucky."*
*(Male, village 2)*

*"To go and catch the rare animals you now have to go deep, to cross where there are the timber companies, as on this side where there is [name of a timber company]. It is only after these societies towards [name of a town] that one can find the calm forest where you will find the elephants, the gorillas, the chimpanzees... And there, you cannot go alone. You have to go as a team. And as hunting is already forbidden, people do not venture so much anymore."*
*(Male village 2)*

### 3.3.2 Total catch composition and hunting intensity

The composition of offtake in village 1 includes primates and slow breeding and sensitive species such as chimpanzee (Pan troglodytes), gorilla and tree pangolins (figure 3-3b). This suggests that there has not been prolonged intensive hunting. In contrast, the composition of offtake from village 2 shows some indication of intensive hunting, based on the high proportion of rodents and faster breeding animals such as porcupine *(Atherurus africanus)* and blue duiker *(Philantomba monticola)* and the lack of rare slow breeding ones such as elephants, giant pangolins *(Smutsia gigantean)* and chimpanzees. Hunters in village 2 appear to be targeting medium to large ungulates such as yellow-backed duiker *(Cephalophus silvicultor)* and sitatunga *(Tragelaphus spekii)* where possible (figure 3-3a), although hunters in village 1 hunt more large mammals overall (figure 3-3c). The results are consistent with optimal foraging theory, in that they suggest that larger, more profitable, mammals are targeted first by hunters (Peres, 2000; Newing, 2001).

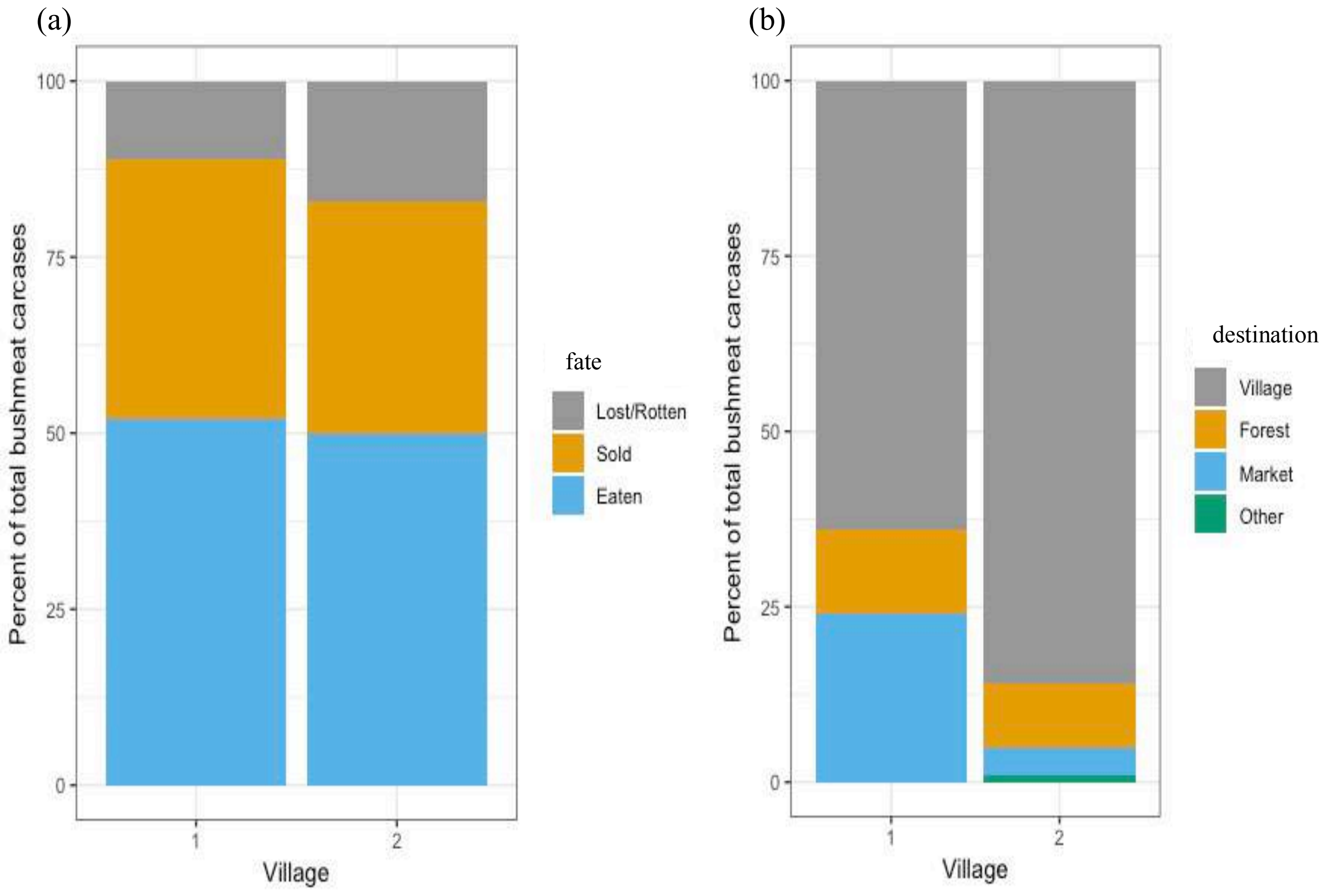
Figure 3-3: Comparison of total offtake and composition obtained from hunter diaries for (a) individual species (b) total offtake according to species order, and (c) offtake according to body mass category



### 3.3.3 The drivers of hunter decision making

Villages may be more or less connected to external demand for wild meat and the infrastructure which facilitates trade, which in turn may affect motivations for hunting. 52% of wild meat in village 1, and 50% in village 2 is eaten for subsistence. 37% in village 1 and 30% in village 2 is sold for income. 11% in village 1 and 14% in village 2 is lost or found rotten. More is lost or rotten in village 2 because hunters here go on fewer hunting trips than in village 1 and therefore are likely to be leaving their snares for longer (table 3-4). I found significant differences in the destination of the wild meat caught; 87% of wild meat was destined for the village, either to be sold or consumed, in village 2, and 62% in village 1. In contrast, a greater proportion of wild meat from village 1 was destined for market (24% in village 1, and 5% in village 2). This may be because it is easier for hunters in village 2 to sell their meat from the village directly, in particular to passers-by, compared to hunters in village 1, where buyers other than those in village are infrequent.

Figure 3-4: Stacked bar graph showing (a) the fate and (b) the destination of the total proportion of offtake with distance from village.




Multinomial mixed effect GLMs were run to investigate which variables are the most important predictors of whether a captured animal was found rotten, lost, eaten or sold. Body mass and hunting method are in the top models in both villages, indicating that these variables are important at both sites. Habitat is in the top model for village 1, while distance from village is in the top model for village 2. With 99% support for village 1, and 87% support for village 2, I chose to accept the top models, which both included a species characteristic (body mass), a hunter behaviour variable (method) and an environmental characteristic (either habitat or distance from village) (table 3-7).

In village 1, those using snares rather than guns were less likely to lose their catch than to eat it (table 3-8). No variables had a significant effect when comparing wild meat that is found rotten vs eaten. Further, farm habitat has a large SE because there are very few data points for animals found rotten on the farm, perhaps because people are at their farms daily, reducing the chance of animals being left to rot. Large mammals are 2.68 times more likely to be sold rather than eaten, a highly significant result. Animals snared rather than gun hunted are significantly less likely to be sold than eaten. Animals hunted on the farm rather than in degraded forest are also significantly less like to be sold rather than eaten.

In summary, these results suggest that people hunt on the farm for their own consumption, using snares, and in a way that has little wastage. By contrast, animals hunted in forest are generally large, sold, and potentially more likely to be lost because they can escape wounded when they are gun-hunted. This makes sense as these are two different styles and purposes of hunting.

Table 3-7: Results from the multinomial GLM with AIC < 2, indicating the characteristics associated with whether wild meat is lost, rotten, sold or eaten.

| Village | Rank | Model | Degrees freedom | Delta AIC | Weight |
|---|---|---|---|---|---|
| 1 | 1 | Body mass + Method +Habitat | 21 | 0 | 99% |
| | 2 | Body mass + Method +Distance village | 19 | 16.25 | <0.001% |
| 2 | 1 | Body mass + Method +Distance village | 13 | 0 | 87% |
| | 2 | Body mass + Method +Distance village + Habitat | 17 | 15.3 | 13% |

Table 3-8: Multinomial fixed effect GLM top model for village 1. Dependent multinomial variable is fate (lost, rotten, sold or eaten). Intercept: Biomass= small, method=gun, habitat=degraded

| Variable | Estimate (SE) | Lower | Odds ratio | Upper | P- value |
|---|---|---|---|---|---|
| Lost vs Eaten | | | | | |
| Intercept | -1.21 (0.50) | 0.11 | 0.29 | 0.79 | 0.02 |
| Body mass medium | -0.18 (0.38) | 0.40 | 0.84 | 1.75 | |
| Body mass large | 0.49 (0.35) | 0.83 | 1.63 | 3.25 | |
| Method snare | -1.43 (0.25) | 0.15 | 0.24 | 0.39 | <0.001 |
| Method dogs/mix | 0.12 (0.60) | 0.35 | 1.13 | 3.66 | |
| Habitat farm | 1.16 (0.82) | 0.06 | 0.31 | 1.56 | |
| Habitat forest | -0.26 (0.39) | 0.36 | 0.59 | 1.66 | |
| Rotten vs Eaten | | | | | |
| Intercept | -2.71 (0.75) | 0.01 | 0.065 | 0.29 | <0.001 |
| Body mass medium | 0.17 (0.59) | 0.38 | 1.19 | 3.80 | |
| Body mass large | 0.59 (0.56) | 0.59 | 1.79 | 5.44 | |
| Method snare | -0.05 (0.38) | 0.45 | 0.95 | 1.99 | |
| Method dogs/mix | 0.28 (1.09) | 0.16 | 1.32 | 11.23 | |
| Habitat farm | 17.21 (2633.53) | 0 | <0.001 | 1 | |
| Habitat forest | -0.52 (0.51) | 0.22 | 0.59 | 1.59 | |
| Sold vs Eaten | | | | | |
| Intercept | -0.46 (0.31) | 0.34 | 0.63 | 1.16 | |
| Body mass medium | -0.14 (0.22) | 0.57 | 0.87 | 1.33 | |
| Body mass large | 0.96 (0.20) | 1.76 | 2.68 | 3.87 | <0.001 |
| Method snare | -1.03 (0.13) | 0.27 | 0.36 | 0.47 | <0.001 |
| Method dogs/mix | -0.53 (0.46) | 0.24 | 0.59 | 1.46 | |
| Habitat farm | -2.95 (1.04) | 0.006 | 0.005 | 0.41 | 0.003 |
| Habitat forest | 0.26 (0.25) | 0.79 | 1.29 | 2.11 | |

In village 2, those using dogs are less likely to find their food rotten compared to those hunting with snares, while those hunting with guns are less likely to lose their catch than to eat it (table 3-9). Further, large mammals are significantly less likely to be found rotten compared to small mammals, which are more frequently hunted with snares. Medium and large mammals are respectively 22 and 13 times more likely to be sold than eaten compared to small mammals, again highly significant results. Finally, species hunted with a gun are 3.97 times more likely to be sold rather than eaten, compared to animals hunted with snares. With every km travelled, animals are 1.13 times more likely to be sold than eaten, supporting the hypothesis that hunters travel further from the village to hunt larger mammals to sell.

In summary, these results suggest that people hunt for their own consumption using snares, resulting in some wastage of small mammals. By contrast, medium and large mammals are commonly gun hunted and sold rather than consumed.

Table 3-9: Multinomial fixed effect GLM top model for village 2. Dependent multinomial variable is fate (rotten, sold or eaten). Intercepts: Body mass = small, method=snares)

| Variable | Estimate (SE) | Lower | Odds ratio | Upper | P-value |
|---|---|---|---|---|---|
| Rotten vs Eaten | | | | | |
| Intercept | -0.95 (0.54) | 0.13 | 0.38 | 1.11 | |
| Distance village | 0.05 (0.03) | 0.98 | 1.05 | 1.13 | |
| Body mass medium | -1.16 (0.38) | 0.15 | 0.31 | 0.66 | 0.002 |
| Body mass large | -3.36 (0.54) | 0.01 | 0.03 | 0.10 | <0.001 |
| Method gun | 1.03 (0.36) | 1.39 | 2.81 | 5.67 | |
| Method dogs/mix | -1.42 (0.43) | 0.10 | 0.24 | 0.56 | <0.001 |
| Sold vs Eaten | | | | | |
| Intercept | -5.01 (0.78) | 0.001 | 0.006 | 0.03 | <0.001 |
| Distance village | 0.12 (0.02) | 1.08 | 1.13 | 1.18 | <0.001 |
| Body mass medium | 3.09 (0.74) | 5.14 | 21.98 | 93.99 | <0.001 |
| Body mass large | 2.53 (0.77) | 2.76 | 12.58 | 57.31 | 0.001 |
| Method gun | 1.38 (0.28) | 2.29 | 3.97 | 6.88 | <0.001 |
| Method dogs/mix | -0.40 (0.29) | 0.38 | 0.66 | 1.16 | |

### 3.3.4 Temporal changes in offtake and hunting motivation

Species offtake in village 1 is quite consistent from August to December (figure 3-5a), with the majority of offtake coming from snares or gun hunting (figure 3-5b). Offtake drops in January when most people in the village are busy preparing their main cash crop, cacao, for market. January and February are also the dry months when people are busy with agricultural duties before the first rains arrive in March.

I hypothesized that there would be more gun hunting during crop planting and cultivation from March-June, as this is when financial expenditure is needed for materials and to buy food to supplement what is available. However, offtake remains low during those months, and increases in the rainy season in both villages, when animals are reported to be found more easily and closer to the villages.

There is a spike in offtake in village 1 in March, attributed to an increase in gun and dog hunting. This increase in offtake is partly in preparation for women's day (8th March). A huge celebration of this was held in village 1 during this study, hosting women's organizations from neighbouring villages. Months of planning and fundraising went into this event, with the launch of a new school that day also:

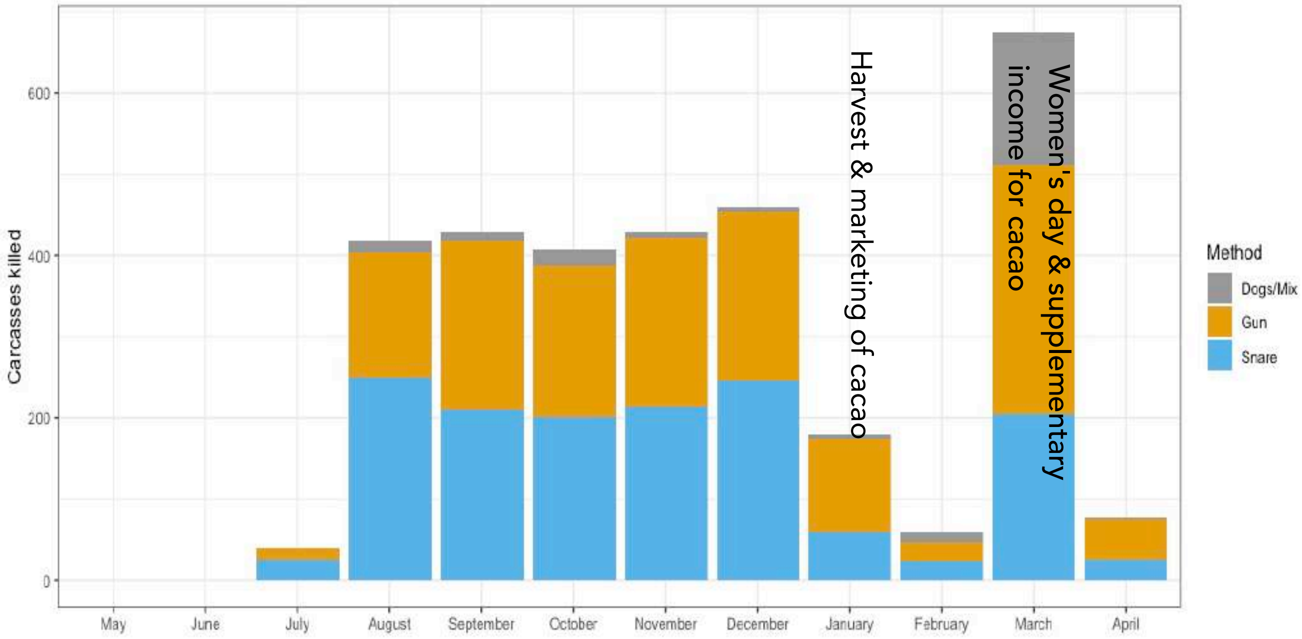
> *"It's (8th March) a big event. All the women work together for months. We have weekly meetings here on Saturday morning to practice the songs and organize the parades. We need to prepare, collect the money, give the orders for food so they (the men) can go and get enough, and make plenty of palm wine. Everyone is busy! The men don't like to give us money for this day, but we are making them. They will eat all the meat and drink the wine so they must pay" (Female, village 1)*

Secondly, hunting appears to provide a financial backstop in times of financial stress. For example, villagers reported that they did not get a good price for their cacao in December this year. This does not happen every year, although apparently it is happening increasingly frequently:

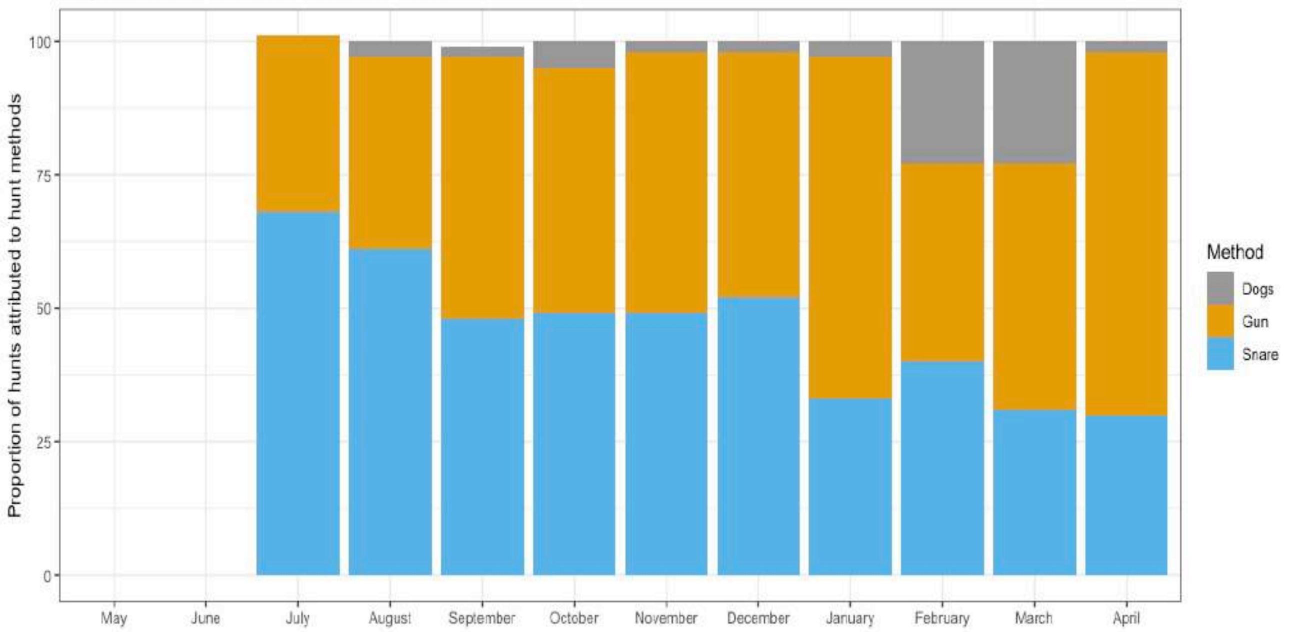
> *"In December, I got the money for my (cacao) crop. I worked so hard for it, but it was bad. They just tell us each year that the price is dropping. I cannot argue, they come here to buy it, so I don't have a choice. I need that money to pay for my children's school, but already it is gone. I have had to find money in other ways. All the animals will be gone this year, they will need to hide."*
> *(Male, village 1)*

Figure 3-5: Village 1 (a) monthly offtake and (b) proportion of offtake attributed to each hunting method. No data was collected in village 1 in May-June.

(a)




(b)



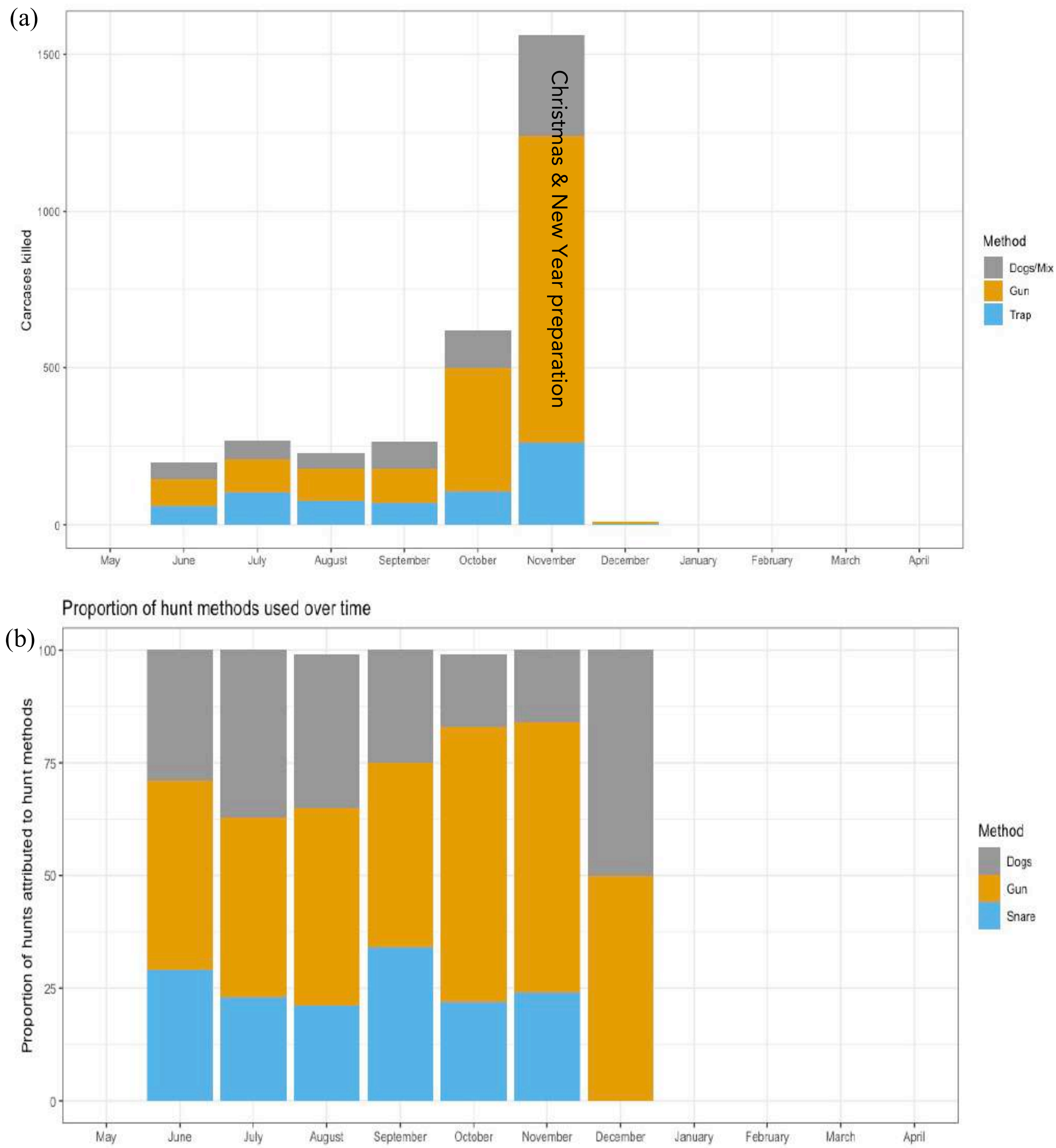
While village 2 farms cacao, they do not farm it to the same extent as village 1 does. Hunting offtake remains relatively low from June to September, the summer months, when hunters harvest their subsistence crops (figure 3-6a). Qualitative data also suggest that hunters in village 2 have the option of alternative work in the timber concessions, and this work is preferred to hunting when it is available. However, village 1 do not have the same alternatives available to them. When overall offtake increases in October and November, in part due to preparations for Christmas and New Year, gun hunting also increases:

*"The New Year's party, we will kill many many animals."*
*(Female village 2)*

*"So many of the boys here don't even have fields anymore, they just hunt and work in the timber concession....people come here to recruit extra help in the summer holidays. It's sad, because they will forget how to farm. In fact, they forget how to hunt too. There's a whole forest out there full of animals, but the boys prefer to work in the concessions instead." (Village chief, village 2)*

The increase in gun hunting in both villages may serve to signal the hunters' motivations; gun hunting generally means larger mammals which can then be sold for additional income. This supports the qualitative evidence that spikes in hunting offtake in both villages are in response to economic stressors (i.e. cacao prices) or cultural events (i.e. Christmas).

Figure 3-6: Village 2 (a) monthly offtake and (b) proportion of offtake attributed to each hunt method. No data was collected in village 2 from January-May.

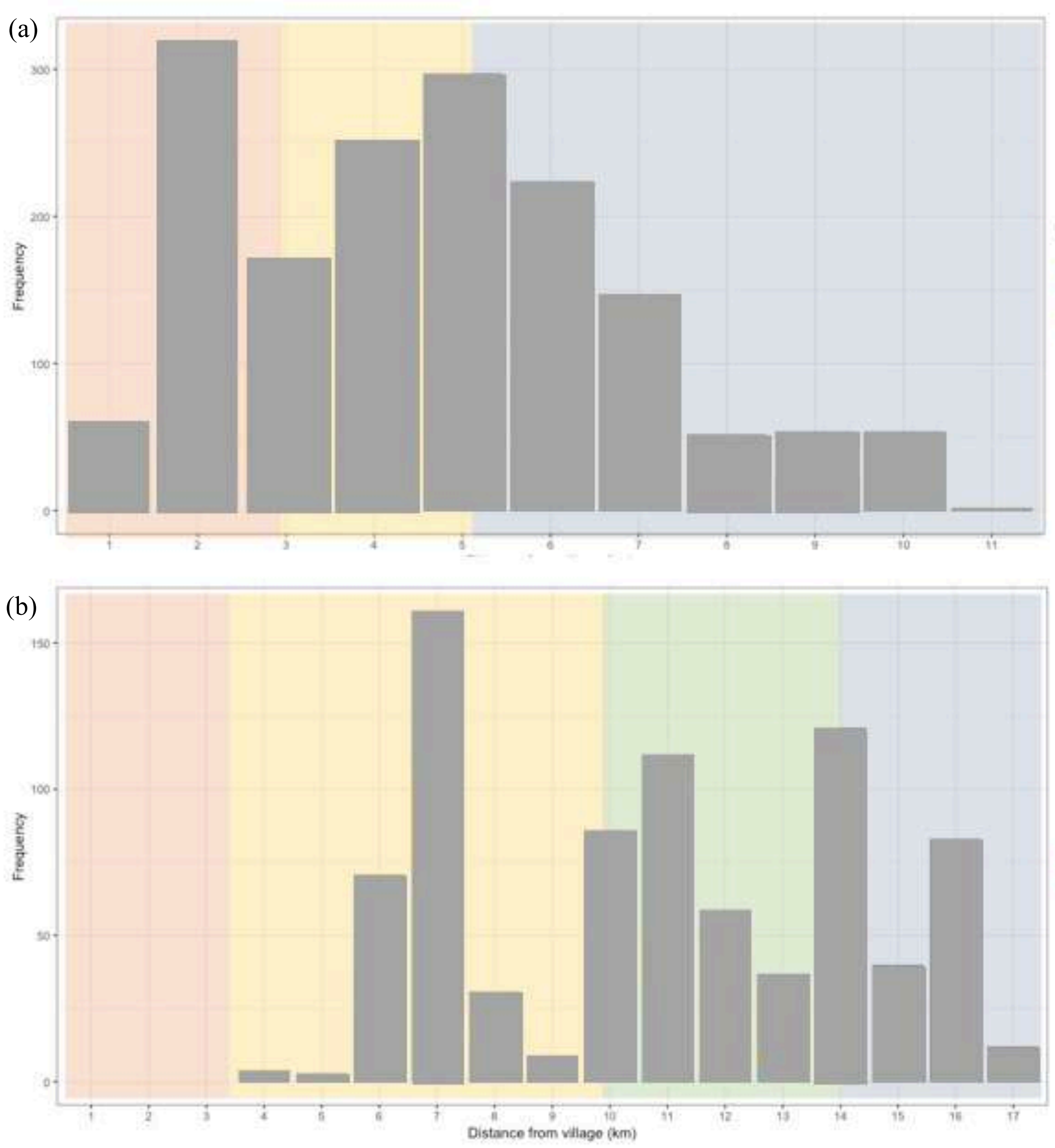


### 3.3.5 The drivers of spatial patterns in wild meat hunting systems

To identify the drivers of spatial patterns in wild meat hunting systems, I first compared the spatial variation in distance travelled, offtake and gear choice. I asked whether there was any indication that hunters travelled further from the village in response to differences in profitability between animals found in different locations, or if any spatial patterns were simply due to depletion of wildlife near the villages. Central place foraging theory dictates that hunters hunt close to the village, then move away from the village as the prey base depletes. This assumes that all prey are equally valuable. Yet, if the potential benefit varies (e.g. if more saleable or profitable species are found in particular habitats) then the cost hunters are willing to incur (which is reflected in the distance they're prepared to go) may also vary in order to maximize their profits.

In village 1, the highest frequency of trips occurs within the agricultural zone and the community forest, less than 5km from the village. The frequency of trips then drops by 50% beyond 5km from village (figure 3-7a). This indicates either that most hunters are only hunting close to the village, while some hunters are choosing to go further, or that all hunters are hunting close to the village, but occasionally travelling further. Appendix B-7 confirms the latter; all hunters travel similar distances - on average 2.5-5km from the village - but all occasionally travel over 7.5km.

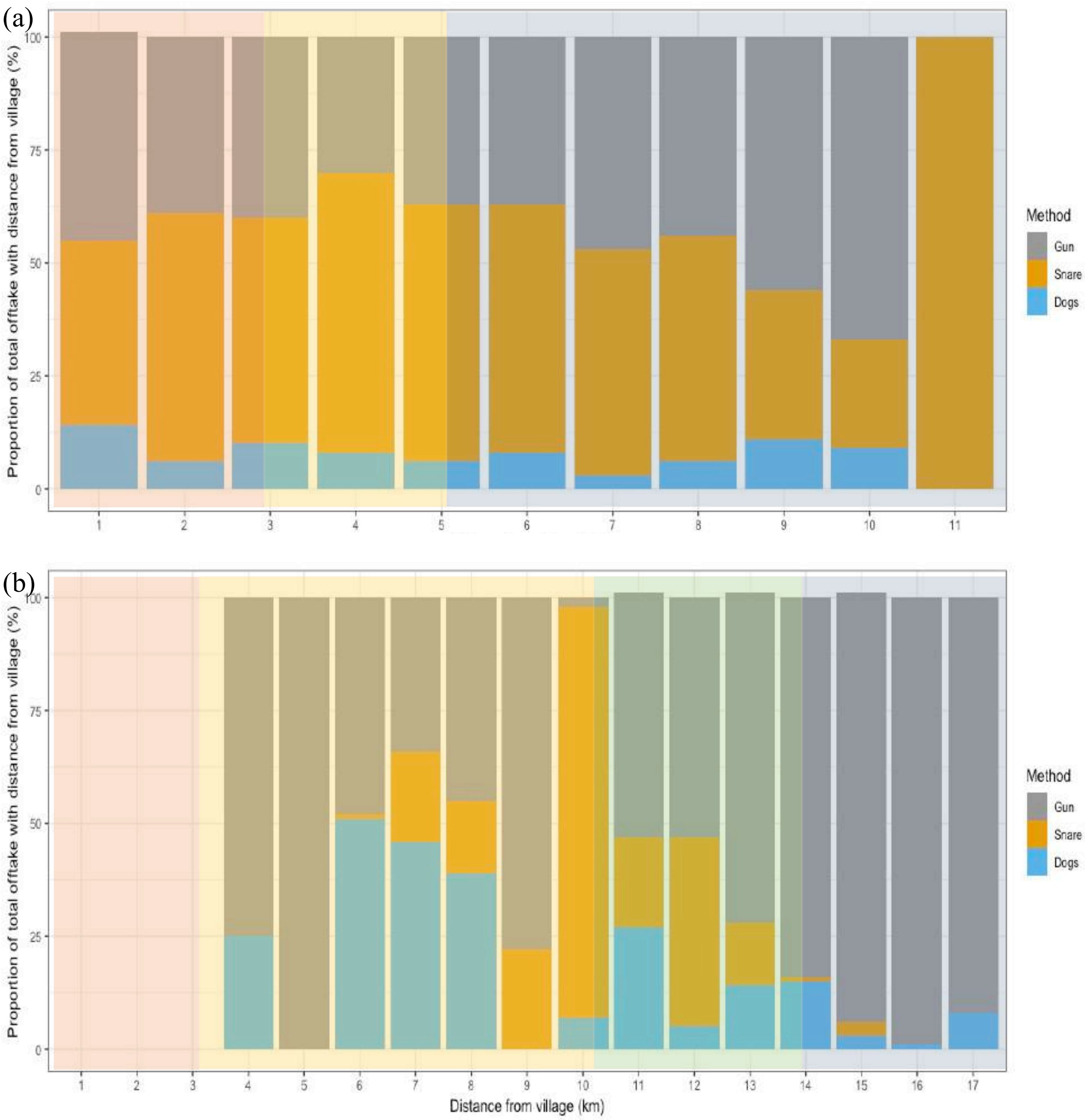
Figure 3-7: Frequency graph of hunting trips against distance walked from (a) village 1 and (b) village 2. Background colours indicate the land uses that may influence hunting trip location, gear use, and frequency. Red=agricultural zone, yellow=community forest, green=timber concession, blue=reserve



In village 2, barely any hunting trips occur within the agricultural zone (<3km from village). The majority of hunting events occur at 7km from the village, and a peak in hunting occurs again at 10km from the village, signalling the boundary of the community forest and timber concession. It appears that wildlife depletion in village 1 isn't yet a major problem due to the availability of animals so close to the village. In contrast, local depletion appears to be an issue in village 2 (figure 3-7b).

If we now take the gear choice used with distance from village into consideration, there is some indication that hunters may be responding to external incentives to hunt in both villages, and enter areas with potential risk to them due to illegality in order to do so. Snares are the most commonly used method for hunting within 8km of village 1 (figure 3-8a). In village 1, the proportion of hunts with snares declines from 8-10km, while the proportion of hunts with guns increases. In village 2, there is clear preference for hunting with guns from 13km. Although gun hunting in village 2 is more preferred than in village 1, the preferred hunt method prior to 13km is more variable (figure 3-8b).

Figure 3-8: Proportion of hunting events attributed to different hunting methods with distance from (a) village 1 and (b) village 2.

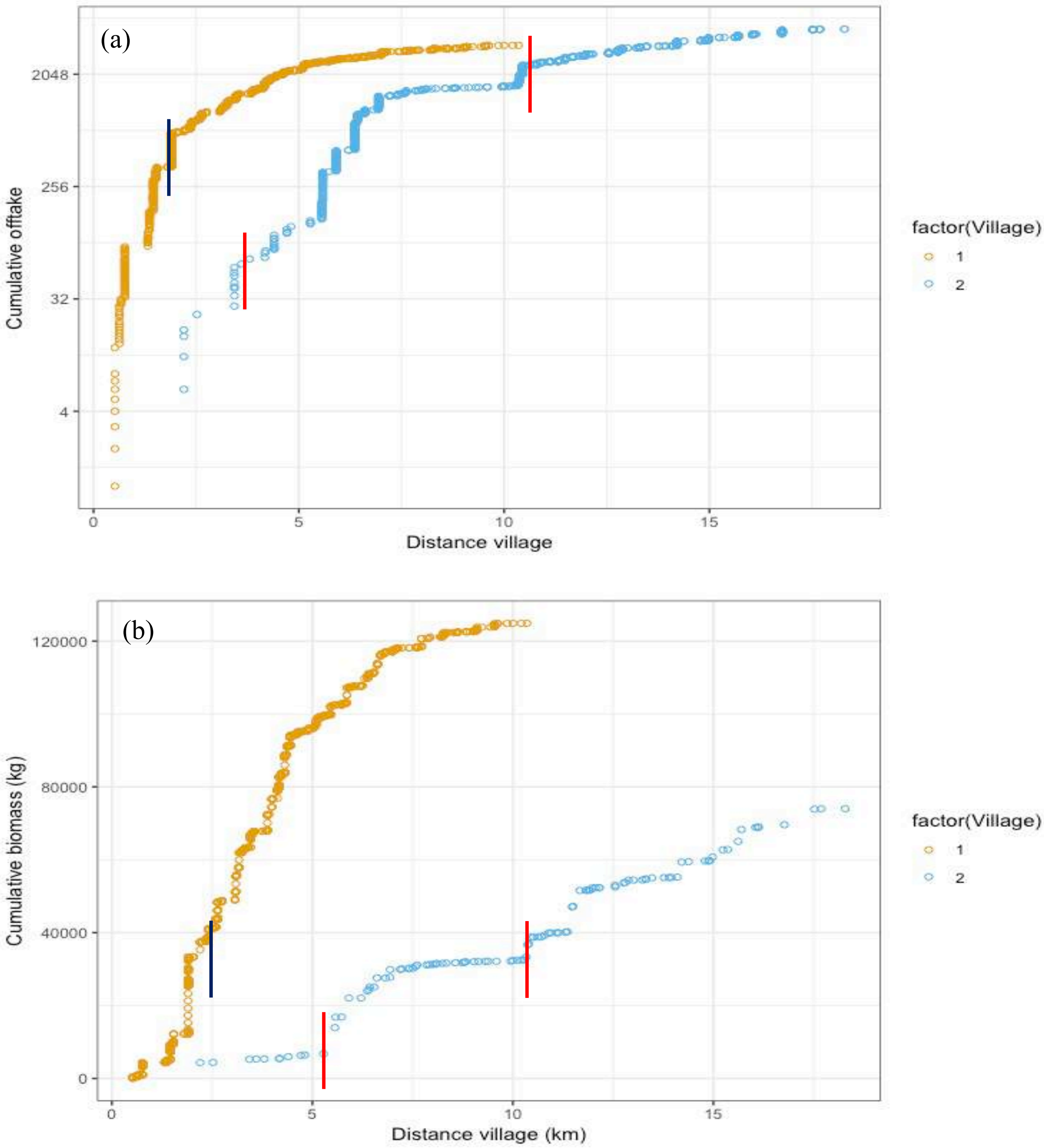


I found it interesting to observe how much more frequently hunters hunted accompanied by dogs in village 2. Qualitative data revealed that this village has a strong culture of dog hunting that is now dying out in the face of new hunt methods, but may still result in higher rates of hunting with dogs compared to village 1:

> *"At the time there was much hunting with dogs. Now it's more the traps, and the weapon. We do not have the good dogs anymore. By the 90's people were hunting regularly with dogs, but now the dogs are gone. Especially the purebred dogs. Hunting dogs. Besides the cable even today is no longer the right cable. Before people were doing 1 2 3 years with a single cable. The cables are no longer solid, we do not know why." (man, village 2)*

The frequency of hunting events with guns in village 2 highlights the intensity of hunting effort further from the village and provides support to the hypothesis that village 2 is hunting in response to connectedness with market-based economies. These graphs also reflect the results of the GLMs (tables 3-8 & 3-9), that hunters are travelling further afield to hunt with guns, which are then sold. Understanding the total offtake and body mass with distance from village also provides insight into the rate of offtake with distance from village (figure 3-9) and further clarification about the motivations for hunters in both systems.

Figure 3-9: Cumulative frequency curves for number of individuals hunted and (b) total biomass with distance from village. Black lines indicate changes in land use from agricultural zone to forest in village 1, while red lines indicate changes from agricultural zone, into timber concession and finally into forest village 2.



The rate of accumulation of individuals hunted begins to slow in village 1 at the end of the agricultural zone (figure 3-9a) while the cumulative offtake in village 2 increases after the agricultural zone finishes and again within the timber concessions. There are also distinct changes in the cumulative biomass rate with changes in land use, again with the end of the agricultural zone and start of the timber concessions (figure 3-9b). The cumulative number of individuals hunted in both villages is comparable (figure 3-9a), but hunters in village 2 travel double the distance to obtain half the body mass (figure 3-9b). The rate of individual offtake plateaus after 5km from village 1, indicating that most hunters remain within 5km of the village, reflecting the results in figure 3-7. The biomass obtained within 5 km indicates that there are plenty of medium and large mammals to support the hunters' needs close to the village. In village 2, very few animals and little biomass is obtained within 5 km, supporting the result that hunters rarely hunt here because of local depletion of medium and large mammals. The number of individuals begins to plateau after 10km from village 2, however the cumulative body mass continues to increase.

The results indicate that village 2 suffers from wildlife depletion, while there is little evidence of this in village 1; their community forest is able to provide them with wild meat. However, hunters in both villages report observed changes in species populations, and in both cases, link the arrival of timber concessions, infrastructure and external pressures with the start of species decline. The extent of decline is different in both villages. In village 1, wildlife is still perceived to be abundant, although some recognize recent declines in abundance due to external pressures:

*"Animals are still abundant, and hunting isn't really a problem. Since the arrival of the logging road, hunting is starting to become a little harder, but it's not a huge issue…..we can hunt close to the village with both snares and guns and still catch plenty*

*of food, especially monkeys which are all over the place. Standing in the village you can head the moneys and chimpanzees calling, and see them jumping in the trees" (Male, village 1)*

The new road in village 1 facilitated access to the community forest by people outside of the community, as well as increased access to markets:

*"Since the logging road arrived last year, animals are scared of the noise, and people from the concession hunt in our forests. Also, people from other places come to hunt in our forests now because they heard about how much meat we have. We have tried to stop them and put in a barrier at our village, but the eco-guards do nothing to stop people. No one is helping us." (Male, village 1)*

In village 2, wildlife depletion has been observed over a long period of time. The lack of wild meat available in the community forest is cause for concern, both for food security and conservation:

*"The animals here have become rare. When you have your snares in the forest, it must be a lot. You must check them a lot and still come home with nothing some days. The community forest is not enough for us anymore…It has been like this a long time." (Male village 2)*

*"So, before they may walk 3-4h, now it's 2 days of walking. Yes, there are some who make up to 2 days of walking to get to where they have to camp." (Female village 2)*

Can you tell me when it became harder to hunt? When did you start to go far to find the animals?;

*"Yes! At the arrival of the forest companies. As soon as the logging companies started the exploitation, it was then that the animals also moved away from the village. They entered in the years 80-85. So, when these companies entered the forests, the animals started to flee. They have moved away from the village." (Male, village 2)*

## 3.4 Discussion

This study provides an assessment of the effects of hunter and village level variables on the spatial and temporal trends in wild meat hunting at two contrasting study sites within the same ecosystem. I found that overall the results are in-line with the hypotheses presented in table 3-2, and that SES frameworks are a useful tool to piece together evidence to help us understand hunting sustainability, given how challenging monitoring sustainability can be in forest environments. Hunters in both villages use hunting for food and livelihoods.

However, the sites present different sustainability challenges due to differences in the way that hunter, village and external pressures interact. Using an SES framework allowed me to investigate the interacting influences of variables at different scales, displayed in figure 3-10. Future studies on wild meat hunting will also benefit from using such an approach, which allows for a more holistic view of sustainability than biological measures alone. This enables more nuanced recommendations to be made on how the system in question could be brought back in line with sustainability if required.

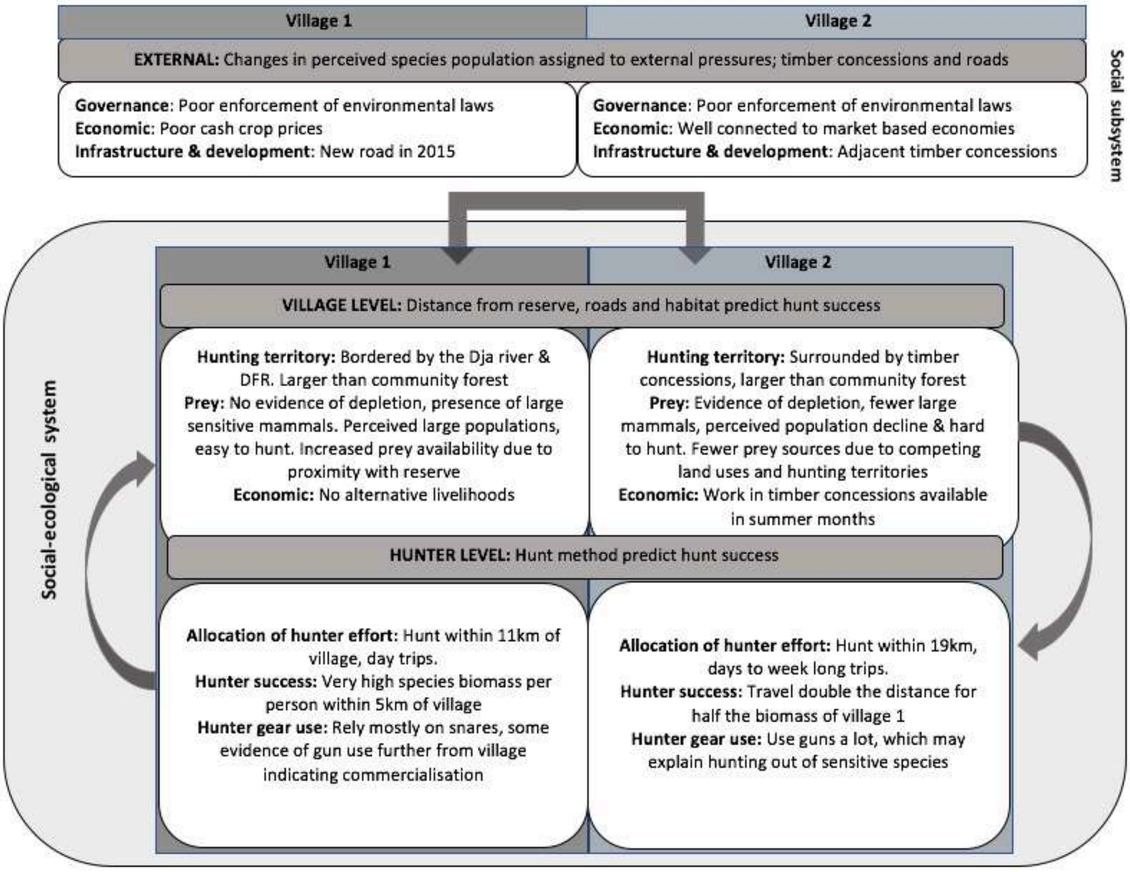



Figure 3-10: SES model completed with key hunter, village and external pressures for each village in this study

### 3.4.1 Spatial variation in hunting offtake is determined by interactions between village and hunter level variables

Village location in relation to the reserve and neighbouring land uses affects prey availability (Gavin, 2007), which in turn influences species offtake and the size of the hunting territory. Studies in West and Central Africa found that reserves act as a refuge for wildlife, encouraging illegal hunting within the reserve, but also providing spill over effects as prey emigrates out of the reserve and into surrounding areas (McNamara et al. 2016; Mavah et al. 2018) resulting in higher offtake for vulnerable species such as primates and large ungulates (Fa et al. 2006). The results of this study support this; village 1, in close proximity to the reserve had a high ratio of primates in their catch composition. In contrast, village 2 is adjacent to other villages, timber concessions and hunting territories, with a more limited source area and reduced prey availability as a result. In another hunting study in south-east Cameroon, Bobo et al. (2015) found that ungulates and primates were the most heavily hunted, but that primates were primarily represented in hunter catch where hunting pressure was highest. While the other strands of evidence point to low hunting intensity in village 1, it may be that the reserve is acting as a good prey reserve, masking the effects of overhunting. Further work is needed to determine the effect either way.

Accessibility is a key determinant of spatial variation in hunting intensity. As hunting is limited by travel costs, hunting intensity is often greatest along roadsides and closest to villages where access is facilitated (Siren et al. 2013, 2015). While hunters in village 1 reduced effort by hunting close to roads where possible, hunters in village 2 did not, contrary to our hypothesis. The recently completed road past village 1 is rarely used, minimizing the noise and human disturbance that may scare away animals (Blom et al. 2005) while allowing ease of access to hunters. In contrast, the road in village 2 is well-established, well-used and lined with villages, which may act as a deterrent for animals or have resulted in historic overhunting, and therefore reduce the benefit of easy access for hunters now. Several studies have found that exploited population densities are lower closer to settlements and major transport routes (Wilkie 1989; Wilkie & Curran 1991; Siren et al 2004), which I also found to be the case in village 2.

