Human-wildlife coexistence at Gola Rainforest National Park, Sierra Leone

Submitted by Tanya Payne

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I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

Tanya Payne

Signature: ………………………………........…

Thesis abstract

In light of global human population growth and increased spatial overlap between human and wildlife populations, effective conservation must increasingly incorporate the patterns of and threats to human-wildlife coexistence. Gola Rainforest National Park (GRNP) is the largest area of the Upper Guinean Forests of West Africa remaining in Sierra Leone and is of vital conservation importance. However, there has been limited research to examine the interactions between wildlife and the human populations that inhabit this landscape. This thesis aims to inform conservation management strategies at GRNP to mitigate the potential negative impacts of interactions between humans and wildlife.

Firstly, in light of investment in the cocoa industry at GRNP, a cross-disciplinary approach was used to investigate wildlife crop foraging at GRNP, a major conservation and development concern. Semi structured interviews (n=71) and participatory risk mapping were used to determine the perceptions of local farmers. It was found that wildlife cocoa foraging was the main concern of GRNP farmers, with other factors such as climate of less importance. To compare perceptions to measured crop damage by three major groups, western chimpanzees, monkeys and squirrels, crop foraging traces were counted at 24 plantations. Monkeys were found to cause more damage than chimpanzees and squirrels, yet tolerance towards chimpanzees was disproportionately negative, suggesting the need for specific knowledge sharing amongst farmers to protect this critically endangered species. Investigation of the spatial characteristics and crop defence strategies that impact the susceptibility of plantations to crop foraging revealed variation between species groups in the most appropriate plantation management initiatives, including land use planning. A local cocoa farming and development initiative was also evaluated, with member farmers found to experience reduced crop foraging and show increased acceptance of wildlife. Our findings suggest community engagement is vital to the success of local conservation.

Secondly, the presence and distribution of wildlife across GRNP and the impacts of proximity to human infrastructure and activities, including hunting, deforestation and agriculture, were explored. A camera trap survey conducted over 13 months resulted in detections of 26 medium to large bodied size mammals, forming the most recent assessment of the GRNP mammal community. Detected species included five globally threatened species, including critically endangered western chimpanzees, and one previously unconfirmed at the national park, giant forest hog. Species richness was found to be similar within two protected forest blocks and in the surrounding community-managed multi-use landscape. Variation between camera trap sites in ecological characteristics and proximity to anthropogenic infrastructure (roads and villages) showed no impact on site species richness, but an occupancy modelling approach revealed different responses between the 11 most detected species. The occupancy of six species was impacted by anthropogenic covariates, highlighting the varied abilities of species to occur in human-impacted environments. This study provides a vital contribution to conservation planning and management, including land use planning and prioritisation of species for continued monitoring in light of changing threats at GRNP.

This body of work contributes vital information about the patterns and impacts of human-wildlife coexistence at GRNP from both ecological and human perspectives, providing management suggestions that promote both the long-term preservation of biodiversity and the needs of local people.

Acknowledgements

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Author's declarations

I, Tanya Payne, conceived and developed the experimental designs of this study with support and guidance from Dr Kimberley Hockings and Dr Tania Minhos. Further guidance and in country advice and support was given by Benjamin Barca (Technical Advisor, GRNP). I coordinated and collected all field data involved in this project with the exception of camera trap footage. Data collection was supported by Mohamed Kanneh (field assistant and undergraduate student at Eastern Polytechnic, Kenema, Sierra Leone), members of the GRNP research and monitoring department and farmers from the forest edge villages of GRNP. The collection of camera trap data was designed and implemented by the research and monitoring department of GRNP. I performed all literature searches, data analyses, figure production and thesis compilation and formatting, with advice and comments from Dr Kimberley Hockings, Dr Tania Minhos and Professor Michael Cant.

Abbreviations and definitions

CSSL: Conservation Society of Sierra Leone

DBH: Diameter at breast height

GCPO: Goleagorbu Cocoa Producers Organisation

GRNP: Gola Rainforest National Park

NTFPs: Non-timber forest products

PRM: Participatory risk mapping

REDD+: Reducing Emissions from Deforestation and Degradation

RSPB: Royal Society for the Protection of Birds

SSIs: Semi-structured interviews

UGF: Upper Guinean Forests of West Africa

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Chapter 1:

General Introduction

1.1 Human-wildlife coexistence in the Anthropocene

It is widely acknowledged that we are in a new geological and ecological epoch, the Anthropocene, in which anthropogenic activities are driving widespread environmental change (Steffen *et al.* 2011; Corlett 2015) and unprecedented rates of global biodiversity loss (Dirzo & Raven 2003; Dirzo *et al.* 2014). Global population growth coupled with increased conversion of natural habitat to other land uses is resulting in increased competition between humans and wildlife for space and resources (Hockings *et al.* 2015). Understanding the threats to and patterns of human-wildlife coexistence across landscapes with varying levels of formal protection is fundamental for effective conservation planning and management.

Local human communities that live alongside protected areas rely on ecosystem services and resources (Norris *et al.* 2010), but population growth and human activities exert increasing pressures on ecosystems and local wildlife. The biodiversity of tropical forests is of global conservation importance, but is particularly vulnerable to human activities (Dirzo & Raven 2003; Laurance *et al.* 2012). Unsustainable logging practices, the harvest of non-timber forest products (NTFPs) and agricultural expansion contribute to deforestation and fragmentation (Dirzo & Raven 2003; Ticktin 2004; Bradshaw *et al.* 2009). Increased agricultural intensity can also result in loss of plant biodiversity (Gibson *et al.* 2011), and lead to conflicts associated with wildlife crop foraging (Humle & Hill 2016; Hill 2018). The resulting disruptions to habitat connectivity and changes to the patterns and distribution of food and other resources can impact the spatial distribution of species across a landscape, with potentially widespread ecological effects (Bradshaw *et al.* 2009; Effiom *et al.* 2013). Furthermore, hunting of wildlife, for both bushmeat and the illegal wildlife trade, can cause unsustainable losses of tropical wildlife and fear of human predation (Brashares *et al.* 2004; Bradshaw *et al.* 2009; Ripple *et al.* 2016). Human infrastructure such as roads contribute to further habitat fragmentation (Goosem 2007) and carry direct wildlife mortality risks (Fahrig & Rytwinski 2009). It is vital to understand the varying impacts and responses of wildlife to close human proximity and activities in order to achieve effective long-term conservation in areas where humans and wildlife coexist.

It is also important to consider the socio-economic impacts experienced by local people that coexist with wildlife. Problematic wildlife behaviour can cause crop and livestock losses, damage to infrastructure and injury and mortality due to wildlife attacks on humans (Madden 2004; Dickman 2010; Peterson *et al.* 2010). Such conflicts can result in financial losses with impacts on food security, health and poverty (Dickman 2010; Kansky *et al.* 2016; Nyhus 2016). In addition, the resulting reduced tolerances of wildlife can prompt retaliatory actions and cause conflicts between different stakeholders (Peterson *et al.* 2010; Redpath *et al.* 2013, 2015; Hockings 2016). Therefore, to ensure support and success, protected area conservation requires collaboration with local human communities (Andrade & Rhodes 2012), as well as the development of appropriate and adaptive conflict mitigation measures (Hockings & Humle 2009).

Although there may be some similarities between protected areas in how human activities are impacting wildlife and vice-versa, human-wildlife coexistence should be studied on a site-specific basis to inform specific and evidence-based conservation (Sutherland *et al.* 2004). Such studies are of particular importance at sites where both conservation and socio-economic development are key but often competing priorities. This thesis provides vital information about the patterns and impacts of human-wildlife coexistence at Gola Rainforest National Park (GRNP) in Sierra Leone, with the objective of informing conservation and management practices that incorporate both human and ecological perspectives.

1.2 Study site

1.2.1 Sierra Leone and the Upper Guinean Forests of West Africa

GRNP is located in south-east Sierra Leone, West Africa. Sierra Leone has experienced rapid rates of human population growth, with a current population of approximately 7.6 million people (World Bank 2018). Economic development and poverty eradication has been hindered by civil war, which lasted from 1991 to 2002, and the outbreak of Ebola from 2014 to 2016 (World Bank 2018). Of

189 countries, Sierra Leone ranks 184th in the United Nations Human Development Index based on life expectancy, education and gross national income (United Nations Development Programme 2018). Development is therefore a continued priority of the Government of Sierra Leone.

GRNP is located at the western extremity of the Upper Guinean Forests of West Africa (UGF). The UGF extend eastwards from Sierra Leone through southern Guinea, Liberia, Cote d'Ivoire, Ghana and Togo and were identified as one of 25 global biodiversity hotspots (Myers *et al.* 2000). However, it is estimated 10 million hectares of the UGF have been lost as a result of rapid human population growth and the associated pressures from logging and agricultural expansion (Norris *et al.* 2010). GRNP is the largest area of the UGF remaining in Sierra Leone and is therefore of vital conservation importance.

1.2.2 Gola Rainforest National Park

GRNP lies adjacent to the Liberian border (between 7°18′ and 7°51′N and 10°37′ and 11°21′W) and covers parts of seven chiefdoms, Malema, Gaura, Nomo, Tunkia, Koya, Barri and Makpele, located within three districts of Sierra Leone, Kailahun, Kenema and Pujehun (Figure 1.1). GRNP comprises three blocks, Gola South, Gola Central and Gola North, totalling 710 km^2 of protected forest. It was established as a forest reserve in 1926 and was extensively logged from the 1960s to the 1980s, particularly intensively in the more accessible Gola South (Lindsell & Klop 2013; Munro & van der Horst 2013). Through the work and collaboration of the Conservation Society of Sierra Leone (CSSL) and the Royal Society for the Protection of Birds (RSPB), GRNP was awarded National Park status in 2011.

GRNP is made up of lowland moist evergreen forest (Lindsell *et al.* 2011), dominated by trees in the Leguminosae family (Klop *et al.* 2008). Total annual rainfall in this region is approximately 3000 mm, with peaks in July and August, and the forests are catchment areas for the Moro, Mahoi, Mano and Moa rivers (Barca *et al.* 2018). Previous line transect surveys suggest GRNP contains over 50 species of medium to large bodied size mammals, including 12 primate and 11 bovid species (Lindsell *et al.* 2011; Hillers 2013; Table S1), as well as at least 330 bird species, including 14 globally threatened or near threatened species (Klop *et al.* 2008). GRNP is of particular conservation significance for flagship species such as white-necked picathartes (*Picathartes gymnocephalus*) and pygmy hippopotamuses (*Choeropsis liberiensis*), both endemic to West Africa. A population of approximately 270 western chimpanzees (*Pan troglodytes verus*) is also found at GRNP (Ganas 2009; Brncic *et al.* 2010), around 5% of the estimated 5,925 individuals found in Sierra Leone (Heinicke *et al.* 2019). Approximately 70% of the chimpanzee population in Sierra Leone is thought to occur outside of protected areas (Garriga *et al.* 2019; Heinicke *et al.* 2019), therefore GRNP likely provides a vital area of habitat for the conservation of this critically endangered species (Humle et al. 2016).

The GRNP forest blocks are immediately surrounded by a ‘leakage belt’, a 4 km wide area, truncated to the east by the Liberian border. The leakage belt is made up of a mosaic of agricultural land, community managed forests and anthropogenic infrastructure, including 122 villages (forest edge communities) inhabited by approximately 25,000 people (Bulte *et al.* 2013). The agricultural practices of these communities largely focus on subsistence rice farming and increasingly on the cultivation of commercial cash crops (Bulte *et al.* 2013). Investment in the development, marketing and export of cocoa grown in the GRNP leakage belt using agroforestry practices encompasses part of a REDD+ (Reducing Emissions from Deforestation and Degradation) initiative in this area, with both livelihood development objectives (i.e. higher yields and quality to increase prices) and ecological objectives (i.e. reduced deforestation, higher biodiversity and higher carbon storage). The increasing dominance of cocoa as a cash crop at GRNP promotes a growing and major conservation concern, as although vital for human development, the growing presence of cocoa is driving increased wildlife crop foraging and the resulting conflicts. Other anthropogenic activities in and surrounding GRNP include illegal mining and logging, the harvest of NTFPs (Munro & van der Horst 2013) and bushmeat hunting (Davies *et al.* 2008; Jones *et al.* 2017).

Previous research at GRNP has focused largely on biodiversity assessments and monitoring (e.g. Ganas 2009; Ganas & Lindsell 2010; Lindsell *et al.* 2011; Vaglio Laurin *et al.* 2014), with particular emphasis on birds (e.g. Monticelli *et al.* 2012; Burgess *et al.* 2017). There has to date been little research focus on understanding the interactions between humans and wildlife at GRNP and how these data might aid conservation planning and management of the national park.

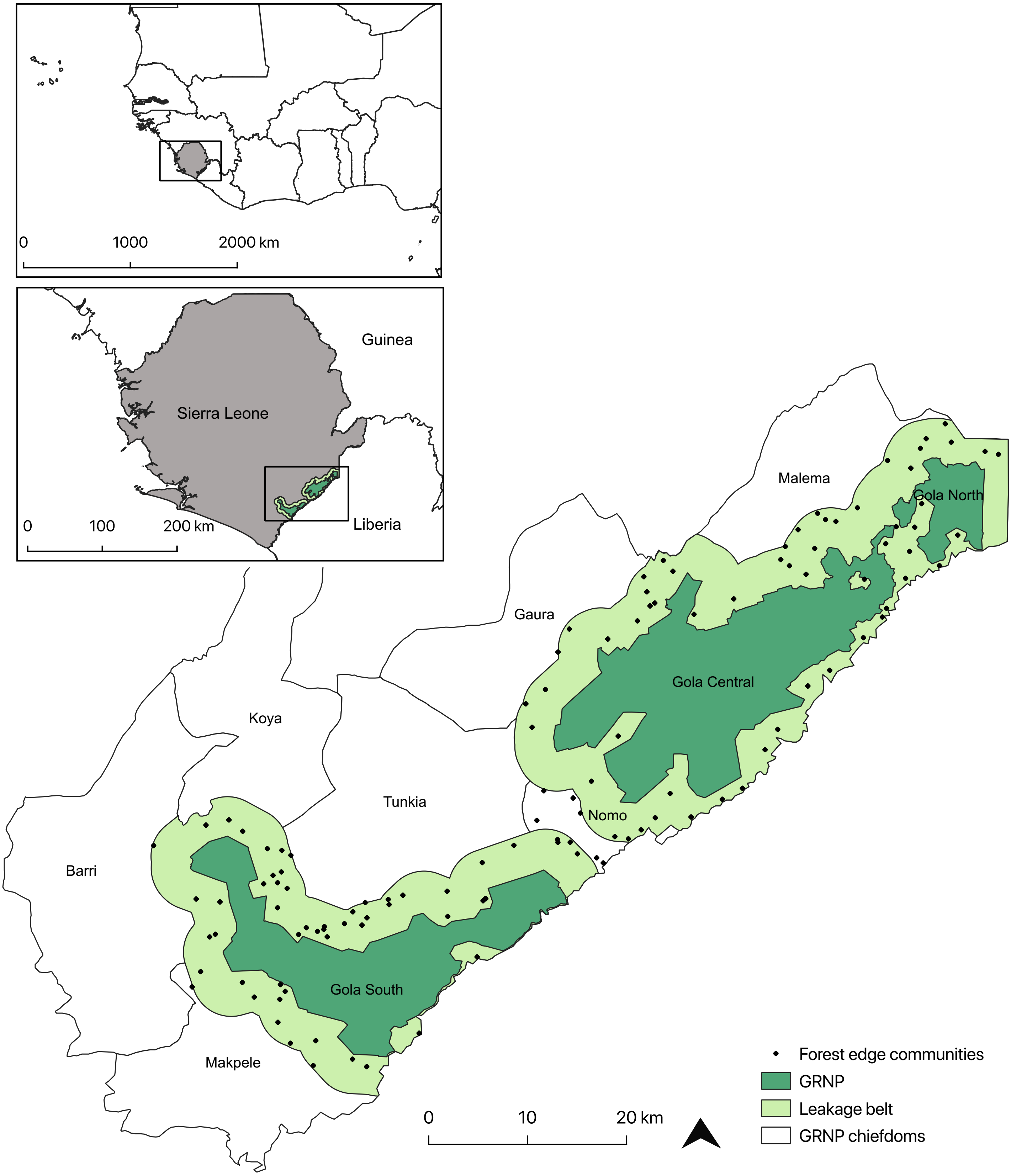


Figure 1.1: Map of Gola Rainforest National Park (GRNP) showing the three forest blocks (Gola Central, Gola South and Gola North), the surrounding leakage belt, the seven GRNP chiefdoms and the 122 forest edge communities (villages). Location of GRNP within West Africa and Sierra Leone inset.

1.3 Thesis aims and objectives

The aim of this thesis was to determine the impacts of human-wildlife coexistence at GRNP and inform conservation management that supports both the long-term preservation of wildlife and the livelihoods of local human communities. Using cross-disciplinary approaches, multiple aspects of human-wildlife coexistence were investigated.

First, we investigated a potential major conservation and livelihood concern at GRNP, wildlife cocoa foraging. By carrying out interviews with farmers and conducting surveys of cocoa plantations we aimed to determine farmer's perceptions of cocoa farming and cocoa foraging species, and compare perceptions of damage to measured damage caused by different species groups. We also aimed to determine the susceptibility of plantations with different spatial (e.g. area, perimeter length, the distance to GRNP, distance to the closest road) and crop defence characteristics (e.g. use of guarding, noise and weapons) and evaluate the conservation value of a local cocoa farming development initiative. The findings of this study provide accurate information to potentially change negative perceptions and tolerances of threatened wildlife and can inform plantation management in order to reduce incidences and impacts of wildlife cocoa foraging.

Second, we examined wildlife presence and distribution across GRNP and how this is impacted by human infrastructure and activities. Using a camera trap survey, we aimed to provide updated information about the species present at GRNP, determine variation in species richness across the national park and determine the impact of proximity to human infrastructure (distance to the closest village and the closest road) and associated activities, as well as ecological factors (tree density and distance to the closest permanent water source), on fine-scale patterns of species richness and species-specific occurrence. The results of this research can be used for informing management initiatives including land use planning and prioritising areas and species for directed conservation efforts and further monitoring.

Chapter 2:

Wildlife cocoa foraging surrounding Gola Rainforest National Park: Understanding perceptions and mitigating negative interactions

Abstract

With increasing conversion of natural habitat to other land uses such as agriculture, understanding the nature of human-wildlife coexistence is fundamental to evidence-based conservation. Negative human-wildlife interactions, such as wildlife crop foraging, can reduce local people's tolerance towards wildlife and national park authorities, and potentially drive retaliatory actions that undermine conservation efforts. In light of extensive recent investment in the cocoa industry surrounding Gola Rainforest National Park (GRNP), this study aimed to understand how farmers perceive sympatric wildlife and explore ways to mitigate and promote tolerance of potentially problematic crop foraging behaviours. Semi-structured interviews (n= 71) and participatory risk mapping showed wildlife cocoa foraging to be the most significant concern for local cocoa farmers, as well as the increased perceived risk of damage associated with western chimpanzees, monkeys and squirrels relative to other crop foraging groups. By quantifying cocoa foraging traces and damage at plantations (n= 24 surveyed plantations), monkeys were found to cause more damage than chimpanzees and squirrels at all plantations. Chimpanzees were disproportionately perceived to be the most damaging species group, suggesting the need for informed knowledge sharing initiatives to strengthen tolerance towards this critically endangered species. The damage caused by crop foraging by different groups varied with the spatial characteristics of plantations and with the crop defence strategies employed by farmers, which may help inform targeted plantation management. For example, plantations located further from roads experienced lower levels of monkey cocoa foraging. Reduced cocoa foraging was also found at plantations owned by members of a local farming initiative, the Goleagorbu Cocoa Producers Organisation, providing evidence for the value of integrated conservation and development projects for mitigation of negative human-wildlife interactions and achieving both socio-economic and conservation objectives at GRNP.

2.1 Introduction

Understanding the patterns and threats of human-wildlife coexistence is fundamental to effective conservation, particularly surrounding protected areas where the activities of local people can result in competition with wildlife for space and resources (Hockings *et al.* 2015). Negative human-wildlife interactions can reduce local people's tolerance towards wildlife, ultimately undermining local conservation efforts (Madden 2004; Andrade & Rhodes 2012; Redpath *et al.* 2013). When developing management plans to promote the long-term preservation of threatened species, it is therefore important to incorporate and evaluate the needs and perceptions of local people (Webber & Hill 2014; Hockings 2016; Spagnoletti *et al.* 2017), and provide appropriate mitigation strategies that reduce incidences of negative human-wildlife interactions (Sitati *et al.* 2005; Linkie *et al.* 2007; Dickman 2010).

Whilst human-wildlife interactions can bring benefits for humans (Frank 2016), for example through the provision of ecosystem services (Soulsbury & White 2015), conflicts can result when humans are negatively impacted by problematic wildlife behaviour (Madden 2004; Woodroffe *et al.* 2005; Frank 2016; Nyhus 2016). Such conflicts most commonly relate to damage to crops and infrastructure, predation of domesticated animals, disease transmission and attacks on humans (Madden 2004; Dickman 2010; Peterson *et al.* 2010). It is increasingly acknowledged that human-wildlife conflicts are also driven by the opposing views of different stakeholders (Peterson *et al.* 2010; Redpath *et al.* 2013, 2015; Madden & McQuinn 2014; Hockings 2016). Communities are directly impacted through damage, financial losses and injury or mortality, and there can be additional indirect effects such as opportunity costs (i.e. loss of ability to undertake other economic activities when having to guard crops or property) and loss of food security (Dickman 2010; Kansky *et al.* 2016; Nyhus 2016). The retaliatory responses of people affected by negative wildlife interactions, which often involve lethal control (Frank 2016; Nyhus 2016), can be major drivers of wildlife declines (Woodroffe et al. 2005; Dickman et al. 2014). Mitigation of negative human-wildlife interactions is therefore a key and growing conservation concern, requiring collaboration and engagement with local people and the development of community-focused conservation strategies (Hockings & Humle 2009; Hockings 2016).