Hunters were most successful in forest habitats compared to farmland or degraded forest, supporting the suggestion that hunters may be targeting forests because they harbour larger mammal species which hunters can both eat and sell (van Vliet & Nasi 2008). As habitat type affects game availability

(Martínez-Cruz, et al. 2017) in both villages hunters are preferentially hunting in forests to target larger mammals and maximize their returns (Peres, 2000; Newing, 2001).

### 3.4.2 Evidence of species depletion and intensive hunting

The catch composition data indicate differences in hunting intensity between the two villages. Rodents, more resilient to overhunting, dominate the catch in village 2 (Wilkie & Carpenter 1999; Rowcliffe et al. 2003) while a greater proportion of primates, sensitive to overhunting, suggests little intensive hunting in village 1 (Duncan et al. 2002). Intensive hunting near village 2 may have resulted in the hunting out of wildlife, where travel time is regularly cited as a major cost compared to village 1; travelling further results in fewer total hunting trips to compensate (table 3-4). The longer average hunting trip length in village 2 may therefore represent a response to species decline, although more work is needed to confirm this.

### 3.4.3 Hunting gear choice affects hunter success

Hunting method was the most significant hunter-level predictor of hunt success. In one study, Gill et al. (2012) found that dependence of rural people on intensive snaring in Equatorial Guinea as a source of income decreases when employment opportunities are available to young men. It may be that a lack of alternatives to snaring could be fuelling the dependence on snaring in village 1. However, our results show that those using snares are often more successful than gun hunters, suggesting that snare hunting is the best option for hunters here regardless of the alternatives avaialble. Similar conclusions have been drawn in other studies (Holmern et al. 2015; Dounias et al. 2016).

### 3.4.4 Wild meat use for food and livelihoods

Wild meat is an important source of protein and livelihoods in both villages (as also found by Zurlini et al. 2006; Coad et al 2009; Alexander et al. 2011). I did not find a distinction between hunters who travelled further afield and those who hunted closer to the villages as has been found in other studies (Okouyi, 2006; van Vliet & Nasi, 2008; Alexander et al. 2011), suggesting that all hunters hunt both for food and for sale. Further work would be beneficial to disentangle hunter typologies present in these villages and track changes over time as external pressures develop.

### 3.4.5 Hunting as a financial backstop

Hunting intensity varies with agricultural activities and cultural events, acting as an important financial backstop in times of financial stress, as also found by Coad et al. (2010). Seasonal Increases in offtake are largely driven by increased gun hunting in both villages, although hunting using dogs or mixed-methods also increases. De Merode (2003) and Solly (2004) observed similar trends in the Congo and in Cameroon in times of financial trouble; guns are used to target larger species for sale. While village 1 hunters do not have alternative income options, some hunters in village 2 work seasonally in timber concessions, which may explain the higher prevalence of guns in village 2. In other studies in the Congo basin, hunters have suggested that they would rather have other work than hunting (Endamana et al. 2010; Gill et al. 2012). Qualitative data from village 2 suggest that hunters prefer work in the timber concessions over hunting, although more evidence on hunter livelihood preferences is needed to confirm this. However, the fact that villagers hunted more in periods of cash shortage, especially when income from cocoa runs out as in village 1 (see Schulte-Herbrüggen et al. 2013)) suggests that there is a lack of alternative livelihoods.

### 3.4.6 Evidence of commercialisation

The catch composition data provided evidence to support the commercialization of hunting in village 2, in the form of increased gun use with distance from village to hunt larger animals which are then sold. I also identified the start of similar commercialization patterns in village 1. Alexander et al. (2014) found that proximity to urban centres goes some way to explaining inter-village differences in species depletion and trends in hunting. In this study, village 2 is well-connected to urban markets and infrastructure (Lindsey et al. 2011; Nielsen & Meilby, 2015; Greengrass, 2016; Wilkie et al. 2016); growing urban prosperity (Fa et al. 2000) and migration (Davies, 2002; Wright 2019), allowing hunters to sell their wild meat directly from the village to passers-by. In another study, areas closer to markets experienced a loss of more-vulnerable species and consequently, species profiles were more stable in these areas than in areas farther away that were still experiencing a decline in these species (Allebone-Webb et al. 2011). The early indications of commercial hunting since the arrival of the logging road in village 1, and the long-established sale of wild meat in village 2 support the findings of other studies that the emergence of market-based economies drives wild meat hunting (Brashares et al. 2011).

### 3.4.7 The typologies of hunting systems

Village 1 is a 'frontier' village (Milner-Gulland, 2006) and of high conservation importance (table 3-10).

Table 3-10: Typology of two contrasting hunting systems in this study, following the typologies set out by Milner-Gulland (2006).

| Village | Conservation priority | Main threat | Primary source of wild meat | Species composition | Hunting system type |
|---|---|---|---|---|---|
| 1 | High | Opening up of primary forest from new logging road. | Forest | Rapid extirpation of vulnerable and large mammals due to hunters from village and external hunters | Frontier |
| 2 | Low/ medium | Long term exploitation, external pressures | Forest and degraded forest | Some vulnerable species, but reduced species population over wide area | Declining-source sink/ becoming mature wild meat market |

The new road provides improved access to market towns and an influx of hunters who come to hunt illegally in this rich and recently exposed forest. This study was able to capture some direct qualitative information about the influx of external hunters into the community forest, but more research on this would be a useful addition to understand the extent of hunting pressure in this newly connected hunting system.

The combined rate of hunting by those within and outside the local community may result in rapid extirpation of wildlife populations, starting with the larger and more vulnerable species, as reflected by the offtake composition. There are some early indications of transition to a commercialized system in village 1;

hunters are occasionally travelling further to hunt large mammals with guns, in order to sell them for income. In their study of the sustainability of duiker hunting by Baka in south-east Cameroon, Yasuoka (2006) found that external pressures could upset the balance of natural resource extraction where a village was previously succeeding, which this study also supports in part. This is not to point blame on the local community, but rather to highlight how systems are highly vulnerable to rapid change.

Action is required to prevent a commercialized system from developing, as once a small society is plugged into external markets, effective conservation becomes less likely (Smith and Wishnie, 2000). Villages should be steered away from hunting large, less resilient mammals and encouraged to hunt smaller more resilient mammals, such as porcupine and blue duiker, which are abundant within the community forest. Village 1 is neglected in terms of agricultural extension and livelihood generation, evidenced by their vulnerability to cash crop prices. This is consistent with findings from the FAO (2013) that volatile cash crop prices result in increased dependence on wild meat; wild meat hunters require more support in agriculture and meeting their minimum needs if wild meat hunting of rare and sensitive species is to be reduced (Nielsen, 2017). Such support may help build greater resilience to financial shocks and stressors.

Village 2 could be a declining source sink system, whereby hunting is occurring over an increasingly wide area, although with spatial variation due to environmental and social factors. Alternatively, it may also be a mature system, with depleted but stable species populations; monitoring over time is required to draw solid conclusions. The catch composition and trends in offtake suggest intensive hunting and species depletion, however. The quotes from hunters also support the view that the system is not in equilibrium. The predominance of gun hunting and an increased proportion of species sold as the distance from village at which they are hunted increases point to a well-established and commercialized system. Trends in hunting here are of concern both for conservation and for food security. Despite the total offtake matching that of village 1, hunters here expend double the effort to hunt half the biomass. Hunters passionately shared their concerns about how hunting is no longer easy and that they are suffering to make a living from wild meat. Hunting practices in village 2 need to be brought back in line with sustainability for the benefit of conservation and for food security and livelihoods. Species population monitoring is required to establish how much hunting can be sustained (perhaps forming the basis for future village-level sustainable use quotas), while the sale of wild meat from the village should be reduced. However, this can only happen as part of an integrated and participatory livelihoods plan.

### 3.4.8 Broader implications for conservation in the Dja faunal reserve

Smith et al. (2003) reported a lack of legal enforcement and capacity in the DFR. During this research, I found the state of enforcement and capacity in the reserve has not changed in the past 15 years. Despite recent activities to improve ranger capacity (UNESCO, 2018), capacity to prevent external threats and illegal hunting remains weak.

Gun hunting has serious implications for arboreal primates (Kumpel et al. 2009; Jost Robinson et al. 2011; Bobo et al. 2015), which may help to explain the lack of primates in village 2's catch composition (Ávila et al. 2017). Although poorly enforced, gun hunting requires a license in Cameroon. However, the government will not license any locally produced or modified shotguns (Akumsi, 2004), which represent a large proportion of the guns present in village 2, an issue that was also reported by Willcox and Nambu, (2007). Although stronger enforcement of this law could have a positive effect on wildlife populations and allow the more sensitive species to recover, it may also have negative consequences for local livelihoods and food security in village 2, which must be strongly considered.

Wild meat has long been of importance to communities around the DFR. Muchaal and Ngandjui (1999) found that 98% of the wild meat consumed in villages around the DFR came from the reserve itself. Yet, considering the highly valuable biodiversity within the reserve, there is comparatively little recent research on the importance of wild meat from both within and around the reserve for local communities. 20 years later, this research shows how two hunting villages hunt in response to individual, village-level and system-wide pressures. The development of commercialized systems in relatively remote areas reflects how motivations to hunt are changing, and pressures on wild meat species are shifting. The emerging external pressures observed are not unique. As roads, mines and timber concessions lead to the development of wild meat markets around the reserve, so too will the motivation to hunt for commercial purposes increase, threatening the most sensitive and rare species first.

# Chapter 4

# Comparing interview methods with camera trap data to inform occupancy models of hunted mammals in forest habitats

(Walking to the next household interview to gather species presence/absence data)

## 4.1 Introduction

Forests in Cameroon are important both for conservation and for local people who depend on forest resources and biodiversity for their livelihoods (Carson et al. 2018). Species population declines, and in certain places, the rapid extirpation of populations due to overhunting (Milner-Gulland & Bennett, 2003), highlight the need for sustainable management of resources, and for rapid yet efficient monitoring methods that are both robust and practical over large spatial and temporal scales. Yet, despite a growing number of monitoring programmes in tropical forests, there is still incomplete knowledge on the efficiencies of different field techniques in assessing mammal populations (Munari et al. 2011).

Monitoring in forest environments is commonly informed by camera traps or sign detection surveys such as line-transects (Karanth et al. 2011; Rich et al. 2017; Beaudrot et al. 2019). Camera trapping remains a key method especially for monitoring shy or secretive species (Rowcliffe & Carbone, 2008). Although the cost of camera trapping has reduced, it is still relatively expensive and time consuming (Silveria et al. 2003), a problem exacerbated by limited funding in many projects. In response, studies are increasingly incorporating local ecological knowledge (LEK) into monitoring programmes. LEK (Huntington 2000) is a cost-effective, and potentially robust method for data collection, utilising the often-detailed ecological knowledge accumulated during the daily lives of people who live or work within or close to the area of interest (Mohd-Azlan et al. 2013;  Turvey et al. 2013). Local people can frequent large areas that are relatively inaccessible (Zeller et al. 2011; Service et al. 2014), increasing the likelihood of encountering species and providing historical information about a population. This can be especially useful for wide-ranging and elusive species (Turvey, 2013, 2015). Importantly, putting local knowledge at the centre of conservation efforts can also increase the likelihood that subsequent conservation activities are locally supported and sustainable (Danielsen et al. 2008).

Several methods exist to gather data from local people (see Newing et al. 2011). Interview-based surveys, acknowledged as a rapid and low-cost method (Mohd-Azlan et al. 2013;  Turvey et al. 2013; Service et al. 2014), are widely used to gather data on species trends and distributions (e.g. Jones et al. 2008; Parry & Peres, 2015; Turvey et al. 2015) in both terrestrial and aquatic habitats (Turvey et al. 2010; Garrote & Pérez de Ayala, 2015). Many studies focus on species with an economic or cultural value (e.g. (Danielsen et al. 2005; Zurlini et al. 2006; Leeney & Poncelet, 2013) and large-bodied vertebrates have also generally received greater attention (e.g. Belant et al. 2013; Brittain et al. 2018). However, in their study of the Hispaniolan solenodon (Solenodon paradoxus) and Hispaniolan hutia (Plagiodontia aedium) in the Dominican Republic and Haiti, Turvey et al. (2014) demonstrate that LEK can be used to assess the status of and threats to small, cryptic species. Diaries are an alternative method that are gaining popularity, often used in studies to gather self-reported data on hunting patterns (Rist et al. 2008; Allebone-Webb et al. 2011; van Vliet et al. 2015; Brook et al. 2018) or wild meat consumption (Kumpel et al. 2009; Broegaard et al. 2017).

As with all monitoring methods, surveys with local people present their own set of challenges and potential biases. For example, Mohd-Azlan et al. (2013) identified several limitations to the robustness of interview data from local people, such as the misidentification of species, different understandings or concepts of what a species is, language barriers between interviewers and local people and the challenge of separating accurate from inaccurate interview data. Further, Garrote & Perez de Ayala (2015) found participants to overestimate compared to camera trap estimates, while Golden et al. (2013) found that participants are subject to recall bias when asked to provide information over a period of time. Yet, several studies that compared the estimates derived from daily and recall interviews found dietary recall surveys to be an effective method for measuring the food consumption choices of households (Day et al. 2001; Baer et al. 2005; Rentsch & Damon, 2013). Additionally, Jones et al. (2008) sought to investigate the impact of recall bias and the quality of data obtained from seasonal rapid interviews compared to daily diaries with resource harvesters in Madagascar, finding that rapid seasonal interviews provide reliable information on quantities, effort, and the spatial pattern of harvesting.

Some studies have assumed that diaries are not subject to bias (e.g. Golden et al. 2013) yet substantial literature exists on the inaccuracy of daily diaries particularly within health studies (e.g. Wiseman et al. 2005), development studies (Beegle et al. 2012) and ethnobiology (Shanley, 1999; Menton et al. 2010). In their comment on Golden et al. (2013), Newing & St John (2013) state that diaries should not be viewed as free of bias, and rather that diary keepers may suffer from reporting fatigue; recall error connected to gaps or delays in completing the data; incomplete knowledge; and of course, intentional errors related to social and cultural norms or the illegality of consuming certain species.

Comparison studies are a useful tool to compare the precision, accuracy and cost of monitoring methods. I searched Google Scholar for studies that were published since 2000 which explicitly sought to compare estimates derived from

LEK to conventional species monitoring methods. Studies that simply incorporated LEK but did not seek to compare estimates with another method were excluded from table 4-1 below, as were studies that apply LEK for
Of the comparison studies listed in table 4-1, two focus on detection efficiency and only two studies look at occurrence data. The majority compare transects to camera traps and no studies are conducted in forest habitats. Only two studies compare daily logs to interviews, and none compare diaries to conventional methods. Six studies compare interview data to either camera or transects, four of which include an assessment of cost. This is important as the involvement of communities should build upon local skills and must be locally affordable (Gaidet et al. 2006).

Table 4-1: Summary of previous relevant studies comparing conventional monitoring methods with LEK based monitoring. Only studies that seek to compare estimates derived from local people with conventional monitoring methods were included in the search.

| Study | Survey approach assessed | Species | Camera traps | Transects | Aerial | Interview | Daily log | Community monitoring | Focus groups | Cost/ feasibility |
|---|---|---|---|---|---|---|---|---|---|---|
| Tomasini & Theilade (2019) | Abundance | Medicinal plants | | X | | X | | | | No |
| Oliveira Lima et al. (2016) | Species richness | Insects | | X | | | | X | | No |
| Caruso et al. (2016) | Occupancy estimates | Carnivores | X | | | X | | | | No |
| Hausser et al. (2016) | Species richness | Mammals | X | X | | X | | | | Yes |
| Garrote & Perez de Ayala (2015) | Occurrence | Iberian Lynx | X | X | | X | | | | Yes |
| Cook et al. (2014) | Expert judgement | Vegetation condition in PAs | | X | | X | | | | Yes |
| Danielsen et al. (2014) | Abundance | Mammals birds & plants | | X | | | | | X | Yes |
| Golden et al. (2013) | Consumption recall | Wild meat | | | | X | X | | | Yes |
| Zukowski et al. (2011) | Catch & scientific data (sex ratio, size) | Crayfish | | X | | X | | | | No |
| Oldekop et al (2011) | Species richness | Ferns | | X | | | | X | | Yes |
| Anadon et al. (2009) | Abundance | Tortoise | | X | | X | | | | Yes |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Jones et al. (2008) | Detection efficiency | Crayfish & firewood | | X | X | | |
| Baer et al. (2006) | Food consumption | Wild meat | | X | X | | No |
| Gaidet-Drapier et al. (2003) | Sampling and detection efficiency | Mammals | X | | | X | Yes |

While there is now a growing wealth of examples of the incorporation of LEK into monitoring studies, there remains in general a lack of research that adequately compares results obtained from local people to data from more conventional methods. In particular, no studies exist that compare daily diary data to both seasonal interviews and a conventional monitoring method, such as camera traps. If locally-based approaches are shown to be a robust monitoring option for mammal species, such a comparison may provide a cost-effective alternative to conventional methods commonly used to monitor mammals in forest habitats.

Ideally, the objective of monitoring is to obtain precise population estimates with a low bias (Singh & Milner-Gulland, 2011). Detection effectiveness is key in determining the success of animal population assessments (Munari et al. 2011). However, all wildlife population monitoring methods suffer from a degree of imperfect detectability. With imperfect detection, a species may be falsely detected when they do not exist (Royle & Link, 2006; Miller et al. 2011) or (more commonly) not detected when it is in fact present (Royle & Nichols, 2003; Royle, 2006). Not accounting for imperfect detection can result in inaccurate estimation of extinction probability, biased inference on habitat preferences (Kery 2004; 2006), or biased population trajectories (Shefferson et al. 2001).

Occupancy modelling is one monitoring approach which accounts for imperfect detection, thus reducing estimate bias. Occupancy is the probability that a species occupies, or uses, a sample unit during a specified period of time during which the occupancy state is assumed to be static (Bailey et al. 2004). Occupancy analysis is widely used for large-scale monitoring programmes as it is relatively inexpensive and easy to implement compared to estimates of absolute abundance (Pollock et al. 2002; Royle & Nichols, 2003; Hedges, 2012). Data collection is relatively simple; presence-absence data is collected through repeat visits to a sample site in search of evidence of the target species. Occupancy modelling (Mackenzie et al. 2002), allows both detectability and occupancy to be estimated in a single-model framework (Mackenzie et al. 2006). Relationships between probabilities of occupancy and detection and either ecological or, in the case of data obtained from local respondents, socio-demographic factors can be investigated by incorporating them as covariates for analysis.

Several studies have combined observational methods with occupancy analysis to gather data on rare or wide-ranging species at large-scales (Martinez-Marti 2011; Puri et al. 2015; Brittain et al. 2018) and over long timescales (Pillay et al. 2011). The approach has been used to identify jaguar corridors in central America (Zeller et al. 2011; Petracca et al. 2013) and identify significant declines in distribution for large carnivores, the Asian elephant and endemic ungulates and primates in India (Pillay et al. 2011).

Despite the growing popularity of the approach, there is a surprising lack of studies that assess local knowledge combined with occupancy analysis in forest environments or for monitoring species commonly hunted for wild meat. The motivation for this study therefore stems from this lack of focus in the literature, and the need for comparison studies that focus on uncertainty. In this study, I evaluate the efficacy and robustness of occupancy models informed by observational data from local people, for a range of rare and commonly hunted mammal species in two contrasting community forests in Cameroon. Using a mixed-method approach, I triangulate estimates of detectability and occupancy from three different methods to assess their precision and accuracy at different scales and provide guidance on future use of these methods for monitoring both threatened and hunted mammals in forest habitats. The research objectives were as follows:

1. Compare estimates of detectability and occupancy for multiple species and assess their levels of precision and accuracy between observational methods (i.e. the term used when referring to both daily diaries and seasonal interviews) and camera traps
2. Identify which environmental and observer variables have the biggest influence on both detectability and occupancy
3. Make recommendations for future use of LEK informed monitoring methods when monitoring mammals in challenging forest habitats

Research questions and hypotheses related to each objectives are outlined in table 4-2.

Table 4-2: Research questions and hypotheses

| Question | Hypothesis |
|---|---|
| 1.Do estimates of species occupancy compare across camera trap and LEK methods at the village level? | Observational data will be comparable to the camera trap estimates at the village level |
| 2. How does the precision and accuracy of occupancy estimates vary between methods and species? | LEK-based estimates will be more accurate and precise than camera-based methods for rare species, such as chimpanzee and gorilla<br><br>Highly abundant species that are well detected by all methods will have comparable site-level occupancy estimates across methods.<br><br>A-priori predictions of how each method will perform when detecting each species are shown in table 4 |
| 3.How do environmental and observer variables account for variation in occupancy and detection? | Experience and sociodemographic variables such as age, gender and frequency of trips will be more important predictors of detectability for interviews than diaries, which are completed by a more homogenous group of participants.<br><br>See table 5 for a-priori predictions of the effect of environmental and observer variables on species occupancy and detection |
| 4.What methods are most cost and time efficient? | Observational methods will be more cost-effective than camera traps (Garrote & Peres de Ayala, 2015; Hausser et al. 2016) |

## 4.2 Methods

In this study, I compare two locally-based approaches to camera trapping. Interviews are collected seasonally, while diaries are collected daily, allowing for recall bias to be assessed (Jones et al. 2008). Further, the types of observer are different for each method; the diaries are completed by a smaller, more homogenous but knowledgeable and experienced group of hunters who may target certain species and areas, while in contrast the interviews are conducted with the whole village, who may go everywhere and also have different levels of ability to detect species. Finally, camera traps were selected as the comparison method because although they have their own related biases, it removes the observer-based biases that are associated with both interviews and diaries. Camera traps have also been frequently compared in previous literature (table 4-1) and are often used for monitoring in forest habitat, therefore present a practical method to compare against the observational methods. Each data collection method presents different strengths and potential biases; efforts were made to account and control for these biases (table 4-3).

Table 4-3: Common bias associated with each monitoring method compared in this study and the steps taken to overcome them.

| Method | Method strengths | Potential bias | Actions taken |
|---|---|---|---|
| Camera traps | Cameras are set in the village community forests, allowing direct comparison between estimates of occupancy from camera traps and observational data.<br><br>Low human interference maximizes chance of detecting cryptic species. | The detection rate of endangered or cryptic species may be too low, resulting in greater uncertainty in the estimate of occupancy. In such cases, perhaps estimates from interviews results in higher detection rate and more robust estimates<br><br>Not all species will be well captured by camera traps. This bias depends on placement, such as height, relation to paths, habitat type. | Camera placed to maximize detections and a good camera survey effort of 1440-1880 days was used to maximize chance of detections.<br><br>Cameras were placed in a grid, close to species signs (e.g. tracks, paths etc.) to maximize detections. Table 4-4 shows the species that I expect to be detected sufficiently by the camera traps, and where I feel detection will either be too low, or absent. This will be taken into consideration when drawing comparisons and making recommendations. |
| Daily diaries | A group of hunters who make regular trips to the forest and are able to reliably recognize and record the location and species detected. This image-based data provides daily detection data, that is in theory less subject to recall bias as there is minimal delay between experience and record. | Relatively small sample size, therefore the precision of the estimates and the range of areas covered by this sample may not be as large as the interview data. The hunters selected could potentially be more or less active or more prone to hiding the truth than the general hunting population. | Hunters that hunt together were not selected, to prevent duplication of records. Only hunters that agreed to take part, and who had shown openness in their answers in previous work were finally selected. |

| | | | |
|---|---|---|---|
| | | Potential errors in data collection | Thorough training was conducted with each participant at their households, to ensure that participants were comfortable with the daily diaries and how to locate and record sightings and hunting events from the participatory map (which all were already familiar with through previous surveys). See section 4.2.2 for more details |
| Interview data | Rapid, cost effective and includes local people in conservation. Participants cover areas that traditional methods cannot under budgetary and logistical restrictions. Information is based on long-term association not just a snapshot in time. | In some cases, the population may not be honest, when the species has an economic or cultural value (Grant & Berket 2007) or if they don't know or haven't observed the species. | No names were recorded on the data sheets, ID codes ensured the respondents identity remained anonymous. Ensuring anonymity to respondents helped reduce reporter bias.<br><br>Responder knowledge was checked prior to participation, by showing a series of photos and asking respondents to identify them. Only data from respondents who provided the correct name of the species, either in French or in the local language, were included in the analysis. |

### 4.2.1 Study area

Two wild meat hunting villages adjacent to the Dja Faunal Reserve (DFR) were selected to draw inferences about the applicability and robustness of the different methods in two systems. See section 2.1.10 for further details.

### 4.2.2 Data collection

Using a mixed-method approach, I obtained estimates of occupancy and detectability for 15 mammal species hunted for wild meat in two contrasting systems. The species included in the study are listed in table 4-4.

*Camera traps*

Bushnell aggressor cameras were used with low glow infrared flash set to medium. The manufacturer-reported trigger speed was 0.14s, with a 1s delay between triggers. Each trigger took 3 photos. The cameras had high sensitivity and were placed 30-45cm off the ground, angled horizontally. No attractants were used. The cameras were in the field from September-November 2017 in village 1, and April-June 2018 in village 2. While different seasons, the weather is relatively similar during these seasons (hot and rainy), so differences in species distributions wouldn't be expected. However, there may be differences in hunting pressure as a result of different livelihood needs at those times of year, which must be considered. A systematic grid layout was used consisting of 30 cameras with a 1km gap between each (Rovero et al. 2013; Burton et al. 2015). Cameras were placed to capture a gradient of distance from each village out towards the reserve and adjacent land uses, but remaining within the limits of the community forest where local people had the right to hunt and access, in order to reduce reporter bias from the observational methods. The placement of some cameras had to be adjusted slightly as they would have fallen in the middle of the village. However, the new placement still respected the 1km spacing.

*Semi-structured interviews*

Semi-structured interviews comprising of simple questions about the presence/absence of the target mammal species were designed and administered by the research team. Semi-structured interviews were conducted once each seasonal (four times in total) according to seasons as identified by key informants in both villages, which ensured that the seasons used were familiar and locally relevant, and respondents could more comfortably recall over that given time. Sightings were recorded as direct observations. Tracks, nests, vocalization and dung were recorded as indirect observations following Munari et al. (2011).

As people in villages regularly travel away for work or study, I was not able to employ a stratified random sample approach, as a representative sample of each demographic variable selected for use in this study was not always available. Further, despite our efforts to interview the same respondents each season, this was not always possible. Therefore, a targeted non-probability sampling strategy was employed, aiming to interview all willing adults within the villages, at least one adult per household. See appendix C-1 for interview protocol.

*Hunter diaries*

Diaries completed my local people have been used to collect data on catch and effort in hunting camps (Rist et al. 2010) to recall dietary intake (Golden et al. 2013); collect data on offtake rates (Kumpel et al. 2009); and meat consumption

(Jenkins et al. 2011). Ten hunters in each village were trained to keep image-based daily diaries from May 2017 to April 2018, providing information on the species they detected and hunted, where the species was detected, the habitat type and date. Species level predictions of how each survey method will perform are found in table 4-4. The daily diaries gather the same information as the seasonal semi-structured interview, but only required a tick or cross in columns with corresponding images. Hunter follows with key informants were conducted opportunistically, following Rist et al. (2008) to ensure that the species sighting data was being entered correctly. See appendix C-2 for the hunter diary data sheet. In contrast to chapter 3, where only species recorded as hunted were included in analysis, all species detections were included in this analysis whether hunted or not.

*Mapping of detections for diaries and interviews*

To locate the species detections from the interview and diary data, a simple map of the village and surrounding community forest was made, combining GPS points of each household, key landmarks and GIS data on rivers, major roads, community forests and the reserve. This basic map of the village was a) used during the initial interviews to ensure that we reach each household and b) used as a base map for participatory mapping.

Subsequently, participatory mapping exercises were held to identify fields, paths, rivers and key landmarks, which help participants to accurately recall, using features familiar to them, where species were detected. Participatory mapping may require a representative sample of informants to capture all the different ways that local people use or access their environment (Corbett, 2009). As such 20 key informants took part in 2 focus group exercises in each village (n=4 focus groups). Features added to the base map during these exercises were then ground-truthed with GPS, create a map that is both representative of areas of local importance and spatially accurate.

To facilitate comparison of estimates of occupancy between methods, it was important that the spatial sampling units used for the interviews and daily diaries, reflected the 1km$^2$ grid used for the camera trapping. This was achieved by placing a 1km$^2$ grid over the research teams copy of the completed map, so that the research team could allocate grid references to detections recorded in the semi-structured interviews and diaries. These maps allowed participants to recall and map species detections using locally relevant landmarks and features, which I could then convert into detection histories for each 1km$^2$ site (van Vliet & Nasi, 2008; Newing, 2011). Following Martinez (2011), individual interviewees are treated as effective repeat spatial and temporal surveys for occupancy analysis. Only detections that were made within the same 30km$^2$ area as the camera traps were included in analysis to facilitate comparison of occupancy estimates between methods.

When mapping from the hunter diary data, a key contact was employed within each village to provide assistance when needed and help to collate the datasheets. In both cases, he was a trusted community member and not associated with the village leader's family. They were involved in the creation of the community forests as a community representative, therefore familiar with reading maps. Participants knew the key contact and were able to approach them for assistance if needed. The accuracy of data collection was checked with opportunistic hunter follows at the start of the data collection (n=9) and the key contact was in weekly contact with me via text and calls where possible, to feedback on progress and notify me of any issues.

Table 4-4: A-priori predictions of how each method will perform when detecting each species. Number of animal icons represents the predicted occupancy estimate range for each method, where one=0-0.33, two=0.34-0.66 and three= 0.67-1. No species icon=predicted zero occupancy. Colours indicate the hypothesised robustness of the occupancy estimate, where green= accurate & precise (robust), yellow= precise but inaccurate, blue=accurate but imprecise, red= inaccurate and imprecise.

| Species | Hunt class | Cryptic | Vocal/ groups | Rare | Hunted | Camera | Interview | Diary | Details |
|---|---|---|---|---|---|---|---|---|---|
| **Servaline genet *(Genetta servalina)*** | - | | | | √ | | | | High enough detection for robust camera and interview estimates, although estimates may be higher from interviews because they are reported to be a nuisance. |
| **Tree pangolin *(Phataginus tricuspis)*** | A | √ | | √ | √ | | | | Too few detections for a precise camera trap estimate. Actively hunted therefore detection may be higher from diaries, although again the detection rate may not be sufficient for precise estimates. |
| **Blue duiker (*Philantomba monticola)*** | - | √ | | | √ | | | | High detection rate will result in precise and accurate camera trap estimates of occupancy. Estimates from interviews may not be very precise due to recall, but the estimates of occupancy will be comparable between methods |
| **Yellow-backed duiker *(Cephalophus silvicultor)*** | B | √ | | | √ | | | | Lower occupancy and detection than blue duiker in community forest, but an important species for wild meat and easily identifiable. I expect them to be at high enough densities for camera traps to produce accurate estimates, although with some uncertainty. Estimates from observational data will be more robust as they are actively hunted |
| **Gorilla *(Gorilla gorilla)*** | A | | | √ | √ | | | | Culturally and economically important species that is well recognized and at times hunted. Occupancy may be too low in community forests for occupancy analysis to be successful using camera traps. However, interviews will result in more detections due to its well-recognized call, nests and value. I expect people to be able to remember where they have seen them in the interviews. |
| **Chimpanzee *(Pan troglodytes)*** | A | | √ | √ | √ | | | | Camera traps may not be best placed to provide robust estimates of chimpanzees as they are both terrestrial and arboreal. Easily recognizable and noisy makes them easier to detect via interviews and diaries. I expect people to be able to remember where they have seen them in observational methods |
| **Brush tailed porcupine *(Atherurus africanus)*** | - | | | | √ | | | | Abundant and commonly hunted in both villages. High abundance may make remembering where species were seen harder because they are seen frequently. |
| **Bongo *(Tragelaphus eurycerus)*** | A | √ | | √ | | | | | Very rare and I expect there to be few to no detections within the community forests. |
| **Giant pangolin *(Smutsia gigantean)*** | A | | | √ | √ | | | | Although very rare, giant pangolin has been reported within the community forest of village 1. However, occupancy may be too low for any detections from the camera traps. As they are well-recognized and actively hunted I expect higher estimated occupancy and density from the interviews. |
| **Forest elephant *(Loxodonta cyclotis)*** | A | | | √ | √ | | | | Rare and low occupancy expected in both community forests. I expect only a few detections within either community forests from interviews as signs are easily recognizable and none from the camera traps due to the relative proximity of the cameras to the village. |
| **Sitatunga *(Tragelaphus spekii)*** | B | √ | | | √ | | | | Mid-low occupancy expected in community forests, I expect detection will be too low for camera traps to be informative. Cryptic and quite shy, but actively hunted and with recognizable signs, so interview and diary estimates will be higher than those from camera traps. |
| **Red river hog *(Potamochoerus porcus)*** | B | | √ | | √ | | | | Mid-high occupancy and detection in community forests with recognizable signs. Actively hunted and occur in groups, I expect the observational methods and camera traps to detect them well. |
| **Putty-nosed monkey *(Cercopithecus nictitans)*** | - | | √ | | √ | | | | Putty-nosed monkey are arboreal; therefore, camera traps are not best placed to result in robust occupancy estimates. Putty-nosed are both abundant and actively hunted, therefore I expect that estimates of occupancy will be high from both observational methods. However, because they are abundant, estimates from seasonal interviews may not be so precise. |

### 4.2.3 Ethics

For details on the ethical procedures followed, see section 2.1.17

### 4.2.4 Sampling and environmental covariates

I included four sociodemographic covariates in analysis of the observational methods that I hypothesised could influence the ability of participants to detect the species. I also included five environmental covariates that I hypothesised could help to explain variation in $\boldsymbol{\psi}$ (occupancy) or p (detection). These included the distance (km) of each detection from the limit of the reserve, and from the village, which serves as a surrogate for human disturbance (see table 4-5). I used the Euclidean distance tool in QGIS 3.0.2 to extract distances for these covariates (QGIS Development Team, 2018). I used Pearson correlation coefficients to test for correlation between environmental covariates. None were removed because the test showed that none were highly correlated (e.g. >0.7). Covariates were standardised before modelling to aid comparisons and model convergence (Reilly et al. 2017). See appendix C-3 for the correlation test results.

Table 4-5: The environmental and observer covariates included in analysis and justification for their inclusion based on the literature.

| Variable | ψ /p | Measure | Justification |
|---|---|---|---|
| Distance roads | Ψ | km | Increased human activity and potentially hunting pressure could result in road avoidance behaviour and decreased abundance, that might be reflected by reduced occupancy. However, positive and neutral associations have been found and more tolerant species might not be affected. (Peres & Lake, 2003; Blom et al. 2005; Laurance et al. 2006; Laurance et al. 2008; Burton et al. 2012; Vanthomme et al. 2013; Ziegler et al. 2016). |
| Distance village | Ψ | km | Rare or shy species may prefer to occupy sites further from the village, while more abundant species may be found all over the community forest. |
| Habitat | Ψ | Semi-deciduous<br>Riparian<br>Farm | For a lot of rainforest specialist species tree cover is an important habitat requirement, but for those species that require clearings or are regular crop pests the cropland might provide favourable habitat. (Arlet & Molleman 2007; Kingdon, 2015; Koerner et al. 2017; Wearn et al. 2017). |
| Distance reserve | Ψ | km | Habitat closer to the reserve where there is less human activity might be valuable, especially for the less tolerant species or those that are overexploited in the village. Little to no trend is expected for more resilient species, such as blue duiker (Muchaal & Ngandjui 1999; Tagg et al. 2015; Bruce et al. 2017; Koerner et al. 2017; Ehlers Smith et al. 2018). |
| Distance river | Ψ | km | Being close to a water source is important for many species and was included as an indicator of habitat quality (MacKenzie & Bailey 2004; Bruce et al. (2017). |

| Observational methods only | ψ /P | Measure | Justification |
|---|---|---|---|
| Age | P | 18-25<br>26-40<br>41-65<br>65+ | Younger respondents may be more active in the forest and go to a wider range of sites for different livelihood activities, resulting in higher likelihood of detection. However, older respondents may also be well experienced at detecting certain species (Walker et al. 2002; Fonkwo et al. 2017). |
| Gender | P | Male<br>Female | Men and women may detect different species more or less, depending on where they go in the forest, the activities they are taking part in and also their level of knowledge of different species and their signs. |
| Frequency of trips | P | Daily<br>Weekly<br><weekly | People who go on more regular trips may be more knowledgeable about forest biodiversity and better able to detect species. Increased frequency of visits also increases the likelihood of detection (Burgman et al. 2016). |
| Time spent in forest per trip | p | <5 days<br>11-15 days<br>>15 days | Participants recalled the average amount of time spent per trip to the forest over a season, and hunters recorded the actual amount of time each trip had taken when completing the hunter diaries. Increased time spent in the forest and increases the likelihood of detecting animals. |

### 4.2.5 Data analysis

Species detection histories were created by arranging the data into presence/absence (1/0) of a species during repeat visits to a site. For camera traps, the sampling occasion was set at 5 days for the camera traps. This was chosen as a compromise between model stability and ensuring an adequate number of repeat visits to each site (Burton et al. 2015). For observational data, respondents were treated as effective repeat surveys. Table 4-6 displays the survey effort for each village.

Table 4-6: The survey effort undertaken for each method in villages 1 and 2 over 2 seasons.

| | Village 1 | | | Village 2 | | |
|---|---|---|---|---|---|---|
| Method | Spatial scale ($km^2$) | Temporal scale (occasion) | Observers/ traps | Spatial scale (km) | Temporal scale (occasion) | Observers/ traps |
| Camera traps | 26 | 2.5 months (15 5-day occasions per site) | 26 | 22 | 2.5 months (15 5-day occasions per site) | 24 |
| Diaries | 13 | 2.5 months (76 1-day occasions per site) | 10 | 12 | 2.5 months (76 1-day occasions per site) | 10 |
| Semi-structured interviews | 26 | 2.5 months (4 observers per site) | 141 | 24 | 2.5 months (4 observers per site) | 109 |

I first calculated the naïve occupancy for each species, which is the number of sites where the species was found divided by the total number of sites, not accounting for variation in detection probability (Linkie et al. 2013). The MacKenzie and Bailey (2004) goodness of fit test was conducted on each global model to produce a c.hat value, indicative of over dispersion (non-independence). If the c.hat value is substantially >1 there is more variation in the data than expected, while a value of close to 1 shows good model fit (MacKenzie and Bailey, 2004). Where the c.hat value was >1, models were compared using the second-order quasi-Akaike Information Criterion (QAICc), a form of AIC for over dispersed data that accounts for small sample sizes (Burnham & Anderson, 2002; Mackenzie & Bailey, 2004; Clark et al. 2009). Single species, single-season occupancy models, originally designed by MacKenzie et al. (2002), were performed using the package "unmarked" in R version 3.4.2 (Fiske & Chandler, 2011; R Core Team, 2017). These models have four key assumptions that should be met (MacKenzie et al. 2006) (table 4-7).

Table 4-7: The key assumptions of single species, single-season occupancy models and how they were accounted for in this study

| Assumption | How the assumption accounted for in this study |
|---|---|
| Sites are closed to changes in occupancy | The interpretation of occupancy estimates has been changed from sites occupied to sites used for wide ranging species |
| Occupancy probability is constant across sites, or variation is accounted for | Variables accounting for occupancy are included in the models |
| Detection probability is constant across sites and occasions, or variation is accounted for | Variables accounting for detection are included in the models |
| Detection of a species at a site is independent of detecting the species at other sites | Model fit was assessed, which can detect non-independence. When non-independence was found, standard errors were inflated to reduce precision in the model estimates accordingly.<br><br>1km$^2$ grid large enough for sample sites to be within different home ranges for many of the species but not all, such as chimpanzee or elephant.<br>Interpretation of occupancy changed from proportion of area occupied, to area used. |

My hypotheses drove which predictor variables to include in the global models (Burnham & Anderson 2010). Minimal adequate models were selected from the global model with the "dredge" function (package MuMIn) (Barton 2012), which searches all predictor combinations and selects models by comparing values of Akaike's information criterion (AICc) (Pinheiro & Bates 2002; Barton, 2012). Models that didn't converge or produced estimates of $p<0.15$ and $\boldsymbol{\psi}=1$ were excluded, because in those cases there it is difficult for the model to distinguish between genuine absence and when the species was present but was not detected (MacKenzie et al. 2002).

Having selected the best and most parsimonious model, I used the "predict" function used to obtain predicted estimates of occupancy across the 30km$^2$ area which then allowed for comparison of estimates at the site-level. I only compared species that had produced robust occupancy estimates from all three survey methods, so as to produce a full methods comparison for these species. Of the 13-total species included in the study, 7 were used for full comparison in village 1, and 2 species in village 2. Pearson correlation tests were used to compare the predicted occupancy estimates from all three survey methods and ascertain whether there were statistically significant similarities or differences in the estimates provided.

## 4.3 Results

For the local knowledge interviews, a total of 141 local people were interviewed in village 1 and each site (1km$^2$) was visited a mean of 106 times (range=42-139, median= 135). Ten hunters completed the hunter diaries and each site was visited a mean of 3.93 times (range=2-10, median=3). The memory cards from three of the 30 cameras malfunctioned and one camera malfunctioned in the field in village 1. Four cameras were located in habitat that got flooded, although they continued to work and were therefore still included in the study. Over a total of 1,730 trap days, 16,050 photos were taken.

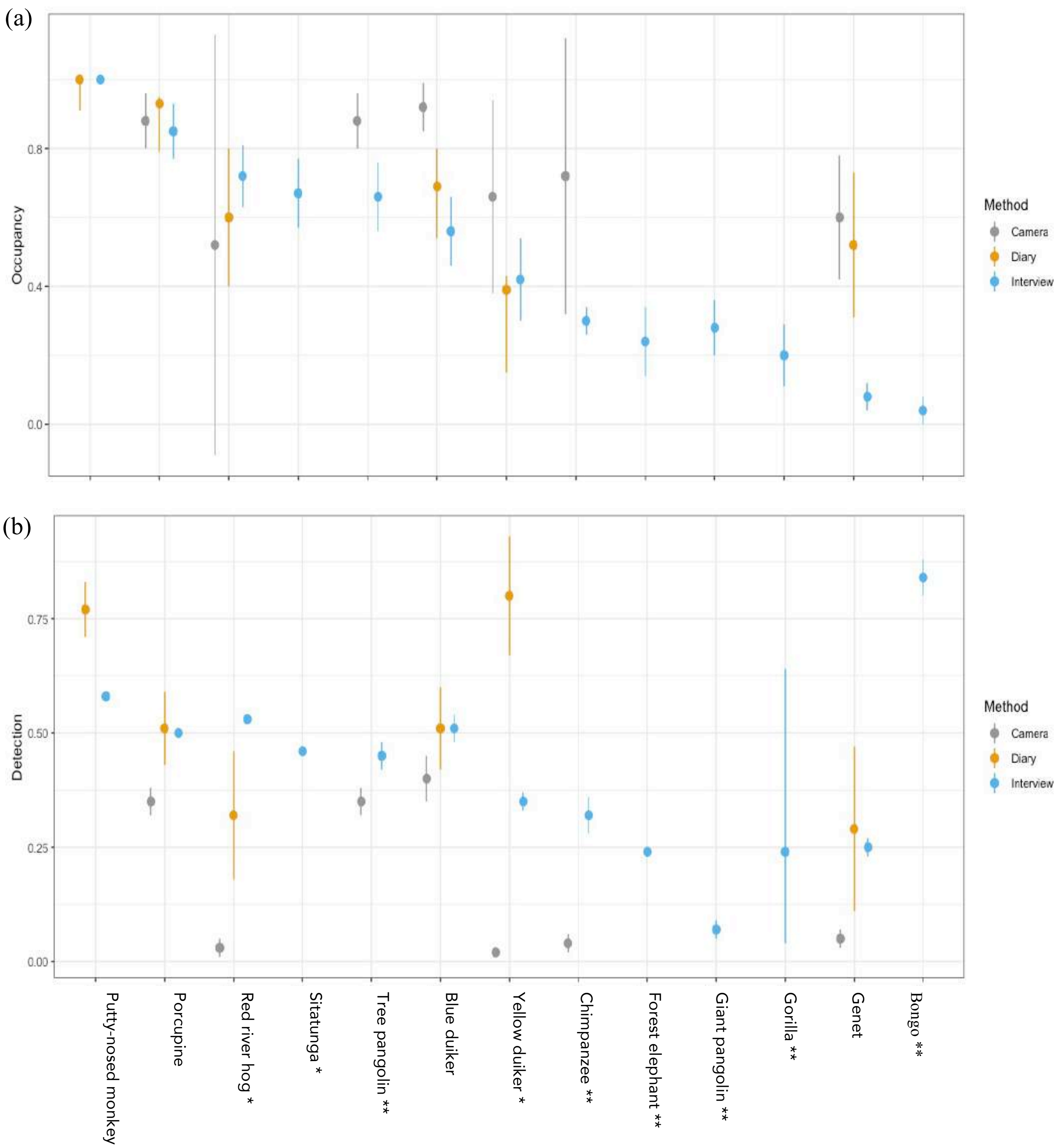
I interviewed a total of 109 local people in village 2, and each site was visited a mean of 16 times (range=2-71, median= 8). Ten hunters completed the hunter diaries and sites were visited a mean of 2.1 times (range=1.4-12.6, median=2). The memory cards for four of the cameras malfunctioned, and two further cameras failed to trigger so were not included in the study.

### 4.3.1 Species level comparisons of predicted occupancy and detection from different survey methods

Survey methods in village 1 provided estimates of occupancy for 13 (interview), 6 (diary) and 7 (camera) species respectively (figure 4-1). Where species estimates from interviews are shown with one or more other survey method (n=8), interview data were in agreement with at least one other method in 6 of those examples (figure 4-1). Interview and camera trap data agreed in five of seven cases. Estimates for porcupine agreed well in village 1 across methods (**Ψ** =0.85-0.93), as did estimates for red river hog (*Potamochoerus porcus)* (**Ψ** = 0.52-0.72). Interview and diary estimates for yellow duiker agreed well (**Ψ** =0.39-0.42). Where hunter diary data were shown with camera trap estimates (n=5), there was strong overlap in estimates between the methods every time. Gorilla and Sitatunga were recorded by only one camera in village 1, insufficient for occupancy analyses for these species.

Estimates of occupancy derived from camera traps resulted in the highest estimate in 4 of 7 cases. As hypothesised in table 4-4, detection rates from the camera traps were low for several rare and cryptic species, notably for chimpanzee (p=0.04) and yellow backed duiker (p=0.02). Unexpectedly, detection was also low for relatively common species like red river hog (p=0.03) and genet (*Genetta servalina)* (p=0.05). Low detectability from the camera trap data for chimpanzee, red river hog and yellow-backed duiker, resulted in wide confidence intervals occupancy estimates and greater uncertainty (figure 4-1).