The frequency and relative intensity of different forms of negative human-wildlife interactions vary across regions and environments (Humle & Hill 2016; Nyhus 2016), but wildlife crop foraging is amongst the most consistently documented forms (Peterson *et al.* 2010). Crop foraging (or more traditionally ‘crop raiding’, but see Humle & Hill 2016; Hill 2018) refers to the loss of and damage to human-cultivated plant crops caused by wild animals via feeding or trampling (Hill 2017a, 2018). The introduction of novel crops to areas within or close to wildlife habitat results in exploitation of this resource by species that can adapt their feeding ecology and show behavioural tolerance to the risks associated with increased proximity to humans (Hockings *et al.* 2012; Hill 2017b, 2018). Crop foraging has been reported across various taxa, for example elephants (Sukumar 1990; Naughton-Treves 1998; Hoare 2012; Wilson *et al.* 2015), hippopotamuses (González *et al.* 2017) and rodents (Arlet & Molleman 2007). Crop foraging is particularly common among primate species (Naughton-Treves 1998; Lee & Priston 2005; Warren 2009; Campbell-Smith *et al.* 2010; Hill & Wallace 2012; Humle & Hill 2016), especially those that show high ecological flexibility such as chimpanzees (reviewed in Hockings & McLennan 2012). Incidences of crop foraging resulting in retaliatory killings of wildlife have been extensively reported (e.g. Katsvanga et al. 2006; McLennan et al. 2012; Choudhury 2004) and various other mitigation methods are used by farmers to defend crops, such as guarding, noise production and the use of weapons, including sling shots and stones (Hockings & Humle 2009). However, these methods are often unsuccessful or their effectiveness is not evaluated (Hill 2018; Junker *et al.* 2018), and can have associated indirect costs, such as increased disease risk when guarding crops at night (Mackenzie & Ahabyona 2012).

Due to growing international demand for cocoa (Lenzen *et al.* 2012; Gilbert 2016), there is widespread investment in cocoa growth and industries across West Africa, where smallholders account for 70% of global production (Wessel & Quist-Wessel 2015). In 2015, as part of a REDD+ (Reducing Emissions from Deforestation and Degradation) initiative, the Goleagorbu Cocoa Producers

Organisation (GCPO) was established to support the development, marketing and export of ‘forest-friendly’ cocoa grown within the leakage belt of Gola Rainforest National Park (GRNP). GRNP is of vital conservation importance, comprising the largest remaining area of the Upper Guinean forests of West Africa (UGF) biodiversity hotspot (Myers *et al.* 2000) in Sierra Leone. In collaboration with GRNP management authorities, the GCPO promotes agroforestry practices. Effective agroforestry of shade-grown crops such as cocoa provides an alternative to more destructive slash-and-burn agriculture, resulting in reduced deforestation and encroachment (Tscharntke *et al.* 2011), higher biodiversity (Rice & Greenberg 2000; Schroth & Harvey 2007) and greater carbon storage (Wade *et al.* 2010; Waldron *et al.* 2015). In addition, the GCPO contributes to socio-economic development by establishing cocoa nurseries, supplying loans and equipment and organising Farmer Field Schools that provide training in harvest, fermentation, drying and storage processes to maximise yields and quality. As an incentive to encourage the adoption of such agroforestry techniques and engagement with conservation authorities, and as a result of the higher quality cocoa produced, cocoa from GCPO farmers is bought by GRNP at a higher market price than is average for the area (Tubbs *et al.* 2017; Katie Sims, pers. comm). With increased investment in cocoa farming at GRNP, there is a growing need to monitor wildlife crop foraging and the impacts on local people’s perceptions and tolerance. Appropriate mitigation strategies also need to be developed to ensure the responses of farmers do not compromise the conservation of GRNP wildlife.

Determining local farmers’ perceptions is essential to understand the full impact of wildlife crop foraging on human livelihoods and wellbeing, and the responses of farmers. Local and personal perceptions of risk may result in disproportionate or misdirected retaliatory responses relative to actual crop damage and loss (Dickman 2010; Hill & Webber 2010; Kansky *et al.* 2016). Perceptions of crop foraging and particular crop foraging species can be driven by social influences (Knight 2000), the experiences and exaggerations of other community members (Dickman 2010; Kansky et al. 2016), species’ physical characteristics and perceived likelihood of human attack (Dickman 2012; Webber & Hill 2014) and specific behaviours and patterns of wildlife crop foraging (Hill & Webber 2010). Comparing the perceptions of crop foraging species to measured crop damage can also help reduce negative perceptions of particular species (Webber & Hill 2014; Spagnoletti *et al.* 2017). This approach might be particularly important for shifting negative focus away from threatened species such as great apes, whose slow life histories limit their ability to recover from population losses resulting from retaliatory killings (Humle & Hill 2016).

Mitigation of negative crop foraging interactions may also involve quantifying the amount of damage at plantations with different characteristics. Plantation management initiatives can be informed by exploiting the characteristics of plantations that are less intensively foraged by wildlife (Sitati *et al.* 2005; Linkie *et al.* 2007; Dickman 2012). For example, through evaluation of the crop protection methods used by different farmers, such as guarding and noise production, it may be possible to aid in immediate reductions of crop foraging via knowledge sharing of effective methods (Campbell-Smith *et al.* 2012; Hsiao *et al.* 2013). Additionally, the spatial characteristics of plantations, such as distance from national park boundaries and settlements, can inform future land use planning (Linkie *et al.* 2007; Gubbi 2012). Integrated livelihood and conservation projects may also be an important aspect of conflict mitigation; they may aid in changing tolerances towards crop foraging species and crop losses and promoting greater acceptance of local conservation efforts (Hockings & Humle 2009; Blomley *et al.* 2010), as well as promoting greater investment in effective crop defence strategies. Evaluation of the effectiveness of such integrated initiatives is lacking, particularly for primate conservation (Junker *et al.* 2018).

This study therefore aimed to determine local perceptions and tolerance towards cocoa foraging wildlife, and inform appropriate mitigation strategies to reduce negative human-wildlife interactions at GRNP. A cross-disciplinary approach was used to (1) determine farmer’s perceptions of wildlife cocoa foraging, (2) compare perceived and measured damage attributed to different species groups, (3) determine the spatial characteristics and crop defence methods that predict susceptibility of plantations to cocoa foraging, and (4) evaluate the effectiveness of the GCPO in terms of mitigating conflict over wildlife crop foraging. The following hypotheses and predictions were tested: (1) Wildlife cocoa foraging would be negatively perceived and a major concern for

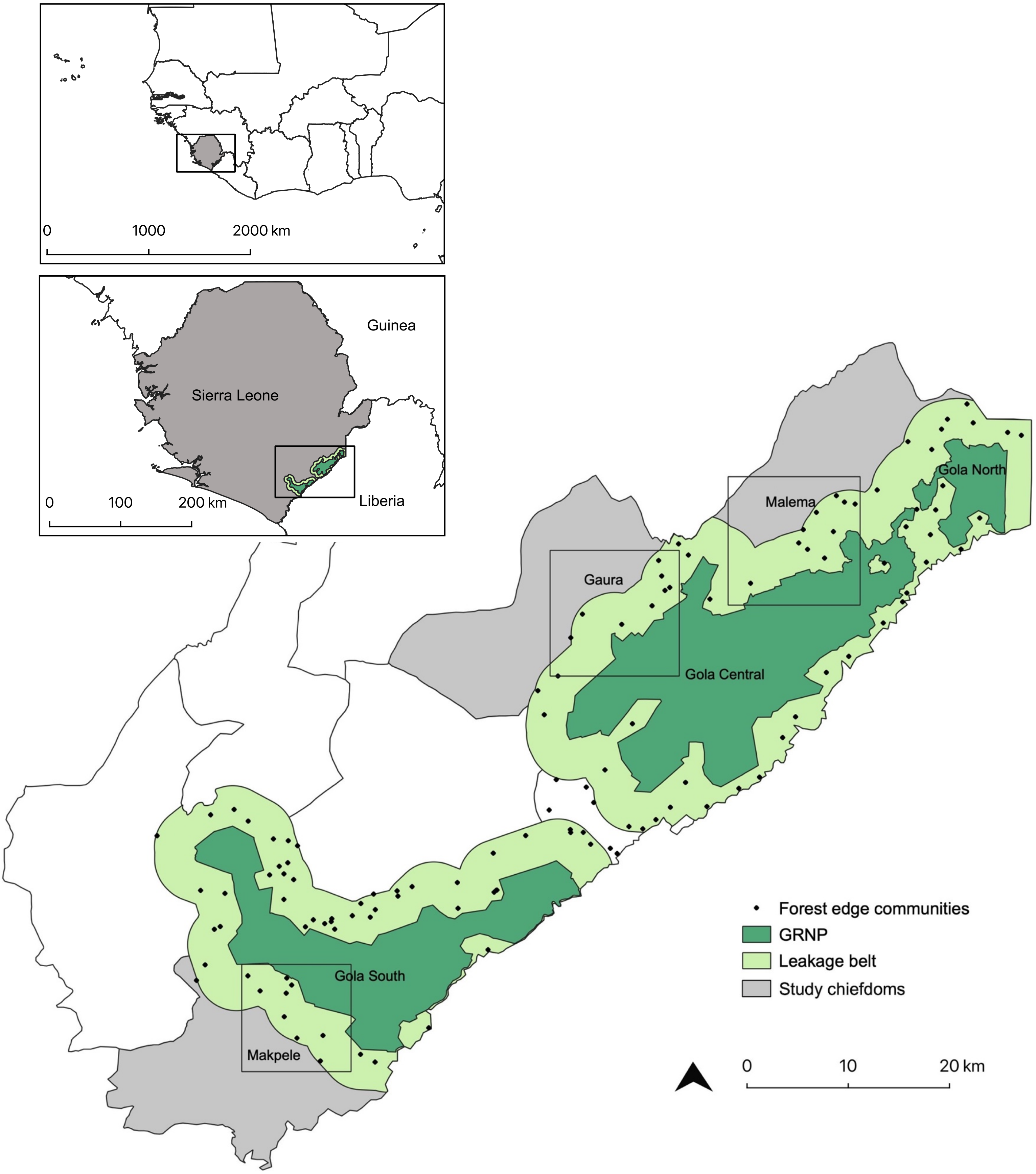
GRNP cocoa farmers, (2) perceptions of damage would vary between species, with larger more visible species blamed disproportionally when compared to measured damage, (3) different plantations will experience varying levels of crop damage according to their spatial features and the crop defence methods used by farmers, with plantations closer to the GNRP boundary and further from human infrastructure experiencing high levels of damage, and (4) plantations would also vary in susceptibility to crop damage based on farmer’s membership of the GCPO, with member farmers experiencing lower rates of crop foraging and perceiving wildlife less negatively than non-members.

2.2 Methods

2.2.1 Study site

Gola Rainforest National Park (GRNP) lies in south-east Sierra Leone (between 7°18′ and 7°51′N and 10°37′ and 11°21′W) and covers parts of the Kailahun, Kenema and Pujehun districts. GRNP is the largest tract of the Upper Guinean forests of West Africa remaining in Sierra Leone and comprises three blocks of lowland moist evergreen forest (Lindsell *et al.* 2011), Gola Central, Gola South and Gola North, totalling an area of 710 km^2. GRNP contains over 50 species of medium to large bodied size mammals (Table S1), including threatened species such as western chimpanzees (*Pan troglodytes verus)*, western red colobus (*Piliocolobus badius)*, Jentink's duiker (*Cephalophus jentinki*) and pygmy hippopotamus (*Hexaprotodon liberiensis)* (Lindsell et al. 2011; Hillers 2013) and at least 330 bird species including threatened white-necked picathartes (*Picathartes gymnocephalus*) and Gola malimbe (*Malimbus ballmanni*) (Klop et al. 2008). Surrounding the GRNP blocks is a 4 km wide leakage belt covering parts of seven chiefdoms. The leakage belt is made up of community-managed forests, agricultural areas and human infrastructure including 122 settlements (forest edge communities) in which approximately 25,000 people live (Bulte *et al.* 2013). This study focused on farmers and plantations located in the leakage belt from three chiefdoms (Figure 2.1), Malema and Gaura (Gola Central) and Makpele (Gola South). To date, the efforts of the GCPO have been focused in Malema and Gaura, but not all farmers and villages in these areas have been reached. The farmers of Makpele have also not yet received the opportunity to become members of the GCPO.

a)


0
1000
2000 km
Guinea
Sierra Leone
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Malema
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b)

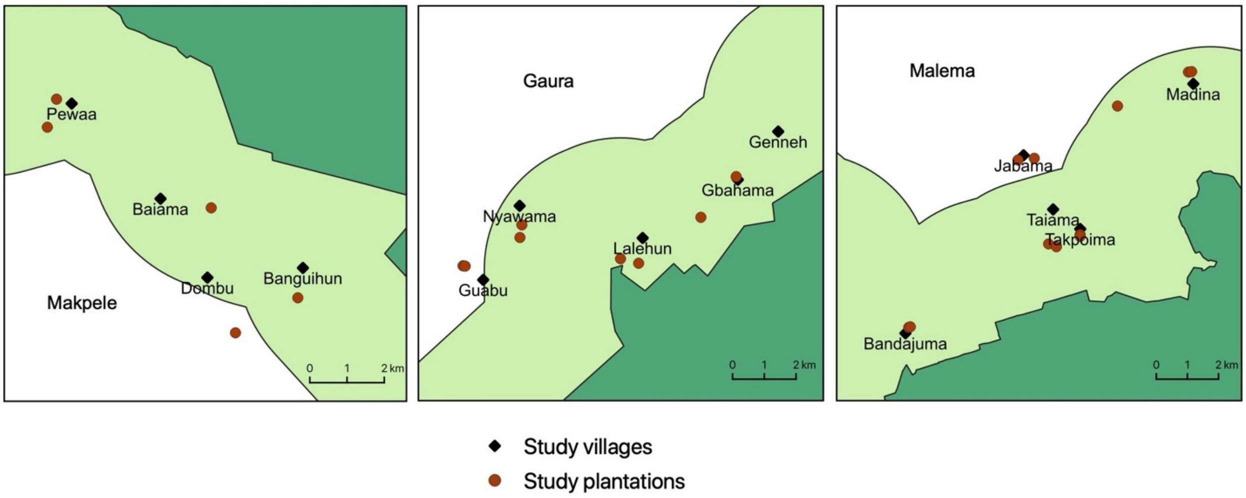


Figure 2.1: (a) Location of study areas within three chiefdoms of GRNP, showing all 122 villages (forest edge communities) and with location of GRNP within West Africa and Sierra Leone inset. (b) Location of villages where semi-structured interviews (n=71) were carried out and plantations (not to scale) where cocoa foraging surveys (n=24) were conducted, within each study chiefdom.

2.2.2 Semi-structured interviews

Semi-structured interviews (SSIs) were conducted with 71 cocoa plantation owners from 14 villages across Gaura (n= 5 villages, n= 19 interviews), Malema (n= 5, n= 27) and Makpele chiefdoms (n= 4, n= 25) (Figure 2.1b). Interviews were carried out between 3rd November and 14th December 2018, coinciding with peak cocoa harvest. Interviews were conducted with prior consultation and permission of village chiefs and the selection of participants was based on recommendations of village chiefs as per local custom, and the availability of farmers. Prior to interviews with each participant, the purpose and aims of the project were explained and the voluntary nature of participation was assured, as was complete anonymity (Newing *et al.* 2010). All interviews were conducted by the international lead researcher and questions and responses were translated between English and the local Mende language by two trained field assistants. All responses were noted by hand.

Interview structure and analyses

Interviewees were asked about planned expansion of cocoa plantations, as well as various farming practices such as crop defence strategies used. Interviews also determined whether farmers perceived any benefits of living close to wildlife, and whether they were involved in the GCPO and if membership brought benefits for the cocoa harvest.

In order to determine the perceptions of wildlife cocoa foraging relative to other risks, interviewees were asked to list all risks that they associate with cocoa farming. Responses were later grouped into categories and analysis followed the participatory risk mapping (PRM) methods outlined by Webber & Hill (2014). For each risk a severity index, S, was calculated: $S = 1+(r-1)/(n-1)$ where r is the position in the order of responses given by the interviewee and n is the total number of risks identified by the interviewee. A mean value between 1 (most severe) and 2 (least severe) for all respondents identifying each risk was calculated. The proportion of interviewees (ranging from 0-1) identifying each risk was used as the incidence index, I. A third index, risk, R, where a higher figure represents a greater perceived risk, was calculated by dividing incidence by severity. The same PRM approach was used to determine perceptions about which species forage on cocoa at GRNP. Farmers were asked to list all the species that visit the plantation to forage on cocoa and due to common use of grouping terms and wildlife rarely being identified to species-levels, responses were categorised prior to analysis.

To allow for direct comparison between the perceived and measured damage caused by each of the study focal groups, farmers were asked to rank chimpanzees, monkeys and squirrels in terms of the amount of damage caused to cocoa crops. Selection of the ‘species groups’ chimpanzees, monkeys and squirrels was based on local language use (i.e. grouping of multiple species) and initial consultation with GRNP cocoa farmers and the GRNP research department. The groups also show observable and distinctive cocoa foraging behaviours and traces (described below). The species group ‘monkeys’ comprises all primates present at GRNP excluding western chimpanzees, one potto species and two galago species (Table S1). Western chimpanzees are considered separately due to distinct cocoa foraging behaviours and consistent separate acknowledgment by farmers. The species group ‘squirrels’ comprises all squirrel species present at GRNP.

2.2.3 Plantation surveys

Surveys of 24 cocoa plantations were carried out between 8th November and 14th December 2018. Plantations surveyed were owned by farmers from 13 villages (1 - 3 per village) across the three study chiefdoms (Gaura, Malema and Makpele, n plantations= 8, 11 and 5 respectively). Selection of plantations was made by village chiefs to adhere to local customs.

Mapping

At each plantation the boundary was walked with the farm owner using two identical Garmin GPSMAP 64st GPS devices. Plantation perimeters were mapped using QGIS software (version 3.6.3, QGIS Development Team 2019) and the area of each plantation was calculated. Perimeter length ranged from 158.43 - 627.10 m (mean ± SD= 340.74 ± 120.09) and plantation area ranged from 1,513.00 - 14,492.13 m^2 (mean ± SD= 5,352.50 ± 3,333.95). See Table S2 and Figure S2 for descriptive statistics and polygon maps of plantations.

Measured crop damage

Traces of cocoa foraging by the three focal species groups, western chimpanzees, monkeys and squirrels were counted along transects. Transects were 4m wide and spaced at 25m intervals across the whole plantation width. Each transect ran from the north to the south boundary; total transect length for each plantation varied due to plantation shape and size, ranging from 59.16 - 365.61 m (mean ± SD= 180.92 ± 88.75). Due to the nature of fieldwork it was not possible to revisit plantations once areal analysis and mapping had been completed. This systematic transect approach ensured that transect length was representative of plantation area (see page 31) and therefore survey effort was accounted for in later analysis.

Training to identify cocoa foraging behaviours and traces left by different species groups was given by members of the GRNP research department and farmers prior to data collection. Chimpanzees remove cocoa pods from trees with force by twisting and pulling leaving scars in the tree bark, crack or bite pods to feed on fruit and swallow seeds whole, including pods that have been removed or pods still attached to trees, and in some cases they produce cocoa seedlings and germinating seeds from faecal remains (as reported at Bossou, Guinea, Hockings et al. 2017). Monkeys remove small cocoa pods leaving the pod fibre attached to the tree (no bark scar damage) and feed on pods or pod fragments on trees and on the ground. Squirrels do not remove whole pods but feed on pods attached to trees, or pods or pod fragments removed by other species. Trace types counted therefore were: sites of damage to cocoa trees, damage to cocoa pods in cocoa trees, damage to cocoa pods and fragments on the ground surrounding cocoa trees, and faecal seedlings (Figure 2.2). Traces were distinguished by characteristics including method of removal, teeth and bite marks, fragment size and evidence of cracking.

Figure 2.2: Examples of cocoa foraging traces counted at cocoa plantations located in the leakage belt of GRNP. (a) Squirrel damage to cocoa pod, (b) monkey damage to cocoa pod, (c) pod removed and damaged by western chimpanzee and (d) damage to cocoa tree caused by western chimpanzee pod removal.

Only traces that could be identified as one of the three species groups were counted. Correct and consistent categorisation of traces was ensured by using the same team during all plantation surveys, consisting of the lead researcher and two trained field assistants, with the addition of each respective plantation owner. The presence of the farmer ensured that no human harvested cocoa was counted and in all plantations it was clarified that humans used knives to harvest cocoa, consistent with other sites in West Africa. For direct comparisons between species groups, only traces estimated as less than one year old were used for analysis. For all other analyses, all counted traces for each species group were used, including where all three species groups were combined to give a total amount of crop foraging traces for each plantation. The majority of traces were less than three months old and so could be accurately attributed to a species group, only those left on trees (i.e. sites of pod removal from trees) allowed for identification of up to one year. If a trace could not be classified as one of the focal species groups it was not counted and therefore not included in any later analysis. The majority of traces were significantly less than one years old and only those

Analyses

All statistical analyses were conducted using R Studio (version 1.1.463, R Core Team 2018).