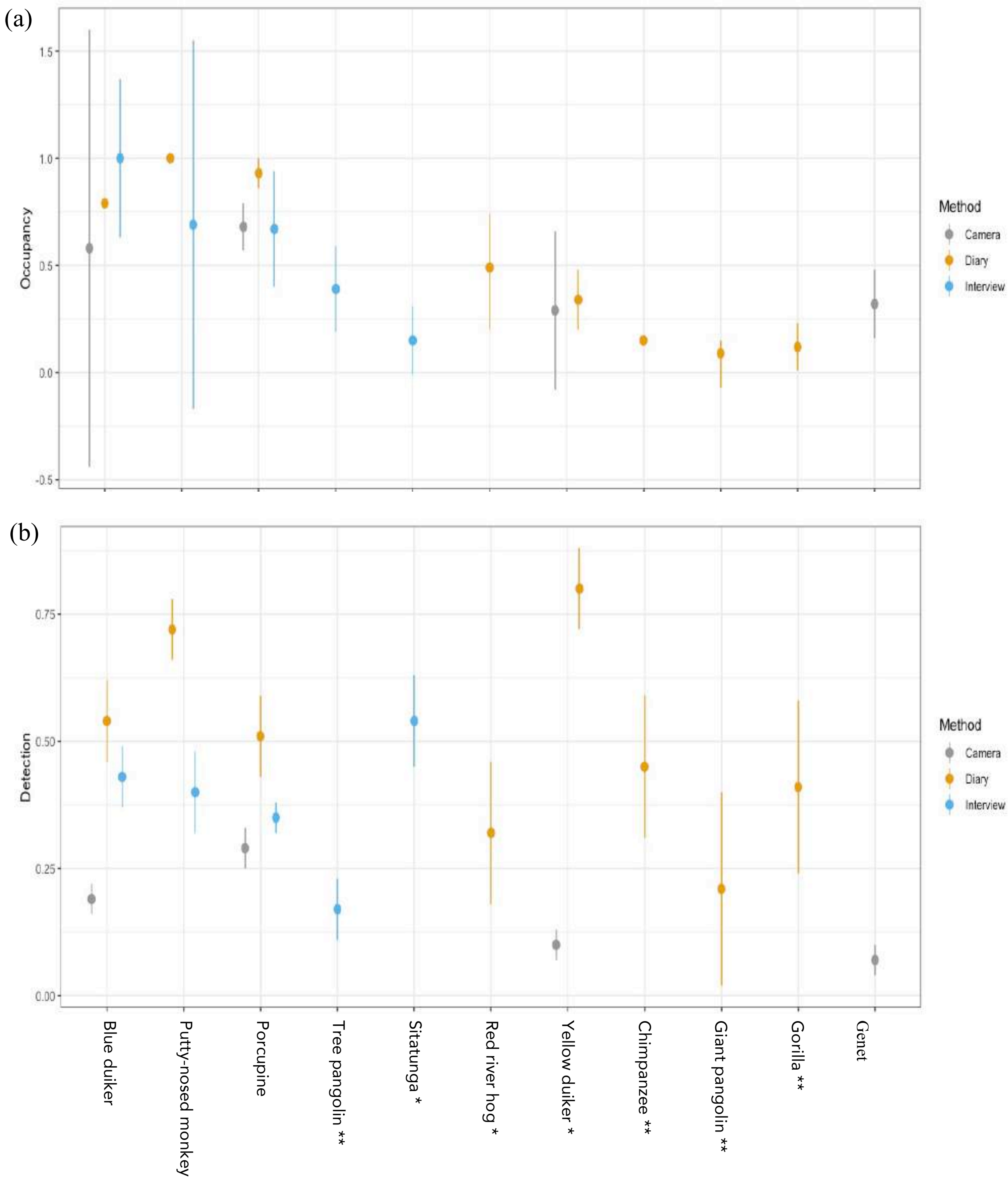
Figure 4-1: The back-transformed estimates of (a) occupancy and (b) detection from the top models in village 1, with their standard errors. See appendices C-4 for the full top models including the covariates. Cameroonian hunting classes are denoted by a *(class B) or ** (class A).



Survey methods in village 2 provided estimates of occupancy for 5 (interview), 8 (diary) and 4 (camera) species respectively. Where species estimates from interviews are shown with one or more other survey method (n=3), interview data were always in agreement with at least one other method (figure 4-2).

Interview-informed estimates ranged from **Ψ** =0.15 for sitatunga to **Ψ** =1 for blue duiker (figure 4-2). Diary-informed estimates ranged from **Ψ**= 0.09 for giant pangolin, to **Ψ** =1 for putty-nosed monkey (*Cercopithecus nictitans).* Camera trap estimates ranged from **Ψ** =0.29 for yellow backed duiker, to **Ψ** =0.68 for porcupine. Diary data was the only data available for rare species such as gorilla (**Ψ**=0.12), chimpanzee (**Ψ** =0.15) and giant pangolin (**Ψ**=0.09). Observational and camera trap data were in agreement in the three cases where camera trap data provided an estimate: blue duiker, porcupine, and yellow backed duiker. Chimpanzee and red river hog were detected by cameras, although not enough for occupancy analysis to be successful. Proven presence of these species in the community forests from the camera trap images suggest that the estimates from interview data may well provide us with a more robust estimate of occupancy compared to camera trap data for these species, in this setting.

Figure 4-2: The back-transformed estimates of (a) occupancy and (b) detection from the top models in village 2 with their standard errors See appendix C-5 for the full top models including the covariates. Cameroonian hunting classes are denoted by a *(class B) or ** (class A).



In summary, these results support the hypothesis that estimates derived from observational methods are largely comparable to camera trap estimates, but with species-level differences. As predicted, detections of rare species were too few from the camera traps, resulting in no model or highly uncertain estimates of occupancy compared to the observational data. Further, where species were detected infrequently by cameras, estimates from observational data may provide a more reliable estimate than camera traps.

Occupancy estimates from interviews were the most precise in village 1 (se=0.07), while diaries provided the most precise estimates in village 2 (se=0.09) (table 4-8). In both cases, estimates of detectability were the most precise from camera traps and interviews, and least precise for diary informed estimates (table 4-8). This may be due to the larger sample size obtainable with interviews compared to diaries and the greater coverage of area achieved within the 30km$^2$ study area as a result. Estimates of detection and occupancy were overall more precise in village 1 than village 2. While precise estimates may be more informative, it may also be that increased precision results in an inaccurate estimate, as upper and lower bounds of the estimate no longer capture the truth.

Table 4-8: Mean standard errors for occupancy estimates from each method averaged across the species

| Method | Village | Average occupancy SE | Average detection SE |
|---|---|---|---|
| Interview | 1 | 0.07 | 0.02 |
| Diary | 1 | 0.41 | 0.11 |
| Camera | 1 | 0.37 | 0.02 |
| Interview | 2 | 0.35 | 0.06 |
| Diary | 2 | 0.09 | 0.11 |
| Camera | 2 | 0.41 | 0.03 |

### 4.3.2 Comparing predicted occupancy and detection from different survey methods at the site-level

Here I have presented only the species with occupancy estimates from all three methods which allow for a full comparison of estimates between methods (table 4-9). In village 1, none of the methods resulted in significant occupancy variables for putty-nosed monkey, red river hog or tree pangolin (*Phataginus tricuspis).* Red river hog and chimpanzee had no significant detection variables. Chimpanzee occupancy increased with distance from river and in semi-deciduous habitat in the interview data. There was no agreement on the significant variables between methods in the top occupancy models (table 4-9). Detection variables were only significant for interview data, of which gender was the most important; women were more likely to detect species compared to men.

Table 4-9: Summary of the significant occupancy and detection covariates for each top model. See appendices C-6 to C-11 for the full top models. Green indicates significant positive and red indicates significant negative relationship between variables and with estimates of occupancy and detection. Significant for; I= Interview, D=Diary, C=Camera. DV=Distance from village, DR= distance from road, DRi= distance from river, H_Farm=Farm habitat, DPA= distance from reserve. Intercept: habitat=riparian, age=18-30, gender=female (G_M = gender male), frequency visit (Freq)=<weekly, time sleep (S_ )=<6 days at a time.

| | Occupancy covariates (psi) | | | | | | | Detection covariates (p) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Hunt class | DV | DR | DRi | H_SD | H_Farm | DPA | Age 31-44 | Age 45+ | G_Male | Freq_ week | Freq_ daily | S_15+ | S_ 11-15 | S_ 6-10 | Don't sleep |
| Blue duiker | | | I | | | | | | | I | I | I | | | | |
| Yellow duiker | B | | | | | | | | | I | | | | | | |
| Putty-nosed monkey | | | | | | | | | | I | | | | | | |
| Genet | | | | | I | | | | | I | | | | | | |
| Porcupine | | | C | | | | | | | I | | | | | | |
| Red river hog | B | | | | | | | | | | | | | | | |
| Chimpanzee | A | | | I | I | | | | | I | | I | | | | |
| Tree pangolin | | | | | | | | | | I | I | | | | | |
| Blue duiker | | | | | | | | | | | | | | | | |
| Porcupine | | C | | | | | | | | | | | | | | |

In village 2, there were no significant occupancy variables for blue duiker, or detection variables for ether blue duiker or porcupine. None of the variables were significant in the diary data, possibly because only a small proportion of the community forest had data so there was insufficient power.

### 4.3.3 Are site-level estimates of occupancy comparable between methods?

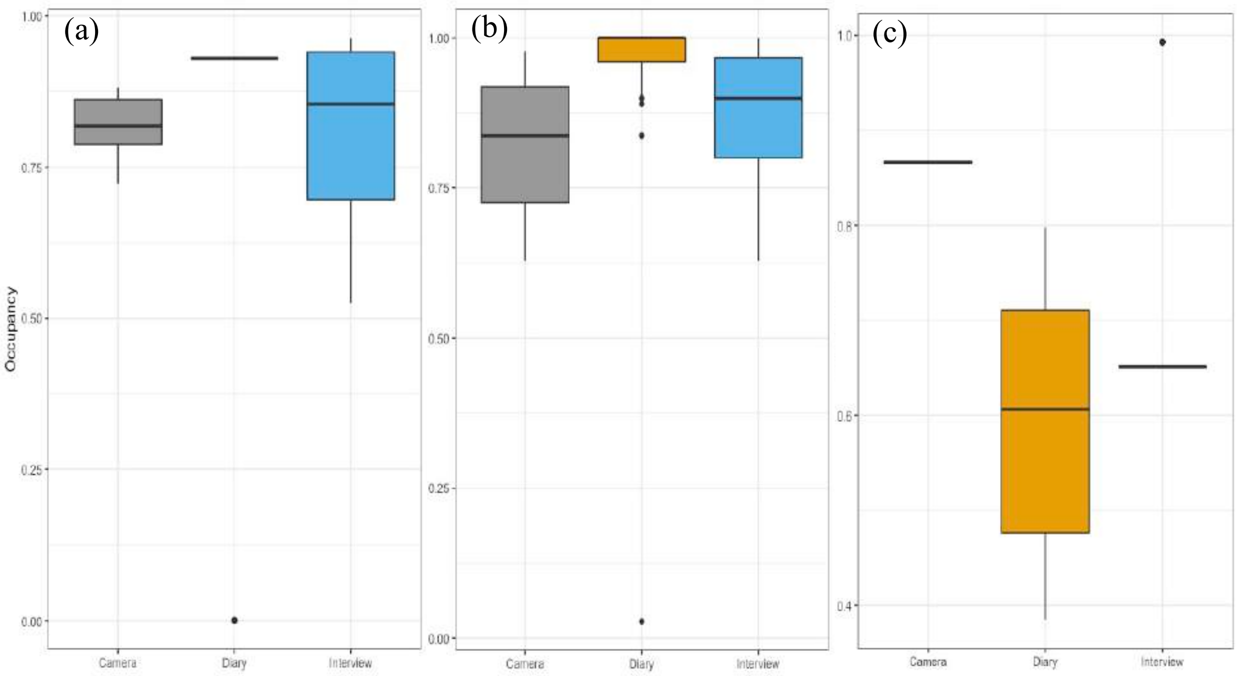
Site-level predicted occupancy in village 1 showed significant correlation between some of the methods (table 4-10). However, it was not always possible to conduct correlation tests because some of the species top models did not include an occupancy variable and therefore estimates of occupancy were constant across all sites. Strong and significant positive correlation was found between site-level predicted occupancy from interview and diary data for red river hog. Further, there is strong evidence to support that site-level estimates of tree pangolin and porcupine occupancy correlated strongly between camera and interview methods in village 1. No significant correlations were found for site-level predicted occupancy in village 2.

Table 4-10: Pearson correlation tests for site-level predicted estimates of occupancy in villages 1 and 2. 0-0.3=weak positive relationship, 0.3-0.5=moderate positive relationship and 0.5-0.7= strong positive relationship. 0- -0.3=weak negative relationship, -0.3-0.5=moderate negative relationship and -0.5-0.7= strong negative relationship. P=<0.05=significance. Green shows significant positive correlation. Where no colour fill, correlation is not significant. NA shows where 1 or more sample in test had constant occupancy across sites.

| | | Village 1 | | | Village 2 | | |
|---|---|---|---|---|---|---|---|
| Species | Hunt class | Interview & diary | Camera & interview | Camera & diary | Interview & diary | Camera & interview | Camera & diary |
| Blue duiker | | NA | -0.34 (0.9) | NA | 0.12 (0.51) | NA | NA |
| Yellow duiker | B | NA | NA | NA | | | |
| Genet | | NA | NA | NA | | | |
| Porcupine | | -0.52 (0.99) | 0.31 (0.05) | -0.18 (0.81) | NA | NA | NA |
| Red river hog | B | 0.53 (0.001) | NA | NA | | | |
| Tree pangolin | A | 0.31 (0.13) | 0.99 (0.001) | 0.3 (0.12) | | | |
| Chimp | A | -0.20 (0.86) | -0.39 (0.97) | -0.04 (0.58) | | | |
| Putty-nosed monkey | | -0.03 (0.56) | NA | NA | | | |

Predicted tree pangolin occupancy from camera and interview were highly correlated with very similar distributions in predicted unit level estimates (figure 4-3). Predicted porcupine estimates from camera and interview were also highly comparable. In both cases, predicted occupancy from diaries varied little between sites because the top model didn't include covariates sites, resulting in highly homogenous estimates of occupancy for these species across the site units. In contrast, estimates of predicted occupancy from the diary data were most variable for red river hog, where again a lack of occupancy variables in the top model resulted in a highly homogenous set of predicted occupancy estimates from interviews, but which were captured within the upper and lower bounds of the diary estimate.

Figure 4-3: Distributions of predicted site-level occupancy for species with significant Pearson correlations between camera trap, diary and interview data in village 1 for a) porcupine; b) tree pangolin and c) red river hog. The median (horizontal lines) marks the mid-point of the data. Coloured boxes show the inter-quartile range, which represents the middle 50% of the data for each monitoring method. Upper and lower whiskers show estimates that fall outside of the middle 50%. Points show outliers, where data lies outside of the expected range.




In summary, while the village-level results showed good agreement between methods, agreement between methods for site-level occupancy estimates is less certain. Of the 6 species with enough data to run comparisons, three species resulted in significant positive correlation between camera and interview and diary and interview estimates. However, as some of the species did not include an occupancy variable in the top model, scope for comparison across more species was limited.

### 4.3.4 What methods are most cost and time efficient?

Survey methods are only of use if they are not only robust but also affordable for sustained periods of time. Observational methods were the cheapest option in both villages, and camera traps the most expensive, even without the start-up cost of buying camera traps (table 4-11). This is because of the high cost of batteries required for long-term monitoring, high staff costs and the cost of data analysis, which is highly laborious. Hunter diaries were cheapest in village 2, while semi-structured interviews were cheaper in village 1. This is because of the additional petrol and accommodation costs that are required for trips to this village, and the high volume of repeat trips required for continual engagement with diary participants and the key contacts.

Conducting semi-structured seasonal interviews require the research team to be in the villages the longest (c=60-68 days), while hunter diaries were the least time intensive (c=36-60 days), as the return check-up visits did not require a full research team. All methods required start-up costs, such as scoping trips and maintaining relationships. However, in terms of equipment or start-up costs, observational methods were very low cost compared to the camera traps.

Table 4-11: Table of costs per trip and in total per year for each method and for each village.

| | Village 1 | | | Village 2 | | |
|---|---|---|---|---|---|---|
| Costs | Interview | Diary | Camera | Interview | Diary | Camera |
| Monitoring trip details | | | | | | |
| Number of staff required per trip | 3 | 1 | 6 | 3 | 1 | 6 |
| Days staff are in the field per trip | 15 | 3 | 21 | 15 | 3 | 21 |
| + Night accommodation required per trip | 2 | 2 | 2 | - | - | - |
| + Total trips needed | 4 | 12 | 2 | 4 | 12 | 2 |
| = Total days staff are in the field per year | 68 | 60 | 46 | 60 | 36 | 42 |
| Number of participants or cameras per survey trip | 141 | 10 | 60 | 100 | 10 | 60 |
| Accommodation costs (£5 per room per night) | | | | | | |
| Accommodation costs for staff (+ 1 driver) per trip | £40 | £20 | £70 | - | - | - |
| Staff costs (£9 each a day) | | | | | | |
| Survey team staff costs: salary and food per trip | £459 | £45 | £1242 | £405 | £27 | £1134 |
| Travel costs | | | | | | |
| Travel costs (petrol) per trip (take to field and return to Yaoundé) | £280 | £280 | £280 | £160 | £160 | £160 |
| Community participation gifts and remuneration | | | | | | |
| Participants gifts costs per trip (£0.80 per gift) | £112.8 | £8 | - | £112.8 | £8 | - |
| Total community monitor payment per key contact, for collecting and checking daily diaries. £6.50 per trip | - | £6.50 | - | - | £6.50 | - |

| | | | | | | |
|---|---|---|---|---|---|---|
| Data entry and analysis costs | | | | | | |
| Data entry costs per trip:<br>- per interview (£0.25)<br>- per daily diary participant (c. 8 data sheets each) (£1)<br>- camera trap data (£500 for 1-month internship) | £35.25 | £10 | £500 | £25 | £10 | £500 |
| Equipment | | | | | | |
| Camera trap start-up costs (£180 per camera) | | | £5400 | - | - | £5400 |
| Batteries £23.2 per camera (8 batteries per camera, 1 replacement) | | | £696 | | | £696 |
| Photocopying and paper per trip (£0.10 per participant) | £14.1 | £1 | - | £10 | £1 | - |
| TOTAL COSTS | | | | | | |
| Cost per trip | £981.15 | £370 | £2435 | £712.8 | £212.5 | £1910 |
| Total cost for the year (excluding camera start-up costs | £3924.6 | £4446 | £5213 | £2851.2 | £2550 | £1624 |
| Total cost (including cameras) | £3924.6 | £4446 | £10,613 | £2851.2 | £2550 | £10543 |

## 4.4 Discussion

There is a need for monitoring methods that are both cost-effective and robust, especially for monitoring in challenging habitats such as forests, and in potentially rapidly changing systems such as wild meat hunting villages and surrounding forests. This study represents one of only a few studies that compares interviews to camera traps to explicitly focus on better understanding uncertainty in estimates of occupancy. It is the first that uses comparisons with a conventional monitoring method to quantify the applicability of occupancy analysis informed by local people for monitoring mammals in forest habitats. The results are broadly in line with the a priori hypotheses (table 4-2).

Firstly, this study finds that where camera trap estimates are available, estimates of occupancy informed by both hunter diaries and interview data are broadly comparable to estimates derived from camera traps at the village scale. However, in village 1 camera trap estimates were consistently higher than those from either observational method. This contradicts Garrote & Pérez de Ayala (2015) who found that estimates from interviews overestimated occurrence and distribution. Occupancy estimates from the camera traps may be inflated due to the comparatively low detection rate, something that has also been observed in other previous studies where detection is low (Mackenzie et al. 2002; Mckann et al. 2012). Previous studies have found that monitoring with local people increases the likelihood of detection by going places we wouldn't usually go with cameras (Turvey et al. 2015; Martínez-Martí et al. 2016). While this study included the same 30km$^2$ area of community forest to enable comparison across three methods, still there is some evidence to support this as the rate of detection was often higher with observational methods compared to the camera traps.

While village scale estimates of occupancy were comparable, the evidence for comparable estimates at the 1km$^2$s was less clear. Three of the 6 species in village 1 had significantly correlated predicted occupancy estimates, but evidence was lacking for the other species, either because a) site-level predicted occupancy was not correlated between methods or b) because the top model for the species did not include occupancy variables, therefore estimates of occupancy were constant across sites, hindering any comparison of site-level correlations. A potential lack of site-level agreement in predicted occupancy has implications for future occupancy studies; if the aim of a study is to obtain overall estimate of occupancy at a village or landscape scale for example, then observational data may be a robust and cost-effective way to do this. However, if the aim of the study is to obtain estimates of the likelihood of occupancy at the site-level in order to predict habitat preferences or identify areas of conservation priority, for example, then more research is needed to understand the way in which results may differ between observational methods and those from camera traps. Brittain et al. (2018) used LEK combined with occupancy analysis to identify areas of conservation priority for forest elephants across the whole eastern region of Cameroon, concluding that the resulting maps of predicted occupancy were in line with up-to-date estimates of forest elephant density and distribution. It may be that in this particular study, the limited sample size rendered the variability in occupancy and detection harder to ascertain and predict, especially for species such as chimpanzee whose home ranges often extend to 15km$^2$ (Bryson-Morrison et al. 2017).

The results revealed species level differences in the effectiveness of different monitoring methods. I hypothesised that the camera trap detection rate would be too low for robust occupancy estimates for arboreal or rare species and the results support this hypothesis; the cameras detected gorilla, sitatunga and chimpanzee, proving their presence within the community forest, but at insufficient frequency for robust occupancy analysis. Therefore, estimates from observational data may have provided a more robust estimate of occupancy for these species. There are several potential reasons for this. Firstly, chimpanzees and gorillas are easily identifiable, making easily recognisable calls and nests, which may also increase detection from observational methods. Previous studies have also found that estimates from local people for primate population trends to be robust (van der Hoeven et al. 2004). Secondly, the observation-based detection rates for these species may be high because they each hold a cultural or economic interest, a trend that has also been reported in other studies (Martinez-Marti et al. 2016). More 'eyes on the ground' from observational methods increases the likelihood of detection for rare species, which has also been shown to be especially useful where species densities are low in other studies (Turvey et al. 2015; Martinez-Marti et al. 2016). In the face of a lack of data or low budgets for extensive or intensive camera trapping, these results suggest that LEK can provide informative estimates of occupancy.

The results show differences between methods in the significant variables that account for occupancy and detection. As expected, sociodemographic detectability covariates were more important in the interview than the hunter diary data, because the group of participants completing the diaries were much more homogenous and equally able to detect the species than the larger and more diverse group of participants taking part in the interviews. Gender played an important role in accounting for detectability in the interview data, with women reporting more detections than men (also see Chapter 5). T may be that the species better detected by women are of greater interest for women than men, and therefore noticed more readily by women. Alternatively, it may be that men who hunt, generally go further away from the community forest, and therefore women, who remain closer to the village during the period of time the camera traps were set, have a higher detection rate for the community forest area included in this study, although this heterogeneity should have been captured by the variables in the occupancy models. To further investigate the effects of gender on detection, a larger area should be used in subsequent studies, spanning a longer period of time.

The inclusion of observer-based and environmental variables in the top occupancy models highlights the importance of accounting for variation in both occupancy and detection. Occupancy analysis should be used to account for biases in detection and occupancy in all methods (Van Strien et al. 2013). However, there are some biases that occupancy analysis cannot account for, such as reporting bias or recall bias, which may affect diary and interview data in different ways. For example, participants may not be willing to report directly when they have hunted or even detected a protected species due to social desirability biases (see Nuno & St John, 2014), while completing anonymous diaries may make the participants more comfortable when reporting such activity compared to direct interviews, but may still suffer from recall bias (Gavin et al. 2010). However, so long as reporting probability isn't too low, occupancy analysis can still in principle give unbiased results by adjusting detectability appropriately. In this study, all efforts were made to minimise these types of bias (table 4-3) and the results do not suggest that respondents held back from reporting species detections.

In their large-scale comparison of wildlife distribution data between interviews with local people and camera traps, Caruso et al. (2016) found that interviews with local people cannot be adequately relied upon. However, as Petracca & Frair, (2017) eloquently pointed out, their study failed to account for differences in site size and location, violated the closure assumption and did not conduct repeat surveys at each site. As such they were unable to control for variable detectability, which this study has shown is highly variable between methods and must be accounted for. Poor survey design leads to flawed inference and calls into question the use of using static camera traps placed for a limited period of time to validate human observations over a long period of time. This study found that within the financial and logistical constraints of fieldwork, observational methods can be a reliable, rapid and cost-effective method of gathering occupancy data at the village scale in poorly understood systems. However, I did find large variability between species and villages in how the observational data relate to the camera trap data, which would require further investigation before robust conclusions can be drawn. Although the comparison conducted in this study was over a small spatial scale, it enabled a direct spatial comparison of estimates derived from camera traps, diaries and interviews. I have ensured that estimates of occupancy can be robustly compared through spatial and temporal matching of sites, over a period of time during which the closure assumption would not be violated.

While a small-scale study permitted me to ensure spatial and temporal matching, a lack of robust estimates from the camera traps, especially in village 2, limited my ability to provide more extensive species comparisons, and for conclusions to be drawn in a broader context. A greater camera trapping effort in terms of both the number of sites and the amount of time allocated may be required in future comparative research, to increase the detection of endangered and cryptic species that may avoid community forest areas. A larger survey area may also help to capture a wider range of environmental covariates and improve the site-level predictions of occupancy. However, increasing survey effort has budgetary implications which would need to be weighed up prior to starting monitoring species with low detection rates (see Guillera-Arroita & Lahoz-Monfort, 2012; Southwell et al. 2019; See Chapter 6).

Camera traps were selected as the comparison method because although they have their own biases, they remove the observer-based biases that are associated with both interviews and diaries. While I do not assume that the camera traps in this study represent the 'truth' in all cases, as is the case in other comparison studies (e.g. Caruso et al. 2016), future studies in locations where the 'truth' is known would be beneficial to assess the robustness of local knowledge.

# Chapter 5

## Eliciting expert judgement to inform monitoring of mammals in Cameroon

(Little girl watches her father as he identifies a series of animals as part of the expert elicitation protocol).

## 5.1 Introduction

Expert judgement represents a valuable source of information, especially when time and resources are stretched, or when extrapolations are required for novel, or uncertain situations (Kuhnert et al. 2010; Burgman et al. 2011). As such, expert judgement is commonly drawn upon for questions relating to climate change (Bamber et al. 2013), risk assessments (Goossens et al. 2001), health (Fischer et al. 2013), public policy (Morgan, 2014) and environmental impact assessment (Knol et al. 2010). Expert judgement has also been used in conservation to fill knowledge gaps in habitat suitability models (Aizpurua et al. 2015) conceptual models of migratory birds (Nelitz et al. 2015) and to provide estimates of species density (van der Hoeven et al. 2004).

There are many different views of what an expert is. An expert can be defined as someone who has knowledge of an issue at an appropriate level of detail and who is capable of communicating their knowledge (Meyer & Booker, 1990). Conventionally, experts are distinguished from lay people by their formal training and technical knowledge (known as their 'substantive' expertise (Stern & Fineberg, 1996; Burgman, 2016). However, Burgman et al. (2011) discuss how this definition of an expert may be too narrow, eliminating "lay experts" whose evidence is specific, concrete, and sensitive to local realities (Yearley, 2000; Irwin, 2001; Burgman et al. 2011). Burgman (2016) later goes on to define experts as people who are considered by their peers and society at large to have specialist knowledge and who are consulted to make an estimate or prediction. Subsequently, Hemming et al. (2017) take an even more broad approach, defining an expert as anyone who has sufficient knowledge to understand the question of interest.

Judgements may include facts or evidence recalled by the expert, inferences to new or undocumented situations, and the integration of disparate sources of information to address new problems (Kaplan, 1992). While experts have a wealth of experience and knowledge to help inform their decision making, asking for judgements differs from asking for information on known facts. As such, expert judgements in law, for example, have been likened to assessing the probability of an event occurring (Burgman, 2016).

Where robust empirical data is lacking, expert judgments may be the only, or most, credible source of information available (Carwardine et al. 2011). Yet as with all sources of information, bias can lead to poor inference. Just as the methods used for wildlife population monitoring are subject to scrutiny, so too should expert knowledge be scrutinized to ensure that uncertainty is quantified and bias is minimized (O'Hagan et al. 2006).

As with all sources of information, judgements are subject to bias that are often influenced by values and conflict of interest (O'Brien, 2000). In turn, this may influence whose judgements we can or cannot rely upon. For example, well-respected experts are often interviewed, as they are expected to have the knowledge required to make an informed judgement, increasing the likelihood that the judgement is robust (Martin et al. 2012). However, Burgman et al. (2011) test this idea, known as the social expectation hypothesis, finding that highly regarded experts were no better at providing accurate judgements than lay people. This is concerning, given that this is the process through which most expert elicitation processes select their participants.

This may be because highly regarded experts have been found to be more susceptible to ‘confirmation bias’ (Nickerson, 1998) confident in their ability and electing not to seek counterfactual evidence. As a result, they can be outperformed by non-experts who choose to use such evidence (Camerer & Johnson, 1991; Koehler, 1993).

This overconfidence often also stems from anchoring, whereby people fix on values that have been previously mentioned either by the participant or interviewer, and adjust their estimates from that point, a finding also echoed in Burgman et al.’s (2006) review of the expert elicitation literature. Experts may be vulnerable to anchoring, because anchoring may be relative to levels of confidence surrounding previous questions, resulting in judgements that have little to do with underlying knowledge. As such, where experts were previously confident in their estimates, they may be more likely to anchor. Tormala & Petty (2007) and Slovic et al. (1977) also found this to be the case in their analysis of the biases exhibited by lay people and experts when making estimates of risk.

Mcbride et al. (2012) assessed the degree to which judgments correspond to the truth (accuracy); the precision with which judgement predicts the true outcome (narrower confidence intervals indicate more precise and therefore informative judgements), and the degree to which the confidence intervals contained the truth as often as specified (calibration), comparing results from groups of experts and students. Experts were often overconfident, while students' 80% confidence intervals captured the truth 76% of the time, exhibiting almost perfect calibration.

To obtain judgements that are informative and robust, structured elicitation processes are commonly followed, a key aim of which is to enable participants to provide high quality estimates that reflect real life (Murphy, 1993) by helping participants to  engage with the uncertainty around their estimations and minimize bias. For example, where experts are asked to give estimates on species density or occupancy, expert elicitation protocols can help them to reach judgements that are in line with true trends, so that the data they provide can be robustly used to inform conservation decision making. Speirs-Bridge et al. (2010) defined a four-step approach to expert knowledge elicitation that requires the lower estimate first, followed by the upper and then the best 'mid-point' estimate, before asking experts to state their confidence in their estimates. Subsequent research supports that this approach reduces overconfidence and anchoring, and helps experts to construct their estimates quantitatively (Hemming et al. 2018).

Further developing the structured elicitation protocols to mitigate bias, Hemming et al (2018) set out the IDEA protocol, which stands for "Investigate," "Discuss," "Estimate" and "Aggregate". First, mixed groups of experts are recruited to answer questions with probabilistic or quantitative responses. Experts must first Investigate the questions and to clarify their meanings, and then to provide their private, individual best guess point estimates and associated credible intervals. Feedback is then provided to experts, relating to their estimates compared to the other experts. Experts are then encouraged to discuss the results, and provide a second and final private estimate. Notably, the purpose of discussion in the IDEA protocol is not to reach consensus but to resolve linguistic ambiguity, promote critical thinking, and to share evidence (Hemming et al 2018).

Another proposed way to overcome the biases of individuals is to avoid reliance on information provided by only one or a few experts, especially where indicators of performance are lacking (Armstrong, 2001). The 'wisdom of the crowds' phenomenon, discussed in detail with case studies by Surowiecki, (2004) shows that group level aggregation of information obtained from individual members often outperforms estimates made by any single member of a group. This phenomenon is supported by Burgman (2016) who found group averages to be consistently reliable, outperforming the best-performing individuals overall. Hemming et al. (2018) also found experts performed no better than non-experts, in their study involving 76 participants with varying levels of expertise, asked to estimate 14 future abiotic and biotic events on the Great Barrier Reef, Australia. The authors concluded also that group rather than individual estimates were more robust.

Van der Hoeven (2004) provides a rare application and assessment of expert judgment elicitation processes, for species monitoring. He asked local experts in Cameroon to give estimations of abundance for 33 mammal species across his study area, which he then converted to density estimates. Expert estimates were combined and averaged to get the pooled local expert opinion (PLEO). He concluded that pooled estimates of density were not only comparable to transect estimates and the wider literature, but were quicker and cheaper to obtain.

One area for potential expansion of the use of expert judgement in conservation is in studies on wild meat. Swan et al. (2017) use elicitation processes with 'conventional' experts to gather data on wild meat hunting. However, examples of structured elicitation processes being applied to local people and using a broader definition of an expert are lacking. Wild meat studies regularly incorporate local ecological knowledge (Azhar et al. 2013; Coad et al. 2013; Funder et al. 2013; Turvey et al. 2014, 2015; Golden &

Comaroff, 2015; Parry & Peres, 2015; Ávila et al. 2017) but do not gather this knowledge using structured elicitation protocols.

The lack of wild meat studies that apply structured elicitation protocols, especially when interviewing local people, is surprising, given the potential benefit of such approaches. Studies of wild meat are often limited by a lack of reliable and current data (Mayor et al. 2019) and by rapidly changing systems (Biggs et al. 2015; Luz et al. 2017). Regular interactions with the species or system of interest provides local people with valuable ecological knowledge, well placing them to give judgements on species populations (Cerqueira et al. 2013; Service et al. 2014) and current threats (Turvey et al. 2013, 2015), especially valuable where it is financially and logistically challenging to obtain timely and robust empirical data (Brittain et al. 2018). As the use of local ecological knowledge to inform conservation decisions increases, the robustness of judgements from local people remains under studied, and several unknowns require significant attention.

One significant unknown is around how to identify those able to provide the most reliable judgements, as sociodemographic variables may influence both perceived and actual levels of knowledge. Wild meat hunting is generally regarded as a male dominated activity in west and central Africa (Kumpel et al. 2009; Conteh et al. 2014; Dounias, 2016). Male hunters are frequently asked to contribute to studies of wild meat hunting (Levi et al, 2011; Parry & Peres, 2015; Barboza et al. 2016; da Silva Neto et al. 2017), while women are frequently interviewed in studies on wild meat preferences (e.g. Morsello et al. 2015; Golden et al. 2016; Luiselli et al. 2017). This is because young men are often thought to be more active hunters (Abere et al. 2016; Dai & Hu, 2017) holding the most up-to-date knowledge of the forest, while older participants may have greater accumulated knowledge. Strong hierarchical traditions between genders, ages and marital status may affect the way in which participants perceive each other's knowledge, resulting in unconscious bias (Rowlands & Warnier, 1988) but whether such variables affect the robustness of judgements has not been studied. Exploring the role of sociodemographic variables, in both perceived and actual levels of knowledge will improve understanding of whose knowledge can be best relied upon when asking for judgements in wild meat hunting systems.

Another unknown is the effect that the target species has on estimate robustness from different participants. Participants may be better able to provide estimates of abundance, density or distribution for wide ranging or abundant species (Turvey et al. 2014) that are regularly encountered. Or, estimates may be more robust for rare species such as forest elephants (Brittain et al. 2018) or large primates (van der Hoeven et al. 2004), encounters with which are easily remembered. Men may be better placed to give estimates for species that are frequently hunted, while women may be better placed to share information about species they encounter more in the farm, or in the areas of forest they frequent. Further, the abundance and distribution of species sensitive to overhunting, such as primates, large ungulates, and species targeted for illegal trade, such as great apes and elephants, may change rapidly; in such systems, estimates from participants may not be able to keep up with the rate of population change, resulting in biased estimates.

Comparison studies are widely used in ecology and conservation to identify the biases at play among different monitoring methods. Mcbride et al (2012), conducted a comparison study whereby expert judgement was compared to known 'truths'. In conservation, acquiring 'true' data may be feasible in systems where counts of absolute abundance are possible, with the use of aerial surveys for example. However, obtaining 'true' values of species distribution or density in forest environments is challenging, especially where projects have limited budgets. The same challenges apply to wild meat hunting systems, where data are lacking and 'true' species occupancy, density or abundance are rarely known. In such cases, estimates from local knowledge may be compared with estimates from commonly used monitoring methods such as transects, camera traps, or from studies with comparable habitats and species (Van der Hoeven et al. 2004).

In this chapter, I apply current expert knowledge elicitation and evaluation methods to assess the robustness of local expert judgements about species occupancy and density made by forest resource dependent participants in Cameroon. I 1) test the relationship between assessments of expertness and other correlates of performance such as such as age, gender, years living in village and time spent in forest; 2) assess the precision, and accuracy of expert estimates of occupancy and density compared to camera trap data and the wider literature; and finally, 3) identify the challenges to eliciting and assessing expert knowledge, developing the expert knowledge elicitation process by broadening its applicability beyond its predominate current usage in a standard developed-economy context (usually with respect to scientists) setting (table 5-1).

Table 5-1: Study research objectives, questions and hypotheses

| Research objective | Research questions | Hypotheses |
|---|---|---|
| 1. Test the relationship between peer-assessed expertise score and other correlates of performance. | 1a: What sociodemographic and performance-based correlates are the most important predictors of peer-assessment scores? | 1) There will be a strong positive correlation between self- and peer-assessed expertise scores (Burgman et al. 2011)<br><br>2) Peer-assessments will be also positively correlated with sociodemographic and experience-based variables such as age, gender, years living in village and time spent in forest. |
| | 1b: Does the social expectation hypothesis (whereby well-respected experts are expected to have the knowledge required to make an informed judgement, increasing the likelihood that the judgement is robust) in the context of wild meat hunters? | 3) There will be no correlation between peer score and actual performance in giving occupancy & density estimates (Burgman et al. 2011) (null-hypothesis) |
| 2. Test the precision and accuracy of expert estimates of occupancy and density, compared to camera trap data and wider literature | 2a. How accurate and precise are judgements from local people compared to estimates from camera traps and wider literature? | 4) Estimates of occupancy are more accurate and precise than estimates of density, because occupancy is easier to observe and perhaps easier to engage with than abundance. |
| | 2b. Do pooled estimates across the whole focus group capture the 'truth' and how do pooled | 5) Pooled expert-opinion performs better than the estimates derived from top-ranked experts in the groups (Hemming et al 2018; van der Hoeven et al. 2004). |

| | | |
|---|---|---|
| | estimates compare to those from high peer score experts?] | |
| | 2c. What is the role of experience variables (years in the village, frequency of forest visits and self-assessment) and sociodemographic variables (age, gender) in the ability to make robust judgements? | 6) Time spent in the forest and years in the village are good predictors of robust judgements for both occupancy and density<br><br>7) Predictors of peer-score, gender and age do not predict ability to make robust judgements for either occupancy or density (Hemming et al 2018; Burgman et al. 2017) (null-hypothesis). |
| | 2d. What species level differences in the robustness of judgements from different participants and methods are observed? | 8) Detections from camera traps will be too low for robust estimates of rare or shy species, such as sitatunga, gorilla, elephant and giant pangolin.<br><br>9) Participants will be better able to engage in estimates of rare species, or species that have economic or livelihood value, than highly abundant or less valuable species. |

## 5.2 Methods

### 5.2.1 Study site

Two contrasting wild meat hunting villages adjacent to the Dja Faunal Reserve in Cameroon were selected for this study, to explore how sociodemographic variables may affect the ability of participants to make judgements in both villages. For study site details please see section 2.1.10

### 5.2.2 Comparison datasets

'True' values of occupancy and species density are not known at either site. Rather, comparison occupancy and density estimates were obtained from camera traps and the wider literature. All methods are subject to biases, which I have tried to account for during analysis and interpretation (see table 5-2).

Comparison occupancy dataset

During September-November 2017 and April-June 2018 I placed 30 Cuddeback E2 passive infrared triggered cameras 30-60 cm above ground level for 60 days in the hunting territories of both villages, in places that may maximize detections (e.g. by a well-used animal trail, fruiting tree or water source). They were placed 1 km apart in a grid configuration, to reflect the spatial sampling units used for the interviews, and facilitate comparison between camera traps and expert judgement estimates. In total, 1440 camera trap days were collected from village 1 and 1880 from village 2. Please see section 4.2.2 for further details on the camera trap survey design.

Comparison density dataset

Comparison estimates of species density were extracted from the wider literature. Studies were selected based on their likeness to the study villages in terms of the date published, habitat and hunting pressure. For certain species, such as chimpanzees, density estimates across the literature were quite consistent, suggesting that those estimates can be used with relative confidence. In contrast, some species, such as sitatunga, had widely varying estimates of density in the literature, in which case estimates from study sites most closely matching mine were emphasised in comparisons. Identifying a robust estimate for giant pangolins is challenging, because so little is known about the species. In a recent review of method to detect and monitor pangolins, Wilcox et al. (2019) only found two published studies of wild giant pangolin populations, both from Cameroon (Ichu et al. 2017; Bruce et al. 2018). In these cases, we must interpret the results with caution, but also recognize the high potential value that expert judgement has for these data-poor species (see appendix D-1 for full table of estimates, their sources and how they were selected).

Table 5-2: Table outlining the different datasets used to collect information, their strengths and potential bias

| Method | Method strengths | Potential bias | Actions taken |
|---|---|---|---|
| Camera traps (occupancy comparison) | Cameras are set in the village community forests, allowing direct comparison between estimates of occupancy from camera traps and interviews.<br><br>Low human interference maximizes chance of detection rare and cryptic species.<br><br>As a widely used method, the biases are well understood and accountable for. | Densities for rare species may be too low to detect them, meaning that the estimate from the camera traps is incorrect, while perhaps the estimate from interviews is closer to the truth. Further camera trap placement (such as height, relation to paths and habitat) may bias estimates in favour of certain species over others | Camera placed to maximize detections and a good camera survey effort of 1440-1880 days was used to maximize chance of detections.<br><br>Species with an expected low density are noted, and comparison results for these species are treated with caution. |
| Wider literature (density comparison) | Estimates from peer reviewed journal papers that use conventional monitoring methods. | Lack of density data. Using estimates from Dja and comparable sites that may not represent the truth. Further, the methods used will also have their own biases which may not be well understood<br><br>A lack of agreement across estimates for some species means estimates for certain species must be treated with caution. | While not from the same site, we have used set criteria to ensure that the estimates that we do use are from systems that most closely resemble the habitat and pressures faced by biodiversity in these study sites. Please see appendices 1 for details of the criteria and the estimates used as comparisons.<br><br>Comparing estimates across many studies using a set criteria allows us to understand where there is agreement, and where there is uncertainty. |

| | | | |
|---|---|---|---|
| Participant judgement (occupancy and density) | Rapid, cost effective and includes local people in conservation. Participants cover areas that traditional methods cannot under budgetary and logistical restrictions and give estimates based on long-term associations, as opposed to snapshots in time. | In some cases, the participants may not be honest, when the species has an economic value (Grant & Berket, 2007), if they think the information will be used against them, or if they have not encountered the species. | No names were recorded on the data sheets, ID codes ensured the participants identity remained anonymous. Ensuring anonymity to participants helped reduce reporting bias. |

### 5.2.3 Expert elicitation method

I identified four groups of 8-12 participants in each village (n=80). In this study, I applied the rule by Hemming et al. (2017) when selecting participants, that they understood the study aim and questions, and were willing to take part. An equal representation of gender was included in each group, as were different ages and levels of forest experience, to allow exploration of how sociodemographic variables affect response robustness and peer-perceived knowledge. For further details on the ethical procedures followed, see section 2.1.17.

To obtain "self" and "peer"-assessment scores of expected performance, each participant introduced themselves to the group, and briefly described their experience with respect to the community forest and biodiversity. Participants were then interviewed individually to privately rank the other members of the group, including themselves, in order of most (score of 10) to least (score of 1) likely to give reliable estimates regarding species distribution and abundance, following Burgman et al. (2011). As such, when referring to 'experts' throughout this study, I use the definition of an expert set out by Burgman et al. (2015), as people who are considered by their peers and society at large to have specialist knowledge.

A participatory map delimiting the village zones of use (Newing, 2011) was created in previous surveys and was familiar to everyone in this study. The maps showed the location of each camera in relation to the village and identified points of reference such as paths, rivers and fruiting trees. Each participant was asked how many of the 30 camera trap sites they believe currently hosted each species in the study. A '1' for predicted presence and a '0' for absence was marked at each camera trap site on the map to gather data on predicted occupancy. A '?+' or '?-' was recorded at each site where participants were uncertain of presence (?+) or absence (?-), and if participants were totally unsure, a '?' was recorded in those sites, which overall provided an indication of participant precision and confidence. To reduce anchoring and prevent dominance bias, interviews were conducted individually and a new map was used for each species and participant.

If a species was reported as present within the community forest, participants were asked to provide their lowest, highest and best guess estimate for the number of individuals present across the total camera trapped area, in that order. Following this method reduces the effect of anchoring and increases the robustness of estimates (Spiers-Bridge et al. 2010; Burgman et al. 2011; Hemming et al. 2018).

Finally, participants were asked for feedback on how they found the questions, and whether it was easier for them to estimate certain species over others. Before finishing, participants were brought back into their original group and thanked with a small gift of cold refreshments, and any questions about the process were answered by the research team. See appendix D-2 for interview protocols used.

### 5.2.4 Analysis methods

To obtain estimates of density that can be compared to the best available literature, the lowest, highest and best estimates of abundance were divided by the total community forest area included in this study ($30km^2$), following van der Hoeven (2004). Species occupancy and density estimates were plotted against estimates from the camera trapping and wider literature, allowing for variation in the precision and accuracy of estimates in both villages to be explored.

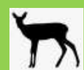
To visualize how well peer-assessed experts perform compared to pooled judgements from all participants, I plotted individual lower, higher and best judgements for each participant, and colour coded to show whether individual estimates were from those deemed an expert by their peers.

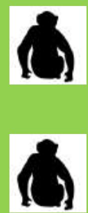
In order to assess the relationship between peer-assessment scores and the accuracy of occupancy and density predictions, we used two-tailed Pearson's correlation tests. Correlation tests were also used to investigate the sociodemographic variables that predict peer-assessment scores and draw inferences about the characteristics associated with robust judgements. To further investigate the importance of self-assessment and gender as predictors of peer-assessment, I combined the mean and self-assessment scores for all workshops and displayed the results as a scatter plot.

To investigate the variables that best predict whether an estimate is in-line with the comparison data (accurate) and informative (precise), generalized linear models (GLMs) with a binomial distribution and a logit-link function were employed using the 'glm' package in R (R Core Team, 2017). Estimates were deemed accurate if the participant's lowest and highest estimate captured the comparison estimate (following McBride et al. 2015). Precision was assessed by the confidence intervals provided. For the purpose of classifying estimates for analysis, a coefficient of variation of  less than 0.3 were considered precise. Separate models for precision and accuracy were also run, the results of which can be found in appendices D-9 to D-14. The top model was the most parsimonious as well as having the most support (AIC weight = 25%).

Binomial GLMs were run to investigate what variables best predicted the robustness of estimates. Robust estimates were defined as those that were both accurate and precise (1), while the remaining were deemed non-robust (0). Explanatory variables in the above binomial GLMs were selected a-priori based on the literature and my experience (appendix B-1). To avoid collinearity among variables in the model, Spearman's correlation coefficients were calculated for pairs of variables. No variables were highly correlated ($r > 0.8$) therefore all variables were included in the models (see appendix D-4for Spearman's rank correlation results).