Variation between species groups

A one-way ANOVA and post-hoc TukeyHSD tests were carried out to determine whether there was significant variation in the mean number of cocoa foraging traces caused by the different species groups across the surveyed plantations. The relative amount of damage (number of traces) was also ranked by species group for each plantation, for comparison with perceived rankings.

Susceptibility of plantations

The number of cocoa foraging traces was divided by plantation area to give a proxy of the amount of damage caused by cocoa foraging at each plantation (by all species groups combined and for each species group individually). Survey effort was therefore controlled for (total transect length was highly correlated with plantation area, Pearson’s correlation test, t_{22}=22.62, r=0.98, p=<0.001). For the purposes of this study, productivity was assumed to be equal across all plantations.

Multiple linear regression models were then used to determine whether various spatial factors impacted the amount of damage caused by each species group and by all groups combined. The response terms, the number of cocoa foraging traces per m^2 attributed to each species group (chimpanzees, monkeys and squirrels respectively) and all three species groups combined were log transformed to meet normality assumptions. The same fixed explanatory variables were used for all models: plantation area, plantation perimeter length, distance to the GRNP boundary, distance to the closest village (permanent settlement, one of 122 forest edge communities) and distance to the closest road (including minor and major roads). Due to lack of systematic land use mapping of the leakage belt surrounding GRNP and heavy fragmentation, the distance to the nearest forest block (beyond GRNP) could not be included in analysis. The distance to the nearest plantation or average distance to neighbouring plantations was also not included as a covariate (therefore attempting to account for spatial autocorrelation) as there are numerous cocoa plantations and unmapped land uses in this area that could not be surveyed within the breadth of this study. The minimum distance to the GRNP boundary, distance to the closest village and distance to the closest road were calculated in QGIS using nearest neighbour analysis with plugin *nnjoin*. Variation inflation factors (VIF) were calculated using R package *car* (Fox & Weisberg 2019) to test for collinearity between model variables and area was removed from the global model as it was highly collinear with perimeter length (VIF value >5). Models were then fit with all possible combinations of fixed effects using the R package *MuMIn* (Barton 2019) and the resulting models were ranked based on Akaike's information criterion corrected for small sample size (AICc). Model averaging of the top model set (ΔAICc <2) was then used for significance testing.

The same modelling approach was used to determine the impact of various crop defence methods used by farmers on the amount of damage caused by cocoa foraging for each individual species group and all species groups combined. The log-transformed number of cocoa foraging traces per m^2 for all species groups combined and each respective species group were fit as response terms. The following variables were fit as explanatory terms (coded as 1 (used) or 0 (not used) based on responses given during SSIs with surveyed plantation owners): use of guarding (continued presence in farm, used by 8 of surveyed plantations), use of noise (includes drumming of pans, drums and trees and shouting, n=21), use of scarecrows (n=3), use of trapping (n=2), use of slingshots (n=5), use of fire (n=2), use of hunting dogs (n=1) and use of surrounding land management (bordering other plantations as opposed to forested areas, n=3).

Impact of GCPO membership

In order to compare the amount of measured cocoa damage between plantations owned by farmers that were members of the GCPO and those that were not, two-sample Welch's t-tests were carried out using the mean log-transformed number of cocoa foraging traces per m^2 attributed to all species groups combined and individually as a function of GCPO membership. GCPO membership of surveyed plantations was determined during SSIs (members n=17, non-members n=7).

2.3 Results

2.3.1 SSIs and perceptions of cocoa farming

Males represented 87.32% of participants and 12.68% were female. The mean age of participants was 46.34 ± SD 16.95 years (range= 20- 100). Farming (including other crop types) was the sole occupation of 74.65% interviewees, with a further 25.35% supplementing incomes with other occupations including petty trading, store keeping, community health care and tourism.

Of those interviewed, 84.51% (60 of 71) of cocoa plantation owners were planning to increase the area of land dedicated to cocoa farming, as direct additions to existing plantations (n=47) and/or by developing new areas (n=16). 53.52% of participants were members of the GCPO; all members were from Gaura and Malema chiefdoms. Of GCPO member farmers, 97.37% reported that it had brought benefits for the harvest, reasons for this were rarely given (due to question omission) but training to increase yields (n=8) and the provision of loans (n=13) and equipment (n=1) were identified. When asked about the benefits that wildlife bring generally, 84.51% reported no benefits to themselves or the community. The most commonly acknowledged benefit was membership of the GCPO (14.08%). Employment benefits and reduced crime were each identified by one individual respectively.

Eight different methods were reported as being used to protect cocoa plantations from wildlife cocoa foraging; noise (used by 88.73% of 71 interviewed farmers), guarding (28.17%), slingshots (14.08%), scarecrows (8.45%), trapping (8.45%), land management (i.e. bordering other plantations as opposed to forested areas) (7.04%), fire (2.82%) and hunting dogs (1.41%).

Perceived risks of cocoa farming

Interviewees listed a number of problems associated with cocoa farming which were grouped into seven risk categories (Table 2.1). Wildlife cocoa foraging showed the highest risk index across all three chiefdoms. There was some variation in the risks identified between chiefdoms, for example, only farmers from Makpele described safety concerns.

Table 2.1: Risk index of perceived risks associated with cocoa farming derived from semi-structured interviews (n=71) with plantation owners at Gola Rainforest National Park. Higher values indicate risks perceived as more significant.

	Total Risk Index	**Gaura Risk Index**	**Malema Risk Index**	**Makpele Risk Index**
Wildlife cocoa foraging	0.778	0.739	0.784	0.802
Death of cocoa	0.319	0.351	0.347	0.270
Financial constraints	0.247	0.215	0.037	0.499
Climate	0.115	0.118	0.140	0.088
Social issues	0.057	0.063	0.037	0.080
Safety concerns	0.047	0.000	0.000	0.134
Insect disturbance	0.019	0.053	0.019	0.000

Example responses given by farmers included in categories:
Wildlife cocoa foraging= animal disturbance, animal damage, animals stealing, crop raiding
Death of cocoa= rotting and disease of pods and trees, no and poor pod growth
Financial constraints= inability to buy food and equipment and pay for farming assistance
Climate= rainfall, flooding, sun/shade exposure (too much, not enough)
Social issues= people stealing cocoa, lack of support, gender and trade issues
Safety concerns= lack of safety equipment and dangerous animals
Insect disturbance= disturbance to cocoa and to farmers during plantation work and harvest

The severity and incidence indices of each risk across all interviewed farmers are shown graphically in the risk map (Figure 2.3). Wildlife cocoa foraging was identified by all interviewees (I= 1.00) and showed the highest severity index (S= 1.29). Few farmers identified insect disturbance (I= 0.03) as a problem associated with cocoa farming at GRNP.

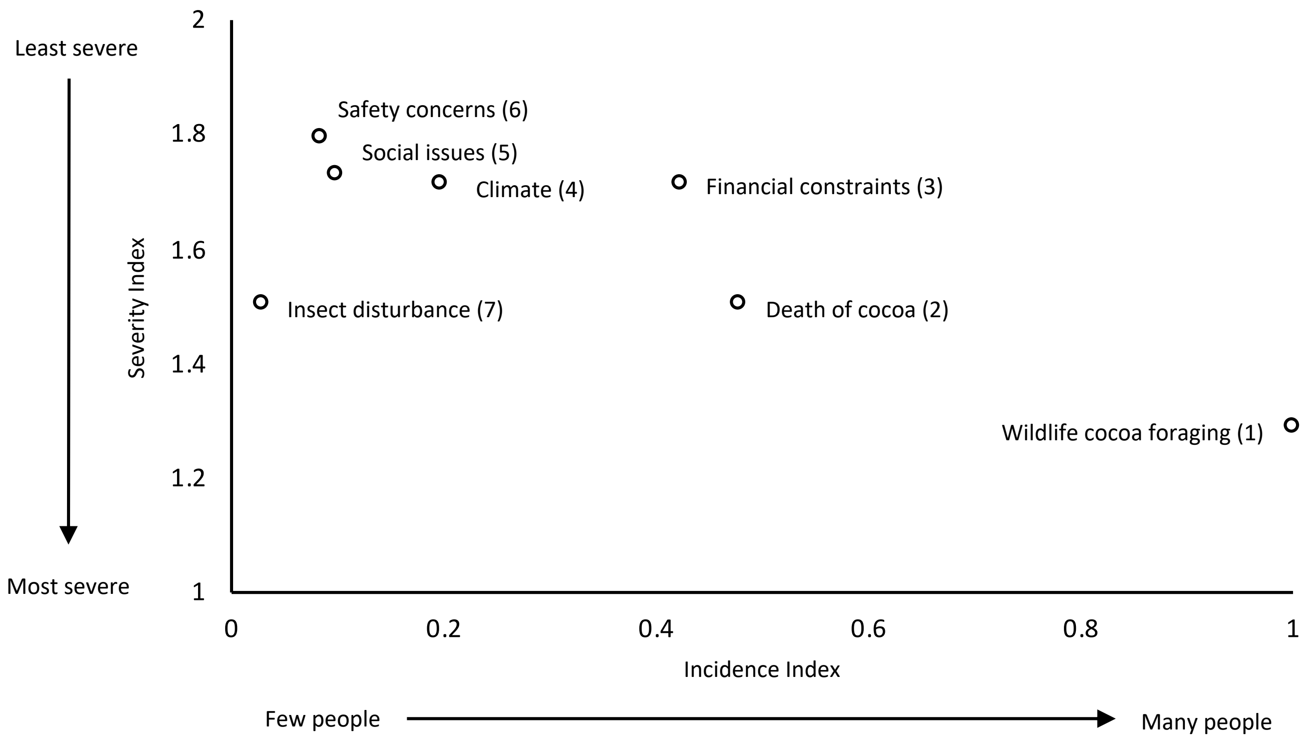


Figure 2.3: Risk map showing GRNP cocoa farmers' perceptions towards wildlife cocoa foraging and other risks associated with cocoa farming identified during semi-structured interviews (n=71). Risk index rank is in parentheses.

Variation between chiefdoms in the risks identified and the incidence and severity indices of each are shown in Figure S3.

2.3.2 Perceptions of cocoa foraging species and measured cocoa damage

Perceptions of cocoa foraging species groups

Cocoa farmers gave a number of responses when identifying animals that forage at cocoa plantations. Animals were rarely identified to species-level; based on grouping terms used by farmers, responses were categorised into seven species groups (Table 2.2). Monkeys, chimpanzees and squirrels showed higher risk indices than any other group across all three chiefdoms.

Table 2.2: Risk index of cocoa foraging species groups derived from semi-structured interviews (n=71) with plantation owners at Gola Rainforest National Park. Higher values indicate species groups perceived as more problematic.

	Total Risk Index	**Gaura Risk Index**	**Malema Risk Index**	**Makpele Risk Index**
Monkeys	0.747	0.731	0.829	0.693
Chimpanzees	0.660	0.671	0.624	0.685
Squirrels	0.585	0.581	0.574	0.601
Other rodents	0.108	0.158	0.041	0.156
Snakes	0.058	0.000	0.042	0.121
Insects	0.021	0.000	0.056	0.000
Ungulates	0.014	0.000	0.059	0.000

The severity and incidence indices are represented graphically in the risk map (Figure 2.4). Monkeys, chimpanzees and squirrels were identified consistently by farmers (I= 1.00, 0.92 and 0.99 respectively), supporting their inclusion as the groups for which measured damage was quantified.

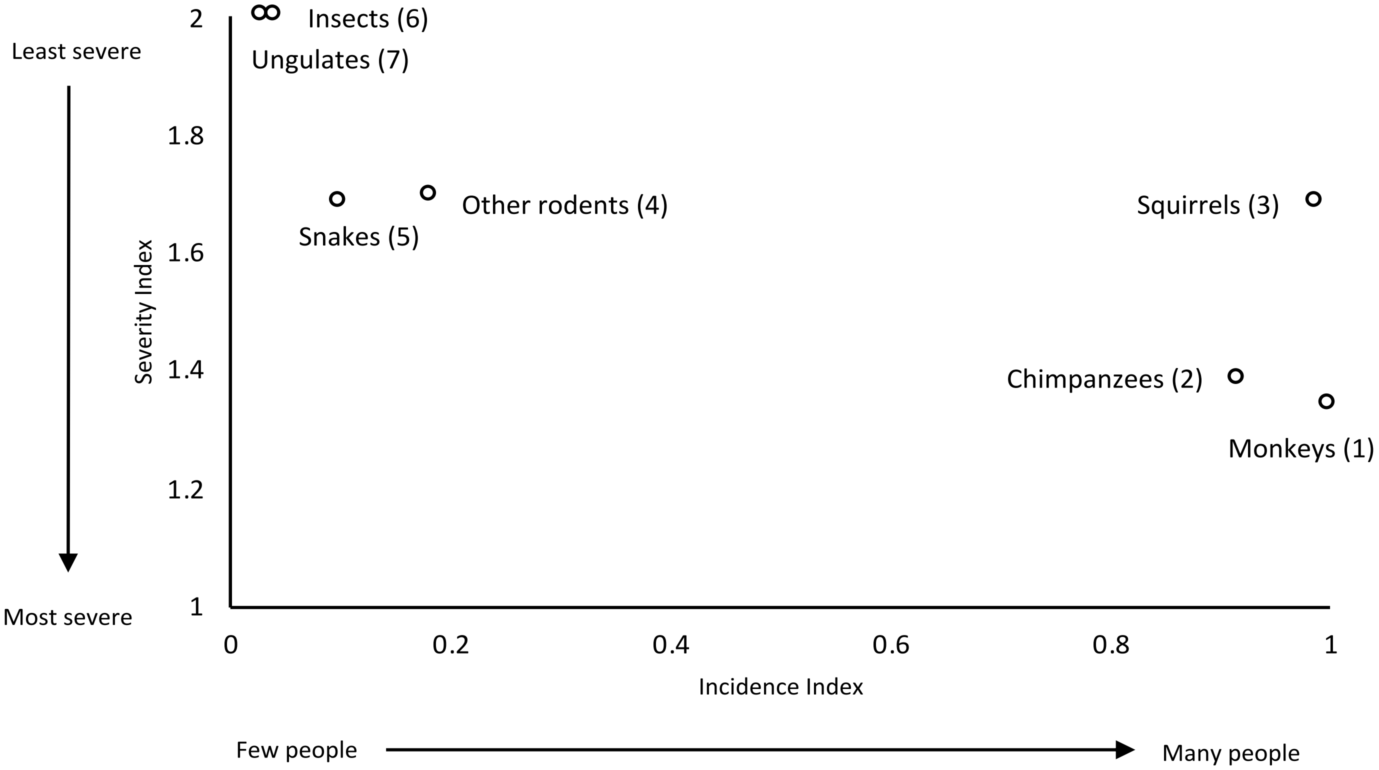


Figure 2.4: Risk map showing GRNP farmers' perceptions of cocoa foraging species groups identified during semi-structured interviews (n=71). Risk index rank is in parentheses.

Variation between chiefdoms in the species groups identified and the incidence and severity indices of each are shown in Figure S4.

Measured crop damage

All surveyed plantations (n= 24) contained chimpanzee and monkey cocoa foraging traces, and squirrel cocoa foraging traces were found in 23 of the plantations. The mean number of cocoa foraging traces (<1 year old) per plantation differed significantly between the three species groups (Figure 2.5, one-way ANOVA, $F_{2,69}$= 42.51, p=<0.001). The mean number of traces attributed to monkeys was significantly higher than that for both chimpanzee and squirrels (TukeyHSD test, p=<0.001). There was no significant difference between the mean number of traces caused by chimpanzees and squirrels (TukeyHSD test, p= 0.26).

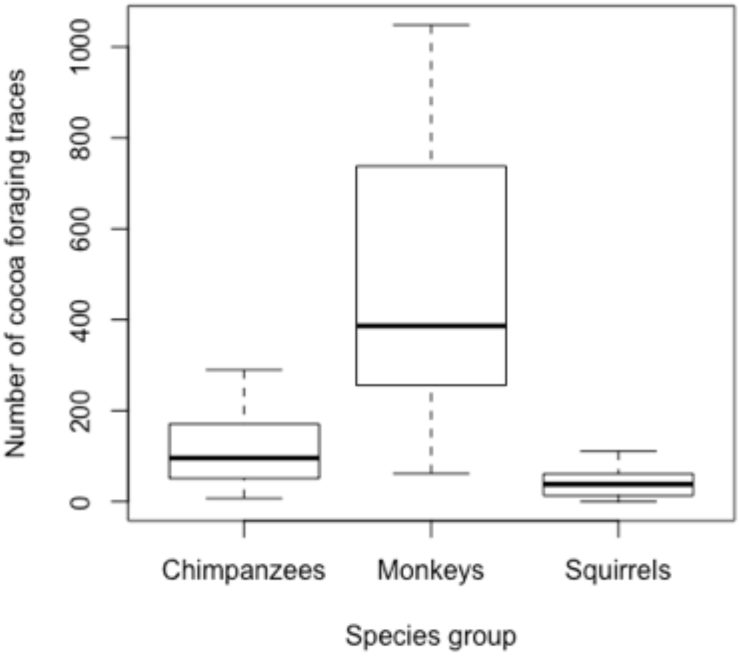


Figure 2.5: The number of cocoa foraging traces per plantation for chimpanzees, monkeys and squirrels counted during surveys of cocoa plantations at GRNP (n=24).

Perceived and measured damage rankings

There was variation between the rankings for perceived and measured damage caused by each species group. Chimpanzees were perceived by the majority (52.11%) of cocoa farmers as the species group causing the most damage to cocoa farms (ranked first), but measured damage rankings showed monkeys were most damaging in all surveyed plantations (Figure 2.6). Monkeys were perceived as most damaging by 36.62% of farmers. Squirrels were perceived as most damaging by 11.27% of farmers. Chimpanzees, monkeys and squirrels were perceived as the least damaging group (ranked third) by 33.33%, 8.33% and 58.33% of farmers respectively (Figure S5). Measured damage caused by chimpanzees ranked second across 87.50% surveyed plantations and third across 12.50% surveyed plantations. Measured damage caused by squirrels ranked second and third across 12.50% and 87.50% plantations respectively.

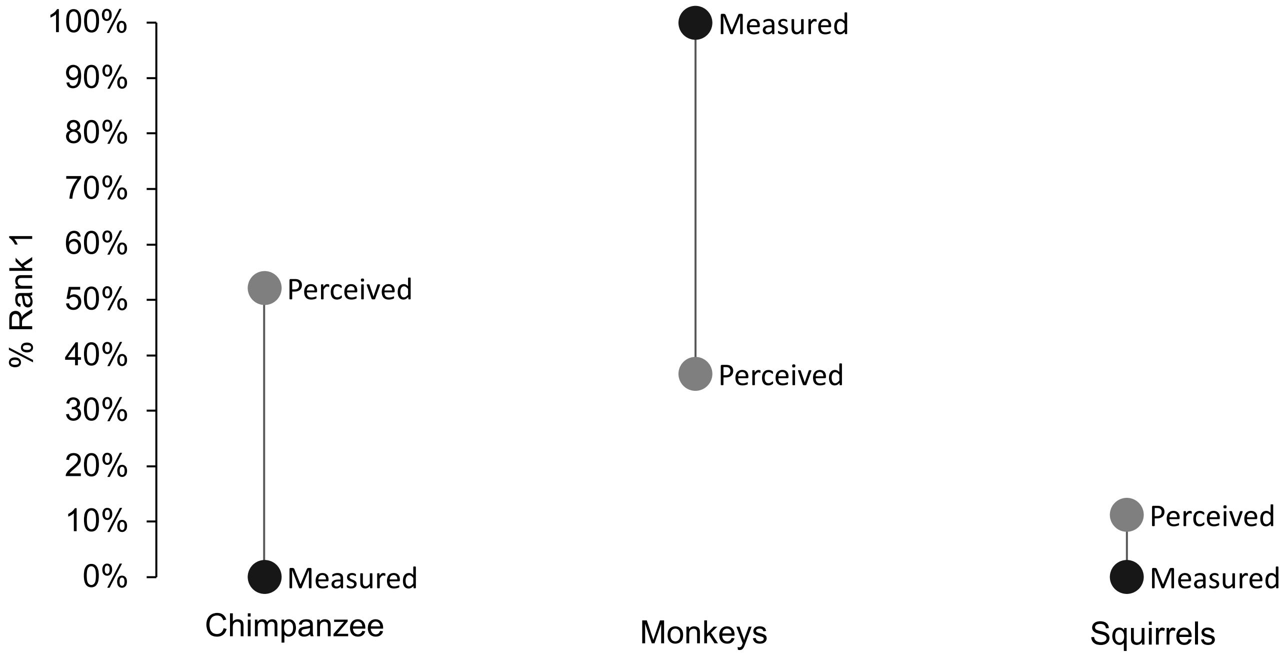


Figure 2.6: Percentages of interviewees (perceived, light grey, n=71) that ranked each species group as the most damaging to cocoa plantations (rank 1), and the percentages of surveyed plantations (measured, dark grey, n=24) in which species group found to be most damaging (rank 1).

2.3.3 Susceptibility of plantations to cocoa foraging

Spatial characteristics

For each of the species groups there was variation in the spatial covariates that contributed to the top models (ΔAIC <2) for predicting the amount of damage (number of cocoa foraging traces per m^2) at each plantation (Table 2.3). Model averaging showed damage caused by all three species groups combined (Figure 2.7a) and by chimpanzees (Figure 2.7b) was significantly higher at plantations further from the GRNP boundary (Table 2.4). Damage caused by monkeys was significantly higher at plantations with shorter perimeter lengths and at sites closer to roads (Figures 2.7c and 2.7d). None of the spatial covariates significantly predicted the amount of damage caused by squirrels.

Table 2.3: Summary table of top models (ΔAICc <2) derived from linear models. + indicates inclusion of covariate in model. Averaging of top models used for significance testing in text.

Response term	perimeter	GRNP	village	road	df	AICc	ΔAICc	AICw
logAll species cocoa foraging traces per m^2	+	+			4	9.68	0.00	0.51
		+			3	10.87	1.20	0.28
	+	+		+	5	11.49	1.81	0.21
logChimpanzee cocoa foraging traces per m^2		+			3	68.31	0.00	0.61
	+	+			4	69.20	0.89	0.39
logMonkey cocoa foraging traces per m^2	+			+	4	49.00	0.00	0.70
	+	+		+	5	50.70	1.74	0.30
logSquirrel cocoa foraging traces per m^2					2	79.90	0.00	0.31
		+			3	80.00	0.03	0.30
			+		3	80.70	0.78	0.21
		+	+		4	80.90	0.97	0.19

Abbreviations: perimeter= plantation perimeter length, GRNP= distance to GRNP boundary, village= distance to closest village, road= distance to closest road

Table 2.4: Averaged coefficients and p values for predictor covariates included in top model set (Table 2.3). Significant predictors (P<0.05) shown in bold.

Response term	Covariates	Estimate	S.E	z value	P value
logAll species cocoa foraging traces per m^2	perimeter	-0.001	0.000	1.909	0.056
	GRNP	**0.000**	**0.000**	**2.415**	**0.016**
	road	0.000	0.000	1.035	0.301
logChimpanzee cocoa foraging traces per m^2	perimeter	-0.001	0.001	0.565	0.572
	GRNP	**0.000**	**0.000**	**2.948**	**0.003**
logMonkey cocoa foraging traces per m^2	**perimeter**	**-0.003**	**0.001**	**2.912**	**0.004**
	GRNP	0.000	0.000	1.064	0.287
	road	**0.000**	**0.000**	**2.001**	**0.045**
logSquirrel cocoa foraging traces per m^2	GRNP	0.000	0.000	0.714	0.475
	village	0.000	0.000	0.565	0.572

Abbreviations: perimeter= plantation perimeter length, GRNP= distance to GRNP boundary, village= distance to closest village, road= distance to closest road

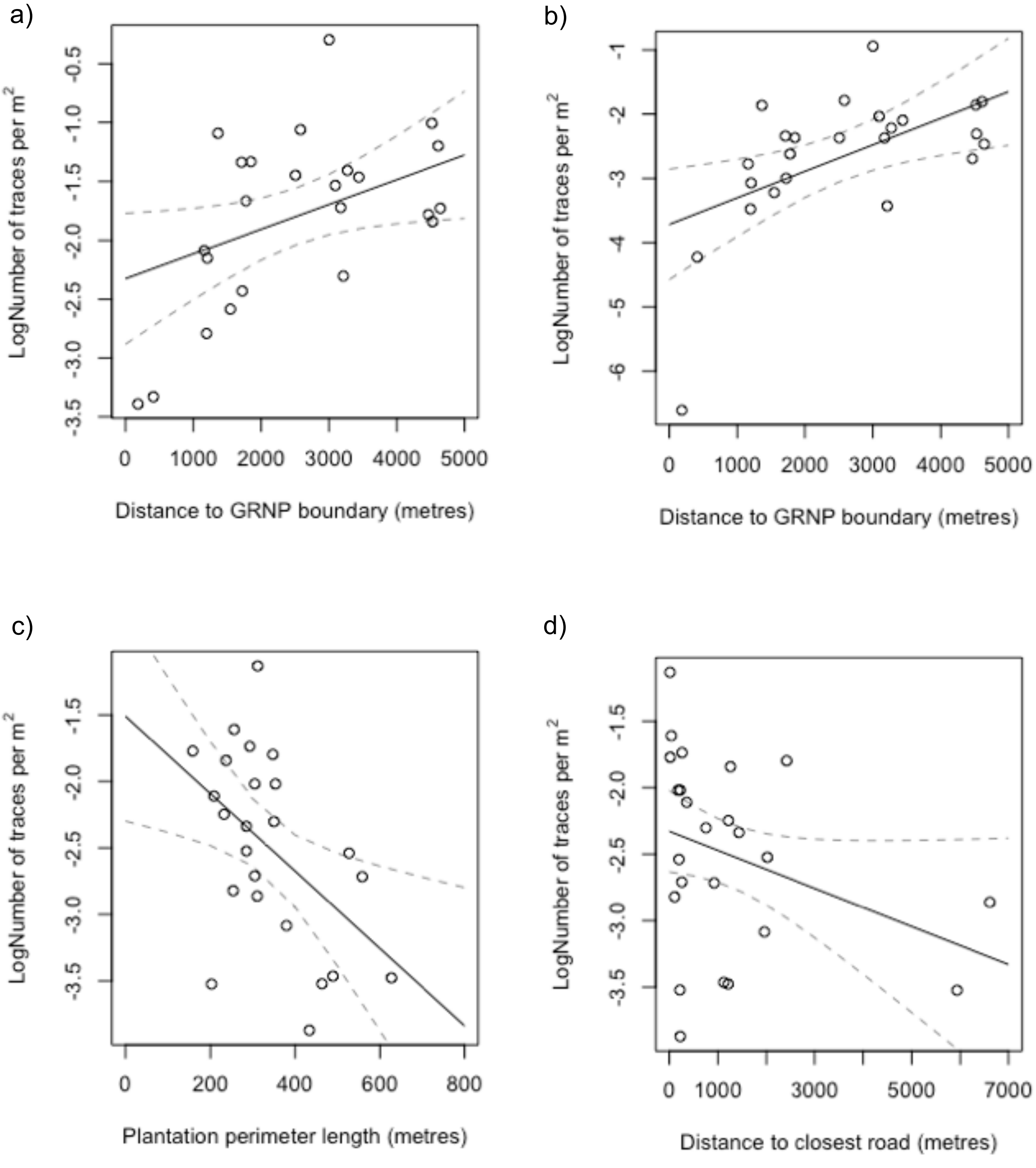


Figure 2.7: Relationship between log number of cocoa foraging traces per m^2 and significant predictors from averaging of linear models, all other variables in model set held at mean values. Circles are actual plantation survey results (n=24). Dashed lines are 95% confidence intervals. Number of (a) traces caused by all species groups predicted by distance to GRNP boundary, (b) traces caused by chimpanzee predicted by distance to GRNP boundary, and traces caused by monkeys predicted by (c) plantation perimeter length and (d) distance to closest road.

Crop defence methods

For each of the species groups there was also variation in the crop defence methods that contributed to the top models for predicting the amount of damage at each plantation (Table 2.5). Model averaging showed significantly lower levels of damage caused by all species groups combined (Figure 2.8a) and chimpanzees (Figure 2.8b) at plantations where owners used hunting dogs (Table 2.6). None of the crop defence methods significantly predicted the amount of damage caused by monkeys and squirrels.

Table 2.5: Summary table of top models (ΔAICc <2) derived from linear model. + indicates inclusion of covariate in model. Averaging of top models used for significance testing.

Response term	**guarding**	**noise**	**scarecrows**	**trapping**	**slingshots**	**fire**	**hunting dogs**	**land management**	**df**	**AICc**	**ΔAICc**	**AICw**
logAll species cocoa foraging traces per m^2							+		3	53.99	0.00	0.71
			+				+		4	55.73	1.74	0.30
logChimpanzee cocoa foraging traces per m^2							+		3	56.20	0.00	0.45
			+				+		4	57.91	1.71	0.19
	+						+		4	57.98	1.77	0.19
		+					+		4	58.17	1.97	0.17
logMonkey cocoa foraging traces per m^2									2	55.95	0.00	0.29
							+		3	56.29	0.35	0.24
								+	3	56.66	0.71	0.20
					+				3	57.42	1.47	0.14
			+						3	57.57	1.62	0.13
logSquirrel cocoa foraging traces per m^2		+							3	78.91	0.00	0.40
									2	79.95	1.04	0.24
	+	+							4	80.19	1.29	0.21
		+				+			4	80.74	1.83	0.16

Table 2.6: Averaged coefficients and p values for predictor variables from the top model set (Table 2.5). Significant predictors ($p<0.05$) shown in bold.

Response term	Covariates	Estimate	S.E	z value	P value
logAll species cocoa foraging traces per m^2	**dogs**	**-1.651**	**0.685**	**2.275**	**0.023**
	scarecrows	0.423	0.414	0.962	0.336
logChimpanzee cocoa foraging traces per m^2	**dogs**	**-4.136**	**0.731**	**5.347**	**<0.001**
	scarecrows	0.450	0.434	0.977	0.328
	guarding	0.320	0.318	0.949	0.343
	noise	-0.398	0.436	0.861	0.389
logMonkey cocoa foraging traces per m^2	dogs	-0.258	0.577	0.438	0.662
	land management	-0.120	0.308	0.379	0.705
	slingshots	0.052	0.187	0.271	0.787
	scarecrows	0.055	0.215	0.250	0.803
logSquirrel cocoa foraging traces per m^2	noise	-1.000	0.823	1.177	0.239
	guarding	0.122	0.324	0.366	0.715
	fire	-0.130	0.449	0.281	0.779

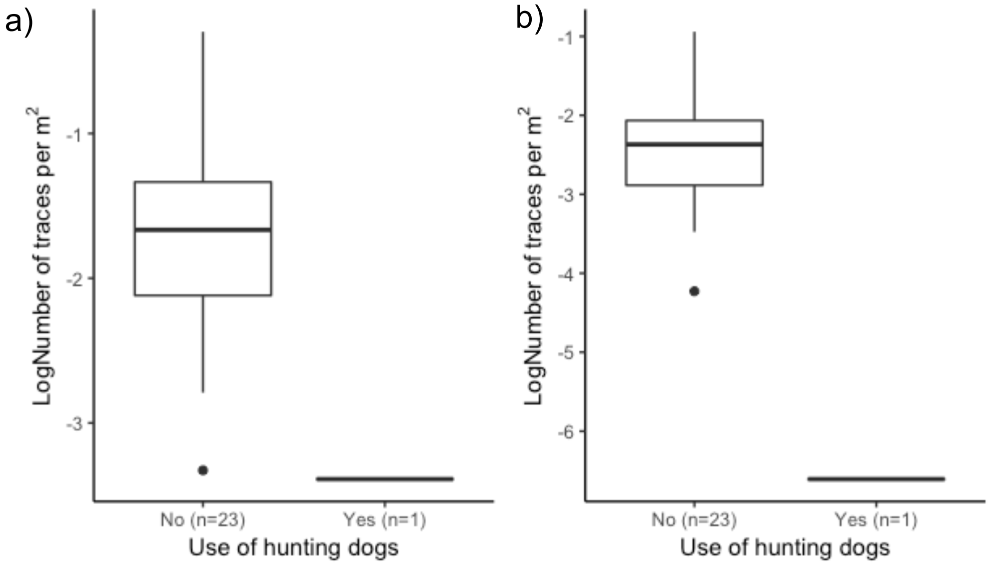


Figure 2.8: Relationship between log number of cocoa foraging traces per m^2 from plantation surveys (n=24) and significant covariates from averaging of linear models. (a) Number of traces caused by all species groups combined predicted by the use of hunting dogs and (b) number of traces caused by chimpanzees predicted by the use of hunting dogs.

2.3.4 Impact of GCPO membership

Plantations owned by members of the GCPO showed significantly lower numbers of cocoa foraging traces for all three species groups combined (Welch two sample t-test, $t_{21.62}$= 3.09, p= 0.01, Figure 2.9a), chimpanzees (Welch two sample t-test, $t_{20.78}$= 2.21, p= 0.04, Figure 2.9b) and monkeys (Welch two sample t-test, $t_{21.59}$= 3.99, p= <0.001, Figure 2.9c) than plantations not associated with the project. There was no significant effect on the number of squirrel cocoa foraging traces (Welch two sample t-test, $t_{10.91}$= -0.78, p= 0.45, Figure 2.9d).

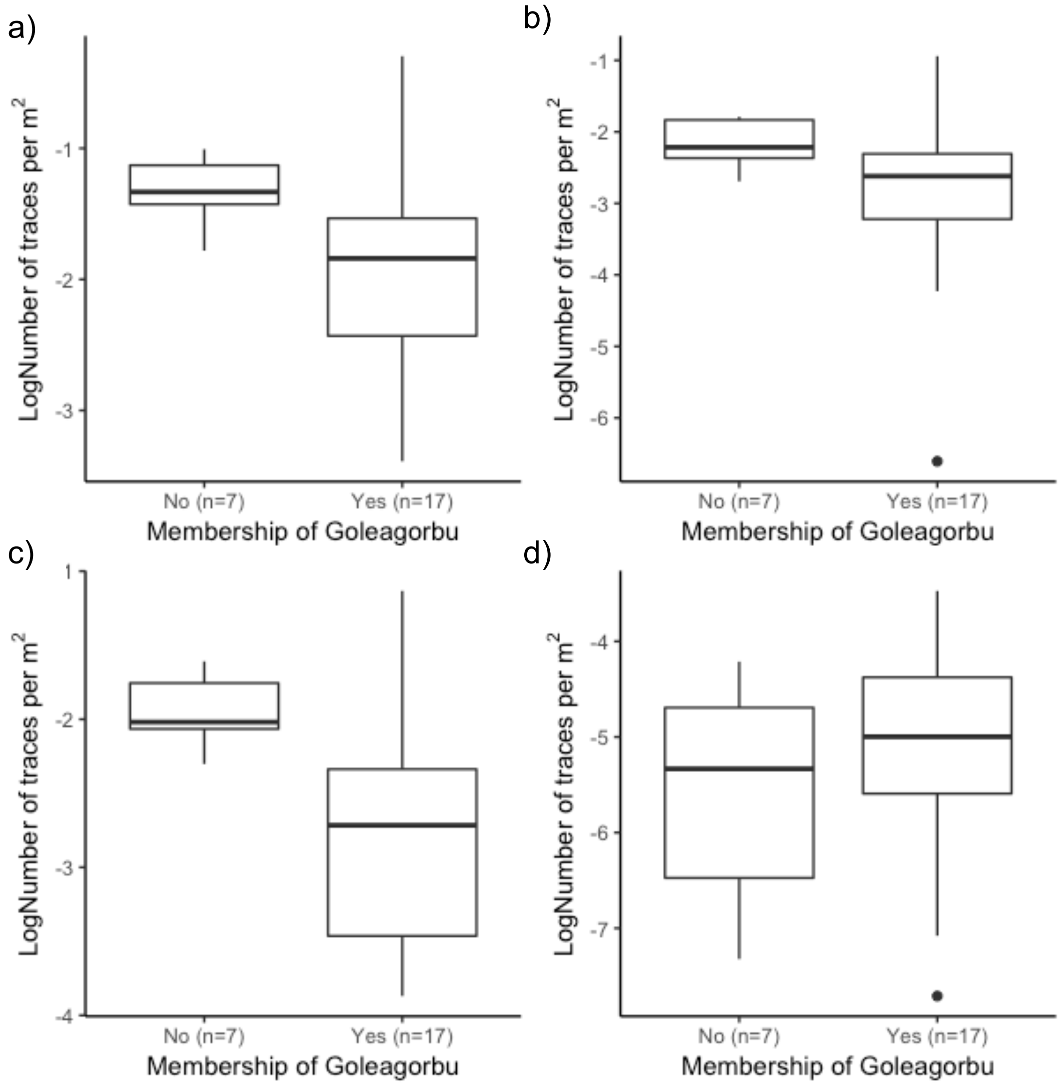


Figure 2.9: Relationship between membership of the GCPO and the log-transformed number of cocoa foraging traces per m^2 caused by (a) all species groups combined, (b) chimpanzees, (c) monkeys and (d) squirrels.

2.4 Discussion

With increasing development of the GRNP cocoa industry, it is vital to understand the impacts of wildlife cocoa foraging on human and wildlife populations and provide management strategies that support sustained conservation and development efforts. In this study we have explored the perceptions of cocoa farmers and how such perceptions relate to measured damage caused by different species groups. Potential conflict mitigation measures have also been investigated, including evaluation of spatial plantation features, crop defence strategies used by farmers and the value of integrated conservation and development initiatives.

2.4.1 Perceived risks of cocoa farming

Cocoa farmers perceived a number of risks associated with cultivating cocoa in close proximity to GRNP including financial constraints and the local climate. As observed in other areas where local economic development relies on the cultivation of commercial crops (Linkie *et al.* 2007; Webber & Hill 2014; Garriga *et al.* 2018), wildlife cocoa foraging was identified as the main risk experienced by GRNP farmers across all three surveyed chiefdoms. This finding may be due in part to the crop type in question (Hockings & Sousa 2012). For example, Spagnoletti *et al.* (2017) found that subsistence maize farmers perceived wildlife crop foraging less negatively than in our study, due to relatively low financial losses compared to losses of cash crops. Strong negative perceptions of wildlife cocoa foraging at GRNP may therefore be driven by the direct financial losses associated with reduced yields of cash crops, and the resulting increased poverty, lack of food security and health impacts (Inskip *et al.* 2013). Further indirect impacts may also reduce tolerance of wildlife foraging, including decreased mobility due to fear of animals (Mayberry *et al.* 2017).

However, other cocoa farming risks identified by GRNP farmers also result in decreased income, such as death of cocoa. Contributing to reduced yields and cocoa losses, diseases such as black pod disease are major causes of plantation abandonment across West Africa (Hislop 1966; Opoku *et al.* 2007; Arlet & Molleman 2010). Furthermore, risks such as safety concerns (i.e. lack of safety equipment) compromise human wellbeing, yet no other issue was perceived with the same incidence, severity and risk indices as wildlife cocoa foraging. As suggested by Webber and Hill (2014), such negative perceptions of crop foraging may result from the identification of crop foraging as an ‘external' issue, seemingly beyond farmer’s control. Farmers may also seek to exaggerate concerns about wildlife crop foraging and blame the conservation work of national park authorities in the hope of securing compensation (Gillingham & Lee 2003; Campbell-Smith *et al.* 2010; Karanth *et al.* 2012). It is also important to note that our study was carried out during the peak harvest of cocoa when wildlife cocoa foraging is also at its highest, perhaps resulting in heightened negative perceptions of wildlife cocoa foraging relative to other issues.

Despite issues associated with cocoa farming raised by GRNP farmers, almost all of those interviewed intended to expand their cocoa growth, highlighting the importance of developing conflict mitigation initiatives in this area. In addition, our findings highlight the importance of managing the expansion of the GRNP cocoa plantations and industry in a manner that identifies and preserves key wildlife habitat outside of the main protected forest blocks. Determining the risks farmers identify with cocoa farming aids in the prioritisation of management initiatives, and provides the opportunity to explore and deliver solutions to other cocoa farming risks identified by farmers which may bring indirect conservation benefits. For example, the introduction of effective insecticides may eradicate insect disturbance, increasing yields and potentially therefore increasing tolerance of crop losses caused by wildlife cocoa foraging (Garriga *et al.* 2018). Such initiatives must however be mindful of potentially decreasing tolerances towards wildlife further due to the high investment costs of insecticides (Knight 2000). Similarly, it is important to note that variation in cocoa productivity at different plantations was not considered as part of this study and the impact this may have on farmer’s perceptions. As an issue identified by many farmers, the loss of cocoa to disease and whether this exacerbates negative perceptions of wildlife is likely an important area for future research.

2.4.2 Perceptions of cocoa foraging species groups and measured damage

Seven species groups were identified by GRNP cocoa farmers as foraging in cocoa plantations, including ungulates, rodents and snakes. Wildlife is likely attracted to the high palatability, spatial concentration and high energy content of cocoa pulp (Riley *et al.* 2013; McLennan & Ganzhorn 2017), making it a potentially important dietary component for many species. PRM analysis shows the increased perceived risk associated with the three study focal groups, monkeys, chimpanzees and squirrels compared to other groups, supporting their inclusion as the focal groups causing the majority of cocoa damage at GRNP.

Across crop foraging studies, chimpanzees are consistently reported where they live in close spatial overlap with farming communities (Hockings & McLennan 2012; Humle & Hill 2016), likely due to high dietary flexibility and cognitive ability to access and process cocoa and mitigate the associated risks (Humle & Hill 2016). Chimpanzees were perceived as the most damaging species group by the majority of interviewed plantation owners when ranking the three study focal groups, however measured crop damage showed that monkeys caused more damage than chimpanzees across all surveyed plantations. This discrepancy between perceived and measured damage may result from the specific crop foraging behaviours of chimpanzees and how these are perceived by farmers. Chimpanzees at GRNP were described to take larger amounts of cocoa than other species every time they visit plantations, removing pods from the farm and swallowing seeds whole, leaving no seeds to harvest. In Uganda, Hill & Webber (2010) found baboons (*Papio anubis*) to be similarly disproportionately blamed for damage, likely due to perceived particularly destructive and ‘greedy’ behaviours. Furthermore, GRNP farmers described the long-term damage to cocoa trees caused by the forceful removal of cocoa pods from trees. The visibility of the scars left on trees may therefore cause farmers to hold chimpanzees to higher account than species that cause less lasting damage when crop foraging. Anthropomorphism of chimpanzees due to similarities between chimpanzees and humans in appearance and behaviour, such as descriptions of chimpanzees as ‘criminals’ by interviewed farmers, may also contribute to negative perceptions due to beliefs that chimpanzee crop foraging is driven by malice and intended harm (Root-Bernstein et al. 2013; Kansky et al. 2016). Large body size and fear of human attack, observed historically in Sierra Leone (Richards 2000) and in other West African chimpanzee populations (Hockings *et al.* 2010), may also drive disproportionately negative views of chimpanzee cocoa foraging damage. Conversely, some farmers perceived chimpanzees to cause less damage than both monkeys and squirrels, largely due to less frequent foraging events at plantations. In other studies, the presence of particularly damaging species such as baboons and cane rats also results in reduced negative tolerance of chimpanzees (Webber & Hill 2014; Garriga *et al.* 2018).