Table 5-3: Table of the expected precision and accuracy of occupancy estimates from each method (camera, interview and literature), based on species characteristics. Number of animal icons represents the predicted occupancy estimate range for each method, where one=0-0.33, two=0.34-0.66 and three= 0.67-1. No species icon=predicted zero occupancy. Red = neither accurate nor precise, green= both accurate and precise, orange= precise but inaccurate, blue= accurate but imprecise.

| Species | IUCN Status | Hunt class | Cryptic/ shy | Vocal | Rare | Hunted | Camera | Interview | Literature | Details |
|---|---|---|---|---|---|---|---|---|---|---|
| Servaline genet (Genetta servalina) | LC | - | | √ | | √ |   |    |   | High enough detection for robust camera and interview estimates, although estimates may be higher from interviews because they are reported to be a nuisance. Few reliable estimates in the literature that can be compared to the interview data. |
| Tree pangolin (Phataginus tricuspis) | VU | A | √ | | | √ | |  |  | Too few detections for a precise camera trap estimate. Actively hunted therefore detection may be higher from interviews, although again the detection rate may not be sufficient for precise estimates. Estimates of density exist in the literature and are relatively similar across different sources |
| Blue duiker (Philantomba monticola) | LC | - | √ | | | √ |    |    |    | High detection rate will result in precise and accurate camera trap estimates of occupancy, although participants may struggle to give estimates of occupancy and density as they are perceived to be everywhere and highly abundant. Recent comparable data from the literature exists, with relatively consistent estimates across data sources. |
| Yellow-backed duiker (Cephalophus silvicultor) | NT | B | √ | | | √ |   |   |   | Lower density and detection than blue duiker in community forest, but an important species for wild meat and easily identifiable. I expect them to be at high enough densities for camera traps to produce accurate estimates, although with some uncertainty. Comparable |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | estimates exist in the literature, although there are large variations in estimates, therefore the comparison estimate may not be accurate. |
| Gorilla (Gorilla gorilla) | CR | A | | | √ | |  |  |  | Culturally and economically important species that is well recognized and at times hunted. Low densities in community forests, possibly too low for occupancy analysis to be successful. However, interviews will result in more detections due to its well-recognized call, nests and value. I expect people to be able to remember where they have seen them and approximately how many. Comparable estimates in the literature exist, with decent agreement |
| Chimpanzee (Pan troglodytes) | EN | A | | √ | √ | √ |  |  |  | Camera traps may not be best placed to provide robust estimates of chimpanzees as they are both terrestrial and arboreal. Easily recognizable and noisy makes them easier to detect via interviews. Consistent estimates in the literature, although chimpanzees are reported to be much more common in village 1 than village 2, which may affect the accuracy of estimates compared to the wider literature |
| Brush tailed porcupine (Atherurus africanus) | LC | - | | | | √ |  |  |  | Abundant and commonly hunted in both villages. High abundance may make precise estimates of density difficult. Comparable estimates from the literature exist, with good agreement |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Bongo (Tragelaphus eurycerus) | NT | A | √ | | √ | | | |  | Very rare and I expect there to be few to no detections within the community forests. No recent comparable estimates were found in the literature. |
| Giant pangolin (Smutsia gigantean) | VU | A | √ | | √ | √ | |  |  | Although very rare, giant pangolin has been reported within the community forest of village 1. However, abundance may be too low for any detections from the camera traps. As they are well-recognized and actively hunted I expect higher estimated occupancy and density from the interviews. Only a few estimates of density exist in the literature. |
| Forest elephant (Loxodonta cyclotis) | VU | A | √ | | √ | √ | |  |  | Rare and low densities expected in both community forests. I expect only a few detections within either community forests from interviews as signs are easily recognizable and none from the camera traps due to the relative proximity of the cameras to the village. Comparable data is available from neighbouring forests, but not from community forests where human presence is higher. |
| Sitatunga (Tragelaphus spekii) | LC | B | √ | | | √ |  |   |   | Mid-low density expected in community forests, I expect detection will be too low for camera traps to be informative. Quite cryptic and shy, but actively hunted and with recognizable signs, so interview estimates will be higher than those from camera traps. Comparison estimates found in the literature but estimates vary greatly. |

Different species are hypothesized to lend themselves better to certain monitoring methods, due to their species characteristics and predicted abundance within the community forests. I expect that highly abundant species such as blue duiker and porcupine will result in many detections from camera traps, improving the precision and accuracy of these estimates. However, participants may struggle to give precise estimates of their occupancy and density, given that they are reportedly so abundant. In contrast, rare species such as gorilla, chimpanzee and giant pangolin may be at too low density to be detected by camera traps. However, many of these species, such as chimpanzees, that are actively hunted by the community, or have distinct signs or calls that will increase the likelihood of these species being detected and reported via the interviews (table 5-3).

## 5.3 Results

A total of 80 participants were included in this study. 37 participants in village 1 and 33 in village 2 identified primarily as farmers, only 2 identified primarily as hunters in total (appendix D-5). 4 male participants were identified by their peers as 'experts' (i.e. had a total peer-assessed score of 10 in both villages. In village 1, peer-assessed experts were mostly young (26-40) while in village 2, older participants were deemed more knowledgeable (appendix D-6).

### 5.3.1 Correlates of peer-assessment scores

Two-tailed Pearson's tests revealed strong correlation in all groups between expectations of performance (peer-assessment) and self-assessment, with coefficients ranging from 0.71 to 0.97 (table 5-4). Being male was the most important predictor of peer-assessed knowledge, exhibited by a strong and significant negative correlation between gender and peer-assessment in all bar one group. Years lived in the village and time spent in the forest were less significant; three groups showed positive correlation between peer-assessment and years lived in village and one group had positive correlation between peer-assessment and time spent in the forest. There was no significant correlation between age and peer-assessment.

Table 5-4: Pearson correlation coefficients and p values (in brackets) between peer-assessments of performance and measures of expertise, and between self-assessed expertise and peer-assessments of expertise. Statistically significant correlations (at p=0.05) are in bold face.

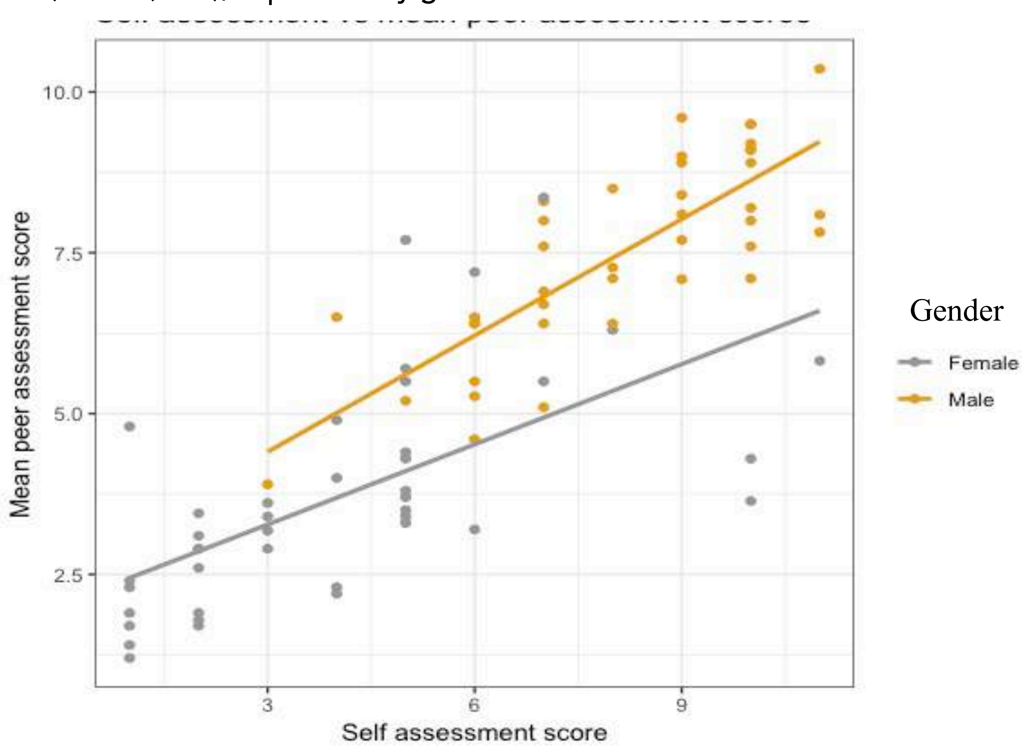
| Village | Group | Peer vs age | Peer vs gender (male) | Peer vs freq forest | Peer vs years village | Peer vs self-assessment |
|---|---|---|---|---|---|---|
| 1 | 1 | 0.34 (0.34) | -0.70 (0.02) | 0.17 (0.63) | 0.06 (0.85) | 0.90 (0.001) |
| 1 | 2 | 0.04 (0.90) | -0.85 (0.002) | 0.07 (0.85) | 0.22 (0.57) | 0.71 (0.02) |
| 1 | 3 | 0.40 (0.26) | -0.88 (0.001) | 0.02 (0.96) | 0.77 (0.01) | 0.97 (0.001) |
| 1 | 4 | 0.62 (0.06) | -0.93 (0.001) | 0.27 (0.45) | 0.84 (0.002) | 0.87 (0.001) |
| 2 | 5 | 0.27 (0.44) | -0.93 (0.001) | 0.15 (0.67) | 0.21 (0.56) | 0.97 (0.001) |
| 2 | 6 | -0.02 (0.9) | -0.81 (0.005) | 0.06 (0.86) | -0.50 (0.14) | 0.97 (0.001) |
| 2 | 7 | -0.13 (0.69) | -0.83 (0.001) | 0.73 (0.007) | 0.69 (0.01) | 0.79 (0.002) |
| 2 | 8 | 0.47 (0.24) | -0.21 (0.61) | 0.25 (0.55) | 0.26 (0.54) | 0.91 (0.002) |

In summary, these results show that participants were most influenced by sociodemographic variables such as gender when deciding on the peer-assessment score to allocate to other members of the group. By contrast, measures of experience did not influence peer-assessment score as hypothesized. This suggests that participants believe men to be the best placed to have access to knowledge and able to make robust estimates above any other consideration.

## 5.3.2 Assessing the social expectation hypothesis

I found very high correlation between peer and self-assessment scores (correlation, r=0.85). Participants believed that men would outperform women, shown by the higher mean self and peer score given to the male sample (figure 5-1). This is not a one-sided belief, as women consistently score themselves lower than men. Self and peer-assessed scores for women mostly sit between 1 and 6, whereas no men scored less than 3.

Table 5-1: Self vs peer-assessment of expertise for all participants on a scale of 1 (worst) to 10 (best), separated by gender.




Individuals with higher peer-assessment scores were less frequently accurate than those with low peer-assessment scores (table 5-5). There is only one very weak non-significant correlation in support of experts in group 4, while the significant correlations show that non-experts were significantly more accurate than experts.

Table 5-5: Correlation coefficients and p-values (in brackets) for comparisons between peer-assessments of expertise and the accuracy of occupancy predictions assessed against independent estimates. A negative correlation indicates that more experienced and better-credentialed participants were more accurate, while a positive correlation indicates the converse. Statistically significant correlations (at p=0.05) are in bold face.

| Workshop and number of participants | Peer-assessment against prediction accuracy of occupancy | Peer-assessment against prediction accuracy of density |
|---|---|---|
| 1, n=10 | 0.65 (0.04) | 0.57 (0.09) |
| 2, n=10 | 0.39 (0.26) | 0.40 (0.24) |
| 3, n=10 | 0.5 (0.13) | 0.08 (0.82) |
| 4, n=10 | 0.59 (0.07) | -0.11 (0.77) |
| 5, n=10 | 0.62 (0.05) | 0.46 (0.17) |
| 6, n=10 | 0.40 (0.25) | 0.44 (0.20) |
| 7, n=11 | 0.69 (0.01) | 0.41 (0.21) |
| 8, n=10 | 0.24 (0.51) | 0.58 (0.07) |
| Average | (0.16) (0.29) | 0.48 (0.50) |

In summary, these results show a strong relationship between peer and self-assessment scores, supporting Burgman et al. (2011) that highly respected participants also believed they will perform better. However, there is no evidence to support the idea that peer-assessed experts provided more accurate results than lay people.

When asked how easy or hard participants found the survey, both men and women frequently reported that the questions were hard, although men more frequently stated that they found the questions easy. Although not a significant variable in predicting peer-assessment, the interview data suggests that participants equated confidence in their ability to make judgements with their experience of the forest. This may explain the heavy male bias in expected knowledge, and a greater general level of confidence shown by men:

"Hard to estimate but I know the animals well so I can estimate well" (Female, village 1)

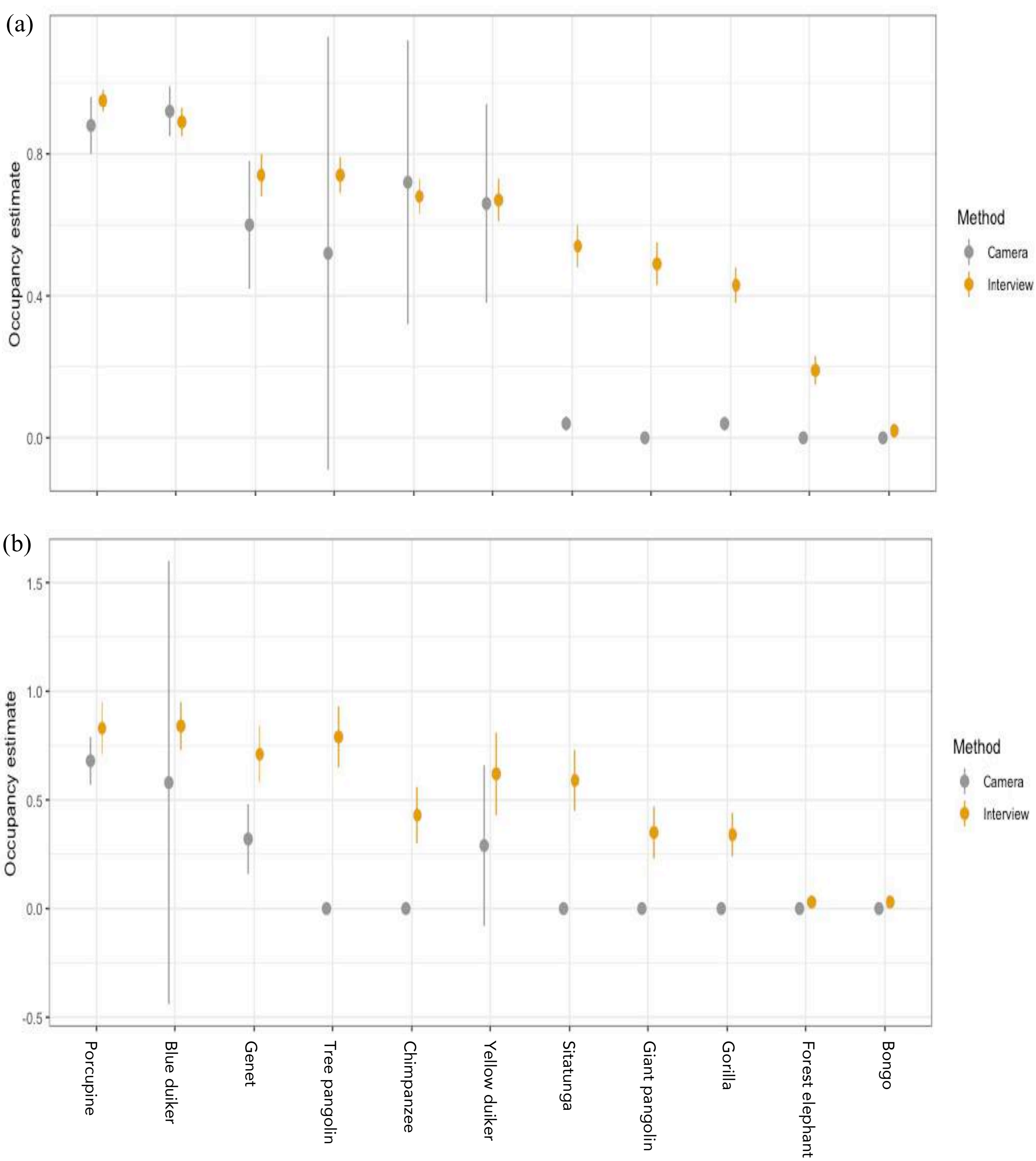
"Questions are easy because I master both the forest and the animals" (Male, village 2)

### 5.3.3 Comparison of participant judgements with camera traps and wider literature

Participant best estimate judgements of occupancy had consistently narrow confidence intervals in both villages, resulting in highly precise estimates that indicate good agreement between participants across all species (figure 5-2). Yet, there were large species-level differences in the accuracy of judgements when compared to the camera trap data.

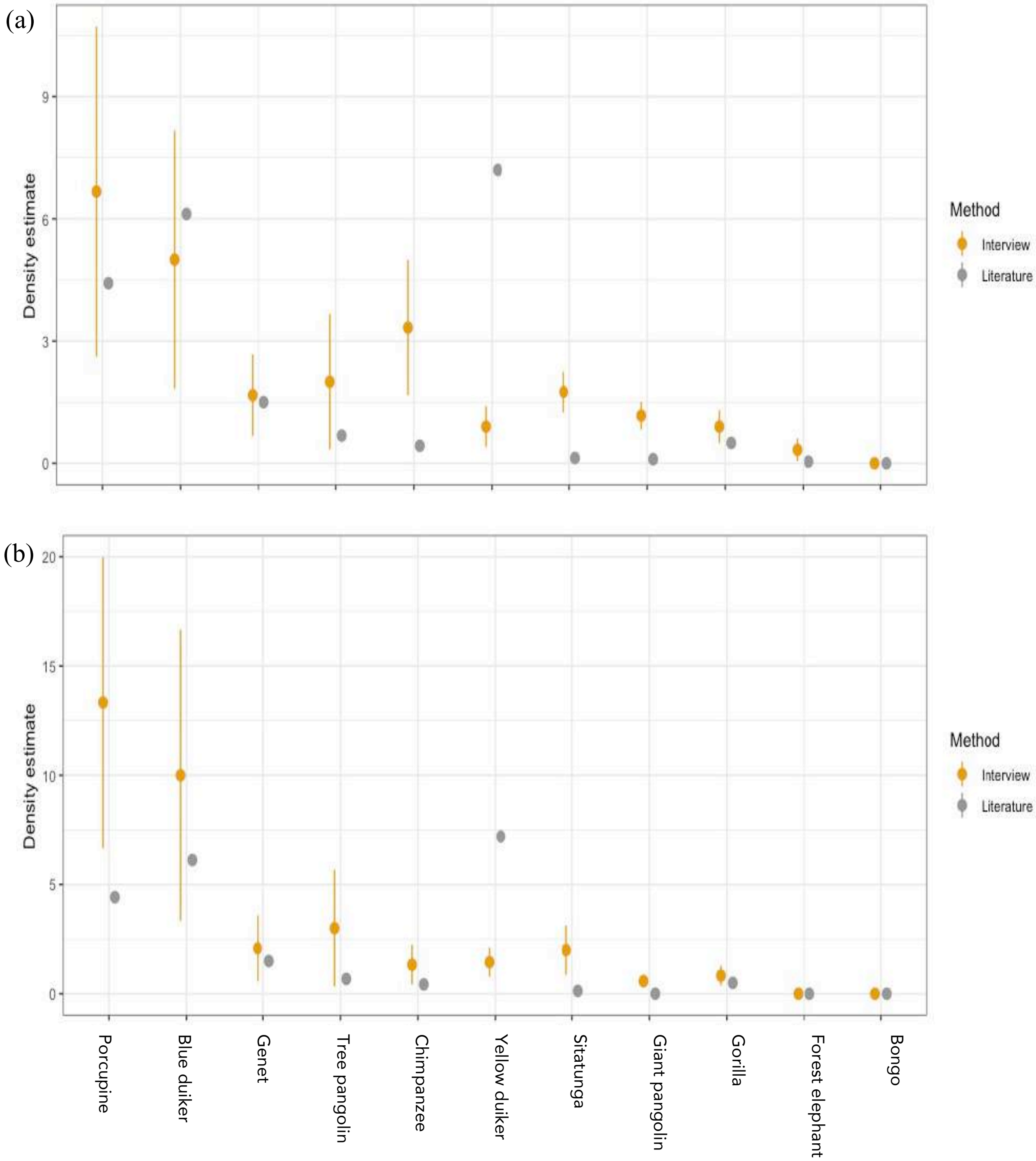
In village 1, participant best judgements sat within the lower and upper bounds of the camera trap data for all species except sitatunga, giant pangolin, gorilla and elephant, where judgements were higher than the camera trap estimates. Occupancy estimates for porcupine and blue duiker, both common and regularly hunted species, were highly comparable with the camera trap data. Contrary to my expectations, the camera trap data in village 2 were only precise for porcupine and genet. Detections of blue and yellow-backed duiker were insufficient for robust estimates, a surprising result given that blue duiker are expected to be one of the most resilient and abundant species in the area. Sitatunga, giant pangolin, gorilla, elephant, bongo, pangolin and chimpanzee were all absent from the camera trap data. Of these species, I only predicted bongo and elephant to be truly absent from the forest, suggesting that the remaining species were at too low densities for robust camera trap estimates.

Figure 5-2: Species occupancy estimates derived from camera traps (grey) and interviews (yellow) in village 1 (a) and village 2 (b). Bars show the range of the responses from participants for each species.



Participant estimates of density were generally precise for all species in both villages (figure 5-3). As expected, estimates for porcupine and blue duiker, both highly abundant and reportedly widespread, were the least precise, possibly because participants found it hard to think about estimates for species density when they were perceived to be so abundant. In both villages, yellow duiker was underestimated compared to the literature, possibly reflecting its shy and cryptic nature, which may diminish the perceived species density. In contrast the best judgements for giant pangolin and sitatunga were overestimated. These inaccuracies could represent rapid changes in species densities that participants have not yet caught up with. Or, it could be that the estimates from the literature were not representative of the densities present in these community forests (appendix D-1). Both giant pangolin and sitatunga are actively targeted by hunters in both villages and sensitive to overhunting, which suggests that for these species the former hypothesis may be more likely. However, it may also be that because both giant pangolin and sitatunga are actively targeted by hunters in both villages and sensitive to overhunting, that hunters have better current knowledge of these species occupancy than the literature estimates.

Figure 5-3: Species density estimates derived from the literature (yellow) and from interviews (grey) in village 1 (a) and village 2( b).




In summary, these results show species and village level differences in the robustness of judgements when compared to both the camera trap occupancy and wider literature density estimates. Judgements are often in line with the comparison data, especially for highly abundant and commonly hunted species. Where estimates are not in line with the comparison data, expert judgements of both occupancy and density were generally higher than the comparison data, particularly for species that are sensitive to overhunting or rare. One possible explanation is intentional overestimation by participants. However, participants were not all hunters, who may be the most motivated to give higher estimates in order to seem like more competent hunters. Further, participants were more likely to state that there were fewer animals in the forest as a result of hunting, than they were to tell me that the forests were highly abundant. This suggests that these species may be experiencing population declines, and participants are presenting estimates that better represent past densities.

When asked which questions participants found easy or hard, both male and female participants in both villages reported that overall, estimating abundance was perceived to be harder than estimating occupancy:

> "Some are easy and some are difficult- the map was ok but estimating the numbers is really difficult." (Male, village 2)

When asked what species were the hardest to report on, some reported that the rare species were the hardest, because they did not encounter them enough to feel that they had valuable insights to share:

> "Questions were easy. Estimating the numbers was the hardest and the elephant and giant pangolin were hard to estimate because I don't see them anymore." (Male, village 2)

> "I can't think for species such as elephant or gorilla, because I don't know where they like to go, what they eat…I don't know where to find them, so how can I show you on this map?" (Female village 1)

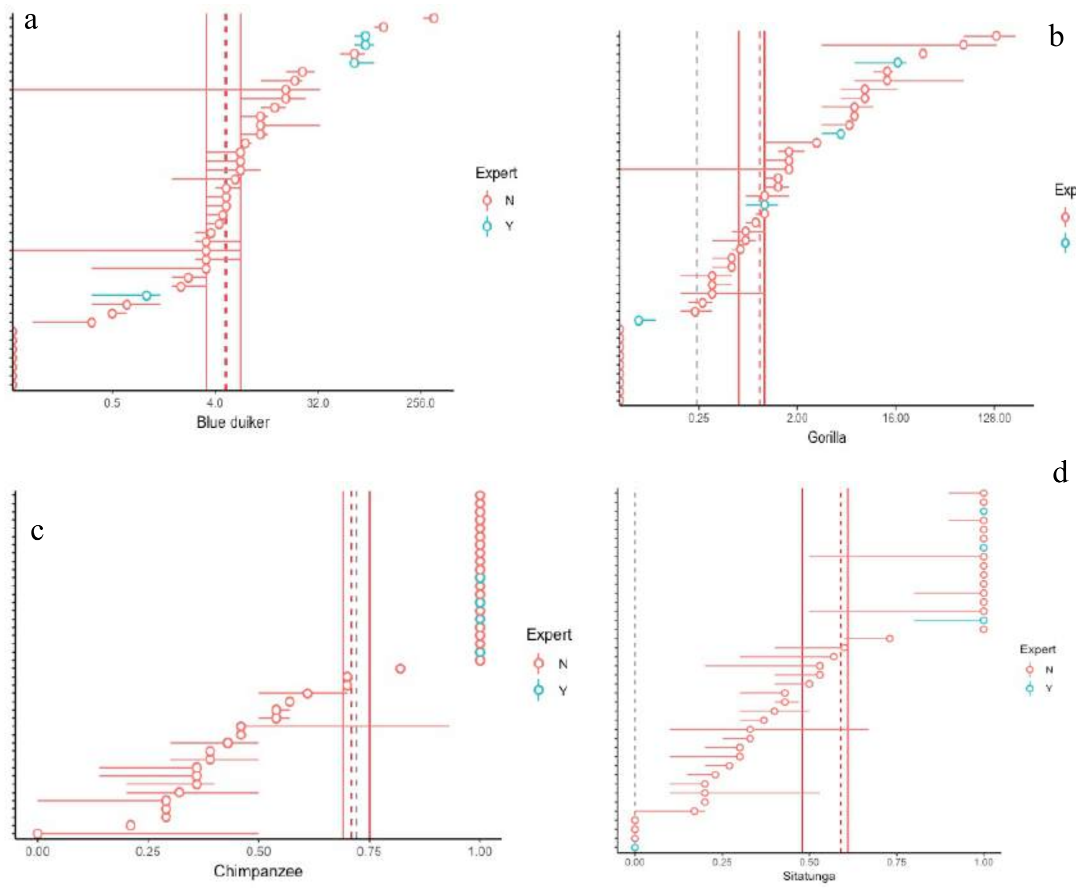
However, giving estimates for highly abundant species was also reported to be hard in both villages:

> "I see the species a lot, I come home with many species when I go into the forest. It's not easy to estimate when you see them all the time. What is your lowest, highest and best estimate for how many loaves of bread there are in London?" (Male, village 1).

### 5.3.4 Comparing individual estimates to pooled estimates

The group judgement for blue duiker exactly matched the estimate from the literature, while no judgement from peer-assessed experts captured the literature estimate (figure 5-4a). The group judgement for gorilla density overestimated compared to the literature, but was still closer to the comparison dataset than estimates from individual experts (figure 5-4b). The group judgement for chimpanzee occupancy captured the camera trap estimate, while again all 4 experts overestimated chimpanzee occupancy (figure 5-4c). One expert was in line with the camera trap estimate for sitatunga in village 2 (of no occupancy), while the group estimate and other experts overestimated occupancy compared to the camera trap. Where estimates are inaccurate, it was often due to overestimations (e.g. figure 5-4b).

Figure 5-4: Examples of Individual vs pooled participant estimates, demonstrating what a range of robust to inaccurate group judgements looks like for both occupancy and density, and how frequently and by how much individual peer-assessed experts were closer to the camera trap or wider literature estimates compared to the pooled judgement (full results are in appendices D-7 and D-8). Horizontal lines represent the lowest to highest estimates per participant, with their best estimate shown as a circle. The grey dashed line shows the estimate from the comparison dataset, so all estimates (horizontal lines) that cross it are comparable to that estimate. The solid vertical red lines show the median lowest and highest estimates from the pooled group estimate, and the red dashed line shows the median best guess for all participants. Where the solid red lines capture the grey line, the average estimate from the participants has captured the independent estimate. a) Blue duiker density, village 1. The grey line is not visible because the median pooled expert opinion is exactly beneath it, b) Gorilla density village 2 c) chimpanzee occupancy village 1 and d) sitatunga occupancy village 2. Red lines= non-expert participants, blue lines= peer -assessed experts participants




In summary, average group judgements were often closer to the comparison estimate than individual estimates from top ranked experts, who usually overestimated (appendices D-7 and D-8). Experts were never all in agreement and never captured the independent estimates at the same time.

### 5.3.5 What variables best predict a participant's ability to make robust judgements?

Participants were able to robustly estimate the density occupancy of two species each. 89% and 68% robustly estimated bongo and forest elephant density at close to zero, possibly reflecting a boundary effect which makes estimates of very rare or absent species easier for participants (table 5.6). The density of commonly hunted and rare, yet well-recognized, species such as pangolin (22%), giant pangolin (14.6%) and chimpanzee (17%) were also more frequently robustly estimated, while more shy and cryptic species such as sitatunga (4.8%) and yellow backed duiker (7.3%), despite being targeted for hunting, were the least robustly estimated. None robustly estimated blue duiker occupancy, again possibly reflecting how highly abundant species make providing precise confidence intervals harder.

Table 5-6: Total number of participants who are able to give a robust estimates for species (a) occupancy and (b) and density across both villages. Those species with >50% of participants giving robust estimates for one or other quantity are highlighted in bold. n=82.

| | Occupancy | Density |
|---|---|---|
| Species | % total sample | % total sample |
| Chimpanzee | **77%** | 17% |
| Bongo | 18% | **89%** |
| Porcupine | **97%** | 8% |
| Blue duiker | 0% | 24.4% |
| Forest elephant | 4.9% | **68.3%** |
| Genet | 33% | 15.8% |
| Gorilla | 15.8% | 14.6% |
| Pangolin | 8.5% | 22% |
| Giant pangolin | 1.21% | 14.6% |
| Sitatunga | 7.3% | 4.8% |
| Yellow backed duiker | 1.21% | 7.3% |

The robustness of density judgements was most strongly affected by participant years in village, gender and species (GLMs; variable importance = 1 in all cases, appendix D-13). Densities of commonly hunted species such as porcupine, blue duiker and pangolin were most likely to be robustly estimated (table 5-7), as well as chimpanzee. Women provided more robust judgements of density than men. Participants who have lived in the village for more than 40 years were more likely to give robust judgements compared to residents of 21-40 years, but less than participants who lived there for less than 21 years. However, while years in village is an important variable as shown by the variable importance of 1 (appendix D-13), the effect of years lived in the village was not significant (table 5-7).

Table 5-7: Summary of the top minimal model produced from binomial GLM, showing the predictors of whether an estimate of density are robust (1) or not (0). Estimates were deemed robust if they were both accurate (lower and upper estimates captured the estimate from the literature) and precise (SE of less than 0.3). This model was both the most parsimonious and had most support (AIC weight 25%).

| Reference | Variable | Estimate (SE) | Lower | Odds ratio | Upper | p-value |
|---|---|---|---|---|---|---|
| | Intercept | 1.12 (0.27) | 1.83 | 3.06 | 5.29 | <0.001 |
| Species reference = | Porcupine | 1.37 (0.43) | 1.73 | 3.92 | 9.58 | 0.002 |
| chimpanzee | Bongo | -2.43 (0.39) | 0.03 | 0.08 | 0.18 | <0.001 |
| | Blue duiker | -1.97 (0.35) | 0.06 | 0.14 | 0.27 | <0.001 |
| | Forest elephant | -2.52 (0.39) | 0.04 | 0.08 | 0.16 | <0.001 |
| | Genet | -2.62 (0.39) | 0.03 | 0.07 | 0.15 | <0.001 |
| | Gorilla | -2.62 (0.39) | 0.03 | 0.07 | 0.15 | <0.001 |
| | Tree pangolin | -2.11 (0.36) | 0.05 | 0.12 | 0.24 | <0.001 |
| | Giant pangolin | -2.84 (0.42) | 0.02 | 0.05 | 0.13 | <0.001 |
| | Sitatunga | -3.84 (0.57) | 0.006 | 0.02 | 0.06 | <0.001 |
| | Yellow-backed duiker | -3.41 (0.49) | 0.01 | 0.03 | 0.08 | <0.001 |
| Gender reference=female | Gender male | -0.65 (0.21) | 0.34 | 0.52 | 0.78 | 0.002 |
| Years reference= | Years 21-40 | -0.46 (0.24) | 0.97 | 1.58 | 2.56 | 0.06 |
| < 20 years | Years >41 | 0.43 (0.27) | 0.37 | 0.64 | 1.09 | 0.11 |

R2 = 0.29 (Hosmer-Lemeshow), 0.28 (Cox-Snell), 0.41 (Nagerkerke). p=<0.05

When selecting the best GLM to explain the robustness of occupancy judgements  (appendix D-14), the second model was the more parsimonious than the top model, and the variable importance of age in the top model only is low (0.52). Therefore, the second model was selected with (AIC weight = 31%, appendix D-14), in which all predictor variables (participant gender, the species, the village and their peer-assessed expert status) had a strong effect on the robustness of occupancy judgements (variable importance =1 in all cases).

Occupancy judgements for chimpanzees and porcupines were most likely to result in robust occupancy judgements, while all other species had a very low likelihood of a robust estimate. For example, there were no robust occupancy judgements for blue duiker at all (see table 5-6) as judgements were too imprecise (table 5-8), which explains the large CI for blue duiker. Contrary to the predictors of robust density, men were more likely to give a robust judgement of occupancy. Expertness also has an effect over and above gender; experts were nearly three times more likely to give robust judgements of occupancy compared to non-experts. Participants in village 2 were over eight times more likely to give a robust judgement compared to village 1. This could suggest either that participants in village 2 were better at giving robust judgements and engaging with uncertainty, or that wildlife populations in village 2 were more stable than those of village 1, allowing participants to give more accurate and precise judgements (table 5-8).

Table 5-8: Summary of the top minimal model produced from binomial GLM, showing the predictors of whether judgements of occupancy were robust (1) or not (0). Judgements were deemed robust if they were both accurate (lower and upper estimates captured the estimate from the camera traps) and precise (with a CV (coefficient of variation) of <0.3).

| Reference | Variable | Estimate (SE) | Lower | Odds ratio | Upper | p-value |
|---|---|---|---|---|---|---|
| | Intercept | 1.12 (0.42) | 1.34 | 3.06 | 7.16 | 0.008 |
| Species reference= | Bongo | -3.44 (0.46) | 0.01 | 0.03 | 0.07 | <0.001 |
| chimpanzee | Porcupine | 2.72 (0.78) | 4.02 | 15.3 | 100.9 | <0.001 |
| | Blue duiker | -20.57 (666.71) | <0.001 | <0.001 | <0.001 | 0.97 |
| | Forest elephant | -5.08 (0.63) | 0.001 | 0.006 | 0.02 | <0.001 |
| | Genet | -2.47 (0.41) | 0.036 | 0.084 | 0.18 | <0.001 |
| | Gorilla | -3.65 (0.47) | 0.009 | 0.025 | 0.06 | <0.001 |
| | Tree pangolin | -4.44 (0.54) | 0.003 | 0.012 | 0.03 | <0.001 |
| | Giant pangolin | -6.54 (1.07) | <0.001 | 0.001 | 0.007 | <0.001 |
| | Sitatunga | -4.62 (0.56) | 0.003 | 0.009 | 0.028 | <0.001 |
| | Yellow backed duiker | -6.54 (1.07) | <0.001 | 0.001 | 0.008 | <0.001 |
| Village reference= 1 | Village 2 | 2.10 (0.33) | 4.37 | 8.20 | 16.25 | <0.001 |
| Gender reference= male | Gender female | -1.02 (0.29) | 0.19 | 0.36 | 0.63 | <0.001 |
| Expert reference= no | Expert- Yes | 0.99 (0.43) | 1.13 | 2.70 | 6.37 | 0.02 |

R2 = 0.57 (Hosmer-Lemeshow), 0.46 (Cox-Snell), 0.69 (Nagerkerke). Model x2 (1) = 567.97 p=<0.001

These results show that certain variables such as village are only important for occupancy, while others are only important for density, such as years in the village. As expected, species varied in the extent to which experts were able to judge density and occupancy robustly. Rare species or species commonly hunted often resulted in the most precise and accurate judgements, while participants were less able to deal with uncertainty where species were deemed highly abundant. While I expected the participants years in the village to have an effect on judgement robustness, and found this to be the case for occupancy judgements, the effect of this variable on the robustness of density judgements was insignificant.

## 5.4 Discussion

This study provides the first known application of modern structured elicitation techniques within the context of a wild meat hunting system. As predicted from the expert elicitation literature (Burgman et al. 2011), those deemed to be the most knowledgeable by their peers are not always better at providing judgements. Rather, a combination of social and species level characteristics affects the robustness of estimates from local participants. However, conclusions drawn about the ability of participants to estimate density must be viewed as initial interferences based on that assumption that the data in the literature are accurate. As suc,h further tests where estimates of density are known are required to validate these findings. Studies that further develop structured elicitation protocols in different cultural contexts may improve the quality of data obtained from local people, and provide greater understanding of the uncertainty that surrounds subsequent estimates. The results and how they meet the hypotheses are summarised in table 5-9.

Table 5-9: Summary of whether the hypotheses are accepted or rejected

| Hypotheses | Accepted | Details |
|---|---|---|
| 1)There will be a strong positive correlation between self and peer-assessment expertise scores (Burgman et al 2011) | Yes | |
| 2) Peer-assessments will be a positively correlated with sociodemographic and experience based variables such as age, gender, years living in village and time spent in forest. | In part | Peer assessment was strongly correlated with gender, but not with experience variables. |
| 3) There will be no correlation between peer score and actual performance in giving occupancy & density estimates (Burgman et al 2011) (null-hypothesis) | No | There was evidence that peer-assessed experts performed less well than non-experts, supporting that the social expectation hypothesis is not always correct (Burgman et al 2011) |
| 4) Estimates of occupancy are more accurate and precise than estimates of density, because occupancy is easier to observe and perhaps easier to engage with than abundance. | No | Initial results suggest that density judgements were more robust than occupancy. Density estimates for rare species such as chimpanzee and elephant were the most precise, suggesting it is easier to engage with uncertainty when density is low. |
| 5) Pooled expert opinion performs better than the estimates derived from top ranked experts in the groups (Hemming et al 2018; van der Hoeven 2004). | Yes | Group judgements frequently outperform individual estimates from peer-assessed experts, and all experts never agreed on a robust estimate. |
| 6) Women and younger participants are less confident and therefore give less precise estimates compared to men and older participants (Hemming et al | In part | |

| | | |
|---|---|---|
| 2018). | | Women were less precise in estimates of occupancy than men, but were overall more robust in estimates of density, reflecting the different types of knowledge that men and women may hold. Age did not have a significant effect on the precision of estimates. |
| 7) Time spent in the forest and years in the village are predictors of robust judgements for both occupancy and density | In part | Years in village was in the top model for robust density estimates, although the effect of years was uncertain. While the RVI=1, the effect of each year's category was insignificant |
| 8) Peer-scored expertise, gender and age do not predict ability to make robust judgements for either occupancy or density (Hemming et al 2018; Burgman et al. 2017) (null-hypothesis). | In part | In these initial results, gender was a significant predictor of robust occupancy and density judgements, but age had no effect. |

### 5.4.1 Sociodemographic predictors of robust estimates

While gender has also been found to be an important predictor of peer-assessed scores in earlier studies of expert judgement elicitation (McBride et al. 2012), the importance of gender as a predictor of actual performance was surprising, as one should not expect to see a difference in performance based on a single demographic variable (although see chapter 3). However, given strongly gendered roles around wildlife and land use, the results make sense in this context. Yet, as predicted from the expert elicitation literature (Burgman et al. 2011), those deemed to be the most knowledgeable by their peers are not always better at providing judgements; contrary to participant expectations, men did not always perform better than women.

Men gave more robust estimates of species occupancy overall, but based on the assumption that the estimates from the literature were accurate, women were significantly more likely to give a robust estimate of density, suggesting gendered differences in the type of knowledge held by men and women in these systems. If this is truly the case, the reasons for this remain unclear; it may be that women are better able to engage with estimates of density than men, as they receive wild meat into the home, cook and prepare it and as a result may be better aware of what proportion of each species is brought into the household. The frequency of species brought into the home may broadly reflect species densities in the community forest. In contrast, men may be better placed to engage in estimates of occupancy, as they are frequently in the forest looking for wild meat and acquiring knowledge on the distribution of these species. Overall, these initial results reinforce differences between genders as a recurrent theme in the judgement and risk literature (Hemming et al. 2018).

These results are also interesting given that men are the only ones recommended as experts by their peers. A clear positive bias towards the perceived knowledge of men is demonstrated, while in reality there is no evidence to support the suggestion that men are any better than women at making robust estimates (cf Hemming et al. 2018). Unconscious bias may have affected how groups perceive their peers and themselves, a bias that is also rife in the traditional expert context (e.g. Baum & Martin, 2018). This unconscious bias may reflect the reasoning that wild meat hunting is a male activity (Newing, 2011) therefore knowledge on mammal species in the forest is equated to male knowledge. While recruiting participants for this study and when approaching households for interviews for other aspects of my work, women often deferred to their husbands at first, stating that they are the ones who go into the forest the most and will be able to help more than they can. While this may be the case for estimates of occupancy and species distribution, it is not the case for estimates of density. The disparity between perceived and actual performance highlights the importance of ensuring that diverse voices are heard. Snowball sampling is often used in conservation, but these results warn against relying on this method for expert elicitation purposes without the ability to validate the judgements provided.

An important conclusion from this research is that pooled opinion, while not always capturing the 'truth', did often perform better than the top experts, who tended to overestimate more than the pooled estimate. This finding supports the 'wisdom of the crowd' phenomenon (Surowiecki, 2004) and provides additional evidence that gathering judgements from groups results in more consistent estimates than relying on estimates from a handful of self and peer identified experts (Armstrong, 2001; van der Hoeven, 2004; Hemming et al 2018). While one or two experts gave accurate estimates for some species, it was never the case that all experts were robust in their estimates, a finding also reflected by Burgman (2016). Unless we are certain that a given expert is going to give a robust estimate for a given species, pooled expert opinion is generally a safer bet than individual opinion (van der Hoeven, 2004). However, while Burgman et al. (2016) found group estimates to be consistently reliable, pooled estimates in these villages tended to overestimate. The same overestimation has been observed in other studies when gathering data from both local people (O'Donnell et al. 2010; Mgawe et al. 2012; Caruso et al. 2016) and conventional experts (Burgman, 2016). This may be a bias, or it may reflect the nature of these rapidly changing hunting systems, especially when compared to questions relating to longer-term change, such as climate related issues that have also used expert judgement (Oppenheimer et al. 2016). More research on the effect of recall bias (see Jones et al. 2008; Golden et al. 2013; Newing & St John, 2013), and how quickly experts can keep up with change in systems such as these, would be highly valuable to improve the performance of expert elicitation in these contexts.

It's important to remember that the density results are based comparison estimates provided in the most comparable literature. As such, the conclusions drawn in this study should only be viewed within the context of assuming the literature is accurate, which may not be the case. Further studies when true density estimates are known would be beneficial to verify these findings.

### 5.4.2 Application of expert elicitation methods in the context of wild meat hunting systems

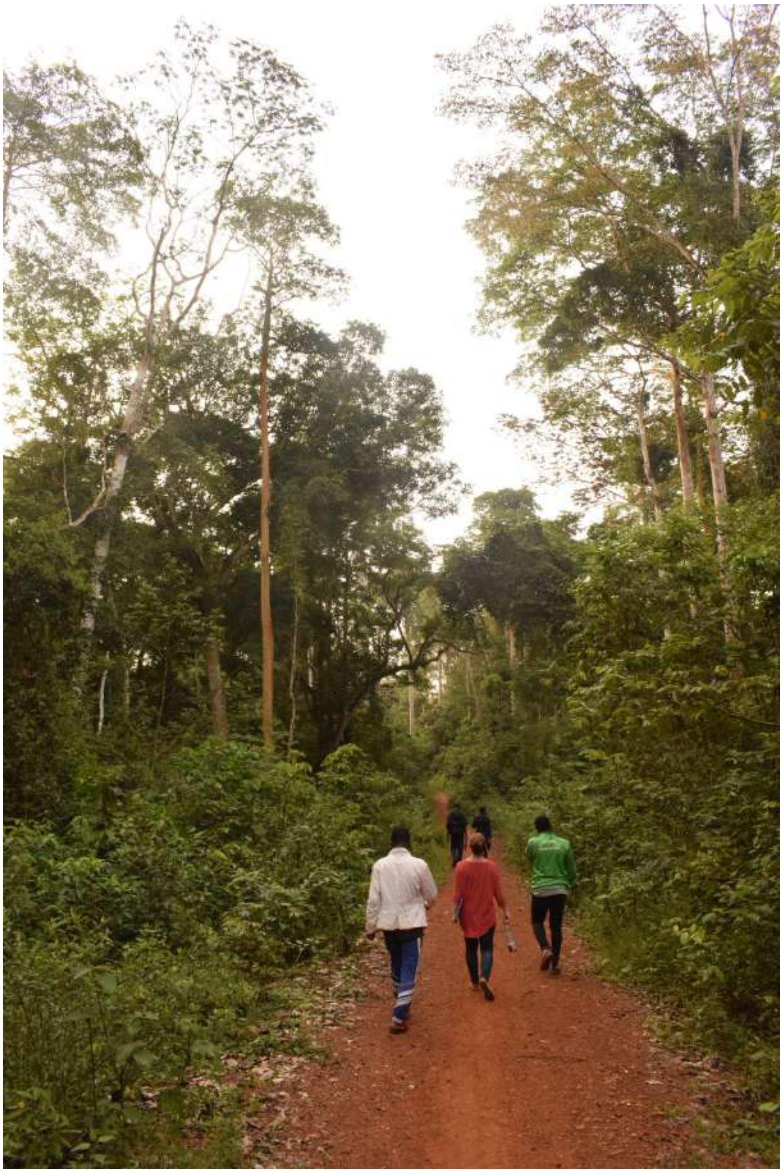
I predicted that experience-based variables such as years in the village, or time spent in the forest would best predict participant ability. However, species was the most important predictor of performance, appearing in all top GLM models. Estimates were more robust for rare or actively hunted species, findings that are also reflected in previous studies whereby participants were better able to give species estimates where the species is of cultural, economic or livelihood value (Martínez-Martí et al. 2016). This work supports the suggestion that judgements from local people may be valuable for conservation, especially for species at low densities, which renders gathering population data from camera traps expensive and time consuming. For example, there are few current estimates of giant pangolin densities available in the literature (Wilcox et al. 2019). In this study it was hard to ascertain the robustness of judgements with a lack of reliable and comparable density or occupancy data. However, participants were able to provide precise estimates of occupancy and density for giant pangolin, suggesting general agreement among participants and demonstrating that expert judgement could be an informative method to gather at least preliminary data on this endangered species where camera trap data are lacking. Similar results have been found in local knowledge studies on forest elephants in east Cameroon (Brittain et al. 2018) jaguars in Central America (Petracca et al. 2017) and for carnivores in mountain landscapes (Farhadinia et al. 2018).

We originally wanted to apply the full IDEA (Investigate, Discuss, Estimate, Aggregate) protocol outlined by Hemming et al. (2017), but getting the group back to discuss and refine their estimates didn't work. They had already given a few hours of their time and needed to get back to their work in the fields and in the households. Further, we found that people were strongly anchored to their first estimate, and did not understand why they would change their estimates based on the opinions of others. This limitation gives some support to Hemming et al. (2017), that participants were reluctant to update their estimates.

This study has found that judgments from local people, regardless of their perceived knowledge status, can provide valuable estimates (Burgman et al. 2011), especially where resources are lacking for robust estimates from conventional methods. As the use of local knowledge for conservation increases, researchers should be encouraged to properly engage with formal elicitation techniques to improve the robustness of their estimates by enabling participants to engage with uncertainty, especially when judgements are more complex (Hemming et al. 2017). More work is needed to see how to best adapt the IDEA technique or other structured elicitation protocols to overcome the challenges of implementing the full elicitation protocol in such a context.

# Chapter 6

## Power to the people: analysis of occupancy models informed by observational data from local knowledge



*(The research team walking to the next household interviews)*

## 6.1 Introduction

### 6.1.1 Species population monitoring

Monitoring is essential to identify trends in population size (Parry & Peres, 2015), assess a species' status (Witmer, 2005; Joseph et al. 2006; IUCN, 2012) the impact of threats (Kumpel et al. 2009) or measure the success of conservation programmes (Lindenmayer et al. 2012). Different state variables (e.g. density, abundance, occupancy) may be used for monitoring, but the choice of state variable should depend upon the monitoring objectives and target species (Joseph et al. 2006). All variables require different levels of effort and cost to monitor, and as such will determine the design and cost of the monitoring programme (Williams et al. 2002). However, many programmes are created without adequate attention to three key questions: 1) Why monitor? 2) What should be monitored? 3) How should monitoring be carried out? (Yoccoz et al. 2001). Poor consideration of these key questions often results in conservation resources being misspent on monitoring programmes which do not have the statistical power to detect a desired level of change (Robinson et al. 2018). Conversely, resources may be wasted on additional surveys which are not required to achieve the statistical power needed to detect a desired level of change.

### 6.1.2 Power analysis

Power analyses have been extensively applied to improve the cost-effectiveness of monitoring strategies for wildlife populations (Hatch, 2003; McDonald-Madden et al. 2010; Meyer et al. 2010); inform the design of presence-absence studies (Field et al. 2005); and determine the number of sites and surveys required to detect a given change with enough power in occupancy studies (Rist et al. 2009) especially helpful for sparsely distributed species (Latif et al. 2018). They help conservation practitioners ensure that monitoring does not waste valuable time and resources, by calculating the probability that a given survey design will detect a true trend (Gerodette, 1987).

Power to detect change is affected by sample size, effect size (the change we want to detect), variance, and the balance between **α** (the desired probability of incorrectly accepting the null hypothesis) and **β** (the desired probability of incorrectly rejecting the null hypothesis) (Legg & Nagy, 2006). Power is traditionally set at 0.8 (i.e. an 80% chance of detecting change over a specified time period), but the required sensitivity of the monitoring programme to detect a particular change depends on the research question.