Significantly more traces attributed to monkeys were found in all plantations compared to those caused by chimpanzees and squirrels. Although sometimes considered to be the most damaging species group by farmers due to large group sizes and frequent visits to cocoa farms, monkeys were most often perceived as less damaging than chimpanzees, perhaps because monkeys were described as more scared of people than chimpanzees. Sooty mangabey, lesser spot-nosed monkey, and Campbell’s monkey were identified by farmers who were able to differentiate between species, suggesting these species in particular have the behavioural flexibility to exploit human-influenced areas and incorporate cocoa in their diets. The significant amount of damage caused by monkeys also infers the need for future studies to determine the relative contribution of different primates at the species-level, likely requiring alternative methods of assessing crop damage such as direct observation of foraging events (e.g. Tweheyo *et al.* 2005) or the deployment of camera traps (e.g. Zak & Riley 2017).

In this study, GRNP farmers also consistently stated that monkey cocoa foraging behaviours often resulted in seeds that could be harvested, unlike chimpanzee foraging. Identification of this positive aspect of monkey crop foraging, despite the large amount of damage caused by monkeys, shows the importance of forming positive associations between crop foraging and farming. Therefore, increasing awareness of chimpanzee faecal dispersal of cocoa, described by a small number of interviewed farmers and observed in Guinea (Hockings *et al.* 2017) may be important in terms of increasing tolerance towards chimpanzees. The extent of direct retaliatory actions targeted at chimpanzees at GRNP is unknown, but non-specific retaliatory actions such as traps may cause injury and mortality (McLennan *et al.* 2012; Garriga *et al.* 2018; Cibot *et al.* 2019). Slow life histories of chimpanzees and other great apes makes them particularly vulnerable to population declines caused by retaliatory actions (Humle & Hill 2016). Further potential positive impacts of wildlife crop foraging should be investigated at GRNP, such as pruning benefits (Siex & Struhsaker 1999), chimpanzee 'guarding' to prevent other crop pests (Watkins 2006; McLennan & Hill 2012), and chimpanzees piling uneaten crop parts (Hockings & Sousa 2012).

Despite causing similar amounts of damage to chimpanzees, perceptions of risk and damage associated with squirrel cocoa foraging was lower than that of chimpanzees and monkeys for most farmers. Small body size and lack of intimidation (Webber & Hill 2014), as well as predominantly nocturnal foraging described by farmers, likely reduce the visibility of such species, reducing detection of crop foraging and increasing tolerance relative to species with greater visibility (Hill 2018). Similarly, reduced visibility and less distinctive cocoa foraging behaviours likely accounts for the lower perceived risk of other rodents, snakes, ungulates and insects shown in the PRM analysis relative to monkeys, chimpanzees and squirrels. These groups also receive less conservation focus, and so perceived risk and damage may be minimised by farmers' ability to control these species with less fear of repercussions. However, as shown in our study and across crop foraging environments, the impact of small species and less visible species is often underestimated (Naughton-Treves 1998; Dickman 2010). Further work should look to examine the crop foraging behaviours of all identified crop foraging groups and quantify the relative contributions to cocoa damage.

2.4.3 Susceptibility of plantations to cocoa foraging

Spatial characteristics

Contrary to most studies that find the greatest amount of crop foraging damage at sites in close proximity to protected area boundaries (Saj *et al.* 2001; Linkie *et al.* 2007; Warren *et al.* 2007; Gubbi 2012), our findings show that plantations further from the GRNP boundary experience higher rates of cocoa foraging damage, particularly caused by western chimpanzees. This finding may reflect the dietary importance of cocoa and other crops for chimpanzees (Hockings & McLennan 2012), with chimpanzees potentially willing to travel and crop forage further from the national park boundary, particularly during periods of wild fruit scarcity (Hockings *et al.* 2009). Furthermore, this finding may also suggest that the community forests and other land uses found within the GRNP leakage belt provide sufficient habitat to support chimpanzee populations, at distances up to 4km from the national park boundary. Both Barca *et al.* (2018) and Ganas (2009) also found evidence of chimpanzee nest sites throughout the leakage belt, further highlighting the need to avoid additional fragmentation of this area and maintain important corridors between forest blocks and between protected and community forest areas.

Distance to the closest village was not an important predictor of crop foraging for any species group. Furthermore, monkey cocoa foraging was found to be higher at sites close to roads, potentially due to the use of roads as corridors (Forman & Alexander 1998; Baker & Leberg 2018). These findings are surprising in light of increased bushmeat hunting intensity and therefore predation risk associated with proximity to settlements (Brashares *et al.* 2011; Jones *et al.* 2017) and roads (Wilkie *et al.* 2000; Vanthomme *et al.* 2013), as well as direct road mortality risks (Forman & Alexander 1998; Fahrig & Rytwinski 2009). Risk mitigation behaviours, such as increased group cohesiveness (Hockings *et al.* 2012), reduced calling rates (Wilson *et al.* 2007) and reduced group size (Baranga *et al.* 2012) that inhibit detection by farmers may result in reduced fear of predation and hence allow species to forage at plantations at varying distances from villages and roads, regardless of human predation risk. Additional research is required to further explore the spatio-temporal risks of poaching on different wildlife species across GRNP, including transboundary research in Liberia, where much of the demand for bushmeat is thought to originate from (Jones et al. 2019).

Plantation perimeter length predicted the amount of cocoa foraging caused by monkeys, where crop foraging was highest at sites with shorter perimeter lengths. This finding suggests that monkeys may prefer to crop forage where plantations are smaller and the distance to surrounding forested areas is reduced, allowing for rapid evasion if detected by farmers (Wallace & Hill 2012).

As also found by Linkie *et al.* (2007) when predicting crop foraging of 11 species, perimeter length was not an important predictor of the amount of cocoa foraging damage for other species groups in this study. This suggests easy access to forested areas and plantation size are not important for reducing the levels of crop foraging for all species groups, likely due to varied risk mitigation strategies and physical characteristics that impact the fear of human predation. Additional study should look to investigate the distance travelled in to plantation interiors by different species groups (Wallace & Hill 2012).

Crop defence methods

A number of crop defence methods were identified during interviews with GRNP cocoa farmers, showing varying rates of use amongst interviewees and the plantation owners of surveyed plantations. The most commonly adopted methods were noise production and guarding, however neither were found to be effective for reducing the levels of crop foraging for any of the species group. Evidence of effective crop defence methods used by farmers is scarce (Linkie *et al.* 2007; Karanth *et al.* 2013; Junker *et al.* 2018), but in north Sumatra noise production and the use of netting were shown to successfully reduce crop losses to orangutans in the short-term (Campbell-Smith *et al.* 2012). In our study, there was some evidence that the use of hunting dogs may be successful to reduce crop foraging damage caused by all species groups combined and by chimpanzees. Whilst uncommon at GRNP, the use of hunting dogs is prevalent across other parts of Sierra Leone (i.e. 49.4% of interviewed farmers in Port Loko district, Garriga *et al.* 2018), and has been observed to prevent crop foraging by chimpanzees in Uganda (McLennan & Hill 2012). However, larger sample sizes are needed to support our evaluations of GRNP farmers' crop defence methods, particularly with regards to the use of dogs. The use of dogs to control crop foraging should be monitored to ensure their use for guarding or early detection (Hsiao *et al.* 2013), as opposed to lethal killings of wildlife. No other crop defence strategies used by GRNP cocoa farmers proved effective, perhaps because we did not account for the intensity of use of crop defences. For example, whilst the use of guarding did not predict crop foraging levels in our study, measures of guarding effort (i.e. density of guards per farm) have proved important for detection and reduced crop losses to elephants in Kenya (Sitati *et al.* 2005).

2.4.4 Evaluation of the Goleagorbu Cocoa Producers Organisation

In this study, plantations owned by members of the GCPO showed significant reductions of cocoa foraging traces compared to non-member plantations for all species groups combined, chimpanzees and monkeys. Despite the agroforestry practices of the GCPO not being directly aimed at reducing crop losses to wildlife cocoa foraging, our findings suggest they may be a vital component of conflict mitigation at GRNP. For example, effective cocoa agroforestry requires clearing of under-storey vegetation (Tscharntke *et al.* 2011), and this practice is encouraged as part of GCPO training workshops (Kate Simms, pers. comm). The increased vegetation clearing in plantations owned by members of the GCPO increases habitat openness, potentially aiding the detection of crop foraging wildlife and thereby deterring wildlife due to increased risks (Hockings & Humle 2009; Hulme *et al.* 2018). Farmers may be further encouraged to effectively manage plantations due to higher cocoa sale prices for better quality cocoa; increased time spent tending cocoa plantations likely results in increased indirect guarding effort and intensity (Katie Simms, pers. comm). However, future work should look to monitor biodiversity in plantations and ensure the decreased levels of crop foraging are not caused by farming practices moving towards ‘full sun cocoa’ (Bitty *et al.* 2015), undermining the agroforestry and ecological objectives of the GCPO.

Our findings suggest the GCPO may also be crucial in terms of changing negative perceptions towards wildlife crop foraging. The majority of members recognised that membership was beneficial for the cocoa harvest, and whilst reductions in wildlife cocoa foraging were not given as a reason for harvest increase, higher yields likely decrease the economic impact of losses to wildlife, inferring increased tolerance. The GCPO also provides a platform for communication between GRNP authorities and local communities, increasing engagement and empowerment and allowing conservation messages to be spread. Furthermore, during interviews some member farmers stated that the support provided by GCPO was a benefit that they associate with the presence of wildlife at GRNP. Local people are therefore starting to associate economic development with the conservation practices of GRNP, increasing support and tolerance, and also reducing demand for direct, often counter-productive, compensation schemes (Bulte & Rondeau 2007; Nyhus et al. 2009).

2.4.5 Conservation and management implications

The findings of this study exemplify the complex nature of investigating and mitigating negative human-wildlife interactions. Understanding the perceptions and views of GRNP cocoa farmers using SSIs provided vital information with which to inform conservation initiatives, whilst also determining the socio-economic impacts and needs for local development. Whilst a number of risks associated with cocoa farmers were identified by GRNP farmers, wildlife cocoa foraging was identified as the most significant barrier to high yields in this area presenting both a development and conservation priority. Yet tackling some of these other risks, i.e. introduction of insecticides, may prove a valuable conservation tool, resulting in increased yields which indirectly promotes greater tolerance of wildlife (Inskip *et al.* 2013; Garriga *et al.* 2018). By determining perceptions of different species groups and evaluating perceptions in the context of measured damage, this study also informs prioritisation of species for conflict mitigation and knowledge sharing programmes that appropriately address farmers concerns and beliefs (Dickman 2010; Webber & Hill 2014). For example, the disproportionate level of blame placed on chimpanzees for damage caused to cocoa plantations presents an opportunity to raise awareness of the lower amount of damage than is perceived and the potential benefits of cocoa foraging in order to change tolerances and reduce retaliatory actions.

To further mitigate conflicts resulting from wildlife cocoa foraging, plantation management that results in reduced levels of wildlife crop foraging and therefore crop losses can be informed from this study. By quantifying damage caused by different species and modelling variation in the amount of damage at plantations with different spatial characteristics our findings can advise future land use planning (Sitati *et al.* 2005; Linkie *et al.* 2007). Different spatial characteristics impacted the crop foraging patterns of the study species groups, but as monkeys were found to be the most damaging group planning should incorporate features that promote low foraging by this group such as longer perimeter lengths and greater distances from roads. However, spatial planning must ensure that there is not a shift towards clearing further community-managed forested areas or encroachment towards the national park. Furthermore, with additional evaluation of the current crop foraging methods used by GRNP farmers our findings can inform reduced use of ineffective methods, potentially providing indirect socio-economic impacts. For example, reduced time spent guarding may allow farmers to carry out other livelihoods in addition to farming and stop children missing school to guard plantations (Harrison *et al.* 2015). Furthermore, whilst there is some preliminary evidence that the use of hunting dogs may be effective to reduce wildlife crop foraging, the potentially negative impact on biodiversity associated with dogs (Doherty *et al.* 2017) presents the need for other novel forms of crop protection to be trialled and evaluated (Hockings & Humle 2009; Hill 2018). This may involve taste aversion interventions (e.g. chilli paste, Hill & Wallace 2012; coffee, O'Brien & Hill 2018) or the planting of wild foods or buffer crops to reduce the dietary significance of cultivated foods (Riley *et al.* 2013), requiring further study of the drivers of species' cocoa foraging behaviours and feeding ecologies at GRNP.

In addition, our findings provide an example of a potentially effective integrated conservation and development project, the GCPO. Through increased incentives and training in effective management, members are experiencing reduced crop foraging benefits that can be directly linked to GRNP and wildlife conservation. Our findings therefore provide evidence for the implementation of the GCPO project and practices across further communities surrounding GRNP, and as a potential model for similar projects elsewhere.

In light of potential shifts in crop foraging dynamics (i.e. seasonal patterns) and differing species-level findings, management should be adaptive, requiring regular review. Monitoring should also ensure conflicts are not displaced to surrounding plantations and that tolerance of other species are not decreased or other forms of human-wildlife conflict inflated (Dickman 2010; Hill & Wallace 2012). This study provides a replicable framework with which to monitor human-wildlife interactions regarding crop foraging and inform management that incorporates local perceptions and socio-economic development, decreases incidences of wildlife cocoa foraging and promotes tolerance towards wildlife and local conservation efforts. Studies such as this are of vital importance for continued preservation of threatened species in environments that are increasingly dominated and impacted upon by anthropogenic population growth and activities.

Chapter 3:

Spatial patterns of wildlife at Gola Rainforest National Park: Investigating the impacts of anthropogenic activities

Abstract

Understanding the presence and distribution patterns of wildlife across a landscape is fundamental to effective biodiversity conservation. Increasingly, management plans must also incorporate the role of anthropogenic activities, including hunting, deforestation and agriculture, in shaping spatial distributions. 25,000 people live in forest-edge villages close to Gola Rainforest National Park (GRNP), the largest remaining area of the Upper Guinean Forests (UGF) biodiversity hotspot in Sierra Leone and of vital conservation concern for threatened species, including the critically endangered western chimpanzee. In order to inform appropriate management initiatives, this study aimed to provide vital information about the wildlife assemblage of GRNP and determine the factors that influence the community and species-specific distribution patterns. Over 13 months and 1,255 trap days, a comprehensive camera trap survey provided the most recent assessment of GRNP wildlife with detections of 26 mammal species, including the giant forest hog previously unconfirmed to occur at GRNP and five that are globally threatened. Modelling variation in species richness across areas of the national park showed comparable species richness in the leakage belt to the protected forest blocks. This demonstrated the importance of further monitoring and land use management of the multi-use land that surrounds the national park. Variation between camera sites in ecological factors and the proximity to human threats showed no impact on community-level responses, but an occupancy modelling framework for a subset of 11 species (Maxwell's duiker, sooty mangabey, forest giant pouched rat, African brush-tailed porcupine, fire-footed rope squirrel, Ogilby's duiker, marsh mongoose, red river hog, bay duiker, Jentink's duiker and western chimpanzee) revealed different impacts and associations between species. For example, the occurrence of six species (Maxwell's duiker, fire-footed rope squirrel, Jentink's duiker, sooty mangabey, African brush-tailed porcupine and bay duiker) was impacted by one or both anthropogenic covariates (distance to closest village and distance to closest road). Our findings provide novel insights

about the abilities of different species to adapt to anthropogenic activities in this area, and highlight the importance of close collaboration with human populations in order to achieve long-term persistence of GRNP wildlife in the face of increasing and changing anthropogenic threats.

3.1 Introduction

Biodiversity conservation requires knowledge of the patterns of wildlife presence and distribution across a landscape and the factors that shape such spatial patterns (Margules & Pressey 2000; Erb *et al.* 2012; Rovero *et al.* 2017). Species distributions are based on complex interactions and trade-offs between biotic and abiotic factors (Wisz *et al.* 2013). For example, habitat features, such as microclimates (Martin 2001) can determine the presence or absence of a species at particular sites, as well as the availability of resources, including food (Barton *et al.* 1992; Wilmshurst *et al.* 2011)**,** nesting resources (Torné-Noguera *et al.* 2014; Carvalho *et al.* 2015) and water sources (Shannon *et al.* 2009). In the era of the 'Anthropocene' (Corlett 2015) and across increasingly human-dominated environments, conservation strategies must also address the contribution that anthropogenic infrastructure and activities, including hunting, agriculture and deforestation, have in shaping the distribution patterns of wildlife.

The Upper Guinean forests of West Africa (UGF) have been identified as one of 25 global biodiversity hotspots (Myers *et al.* 2000). However, the landscape has been heavily fragmented to less than 10% of the original primary forest due to increasing human populations and associated activities (Myers *et al.* 2000; Norris *et al.* 2010). Strategies to conserve remaining forested areas and maintain biodiversity in this area are therefore urgently needed and must involve close collaboration with local human populations to determine and mitigate potential negative impacts of human activities on wildlife populations (Bersacola *et al.* 2018). At the western extremity of the UGF is Gola Rainforest National Park (GRNP), the largest remaining area of lowland rainforest in Sierra Leone (Lindsell & Klop 2013) and therefore of vital conservation importance. The three forest blocks of GRNP are surrounded by a 4 km wide leakage belt which comprises a mosaic of community-managed forests, agricultural areas and human infrastructure including 122 settlements. The wildlife of GRNP, including critically endangered western chimpanzees (*Pan troglodytes verus*)*,* is therefore exposed to the activities of 25,000 people that live alongside the national park, which may impact the spatial dynamics of GRNP wildlife and present an important consideration for conservation and land use planning.

Deforestation and habitat fragmentation at GRNP has historically been driven by commercial logging, which occurred between 1960 and 1989 with greatest intensity in the southern block of the national park (Lindsell & Klop 2013; Munro & van der Horst 2013). Small-scale illegal logging continues (Hillers 2013) and further habitat modification results from the extraction of at least 39 non-timber forest products (NTFPs) for food, medicines, craft and construction (Munro & van der Horst 2013). Slash-and-burn agricultural practices largely for subsistence rice farming (Bulte *et al.* 2013), and the development of roads, contribute to further deforestation and fragmentation (Wilkie *et al.* 2000; Robinson & Bennett 2004; Vanthomme *et al.* 2013). Habitat modification and fragmentation results in disruptions to the availability and distribution of resources (McCarthy *et al.* 2017), as well as increased edge effects (Laurance *et al.* 2011), potentially causing shifts in the spatial distribution of species in order to meet their ecological needs. The ability to survive and disperse in more heavily modified sites and the spatial responses of species to the impacts of deforestation will likely depend on species-specific habitat preferences and the ecological flexibility of species (Erb *et al.* 2012).

Other agricultural practices may further impact species distributions. The introduction of novel food sources due to recent investment and expansion of commercial crop plantations and industries, such as cocoa, coffee and oil palm (Bulte *et al.* 2013), may cause increased occurrence of species that show dietary flexibility at sites close to villages and plantations (Hockings *et al.* 2012; Hill 2017b, 2018). Conversely, the resulting retaliatory actions and crop defence strategies used by farmers to prevent wildlife crop foraging (McLennan *et al.* 2012; Hill 2018; Chapter 2) may lead to spatial avoidance of areas associated with agriculture due to fear of humans as predators. Illegal bushmeat hunting also remains prevalent within and surrounding GRNP despite law enforcement and ranger patrols (Davies *et al.* 2008; Jones *et al.* 2017). Species targeted for bushmeat may experience unsustainable population losses (Brashares *et al.* 2004; Bradshaw *et al.* 2009; Ripple *et al.* 2016) and will likely adjust behaviour and distribution patterns to avoid the higher hunting intensity and therefore predation risk associated with proximity to human settlements (Brashares *et al.* 2011; Jones *et al.* 2017). In addition, roads may show similar effects on wildlife as they increase access to forest interiors for hunters (Wilkie *et al.* 2000;

Vanthomme *et al.* 2013) and carry direct mortality risks (Forman & Alexander 1998; Fahrig & Rytwinski 2009).

Widespread ecological impacts can result from changes to wildlife occurrence patterns, including disruptions to species interactions (Tylianakis *et al.* 2007) and disease dynamics (Bradshaw *et al.* 2009), as well as decreased plant regeneration (Effiom *et al.* 2013). It is therefore vital to understand the community-level and species-specific responses of wildlife to human activities in order to inform conservation that incorporates and promotes positive long-term human-wildlife coexistence, requiring up-to-date assessments of the wildlife present at GRNP and species distribution patterns.

Carrying out surveys of elusive tropical forest species over large spatial scales can be resource and time consuming, particularly where species occur at low densities. Non-invasive camera traps are increasingly used over traditional monitoring methods, such as line transects (Plumptre 2000; Kouakou *et al.* 2009), to provide information that can be used to understand patterns of community composition and species distributions (Tobler *et al.* 2008; Burton *et al.* 2015; Wearn & Glover-Kapfer 2019). Furthermore, the placement of camera traps across a heterogenous environment allows for comparisons between sites at different proximities to human disturbance and activities, and with different ecological characteristics. The impacts of such variation can be assessed at the community-level through comparisons of species richness and species-specific trends can be identified using an occupancy modelling framework. Occupancy modelling accounts for imperfect detection (where species might be present but not detected) by using detection/non-detection data over repeated sampling occasions to derive detection probability (MacKenzie *et al.* 2002). Occupancy probability, the probability that a species occurs at a site, can then be estimated and modelled as a function of anthropogenic and ecological site characteristics (MacKenzie *et al.* 2002, 2006, 2017). Occupancy has been used widely in ecological research across a range of scales and environments (e.g. Burton *et al.* 2012; Erb *et al.* 2012; McCollum *et al.* 2017, 2018; Rich *et al.* 2017; Van der Weyde *et al.* 2018) and is increasingly used in tandem with camera trapping (Burton *et al.* 2015; Sollmann 2018).

In this study, a camera trap survey was used to determine the factors that influence the patterns of wildlife presence and distribution at GRNP and inform appropriate management initiatives. We aimed to: (1) provide an updated baseline assessment of the mammal community at GRNP, (2) determine variation in species richness across and surrounding GRNP, and (3) assess the fine-scale impacts of anthropogenic and ecological factors on species richness and species-specific occurrence at different sites. The following hypotheses and predictions were tested: (1) species richness would vary between the areas of the national park, with highest species richness expected in Gola Central compared to more heavily logged Gola South and the leakage belt, and (2) species richness and species occupancy would largely decline with increased proximity to human infrastructure, but variation was expected between species depending on ecological characteristics of sites and species' habitat requirements and abilities to persist alongside human activities.

3.2 Methods

3.2.1 Study site

The study was conducted at Gola Rainforest National Park (GRNP, Figure 3.1), a 710km^2 area of primary forest covering parts of the Kailahun, Kenema and Pujehun districts of south-east Sierra Leone and bordering Liberia (between 7°18′ and 7°51′N and 10°37′ and 11°21′W). GRNP is the largest area of the Upper Guinean forests of West Africa remaining in Sierra Leone and is made up of lowland moist evergreen high forest (Lindsell *et al.* 2011). GRNP comprises three forest blocks, Gola South, Gola Central and Gola North, surrounded by a 4km wide leakage belt (truncated to the east by the Liberian border). The leakage belt is made up of agricultural land, community-managed forested areas and anthropogenic infrastructure, including minor and major roads and 122 villages inhabited by approximately 25,000 people (Bulte et al. 2013). Previous line transect surveys (Lindsell et al. 2011) and unpublished camera trap surveys (Koroma 2012; Hillers 2013; Keifala 2013) suggest GRNP contains over 50 species of medium to large bodied size mammals (Table S1).

3.2.2 Camera trap survey

Reconyx PC800 Hyperfire motion sensor camera traps were deployed in 33 x 1 km^2 plots (Figure 3.1) between December 2017 and January 2019 in Gola Central (n= 21), Gola South (n= 8) and in the leakage belt between Gola Central and Gola North (n= 4). Plots were selected based on previous surveys of the whole national park carried out in 2011 and 2013 which revealed potential for high biodiversity at these sites (Hillers 2013) and a more comprehensive sampling of the whole national park was limited by the GRNP research budget and resources. Due to disruptions to the standard camera trap sampling schedule of GRNP and poor accessibility, cameras were not placed in or retrieved from Gola North with sufficient time to allow inclusion in this study. Cameras were moved between sites and were deployed at each site once during the study period. One unbaited camera was attached to a tree within a 10- metre radius of each plot centre, 30 - 50cm above ground level. Cameras were programmed to take three photos per trigger event with a <1 second interval between photos and no delay between trigger events. Cameras were functional 24 hours per day and deployed for between 36 and 51 days (mean days= 41.03, SD= 7.91). Three of the cameras malfunctioned during deployment and for subsequent analysis photos from 30 sites were used (Gola Central n= 20, Gola South n= 7, and leakage belt n= 3), resulting in a total of 1,255 trap days.

Data management

Species identification of camera trap photos was carried out manually and all images of humans were deleted immediately (Sandbrook *et al.* 2018). Intervals of 60 minutes were used to distinguish between independent events. The filename of each photo was renamed to include the species name and organised into species folders within specific camera trap site folders. The resulting file directory was used with R package *camtrapR* (Niedballa *et al.* 2016) to derive descriptive information. Naïve occupancy for each species was calculated as the proportion of sites it was detected at on at least one occasion. Relative detection frequency (trap rate per 100 days) was calculated as the number of independent detections divided by the total number of trap days across all cameras, multiplied by 100. For occupancy analyses, *camtrapR* was also used to create binary detection histories for each species showing detection (1) and non-detection (0) at each site during each sampling period. Sampling periods of 6 days were used (where increased accuracy of occupancy estimates results from highest number of sampling periods but whilst also ensuring sufficient detections within detection histories of species for model convergence) (MacKenzie *et al.* 2002; Burton *et al.* 2015), resulting in a range of 4- 9 periods per site, depending on the number of trap days.

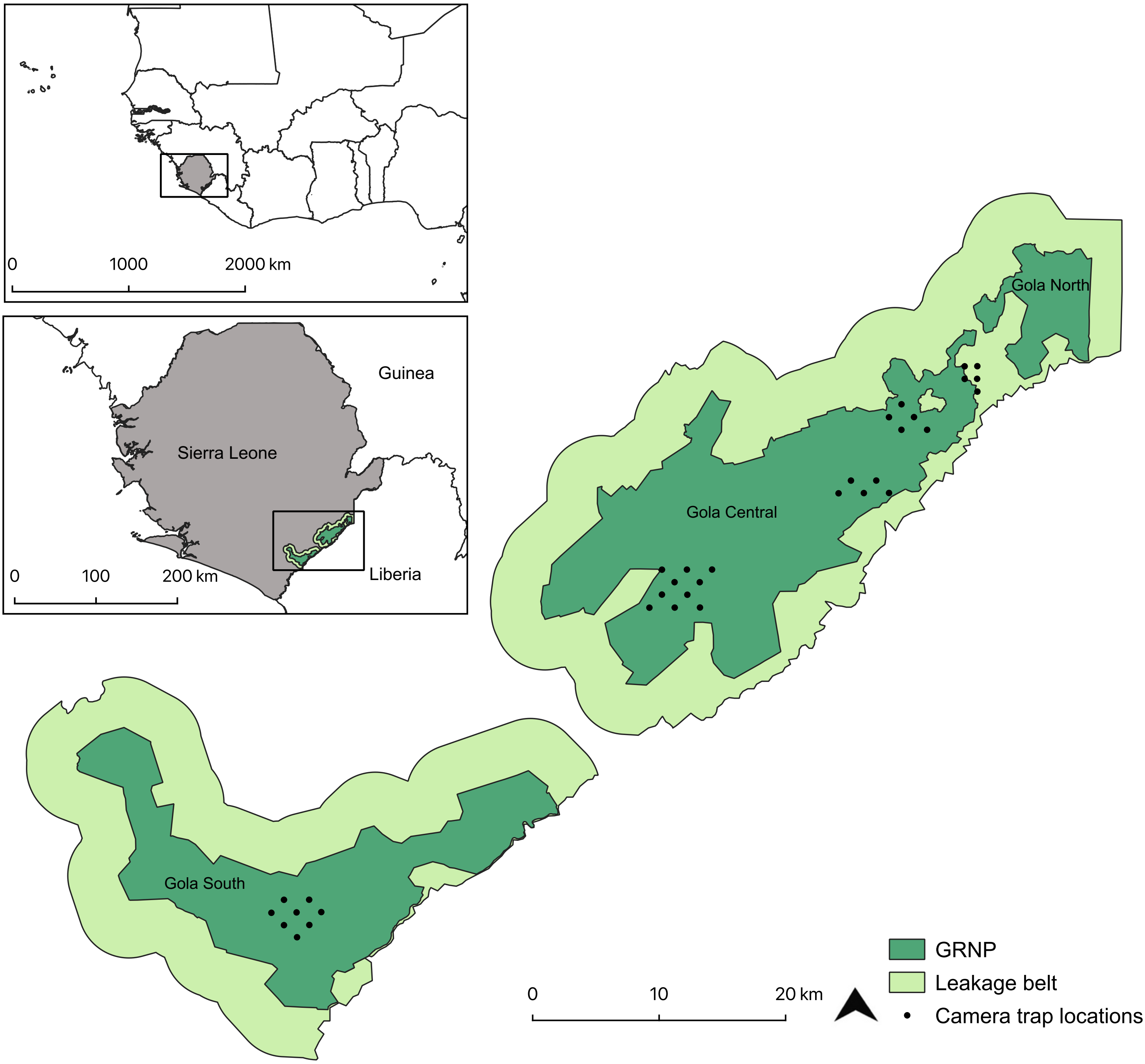


Figure 3.1: Location of camera trap sites (n=33) at Gola Rainforest National Park (GRNP) showing the three protected forest blocks, Gola South, Gola Central and Gola North and the surrounding leakage belt. Location of GRNP within West Africa and Sierra Leone inset.

3.2.3 Statistical analyses

Anthropogenic and ecological covariates

During deployment, the number of trees of all DBH (diameter at breast height) classes was counted within a 5- metre radius of each camera site. This count was divided by area (78.54 m^2) to give an estimation of tree density (per m^2) at each site. Other covariates were derived using nearest neighbour analysis in QGIS (version 3.6.3, QGIS Development Team 2019) with plugin *nnjoin.* The minimum distance from each camera trap site to the closest permanent water source (i.e. stream or river) was calculated for each site as a further habitat feature. Due to incomplete habitat assessments during camera deployment and collection no further ecological covariates (e.g. canopy cover, NDVI) were available for inclusion in analyses. The minimum distance from each camera trap site to the closest village (permanent settlement) and minimum distance to the nearest road (including both major and minor roads) were calculated as proxies of human disturbance and activities. The minimum distance from each site to the national park boundary was also considered as a covariate, but due to significant correlation with distance to closest village (Shapiro wilks test, r^2= 0.60, p= <0.001), it was excluded from analysis to avoid issues of multicollinearity. All covariates were standardised by subtracting the mean and dividing by the standard deviation to give z scores used for further analysis.

Species richness

All statistical analyses were conducted in R studio (version 1.1.463, R Core Team 2018). Species richness was estimated for each site as the number of species detected. To control for variation in sampling effort (number of trap days), species richness at each site was divided by the shortest number of trap days (across all sites) to give standardised species richness. A one-way ANOVA was used to determine whether mean standardised species richness varied significantly between the three surveyed areas, Gola Central, Gola South and the leakage belt.

A linear model was used to determine the impacts of site-level variation in anthropogenic and ecological covariates on species richness. Standardised species richness was fit as the response term. Distance to the closest village, distance to the closest road, tree density and distance to the closest permanent water source were fit as explanatory terms. Models were fit with all combinations of fixed effects using the R package *MuMIn* (Barton 2019) and ranked based on Akaike's information criterion corrected for small sample size (AICc). Model averaging of the top models (ΔAICc <2) was then used for significance testing.

Occupancy modelling framework

The R package *unmarked* (Fiske & Chandler 2011) was used to run single-species single-season occupancy models for the 11 species with sufficient detections for model convergence (independent detections ≥10) in order to determine species-specific responses to site-level variation in anthropogenic and ecological covariates. Distance to the closest village, distance to the closest road, tree density and distance to the closest permanent water source were considered as possible predictors of both detection and occupancy probability. For each species, a two-step approach was used for model selection due to the large number of possible models. First, occupancy probability was held constant ($\Psi(.)$) in order to determine the combinations of covariates that best predicted detection probability. The covariates in the resulting top model (based on AICc ranking) were then held constant for testing of combinations of occupancy covariates. The R package *MuMIn* was used in combination with *unmarked* to fit all combinations of covariates in each step. The resulting occupancy models were ranked based on Akaike's information criterion corrected for small sample size (AICc). Model averaging of the top model set (ΔAICc <2) was then used to determine the relative significance of covariates for predicting the occupancy probability of each species by summing the Akaike's model weights (AICw) of those that contained each covariate. Mean occupancy probabilities (proportion of sites where species occurs, accounting for imperfect detection) for each species were determined by taking an average of back-transformed model estimates whilst holding covariates to their mean values.

3.3 Results

3.3.1 Camera trap survey

Across 1,255 total trap days, a total of 425 independent detections were recorded of 26 mammal species from 4 taxonomic groups (Figure 3.2). The number of independent detections per camera ranged from 1-82 (mean ± SD= 14.17 ± 16.12) and the number of species captured varied from 1-12 (mean ± SD= 4.5 ± 2.57). The most commonly detected species (128 independent detections) was Maxwell's duiker (Table 3.1). Other species detected ≥10 times were sooty mangabey, forest giant pouched rat, African brush-tailed porcupine, fire-footed rope squirrels, Ogilby's duiker, marsh mongoose, red river hog, bay duiker, Jentink's duiker and western chimpanzee. 15 species were detected <10 times, of which eight were detected once. Naïve occupancy and relative detection frequency for each species are shown in Table 3.1 and Figure S6. Although not included in any further analysis, there were detections of two bird species, white-breasted guineafowl (*Agelastes* meleagrides) and crested guineafowl (*Guttera verreauxi*).

Figure 3.2: Examples of captured images from camera trap (n=30) survey at Gola Rainforest National Park. (a) Maxwell's duiker, (b) sooty mangabeys, (c) African brush-tailed porcupine, (d) western chimpanzee, (e) red river hog, and (f) Jentink's duiker

Table 3.1: List of mammal species detected during camera trap survey of Gola Rainforest National Park, Sierra Leone. Number of independent detections, relative detection frequency (trap rate per 100 days) and naïve occupancy (proportion of total sites with at least one detection) shown for each species.

	Species (by Order)	**IUCN status**	**Independent detections**	**Relative detection frequency**	**Naïve occupancy**
Primates					
1	Sooty mangabey *Cercocebus atys*	NT	58	4.62	0.60
2	Western chimpanzee *Pan troglodytes verus*	CR	10	0.80	0.17
3	Western red colobus *Piliocolobus badius*	EN	2	0.16	0.07
4	Diana monkey *Cercopithecus diana*	VU	1	0.08	0.03
5	Olive colobus *Procolobus verus*	NT	1	0.08	0.03
Carnivora					
6	Marsh mongoose *Atilax paludinosus*	LC	16	1.27	0.27
7	Pardine genet *Genetta pardina*	LC	6	0.48	0.13
8	Honey badger *Mellivora capensis*	LC	2	0.16	0.07
9	African civet *Civettictis civetta*	LC	1	0.08	0.03
10	African palm civet *Nandinia binotata*	LC	1	0.08	0.03
Cetartiodactyla (even-toed ungulates)					
11	Maxwell's duiker *Philantomba maxwelli*	LC	128	10.20	0.67
12	Ogilby’s duiker *Cephalophus ogilbyi*	LC	20	1.59	0.17
13	Red river hog *Potamochoerus porcus*	LC	14	1.12	0.20
14	Bay duiker *Cephalophus dorsalis*	NT	12	0.96	0.20
15	Jentink’s duiker *Cephalophus jentinki*	EN	11	0.88	0.17
16	Zebra duiker *Cephalophus zebra*	VU	5	0.40	0.13
17	Bongo *Tragelaphus eurycerus*	NT	2	0.16	0.07
18	Water chevrotain *Hyemoschus aquaticus*	LC	3	0.24	0.03
19	Giant forest hog *Hylochoerus meinertzhageni*	LC	1	0.08	0.03
20	Black duiker *Cephalophus niger*	LC	1	0.08	0.03
Rodentia					
21	Forest giant pouched rat *Cricetomys emini*	LC	54	4.30	0.47
22	African brush-tailed porcupine *Atherurus africanus*	LC	48	3.82	0.30
23	Fire-footed rope squirrel *Funisciurus pyrropus*	LC	22	1.75	0.27
24	Slender-tailed squirrel *Protoxerus aubinnii*	NT	4	0.32	0.13
25	Red-legged sun squirrel *Heliosciurus rufobrachium*	LC	1	0.08	0.03
26	Striped ground squirrel *Xerus erythropus*	LC	1	0.08	0.03

IUCN status: LC= Least concern, NT= Near threatened, VU= Vulnerable, EN= Endangered, CR= Critically endangered

3.3.2 Species richness across GRNP

When standardised by shortest sampling effort, there was no statistically significant variation in mean species richness across the three surveyed areas of the national park (Figure 3.3, one-way ANOVA, $F_{2,17}$= 0.80, p= 0.46).

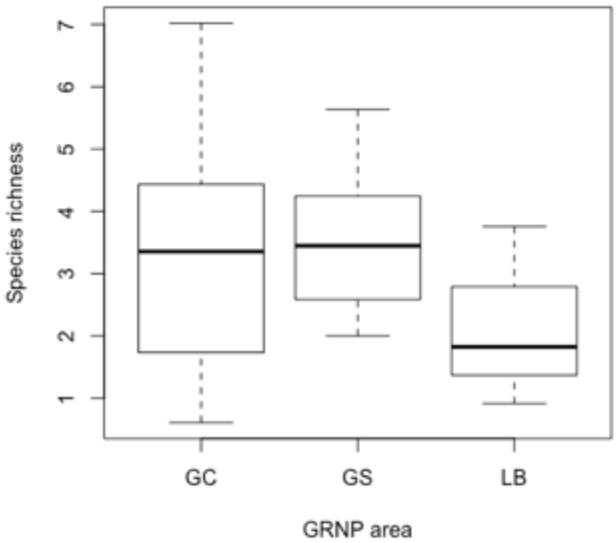


Figure 3.3: Standardised mean species richness (standardised by shortest sampling effort) at three locations of Gola Rainforest National Park. GC= Gola Central, GS= Gola South, LB= Leakage belt between Gola Central and Gola North.

3.3.3 Impacts of site-level anthropogenic and ecological covariates

Species richness

Between the camera trap sites, standardised species richness ranged from 0.61-7.02 (mean ± SD= 3.28 ± 1.63). The top model set (ΔAICc <2) for predicting standardised species richness contained two models (Table 3.2). Based on AICc ranking, the null model was most highly supported, and the second contained the covariate distance to the closest village. Model averaging showed support for a non-significant positive effect of distance to the closest village on species richness (Figure 3.4, p= 0.61).

Table 3.2: Summary table of top models (ΔAICc<2) derived from linear model with standardised species richness as response term. + indicates inclusion of covariate in model.

	village	**road**	**trees**	**water**	**df**	**AICc**	**ΔAICc**	**AICw**
Standardised species richness					2	117.90	0.00	0.63
	+				3	118.96	1.06	0.37

Species richness standardised by shortest number of trap days. All covariates standardised to z scores. Abbreviations: village= distance to closest village, road= distance to closest road, trees= tree density, water= distance to closest permanent water source.

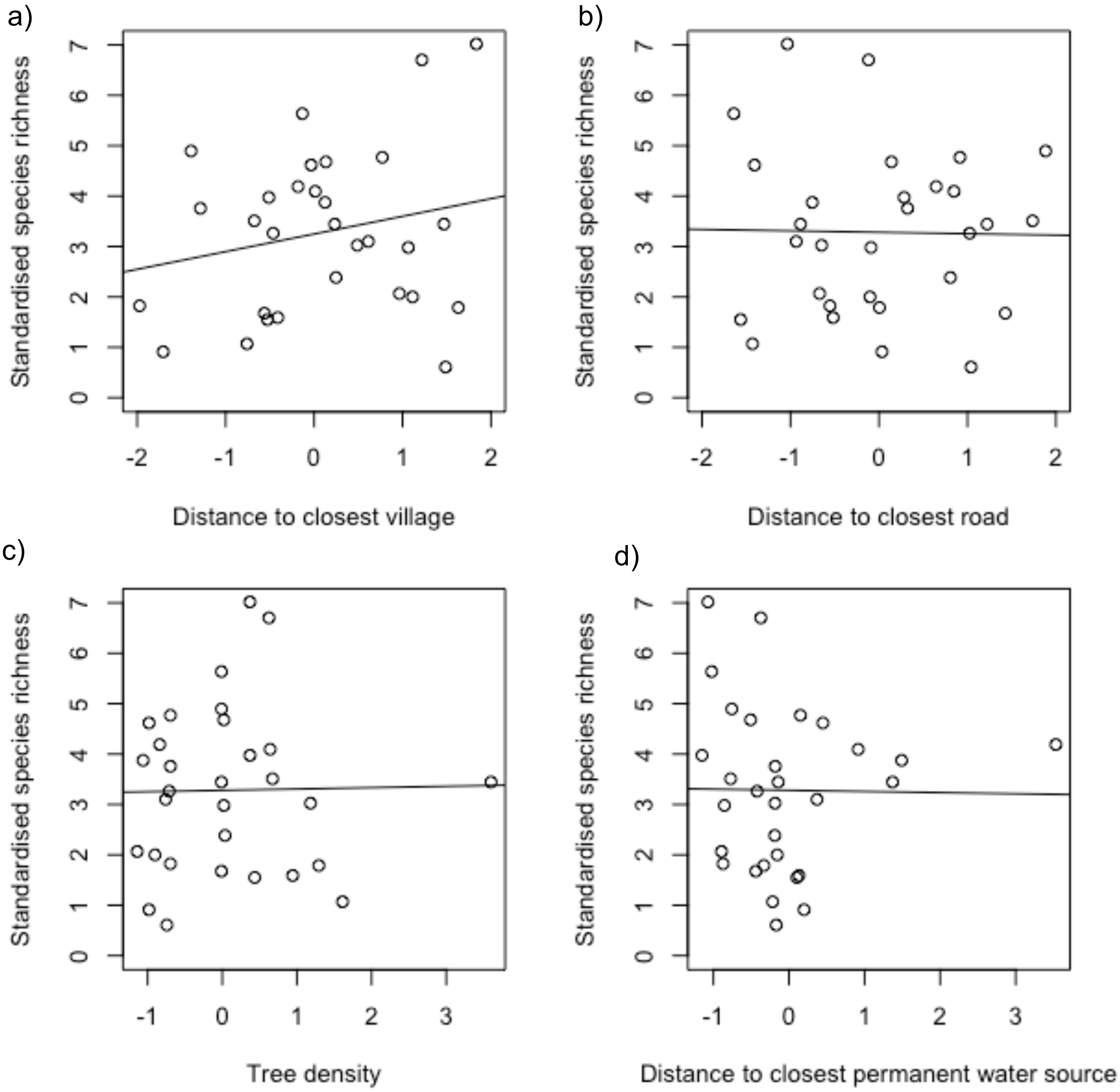


Figure 3.4: Effects of anthropogenic and habitat covariates on standardised species richness (standardised by shortest sampling effort) at camera trap sites in Gola Rainforest National Park. All covariates standardised to z scores, (a) distance to closest village, (b) distance to closest road, (c) tree density and (d) distance to closest permanent water source.