Power can be maximised through careful survey design. For example, Blanchard et al. (2007) compared the power of different survey designs (fixed, fixed stratified, random, or random stratified) to detect known trends in the abundance of depleted fish populations, finding that to monitor species depletion, emphasis in survey design should be placed on coordinating the timing, areas of coverage, and methods of sampling. Hockley et al. (2005) use Monte Carlo simulations to investigate the power of density estimates of crayfish from local people, finding that the willingness of participants to contribute to monitoring was too low to result in statistical power to detect change. Jones et al. (2008) found that, because of differences in individual participants' activity or harvesting skill, the power to detect population trends improved by using the same observers in repeat surveys.

### 6.1.3 Power of occupancy models

Occupancy is the probability that a species occupies, or uses, a sample unit during a specified period of time during which the occupancy state is assumed to be static (Bailey et al. 2014). The presence-absence data required for occupancy analysis is collected through repeat visits to individual sites. The relative ease and cost-effectiveness of presence-absence data collection means that occupancy is often used in large-scale monitoring programmes, and is especially useful in projects with low budgets (Hedges, 2012; Blanc et al. 2014; Geyle et al. 2018).

When designing an occupancy study, three principal decisions affect power to detect change: 1) what data to collect;  2) the number of sites surveyed; and 3) the number of repeated surveys (occasions) at each site (Mackenzie & Royle, 2005). The financial and logistical constraints that apply to many monitoring programmes mean there is often a trade-off between the number of sites that can be surveyed and repeat visits. In these cases, the survey design should bear in mind the target species; for a rare species it is more efficient to survey more units less intensively, while for common species, fewer sampling units should be surveyed more intensively (Mackenzie & Royle 2005).

Many studies fail to account for imperfect detection, which is a common issue for rare or cryptic species (Yoccoz et al. 2001). However, imperfect detection can be addressed in occupancy models by estimating the probability that the observed occupancy status matches the actual occupancy status (Mackenzie & Nichols, 2004; Miller et al. 2015). The maximum likelihood model (MacKenzie et al. 2002), allows both detectability and occupancy to be estimated in a single-model framework by building a detection history from presence-absence data to construct a probability model, if five main assumptions are met (Mackenzie et al. 2006) (see table 4-7). Relationships between occupancy probabilities and detection and socio-demographic or ecological factors can be investigated by incorporating them as covariates in the analysis.

Power analyses rarely account for detection probability, which can substantially affect the power of data to detect trends based on occupancy models (MacKenzie, 2005). To address this issue, Guillera-Arroita and Lahoz-Monfort, (2012) developed and tested a formula for power analysis of occupancy data that accounts for occupancy and detection probabilities. They tested the formula with simulations to verify that the formula correctly calculated power and the number of sites that must be surveyed to achieve such power.

Other studies have since applied the formula to simulations for rare and elusive frog species (Barata et al. 2017), broad scale occupancy assessments of Hawk (Johnson et al. 2019) and with empirical data to identify grizzly bear range shifts in Canada (Steenweg et al. 2016) or the threatened brush-tailed rabbit rat (Geyle et al. 2018), finding the formula to be an effective way to assess the power of different occupancy study designs.

Earle (2017) first applied the formula to a real-life community monitoring scenario in Madagascar, in which patrols were nested within villages. She used the formula to test both the effect of different numbers of patrols and different numbers of villages on the power of occupancy models to detect change and the cost of monitoring under different budgetary scenarios. She found that the current monitoring regime used in Madagascar did not have the power to detect change for many of the species included in the study. Her study provided support for the need to conduct power analysis to ensure that the survey design is suitable for the question being addressed prior to starting a monitoring programme.

While the use of this formula is growing, there are currently no examples of its application to occupancy models informed by interview data from local people. Furthermore, no studies exist to explicitly compare the power of occupancy models informed by observation data to more standard monitoring methods such as camera traps, or to explore the effect of occupancy and detection on the power to detect change in forest habitats. Growing threats to biodiversity in the tropics have resulted in a need for effective monitoring that balances scientific rigor with practical feasibility (Rist et al. 2010). With the rise of interview-based monitoring methods in response to this need, studies are required to identify the optimal survey strategy to ensure power to detect an appropriate level of change in the most cost-effective way.

### 6.1.4 Aim & research questions

The aim of this research is to explore the power of occupancy models to detect change under different survey designs and budgetary scenarios, informed by three monitoring approaches. I assess the power of camera trap data, seasonal interviews and daily diaries collected from local people to detect change in occupancy for 14 different mammal species, either commonly hunted for wild meat or of conservation interest. In this study I ask the following questions:

1. What is the statistical power to detect trends in populations, across the different data collection methods?
2. How does the statistical power of each method to detect change vary with varying numbers of repeat visits and sites?

3. What is the optimal survey design and method to detect change for the different species on a given budget?

## 6.2 Methods

### 6.2.1 Study site

See section 2.1.10 for study side description

### 6.2.2 Data collection

Camera traps, semi-structured interviews and daily diaries were used to collect presence/absence data. See section 4.2.2 for details of the methodologies used. However please note for this chapter, all sites with data were included in the study; occupancy analysis was not restricted to the same $30km^2$ area as per chapter 4.

### 6.2.3 Data analysis

*Occupancy models*

Detection histories for each $1km^2$ site were created by arranging the data into presence/absence (1/0) of a species during repeat visits to a site. For camera traps, the sampling occasion was set at 5 days for the camera traps. This was chosen as a compromise between model stability and ensuring an adequate number of repeat visits to each site (Burton et al. 2015). For observational data, individual participants were treated as repeat surveys. Single species, single-season occupancy models, originally designed by MacKenzie et al. (2002), were performed using the package "unmarked" in R version 3.4.2 (Fiske & Chandler, 2011; R Core Team, 2017). See table 4-5 for a table of the covariates included in the occupancy analysis and why they were included.

*Power analysis of occupancy models*

Guillera-Arroita and Lahoz-Montfort's (2012) formula in R was used to assess and compare the power of occupancy models under different scenarios of survey design and budgets. In all analyses the significance threshold alpha ($\alpha$) was set at 0.05 and the desired power was 0.8, in keeping with the standard in ecology. The questions I explored were:

- What is the statistical power of the current regime to detect trends?

  The power of each species to detect trends between two sampling periods of 10%, 30%, 50% and 80% growth and decline (relative proportional change = R, where $R < 0$ is a decline, $R > 0$ is growth), given the sampling approach employed in this study. The scenario is that: Data from interviews and diaries were opportunistic. Participants were not 'monitors'; they did not collect data systematically, or survey the forest in a systematic way. The number of repeat samples, participants and sites varied with each survey, so the median number of sites and repeat visits per site were used for analysis.

- How does statistical power change with varying number of sites and repeat visits?

  The number of $1km^2$ sites was halved, doubled and tripled and the number of repeat surveys per site were held constant. The process was then repeated, holding the number of sites constant, and halving, doubling and tripling the number of repeat visits (table 6-1). Halving, doubling and tripling is a common approach in studies of power to determine optimal survey design.

- What is the most robust and cost-effective monitoring strategy to detect trends?

  I determined the minimum number of sites and repeat visits required to detect decline and growth of 10%, 30%, 50% and 80% with 80% power, and the costs of detecting 50% growth and decline.

Table 6-1: Outline of the current, halved, doubled and tripled regime in terms of the total participants, sites and visits, per monitoring method

| | | Village 1 | | |
|---|---|---|---|---|
| Method | Plan | Total participants | Total sites (S) | Median repeat visits (K) |
| Interview | This study | 141 | 184 | 4 |
| Diary | This study | 10 | 175 | 2 |
| Camera | This study | 26 | 26 | 12 |
| Interview | Double | 282 | 368 | 8 |
| Diary | Double | 20 | 350 | 4 |
| Camera | Double | 52 | 52 | 24 |
| Interview | Triple | 423 | 552 | 12 |
| Diary | Triple | 30 | 525 | 6 |
| Camera | Triple | 78 | 78 | 36 |
| Interview | Half | 71 | 92 | 2 |
| Diary | Half | 5 | 88 | 1 |
| Camera | Half | 13 | 13 | 6 |

*Cost data and management strategies*

The costs of achieving 80% power over a three-year monitoring period were calculated based on the approximate costs incurred while using these monitoring methods over the past 3 years with ZSL in Cameroon. There are differences in cost between villages depending on the distance required to travel, and the costs required to reach this study village are quite high compared to other, less remote, villages. However, the animals in this study village may be less depleted than in other villages, meaning that the effort required to detect changes with 80% power may be less than it would be elsewhere. The total yearly cost for monitoring was divided by the number of trips required to conduct interviews, collect diary data, or check the camera traps, in order to obtain the cost per trip required for each monitoring method. This gives an indication of the overall costs for each method and strategy, and how those costs may vary according to different staff requirements, number of trips or the number of participants and camera trap days required under each scenario. Although I calculated the approximate start-up costs required for camera trapping, these costs were not included in the calculations. This is because the one-off cost of buying cameras can easily be added onto the total cost of monitoring if necessary. Adapting the formula developed by Earle (2017), I calculated the total cost (C) in GBP£ of a monitoring regime as:

$$C = Y*S*K*a \text{ (Equation 6.1)}$$

where Y is the number of years the monitoring project will operate, S is the number of villages included in the monitoring, K is the number of repeat survey visits to the village per year, and a is the cost per repeat survey. Table 6-2 outlines the monitoring scenarios used for each monitoring method and table 6-3 shows the cost breakdown used to calculate the cost of monitoring under each scenario.

Table 6-2: Outline of the different monitoring scenarios for interviews, camera traps and daily diaries.

| Strategy | Interview | Camera | Diary |
|---|---|---|---|
| A Year round, high intensity | 4 surveys, once a season. All potential participants are interviewed by 3 staff members. | 60 cameras are set over 12 months. 6 trips to change cameras every 2 months. Team of 6 | 10 participants, constant monitoring for 1 year. 1 village monitor and 1 team member |
| B Year round, low intensity | 4 surveys, once a season. 50% of participants are interviewed by 2 staff members. | 30 cameras are set over 12 months. 6 trips to change cameras every 2 months. Team of 3 | 5 participants, constant monitoring for 1 year. 1 village monitor and 1 team member |
| C Seasonal, high intensity | 1 survey, covering 1 season. All potential participants are interviewed by 3 staff members. | 60 cameras are set over 2 months. Team of 6 to set and collect cameras (2 trips) | 10 participants, constant monitoring for 2 months. 1 village monitor and 1 team member |
| D Seasonal, low intensity | 1 survey, covering 1 season. 50% of participants are interviewed by 2 staff members. | 30 cameras are set over 2 months. 2 trips to set and collect cameras by a team of 3 | 5 participants, constant monitoring for 2 months. 1 village monitor and 1 team member |

Table 6-3: Description and cost per trip for each monitoring strategy, and total costs for the year based on number of trips required

| | Interview | | | | Diary | | | | Camera | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Unit | A | B | C | D | A | B | C | D | A | B | C | D |
| Monitoring trip details | | | | | | | | | | | | |
| Number of staff required per trip | 3 | 2 | 3 | 2 | 1 | 1 | 1 | 1 | 6 | 3 | 6 | 3 |
| Days staff are in the field per trip | 15 | 8 | 15 | 7 | 3 | 3 | 3 | 3 | 21 | 21 | 21 | 21 |
| + Nights accommodation required per trip | 8 | 8 | 2 | 2 | 24 | 12 | 4 | 2 | 12 | 12 | 4 | 4 |
| + Total trips needed | 4 | 4 | 1 | 1 | 12 | 6 | 2 | 1 | 6 | 6 | 2 | 2 |
| = Total days staff are in the field per year | 60 | 30 | 15 | 7 | 36 | 18 | 6 | 3 | 126 | 126 | 42 | 42 |
| Number of participants or cameras per survey trip | 100 | 50 | 100 | 50 | 10 | 5 | 10 | 5 | 60 | 30 | 60 | 30 |
| Accommodation costs (£5 per room per night) | | | | | | | | | | | | |
| Accommodation costs for staff (+ 1 driver) per trip | £160 | £120 | £40 | £30 | £240 | £120 | £40 | £20 | £420 | £240 | £140 | £80 |
| Staff costs (£9 each a day) | | | | | | | | | | | | |
| Survey team staff costs: salary and food per trip | £405 | £144 | £405 | £126 | £27 | £27 | £27 | £27 | £1134 | £567 | £1134 | £567 |
| Travel costs | | | | | | | | | | | | |
| Travel costs (petrol) per trip (take to field and return to Yaoundé) | £280 | £40 | £80 | £40 | £280 | £4 | £8 | £4 | £280 | - | - | - |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Community participation gifts and remuneration | | | | | | | | | | | | |
| Participants gifts costs per trip (£0.80 per gift) | £112.8 | | | | £8 | | | | | | | |
| Total community monitor payment per key contact, for collecting and checking daily diaries. £6.50 per trip | - | - | - | - | £6.50 | £78 | £12 | £12 | - | - | - | - |
| Data entry and analysis costs | | | | | | | | | | | | |
| Data entry costs per trip:<br>- per interview (£0.25)<br>- per daily diary participant (c. 8 data sheets each) (£1)<br>camera trap data (£500 for 1-month internship) | £141 | £70.5 | £35.25 | £17.75 | £182.5 | £66.25 | £22 | £11 | £3000 | £1500 | £500 | £250 |
| Equipment | | | | | | | | | | | | |
| Camera trap start-up costs (£180 per camera) | - | - | - | - | - | - | - | - | £10,800 | £5400 | £10,800 | £5400 |
| Photocopying and paper per trip (£0.10 per participant) | £28 | £14 | £7 | £3.50 | £182.5 | £66.25 | £22 | £11 | - | - | - | - |
| Batteries £23.2 per camera (8 batteries per camera, 1 replacement) | | - | - | - | - | - | - | - | £700 | £350 | £175 | £87 |
| TOTAL COSTS | | | | | | | | | | | | |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cost per trip | £894 | £468.5 | £647.25 | £297.25 | £720 | £363.25 | £199 | £153 | 5334 | 2737 | 1854 | 977 |
| Total cost (excluding cameras) | £3576 | £1874 | £647.25 | £297.25 | £8718 | £2257.5 | £410 | £165 | £32,004 | £16,422 | £3883 | £2041 |
| Total cost (including cameras) | | | | | | | | | £42,804 | £21,822 | £14,683 | £7441 |

## 6.3 Results

### Summary of occupancy analysis results

All occupancy models using the interview data converged, while for the diary data, the occupancy model for forest elephant did not converge. Only 7 of the 14 occupancy models for the camera traps converged (see tables 6-4 to 6-6). While occupancy variables were in all the top models using the interview data, few were significant, whereas all occupancy variables were significant in the models using the diary data. The diary and interview data showed that occupancy for rare species such as african golden cat (*Caracal aurata*), gorilla, giant pangolin and chimpanzee reduced with distance from the reserve, although this effect was only significant in the diary data models. Data from interviews and diaries suggests that gorilla occupancy decreased with distance from the village. Consistently higher occupancy in semi-deciduous forests, and increased occupancy with distance from river for some species, suggests that the animals detected may have avoided proximity to rivers and the dense riparian swap habitat adjacent to them during the small rainy season, when this data was collected. For the camera trap data, proximity to the reserve was the only significant detection variable for all species.

Detection variables were mostly significant for both the interview and diary data, suggesting that detection variables were more important than occupancy variables in explaining trends in occupancy using these monitoring methods. Observer age was an important predictor of species detection; younger participants were better able to detect animals from the interview data, while older participants in the daily diaries study were better able to detect animals.

In both interview and diary data, the time participants slept in the forest and the frequency of their visits to the forest were important predictors of their ability to detect animals; the diary data suggest that those who slept in the forest for 15 days or more were more likely to detect animals.

Table 6-4: Back-transformed estimates of occupancy (Ψ) from the top models for each species, using the interview data, with untransformed variable coefficients. Bright red indicates a significant negative effect of the variable on occupancy estimates, while bright green indicates a significant positive effect. Dull green and red indicate that variables were included in the top model, but were not significant. DV=Distance from village, DR= distance from road, DRi= distance from river, DPA= distance from reserve. Intercept: habitat=riparian, age=18-30, gender=female (G_M = gender male), frequency visit (Freq)=more than 3 times a week, time sleep (S_ )=<6 days at a time. Total sites=184

| | | | Occupancy variables | | | | | Detection variables | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Sites with detection | naïve Ψ | DV | DR | DR | HSD | DPA | Age 31-44 | Age 45+ | G_M | Freq 2-3 week | Freq daily | S_ >15 | S_ 11-15 | S_ 6-10 | Ψ | SE | p | SE |
| Blue duiker | 121 | 0.65 | | | - | | + | | | + | + | + | + | + | + | 0.7 | 0.03 | 0.4 | 0.04 |
| Bongo | 25 | 0.13 | + | + | | + | | | | + | | | | | | 0.06 | 0.03 | 0.25 | 0.01 |
| Sitatunga | 95 | 0.51 | - | | | | | - | - | | + | + | | | | 0.59 | 0.04 | 0.27 | 0.001 |
| Yellow backed duiker | 108 | 0.59 | - | | | + | | | | - | | | + | + | + | 0.57 | 0.06 | 0.33 | 0.01 |
| African golden cat | 35 | 0.19 | - | | | + | | | | - | | | | | | 0.08 | 0.03 | 0.15 | 0.01 |
| Genet | 67 | 0.36 | | - | + | + | | | | | - | - | - | - | - | 0.24 | 0.06 | 0.17 | 0.01 |
| Gorilla | 99 | 0.53 | - | + | + | | - | - | + | - | | | | | | 0.71 | 0.04 | 0.33 | 0.007 |
| Chimpanzee | 100 | 0.54 | | | + | | - | - | - | | - | - | + | + | + | 0.62 | 0.04 | 0.35 | 0.01 |
| Putty-nosed monkey | 154 | 0.83 | | - | | | | - | - | + | - | - | + | + | + | 0.91 | 0.02 | 0.54 | 0.01 |
| Giant pangolin | 89 | 0.48 | | | + | | - | - | - | | | | + | + | + | 0.57 | 0.04 | 0.26 | 0.01 |
| Tree pangolin | 110 | 0.59 | | - | | + | | - | - | | - | - | + | + | + | 0.52 | 0.06 | 0.39 | 0.01 |
| Porcupine | 146 | 0.79 | | | | | | - | - | | | | - | + | + | 0.85 | 0.03 | 0.47 | 0.01 |
| Forest elephant | 92 | 0.5 | + | | | | | - | - | + | | | | | | 0.60 | .04 | 0.26 | 0.007 |

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Red river hog | 115 | 0.62 | + | | + | | - | - | | | | + | + | + | 0.54 | 0.06 | 0.32 | 0.01 |

Table 6-5: Back-transformed estimates of occupancy and detection from the top models for each species, informed by the diary data, with untransformed variable coefficients. Bright red indicates a significant negative effect, while bright green indicates a significant positive effect. Dull green and red indicate that variables were included in the model, but not significant. DV=Distance from village, DR= distance from road, DRi= distance from river, DPA= distance from reserve. Intercept: habitat=riparian, age=18-30, gender=female (G_M = gender male), frequency visit (Freq)=more than 3 times a week, time sleep (S_ )=<6 days at a time. Total sites=175

| | | | Occupancy | | | | | Detection probability | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Sites with detection | Naive Ψ | DV | DR | DRi | H_SD | DPA | Age 31-44 | G_M | Freq 2-3 week | Freq daily | S_ >15 day | S_ 11-15 | S_ 6-10 | Ψ | SE | p | SE |
| Blue duiker | 96 | 0.54 | | | | | | + | - | + | + | + | + | - | 0.71 | 0.05 | 0.48 | 0.16 |
| Bongo | 85 | 0.48 | + | | | | | + | + | + | + | + | + | - | 0.72 | 0.08 | 0.44 | 0.16 |
| Sitatunga | 77 | 0.38 | | | - | + | + | | | | | + | + | - | 0.11 | 0.08 | 0.13 | 0.06 |
| Yellow backed duiker | 79 | 0.45 | | + | | | | | | | | | | | 0.45 | 0.06 | 0.39 | 0.04 |
| African golden cat | 67 | 0.12 | | | | | - | | | | | | | | 0.07 | 0.03 | 0.77 | 0.05 |
| Genet | 84 | 0.48 | | | | | | + | + | - | + | + | - | - | 0.71 | 0.08 | 0.49 | 0.18 |
| Gorilla | 88 | 0.50 | - | + | | | | | | | | + | - | + | 0.72 | 0.06 | 0.37 | 0.06 |
| Chimpanzee | 99 | 0.56 | | | | | - | | - | | | | | | 0.79 | 0.04 | 0.84 | 0.07 |
| Putty-nosed monkey | 116 | 0.66 | | | | | | | | - | + | + | - | - | 0.89 | 0.03 | 0.76 | 0.06 |
| Giant pangolin | 91 | 0.52 | | | + | | | | | - | + | + | - | - | 0.74 | 0.05 | 0.49 | 0.11 |
| Tree pangolin | 98 | 0.56 | | | | | | | | - | + | + | - | - | 0.72 | 0.04 | 0.59 | 0.12 |
| Porcupine | 95 | 0.54 | | | | | | | | | | + | - | - | 0.74 | 0.04 | 0.41 | 0.06 |
| Red river hog | 96 | 0.54 | | | | + | | + | | | | + | - | - | 0.51 | 0.09 | 0.24 | 0.07 |

Table 6-6: Occupancy and detection probability estimates for each species based on the top-ranked model(s) using camera trap data. Bright red indicates a significant negative effect, while bright green indicates a significant positive effect. Dull green and red indicate that variables were included in the model, but not significant. DR= distance from road, DRi= distance from river, DPA= distance from reserve. Total sites=26

| Species | Occupancy | | | | Detection probability | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | DR | DR | DPA | Tree cover | Slope | DR | DPA | Ψ | SE | p | SE |
| Genet | | | | | | | + | 0.60 | 0.18 | 0.05 | 0.02 |
| Tree pangolin | - | | | | | | | 0.52 | 0.61 | 0.03 | 0.02 |
| Blue duiker | + | | | | | | - | 0.92 | 0.07 | 0.40 | 0.03 |
| Yellow-backed duiker | | | | | | | - | 0.66 | 0.28 | 0.02 | 0.01 |
| Red river hog | | | | | | + | | 0.58 | 0.14 | 0.10 | 0.03 |
| Chimpanzee | - | | | + | | | | 0.72 | 0.40 | 0.04 | 0.02 |
| Brush tailed porcupine | - | | - | | | - | + | 0.88 | 0.08 | 0.35 | 0.03 |

### 6.3.1 Power of the survey design employed in this study to detect trends

The interview survey design used in this study was able to detect growth and decline of 30-80% depending on the species, but only species with a probability of occupancy or detection > 0.25 had 80% power to detect some level of change (table 6-7). The current camera trap plan was also able to detect between 30-80% growth, but unlike the interview data all 8-species detected by the cameras had 80% power to detect some level of change, regardless of their probability of occurrence and detection. However, only 80% declines could be detected for yellow backed duiker and red river hog, both of which had a very small likelihood of detection ($p=<0.03$). Eight of the 10 species observed in the diary data had a power to detect changes of 80-30%. Diary data allowed the greatest power to detect change in primate and pangolin, while there was little power to detect change in ungulate species occupancy.

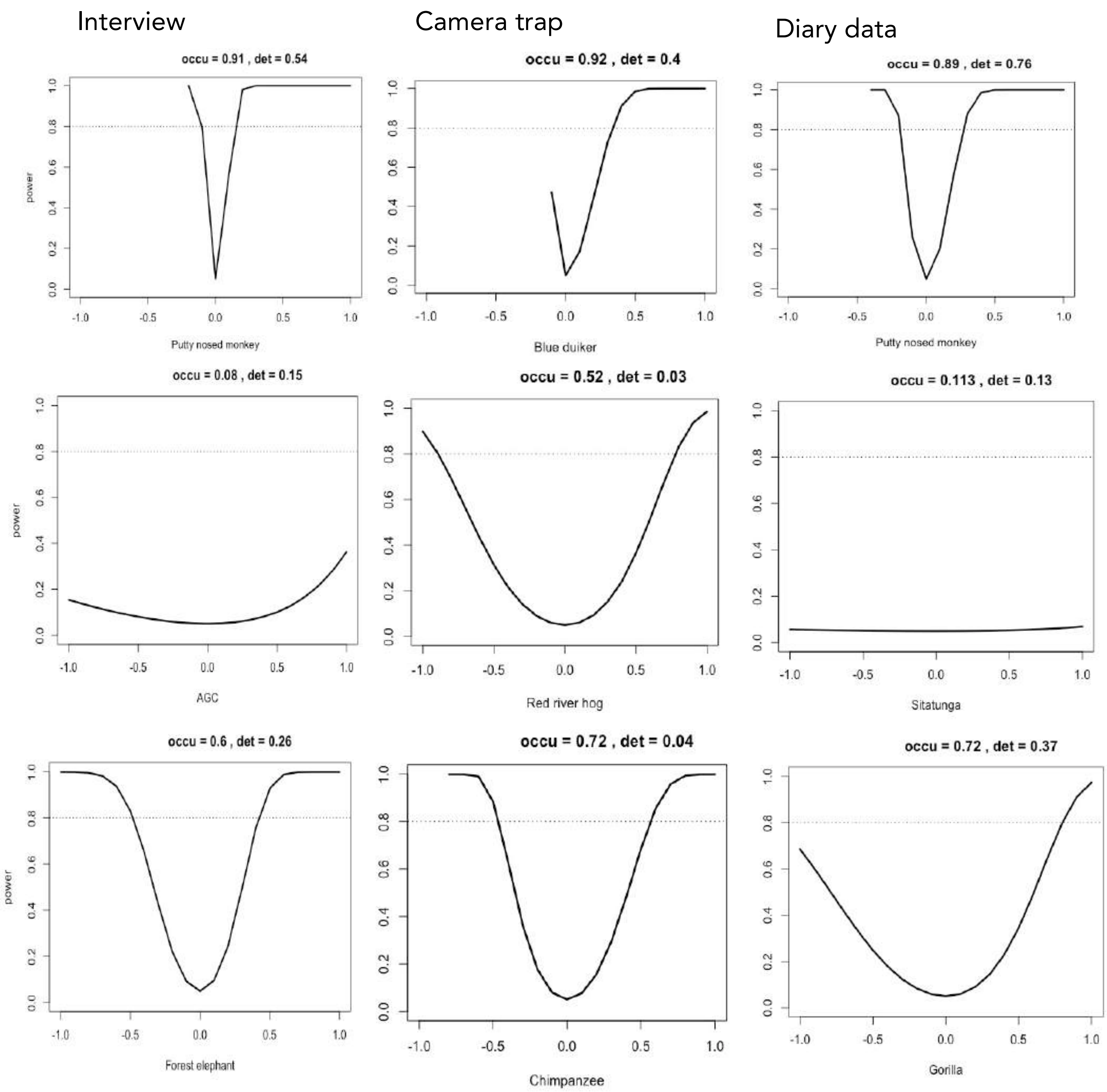
Table 6-7. Power to detect growth and decline in species occupancy between two seasons under the current monitoring plan for interviews, daily diaries and camera trapping. Grey cells indicate power > 80%. NA = not applicable because the rate of growth or decline is not possible, given the estimated probability of occupancy. IUCN Red List categories: LC = least concern, NT = near threatened, VU = vulnerable, EN = endangered, CR = critically endangered.

| Species | IUCN | Ψ | p | 80% decline | 50% decline | 30% decline | 10% decline | 10% growth | 30% growth | 50% growth | 80% growth |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Interview | | | | | | | | | | | |
| Blue duiker | LC | 0.7 | 0.4 | 1 | 1 | 0.99 | 0.19 | 0.19 | 0.99 | 1 | 1 |
| Bongo | NT | 0.06 | 0.25 | 0.49 | 0.28 | 0.15 | 0.05 | 0.05 | 0.11 | 0.16 | 0.22 |
| Sitatunga | LC | 0.59 | 0.27 | 0.99 | 0.99 | 0.94 | 0.09 | 0.09 | 0.85 | 0.98 | 0.99 |
| Yellow duiker | LC | 0.57 | 0.33 | 1 | 0.99 | 0.98 | 0.12 | 0.11 | 0.96 | 0.99 | 0.99 |
| African golden cat | VU | 0.08 | 0.15 | 0.28 | 0.16 | 0.10 | 0.05 | 0.05 | 0.08 | 0.10 | 0.14 |
| Giant pangolin | VU | 0.57 | 0.26 | 0.99 | 0.99 | 0.91 | 0.09 | 0.08 | 0.08 | 0.97 | 0.99 |
| Tree pangolin | VU | 0.52 | 0.39 | 1 | 0.99 | 0.99 | 0.13 | 0.12 | 0.98 | 0.99 | 1 |
| Gorilla | CR | 0.71 | 0.33 | 1 | 0.99 | 0.99 | 0.15 | 0.15 | 0.99 | 0.99 | 1 |
| Chimpanzee | EN | 0.62 | 0.35 | 1 | 0.99 | 0.99 | 0.14 | 0.14 | 0.99 | 0.99 | 1 |
| Putty-nosed monkey | LC | 0.91 | 0.54 | 1 | 1 | 1 | 0.57 | 0.79 | NA | NA | NA |
| Genet | LC | 0.24 | 0.17 | 0.78 | 0.50 | 0.35 | 0.05 | 0.05 | 0.17 | 0.28 | 0.41 |
| Porcupine | LC | 0.85 | 0.47 | 1 | 1 | 1 | 0.36 | 0.43 | 1 | NA | NA |
| Forest elephant | VU | 0.60 | 0.26 | 0.99 | 0.99 | 0.92 | 0.09 | 0.09 | 0.83 | 0.98 | 0.99 |
| Red river hog | LC | 0.54 | 0.32 | 1 | 0.99 | 0.97 | 0.11 | 0.11 | 0.92 | 0.99 | 0.99 |
| Camera traps | | | | | | | | | | | |
| Blue duiker | LC | 0.92 | 0.40 | 1 | 0.99 | 0.98 | 0.17 | 0.47 | NA | NA | NA |
| Yellow duiker | LC | 0.66 | 0.02 | 0.88 | 0.59 | 0.31 | 0.05 | 0.05 | 0.25 | 0.44 | 0.67 |
| Tree pangolin | VU | 0.88 | 0.35 | 1 | 0.99 | 0.96 | 0.14 | 0.23 | NA | NA | NA |

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Chimpanzee | EN | 0.72 | 0.04 | 0.99 | 0.96 | 0.68 | 0.07 | 0.07 | 0.88 | 0.99 | NA |
| Genet | LC | 0.60 | 0.05 | 0.99 | 0.90 | 0.57 | 0.06 | 0.07 | 0.66 | 0.96 | 0.99 |
| Red river hog | LC | 0.52 | 0.03 | 0.93 | 0.68 | 0.36 | 0.06 | 0.05 | 0.31 | 0.56 | 0.80 |
| Porcupine | LC | 0.88 | 0.35 | 1 | 0.99 | 0.96 | 0.14 | 0.22 | NA | NA | NA |
| Daily diaries | | | | | | | | | | | |
| Blue duiker | LC | 0.71 | 0.48 | 0.99 | 0.92 | 0.60 | 0.07 | 0.07 | 0.49 | 0.78 | 0.94 |
| Sitatunga | LC | 0.113 | 0.13 | 0.06 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 |
| Yellow duiker | LC | 0.45 | 0.39 | 0.72 | 0.48 | 0.25 | 0.05 | 0.05 | 0.17 | 0.28 | 0.42 |
| African golden cat | VU | 0.07 | 0.77 | 0.18 | 0.12 | 0.08 | 0.05 | 0.05 | 0.07 | 0.10 | 0.13 |
| Giant pangolin | VU | 0.74 | 0.49 | 0.99 | 0.95 | 0.66 | 0.07 | 0.07 | 0.55 | 0.84 | 0.97 |
| Tree pangolin | VU | 0.72 | 0.59 | 0.99 | 0.99 | 0.85 | 0.08 | 0.08 | 0.84 | 0.99 | 0.99 |
| Gorilla | CR | 0.72 | 0.37 | 0.91 | 0.65 | 0.34 | 0.06 | 0.06 | 0.24 | 0.41 | 0.59 |
| Chimpanzee | EN | 0.79 | 0.84 | 1 | 0.99 | 0.99 | 0.16 | 0.19 | 1 | NA | NA |
| Putty-nosed monkey | LC | 0.89 | 0.76 | 1 | 1 | 0.99 | 0.20 | 0.26 | NA | NA | NA |
| Genet | LC | 0.71 | 0.49 | 0.99 | 0.93 | 0.63 | 0.07 | 0.06 | 0.52 | 0.81 | 0.96 |
| Porcupine | LC | 0.74 | 0.41 | 0.97 | 0.78 | 0.45 | 0.06 | 0.06 | 0.33 | 0.56 | 0.77 |
| Red river hog | LC | 0.51 | 0.24 | 0.33 | 0.19 | 0.11 | 0.05 | 0.05 | 0.08 | 0.12 | 0.16 |

The ability to detect change with 80% power depends on occupancy across all methods, although the level of occupancy required to achieve 80% power differs across methods (figure 6-1). For interview data, species with $\psi$ =<0.08 and p=<0.25 had no power to detect proportional change in occupancy. All animals detected by camera traps had $\psi$ =>0.52, and therefore achieved 80% power to detect some proportion of change, even when p=<0.02 (e.g. yellow-backed duiker). With diary data, species with $\psi$ =<0.51 did not have the power to detect change, even where detection was high (e.g. african golden cat detected in daily diaries, $\psi$ =<0.07, p=0.77).

The level of change that can be detected with 80% was influenced by the species occupancy and detection and again the occupancy or detection required to detect a proportional change varied with each method. To detect a proportional growth or decline of r=0.25, animals detected by interview data required $\psi$ =>0.7 as achieved for blue duiker, gorilla, putty-nosed monkey and porcupine; diary data required $\psi$ =>0.79 as achieved by chimpanzee and putty-nosed monkey; and camera trap data required $\psi$ =>0 92, only achieved by blue duiker. For interview and camera data, detection must be p=>0.4 to detect growth or decline of r=0.25 while for diary data, p must be =>0.84, higher than the other two methods.



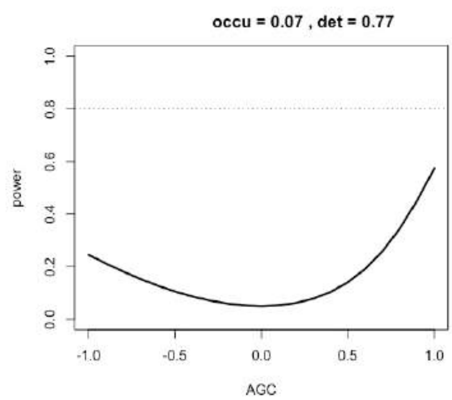




Figure 6-1: Power curves (where α = 0.05) and the relative proportion of change in occupancy (R)[1] that can be detected between two sampling periods for species with a range of high and low probabilities of occupancy and detection. R < 0 is a decline, R > 0 is growth. Left=interview data: high occupancy and detection = putty-nosed monkey; high occupancy, low detection = forest elephant; low occupancy and detection = African golden cat (AGC). Middle=camera traps: high occupancy and detection= blue duiker. High occupancy and low detection= red river hog and chimpanzee. Right= diary: high occupancy and detection = putty-nosed monkey; Low occupancy and detection=sitatunga. High occupancy low detection=gorilla. Low occupancy high detection=african golden cat. See appendix E-2 for power

### 6.3.2 Statistical power of survey designs with varying numbers of repeat visits and sites

There was no power to detect change in two of the 14 species using the interview data. Tripling the number of sites visited slightly improved the power to detect large proportional declines of r=>0.5, but did not greatly improve the power to detect growth. However, doubling or tripling the number of repeat visits did allow the power to detect large proportional changes of r=1 for bongo and african golden cat, both with $\psi$ =<0.08. Yet, detecting a proportional decline of this size may not be much help to conservation managers interested in monitoring these species of conservation interest. Halving the repeat visits resulted in a loss of power where $\psi$ =<0.24 and reduced the ability to detect change where $\psi$ =>0.24 (figure 6-2). Increasing the number of sites surveyed improved the power to detect smaller levels of change where there was already 80% power to detect change.

Camera trap data were less able to detect small proportional changes than interview data; only for blue duiker, porcupine and tree pangolin could growth of r=<0.25 be detected if the current number of sites were doubled or tripled. For no species could proportional declines of r=<0.25 be detected, despite doubling or tripling survey efforts. Halving the number of repeat visits resulted in a loss of power for red river hog and yellow backed duiker, both of which had low detection rates (p=<0.03) (figure 6-3) and a large reduction of power to detect smaller levels of change for all species. However, halving the number of sites did not have such a strong effect on the camera trap data. The diary data had no power to detect change for four of the 12 species, even when the number of sites were doubled or tripled. Doubling or tripling the number of repeat visits per site improved power for yellow backed duiker ($\psi$ =0.45) and red river hog ($\psi$ =0.54). Furthermore, doubling or tripling the number of repeat visits gave power to detect r=<0.25 for porcupine, genet, blue duiker, gorilla, bongo, tree pangolin, and giant pangolin whereas before these had power to detect proportional change of only r=>0.5. Where occupancy is very low, such as African golden cat ($\psi$ =0.07) and sitatunga ($\psi$ =0.11), power was relatively unaffected by changes in survey effort. Halving the total number of repeat surveys resulted in loss of power for all (figure 6-4).

In summary, 80% power to detect change was rarely achieved where occupancy or detection were already very low. Where occupancy was high and detection low, increasing the number of repeat visits substantially increased the ability to reach 80% power. Examples of this effect are especially prominent in the diary data (figure 4). Increasing survey effort in this way may be worthwhile where the species is of conservation interest, or for animals hunted for wild meat that may be important for local people. Where 80% power to detect change is already achievable, increasing the number of repeat visits allowed a lower level of proportional growth or decline to be detected. This may be of use in particular where the animals are of conservation interest or highly sensitive to change, meaning that monitoring that can capture smaller changes in population may be worthwhile. As the median number of repeat visits per site for diary data was low (n=2), doubling or tripling the number of visits greatly improved power compared to the current monitoring plan in many cases (figure 6-4). For interview cases where $\psi$ => 0.52, halving the number of repeat visits from 4 to 2 was possible, but resulted in less ability to detect smaller levels of change. Doubling or tripling the number of repeat visits to 8 or 12 was especially effective where $\psi$= <0.24 (figure 6-2). For cameras, halving the repeat visits from 12 to 6 reduced the power below 80% in many cases where p=<0.03 (Figure 6-3).

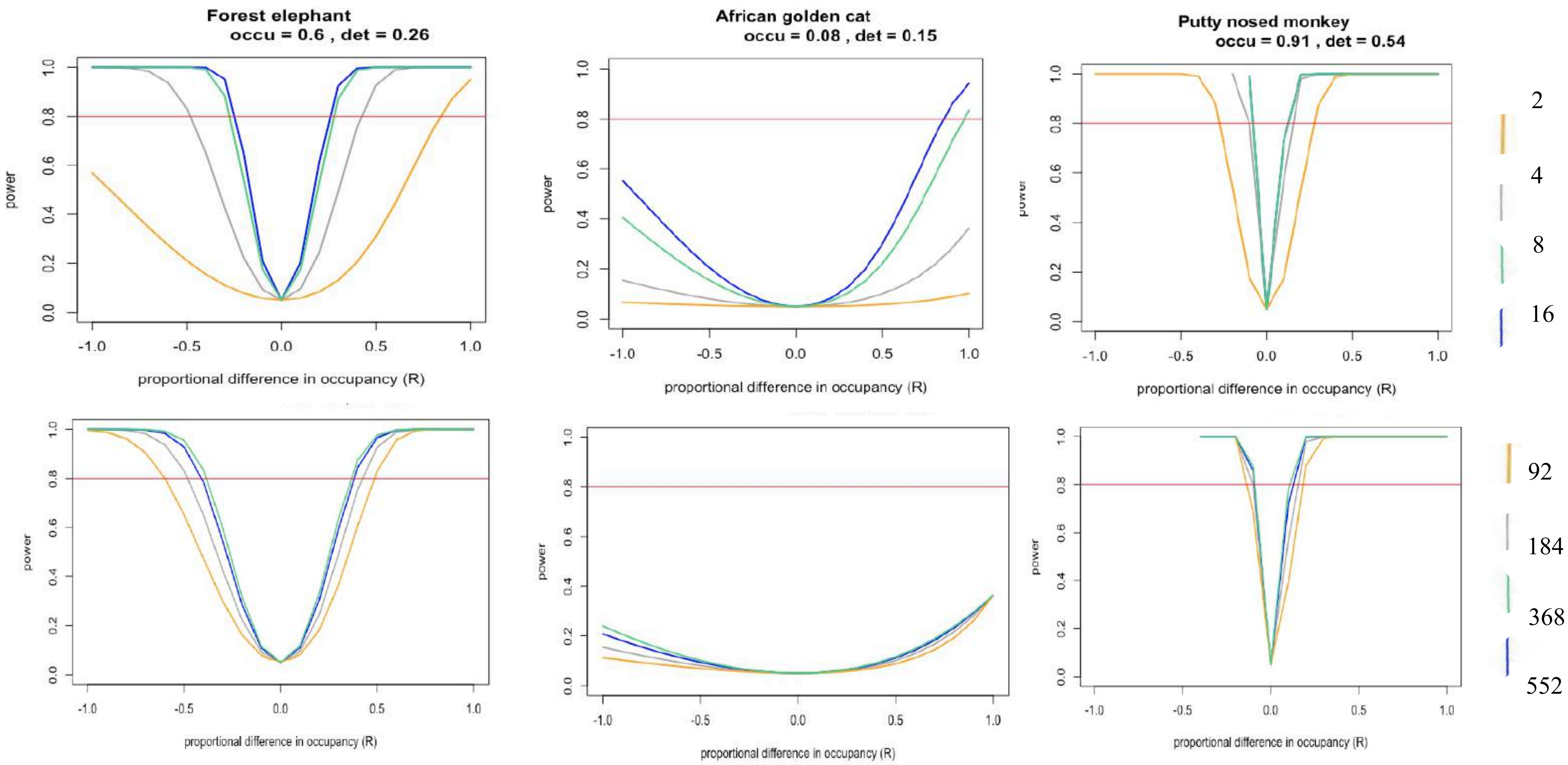



Figure 6-2: Power curves for the interview data, showing the relative proportion of change in occupancy that can be detected (R, where R > 0 is a decline, R < 0 is growth) for various numbers of repeat visits per site (K) (top row) and total number of sites (S, bottom row) for forest elephant, african golden cat and putty-nosed monkey. See appendix E-3 for power curves for all species captured by interview data. Grey lines indicate current strategy where S = 184 and repeat interviews (I) is= 141. Red dotted line is a power of 0.8 to detect change between two periods.

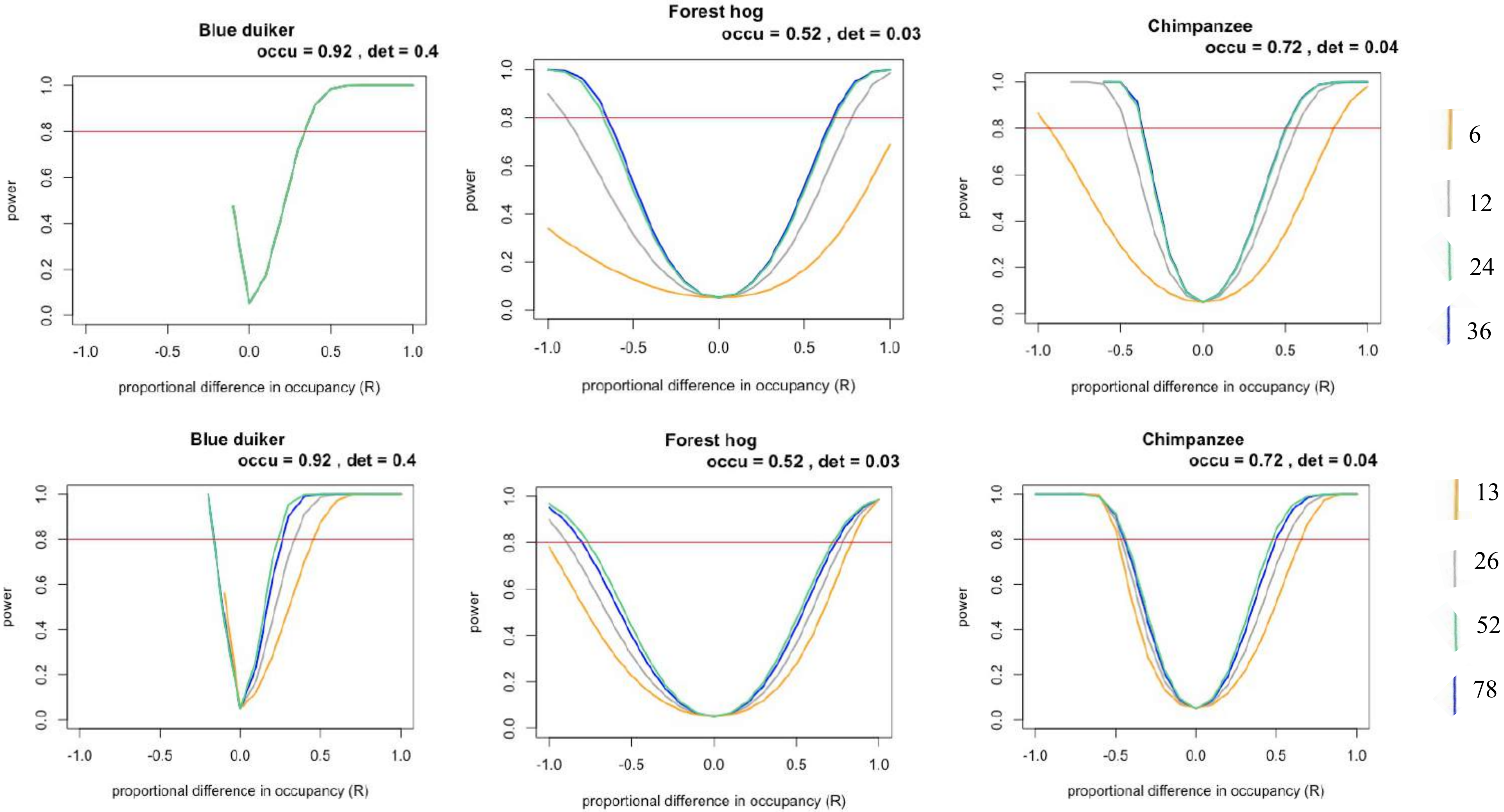



Figure 6-3: Power curves for the camera data, showing the relative proportion of change in occupancy that can be detected (R, where R > 0 is a decline, R < 0 is growth) for various number of repeat visits per site (K) (top row) and total number of sites (bottom row) for blue duiker, red river hog and chimpanzee. See appendix E-5 for power curves for all species captured by camera trap data. Grey lines indicate current strategy where S = 26 and K= 60. Red dotted line is a power of 0.8 to detect change between two periods.