Species occupancy

Occupancy modelling for the species with sufficient detections resulted in a model set of 29 models across 11 species (Table 3.3). Detection probability was predicted by different combinations of covariates for each species. The most common predictors of detection probability were distance to the closest village and distance to the closest permanent water source, each appearing in the top detection model for three species. Both distance to the closest road and tree density featured in the top detection model for two species. Model derived estimates of mean detection probability ranged from 0.10- 0.43 (Table 3.4).

For each species, different combinations of covariates featured in the top model set for predicting occupancy probability (ΔAICc <2, Table 3.3). The relative significance of covariates and the direction of effects varied between species (Table 3.4). Overall, distance to the closest village and distance to the closest permanent water source were found to be the most common predictors of occupancy probability across the 11 species. Mean model-derived estimates of occupancy probability for the 11 species ranged from 0.17- 0.79 (Table 3.4) and was higher than naïve occupancy for 10 of the 11 species (Figure S8).

Distance to the closest village was found to impact the occupancy probability (appeared in top model set) of six species. Maxwell’s duiker, fire-footed rope squirrel and Jentink’s duiker showed higher occupancy at sites further from villages, whereas sooty mangabey, African brush-tailed porcupine and bay duikers showed higher occupancy closer to villages. This covariate showed high relative significance (AICw >0.5) for predicting occupancy of African brush-tailed porcupine (Figure S7). Distance to the closest road impacted occupancy probability of four species. Occupancy of Maxwell’s duiker and sooty mangabey was higher at sites closer to roads, whereas fire-footed rope squirrel and bay duiker showed higher occupancy at sites further from roads. Tree density contributed to the top model set for four species, showing higher occupancy of Maxwell’s duiker, African brush-tailed porcupine, fire-footed rope squirrel and marsh mongoose at sites with higher tree density. This covariate showed high relative significance for predicting occupancy probability of Maxwell’s duiker, African brush-tailed porcupine and marsh mongoose. Distance to the closest permanent water source impacted occupancy for six species.

Sooty mangabey and chimpanzees occurred at higher rates closer to permanent water sources, whereas African brush-tailed porcupine, fire-footed rope squirrel, marsh mongoose and red river hog showed the opposite response. This covariate showed high relative significance for predicting occupancy of African brush-tailed porcupine.

Table 3.3: Top model set (ΔAICc <2) for occupancy modelling of species with 10 or more independent detections during camera trap survey of Gola Rainforest National Park. Best model for predicting detection probability (p) that was held constant when investigating occupancy probability shown first for each species. Models ranked by Akaike's Information Criterion corrected for small sample size (AICc). K: Number of parameters in model, ΔAICc: delta AICc, AICw: Akaike's model weights.

Species	Model	K	AICc	ΔAICc	AICw
Maxwell's duiker	*p(water)*				
	Ψ(trees)	4	228.80	0.00	0.36
	Ψ(road, trees)	5	230.21	1.41	0.18
	Ψ(.)	3	230.37	1.57	0.16
	Ψ(village, trees)	5	230.45	1.65	0.16
	Ψ(village)	4	230.66	1.86	0.14
Sooty mangabey	*p(road)*				
	Ψ(.)	3	201.10	0.00	0.31
	Ψ(village, water)	5	201.33	0.23	0.27
	Ψ(water)	4	202.38	1.28	0.16
	Ψ(road)	4	202.64	1.54	0.14
	Ψ(village)	4	202.97	1.87	0.12
Forest giant pouched rat	*p(village, trees)*				
	Ψ(.)	4	163.72	0.00	1.00
African brush-tailed porcupine	*p(trees)*				
	Ψ(village, trees, water)	6	89.30	0.00	1.00
Fire-footed rope squirrel	*p(.)*				
	Ψ(.)	2	94.50	0.00	0.37
	Ψ(water)	3	95.84	1.34	0.19
	Ψ(road)	3	96.19	1.69	0.16
	Ψ(trees)	3	96.34	1.84	0.15
	Ψ(village)	3	96.35	1.85	0.15
Ogilby's duiker	*p(village, road, water)*				
	Ψ(.)	5	75.90	0.00	1.00
Marsh mongoose	*p(water)*				
	Ψ(trees)	4	90.90	0.00	0.35
	Ψ(trees, water)	5	90.94	0.04	0.35
	Ψ(.)	3	91.21	0.31	0.30

Red river hog	*p(.)*				
	Ψ(.)	2	84.00	0.00	0.53
	Ψ(water)	3	84.25	0.25	0.47
Bay duiker	*p(village)*				
	Ψ(.)	3	79.50	0.00	0.52
	Ψ(village, road)	5	80.68	1.18	0.29
	Ψ(road)	4	81.46	1.96	0.19
Jentink's duiker	*p(.)*				
	Ψ(.)	2	71.20	0.00	0.66
	Ψ(village)	3	72.50	1.30	0.34
Western chimpanzee	*p(.)*				
	Ψ(.)	2	58.40	1.00	0.65
	Ψ(water)	3	59.60	1.20	0.35

Abbreviations: village= distance to closest village, road= distance to closest road, trees= tree density, water= distance to closest permanent water source. All variables standardised to z-scores.

Table 3.4: Model-averaged estimates of mean occupancy and detection probabilities (± SE) for each species, and the summed AIC weights of anthropogenic and habitat covariates appearing in the top model set for predicting occupancy probability of each species. Covariates with high model support for impacting occupancy (summed AICw >0.5) shown in bold. Direction of effect shown in parentheses.

			Occupancy covariates			
Species	**Mean Ψ (± SE)**	**Mean p (± SE)**	**village**	**road**	**trees**	**water**
Maxwell's duiker	0.794 (0.112)	0.428 (0.049)	0.30 (+)	0.18 (-)	**0.69 (+)**	
Sooty mangabey	0.749 (0.110)	0.316 (0.049)	0.39 (-)	0.14 (-)		0.43 (-)
Forest giant pouched rat	0.607 (0.124)	0.238 (0.054)				
African brush-tailed porcupine	0.310 (0.098)	0.101 (0.032)	**1.00 (-)**		**1.00 (+)**	**1.00 (+)**
Fire-footed rope squirrel	0.248 (0.087)	0.328 (0.078)	0.14 (+)	0.16 (+)	0.15 (+)	0.19 (+)
Ogilby's duiker	0.217 (0.087)	0.305 (0.082)				
Marsh mongoose	0.484 (0.212)	0.123 (0.061)			**0.70 (+)**	0.35 (+)
Red river hog	0.328 (0.134)	0.174 (0.069)				0.47 (+)
Bay duiker	0.315 (0.139)	0.112 (0.046)	0.29 (-)	0.48 (+)		
Jentink's duiker	0.186 (0.086)	0.252 (0.086)	0.34 (+)			
Chimpanzee	0.173 (0.105)	0.172 (0.083)				0.35 (-)

Abbreviations: village= distance to closest village, road= distance to closest road, trees= tree density, water= distance to closest permanent water source.

3.4 Discussion

3.4.1 Mammal assessment

Prior to this study, the most recent published assessment of mammal species found within GRNP was conducted between 2005 and 2007 by Lindsell *et al.* (2011), and involved line transect surveys requiring direct observations of species or species traces. Lindsell *et al.* (2011) found evidence of 44 mammal species, of which 22 were detected in the current study along with the addition of four further species: honey badger, forest giant-pouched rat, Jentink's duiker and giant forest hog (Table S1). The presence of honey badgers, giant pouched rats and Jentink's duiker at GRNP has previously been confirmed during early, but largely unpublished, camera trap deployment (Ganas & Lindsell 2010; Hillers 2013). However, the current survey provides the first reported evidence for the presence of the giant forest hog at GRNP, a previously unconfirmed species (Merz 1986; Klop *et al.* 2008). Five of the 12 primates previously found at GRNP were detected during this survey, with relatively frequent detections of semi-arboreal sooty mangabeys and western chimpanzees. Arboreal species were detected less readily. Other notable species not detected were the highly elusive pygmy hippopotamus, an endangered species of particular conservation priority for GRNP which has been previously recorded using stream surveys and camera traps (Hillers 2013; Keifala 2013), and African forest elephants, providing potential further evidence of population declines across the national park (Merz 1986; Lindsell *et al.* 2011). These findings suggest that whilst GRNP supports a diverse mammal community, further monitoring is needed to ensure species that were not detected in this survey are still present and not declining. Adjustments to the sampling protocol such as the placement of cameras to target particular species, increased sampling effort (i.e. trap days, Tobler *et al.* 2008) and the use of arboreal cameras (e.g. Whitworth *et al.* 2016; Bowler *et al.* 2017) may increase the number of species detected and provide further data for investigation of species and community-level spatial dynamics.

3.4.2 Species richness across GRNP

The leakage belt of GRNP was found to support a species richness similar to that of Gola Central and Gola South. This suggests that despite the increased fragmentation and degradation of the leakage belt, and decreased law enforcement and protection compared to the two forest blocks, wildlife is able to survive and may be attracted to the leakage belt. This differs from findings of other studies that show reductions in species richness in areas adjacent to national parks and areas with reduced levels of protection (Rich *et al.* 2016; Hegerl *et al.* 2017). It is important to note that only three of the camera trap sites were located outside of the national park and each was positioned within 1km from the GRNP boundary. Whilst our findings would therefore benefit from a more comprehensive camera trap survey of the leakage belt with cameras placed across different land uses of this area, we provide evidence for the conservation value of the leakage belt and the importance of managing future land use changes and collaborating with local communities to ensure the preservation of this area of important multi-use habitat.

There was also no difference in species richness between the Gola Central and Gola South forest blocks despite the greater pressure of historical logging in Gola South (Lindsell & Klop 2013; Munro & van der Horst 2013). More intensively logged areas of tropical forest have been shown to have reduced species richness, for example avian species richness across the tropics shows declines in logged areas with little evidence of recovery to pre-logging levels (Lamanna & Martin 2016), and across the Brazilian Amazon, mammal and bird species richness declines by up to 8 species for every additional 10% of forest cover lost (Ochoa-Quintero *et al.* 2015). Conversely, our findings suggest historical logging might not have resulted in long-term reductions to mammalian species richness, however potential impacts on community composition and the relative abundance of species should be explored (Lamanna & Martin 2016).

3.4.3 Anthropogenic impacts on species richness and species occupancy

Proximity to villages

Lower species richness was predicted closer to the 122 forest edge villages within the GRNP leakage belt due to increased proximity to anthropogenic activities and threats. Across tropical Africa where the demand for bushmeat continues to grow (Ripple *et al.* 2016), proximity to human settlements is associated with increased hunting likelihood and intensity (Laurance *et al.* 2006; Nielsen 2006; Brashares *et al.* 2011; Lindsey *et al.* 2013; Nuno *et al.* 2013; Koerner *et al.* 2017). Jones *et al.* (2017)

reported evidence of the same relationship within GRNP with hunting traces (human trails, snares, shot gun cartridges and overnight camps) found at higher rates closer to villages. Proximity to GRNP villages is also associated with increased human activity due to the collection of NTFPs (Munro & van der Horst 2013) and the potential retaliation of farmers due to wildlife crop foraging (Chapter 2). However, our findings show that although distance to the closest village was the only covariate to impact species richness, it showed a non-significant effect, suggesting hunting pressure and other human activities do not have a clear negative impact on species richness, contrary to the findings of other studies (e.g. Koerner *et al.* 2017).

Species-specific occupancy probabilities were also predicted to decline with proximity to villages. Lower occupancy estimates at sites closer to villages were observed for Maxwell's duiker, fire-footed rope squirrel and Jentink's duiker. Maxwell's duiker is the most commonly hunted species from GRNP based on bushmeat market surveys and analysis of hunting bags (Davies *et al.* 2008; Koroma 2012). Therefore, despite showing the highest overall occupancy of any detected species it is likely the fear of human predation results in lower occurrence at sites with higher human predation risk due to avoidance behaviours or direct population declines caused by hunting. Similar patterns of village avoidance have been observed across tropical forest environments by African forest elephants (Buij et al. 2007), collared mangabey (Vanthomme *et al.* 2013) and two Amazonian monkey species (Bowler *et al.* 2017). Although Jentink's duiker was the only threatened species of the three showing low occupancy close to villages in this study, the depletion or spatial avoidance of species at sites close to villages may have indirect and widespread effects. For example, the loss of important prey species may have cascading impacts on populations of predatory species in these areas (Ripple *et al.* 2016).

Contrary to our general predictions, the top model sets for sooty mangabey, African brush tailed porcupine and bay duiker showed distance to the nearest village negatively impacting occupancy probability. Despite being a relatively prevalent species in local bushmeat markets (Davies *et al.* 2008), African brush tailed porcupine showed higher occupancy closer to villages, suggesting they may rely on nocturnal activity patterns to avoid human predation. Tolerance of this species to occur in heavily hunted areas was also found by Laurance *et al.* (2006) and it has also been observed with high occupancy in agricultural areas (Vanthomme *et al.* 2013). Attraction to agricultural resources may account for the higher occupancy of sooty mangabey, a frequently identified cocoa foraging species by GRNP farmers (Chapter 2), at sites closer to villages. With further camera trapping of the leakage belt and an increase in cameras close to villages (in this study the shortest distance was 1 km from the nearest village) we would expect to increase detections of other crop foraging species such as western spot-nosed monkey and Campbell's monkey.

It was also found that the occupancy probability of five species showed no response to the proximity of human settlements, suggesting these species are not targeted by hunters and can occupy sites with varying human predation risk. For example, forest giant pouched rat and marsh mongoose contributed less than 1 and 0% respectively to sales at bushmeat markets close to GRNP (Davies *et al.* 2008). Neutral responses of species to human activities may also result from behavioural adaptations to mitigate risks associated with proximity to villages, for example increasing vigilance behaviours (Ciuti *et al.* 2012), adjusting grouping patterns (Hockings *et al.* 2012), increasing movement speeds (Douglas-Hamilton *et al.* 2005) or showing temporal avoidance (Gaynor *et al.* 2018). Similar studies have also found no response to village proximitiy by a number of species (e.g. Ogilby's duiker, Vanthomme *et al.* 2013; carnivore species, Van der Weyde *et al.* 2018).

Our findings may also suggest that hunting pressure did not vary greatly between the camera sites despite varying proximity to villages. Jones et al. (2017) determined that GRNP hunters can travel up to 8 km during hunting trips and in the current study the furthest distance from any camera trap site to the closest village was 6.2 km. Where available, other metrics of hunting pressure may be used to explore impacts on species richness and occurrence such as ranger patrol observations and reports (e.g. Moore et al. 2019), land ownership (e.g. Erb et al. 2012) and population density, where population sizes correlate with bushmeat demand and the number of hunters (Nielsen 2006). At GRNP in particular, distance to Liberian bushmeat markets may show declines in species richness and species occurrence as much of demand is thought to be driven from Liberia (Jones et al. 2019).

Proximity to roads

Whilst the larger roads of GRNP were developed for logging purposes, smaller roads are also used, predominantly by motorcycles and pedestrians to reach forest edge villages from larger towns, move between villages and access forest interiors. Hence, proximity to roads is associated with increased hunting intensity, aiding hunters access to otherwise remote areas (Robinson *et al.* 1999; Wilkie *et al.* 2000; Vanthomme *et al.* 2013). Roads also cause fragmentation of continuous forests, increasing edge habitats and disrupting connectivity (Wilkie *et al.* 2000; Arima *et al.* 2008; Fahrig & Rytwinski 2009; Vanthomme *et al.* 2013) and have potential direct mortality risks due to vehicle collisions (Forman & Alexander 1998; Fahrig & Rytwinski 2009; Erb *et al.* 2012). It was therefore predicted that species richness and species-specific occupancy probabilities would be lower at sites closer to the roads located in and surrounding GNRP. Higher abundances and occupancy are commonly associated with increased distance from roads (Laurance *et al.* 2006, 2008; Vanthomme *et al.* 2013) and in our study, the occupancy patterns of two species met our predictions. Bay duikers, a common bushmeat target in GRNP (Davies *et al.* 2008), and fire-footed rope squirrels showed lower occupancy close to roads.

Our findings also showed variation in species-specific responses. For example, Maxwell's duiker and sooty mangabeys showed higher occupancy at sites closer to roads, perhaps to exploit roadside foraging resources (Forman & Alexander 1998; Erb *et al.* 2012) or use the roads as corridors to facilitate movement across fragmented landscapes (Forman & Alexander 1998; Baker & Leberg 2018). Furthermore, species richness and the occupancy probability of the majority of modelled species showed no response to distance from the closest road, perhaps due to lowered perception of risk for some species due to small home ranges and therefore reduced likelihood of crossing roads (Fahrig & Rytwinski 2009), or the use of risk mitigation behaviours (e.g. increased waiting times shown at larger roads by chimpanzees at Bossou, Guinea, Hockings *et al.* 2006). The lack of negative responses to close road proximity may also result from the relatively large distances between camera sites and the roads at GRNP (shortest distance was 5.1 km), suggesting the need for future cameras placed closer to access routes in order to assess the impacts of immediate vicinity to roads. The intensity of pedestrian and vehicle traffic (e.g. Vanthomme *et al.* 2013) should potentially be incorporated in future analyses, as well as the impact of other road types. Tourism is a relatively recent development at GRNP but where roads and tracks are commonly used by tourists, potential patterns of disturbance to wildlife and deterrence of hunters should also be explored (e.g. Moore *et al.* 2019).

3.4.4 Ecological impacts on species richness and species occupancy

In addition to the impacts of human infrastructure and activities, species richness and species occupancy patterns were predicted to vary as a result of habitat differences between camera sites. Contrary to our predictions and the findings of similar studies (e.g. Gerber *et al.* 2012; Rich *et al.* 2016), there was found to be no effect of tree density or distance to the closest permanent water source on species richness. However, as predicted due to the different ecological requirements and habitat preferences of species, species-specific occupancy patterns showed varied responses to ecological covariates. Tree density and distance to the closest permanent water source predicted the occupancy of four and six species respectively, but for others there was no impact of one or either of these ecological factors. These findings reflect the discrepancies often found between species in the impacts of habitat features (Schuette *et al.* 2013; Paredes *et al.* 2017; Rich *et al.* 2017). However, our species-specific findings did not always reflect those of other study areas, for example Burton *et al.* (2012) and Rovero *et al.* (2017) found no impact of habitat features on the occupancy of marsh mongoose and in Tiwai island that neighbours GRNP, McCollum *et al.* (2018) found the occupancy of Maxwell’s duiker to be impacted by distance to water sources.

The occupancy probabilities of African brush-tailed porcupine, fire-footed rope squirrel and marsh mongoose were predicted by both tree density and distance to the closest stream, suggesting stronger preferences for particular habitat structures than other species and potentially lower ability to survive in human-altered environments. Both tree density and distance to water source can potentially be considered anthropogenic covariates and should be explored in future, for example the potential use of streams and rivers as routes to access forest interiors used by hunters (Bowler *et al.* 2017). In addition, whether species responses to habitat features vary seasonally, for example due to lower water availability in the dry season, could be explored by increasing sampling effort to allow for sufficient sample sizes for comparisons of the dry and rainy seasons (e.g. Bersacola, in prep).

It is likely that the spatial dynamics of species richness and species occupancy are impacted by unmodeled variation between camera sites such as the presence or density of preferred nesting or food plant species, for example western chimpanzees show preferences for particular tree species for nesting (in GRNP, Barca *et al.* 2018; in Guinea-Bissau, Carvalho *et al.* 2015). Our analyses would therefore benefit from more comprehensive habitat assessment during camera deployment and the inclusion of an increased number of ecological variables, particularly when exploring community-level responses. Additionally, analysis of co-occurrence patterns may reveal impacts on spatial dynamics due to interactions between species, such as predator-prey dynamics (e.g. Burton et al. 2012)*,* competition for resources (e.g. Ramesh et al. 2017), and the benefits of inter-species groups (e.g. terrestrial species as sentinels, McGraw & Bshary 2002*)*.