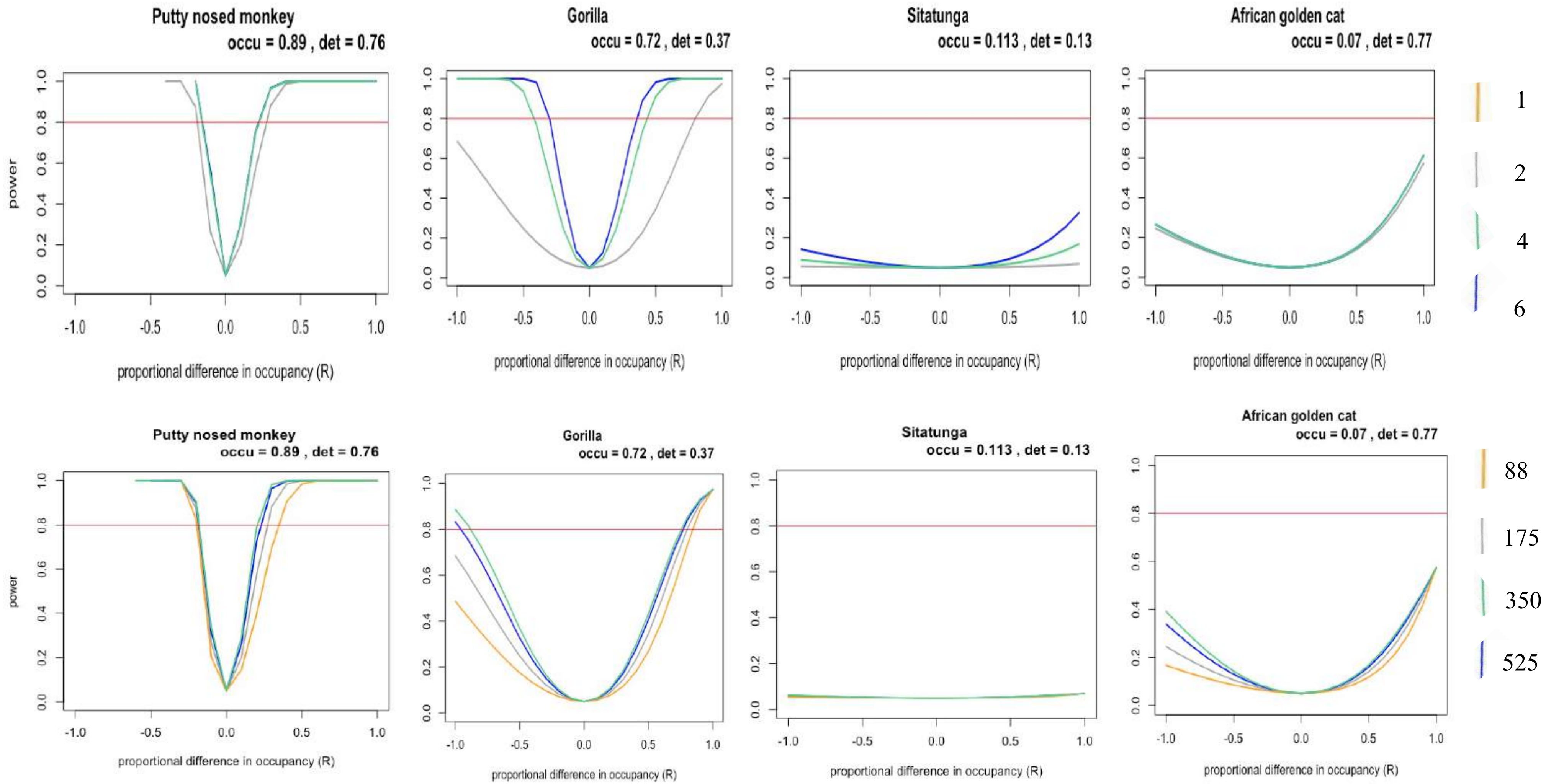



Figure 6-4: Power curves for diary data, showing the relative proportion of change in occupancy that can be detected (R, where R > 0 is a decline, R < 0 is growth) for various number of repeat visits per site (K) (top row) and total number of sites (bottom row) for blue duiker, red river hog and chimpanzee. See appendix E-4 for power curves for all species captured by diary data. Grey lines indicate current strategy where S = 26 and I = 10. Red dotted line is a power of 0.8 to detect change between two periods.

### 6.3.3 Identifying the most efficient survey effort

The survey effort required to detect 10% change was prohibitive for interview data, unless $\psi$ =>0.85. Overall, it was easier to detect decline than growth, a difference which was more pronounced when $\psi$ =<0.24 and p=<0.25. Decreasing detectable change from 80% to 50% was possible in many cases with not much additional effort. However, much more effort was required to detect 30% change, especially where p = <0.4. A minimum of 3-4 repeat visits per site were required for interview data to reduce the number of sites required for survey by about half (figure 6-5).

As with the interview data, it was easier to detect decline than growth in the diary data, a difference that was more pronounced when $\psi$ =<0.1. Again, the sites required for survey reduced by about half after 4 repeat visits, as did the gap in effort required to decrease detectable change from 80%-30%. However, where $\psi$ =<0.51 and p= <0.39, greater effort was required to achieve 30% change. A large difference in effort was required to detect 10% change. However, this effort may often be prohibitive; 500-600 sites with 8 repeat surveys were required for all except for putty-nosed monkey, with $\psi$ =0.89, which required a lower survey effort of 100 sites and 4 repeat surveys.

Species with $\psi$= >0.88 detected by the camera trap data required substantially fewer repeat visits to detect a given change than those with a lower occupancy; 10 rather than 40 repeat visits across the same number of sites. Furthermore, detecting 30% change could be achieved with the same effort that is required to detect 80% change if $\psi$ =>0.88. Overall, the minimum survey effort required when $\psi$ =<0.88 was fewer than 100 sites, with 40 repeat visits. The minimum survey effort required to detect 30% change where $\psi$ =>0.88 was fewer than 10 repeat visits and fewer than 40 sites. The minimum survey effort required to detect 80-50% change where $\psi$ =<0.88 was 40-60 repeat visits and fewer than 100 sites, but detecting 30% change required at least 40-60 repeat visits to reduce the total sites required to 200.

In summary, a similar effort is required to detect 50% as 80% change for all methods, except where $\psi = < 0.13$ (diary data), $\psi = < 0.03$ (camera) or $\psi = < 0.6$ (interviews). In most cases, the power to detect growth and decline at each level of change was comparable after 5 repeat visits, other than for 10% change which required significantly more survey effort. Overall, where $\psi = > 0.54$, 200 sites and 4 repeat visits were required to detect at least a 30% change. However, in my current survey design I had camera trap data from only 26 sites, meaning that achieving the necessary survey effort using cameras could be a challenge if faced with financial and logistical limitations similar to those I faced when conducting this study. Surveying across 200 sites is more feasible using observational methods, as I had data from 184 sites using interviews, and 175 using hunter diary data.

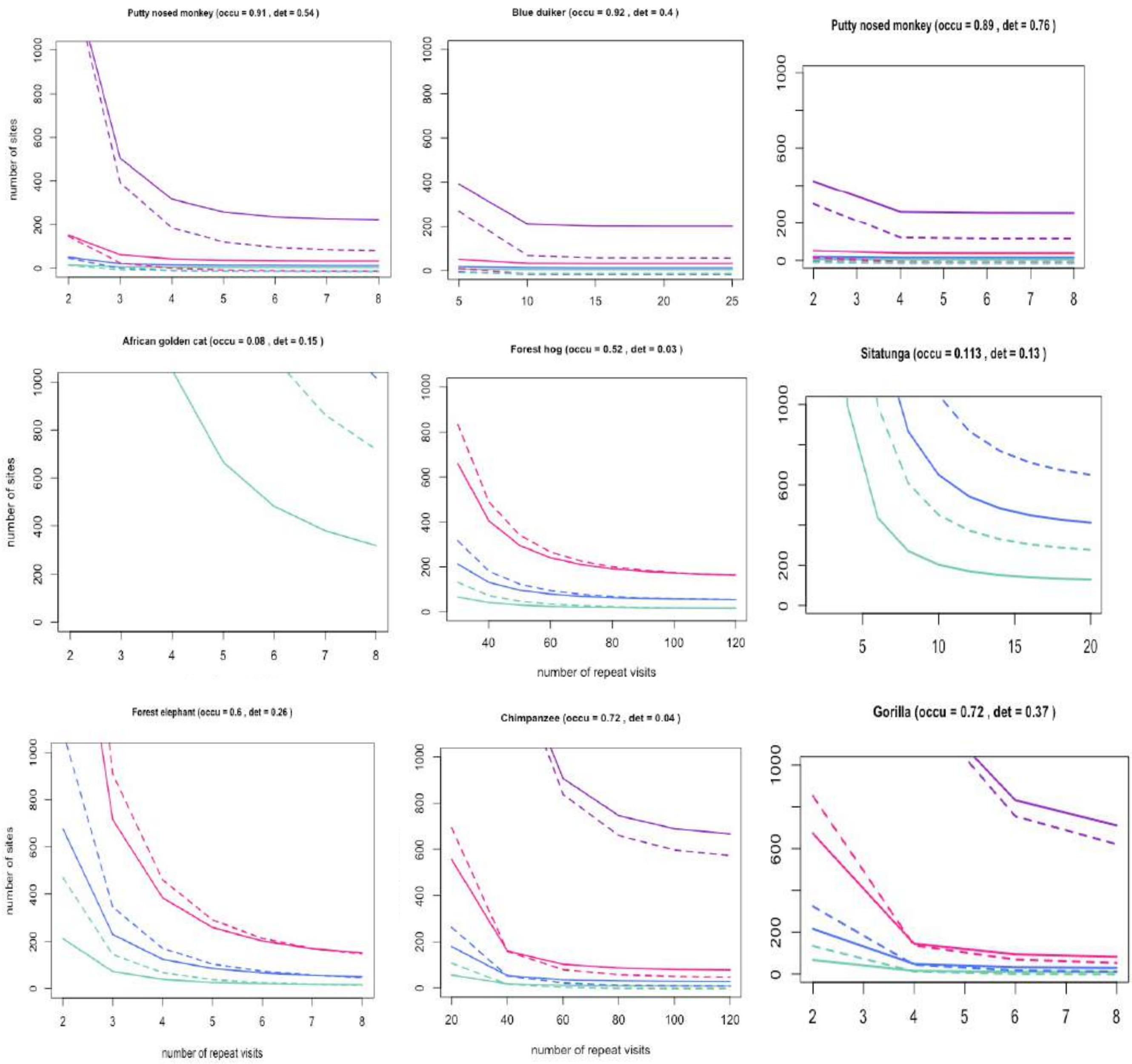
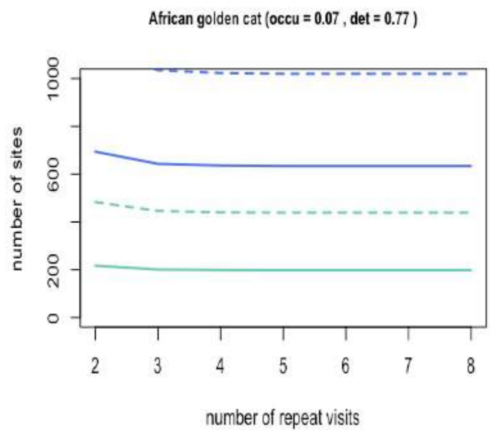



Figure 6-5: The number of repeat visits and total sites required to achieve 80% power to detect growth (dashed line) and decline (solid line) of 80% (green), 50% (blue), 30% (pink) and 10% (purple) for the given number of sites and repeat visits in two seasons. Left column= interviews, middle column= camera traps, right column-daily diaries. See appendix E-6- E-8 for all species power curves from the three different methods.



For species with $\psi$ = >0.85, it may be possible to detect 10% change with 100 sites and at least 4 (observational methods) or 10 (camera) repeat visits. The costs of a three year monitoring project that would allow for 50% growth and decline in occupancy to be detected at the village level with 80% power are shown in table 6-8. Interviews allowed for critically endangered gorilla to be monitored at a cost of <£500,000 using the second most intensive monitoring strategy. Less intensive scenarios C-D for interviews allow for 50% change in gorilla occupancy over 3 years to be detected with a budget of <£250,000. All scenarios using diary data allow for monitoring gorillas with a budget of £500,000 or less. However, gorillas were not detected often enough for monitoring with cameras to be possible. It is possible to monitor several different species at once, for the price of the most expensive species for a given method. For example, alongside gorillas, diaries would allow us also to effectively monitor giant pangolin, chimpanzee and tree pangolin, as well as blue duiker, genet and porcupine.

All scenarios using daily diaries allow the endangered chimpanzee to be monitored with a budget of less than £100,000, while interview scenarios B-D allow monitoring for under £750,000. No camera trap scenarios allow detection of 50% change in chimpanzee occupancy for under £750,000. No camera trap scenarios allow pangolins to be monitored for less than £750,000. In contrast, the cost of monitoring pangolins with diary data is under £250,000 in all scenarios. Scenario B allows for monitoring under £750,00, while the least intensive scenario D allows for monitoring under £500,000.

Diary data provides the cheapest monitoring method overall, especially under scenarios B-D. Interviews and diary data allow for more species of conservation interest to be monitored more intensively, for less money, compared to camera trap data. In particular, growth and declines of 50% for gorilla, chimpanzee and pangolin are detectable for under £750,000 using observational methods, while the same animals are either not detected, or would be prohibitively expensive to monitor sufficiently to detect 50% changes in occupancy, using camera traps.

Table 6-8: Cost of a three-year monitoring project in two villages that allows for 50% growth or decline in occupancy to be detected with 80% power. Dark grey indicates < £250,000, mid grey indicates £251,000-£500,000 and light grey indicates £500,000-£750,000 cost of monitoring over 3 years. Total costs exclude the one-off start-up costs of buying camera traps. Costs are expressed to the nearest thousand (£K)

| Interview | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Species | IUCN | Scenario A ££ (Year-round, high intensity) | | Scenario B ££ (Year round, low intensity) | | Scenario C ££ (Seasonal, high intensity) | | Scenario D ££ (Seasonal, low intensity) | |
| | | 50% decline | 50% growth | 50% decline | 50% growth | 50% decline | 50% growth | 50% decline | 50% growth |
| African golden cat | VU | 647,083,000 | 107,817,000 | 602,256,750 | 100,365,630 | 832,039,875 | 138,658,814 | 215,152,000 | 358,486,000 |
| Blue duiker | LC | 579,000 | 676,000 | 304,000 | 354,000 | 419,000 | 489,000 | 255,000 | 358,000 |
| Chimpanzee | EN | 893,000 | 1,191,000 | 468,000 | 624,000 | 647,000 | 862,000 | 297,000 | 396,9000 |
| Genet | LC | 13,010,000 | 21,386,000 | 6,818,000 | 11,207,000 | 9,419,000 | 15,484,000 | 4,326,000 | 7,111,000 |
| Giant pangolin | VU | 1,963,000 | 2,969,000 | 1,029,000 | 1,556,000 | 1,421,000 | 2,150,000 | 6,523,000 | 987,000 |
| Gorilla | CR | 853,000 | 1,118,000 | 447,000 | 586,000 | 617,000 | 810,000 | 284,000 | 372,000 |
| Red river hog | LC | 1,303,000 | 1,875,000 | 683,000 | 982,000 | 944,000 | 1,357,000 | 433,000 | 623,000 |
| Pangolin | VU | 909,000 | 1,215,000 | 476,000 | 637,000 | 658,000 | 880,000 | 302,000 | 404,000 |
| Porcupine | LC | 281,000 | 169,000 | 148,000 | 89,000 | 204,000 | 122,000 | 94,000 | 56,000 |
| Putty-nosed monkey | LC | 177,000 | 252,000 | 93,000 | 132,000 | 128,000 | 183,000 | 59,000 | 84,000 |
| Sitatunga | LC | 1,722,000 | 2,575,000 | 902,000 | 1,349,000 | 1,247,000 | 1,864,000 | 572,000 | 856,000 |
| Yellow backed duiker | LC | 1,134,000 | 1,601,000 | 595,000 | 839,000 | 821,000 | 1,159,000 | 377,000 | 532,000 |
| Diary | | | | | | | | | |
| African golden cat | VU | 5,556,000 | 8,942,000 | 2,803,000 | 4,512,000 | 1,535,000 | 2,472,000 | 1,181,000 | 1,900,000 |
| Blue duiker | LC | 406,000 | 363,000 | 205,000 | 183,000 | 112,000 | 100,000 | 86,000 | 77,000 |
| Chimpanzee | EN | 52,000 | 69,000 | 91,539 | 8718 | 50,000 | 5000 | 39,000 | 3400 |
| Genet | LC | 293,000 | 173,000 | 148,000 | 87,000 | 81,000 | 48,000 | 62,000 | 37,000 |
| Giant pangolin | VU | 268,000 | 130,000 | 135,000 | 65,000 | 74,000 | 36,000 | 57,000 | 28,000 |
| Gorilla | CR | 415,000 | 380,000 | 209,000 | 192,000 | 115,000 | 105,000 | 88,000 | 81,000 |
| Red river hog | LC | 1,546,000 | 2,264,000 | 780,000 | 1,142,000 | 427,000 | 626,000 | 329,000 | 481,000 |
| Pangolin | VU | 251,000 | 95,000 | 126,000 | 48,000 | 69,000 | 26,000 | 53,000 | 20,000 |
| Porcupine | LC | 337,000 | 242,000 | 170,000 | 122,000 | 93,000 | 67,000 | 72,000 | 51,000 |
| Putty-nosed monkey | LC | 121,000 | 112,000 | 61,000 | 57,000 | 33,000 | 31,00 | 26,000 | 234,000 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Sitatunga | LC | 27,631,0020 | 45,723,000 | 13,940,000 | 23,068,000 | 7,637,00 | 12,637,00 | 5,872,000 | 9,716,000 |
| Yellow backed duiker | LC | 812,000 | 1,037,000 | 410,000 | 523,000 | 224,000 | 287,000 | 173,000 | 220,000 |
| Camera trap | | | | | | | | | |
| Blue duiker | LC | 88,000 | 107,000 | 88,000 | 107,000 | 120,000 | 40,000 | 88,000 | 107,000 |
| Chimpanzee | EN | 34,924,000 | 33,284,000 | 17,407,000 | 17,079,000 | 11,791,000 | 11,569,000 | 6,214,000 | 6,096,000 |
| Genet | LC | 36,485,000 | 36,485,000 | 18,721,000 | 18,721,000 | 12,681,000 | 12,681,000 | 6,683,000 | 6,683,000 |
| Red river hog | LC | 83,850,000 | 115,855,000 | 43,0256,000 | 59,448,000 | 29,145,000 | 40,269,000 | 15,358,000 | 21,220,000 |
| Pangolin | VU | 81,930,000 | 112,654,000 | 42,040,000 | 57,805,000 | 28,477,000 | 39,156,000 | 15,007,000 | 20,634,000 |
| Porcupine | LC | 107,000 | 73,000 | 107,000 | 73,000 | 107,000 | 73,000 | 107,000 | 73,000 |
| Yellow backed duiker | LC | 128,016,000 | 189,464,000 | 65,688,000 | 97,218,000 | 44,496,000 | 65,854,000 | 23,448,000 | 34,703,000 |

## 6.4 Discussion

Growing threats to biodiversity in the tropics mean there is an increasing need for effective monitoring that balances scientific rigor with practical feasibility (Rist et al. 2010). To achieve this balance, it is important to understand whether the monitoring goal is achievable using the survey design selected to avoid wasting valuable conservation resources. Conservation practitioners may have more or less power to detect trends in occupancy depending on the underlying occupancy of the species, the monitoring method used, the intensity of the sampling strategy (which is budget-dependent), and the species detectability (which depends both on species characteristics, the method and observer characteristics). Therefore, knowing what survey method and survey design will provide the greatest power in the most efficient way, prior to starting any monitoring programme, is vital to ensure that monitoring is not a waste of time (Robinson et al. 2018) and that monitoring goals are achieved in the most efficient way possible.

This study is the first to apply the formula developed by Guillera-Arroita & Lahoz-Monfort (2012), which accounts for imperfect detection, to occupancy models informed by interviews with local people. Furthermore, this is the first study that compares the power of locally informed methods with camera traps, and to identify the monitoring strategies that are best suited to different species, to ensure monitoring is both effective and efficient.

While bearing in mind that this study was conducted in just one village, the occupancy models identified some interesting variables that affect species occupancy, mirroring findings in the literature. Occupancy analysis from interviews and diaries suggested that gorilla occupancy decreases with distance from the village, possible due to the presence of several large fruiting trees in proximity to the villages. Other studies of occupancy analysis informed by interviews with local people in central Africa have also found gorillas to be the only species in their study whose occupancy is not affected by proximity to human disturbance (e.g. Martínez-Martí et al. 2016). Overall, variables that account for occupancy were less significant for interview-based models than for diary data models. It may be that people from the interview data are spread out across the landscape, so that species are detected more uniformly, whereas hunters completing the diary data stick to more distinct areas where they can hunt successfully, which is then reflected in the occupancy data from the diaries.

Age, gender, the number of trips to the forest and the number of nights spent sleeping in the forest are all important detection variables included in the top models for the diary data, while the number of nights slept in the forest was insignificant in the top models for the interview data. The interview data showed that young responders were more likely to detect species than older participants, possibly reflecting greater forest activity by younger participants in the village-wide interviews. In contrast, older participants in the hunter diary study were better able to detect species than younger participants, which may reflect the role that experience has in a hunter's ability to hunt using different methods, and detect different species. For example, Kumpel et al. (2009) found that younger hunters put in more effort and had a greater trapping success, while in contrast Walker et al. (2002) found hunter success to peak at an older age. However, a more substantial study would be required to verify these trends in other villages.

Under the monitoring plan used in this study village, all methods had sufficient power to detect 50% change for between 58-78% of the species captured by that monitoring method. Yet, where occupancy or detection was already very low, 80% power to detect change was rarely achieved, despite increasing survey effort. For example, African golden cat was too elusive for any change to be detected with 80% power, regardless of increased monitoring effort, which further highlights the need for power analyses to be prioritised when developing wildlife population monitoring programmes (Guillera-Arroita et al 2012; Southwell et al. 2018). For example, as part of an ongoing camera trap monitoring programme, Bruce et al. (2018) reported the first documentation of African golden cat within the Dja Faunal Reserve. Their occupancy analysis showed African golden cat to have an occupancy of 0.41 and a low detection of 0.13. As such, if repeated, this study may not have sufficient power to detect change in african golden cat occupancy. Programmes that monitor rare or elusive species such as these should conduct power analyses to identify whether or not it is possible to have the power to detect change with the financial and logistical resources available to them.

An interesting result is that a similar effort is needed to detect 80% and 50% changes in populations. However, smaller levels of change are only detectable above a certain occupancy threshold, which again differs with method and species. Conservation practitioners need to weigh up the costs and benefits of attempting to detect small proportional changes for their target species, given the high cost and effort which may be required (Southwell et al 2018). However, we must also bear in mind that multiple species can be monitored for the cost of the most expensive species to monitor. As such, if several species are the target of a monitoring programme, the costs to monitor per species are divided by the cost of the most expensive species.

The results from this village identified species-level differences in the power to detect change across the different methods. It's interesting to note that, for primates and pangolins, diary data had the greatest power to detect change. Primates and pangolins are heavily hunted in this region. Since diary respondents in this village were active gun or snare hunters, these results may reflect more reliable knowledge of the animals that participants of the hunter diaries are actively targeting (Martinez-Marti et al 2016). It may also be that hunters completing the diaries are actively visiting sites where giant pangolins are more likely to be found, therefore increasing their chances of detecting them.

Occupancy and detection dictate the optimal strategy required to increase power. Where occupancy and detection were high, increasing the number of repeat visits per site had a greater effect on the power to detect change than increasing the total number of sites surveyed. The results are in line with the recommendations made by Mackenzie & Royle (2005), that it is more efficient to survey more sampling units less intensively when targeting rare species or where detection is low while for common species, fewer sampling units should be surveyed more intensively. In their study of the challenges of monitoring biodiversity for ecosystem services in Madagascar, Sommerville et al (2011) found that only the most common species had sufficient power on which to base a 'payments for environmental services' scheme.

A minimum of 3 repeat visits is usually required for robust occupancy analysis (Mackenzie & Royle 2005) which was not always achieved from the diary data (median= 3). The significant improvement in power when repeat visits were doubled or tripled (figure 6-4 and appendix E-4) supports that more survey effort would be required to ensure improved power to detect change using this approach, if monitoring were to continue. Something to consider for future research is that monitoring with camera traps uniquely allows us to increase detectability at the expense of the number of repeat visits, by lengthening the occasion length. The impact of different occasion lengths on detection and the resulting power to detect change is worth further exploration.

I found significant differences in the financial investment required for monitoring with each method, regardless of the scenario implemented. While camera traps performed well for abundant ungulates and rodents, the cost to implement effective monitoring in this village to detect 50% growths or declines in occupancy was prohibitive for all species except blue duiker and porcupine. These results reflect the findings of several studies which have now found monitoring that incorporates local knowledge to be a highly cost-effective option (Danielsen et al. 2010; Turvey et al. 2014; Parry & Peres, 2015), especially useful where data is lacking, or in challenging habitats such as forest environments (Turvey et al 2015; Martinez-Marti et al. 2016).

In future, it could be interesting to run a similar cost analysis as carried out in this study to see what species could be monitored with a given amount of power and for a given amount of money. For example, if an NGO has £500,000 or £30,000 and wanted to detect 50% change in occupancy with 80% power, which species could be robustly monitored with each method and the money they have available. An analysis like this could help NGOs such as ZSL in Cameroon to better allocate their funds by identifying the most cost-effective monitoring methods required to monitor their target species, and how to ensure they maximise the returns on the money they allocate to monitoring.

In exploring the differences in power between methods and species, I used one study village only. Future work is required to assess these methods on a larger scale, to assess the effect that different levels of hunting depletion have on the cost-effort ratio for each method and species, and to see if the same general rules apply in other case studies. Southwell et al. (2019) developed a simulation framework to perform spatially explicit power analysis of monitoring programmes for detecting temporal trends in occupancy for multiple species. Users must specify the number and location of sites, the frequency and duration of surveys, and the type of detection method for each species, for the framework to estimate power to detect occupancy trends, both across the landscape, but also within nested management units. In future work it could be interesting to apply this framework to better understand the power of methods spatially explicit power as opposed to landscape only.

Power analysis that accounts for imperfect detection is a valuable tool to assess the effort required to monitor different species, and identify the methods that may be most applicable to monitor different animals. The formula developed by Gurutzeta-Guillera & Lahoz Montfort (2012) allows us to take account of detection, which affects the power to detect different levels of change, and therefore the effort and cost required to reach 80% statistical power. Species such as African golden cat and bongo may have such low detection probability that achieving power to detect any helpful level of change is not viable for projects with small budgets. However, where detection rates are higher, species can be monitored using observational data with a small budget.

# Chapter 7

# Discussion

## 7.1 Introduction

The aim of this thesis was to evaluate the uncertainties and biases of different monitoring methods in the context of a complex and dynamic socioecological system. In a world of increasing need for biodiversity data, biodiversity assessment and monitoring with local people is becoming more important, both for the sustainability of conservation initiatives (Danielsen et al. 2010) and as a tool for building stronger relations between conservationists and local communities (Elbroch et al. 2011; Beland et al. 2013; Mohd-Azlan 2013). The practical value that incorporating local ecological knowledge (LEK) into conservation brings is becoming clearer, but the need for better incorporation of LEK is also reinforced by moral arguments (e.g. it is ethically just to involve local people and their knowledge in the development of conservation initiatives and in the protection of natural resources), and a policy requirement (Target 18 of the Aichi Biodiversity Targets; IPBES deliverable 1c). The cost-effectiveness of using LEK for monitoring compared to conventional methods has received significant attention (Anadon et al 2006; Rist et al 2010; Golden et al 2013; Hausser et al 2016). However, the degree to which data obtained from LEK are robust when used to monitor species in tropical forests, and in particular species hunted for wild meat, is under-researched.

In this thesis, I used social and ecological research methods to improve understanding of the biases associated with occupancy models informed by observational data from local people, and to explore the practical application of these approaches for monitoring in community forests adjacent to the Dja Faunal Reserve (DFR) in Cameroon. I started by contextualising the potential for LEK as a monitoring tool for wild meat species, by developing an understanding of the social-ecological system in which it would occur. This first chapter also sets the scene to understand why we observe some of the differences in species occupancy and detection in subsequent chapters. I then compared estimates of occupancy obtained from seasonal interviews and daily diaries with estimates derived from camera traps set in the same area and over the same time, to investigate how each monitoring method is affected by uncertainty, and identify species-level differences in method performance. Next, I used modern knowledge elicitation methods to gain an understanding of the uncertainty surrounding estimates from local people, and to identify whose knowledge can be robustly used for monitoring and under what circumstances. Finally, I conducted power analyses that account for detectability to explore the practical applicability of LEK informed methods and camera-traps for monitoring different species under varying budgetary and logistical constraints.

## 7.2 Contribution to knowledge

This thesis advanced knowledge on the integration of LEK for wildlife population monitoring of species hunted for wild meat, using two wild meat hunting systems adjacent to the Dja Faunal Reserve as case studies. In this section, I summarize the results of the thesis, and how the results relate to the broader literature, before discussing the overarching themes that emerged from this work and identifying areas for future research and development. My research aims were to: 1) Identify the individual and village level drivers of hunting, and the current threats to hunted species in the Dja Region; 2) Investigate how observational and camera trap methods, when combined with occupancy analysis, are affected by different types of uncertainty within the case study, 3) Explore the trade-offs between cost, precision and accuracy that affect observational and camera trap methods when combined with occupancy analysis and finally 4) Identify barriers to and the potential for the successful integration of LEK into wildlife population monitoring in the Dja region and more broadly.

### 7.2.1 Village level differences in threats and hunting intensity

Wild meat is a valuable non-timber forest product across tropical Africa, helping people to meet their food and livelihood needs (Nasi et al 2011). Yet, hunting systems can quickly be knocked out of equilibrium by environmental, economic or political shocks. Social-ecological systems thinking can provide a more nuanced understanding of the heterogeneity and uncertainty surrounding the effect of overhunting on different species and between locations, allowing for context specific recommendations to move a hunting system closer to sustainability if required (van Vliet et al 2015). In chapter 3, I used the social-ecological systems approach advocated by van Vliet et al. (2015) to combine social and biological data at the village and hunter scales and draw inferences about the intensity of hunting and the role of wild meat hunting in two contrasting systems. This study represents one of only a few empirical applications of the SES framework, and the first to inform such a framework with daily hunter diaries.

I found village level differences in species depletion. While pre-existing differences in species densities could have been a reason for the observed differences, the fact that those in village 2 decided to settle in that specific spot when the village was established (see section 2.6.5 chapter 2), suggests that the forests adjacent to village 2 were once highly abundant. The qualitative data collected during interviews in chapter 3 further supports that participants remember when rare animals used to wonder into the villages which no longer happens. As such, I concluded that these differences are a result of village-level hunting intensity and hunter level differences in hunting method, driven in turn by different motivations for hunting. For example, gun hunting is more prevalent in village 2, while wire snares are the preferred hunt method in village 1. Avila et al. (2019) remark that large-bodied species became rarer due to increased gun-hunting. Given the increase rate of gun hunting large mammals with distance from village, and an increase in gun hunting during times of celebration, it is feasible that gun hunting reflects the desire to hunt larger mammals for sale and maximize returns on effort. Conversely, increased gun hunting may be a response to species depletion; gun hunting is thought to be more efficient than trapping (Damania et al. 2005), and a switch to more efficient hunting techniques could be a response to declining prey availability.

Qualitative data combined with the hunting offtake data from diaries in village 1 identified that the beginnings of a commercial hunting system has coincided with the development of a new logging road (see chapter 3). Previous studies have argued that hunters travel along logging roads in the Congo Basin to obtain wild meat, causing hunting pressure to increase to at times unsustainable levels (Wilkie & Carpenter, 1999; Wilkie et al. 2000; Barnes, 2002; Fa et al 2003; Yakousa et al. 2006; Kleinschroth et al. 2019). Interview data from village 1 revealed that since the logging road was cleared, outsiders are travelling as far as 80km from the town of Lomie, to hunt in the community forest, as hunting is reportedly now too hard close to Lomie, east of the reserve. Tvan der Wal & Nku (1999) also found encounter rates for chimpanzee and gorilla to the east of the reserve to be diminished. However, the results also suggest that some hunters in the village are responding to this new external demand for wild meat and hunting themselves.

The presence of medium and large-bodied species of conservation concern such as gorilla, chimpanzee, and pangolin demonstrates the high conservation value of the community forest immediately surrounding village 1, despite the prevalence of wild meat hunting in the region. In their study of species defaunation around villages in Central Africa, Beirne et al. (2019) also concluded that the community forests can be of high conservation value despite wild meat hunting, especially where villages are more remote and close to protected areas.

Wild meat plays an important role for both income and subsistence for the inhabitants of both village 1 and 2. As such, it is important that future efforts to curb wild meat consumption also acknowledge both these important proximate drivers of hunting for rural communities around the DFR and further afield. For example, projects that establish livelihood or protein alternatives tend to deal with one element of the drivers to hunt (e.g. either financial or for subsistence) whereas a more integrated approach that tackles both the key hunter-level drivers may be more effective. The impact that logging roads has on facilitating the commercialisation of hunting systems is well documented (see Yasuoka et al. 2015; Kleinschroth et al. 2019). Yet, work that seeks to improve the resilience of hunting systems in the face of growing external pressures, from changing land-use and the expansion of logging roads, for example, are lacking. This is perhaps because the wider economic drivers of hunting are seen to be too complex, insurmountable, or beyond the jurisdiction of conservation.

The SES framework provides a helpful approach to identify, not just the hunter and village level drivers of hunting, but how the local and regional context that wild meat hunters are situated within may also affect hunting patterns (Nasi et al 2008). With the acknowledgement of the threat that hunting for wild meat presents for biodiversity in the DFR and elsewhere (Ripple et al. 2016), more nuanced approaches such as those that adopt a SES framework approach, should be quickly adopted into studies of wild meat and species monitoring. This would identify and more effectively tackle potential threats to both biodiversity and food security.

In this study I remarked how outdated much of the literature that aims to quantify hunting pressure and the importance of wild meat for food security and livelihoods.

Dethier (1995) found that ungulates made up 88% of the wild meat captured in the DFR, followed by other (6%), rodents (5%), and primates (4%). In this study, ungulates made up 53% of the wild meat capture in village 2, followed by rodents (22%), other (13%) and primates (12%). The proportion of ungulates hunted is significantly lower than in Dethier's study, while the proportion of rodents in village 2 is much higher. The change in ungulate: rodent ratio may again signify a decline in the availability of larger mammals in the forests close to village 2.

Robust and up to date data is essential to understand if conservation interventions are having an impact both on participating communities, but also on biodiversity, yet studies on hunting intensity conducted in the late 1990's are still referenced today. The community forests around the DFR still host a range of species of global conservation concern and potentially at densities that equal those within the reserve (see later section on the obtaining robust estimates from people, section 7.2.2). Therefore, an increased focus on understanding the intensity of wild meat hunting both within the DFR and in surrounding forests should be a priority for conservation actors active in the Dja region.

### 7.2.2 Bias that affects local knowledge when used for species population monitoring

Rapid and cost-effective monitoring methods that are both robust and applicable over large spatial and temporal scales are needed so that trends in species populations and the drivers of those changes be identified in time for remedial action to be taken. In chapter 4, I used a mixed-method approach to triangulate estimates of detectability and occupancy obtained from daily hunter diaries, seasonal interviews and camera traps. I then assessed the precision, accuracy and comparability of estimates at different scales and provided guidance on the future use of these methods for monitoring both threatened and hunted mammals.

Previous studies that seek to compare monitoring data from LEK to conventional methods tend to focus on a single species. However, by comparing estimates for multiple species as I have in this study, I was able to identify species-level differences in performance for each monitoring method. Species characteristics have a role in determining which methods are most applicable. For example, bold, diurnal species such as gorilla and chimpanzee, both of economic importance to hunters, were readily detected using interview methods in chapter 4. Hunters may target areas where they know certain species can be found, increasing their chances of detection (Service et al. 2014; Turvey et al. 2013, 2015; Mohd-Azlan et al. 2013).

Results for shy or cryptic species, in particular sitatunga, bongo and duiker species such as yellow-backed duiker were less certain. Interview data was the only data type to provide estimates for these species in chapter 4 and as such comparisons were not possible, while comparisons in chapter 5 gave mixed results for yellow-backed duiker. Given the shy nature of these species and the challenging habitats in which they occupy (IUCN 2016), more targeted survey effort may be required in future to enable a more robust comparison of survey methods for these species.

In this study, camera traps were only able to produce occupancy estimates for two species of conservation concern (tree pangolin and chimpanzee). Furthermore, the estimate for chimpanzee was highly imprecise due to very low detectability, despite previous studies reporting that these species are curious about cameras which may increase their detectability with this method (Meek et al. 2016). It may be that species of conservation concern are at low densities in these community forests due to human disturbance rendering them harder to detect, although some studies find that the non-invasive nature of camera traps increases the likelihood of detecting shy or cryptic species (Silveira et al. 2003; Rowcliffe et al. 2008; Rovero & Marshall, 2009). In many studies, camera traps are set to maximize the chance of detection for the target species, while in this multi-species study, this not possible. As such, it may be that the camera trap placement biased the chances of detecting rare species in favour of the LEK informed methods, because local people are able to travel around freely and access parts of the forest that camera traps would not usually be set (Zeller et al. 2011; Service et al. 2014).

Table 7-1 summerises the key biases that are commonly encountered during wildlife population monitoring, and the potential solutions to these biases as foudnin this study.

Table 7-1: Table summarising the key biases that can occur in wildlife population monitoring, and the potential solutions to these biases as found in this study.

| Potential consequences of not following good monitoring practice | Cause of issue | Potential solution |
|---|---|---|
| Spatial bias | Conventional methods that are limited spatially due to cost/resources | Interview -based approaches reach a wider area and can be used to fill gaps and identify areas to focus subsequent more intensive surveys |
| Heterogeneous detection being mistaken for differences in occupancy | Not using occupancy analysis | Use occupancy analysis which account for imperfect detection<br><br>Use multiple methods to triangulate estimates from different sources |
| Expert judgements that do not reflect the 'truth' | Overconfidence<br><br>Overestimation | Use expert elicitation protocols to capture uncertainty<br><br>Unless certain of who will provide the best information, interview many people and use the pooled estimates to draw on the 'wisdom of the crowds' .<br><br>Use multiple methods to triangulate estimates from different sources and get an understanding of uncertainty |
| Lack of power to detect change | Poor survey design that does not account for species detection | Use power analysis that account for detection to ascertain if monitoring is feasible prior to starting monitoring<br><br>Spend realistic amounts of money to increase power |

| | | |
|---|---|---|
| | | Change the research question to allow a more simple monitoring metric (simple presence/absence)<br><br>One-off surveys where the power to detect change over time is not feasible (e.g. inventory approach) |

### 7.2.3 Eliciting expert judgement to inform monitoring of mammals in Cameroon

Expert judgement represents a valuable source of information, especially when time and resources are stretched, or when extrapolations are required for novel or uncertain situations (Kuhnert et al. 2010; Burgman et al. 2011). As with all sources of information, however, expert judgment is subject to bias that are often influenced, both consciously and subconsciously, by values and judgements. Structured elicitation protocols have been developed to help experts engage with uncertainty and minimize consequent bias when providing judgements that can be helpful for informing conservation action (Hemming et al. 2017).  In chapter 5, I applied these modern expert knowledge elicitation and evaluation methods to assess the robustness of judgements made by local people about species occupancy and densities within their community forest. The novelty of this approach is twofold. Local people in a developing country are rarely consulted using expert elicitation methods, which are usually applied to scientists (e.g. Goossens et al. 2001; Knol et al. 2010; Bamber et al. 2013; Aizpura et al. 2015). Further, this is the first time that these elicitation approaches have been applied in a wild meat hunting context.

Estimates of occupancy from local people were similar to camera trap informed estimates for abundant and commonly hunted species such as blue duiker and porcupine, reflecting the results of chapter 4. In contrast, abundant species such as porcupine and blue duiker resulted in imprecise estimates of density while again species of conservation concern such as gorilla and elephant provided comparable estimates to the wider literature, although again caution is required with this interpretation as the comparison literature used may not reflect the local reality.

As with chapter 4, gender played a significant role in predicting the type of knowledge held (with men knowing more about species occupancy and women about species density), but there was no evidence that overall expertise was gender-related. Other studies have also found awareness and judgements to differ between gender; Graef & Uckert (2018) examined scientists' assessments of strategies to upgrade the food security of subsistence farmers in Tanzania, and found that the impact assessments of scientists differed based on their gender. Further, the significance of gender above any other variable as an expected predictor of knowledge is consistent with studies conducted in Australia (e.g. Hemming et al 2018; McBride et al. 2012). Hitomi & Loring (2018) found that LEK research in the circumpolar north is biased towards male knowledge-holders, who are usually elders or hunters, outnumbering women and youth 2:1. Unconscious bias that places male knowledge above that held by women is a reoccurring theme in a range of different cultural contexts and for experts in different fields, including ecology and natural resource management. As such, unconscious bias must be better accounted for in future studies (Baum & Martin 2018) and the impact of male and female awareness should be better integrated into expert elicitation protocols, by ensuring that both men and women are included in studies in consistent ways so that their results can be compared, and conducting further research into the role that gender plays in the type of knowledge held by participants in a range of forest-dependent social-ecological systems.

### 7.2.4 Trade-offs between cost and robustness when monitoring species hunted for wild meat

Species population monitoring programmes require different levels of effort and cost depending on the monitoring objectives (Williams et al 2002). Yet, poor consideration of both the objectives and design of monitoring programmes often results in conservation resources being misspent on monitoring activities which do not result in statistical power to detect a desired level of change (Robinson et al. 2018). Power analyses are commonly used to improve the cost-effectiveness of monitoring strategies for wildlife populations by calculating the probability that a given survey design will detect a true trend (Gerodette, 1987). Yet, such power analyses have rarely accounted for species detection probability, which can substantially affect the power of a monitoring programme to detect trends based on occupancy models (MacKenzie et al. 2005). In chapter 6, I applied the formula developed by Guillera-Arroita & Lahoz-Monfort (2012) to assess the power of seasonal interviews, hunter diaries and camera traps to detect change under the current study survey design, and calculate the effort and cost required to detect 50% change in occupancy for different mammal species. This is the first study to apply the formula to LEK informed data collected in a wild meat hunting system.

Results of this study provide insight into how conservation practitioners can maximise their efforts and design. Species occupancy and detection dictate the optimal strategy to increase power; Increasing the number of repeat visits per site substantially increased the ability to achieve 80% power to detect change, where species occupancy was high and detection was low, such as for yellow-backed duiker and red river hog. In their study on identifying the optimal monitoring strategy to detect rule breaking behaviour in Sierra Leone, Jones et al. (2017) also found that power could be improved by increasing the number of visits per site.

Interview data has repeatedly been shown to be a cheaper alternative to camera traps or other conventional monitoring methods (e.g. Gaidet-Drapier et al. 2006; Turvey et al 2012; Service et al. 2014; Taubmann et al. 2016; Martinez-Marti et al. 2017; Brittain et al. 2018) and chapters 4 & 5 found that LEK methods are often the only method that can provide robust occupancy data for species of conservation concern. However, chapter 6 showed that once detection is accounted for, achieving power to detect trends in species populations over time is not feasible under the current study design, nor in fact with any survey effort that is realistic under NGO budgetary restrictions. For example, the number of sites required to be surveyed to detect 80% change in occupancy for sitatunga exceeded 1000 sites which is unfeasible for most conservation budgets if sufficient camera traps needs to be bought. Similar results are reported by Barata et al (2017), who found that detecting small population changes for rare and cryptic species is particularly challenging as they may require a large number of sites to be surveyed. Further, Sommerville et al. (2011) examined the use of species and threat indicators in a community-based payments for ecosystem services scheme in Madagascar. They explored a range of approaches to detect change in the proportions of observations of each species and concluded that only the most common species had sufficient power on which to base a 'payments for environmental services' scheme.

These results add to the body of literature that highlights the importance of accounting for species characteristics in the survey design process to ensure the monitoring programme has sufficient power while reducing the waste of limited conservation resources (Brashares & Sam, 2005; Lindenmayer & Likens, 2009). Where the target species is of a low detectability, conservation actors must consider before monitoring whether the logistical resources or money available to monitor the target species, and use power analysis that account for detection to help determine this (McDonald-Madden et al. 2010; Earle 2017).

## 7.3 Global themes

In this study, several cross-cutting themes emerged which must be addressed if LEK is to be used more effectively and robustly for wildlife population monitoring, both in tropical forest settings such as my case study, and more generally. The first relates to the practical application of LEK for wildlife population monitoring, including getting the most robust estimates from participants and the cost-effectiveness trade off. I then discuss the moral and ethical implications of incorporating LEK into wildlife population monitoring and, the mechanisms and approaches required to ensure a greater integration of LEK monitoring in a way that is in-line with policy recommendations.

### 7.3.1 Obtaining robust estimates from people

When discussing changes in their hunting systems, participants in village 1 said that they perceived populations of species sensitive to overhunting such as pangolin, giant pangolin and chimpanzee to be declining since the arrival of the logging road (chapter 3). However, estimates of occupancy for species sensitive to overhunting, such as large ungulates, great apes and elephants, were overestimated compared to the camera trap data, as were estimates of density in village 1 compared to the wider literature (chapter 5). Given that some of the comparative estimates from the literature were estimates derived from within the Dja reserve, one may expect literature estimates to be higher than those from interviews, given the higher protection awarded to biodiversity in the reserve.

One explanation is that the occupancy and density of species sensitive to overhunting were overestimated because those species were important to participants. Tomasini & Theilade (2019) compared plot assessments with LEK based data on the abundance of medicinal plants, finding that estimates were comparable, but LEK data was more focussed on the harvestable resource; i.e. certain individuals and plant parts. Similar patterns are reflected in chapter 6, with hunter diary data resulting in the greatest detectability and power to detect change for species of economic importance to hunters like primates and pangolins compared to interviews or camera trap data. It's possible then that participants actively look for the species that are important to them economically, and as such are 'on the lookout' for these species and their signs, as opposed to other animals of less interest to them (Danielsen et al. 2005; Zurlini et al. 2006; Leeney & Poncelet, 2013).

Alternatively, it may be that the culture within the study systems encourages overestimates. In Lunn & Dearnen's (2006) study of small-scale fishermen in Thaliand, the fishing culture was offered as a potential reason why fishermen were found to overestimate their catch and effort compared to what the researchers observed. However, other studies have also found that estimates from key informants tended towards the mean, with those who harvested a below average amount overestimating, and those harvesting an above average underestimating (Jones et al 2008). In chapter 5, I found that estimates from peer assessed experts; all men and seen to know a lot because they hunt a lot and spend a lot of time in the forest, tended to overestimate compared to others in the group and compared to the pooled estimates.

Change blindness is another possibility outlined in a study by Papworth et al. (2009), describe how shifting baseline syndrome or change blindness can influence the validity of local ecological knowledge (see table 2-3). In my study, participants may be referring to past levels of occupancy and density for species sensitive to overhunting, where populations of these species may be changing quickly. Qualitative data collected for chapter 3, showed how participants of all ages were aware of the changes in species populations, suggesting that knowledge transfer is still occurring from older to younger generations.

> *"Before the animals were closer to the village than now, they were just eating by the road. There are animals that are not even found here before going 30 km"*
> *Male, village 2, aged 45+*

> *"To go and catch the rare animals you now have to go deep, to cross where there are the timber companies, as on this side where there is [name of a timber company]."*
> *Male, village 2, aged 18-25*

In this study, change blindness appears to occur when participants are asked to provide estimates for these particular species, even though they are conscious of change when discussing perceived population trends more generally. In chapter 4 LEK informed estimates are either comparable or slightly lower than estimates derived from camera traps, showing that participants are able to recall and record what they have seen. This is interestingly contradictory as it suggests that while participants recognise the increased effort they have to go to in order to hunt, or the reduced numbers of animals compared to years past as they do in chapter 3, they do not translate this knowledge into updated estimates when asked in elicitation processes in chapter 5.

If change blindness is occurring for species with rapidly declining populations, how long does it take for participants to adjust to the reality of newly depleted species populations and provide estimates that reflect the new truth? Or, will they always be subject to 'change blindness?'. If participants were able to 'catch up' with current species occupancy or densities, it may constitute shifting baseline syndrome, which presents another challenge for the robustness of LEK based data (Papworth et al 2009).

Another potential reason for the disconnect between density estimates, and one which raises questions about the status quo of conservation efforts in the Dja region, is that densities of protected species such as chimpanzee, giant pangolin and tree pangolin are higher in the community forest than the DFR and adjacent reserves, where the comparative density estimates were obtained (appendix D-1). For example, in chapter 5 I assumed forest elephant density to be 0 in the community forests for both villages as recent estimates from within the DFR were as low as 0.04/km$^2$ (ZSL, 2018) and as such, I thought that densities outside the protection of the DFR may be lower still. However, chapter 5 density estimates from participants in village 1 better reflect the densities recently observed by ZSL (2018), which also supports the finding in chapter 3 that the community forest around village 1 is home to species of global conservation concern that are of high enough densities to lead hunters from far away towns to travel and hunt in the forest. If they are accurate estimates, this suggests that densities of forest elephants and perhaps other protected species of conservation interest in that community forest are in line with, or perhaps higher than, densities from within the reserve itself. In their study of chimpanzee and gorilla densities in community forests adjacent the DFR, Dupain et al (2004) also found that densities were comparable to those found within the reserve. As such, there is perhaps a need to better recognise the value of community forests for conservation around the DFR, and the important role that communities can play in the conservation of wildlife, when left to their own devises. Greater focus on monitoring in non-protected land adjacent to protected areas is required and there may be scope to develop alternative conservation actions to protect these important populations (see Dupain et al 2004).

### 7.3.2 Sustainable and effective monitoring methods

Chapters 4, 5 and 6 highlight the need to consider the characteristics of the target species, and what methods may be most effective as a result. However, a call for increased focus on species level differences is not  new a (e.g. Singh & Milner-Gulland 2011; Munari et al 2011; Belant et al 2013; Yoccoz et al 2001). Still, many studies do not make the need to consider the target species an explicit part of their survey design. This study has drawn robust comparisons to investigate how each monitoring method performs across multiple species, allowing insight into when and how each method performs best. Using this insight, conservation practitioners can consider the best methods available to them, given the detectability of their target species, the time and logistical resources they have available, and the budget they have to spend on monitoring.

There is a need for cost-effective monitoring options in the DFR, where financial and logistical resources are limited, as well as more broadly to ensure that conservation resources are used as efficiently as possible. Limited finances may be stretched high transport and staff costs, and monitoring in dense habitat where detectability is low further adds to the need for intensive survey effort and therefore again increased costs if power to detect change is to be realised at all. Approaches that cost more than £250,000 over 3 years may be prohibitive for many NGOs working in Cameroon or for Parks authorities. Based on the annual budget that ZSL Cameroon has available for monitoring in the DFR, this study indicates that the species which can be effectively monitored using camera traps are significantly limited to porcupine and blue duiker; not the usual targets of conservation monitoring activities.

These results leaves us with a problem; what to do when conservation actors need data on species with low detectability, but can't achieve statistical power when constrained by an average NGO's monitoring budget? One option to improve the efficiency of programmes that aim to monitor changes in occupancy is to stop surveying a site after the first detection, although this is only recommended if the cumulative detection probability at occupied sites is close to 1 (Guillera-Arroita & Lahoz-Monfort, 2017). Therefore, this approach may be suitable for common species such as porcupine, but is not recommended if rarer species are included within the monitoring programme.

Chapter 6 revealed that hunter diary data allowed the most species to be monitored for < £250,000. Further, multiple species could also be monitored using the same method for the price of the most expensive species to render the approach even more cost-effective. The low costs associated with this method means that increasing survey effort to achieve a 50% power to detect change were not prohibitive, as they often were with interview data and camera traps. Kumpel et al (2010) also used hunter diaries to monitor wild meat offtake and consumption in Equatorial Guinea and Rist et al (2010) use hunter diaries to gathered data on wild meat-hunting catch and effort. The authors found that the locally-based monitoring method can offer accurate, cost-effective, and sufficiently powerful way to monitor the status of natural resources, although their power analysis did not account for detection. Yet overall, the application of hunter diaries is relatively limited in studies of wild meat; Ibbett and Brittain (2019) found that of the 185 articles reviewed that use social science methods to investigate wild meat hunting, 95% used one-off interviews or questionnaires, while only 5% used other methods, such as hunter diaries. This suggests there is greater scope to apply hunter diary methods in wild meat hunting studies, but caution is advised in the integration of such approaches into a monitoring programme. Earle (2017) demonstrated how monitoring programmes have suffered because of a lack of engagement by staff who were supposed to work with the local people. Selinske et al (2014) highlight the motivations that can keep landowners in South Africa engaged in engaged and participating in the CapeNature programme. The authors found that social learning was the most important predictor of participant satisfaction and their continued involvement in a conservation programme. However, they also found that satisfaction with the conservation programme post-enrolment was in part reliant on their expectations being met and on their interactions with the conservation actor. This reflects the findings of Earle (2017) and offers a potential reason why community monitors in her study may have 'drop-off' the monitoring programme due to the insufficient follow up and ongoing engagement from NGO staff. As such, community monitoring programmes that employ participatory methods must ensure that they factor in sufficient time and budget for follow-up and engagement activities throughout the course of the monitoring programme. Although budget in the project costings for chapter 6 was allocated to cover the costs of continuous key informant engagement with those completing the hunter diaries, more budget would need to be allocated in reality to allow sustainable and ongoing engagement between the NGO staff and village monitors.

Taking all the cost into account, it may then be that robust occupancy analysis is too expensive or requires an excessively large survey effort for species with low detections using any method. In such cases, it may be better to revert back to presence only, or presence-absence surveys which are further down the information hierarchy set out by Guillera-Arroita (2017) and provide us with more limited information, but can tell us whether or not, for example, gorilla are present in a given location.  Additionally, monitoring trends in reported threats instead may require less survey effort and as such could also offer a cost-effective alternative to gathering data on species population trends (see Wilcox et al. 2019).

While monitoring is typically used to understand rates of change or the effects of management practices on wildlife populations and habitats, an inventory approach is commonly conducted to determine the distribution and composition of wildlife and wildlife habitats, especially in areas where such information is lacking (Morrison et al. 2008). Earle et al (2017) proposed also that the inventory approach offered a more cost-effective approach which may be of particular value when monitoring species with low detection.

Furthermore, she found that this approach may provide greater flexibility for community-based monitors because they do not need to adhere to the strict assumptions that relate to occupancy analysis, allowing monitoring to better fit in with peoples' livelihoods. The issue returns again then to our research objectives; do we need to know more than 'are there gorillas in the Dja'? If so, how much effort is needed to estimate their occupancy, and is it worth it? In such situations, incorporating LEK may be the best option available to conservation actors to gather  and cost-effective data, as the results of this study support.

If gathering more data about a species or ecosystem does not change the cost-efficiency of management decisions, the new information will likely not result in improved management performance (e.g. Lindsey et al. 2005). Value-of-information analysis (VOI) may help to resolve the trade-off between spending more funds and gaining new information, by evaluating how much management performance could improve if new information was gained (Field et al 2004). VoI has been used to identify how much investment would be useful to gain more information about koala survival and threats (Maxwell et al 2014) while Runge et al. (2011) used expert knowledge to develop preliminary predictions of management response under a series of hypotheses before VoI was used to determine how much management could improve if uncertainty around these hypotheses was resolved. Despite increasing use of the VoI approach in conservation more broadly, there are few examples where this analysis has been used to increase the efficiency and target responses to studies of wild meat. As such, VoI could be a useful additional tool for the wild meat monitoring toolbox.

This study shows that without first considering whether there is a need to monitor at all (Yoccoz et al 2001; McDonnald Madden et al. 2010), followed by adequate prior consideration of the effort required to detect changes with appropriate power (Singh & Milner-Gulland 2011), we are potentially wasting time and resources. This could result in inadequate and misleading information, which is dangerous because it gives the impression that useful monitoring and conservation has taken place when it has not (Legg & Nagy 2006). A review of the power to detect change should be a fundamental part of the planning and design phase of any monitoring project, as has been widely been discussed in the scientific literature for some years now (Legg & Nagy 2006). Otherwise, we may not invest in monitoring a site that has a population of a target species because we didn't know it was there, or we may make inferences and policy recommendations based on information we believe to be true, but without having any power to detect change in species occupancy under the current plan or budget. It may be that researchers are not using these approaches due to a lack of understanding of the statistics behind the model. However, the new formula (Guillera-Arroita & Lahoz-Monfort, 2012) allows for relatively straight forward calculation of power and cost, once the initial values for simulations have been calculated. Training for conservation actors in NGOs and government in basic power analysis and statistics could be valuable to build capacity and enable monitoring resources to be saved or spent sensibly.

## 7.4 Future directions

This thesis contributes to our understanding of the social and ecological variables that can cause bias when using LEK for wildlife population monitoring in villages adjacent to the Dja Faunal Reserve in Cameroon. It also demonstrated the real-life applicability of monitoring using LEK, for a range of different species, and the cost-effectiveness of doing so when trying to achieve power to detect a given level of change. There is however, much more to learn about the robust integration of LEK into monitoring species hunted for wild meat, and more generally about how we should design and implement monitoring programmes that incorporate LEK moving forward.

### 7.4.1 More socioecological research on wild meat hunting is needed in and around the DFR.

There is a lack of research in and around the DFR that focusses on assessing hunting pressure and understanding the importance of wild meat for food security and local livelihoods. Studies in and around the DFR have focussed on the impact of logging on biodiversity (Oke, 2009; Betti et al 2004); human-wildlife conflict between biodiversity and subsistence farmers (Arlet et al 2007; 2010); wildlife population monitoring of mammals within the reserve (Bruce et al 2017; 2018) and some studies examine the governance of the DRF and issues of corruption (Peh et al 2010). The DFR is frequently mentioned in publications that discuss bushmeat on a regional or global scale (Bennet et al. 2007 Ziegler et al 2015). The threat that wild meat hunting poses to biodiversity in and around the DFR is frequently cited in the literature. Yet, since Muchaal et al (1999) examined the impact of hunting on wildlife populations and quantified the importance of wild meat for livelihoods and food security, little work has specifically focussed on the socioeconomic importance of wild meat and the level of hunting pressure from communities. A recent study aimed to identify the level of anthropogenic pressure on great apes and elephants from within the reserve (Farfan et al 2018), finding that the threat of illegal hunting is highest in the centre of the reserve, but again not tackling the socioeconomic variables that may be driving the observed trends. Avila et al (2019) interpreted long-term trends in wild meat harvest in three villages near the DFR, collected from daily hunter diaries. While they identify important biological trends and changes in hunted mean body mass indicator (MBMI) (Ingram et al. 2015) from 2003-2016, there is little acknowledgement of the social dimensions or economic drivers of hunting within the study that may also be driving the observed trends.

Despite its global importance for biodiversity, the state of conservation of the DFR is precarious, due to the continuing impact of uncontrolled commercial hunting and other illegal activities. Thousands of forest dependent people live around the periphery of the DFR, and surrounding community forests may also provide valuable conservation land, as this study in village 1 in particular has demonstrated. A lack of understanding of the trends and drivers of population trends is not only a threat to biodiversity, but also on the livelihoods and food security of the communities living adjacent to the DFR.

### 7.4.2 Scaling-up comparison studies

While the village-level results showed good agreement between methods, agreement between site-level occupancy estimates were less convincing, because a) comparison datasets from camera traps were not available for many species and as such I was only able to compare site-level estimates for porcupine, tree pangolin and chimpanzee and b) where comparison estimates from camera traps were available, the results were highly mixed; site-level predicted occupancy from camera and interviews for tree pangolin were highly corroborative, while estimates for red river hog differed greatly.

It is important to improve our understanding of how environmental variables at a finer scale can influence the distribution of species. It could be that the comparison across 30km$^2$, was too small and as such didn't provide enough spatial variation. The occupancy analysis in chapter 6 that was not restricted to 30km$^2$ resulted in much more significance in variables. As such perhaps a larger-scale comparison could be beneficial in understanding the environmental and observer variables that affect occupancy and detection.

Van Strien et al (2013) investigated whether occupancy models can correct for the observation, reporting and detection biases in opportunistic data. They compared trends in occupancy of butterfly and dragonfly species derived from opportunistic data with those derived from standardized monitoring data and ensured that all data came from the same grid squares and years to avoid geographical bias. As I also found in this study, occupancy models were able to control for the common biases encountered with opportunistic data, enabling occupancy informed by interview data to be monitored for species groups and regions where it is not feasible to collect standardized data on a large-scale. The authors also found that trends in opportunistic and monitoring data were well-matched at the site-level and as such, opportunistic data can be used for monitoring purposes if occupancy models are used for analysis. This study was closely followed by Polfus et al (2014), who examined the strengths and weaknesses of predicting woodland caribou habitat selection based on western science and TEK-based models in British Columbia. They demonstrated again that TEK-based habitat models can effectively inform recovery planning.

While Polfus et al (2014) conducted their study across over 11,000 $km^2$, the comparable area covered in this study was substantially smaller and as such may not have been large enough to detect variation in occupancy at the finer scale. A larger-scale study in several villages all around the Dja would not only provide more data to do a more comprehensive comparison of site-level estimates of occupancy across methods, but would also serve to understand further if the results obtained in this study are globally true and to draw conclusions that are applicable for the Dja region as a whole, allowing further relevant recommendations for conservation management across the Dja region.

### 7.4.3 Use of more sophisticated occupancy models

While I accounted for false-negatives in this study (MacKenzie et al 2002), I did not use the more recent occupancy models developed by Royle & Link (2006) or Miller et al (2011) that also allow for false-positives to be accounted for. False-positives have been found to occur, for example, where there is a chance for species misidentification (Molinari-Jobin et al. 2012). In this study, there was the potential for false-positives for species of economic or cultural value, such as giant pangolin or african golden cat, or for species such as sitatunga or bongo, which may be confused with each other. Checks were put into place prior to the interviews to ensure that participants did not falsely identify the species. Further, the results show that as expected, participants recorded that Bongo were almost absent, which overall suggesting that none, or very little, misidentification occurred.

In this study, there were many sites with only one or two detections, because participants were not surveyors and as such were not evenly distributed across the landscape. Therefore, models that account for false-positives may well have dramatically reduced estimations of occupancy by considering all sites with one detection to be a false-positive (Petracca et al 2017). As such, I decided not employ them in this study, and the results compared to the camera trap data indicate that estimates from participants were not globally overestimated. However, if participants had provided data across the landscape more evenly, I would recommend exploring occupancy models that account for false detections, the importance of which are being increasingly recognized (Berigan et al 2018; Petracca et al 2017; Miller et al 2011; Royle and Link 2006).

## Knowing 'the truth'

There have been calls for studies that assess LEK-informed occupancy models where the truth is known (Petracca et al 2017), which, for studies of rare species in particular, could be helpful to disentangle where estimates from LEK or camera traps are more robust. However, in this case study, I didn't know 'the truth' in terms of the actual occupancy or density of species at either site. While assessing the estimates derived from LEK compared to conventional methods was a primary aim of this thesis, I wanted to do so in a context where LEK informed occupancy would be of actual value. There is little practical value in applying LEK methods in a context where 'the truth' can be identified within the financial and logistical constraints that usually apply to monitoring programmes. However, a "model" system, where the truth is known would be a helpful approach to further explore the comparability of estimates across methods. For example, Lubow et al (2016) estimated the error of aerial surveys when counting a known number of feral horses in the United States, to improve the application of the method and reduce bias. Moore et al (2011) planted a known number of invasive Hawkweed plants over a 2ha area to test for observer effects in a volunteer programmes looking for invasive plants. As in this study, the authors found that experience variables has no effect on the participants ability to detect the plants. Keane et al (2018) and O'Kelly et al (2018) followed a similar approach, setting fake snares in Cambodia to determine the variables that affect snare detection rates by rangers and their detection rate against the "true" number of snares set. Comparison studies in the Serengeti where species densities are known have been carried out to assess bias of different monitoring methods (e.g. Norton-Griffiths et al. 1978). In Cameroon however the densities of species, even in hunting zones or safaris where permits are sold to hunt large game, are not known., limiting our ability to establish model systems in the Dja region.

### 7.4.4 Develop culturally appropriate exert elicitation protocols

Using expert elicitation protocols allowed me to capture uncertainty around estimates that would otherwise had been missed if I had asked participants for a straight answer, as if often used in interviews. The results lead me to encourage the uptake of these protocols as a standard when using LEK for monitoring, as they can help us better understand uncertainty and make more honest and robustly informed decisions . However, the initial trialling of the expert elicitation protocols wasn't straightforward. I first tried to trial the protocol on my first data collection trip to the villages, but people did not really engage in the questions and seemed sceptical of the protocol, asking me why I kept asking them for lower, then upper estimates rather than a straight estimate. When I asked them if they were certain of their answers, some seemed to be a bit confused about what I was trying to achieve. I think in hindsight that I was still quite nervous about asking people to spend significant amounts of time on an exercise, when I knew they were particularly busy at that time of year with agricultural activities. Having spent a further 6 months working in the villages, I tried again, this time more confident in my ability to facilitate the sessions and with greater cultural understanding of the time cost I was asking participants to spend and how best to communicate the purpose of my work. The expert elicitation sessions worked well the second time around, but still participants were hesitant to update their estimates. This is in part because it involved spending a further participants felt that the estimates they had given were correct and they did not understand why coming together to discuss the results would cause them to change their estimates. There is also a time cost; the elicitation process is not quick, despite being more established in the village and confident in the protocol, I didn't want to keep them any longer than absolutely necessary. Future work needs to focus on developing a more cultural relevant version of the IDEA protocol to ask participants to reflect upon and, if necessary, reconsider their estimates. Such methods would need to be rapid, in recognition of the time-burden such research can place upon households during busy agricultural seasons. Further, there are quite distinct approaches between expert elicitation protocols and those commonly used to engage local people, both with their own strengths and weaknesses. A further question for research could be to explore to which circumstance each is suited.

Monitoring species hunted for wild meat is not easy. Many studies rely on purely biological measures of sustainability which provide a snapshot in space and time which in turn encourages a static assessment of hunting sustainability. Furthermore, common methods to assess offtake such as market surveys do not allow for spatially explicit understanding of where the species was hunted. Chapter 4 outlined the application of an icon-based, daily hunter diary that was completed by village participants  to collect spatially explicit hunting offtake data over the course of 9 months. This method presents a helpful contribution to the toolbox for monitoring species hunted for wild meat. Shaffer et al (2018) integrate quantitative data on hunting offtake of primates collected with hunter self-monitoring approaches and semi-structured interviews in an indigenous reserve in Guyana. By incorporating this data into spatially explicit biodemographic models they were able to assess the sustainability of four primate species. Future studies on wild meat hunting should consider a similar approach to gather spatially explicit offtake data in a rapid and cost-effective way.

## 7.5 Moral implications

This thesis has focused on identifying and accounting for the biases present when using LEK and conventional methods for monitoring species hunted for wild meat. It has not, however, addressed the wider complexity that surrounds the use of LEK for conservation. In this section, I will discuss some of the moral and ethical implications of incorporating LEK into wildlife population monitoring, including the discourse on the ethical implications of comparing LEK with western science.

Conservation research that involves local people is often extractive; rather than being applied to tackle problems deemed important by local people, LEK is often used to rapidly gather cost-effective data that helps to answer conservation questions important to researchers (e.g. Brittain et al 2018; Turvey et al 2013; 2015). This study is no different, in that it was extractive. As I was aware of the extractive nature of the study, efforts were made to be as participatory as possible where the study allowed, for example with the use of participatory mapping methods and the design of the daily diary approach with the hunters. Yet, the data collected did not primarily serve to address questions of importance to participants or their communities. While the benefits of LEK for conservation are becoming clearer, the benefits that incorporating LEK into conservation bring to local people, are at times questionable (Briggs, 2005; Popova, 2013).

The argument for the extractive nature of this study was that it provides the most robust evidence to date on the applicability of LEK for monitoring species of conservation interest in the Dja region, as well as broader and more generalisable lessons for the field. As such, the results could enable a more robust integration of LEK-informed wildlife population monitoring into conservation practice for a range of mammal species. I hope that this study facilitates the next step that is required in the Dja region, and more broadly, that where desired, monitoring methods that are based on LEK can be integrated into participatory, community-led initiatives. This could provide benefits for the local communities that I worked with, albeit indirectly and with a timelag.

However, the morality of drawing comparisons between local knowledge and western science has long inspired discussion. Sillitoe et al (2003) argues that "science is no less culturally located than other knowledge traditions, yet the scientific perspective is often privileged to distinguish it from others' knowledge traditions". This sentiment is echoed by other authors, who argue that comparing other knowledge traditions with Western science is unacceptable, because it overlooks differences within and similarities between various local and scientific perspectives (Agrawal 1995; Parkes 2000). Gilchrist & Mallory (2005) used case studies to compare LEK derived estimates against estimates derived from conventional monitoring methods for assessing bird populations. They found that estimates were often highly comparable across methods, and provide some of the most comprehensive evidence in support of the application of LEK for wildlife population monitoring at that time. However, Brook & McLachlan (2005) disagreed that the comparison should have been drawn in the first place, because the results neglected to discuss the wide range of benefits that LEK has been found to bring, and as such simplified LEK by trying to fit it into the framework of western science.

There is no doubt that the value of LEK extends far and beyond wildlife population monitoring, and should be seen as more than an approach to help meet conservation and community objectives. I do not believe that information from local people has to fit into a 'western science' framework in order to be informative. However, in the very specific context of wildlife population monitoring, we must have evidence that the data used to draw inferences and make species management decisions are robust, for LEK to be considered a viable and robust method for species population monitoring (Gilchrist & Mallory 2007). For example, having collected the data for my MSc paper on forest elephant distribution and relative abundance in the timber concessions of eastern Cameroon, I contacted IUCN to add my data to their CITES MIKE database, but was told that the data I had was not in an admissible format as it came from interviews. I was also told in at a workshop on monitoring methods that the use of LEK for monitoring is not and will not become a serious monitoring method. While IPBES admits LEK derived data to inform decision making, I personally have experienced two cases where data of this standard was not deemed robust enough for inclusion in monitoring protocols of global databases, where I believe that data would have been highly valuable for species conservation. Despite the growing popularity of integrating LEK into wildlife population monitoring efforts, a lack of evidence on the robustness of data obtained from local people may be one of the greatest barriers to the meaningful application of local knowledge for wildlife population monitoring, and limit the extent to which local voices are represented in relevant conservation policies. Double standards are also operating between warm words and actual practice of incorporating LEK into monitoring or conservation.

Many of the studies that reported large disparities between estimates from LEK and conventional methods did not use occupancy analysis to incorporate variables that account for heterogeneity in occupancy and detection. Failure to account for this heterogeneity may have resulted in the biases reported, such as overestimation of species range (e.g. Garrote & Ayala 2015). In this study, species detections obtained from daily diaries and interviews with local people were integrated into an occupancy framework; the resulting estimates of occupancy were often comparable with camera trap estimates, or provided believable estimates where camera trap data could not, due to poor detections. The study supports a growing body of evidence that LEK can be robustly incorporated into wildlife population monitoring to cover large areas and gather data on species that are costly to monitor using conventional methods, when combined with occupancy analysis to account for heterogeneity in detection (Brittain et al 2018; Martinez-Marti et al 2018). As such, I suggest that combining LEK with occupancy analysis could act as a 'gateway' to allow for greater acceptance of LEK as a robust wildlife population monitoring method.

Of course, data gathered in an occupancy framework is far from representative of traditional understandings of biodiversity in the Dja reserve, which again presents a moral dilemma. While LEK in many fields is incorporated in a qualitative way which allows for a more holistic, qualitative understanding of the system or species, through open interviews and participatory approaches, the occupancy framework risks reducing this rich knowledge to biological or western science's 'need to know' terms. However, occupancy analysis does not need to be used in isolation. In this study, I combined the simple presence/absence-based questions required for occupancy analysis with open questions and informal interviews. This allowed me to gather rich, qualitative information that was deemed important by those I spoke to, such as on perceived threats and drivers of these threats to their livelihoods and natural resources, and problems they associate with their development, such as the lack of proper roads and falling cash crop prices.

This qualitative element of research led to my postdoctoral research, working to understand the drivers of wild meat consumption from the perspective of local people in order to co-design more sustainable and locally relevant alternatives, where desired, that have greater impacts for conservation and food security. This research is taking place in the same villages where I conducted my PhD. As such, an important and direct consequence of the PhD research and my engagement in LEK methods is that subsequent work is going to respond to some of the issues they were raising in conversations with me during my PHD research, such as the annoyance towards NGOs for telling them not to eat wild meat but not bringing viable alternatives. As such, this initial PhD research of an extractive nature is resulting in research that directly responds to and benefit community priorities and needs.

Martinez-Marti (2011) combined questions relating to occupancy analysis with semi-structured interview to gather data on threats to leopard, golden cat and fifteen other species Equatorial Guinea, Similarly to Martinez-Marti, I conclude that combining open questions and interviews with occupancy analysis results in information pertinent for conservation, while also developing an understanding of the key issues facing communities and how subsequent conservation projects may be able to help. Local people are the first to suffer if the species they depend upon for their livelihoods and subsistence decline. As such, their knowledge of the system in which they live should be front and centre of efforts to conserve the biodiversity upon which they depend.

The straightforward and rapid nature of the presence/absence questions required for occupancy analysis lent itself well to interviews with participants who were often busy with other tasks. The greatest challenge was not a lack of understanding of the questions, or species misidentification as is often reported in similar studies (Miller et al. 2011; McKelvey et al. 2008; Molinari-Jobin et al. 2012), but rather how best to ensure that the location of the detections were accurately recorded. Participants were not all familiar with conventional maps, but held a more detailed and high-resolution knowledge of their surroundings than GIS data could provide me. For example, every river tributary, whether it was seasonal or flowed all year round, had a name. Paths cutting across the rivers and into the forest had individual names. The location of individual fruiting trees, fishing areas and hunter cabins were known, as was the location of an area out of bounds for spiritual purposes. There have been several different methods used to define a site, such as using well defined jurisdictional boundaries (Karanth et al 2009), pre-defined areas such as  forest reserves (Brittain et al 2018), or using a drawn map on which participants and researchers are both able to interpret and define spatial extents on maps (Petracca et al 2013; Martinez-Marti et al 2016). As I defined a site as $1km^2$, a grid was overlain on the research team's copy of the participatory map, so as not to confuse participants and force them to link detections to a 'scientific' $1km^2$ unit, totally unfamiliar to them. Detections were identified on participant maps using triangulation between their detailed knowledge of rivers, paths, landmarks and landuse borders, and assigned to the relevant $1km^2$ grid reference on the researchers' map. This approach not only allowed for participants to recall where they had made detections based on a map that they were familiar with, but allowed the researchers to integrate that local knowledge into occupancy modelling in a way that is spatially accurate to $1km^2$.

Future studies that implement this approach should strive to move beyond extraction, towards finding truly participatory applications for the approach. However, to date, a 'full partnership' approach (Karnieli-Miller et al. 2009) and meaningful engagement with local communities in the Dja region and further afield has often been hampered by clashing priorities with conservation actors and unethical social research conduct. For example, failed promises of development as a result of conservation have resulted in frustration and a lack of confidence in conservation efforts (see Ndobe et al. 2007 report on the long-term impact of the ECOFAC project on local communities). As such, participants in village 2 often voiced their frustration at conservation NGOs, and the lack of progress seen in their development, despite the promises of NGOs and a long history of 'social engagement' in conservation and development efforts:

> *"You people (referring to conservation NGOs) have been coming here for decades, telling us not to eat meat, confiscating our bushmeat, "sensitizing" us against eating bushmeat. We are happy not to eat bushmeat, but you must bring alternatives and stop harassing us. What are we meant to do when you keep telling us not to hunt, but we don't have any other option? Then, you come here asking us questions about food, about the animals, and then you leave and we never hear from you again. Then a year later another person comes asking the same questions, and the same thing happens. We are sick of answering questions and still nothing changes. What are you doing with all this information? Do you go home and get rich from it while we stay poor?"*
> *Woman, village 2*

The quote above is telling, because they assume that everyone that comes to the villages works for an NGO, with a specific remit to stop them from eating and hunting wild meat. I quickly discovered that while I was supported in the field by ZSL, that it was best to introduce myself as a student working with a university. I explained that I wasn't there to gather information on them and to pass it onto an NGO, who would then work with the ecoguards to come to their village and arrest them all. Whether or not this type of situation has happened before I don't know, but there is a strong sense that staff working for NGOs on the topic of wild meat are a potential threat to their way of life and also to their personal safety. The impartiality that being a student researcher with a university was marked; I was twice asked by quite confrontational villagers to prove that I was not a spy for prominent international NGO's operating in the area. Having presented them with my student card, their demeanour towards me immediately relaxed and over the months I was there they began to talk openly about their lives and hunting experience While of course my demeanour in the village played an important role in how I was perceived, being a student researcher and making my impartiality towards the subject of wild meat hunting known upon arrival in the villages I believe made a big difference in how I and my work were perceived.

## 7.6 Ethical solutions

When incorporating local knowledge for conservation efforts, it's important that conservation researchers better recognize the uneven power dynamics that often exist when working with people and the impact our work can have, both on the participants and the wider community. However, a lack of robust ethical reporting in published studies of wild meat hunting can perpetuate the impression that ethical considerations when conducting research with local people are not important, or result in research that is of poor ethical quality when other researchers look to the published literature for guidance. In turn, this can lead to real damage to local people involved in research and their communities (Ibbett & Brittain, 2019). For example, unethical research with local people can and has resulted in disappointment and mistrust towards conservation efforts. As such, Brittain et al (in review, appendix A-2) highlight the need for better training of early career researchers, to recognize how conflicts of interests or uneven power dynamics when carrying out research with local people can result in unethical research that negatively impacts the research participants, their communities and also the validity of the research finding themselves. Such steps would be a highly valuable addition to facilitate the ethical and robust integration of LEK into wildlife conservation monitoring efforts.

If implemented properly, harnessing LEK can provide an opportunity to help to address uneven power dynamics between conservation and the people that conservation research has so frequently imposed upon. Stilltoe (2000) discussed how local people can fight cultural imperialism by seizing opportunities to assert a place for their knowledge. Conservation can go beyond 'do no harm'; researchers can become a valuable external ally for participants and research can act to give something back to participants, who may have less power than other actors (Brittain et al. in review). For example, when local knowledge is combined with western approaches to mapping,  the results can also provide a powerful tool to help local people take ownership of their natural resources, and importantly, to collect data on issues that are important to them, thus readdressing the power dynamic that is often so out of balance in the design of conservation research. McCall & Minang (2005) describe the use of participatory mapping in community forest planning and management in Cameroon, concluding that the participatory mapping process increased understanding, empowerment, good governance, and improved the relationship between the community and the government. The Extreme Citizen Science programme (ExCites) is an example of such bottom-up practices that takes into account local needs, practices and culture to design and build new devices and knowledge creation processes (Lewis, 2007). For example, an image-only app (Sapelli) was co-developed and used by indigenous hunter-gatherers and local NGOs to report on illegal logging in the Congo Basin, helping them take ownership of their forest and natural resources in the face of illegal use by outsiders. While scientific knowledge is anchored culturally in Western society, hybridization is occurring and blurring distinctions between both knowledge types (Sillitoe 2000).

I had originally intended to use Sapelli to gather species detection and hunting offtake data from hunters in both villages. Having bought the phones and co-designed icons with the hunters that could be used in the app to represent the animals they detected, and additional information such as the habitat they were in and the hunt method used if applicable, the GPS n the phones was not strong enough for the location data to upload, which prevented the hunter from moving through to the other steps in the app to complete and save the data. As such, the icon-based diary method was used instead and could also potentially be used as a tool for self-management of natural resources by local communities interested in monitoring their wildlife, or even threats to it such as illegal hunting in the forest, especially in situations where technology doesn't pull through. Hunters reacted well to the icon-based diary, aided by the consultation and discussions we had when designing the icon intended for the Sapelli app. As the same icons were used in the hunter diary, the icons were already familiar to all the hunters, and they were in some way invested in the process. With relatively low startup costs in terms of development and training, and very low ongoing financial costs, the method could be used by communities to self-monitor the species or threats important to them, with ongoing input from researchers for analysis of the data (see previous warnings about the need for ongoing support and engagement with partners to avoid drop out), and to at times compare data obtained from participants, with field based observations, to assess if underreporting is occurring, and how to account for such bias if that is the case.

The question lies in how the field of conservation can now provide the mechanisms to allow communities to reach out to researchers or actors with problems that are important to them, so that conservation can truly benefit and cater for the needs of communities, as opposed to communities becoming involved in projects only once conservation researchers or NGOs perceive there to be a problem and contact the communities. One example of successful locally-demanded research by an outsider, that informed local management, comes from One People One Reef programme, in Yap, found in the outer islands of Micronesia. The programme claims a unique approach to adaptive management and conservation in the outer islands of Micronesia. The focus is on helping local communities achieve their goals of self-sufficiency and protection of their environment through capacity building and the careful combination of modern science and local tradition. The programme relies on a mutual two-way flow of knowledge between communities and scientists to achieve workable solutions to natural resource management issues and provides a successful example of how indigenous knowledge and western science can collaborate effectively to meet the needs of local people (Crane 2017).

## 7.7 Policy implications

Despite calls for the integration of local knowledge with conventional scientific knowledge for decision-making about biodiversity and natural resources (Fazey et al., 2006; Raymond et al., 2010), this is rarely reflected in practice (Sutherland et al 2014). The Local Biodiversity Outlooks report (2016) identifies major gaps in the mainstreaming of traditional knowledge and customary systems in processes related to the Strategic Plan for biodiversity (2011-2020).

Many Parties have yet to develop effective mechanisms that allow local people to participate in the development of national biodiversity strategies and action plans (Local Biodiversity Outlooks report 2016). Sutherland et al (2014) reported high hopes for the mechanisms that encourage the integration of LEK into the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), which informs policy and decision-making on biodiversity and ecosystem services and attempts to synthesise and apply multiple forms of evidence to bridge knowledge systems (Tengo et al. 2017)  In theory, presenting approaches from local and indigenous knowledge alongside approaches from western science without prioritizing either is a step in the right direction towards the integration of new knowledge systems into conservation, as Crane (2017) have highlighted. However often, IBPES works largely by conventional scientific approaches and procedures, and those reading the recommendations are usually from a western science framework. As such, it is likely that they will be biased towards electing to choose recommendations or approaches from a western science perspective.

The inclusion of "other effective area-based conservation measures" (OECMs) in the final negotiations of Aichi Target 11, can potentially address the misalignment between Indigenous and local communities' traditional approaches on one hand and Western scientific approaches to conservation, culture and nature on the other (Jonas et al 2017). However, a major challenge lies in measuring the impact that local and indigenous peoples has on the OECM they are custodians of. This study has shown how the integration of local knowledge gathered from diaries, interviews and participatory mapping processes can produce understanding of current and emerging threats to biodiversity when applied in a SES framework, and provide robust estimates for a range of mammal species in forest environments when applied within an occupancy framework. As such, monitoring that follows the approaches set out in this study could be well suited to monitoring by local people in OECM areas and could provide a means through which local people can take ownership of their forests and identify threats to their natural resources.

## 7.8 Conclusion

This thesis explored the application of local ecological knowledge for monitoring threatened species and those hunted for wild meat. LEK is increasingly recognized as a robust and cost-effective approach to monitoring that can also act as a tool to build better links between local people and conservation practitioners. However, there has been a serious lack of focus on assessing sources of uncertainty when monitoring using LEK. As this case study demonstrates, LEK is valuable for identifying new and emerging threats within a SES framework and providing robust and cost-effective estimates of occupancy for rare species at a village level where camera trap data cannot. However, several challenges need to be overcome. Firstly, while estimates of occupancy at the village level were highly comparable with camera trap data, evidence for site-level comparability was lacking. For interview-based occupancy analysis to be effectively used to identify habitat preferences and map relative occupancy at the site-level, a larger scale study is required that allows for greater variation in variables that account for occupancy at a site-level scale (1km$^2$). Secondly, the way in which we as conservationists recruit participants for inclusion into LEK studies needs reconsidering; those considered 'experts' were rarely those who provided the most robust estimates, warning us against snowball sampling that is often used in such contexts. Species detection and occupancy hugely affect the effort and therefore cost required to detect a given level of change. Greater efforts are required to integrate power analyses that consider detectability into the design of monitoring projects, to ensure that the most effective and efficient monitoring methods are applied and that we are not wasting valuable conservation resources. Given the lessons learned in this study that contribute to a better understanding of the applicability of LEK based monitoring, future studies should strive to implement this approach within truly participatory programmes.

# References

Abere, S. A., Lateef, F. L. and Lameed, G. A. (2016) 'Assessment of Hunters and Other Rate of Illegal Activities in Afi-Mbe-Okwango Division, Cross River State, Nigeria', Natural Resources, 07(05), pp. 287–294. doi: 10.4236/nr.2016.75025.

Abernethy, K. A. and Ndong Obiang, A. M. (2010) Bushmeat in Gabon.Technical Report, Ministère des Eaux et Forets. Libreville, Gabon.

Abram, N. K. et al. (2015) 'Mapping perceptions of species' threats and population trends to inform conservation efforts: The Bornean orangutan case study', Diversity and Distributions, 21(5), pp. 487–499. doi: 10.1111/ddi.12286.

Aizpurua, O. et al. (2015) 'Reconciling expert judgement and habitat suitability models as tools for guiding sampling of threatened species', Journal of Applied Ecology, 52(6), pp. 1608–1616. doi: 10.1111/1365-2664.12515.

Akumsi, A. (2003) 'Community participation in wildlife management: The Mount Cameroon experience.', Unasylva, 54, pp. 37–42.

Alexander, J. S. et al. (2015) 'The role of bushmeat in a West African agricultural landscape', Oryx, 49(4), pp. 643–651. doi: 10.1017/S0030605313001294.

Allebone-Webb, S. M. et al. (2011) 'Utilización de Datos de Mercado para Evaluar la Sustentabilidad de la Cacería de Vida Silvestre en Guinea Ecautorial', Conservation Biology, 25(3), pp. 597–606. doi: 10.1111/j.1523-1739.2011.01681.x.

Anadon, J. D. et al. (2009) 'Evaluation of Local Ecological Knowledge as a Method for Collecting Extensive Data on Animal Abundance', Conservation Biology, 23(3), pp. 617–625. doi: 10.1111/j.1523-1739.2008.01145.x.

Athreya, V. et al. (2015) 'Spotted in the News: Using Media Reports to Examine Leopard Distribution, Depredation, and Management Practices outside Protected Areas in Southern India', Plos One, 10(11), p. e0142647. doi: 10.1371/journal.pone.0142647.

Ávila, E. et al. (2017) 'Interpreting long-term trends in bushmeat harvest in southeast Cameroon', Acta Oecologica, pp. 1–9. doi: 10.1016/j.actao.2017.09.007.

Azhar, B. et al. (2013) 'Contribution of illegal hunting, culling of pest species, road accidents and feral dogs to biodiversity loss in established oil-palm landscapes', Wildlife Research, 40(1), pp. 1–9. doi: 10.1071/WR12036.

Bailey, L. L., Simons, T. R. and Pollock, K. H. (2004) 'Estimating Site Occupancy and Species Detection Probability Parameters for Terrestrial Salamanders', Ecological Applications, 14(3), pp. 692–702. doi: 10.1890/03-5012.

Barber, M. et al. (2015) 'The persistence of subsistence: Qualitative social-ecological modeling of indigenous aquatic hunting and gathering in tropical Australia', Ecology and Society, 20(1). doi: 10.5751/ES-07244-200160.

Barboza, R. R. et al. (2016) 'The role of game mammals as bushmeat In the Caatinga, northeast Brazil', Ecology and Society, 21(2). doi: 10.5751/ES-08358-210202.

Baum, J. K. and Martin, T. G. (2018) 'It is time to overcome unconscious bias in ecology', Nature Ecology & Evolution, 2(2), pp. 201–201. doi: 10.1038/s41559-017-0441-y.

Beaudrot, L. et al. (2019) 'Detecting tropical wildlife declines through camera-trap monitoring: An evaluation of the Tropical Ecology Assessment and Monitoring protocol', Oryx, 53(1), pp. 126–129. doi: 10.1017/S0030605318000546.

Belant, J., Hofer, H. and Wilting, A. (2013) 'Comparison of methods for detecting and surveying tropical carnivores', The Raffles Bulletin of Zoology, 28(28), pp. 109–113.

Benítez-López, A. et al. (2017) 'The impact of hunting on tropical mammal and bird populations', Science, 356(6334), pp. 180–183. doi: 10.1126/science.aaj1891.

Berkes, F., Colding, J. and Folke, C. (2000) 'Rediscovery of traditional ecological knowledge as adaptive management.', Ecological Applications, 10, p. 1251–1262.

Biggs, R. et al. (2015) 'Strategies for managing complex social-ecological systems in the face of uncertainty: Examples from South Africa and beyond', Ecology and Society, 20(1). doi: 10.5751/ES-07380-200152.

Blanc, L. et al. (2014) 'Improving abundance estimation by combining capture-recapture and occupancy data: example with a large carnivore', Journal of Applied Ecology. 51(6), pp. 1733–1739. doi: 10.1111/1365-2664.12319.

Blanchard, J. L., Maxwell, D. L. and Jennings, S. (2007) 'Power of monitoring surveys to detect abundance trends in depleted populations: the effects of density-dependent habitat use, patchiness, and climate change', ICES Journal of Marine Science, 65(1), pp.

111–120. doi: 10.1093/icesjms/fsm182.
Blom, A. et al. (2005) 'Factors influencing the distribution of large mammals within a protected central African forest', Oryx, 39(4), pp. 381–388. doi: 10.1017/S0030605305001080.
Bobo, K. S. et al. (2015) 'Bushmeat Hunting in Southeastern Cameroon : Magnitude and Impact on Duikers (Cephalophus spp.)', African study monographs, 51, pp. 119–141. doi: 10.14989/197202.
Boddicker, M., Rodriguez, J. and Amanzo, J. (2002) Indices for Assessment and Monitoring of Large Mammals Within an Adaptive Management Framework, Environ Monit Assess. doi: https://doi.org/10.1023/A:1015225022101.
Bowkett, A. et al. (2006) 'How do you monitor rare and elusive mammals? Counting duikers in Tanzania and Zimbabwe.', in Animals, Zoos and Conservation, pp. 21–28.
Brashares, J. S. et al. (2011) 'Economic and geographic drivers of wildlife consumption in rural Africa', Proceedings of the National Academy of Sciences, 108(34), pp. 13931–13936. doi: 10.1073/pnas.1011526108.
Brashares, J. S. and Sam, M. K. (2005) 'How Much is Enough? Estimating the Minimum Sampling Required for Effective Monitoring of African Reserves', Biodiversity and Conservation, 14(11), pp. 2709–2722. doi: 10.1007/s10531-005-8404-z.
Briggs, J. (2005) 'The use of indigenous knowledge in development: problems and challenges', Progress in Development Studies.
Brittain, S. et al. (2018) 'Combining local knowledge and occupancy analysis for a rapid assessment of the forest elephant Loxodonta cyclotis in Cameroon's timber production forests', Oryx, pp. 1–11. doi: 10.1017/S0030605317001569.
Broegaard, R. B. et al. (2017) 'Wild food collection and nutrition under commercial agriculture expansion in agriculture-forest landscapes', Forest Policy and Economics. 84, pp. 92–101. doi: 10.1016/j.forpol.2016.12.012.
Bruce, T. et al. (2017) 'Camera-Trap Survey for Larger Terrestrial Wildlife in the Dja Biosphere Reserve, Cameroon: Diversity & Intactness of the Larger Vertebrate Fauna'. doi: 10.13140/RG.2.2.20019.94245.
Bruce, T. et al. (2018) 'Locating giant ground pangolins (Smutsia gigantea) using camera traps on burrows in the dja biosphere reserve, Cameroon', Tropical Conservation

Science, 11, pp. 0–4. doi: 10.1177/1940082917749224.
Buckland, S. T. et al. (2001) Introduction to Distance Sampling: Estimating Abundance of Biological Populations. Oxford University Press Inc., New York, USA.
Buckland, S. T. et al. (2015) Distance Sampling: Methods and Applications. Cham: Springer International Publishing (Methods in Statistical Ecology). doi: 10.1007/978-3-319-19219-2.
Burgman, M. et al. (2006) 'Eliciting Expert Judgments: Literature Review', Risk Analysis, pp. 1–71.
Burgman, M. et al. (2011) 'Redefining expertise and improving ecological judgment', Conservation Letters, 4(2), pp. 81–87. doi: 10.1111/j.1755-263X.2011.00165.x.
Burgman, M. (2016) Trusting Judgements How to Get the Best out of Experts. 1st edn. Cambridge University Press. Available at: www.cambridge.org/9781107112087.
Burnham, K. P., Anderson, D. R. and Laake, J. L. (1980) 'Estimation of Density from Line Transect Sampling of Biological Populations', Wildlife Monographs, (72), pp. 3–202.
Burton, A. C. (2012) 'Critical evaluation of a long-term, locally-based wildlife monitoring program in West Africa', Biodiversity and Conservation, 21(12), pp. 3079–3094. doi: 10.1007/s10531-012-0355-6.
Camerer, C. and Johnson, E. (1991) 'The process – performance paradox in expert judgment. How can experts know so much and predict so badly?', in K.A. Ericsson and J. Smith. Toward a General Theory of Expertise. Prospects and Limits. Cambridge.: Cambridge University Press, pp. 195–217.
Carson, S. et al. (2018) 'Visions from Local Populations for Livelihood-Based Solutions to Promote Forest Conservation Sustainability in the Congo Basin', Human Ecology. Human Ecology, 46(6), pp. 887–896. doi: 10.1007/s10745-018-0036-5.
Caruso, N. et al. (2016) 'Carnivore occurrence: do interview-based surveys produce unreliable results?', Oryx, 51(02), pp. 1–6. doi: 10.1017/S0030605315001192.
Ceballos, G. and Ehrlich, P. (2002) 'Mammal population losses and the extinction crisis', Science, 296, p. :904-907. doi: http://dx.doi. org/10.1126/science.1069349.
Cerqueira, M. C. et al. (2013) 'Rare or elusive? A test of expert knowledge about rarity of Amazon forest birds', Diversity and Distributions. Edited by M. Burgman, 19(7), pp. 710–721. doi: 10.1111/ddi.12033.

Chaudhuri, A & Christofides, T (2013) Indirect Questioning in Sample Surveys. pp 177 pages; Springer.
Coad, L. et al. (2010) 'Distribution and Use of Income from Bushmeat in a Rural Village, Central Gabon', Conservation Biology, 24(6), pp. 1510–1518. doi: 10.1111/j.1523-1739.2010.01525.x.
Coad, L. et al. (2013) 'Social and Ecological Change over a Decade in a Village Hunting System, Central Gabon', Conservation Biology, 27(2), pp. 270–280. doi: 10.1111/cobi.12012.
Conteh, A., Gavin, M. C. and Solomon, J. N. (2014) 'Quantifying illegal hunting: A novel application of the quantitative randomised response technique', Biological Conservation. Elsevier Ltd, 189, pp. 16–23. doi: 10.1016/j.biocon.2015.02.002.
Craig, B. A. and Reynolds, J. E. (2004) 'Determination of a manatee population along the atlantic coast of Florida using a bayesian approach with temperature adjusted aerial survey data', Marine Mammal Science, 20(3), pp. 386–400. doi: 10.1111/j.1748-7692.2004.tb01168.x.
Croissant, Y. (2019) 'mlogit: Multinomial Logit Models. R package version 1.0-1. '.
Dai, C. and Hu, W. (2017) 'Hunting strategies employed by bird hunters with economic pursuit in the city of Guiyang, Southwest China', Journal for Nature Conservation. Elsevier, 40(September), pp. 33–41. doi: 10.1016/j.jnc.2017.09.005.
Danielsen, F. et al. (2003) 'Biodiversity monitoring in developing countries: What are we trying to achieve?', Oryx, 37(4), pp. 407–412. doi: 10.1017/S0030605303000735.
Danielsen, F. et al. (2005) 'Does Monitoring Matter? A Quantitative Assessment of Management Decisions from Locally-based Monitoring of Protected Areas', Biodiversity and Conservation, 14(11), pp. 2633–2652. doi: 10.1007/s10531-005-8392-z.
Danielsen, F. et al. (2009) 'Local Participation in Natural Resource Monitoring: a Characterization of Approaches', Conservation Biology, 23(1), pp. 31–42. doi: 10.1111/j.1523-1739.2008.01063.x.
Danielsen, F. et al. (2010) 'Environmental monitoring: the scale and speed of implementation varies according to the degree of peoples involvement', Journal of Applied Ecology, 47(6), pp. 1166–1168. doi: 10.1111/j.1365-2664.2010.01874.x.
Danielsen, F. et al. (2014) 'Testing Focus Groups as a Tool for Connecting Indigenous

and Local Knowledge on Abundance of Natural resources with Science-Based Land Management Systems', Conservation Letters, 7(4), pp. 380–389. doi: 10.1111/conl.12100.
Danielsen, F., Burgess, N. D. and Balmford, A. (2005) 'Monitoring Matters: Examining the Potential of Locally-based Approaches', Biodiversity and Conservation, 14(11), pp. 2507–2542. doi: 10.1007/s10531-005-8375-0.
Davies, G. (2002) 'Bushmeat and international development', Conservation Biology, 16(3), pp. 587–589. doi: 10.1046/j.1523-1739.2002.01636.x.
Dethier, M. (1995) Etude chasse. Projet ECOFAC- Composante Cameroun. Ministry of the Environment, Yaoundé, Cameroon, and ECOFAC, Brussels, Belgium.
Diendhiou, Y. and Diawara, B. (2015) Rapport de mission de suivi reactif conjointe UNESCO/UICN a la reserve de faune du Dja. Yaounde.
Dirzo, R. et al. (2014) 'Defaunation in the anthropocene', Science, 345, p. 401–406.
Conventional on Biological Diversity (CBD). (1993) Available at: http://www.cbd.int/convention/text/.,.
Dounias, E. (2016) 'From Subsistence to Commercial Hunting: Technical Shift in Cynegetic Practices Among Southern Cameroon Forest Dwellers During the 20th Century', Ecology and Society, 21(1), p.23. doi: 10.5751/ES-07946-210123.
Duda, R., Gallois, S. and Reyes-García, V. (2017) 'Hunting Techniques, Wildlife Offtake and Market Integration. a Perspective From Individual Variations Among the Baka (Cameroon)', African Study Monographs, 38(2), pp. 97–118. doi: 10.14989/225259.
Egbe, S. E. and Institute, O.-O. D. (2001) 'The Law, Communities and Wildlife Management in Cameroon', Rural Development Forestry Network, pp. 1–13. Available at: http://www.odi.org.uk/sites/odi.org.uk/files/odi-assets/publications-opinion-files/1216.pdf.
Elphick, C. S. (2008) 'How you count counts: the importance of methods research in applied ecology', Journal of Applied Ecology, 45(5), pp. 1313–1320. doi: 10.1111/j.1365-2664.2008.01545.x.
Endamana, D. et al. (2010) 'A framework for assessing conservation and development in a Congo Basin forest landscape.', Tropical Conservation Science, 3, pp. 262–281.
Evely, A. C. et al. (2008) 'The influence of philosophical perspectives in integrative

research: A conservation case study in the Cairngorms National Park', Ecology and Society, 13(2). doi: 10.5751/ES-02679-130252.
Fa, J., et al. (2006) 'Getting to grips with the magnitude of exploration: Bushmeat in the Cross- Sanaga rivers region, Nigeria and Cameroon.', Biological Conservation, 129, p. 497–510. Available at: htpp:/dx.doi.org/10.1016/j. biocon.2005.11.031.
Fa, J., et al. (2015) 'Correlates of bushmeat in markets and depletion of wildlife', Conservation Biology, 29(3), pp. 805–815. doi: 10.1111/cobi.12441.
Farhadinia, M. S. et al. (2018) 'Citizen science data facilitate monitoring of rare large carnivores in remote montane landscapes', Ecological Indicators, 94(April), pp. 283–291. doi: 10.1016/j.ecolind.2018.06.064.
Fazey, I., Fazey, J. A. and Fazey, D. M. (2005) 'Ecology and Society: Learning More Effectively from Experience', Ecology And Society, 10(2). Available at: http://www.ecologyandsociety.org/vol10/iss2/art4/.
Folke, C. (2004) 'Traditional Knowledge in Social-Ecological Systems', Ecology and Society, 9(3). Available at: https://www.jstor.org/stable/26267675.
Foster, R. J. and Harmsen, B. J. (2012) 'A critique of density estimation from camera-trap data', The Journal of Wildlife Management, 76(2), pp. 224–236. doi: 10.1002/jwmg.275.
Fox, H. E. et al. (2006) 'Perceived Barriers to Integrating Social Science and Conservation', Conservation Biology, 20(6), pp. 1817–1820. doi: 10.1111/j.1523-1739.2006.00598.x.
Franzen, M. (2006) 'Evaluating the sustainability of hunting: A comparison of harvest profiles across three Huaorani communities', Environmental Conservation, 33(1), pp. 36–45. doi: 10.1017/S0376892906002712.
Funder, M. et al. (2013) 'Reshaping Conservation: The Social Dynamics of Participatory Monitoring in Tanzania's Community-managed Forests', Conservation and Society, 11(3), p. 218. doi: 10.4103/0972-4923.121011.
Gaidet-Drapier, N. et al. (2006) 'Cost and Efficiency of Large Mammal Census Techniques: Comparison of Methods for a Participatory Approach in a Communal Area, Zimbabwe', Biodiversity & Conservation, 15(2), pp. 735–754. doi: 10.1007/s10531-004-1063-7.

Garrote, G. and Pérez de Ayala, R. (2015) ‘Assessing unverified observation data used for estimating Iberian lynx distribution’, European Journal of Wildlife Research, 61(5), pp. 801–806. doi: 10.1007/s10344-015-0941-5.
Gavin, M. C. (2007) ‘Foraging in the fallows: Hunting patterns across a successional continuum in the Peruvian Amazon’, Biological Conservation, 134(1), pp. 64–72. doi: 10.1016/j.biocon.2006.07.011.
Gavin, M. C., Solomon, J. N. and Blank, S. G. (2010) ‘Measuring and Monitoring Illegal Use of Natural Resources’, Conservation Biology, 24(1), pp. 89–100. doi: 10.1111/j.1523-1739.2009.01387.x.
Geyle, H. M. et al. (2018) ‘Towards meaningful monitoring: A case study of a threatened rodent’, Austral Ecology, pp. 223–236. doi: 10.1111/aec.12667.
Gibson, R.B. (2006)'Beylong the pillars: Sustainability assessment as a framework for effective integration of social economic and ecological considerations in significant decision-making' (2006), Journal of Environmental Assessment Policy and Management, 08(03), pp. 259–280. doi: 10.1142/S1464333206002517.
Gill, D. J. C. et al. (2012) ‘Drivers of Change in Hunter Offtake and Hunting Strategies in Sendje, Equatorial Guinea’, Conservation Biology, 26(6), pp. 1052–1060. doi: 10.1111/j.1523-1739.2012.01876.x.
Giraud, C. et al. (2015) ‘Capitalizing on opportunistic data for monitoring relative abundances of species.’, Biometrics. doi: 10.1111/biom.12431.
Golden, C. D. et al. (2013) ‘Assessing the accuracy of interviewed recall for rare, highly seasonal events: The case of wildlife consumption in Madagascar. Animal Conservation, 16:597-203
Golden, C. D. et al. (2016) ‘Ecosystem services and food security: Assessing inequality at community, household and individual scales’, Environmental Conservation, 43(4), pp. 381–388. doi: 10.1017/S0376892916000163.
Golden, C. D. and Comaroff, J. (2015) ‘Effects of social change on wildlife consumption taboos in northeastern Madagascar’, Ecology and Society, 20(2), pp. art41-10. doi: 10.5751/ES-07589-200241.
González-Marín, R. M. et al. (2017) ‘Regaining the traditional use of wildlife in wetlands on the coastal plain of Veracruz, Mexico: ensuring food security in the face of global

climate change', Regional Environmental Change, 17(5), pp. 1343–1354. doi: 10.1007/s10113-016-0955-x.
Gouwakinnou, G. N. et al. (2011) 'Local knowledge, pattern and diversity of use of Sclerocarya birrea', Journal of Ethnobiology and Ethnomedicine, 7, pp. 1–9. doi: 10.1186/1746-4269-7-8.
Greengrass, E. (2016) 'Commercial hunting to supply urban markets threatens mammalian biodiversity in Sapo National Park, Liberia', Oryx, 50(3), pp. 397–404. doi: 10.1017/S0030605315000095.
Guillera-Arroita, G. (2017) 'Modelling of species distributions, range dynamics and communities under imperfect detection: advances, challenges and opportunities', Ecography, 40(2), pp. 281–295. doi: 10.1111/ecog.02445.
Guillera-Arroita, G. and Lahoz-Monfort, J. J. (2012) 'Designing studies to detect differences in species occupancy: Power analysis under imperfect detection', Methods in Ecology and Evolution, 3(5), pp. 860–869. doi: 10.1111/j.2041-210X.2012.00225.x.
Hastie, T. and Fithian, W. (2013) 'Inference from presence-only data; the ongoing controversy', Ecography, 36(8), pp. 864–867. doi: 10.1111/j.1600-0587.2013.00321.x.
Hatch, S. (2003) 'Statistical power for detecting trends with applications to seabirds monitoring.', Biological Conservation, 111, pp. 317–329.
Hedges, S. (2012) Monitoring Elephant Populations and Assessing Threats: A Manual for Researchers, Managers and Conservationists. India: Universities Press (India) Private Limited.
Hemming, V. et al. (2017) 'A practical guide to structured expert elicitation using the IDEA protocol', Methods in Ecology and Evolution, 9(1), pp. 169–180. doi: 10.1111/2041-210X.12857.
Hemming, V. et al. (2018) Eliciting improved quantitative judgements using the IDEA protocol: A case study in natural resource management, PLoS ONE. doi: 10.1371/journal.pone.0198468.
Hiby, L. and Krishna, M. B. (2001) 'Line transect sampling from a curving path', Biometrics, 57(3), pp. 727–731. doi: 10.1111/j.0006-341X.2001.00727.x.
van der Hoeven, C. A., de Boer, W. F. and Prins, H. H. T. (2004) 'Pooling local expert opinions for estimating mammal densities in tropical rainforests', Journal for Nature

Conservation, 12(4), pp. 193–204. doi: 10.1016/j.jnc.2004.06.003
Hinsley, A., Keane, A., St John, F.A.V., Ibbett, H., Nuno, A (2018). Asking sensitive questions using the unmatched count technique: Applications and guidelines for conservation. Methods in Ecology and Evolution. 10:308-319
Holmern, T. et al. (2015) 'Intraspecific prey choice of bushmeat hunters outside the Serengeti National Park, Tanzania: A preliminary analysis', African Zoology, 41(1), pp. 81–87. doi: 10.3377/1562-7020(2006)41.
Houehanou, T., et al. (2011) 'Valuation of local preferred uses and traditional ecological knowledge in relation to three multipurpose tree species in Benin (West Africa).', Forest Policy and Economics 1, 13, pp. 554–562.
Hockley, N., Jones, J., Andriahajaina, F.B., Manica, A., Ranambitsoa, E.M., Randriamboahary, J.A. (2005) Biodiversity and Conservation. 14: 2795-2806
Howe, E. J. et al. (2017) 'Distance sampling with camera traps', Methods in Ecology and Evolution, 8(11), pp. 1558–1565. doi: 10.1111/2041-210X.12790.
Huntingdon, H. P. (2000) 'Using traditional ecological knowledge in science: methods and applications', Ecological Applications, 10, pp. 1270–1274.
Ibbett, H. and Brittain, S. (2019) 'Conservation publications and their provisions to protect research participants', Conservation Biology.
Ichikawa, M., Hattori, S. and Yasuoka, H. (2016) 'Bushmeat Crisis, Forestry Reforms and Contem- porary Hunting Among Central African Forest Hunters.', in In (V. Reyes-García & A. Pyhälä, eds.) Hunter-gatherers in a Changing World. New-York: Springer, pp. 59–75.
Irwin, A. (2001) Citizen engagement in science and technology policy: a commentary on recent UK experience, Pimbert and Wakeford.
IUCN (2012) IUCN Red List of threatened species. Version 2012.2. Cambridge, UK: Available at: http://www.iucnredlist.org/.
Jachmann, H. (2002) 'Comparison of aerial counts with ground counts for large African herbivores', Journal of Applied Ecology, 39(5), pp. 841–852. doi: 10.1046/j.1365-2664.2002.00752.x.
Jeanmart, P. (1998) Tentative d'élaboration d'un plan de gestion de la chasse villageoise dans la réserve de faune du dja. Rapport eCoFAC—Cameroun (Conservation

et utilisation Ra- tionnelle des Écosystèmes Forestiers d'Afrique Centrale), agreCo, Bruxelles.

Jenkins, R. K. et al. (2011) 'Analysis of patterns of bushmeat consumption reveals extensive exploitation of protected species in eastern Madagascar ', PLoS One. doi: 10.1371/journal.pone.0027570 .

Jolliffe, I. T. and Cadima, J. (2016) 'Principal component analysis: a review and recent developments', Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences, 374(2065), doi: 10.1098/rsta.2015.0202.

Jones, J. P. G. et al. (2008) 'Testing the use of interviews as a tool for monitoring trends in the harvesting of wild species', Journal of Applied Ecology, 45(4), pp. 1205–1212. doi: 10.1111/j.1365-2664.2008.01487.x.

Joseph, L. N. et al. (2006) 'Presence-absence versus abundance data for monitoring threatened species.', Conservation biology : the journal of the Society for Conservation Biology, 20(6), pp. 1679–87. doi: 10.1111/j.1523-1739.2006.00529.x.

Joseph, L. N. et al. (2006) 'Presence – absence versus abundance data for monitoring threatened species', Conservation Biology, 20, pp. 1679–1687.

Jost Robinson, C. A., Daspit, L. L. and Remis, M. J. (2011) 'Multi-faceted approaches to understanding changes in wildlife and livelihoods in a protected area: A conservation case study from the Central African Republic', Environmental Conservation, 38(2), pp. 247–255. doi: 10.1017/S0376892910000949.

Kamgaing, T. O. W., Dzefack, Z. C. B. and Yasuoka, H. (2019) 'Declining Ungulate Populations in an African Rainforest: Evidence From Local Knowledge, Ecological Surveys, and Bushmeat Records', Frontiers in Ecology and Evolution, 7(July), pp. 1–13. doi: 10.3389/fevo.2019.00249.

Karanth, K. . et al. (2011) 'Monitoring carnivore populations at the landscape scale: occupancy modelling of tigers from sign surveys. Journal of Applied Ecology, 48, pp. 1048–1056.

Karlin, M. and De La Paz, G. (2015) 'Using camera-trap technology to improve undergraduate education and citizen-science contributions in wildlife research', The Southwestern Naturalist, 60(2–3), pp. 171–179. doi: 10.1894/swnat-d-14-00005.1.

Kéry, M. and Schmid, H. (2004) 'Monitoring programs need to take into account

imperfect species detectability', Basic and Applied Ecology, 5(1), pp. 65–73. doi: 10.1078/1439-1791-00194.
Kinahan, A. and Bunnefeld, N. (2012) 'Effectiveness and cost efficiency of monitoring mountain nyala in Bale Mountains National Park, Ethiopia', Endangered Species Research, 18, pp. 105–114.
Kleinschroth, F. et al. (2019) 'Road expansion and persistence in forests of the Congo Basin', Nature Sustainability. Springer US, 2(7), pp. 628–634. doi: 10.1038/s41893-019-0310-6.
Koehler, J. (1993) 'The influence of prior beliefs on scientific judgments of evidence quality', Organizational Behavior and Human Decision Processes, 56, p. 28–55.
Koppert, G. J. et al. (1996) 'onsommation alimentaire dans trois populations forestières de la région côtière du Cameroon: Yassa, Mvae et Bakola.', in In L'alimentation en forêt tropicale, interactions bioculturelles et perspectives de développement. Eds. C. M. Hladik, A. Hladik, H. Pagezy, O. F. Linares, G. J. A. Koppert, & A. Froment, UNESCO edition. Vol. 1. Paris: UNESCO, pp. 477–496.
Kumpel, N. F. et al. (2009) 'Assessing Sustainability at Multiple Scales in a Rotational Bushmeat Hunting System', Conservation Biology, 24(3), pp. 861–871. doi: 10.1111/j.1523-1739.2010.01505.x.
Kümpel, N. F. et al. (2009) 'Trapper profiles and strategies: Insights into sustainability from hunter behaviour', Animal Conservation, 12(6), pp. 531–539. doi: 10.1111/j.1469-1795.2009.00279.x.
Larrucea, E. S. et al. (2007) 'Censusing Bobcats Using Remote Cameras', Western North American Naturalist, 67(4), pp. 538–548. doi: 10.3398/1527-0904(2007)67[538:cburc]2.0.co;2.
Leeney, R. H. and Poncelet, P. (2013) 'Using fishers' ecological knowledge to assess the status and cultural importance of sawfish in Guinea-Bissau', Aquatic Conservation: Marine and Freshwater Ecosystems, 25(3), pp. 411–430. doi: 10.1002/aqc.2419.
Legg, C. J. and Nagy, L. (2006) 'Why most conservation monitoring is, but need not be, a waste of time', Journal of Environmental Management, 78(2), pp. 194–199. doi: 10.1016/j.jenvman.2005.04.016.
Levi, T. et al. (2011) 'Spatial tools for modeling the sustainability of subsistence hunting

in tropical forests', Ecological Applications. doi: 10.1890/10-0375.1.
Lewis, J. (2007) 'Enabling forest people to map their resources & monitor illegal logging in Cameroon ', Before Farming: the archaeology and anthropology of hunter-gatherers . Available at: http://discovery.ucl.ac.uk/1361446/.
Lindenmayer, D. B. et al. (2012) 'Improving biodiversity monitoring', Austral Ecology, 37(September 2011), pp. 285–294. doi: 10.1111/j.1442-9993.2011.02314.x.
Lindenmayer, D. B. and Likens, G. E. (2009) 'Adaptive monitoring: a new paradigm for long-term research and monitoring', Trends in Ecology & Evolution, 24(9), pp. 482–486. doi: 10.1016/j.tree.2009.03.005.
Lindsey, P. A. et al. (2011) 'Dynamics and underlying causes of illegal bushmeat trade in Zimbabwe', Oryx, 45(1), pp. 84–95. doi: 10.1017/S0030605310001274.
Ling, S. and Milner-Gulland, E. J. (2006) 'Assessment of the sustainability of bushmeat hunting based on dynamic bioeconomic models', Conservation Biology, 20(4), pp. 1294–1299. doi: 10.1111/j.1523-1739.2006.00414.x.
Linkie, M. et al. (2013) 'Cryptic mammals caught on camera: Assessing the utility of range wide camera trap data for conserving the endangered Asian tapir', Biological Conservation. Elsevier Ltd, 162, pp. 107–115. doi: 10.1016/j.biocon.2013.03.028.
Luiselli, L. et al. (2017) 'Rehashing bushmeat – Interview campaigns reveal some controversial issues about the bushmeat trade dynamics in Nigeria', Revue d'Ecologie (La Terre et la Vie), 72(1), pp. 3–18.
Luz, A. C. et al. (2017) 'Continuity and change in hunting behaviour among contemporary indigenous peoples', Biological Conservation, 209, pp. 17–26. doi: 10.1016/j.biocon.2017.02.002.
Lyra-Jorge, M. C. et al. (2008) 'Comparing methods for sampling large- and medium-sized mammals: camera traps and track plots', European Journal of Wildlife Research, 54(4), pp. 739–744. doi: 10.1007/s10344-008-0205-8.
MacKenzie, C. A., Chapman, C. A. and Sengupta, R. (2012) 'Spatial patterns of illegal resource extraction in Kibale National Park, Uganda', Environmental Conservation, 39(01), pp. 38–50. doi: 10.1017/S0376892911000282.
MacKenzie, D.I (2005) 'What are the issues with presence-absence data for wildlife managers?', Journal of Wildlife Management, 69(3), pp. 849–860. doi: 10.2193/0022-

541X(2005)069.
Mackenzie, D.I et al. (2011) 'An integrated model of habitat and species occurrence dynamics', Methods in Ecology and Evolution, 2, pp. 612–622.
Mackenzie, D.I. et al. (2002) 'Estimating site occupancy rates when detection probabilities are less than one', Ecology, 83(8), pp. 2248–2255.
Mackenzie, D.I. and Bailey, L.L. (2004) 'Assessing the fit of site-occupancy models', Journal of Agricultural, Biological, and Environmental Statistics, 9(3), pp. 300–318. doi: 10.1198/108571104X3361.
Mackenzie, D.I. and Nichols, J. D. (2004) 'Occupancy as a surrogate for abundance estimation', Animal Biodiversity and Conservation, 1(27), pp. 461–467.
Mackenzie, D. I. and Royle, J. A. (2005) 'Designing occupancy studies: general advice and allocating survey effort', Journal of Applied Ecology, 42(6), pp. 1105–1114. doi: 10.1111/j.1365-2664.2005.01098.x.
Martin, T. G. et al. (2012) 'Eliciting Expert Knowledge in Conservation Science', Conservation Biology, 26(1), pp. 29–38. doi: 10.1111/j.1523-1739.2011.01806.x.
Martínez-Cruz, A. L., Juárez-Torres, M. and Guerrero, S. (2017) 'Assessing Impacts From Climate Change on Local Social-ecological Systems in Contexts Where Information is Lacking: An Expert Elicitation in the Bolivian Altiplano', Ecological Economics, 137, pp. 70–82. doi: 10.1016/j.ecolecon.2017.03.003.
Martínez-Martí, C. et al. (2016) 'Integrating occurrence and detectability patterns based on interview data: a case study for threatened mammals in Equatorial Guinea', Scientific Reports, 6, p. 33838. doi: 10.1038/srep33838.
Martinez -Marti, C. (2011) The leopard (Panthera pardus) and the golden cat (Caracal aurata) in Equatoria; Guinea: A national assessment of status, distribution and threat.
Mayor, P. et al. (2019) 'Assessing the minimum sampling effort to reliably monitor wild meat trade in urban markets', Frontiers in Ecology and Evolution, 7(May), p. 180. doi: 10.3389/FEVO.2019.00180.
Mcbride, M., Fidler, F. and Burgman, M. (2012) 'Evaluating the accuracy and calibration of expert predictions under uncertainty: predicting the outcomes of ecological research', Diversity and Distributions, 18(8), pp. 782–794. doi: 10.1111/j.1472-4642.2012.00884.x.

McClintock, B. T. et al. (2010) 'Experimental investigation of observation error in anuran call surveys', Journal of Wildlife Management, 74, pp. 1882–1893.
McCrea, R and Morgan, B. J.(2015) Analysis of Capture-Recapture Data. Edited by N. Keiding et al. London & New York: Taylor & Francis Group, LLC.
McDonald-Madden, E. et al. (2010) 'Monitoring does not always count', Trends in Ecology & Evolution, 25(10), pp. 547–550. doi: 10.1016/j.tree.2010.07.002.
McKelvey, K., Aubry, K. and Schwartz, M. (2008) 'Using anecdotal occurrence data for rare or elusive species: the illusion of real- ity and a call for evidentiary standards', BioScience, 58(6), pp. 549– 555.
McNamara, J. et al. (2016) 'Characterising wildlife trade market supply-demand dynamics', PLoS ONE, 11(9), pp. 1–18. doi: 10.1371/journal.pone.0162972.
Meek, P. et al. (2016) 'Are we getting the full picture? Animal responses to camera traps and implications for predator studies ', Ecol Evol pp. 3216–3225. doi: 10.1002/ece3.2111 .
Meijaard, E. et al. (2011) 'Why Don't We Ask? A Complementary Method for Assessing the Status of Great Apes', PLoS ONE. Edited by S. Turvey, 6(3), p. e18008. doi: 10.1371/journal.pone.0018008.
De Merode, E., Homewood, K. M. and Cowlishaw, G. (2004) 'The value of bushmeat and other wild foods to rural households living in extreme poverty in Democratic Republic of Congo', Biological Conservation, 118(5), pp. 573–581. doi: 10.1016/j.biocon.2003.10.005.
Meyer, M. A. and Booker, J. M. (1990) Eliciting and analyzing expert judgment: a practical guide. Washington, DC. Miller.
Mgawe, P. et al. (2012) 'Factors affecting bushmeat consumption in the Katavi-Rukwa ecosystem of Tanzania', Tropical Conservation Science, 5(4), pp. 446–462. doi: 10.1177/194008291200500404.
Miller, D. A. W. et al. (2011) 'Improving occupancy estimation when two types of observational error occur: non-detection and species misidentification', Ecology, 92(7), pp. 1422–1428. doi: 10.1890/10-1396.1.
Miller, D. A. W. et al. (2012) 'Joint estimation of habitat dynamics and species interactions: disturbance reduces co-occurrence of non-native predators with an

endangered toad', Journal of Animal Ecology, 81(6), pp. 1288–1297. doi: 10.1111/j.1365-2656.2012.02001.x.
Miller, D. A. W. et al. (2015) 'Performance of species occurrence estimators when basic assumptions are not met: a test using field data where true occupancy status is known', Methods in Ecology and Evolution. 6(5), pp. 557–565. doi: 10.1111/2041-210X.12342.
Milner-Gulland, E. . (2006) 'Developing a framework for assessing the sustainability of bushmeat hunting', in Ed. D. Lavigne, Wildlife Conservation: In Pursuit of Ecological Sustainability.
Milner-Gulland, E.J and Rowcliffe, M. (2007) Wildlife Conservation and Sustainable Use: a Handbook of Techniques. Oxford University Press.
Milner-Gulland, E. J. (2012) 'Interactions between human behaviour and ecological systems', Philosophical Transactions of the Royal Society B: Biological Sciences, 367(1586), pp. 270–278. doi: 10.1098/rstb.2011.0175.
Milner-Gulland, E. J., Akcakaya, H. R. and Akçakaya, H. R. (2001) 'Sustainability indices for exploited populations', Trends in Ecology & Evolution, 16(12), pp. 686–692. doi: 10.1016/S0169-5347(01)02278-9.
Milner-Gulland, E. J. and Bennett, E. L. (2003) 'Wild meat: the bigger picture', Trends in Ecology & Evolution, 18(7), pp. 351–357. doi: 10.1016/S0169-5347(03)00123-X.
Mohd-Azlan, J., Belant, J. L. and Meilaard, E. (2013) 'Can secondary information inform about population trends of carnivores in Borneo?', The Raffles Bulletin of Zoology, (28), pp. 1–8. Available at: http://espace.library.uq.edu.au/view/UQ:319058/UQ319058_OA.pdf (Accessed: 7 December 2015).
Molinari-Jobin, A. et al. (2012) 'Monitoring in the presence of species mis-identification: the case of the Eurasian lynx in the Alps.', Animal Conservation Conserv, 15, pp. 266–273.
Moller, H. et al. (2004) 'Combining science and traditional ecological knowledge: monitoring populations for co-management', Ecology and Society, 9(3), p. 2. doi: 10.1016/j.anbehav.2004.02.016.
Moon, K. and Blackman, D. (2014) 'A Guide to Understanding Social Science Research for Natural Scientists', Conservation Biology, 28(5), pp. 1167–1177. doi:

10.1111/cobi.12326.
Morgan, M. G. (2014) 'Use (and abuse) of expert elicitation in support of decision making for public policy', Proceedings of the National Academy of Sciences, 111(20), pp. 7176–7184. doi: 10.1073/pnas.1319946111.
Morrison, M et al. (2008) 'Inventory and Monitoring Studies', in Wildlife Study Design. Springer Series on Environmental Management. New York, NY: Springer, pp. 267–312. doi: 10.1007/978-0-387-75528-1_7.
Morsello, C. et al. (2015) 'Cultural attitudes are stronger predictors of bushmeat consumption and preference than economic factors among urban amazonians from brazil and colombia', Ecology and Society, 20(4). doi: 10.5751/ES-07771-200421.
Msoffe, F. U. et al. (2009) 'Participatory wildlife surveys in communal lands: a case study from Simanjiro, Tanzania', African Journal of Ecology, 48(3), pp. 727–735. doi: 10.1111/j.1365-2028.2009.01170.x.
Muchaal, P. K. and Ngandjui, G. (1999) 'Impact of village hunting on wildlife populations in the western Dja reserve, Cameroon', Conservation Biology, 13(2), pp. 385–396. doi: 10.1046/j.1523-1739.1999.013002385.x.
Munari, D. P., Keller, C. and Venticinque, E. M. (2011) 'An evaluation of field techniques for monitoring terrestrial mammal populations in Amazonia', Mammalian Biology, 76(4), pp. 401–408. doi: 10.1016/j.mambio.2011.02.007.
Murphy, A. (1993) 'What is a good forecast? An essay on the nature of goodness in weather forecasting.', Am Meterol Soc, 8, pp. 281–293.
Nakashima, Y., Fukasawa, K. and Samejima, H. (2018) 'Estimating animal density without individual recognition using information derivable exclusively from camera traps', Journal of Applied Ecology, 55(2), pp. 735–744. doi: 10.1111/1365-2664.13059.
Nasi, R., Taber, A. and van Vliet, N. (2011) 'Empty forests, empty stomachs? Bushmeat and livelihoods in the Congo and Amazon Basins', International Forestry Review, 13(3), pp. 355–368. doi: 10.1505/146554811798293872.
Nasi, R. and van Vliet, N. (2011) 'Measuring the abundance of wildlife populations in Central African logging concessions', Unasylva, 62(238), pp. 49–55.
Neba, A. (1999) Modern Geography of the Republic of Cameroon. Bamenda: Neba Publishers.

Nelitz, M. A. et al. (2015) 'Addressing complexity and uncertainty: conceptual models and expert judgments applied to migratory birds in the oil sands of Canada', Ecology and Society, 20(4), p. art4. doi: 10.5751/ES-07906-200404.
Newing, H. (2011) Conducting Research In Conservation: A social science perspective, Journal of Chemical Information and Modeling. doi: 10.1017/CBO9781107415324.004.
Newing, H. and St John, F. A. V (2013) 'Wildlife consumption and recall accuracy - but is it recall of hunting, of cooking or of eating?', Animal Conservation, 16(6), pp. 606–607. doi: 10.1111/acv.12092.
Ngnegueu, P. and Fotso, R. (1996) Chasse villagoise et consequences pour la conservation de la biodiversite dans la reserve de biosphere du Dja. ECOFAC: Yaounde.
Nickerson, R. (1998) 'Confirmation bias: a ubiquitous phenomenon in many guises.', Review of General Psychology ., 2, pp. 175–220.
Nielsen, M. R. and Meilby, H. (2015) 'Hunting and trading bushmeat in the Kilombero Valley, Tanzania: Motivations, cost-benefit ratios and meat prices', Environmental Conservation, 42(1), pp. 61–72. doi: 10.1017/S0376892914000198.
Nuno, A., Bunnefeld, N. and Milner-Gulland, E. J. (2013) 'Matching observations and reality: Using simulation models to improve monitoring under uncertainty in the Serengeti', Journal of Applied Ecology, 50, pp. 488–498. doi: 10.1111/1365-2664.12051.
Nuno, A., Milner-Gulland, E. J. and Bunnefeld, N. (2014) 'Detecting abundance trends under uncertainty: the influence of budget, observation error and environmental change', Animal Conservation, 18, p. n/a-n/a. doi: 10.1111/acv.12175.
Nuno, A. and St John, F. A. V (2014) 'How to ask sensitive questions in conservation: A review of specialized questioning techniques', Biological Conservation. Elsevier Ltd, 189, pp. 5–15. doi: 10.1016/j.biocon.2014.09.047.
NVivo (2011) 'Qualitative data analysis software' QSR International Pty Ltd. .
O'Brien (2000) Making better environmental decisions: an alternative to risk assessment. Cambridge, MA.: MIT Press.
O'Donnell, K. P., Pajaro, M. G. and Vincent, A. C. J. (2010) 'How does the accuracy of fisher knowledge affect seahorse conservation status?', Animal Conservation, 13(6), pp.

526–533. doi: 10.1111/j.1469-1795.2010.00377.x.
Okouyi, J. (2006) Savoirs locaux et outils modernes cynégétiques: développement de la filière commerciale de viande de brousse à Makokou (Gabon). L'Université d'Orléans, Orléans, France.
Oppenheimer, M., Little, C. M. and Cooke, R. M. (2016) 'Expert judgement and uncertainty quantification for climate change', Nature Climate Change. Nature Publishing Group, 6(5), pp. 445–451. doi: 10.1038/nclimate2959.
Parry, L. and Peres, C. (2015) 'Evaluating the use of local ecological knowledge to monitor hunted tropical-forest wildlife over large spatial scales', Ecology and Society, 20(3). Available at: http://www.ecologyandsociety.org/vol20/iss3/art15/ES-2015-7601.pdf.
Parry, L. and Peres, C. A. (2015) 'Evaluating the use of local ecological knowledge to monitor hunted tropical-forest wildlife over large spatial scales', Ecology and Society, 20(3). Available at: http://www.ecologyandsociety.org/vol20/iss3/art15/ES-2015-7601.pdf.
Pasgaard, M. et al. (2017) 'Unfolding scientific expertise and security in the changing governance of Ecosystem Services', Geoforum. Elsevier Ltd, 84, pp. 354–367. doi: 10.1016/j.geoforum.2017.02.001.
Peres, C. A. (2000) 'Evaluating the impact and sustainability of subsistence hunting at multiple Amazonian forest sites.', in J. Robinson, and E. Bennett, editors. Hunting for Sustainability in Tropical Forests. New York, USA: Columbia University Press, pp. 31–57.
Petracca, L. S. et al. (2017) 'Robust inference on large-scale species habitat use with interview data: The status of jaguars outside protected areas in Central America', Journal of Applied Ecology, (January), pp. 1–12. doi: 10.1111/1365-2664.12972.
Petracca, L. S. and Frair, J. L. (2017) 'When methodological flaws limit inference: A response to Caruso et al.', Oryx, 51(2), p. 208. doi: 10.1017/S0030605316000788.
Petracca, L. S., Ramírez-Bravo, O. E. and Hernández-Santín, L. (2013) 'Occupancy estimation of jaguar Panthera onca to assess the value of east-central Mexico as a jaguar corridor', Oryx. Cambridge University Press, 48(01), pp. 133–140. doi: 10.1017/S0030605313000069.
Phillips, S. J. et al. (2009) 'Sample selection bias and presence-only distribution models:

Implications for background and pseudo-absence data', Ecological Applications, 19(1), pp. 181–197. doi: 10.1890/07-2153.1.
Phillips, S. J. and Elith, J. (2013) 'On estimating probability of presence from use-availability or presence-background data', Ecology, 94(6), pp. 1409–1419. doi: 10.1890/12-1520.1.
Pillay, R. et al. (2011) 'Patterns of spatiotemporal change in large mammal distribution and abundance in the southern Western Ghats, India 1', Biological Conservation, 144, pp. 1567– 1576.
Pillay, R. et al. (2014) 'Accounting for false positives improves estimates of occupancy from key informant interviews', Diversity and Distributions, 20(2), pp. 223–235. doi: 10.1111/ddi.12151.
Pollock, K. H. et al. (2002) 'Large scale wildlife monitoring studies: Statistical methods for design and analysis', in Environmetrics. doi: 10.1002/env.514.
Popova, U. (no date) 'Conservation, Traditional Knowledge, and Indigenous Peoples ', American Behavioral Scientist , pp. 197–214. doi: 10.1177/0002764213495043 .
Poulsen, J. R. et al. (2009) 'Bushmeat supply and consumption in a tropical logging concession in northern Congo', Conservation Biology, 23(6), pp. 1597–1608. doi: 10.1111/j.1523-1739.2009.01251.x.
Puri, M. et al. (2015) 'Multiscale distribution models for conserving widespread species: the case of sloth bear Melursus ursinus in India', Diversity and Distributions. Edited by B. Wintle, 21(9), pp. 1087–1100. doi: 10.1111/ddi.12335.
Refsnider, J. M., Mitchell, T. S. and Warner, D. A. (2011) 'A generalized method to determine detectability of rare and cryptic species using the ornate box turtle as a model', Ecology, Evolution and Organismal Biology, 6. doi: 10.1002/wsb.14 Original.
Republic of Cameroon (2012) 'National biodiversity strategy and action plan version II (NBSAP II)
Rich, L. N. et al. (2017) 'Assessing global patterns in mammalian carnivore occupancy and richness by integrating local camera trap surveys', Global Ecology and Biogeography, 26(8), pp. 918–929. doi: 10.1111/geb.12600.
Rist, J. et al. (2008) 'Evaluating measures of hunting effort in a bushmeat system', Biological Conservation, 141(8), pp. 2086–2099. doi: 10.1016/j.biocon.2008.06.005.

Rist, L., Shaanker, R. U. and Ghazoul, J. (2010) 'The Use of Traditional Ecological Knowledge in Forest Management : an Example from India', Ecology And Society, 15(1), p. 3.

Robinson, J. G. and Bennett, E. L. (2004) 'Having your wildlife and eating it too: an analysis of hunting sustainability across tropical ecosystems', Animal Conservation, 7(4), pp. 397–408. doi: 10.1017/S1367943004001532.

Robinson, J. G. and Bodmer, R. E. (1999) 'Towards wildlife management in tropical forests.', Journal of Wildlife Management, 63, pp. 1–13.

Robinson, J. G. and Redford, K. H. (1991) 'Sustainable harvest of Neo-tropical forest animals', in Neotropical wildlife use and conservation. Chicago: University of Chicago Press, pp. 415–430.

Rosin, C. and Poulsen, J. R. (2016) 'Hunting-induced defaunation drives increased seed predation and decreased seedling establishment of commercially important tree species in an Afrotropical forest', Forest Ecology and Management, 382, pp. 206–213. doi: 10.1016/j.foreco.2016.10.016.

Rovero, F. and Marshall, A. R. (2009) 'Camera trapping photographic rate as an index of density in forest ungulates', Journal of Applied Ecology, 46(5), pp. 1011–1017. doi: 10.1111/j.1365-2664.2009.01705.x.

Rowcliffe, J. and Carbone, C. (2008) 'Surveys using camera traps: are we looking to a brighter future?', Animal Conservation, 11.

Rowcliffe, M. et al. (2008) 'Estimating animal density using camera traps without the need for individual recognition', Journal of Applied Ecology, 45, pp. 1228–1236. doi: 10.1111/j.1365-2664.2008.0.

Rowcliffe, M., Milner-Gulland, E. J. and Cowlishaw, G. (2005) 'Do bushmeat consumers have other fish to fry?', Trends in Ecology and Evolution, 20(6 SPEC. ISS.), pp. 274–276. doi: 10.1016/j.tree.2005.03.007.

Royle, J.A. and Nichols, J.D.(2003) 'Estimating abundance from repeated presence-absense', Ecology, 84(3), pp. 777–790.

Royle, J. A. (2006) 'Site occupancy models with heterogeneous detection probabilities.', Biometrics, 62(1), pp. 97–102. doi: 10.1111/j.1541-0420.2005.00439.x.

Royle, J. A. and Link, W. A. (2006) 'Generalized site occupancy models allowing for false

positive and false negative errors. Ecology, 87, 835–841'.
Royle, J. A. and Nichols, J. D. (2003) 'Estimating abundance from repeated presence-absence data or point counts', Ecology, 84(3), pp. 777–790.
Saska, P. et al. (2013) 'Temperature effects on pitfall catches of epigeal arthropods: A model and method for bias correction', Journal of Applied Ecology, 50(1), pp. 181–189. doi: 10.1111/1365-2664.12023.
Schulte-Herbrüggen, B. et al. (2013) 'The Importance of Bushmeat in the Livelihoods of West African Cash-Crop Farmers Living in a Faunally-Depleted Landscape', PLoS ONE. 8(8), p. e72807. doi: 10.1371/journal.pone.0072807.
Seber, G. A. F. (1986) 'A Review of Estimating Animal Abundance', Biometrics, 42(2), p. 267. doi: 10.2307/2531049.
Service, C. N. et al. (2014) 'Indigenous Knowledge and Science Unite to Reveal Spatial and Temporal Dimensions of Distributional Shift in Wildlife of Conservation Concern', PLoS ONE. 9(7), p. e101595. doi: 10.1371/journal.pone.0101595.
Sethi, G. et al. (2005) 'Fishery management under multiple uncertainty' Journal of Environmental Economics and Management, 50, pp. 300–318.
Shea, C. et al. (2011) 'Misidentification of freshwater mussel species (Bivalvia: Unionidae): contributing factors, management implications, and potential solutions', Bulletin of the North American Benthological Society, 30, pp. 446–458.
Shefferson, R. P. et al. (2001) 'Estimating dormancy and survival of a rare herbaceous perennial using mark-recapture models', Ecology, 82(1), pp. 145–156. doi: 10.1890/0012-9658(2001)082[0145:EDASOA]2.0.CO;2.
da Silva Neto, B. C. et al. (2017) 'Assessment of the hunting of mammals using local ecological knowledge: an example from the Brazilian semiarid region', Environment, Development and Sustainability, 19(5), pp. 1795–1813. doi: 10.1007/s10668-016-9827-2.
Silveira, L., Jácomo, A. T. A. and Diniz-Filho, J. A. F. (2003) 'Camera trap, line transect census and track surveys: a comparative evaluation', Biological Conservation, 114(3), pp. 351–355. doi: 10.1016/S0006-3207(03)00063-6.
Singh, N. J. and Milner-Gulland, E. J. (2011) 'Monitoring ungulates in Central Asia: current constraints and future potential', Oryx, 45(01), pp. 38–49. doi:

10.1017/S0030605310000839.
Sirén, A. H. et al. (2012) 'Distance Friction and the Cost of Hunting in Tropical Forests', Ssrn. doi: 10.2139/ssrn.2188389.
Smallwood, S. and Fitzhugh, L. (1995) 'A track count for estimating mountain lio Felis concolor californica population trend', Biological Conservation, 71.
Smith, E. A. and Wishnie, M. (2000) 'C Onservation and S Ubsistence in small scale societies', Forestry, pp. 493–524. doi: 10.1146/annurev.anthro.29.1.493.
Solly, H. (2004) Bushmeat hunters and secondary traders: making the distinction for livelihood improvement.
Southwell, D. M. et al. (2019) 'Spatially explicit power analysis for detecting occupancy trends for multiple species', Ecological Applications, 0(0), p. e01950. doi: 10.1002/eap.1950.
Speirs-Bridge, A. et al. (2010) 'Reducing over-confidence in the interval judgements of experts', Risk Analysis, 30, pp. 512–523.
St John, F. A. V et al. (2016) 'Research ethics: Assuring anonymity at the individual level may not be sufficient to protect research participants from harm', Biological Conservation, 196, pp. 208–209. doi: 10.1016/j.biocon.2016.01.025.
Starkey, M. (2004) Commerce and subsistence: the hunting, sale and consumption of bushmeat in Gabon. University of Cambridge.
Stephens, P. A. et al. (2015) 'Management by proxy? The use of indices in applied ecology.', Journal of Applied Ecology, 52, pp. 1–6.
Stern, P. C. and Fineberg, H. (1996) Understanding risk: informing decisions in a democratic society. Washington, DC. Sternberg: Press, National Academy.
Van Strien, A. Van Swaay, C. and Termaat, T. (2013) 'Opportunistic citizen science data of animal species produce reliable estimates of distribution trends if analysed with occupancy models', Journal of Applied Ecology, 50(6), pp. 1450–1458. doi: 10.1111/1365-2664.12158.
Surowiecki, J. (2004) The Wisdom of Crowds. New York: Random House.
Sutherland, W. (2006) 'Predicting the ecological consequences of environmental change: a review of the methods', Journal of Applied Ecology, 43, pp. 599–616.
Sutherland, W. J. et al. (2014) 'How can local and traditional knowledge be effectively

incorporated into international assessments?', Oryx, 48(1), pp. 1–2. doi: 10.1017/S0030605313001543.
Swan, N., Barlow, J. and Parry, L. (2017) 'Expert elicitation as a method for exploring illegal harvest and trade of wild meat over large spatial scales', Oryx, 51(2), pp. 298–304. doi: 10.1017/S0030605315001167.
Tagg, N. et al. (2018) 'A zoo-led study of the great ape bushmeat commodity chain in Cameroon', International Zoo Yearbook, pp. 1–12. doi: 10.1111/izy.12175.
Taylor, G. et al. (2015) 'Synthesising bushmeat research effort in West and Central Africa - a new regional database', Biological Conservation, 181, pp. 199–2205. doi: doi.org/10.1016/j.biocon.2014.11.001.
De Thoisy, B., Brosse, S. and Dubois, M. . (2008) 'Assessment of large-vertebrate species richness and relative abundance in neotropical forest using line-transect censuses: what is the minimal effort required?', Biodiversity Conservation, 17, pp. 2627–2644.
Tracey, J. P., Fleming, P. J. and Melville, G. J. (2005) 'Does variable probability of detection compromise the use of indices in aerial surveys of medium-sized mammals?', Wildlife Research, 32, pp. 245–252.
Turvey, S. T. et al. (2013) 'Can local ecological knowledge be used to assess status and extinction drivers in a threatened freshwater cetacean?', Biological Conservation, 157(June), pp. 352–360. doi: 10.1016/j.biocon.2012.07.016.
Turvey, S. T. et al. (2014) 'Is local ecological knowledge a useful conservation tool for small mammals in a Caribbean multicultural landscape?', Biological Conservation, 169, pp. 189–197. doi: 10.1016/j.biocon.2013.11.018.
Turvey, S. T. et al. (2015) 'Interview-based sighting histories can inform regional conservation prioritization for highly threatened cryptic species', Journal of Applied Ecology. Edited by M. Hayward, 52(2), pp. 422–433. doi: 10.1111/1365-2664.12382.
Turvey, S. T., Barrett, L. A. and Yujiang, H. (2010) 'Rapidly Shifting Baselines in Yangtze Fishing Communities and Local Memory of Extinct Species', Conservation Biology, 24(3), pp. 778–787. doi: 10.1111/j.1523-1739.2009.01395.x.
UNESCO (2018) State of conservation. Dja Faunal Reserve (Cameroon).
van Vliet, N. (2011) Livelihood alternatives for the unsustainable use of bushmeat.

Van Vliet, N. et al. (2015) 'Ride, shoot, and call: wildlife use among contemporary urban hunters in Trampas Fronteiras, Brazilian Amazon', Ecology and Society, 20(3), p. art8. doi: 10.5751/ES-07506-200308.
van Vliet, N., Fa, J. E. and Nasi, R. (2015) 'Managing hunting under uncertainty: from one-off ecological indicators to resilience approaches in assessing the sustainability of bushmeat hunting', Ecology and Society 20(3):, 20(3), p. 7. Available at: http://dx.doi.org/10.5751/.
van Vliet, N. and Nasi, R. (2008) 'Hunting for Livelihood in Northeast Gabon : Patterns , Evolution , and Sustainability', Ecology And Society, 13(2), p. 33. doi: 10.2193/2007-149.
Wegge, P., Pokheral, C. P. and Jnawali, S. R. (2004) 'Effects of trapping effort and trap shyness on estimates of tiger abundance from camera trap studies', Animal Conservation, 7(3), pp. 251–256. doi: 10.1017/S1367943004001441.
Wilkie, D. S. et al. (2016) 'Eating and conserving bushmeat in Africa', African Journal of Ecology, 54(4), pp. 402–414. doi: 10.1111/aje.12392.
Wilkie, D. S. and Carpenter, J. F. (1999) 'Bushmeat hunting in the Congo Basin: An assessment of impacts and options for mitigation', Biodiversity and Conservation, 8(7), pp. 927–955. doi: 10.1023/A:1008877309871.
Willcox, A. S. and Nambu, D. M. (2007) 'Wildlife hunting practices and bushmeat dynamics of the Banyangi and Mbo people of Southwestern Cameroon', Biological Conservation, 134(2), pp. 251–261. doi: 10.1016/j.biocon.2006.08.016.
Willcox, A.S et al. (2019) Ealuating methods for detecting and monitoring pangolin populations Global Ecology and Conservatio. 17:1-25
Williams, B. K., Nichols, J. D. and Conroy, M. J. (2002) Analysis and management of animal populations. New York: Academic Press, San Diego, CA, USA.
Witmer, G. W. (2005) 'Wildlife population monitoring: some practical considerations', Wildlife Research, 32, pp. 259–263. doi: 10.1071/WR04003.
World Bank (2013) 'Country profile guide country profile guide', 735(Ii), p. 2013.
Wright, J. H. and Priston, N. E. C. (2010) 'Hunting and trapping in Lebialem division, Cameroon: Bushmeat harvesting practices and human reliance', Endangered Species Research, 11(1), pp. 1–12. doi: 10.3354/esr00244.

Yasuoka, H. et al. (2015) 'Changes in the composition of hunting catches in southeastern cameroon: A promising approach for collaborative wildlife management between ecologists and local hunters', Ecology and Society, 20(4). doi: 10.5751/ES-08041-200425.
Yasuoka, H. and Mono, A. S. (2006) 'The sustainability of duiker (cephalophus spp.) hunting for the baka hunter-gatherers in southeastern cameroon', African Study Monographs, 33 (Supplements), pp. 95–120. doi: 10.1017/S0030605302000443.
Yearley, S. (2000) 'Making systematic sense of public discontents with expert knowledge: two analytical approaches and a case study.', Public Understanding of Science, 9, pp. 105–122.
Yoccoz, N. G., Nichols, J. D. and Boulinier, T. (2001) 'Monitoring of biological diversity in space and time', Trends in Ecology & Evolution, 16(8), pp. 446–453. doi: 10.1016/S0169-5347(01)02205-4.
Zeller, K. A. et al. (2011) 'Integrating occupancy modeling and interview data for corridor identification: A case study for jaguars in Nicaragua', Biological Conservation. doi: 10.1016/j.biocon.2010.12.003.
Ziegler, S. et al. (2015) 'Mapping bushmeat hunting pressure in Central Africa Accepted : Biotropica', Biotropica, 0(November), pp. 1–8.
ZSL (2018) 'Cameroon Annual Report Letter from the Country Director'.
Zurlini, G. et al. (2006) 'Disturbance patterns in a socio-ecological system at multiple scales', Ecological Complexity, 3(2), pp. 119–128. doi: 10.1016/j.ecocom.2005.11.002.