3.4.5 Western chimpanzees at GRNP

Western chimpanzee was the only critically endangered species detected during this survey and are a flagship species for GRNP. Our findings provide important data about this understudied population which may be of benefit to the updated conservation action plan for western chimpanzees currently under development. Previous nest count surveys estimated a population density of 0.27 individuals per km^2 (Ganas 2009), suggesting GRNP contains approximately 270 chimpanzees (Brncic *et al.* 2010), around 5% of the estimated 5925 individuals found across Sierra Leone (Heinicke *et al.* 2019). The relative detection rate and naïve and modelled occurrence of chimpanzees found in this study are similar to that of other parts of Sierra Leone (Garriga 2013; Garriga *et al.* 2019), reaffirming the conservation importance of GRNP for this species. Chimpanzees were detected in Gola Central and Gola South, confirming their presence in both blocks, as found during previous nest count surveys (Ganas 2009; Barca *et al.* 2018). However, Ganas (2009) and Barca *et al.* (2018) also found evidence of chimpanzees in the leakage belt but there were no detections beyond the national park boundaries in the current study. Due to widely reported chimpanzee crop foraging at cocoa farms in the leakage belt (Chapter 2), this finding likely results from limited sampling effort and demonstrates the need for more extensive study of how chimpanzees are using the unprotected community forests. Future work should also look to establish chimpanzee-specific camera trap sampling covering the whole national park, with cameras placed along known chimpanzee paths to maximise detections for spatial analyses. Such sampling will also be invaluable to gather information on chimpanzee feeding behaviours and patterns in this area and would be complemented by determining patterns of wild food availability.

Occupancy dynamics

The only covariate to appear in the top model set for predicting chimpanzee occupancy probability was distance to the closest permanent water source. This suggests chimpanzees across GRNP are tolerant to variation in tree density and the impacts of proximity to human infrastructure and activities, likely due to high behavioural flexibility (Hockings *et al.* 2015). For example, the risks of proximity to human infrastructure may be mitigated by behaviours such as increased group cohesiveness (Hockings *et al.* 2012) and reduced calling rates (Wilson *et al.* 2007) when foraging on crops, and adaptive behaviours during road crossings (Hockings *et al.* 2006; Cibot *et al.* 2015). Similarly, the crop defence strategies reported by cocoa farmers, such as noise production and guarding, showed little effect in deterring chimpanzees from foraging on crops (Chapter 2). Despite high frugivory and potential tool use to access drinking water from tree hollows (Tonooka 2001), there was a some evidence of higher occupancy closer to permanent water sources, perhaps due to the availability of plant food resources closer to water sources and potentially aquatic foraging opportunities (Boesch *et al.* 2016). Where chimpanzees are found outside of protected areas in Sierra Leone, chimpanzee occupancy has been found to reduce with increased proximity to roads (Garriga *et al.* 2019), highlighting the importance of further fine-scale monitoring of the impact of human infrastructure on GRNP chimpanzee distribution.

3.4.6 Conservation and management implications

This study comprises one of small number that have applied camera trap data collection within the UGF (e.g. Abdulai Barrie et al. 2007; Collen et al. 2011; Hillers et al. 2017; Cappelle et al. 2019) and within Sierra Leone (e.g. Ganas & Lindsell 2010; McCollum et al. 2017, 2018). The value of using camera traps to monitor mammal communities in tropical forests is reaffirmed by detections of 26 mammal species, including one previously unconfirmed species, with relatively low labour intensity. However, continued systematic monitoring is vital to confirm that populations of undetected species or those that were detected infrequently have not decreased. Future camera trap surveys should look to cover the whole areal extent of all GRNP blocks (including Gola North) and be arranged at equal distances from one another to allow for more detailed analysis of community and species-level responses to both ecological and anthropogenic heterogeneity (and account for auto-correlation). Our findings also support the need for monitoring focused in the leakage belt to determine how different land uses impact species distribution patterns. Future land use planning should look to maintain the connectivity and important habitat this area provides.

We found no impact of anthropogenic activities on species richness, but the occupancy modelling framework suggests the distribution patterns of many species at GRNP are impacted by the presence of anthropogenic infrastructure and activities. In Brncic *et al.* 2015, a country-wide assessment of large mammal distribution across Sierra Leone, similar patterns of varied responses among different species to human proximity and infrastructure were also found. Together these findings confirm the importance of studying species-specific responses when determining the impacts and patterns of human-wildlife coexistence across an environment, as considering community-level responses alone may mask the declines or shifts in distribution of individual species (Brncic *et al.* 2015). Our findings can therefore be used to monitor and predict the impacts on species and the abilities of different species to survive future human-induced changes. For example, hunting of the species found to have lower occurrence at sites close to villages and roads should be monitored closely, and ranger patrol routes and increased protection of specific species can also be informed from our findings. Increased numbers of detections are needed to enhance the accuracy of occupancy estimates and the significance of associations with covariates, but our methodology accounts for imperfect detection (MacKenzie *et al*. 2002) and provides a replicable framework on which to base further ecological surveys and monitoring at GRNP.

As the populations of the forest edge human communities of GRNP grow, and agricultural demand and other anthropogenic activities increase, studies such as this are of vital importance. The impact of humans on wildlife must be monitored and managed, and conservation must incorporate close collaboration with human communities to mitigate such impacts and promote positive human-wildlife coexistence. Providing alternative livelihoods through integrated conservation and development projects, for example the agroforestry cocoa project (GCPO) in this area (Chapter 2), will likely be vital to increase local tolerance towards wildlife conservation and decrease dependency on damaging practices such as slash-and-burn agriculture and bushmeat hunting.

Chapter 4:

General conclusions and recommendations for future research

As the global human population grows and humans and wildlife are increasingly found in close proximity to one another and competing for space and resources (Hockings *et al.* 2015), understanding the patterns and threats associated with human-wildlife coexistence is fundamental to effective and long-term conservation. Across tropical Africa, both socio-economic development and the conservation of biodiversity are key priorities, often reliant on the engagement of groups with competing interests. There is hence a pressing need to deliver cross-disciplinary research that provides site-specific management initiatives that incorporate both human and ecological perspectives. At Gola Rainforest National Park (GRNP), the livelihoods of 25,000 people are dependent on forest resources (Bulte et al. 2013) and a large number of globally threatened and endemic species inhabit this area (Klop *et al.* 2008; Lindsell *et al.* 2011; Hillers 2013; Table S1), yet little research focus has yet been placed on understanding the interactions between humans and wildlife and how best to manage such interactions. This thesis aimed to address this research gap by exploring multiple aspects and impacts of human-wildlife coexistence at GRNP. In this chapter, I highlight the key findings of this body of work and detail how they can contribute to informing appropriate and evidence-based conservation and management strategies.

4.1 Wildlife cocoa foraging at GRNP

For humans, living alongside protected areas comes with socio-economic challenges due to problematic wildlife behaviour (Dickman 2010; Peterson *et al.* 2010). If not addressed the impacts can be detrimental to wildlife where retaliatory actions limit the effectiveness of conservation initiatives (Madden 2004; Redpath *et al.* 2013). One focus of this study was therefore to explore wildlife cocoa foraging at GRNP, an increasing source of negative human-wildlife interactions both at GRNP and across agricultural landscapes (Peterson *et al.* 2010).

4.1.1 Interdisciplinarity

This study incorporated semi-structured interviews within the methodology enabling direct exploration of the perceptions and opinions of cocoa farmers in this area. Drawing on tools and solutions from multiple disciplines, including social science approaches, is increasingly acknowledged as integral to solving conflicts and mitigating negative impacts that arise where humans and wildlife communities coexist (Dickman 2010; Nyhus 2016). Semi-structured interviews (SSIs) proved an efficient and useful tool to draw on local knowledge, providing both quantitative and qualitative information on which to assess the scope of local concerns and address them appropriately (Rust *et al.* 2017).

4.1.2 Key findings and management suggestions

The key findings and potential management implications of Chapter 2 are summarised in Table S3. SSIs and participatory risk mapping (PRM) revealed that farmers associate a number of threats with cocoa farming, including climate and safety concerns, but across all chiefdoms wildlife crop foraging was found to be the most significant issue. This finding highlights the need to increase tolerance towards wildlife, and is representative of farmers perceptions across many similar landscapes (Linkie *et al.* 2007; Webber & Hill 2014; Garriga *et al.* 2018). Novel solutions such as mitigating other risks identified, for example exploring the use of insecticides, may be vital for achieving sustainable human-wildlife coexistence. SSIs also revealed the particularly negative perceptions towards the study focal groups, western chimpanzees, monkeys and squirrels, relative to other species groups and therefore the importance of conflict mitigation targeted at these groups, particularly for critically endangered western chimpanzees.

This study goes beyond the methodology of most research addressing wildlife crop foraging, combining the perceptions of farmers with metrics of measured crop damage (Spagnoletti *et al.* 2017). By doing so, we were able to compare perceived damage caused by each species group relative to measured damage. A key finding here was the disproportionate amount of blame placed on chimpanzees, likely due to body size, anthropomorphism and distinctive foraging behaviours that drive negative perceptions in other areas (Richards 2000; Root-Bernstein *et al*. 2013), despite monkeys having caused more damage across all plantations. It is therefore suggested that a focus of community engagement and awareness schemes should be on adjusting negative tolerances of chimpanzees, requiring research in to the specific chimpanzee foraging behaviours and a greater understanding of the factors shaping farmers perceptions of this species.

In addition to addressing the perceptions of farmers, directly reducing crop losses to wildlife remains a central component of conflict mitigation (Hill 2015). Although our analysis would benefit from greater consideration of variation in plantation productivity (e.g. measures of tree density and loss of pods to disease) (Hill 2018) and increased sample sizes, both land use planning and crop defence use can be informed from our findings. For example, distance to the closest village did not predict crop foraging for any species group and so is less important than other factors when establishing new areas for plantations. As is common among similar studies (Junker *et al.* 2018), there was little evidence of farmers' crop defence methods proving a successful deterrent to wildlife. This finding may bring broader socio-economic impacts (i.e. reduced time and resources spent guarding crops) and presents the case for the development and trial of novel approaches. We also found evidence of the value of integrated conservation and development projects, in this case the GCPO, in reducing crop foraging and shaping positive associations towards local conservation efforts. We therefore suggest the introduction of this project across the whole GRNP region and the use of similar projects elsewhere.

4.1.3 Future research

Whilst this study presents a broad overview of perceptions and factors that influence cocoa foraging at GRNP, there remains key areas of research yet to be addressed. Cocoa is increasingly a dominant cash crop at GRNP, but detailed study of the species and patterns associated with crop foraging of other crop types will help inform further agricultural practices and land use management (i.e. investigating reported elephant crop foraging at subsistence rice farms at GRNP). Such work should be complimented by studies of the dietary ecologies of species in this area, including the relative contribution of crops, and associations with patterns of wild food availability (e.g. chimpanzees at Cantanhez National Park, Bessa *et al.* 2015).

Our findings relating to the impact and importance of the GCPO is a particularly interesting area in which more in-depth study is critical. It is increasingly acknowledged that protected area conservation must incorporate a better understanding of conflicts between stakeholders and the role that projects such as this can play in bridging the gap between conservationists and those living alongside wildlife populations (Redpath 2013; Hill 2015).

4.2 Spatial distribution at GRNP

In order to further explore human-wildlife coexistence at GRNP, this thesis also addressed the presence and distribution of wildlife at GRNP and the impacts that close proximity to human infrastructure and activities have on community-level and species-level responses. This study contributes to a growing body of research that acknowledges that even protected areas are not unaffected by human influence (Hockings *et al.* 2015) and provides vital information regarding the ability of species to persist in areas impacted to different extents by human populations.

4.2.1 Key findings and management suggestions

The key findings and management implications of chapter 3 are summarised in Table S4. A primary aim of this study was to establish an updated baseline assessment of medium-to-large bodied size mammals at GRNP. Our camera trap survey provided the first reported evidence of the giant forest hog and confirmed the presence of several globally threatened species. These findings are invaluable contributions to quantifying and justifying the conservation value of the national park, as well as informing future species-specific surveying. High biodiversity found at GRNP is particularly important in light of reduced species richness found across the majority of the south of Sierra Leone relative to northern areas (Brncic *et al*. 2015). In comparing species richness in the leakage belt to within the protected forest blocks, we found that the leakage belt is likely of high conservation value. Future land use planning should therefore look to maintain connectivity, despite the expansion of agricultural industries. Increased research effort is particularly suggested for this area, as well as collaboration with those that live in communities in the leakage belt.

In determining community-level responses to ecological and anthropogenic variation at camera sites, there was found to be no impact on species richness but occupancy modelling revealed that species-specific responses were varied. This promotes the use of additional community-level metrics in future research, such as community composition (e.g. Effiom *et al.* 2013) to ensure responses are not masked. Our species-specific findings highlight the importance of studying the differences between species in their ability to adapt to human-impacted environments. We suggest that those species that were impacted by proximity to roads and villages (Maxwell's duiker, sooty mangabey, African brush-tailed porcupine, fire-footed rope squirrel, bay duiker and Jentink's duiker) should be prioritised for further monitoring. In addition, whilst not all species showed reduced occurrence rates at sites associated with greater intensity of hunting, agriculture and deforestation, we emphasise the need to gain community support and involvement in conservation in this area and monitor changing threats.

4.2.3 Future research

In this study we integrated both metrics of ecological and anthropogenic variation between different camera sites into analysis of species presence and distribution, but there is further scope to include a larger number of covariates. Through the application of similar social science methods to those used in Chapter 2, a more detailed assessment of variation in the intensity of human activities across GRNP may be possible, for example gaining an understanding of the NTFPs extracted from the forest by villagers (e.g. Dash *et al.* 2016). Assessments of species abundance, as well as behavioural research such as specific risk mitigation behaviours of GRNP populations (e.g. road crossing, Hockings *et al*. 2006) will also be beneficial to fully explore the impact of anthropogenic proximity on wildlife.

Effective conservation of GRNP that adapts to changing threats relating to human-wildlife coexistence is particularly important in a country that is experiencing rapid population growth and economic development. This thesis provides a multi-faceted investigation of the interactions between humans and wildlife at this previously understudied site. It contributes to informing management that recognises the needs of human communities and helps achieve long-term conservation of threatened species. Work such as this is essential to recognise the intricacies of human and wildlife persistence and is vital to our efforts to preserve global biodiversity.

Supplementary Materials

Chapter 1 Supplementary materials

(Overleaf)

Table S1: Larger mammal species recorded in GRNP, IUCN red list status (2019) and location of records. Adapted from Lindsell et al. 2011 and Hillers 2013

Family	Species	IUCN status	Chapter 3 Survey	Published records	Unpublished records*
Order: Rodentia					
Sciuridae	Fire-footed rope squirrel *Funisciurus pyrropus*	LC	**Yes**	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Sciuridae	Slender-tailed squirrel *Protoxerus aubinnii*	NT	**Yes**	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Sciuridae	Red-legged sun squirrel *Heliosciurus rufobrachium*	LC	**Yes**		Dowsett-Lemaire & Dowsett 2007^{4}, Hillers 2013^{3}
Sciuridae	Striped ground squirrel *Xerus erythropus*	LC	**Yes**	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Sciuridae	Forest giant squirrel *Protoxerus stangeri*	LC	No	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Sciuridae	Western palm squirrel *Epixerus ebii*	LC	No	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Sciuridae	Green bush squirrel *Paraxerus poensis*	LC	No	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Nesomyidae	Forest giant pouched rat *Cricetomys emini*	LC	**Yes**		Hillers 2013^{3}
Hystricidae	African brush-tailed porcupine *Atherurus africanus*	LC	**Yes**	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Hystricidae	Crested porcupine *Hystrix cristata*	LC	No		Dowsett-Lemaire & Dowsett 2007^{4}
Anomaluridae	Beecroft's scaly-tailed squirrel *Anomalurus beecrofti*	LC	No	Lindsell et al. 2011^{1}	
Order: Pholidota					
Manidae	White-bellied pangolin *Phataginus tricuspis*	VU	No	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Manidae	Giant ground pangolin *Smutsia gigantea*	VU	No		Hillers 2013^{3}
Manidae	Long-tailed pangolin *Uromanis tetradactyla*	VU	No		Davies 1987^{5}, Hillers 2013^{3}
Order: Carnivora					
Nandiniidae	African palm civet *Nandinia binotata*	LC	**Yes**	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Viverridae	African civet *Civettictis civetta*	LC	**Yes**	Lindsell et al. 2011^{1}	Hillers 2013^{3}
Viverridae	Pardine genet *Genetta pardina*	LC	**Yes**		Davies 1987^{5}
Felidae	African golden cat *Caracal aurata*	VU	No	Lindsell et al. 2011^{1}	Davies 1987^{5}

Family	Species	IUCN status	Chapter 3 Survey	Published records	Unpublished records*
Felidae	Leopard *Panthera pardus*	VU	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Herpestidae	Marsh mongoose *Atilax paludinosus*	LC	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Herpestidae	Common slender mongoose *Herpestes sanguineus*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Herpestidae	Egyptian mongoose *Herpestes ichneumon*	LC	No		Davies 1987[5]
Herpestidae	White-tailed mongoose *Ichneumia albicauda*	LC	No		Davies 1987[5]
Herpestidae	Long-nosed cusimanse *Crossarchus obscurus*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Mustelidae	Honey badger *Mellivora capensis*	LC	**Yes**		Hillers 2013[3]
Mustelidae	Spotted-necked otter *Hydrictis maculicollis*	NT	No	Lindsell et al. 2011[1]	
Mustelidae	African clawless otter *Aonyx capensis*	NT	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Order: Primates					
Galagidae	Demidoff's dwarf gelago *Galagoides demidovii*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Galagidae	Thomas's dwarf gelago *Galagoides thomasi*	LC	No		Hillers 2013[3]
Lorisidae	Potto *Perodicticus potto*	LC	No	Lindsell et al. 2011[1]	
Cercopithecidae	Sooty mangabey *Cercocebus atys*	NT	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Western red colobus *Piliocolobus badius*	EN	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Diana monkey *Cercopithecus diana*	VU	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Olive colobus *Procolobus verus*	NT	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Western pied colobus *Colobus polykomos*	VU	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Campbell's monkey *Cercopithecus campbelli*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Lesser spot-nosed monkey *Cercopithecus petaurista*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Cercopithecidae	Green monkey *Chlorocebus sabaeus*	LC	No		
Hominidae	Chimpanzee (western) *Pan troglodytes verus*	CR	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Order: Cetartiodactyla					
Suidae	Red river hog *Potamochoerus porcus*	LC	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Suidae	Giant forest hog *Hylochoerus meinertzhageni*	LC	**Yes**		

Family	Species	IUCN status	Chapter 3 Survey	Published records	Unpublished records*
Hippopotamidae	Pygmy hippopotamus *Hexaprotodon liberiensis*	EN	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Tragulidae	Water chevrotain *Hyemoschus aquaticus*	LC	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Royal antelope *Neotragus pygmaeus*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	African forest buffalo *Syncerus caffer nanus*	NT	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Bushbuck *Tragelaphus scriptus*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Bongo *Tragelaphus eurycerus*	NT	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Bay duiker *Cephalophus dorsalis*	NT	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Maxwell's duiker *Philantomba maxwelli*	LC	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Black duiker *Cephalophus niger*	LC	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Ogilby's duiker *Cephalophus ogilbyi*	LC	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Yellow-backed duiker *Cephalophus silvicultor*	NT	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Zebra duiker *Cephalophus zebra*	VU	**Yes**	Lindsell et al. 2011[1]	Hillers 2013[3]
Bovidae	Jentink's duiker *Cephalophus jentinki*	EN	**Yes**	Ganas and Lindsell 2009[2]	Hillers 2013[3]
Order: Hyracoidea					
Procaviidae	Western tree hyrax *Dendrohyrax dorsalis*	LC	No	Lindsell et al. 2011[1]	Hillers 2013[3]
Order: Proboscidae					
Elephantidae	African Forest Elephant *Loxodonta cyclotis*	VU	No	Lindsell et al. 2011[1]	Hillers 2013[3]

*= grey literature (internal reports)

[1] Lindsell, J.A., Klop, E. & Siaka, A.M. (2011). The impact of civil war on forest wildlife in West Africa: mammals in Gola Forest, Sierra Leone. *Oryx*, 45, 69–77.

[2] Ganas, J., Lindsell, J.A., (2010). Photographic evidence of Jentink's duiker in the Gola Forest Reserves, Sierra Leone. *African Journal of Ecology* 48, 566–568.

[3] Hillers, A. (2013). Final report on the activities of the Research Unit Across the River Transboundary Peace Park for Sierra Leone and Liberia, 1–145.

[4] Dowsett-Lemaire, F. & Dowsett, R.J.D. (2007). Faunistic Survey of Gola Forest (Sierra Leone) in January–February 2007, with an Emphasis on Birds. Royal Society for the Protection of Birds, Sandy, UK.

[5] Davies, A.G. (1987). The Gola Forest Reserves, Sierra Leone: Wildlife Conservation and Forest Management. IUCN, Gland, Switzerland.

Chapter 2 Supplementary materials

Tanya Payne, MSc by Research student, University of Exeter

Cocoa farming and wildlife perceptions surrounding the Gola Rainforest National Park

Interview number: Chiefdom:
Date: Plantation name (if surveyed):
Village:

Notes (e.g. location of interview, other people present):

<u>Part 1: Interviewee information</u>

Age (or year of birth):
Gender:
Other occupation:

<u>Part 2: Cocoa farming practices and GCPO</u>

1. Do you plan any expansion to your cocoa plantations?

   ☐ Yes ☐ No

   ☐ Expanding current plantations ☐ Establish new plantations

2. Are you a member of the Goleagorbu programme (GCPO)?

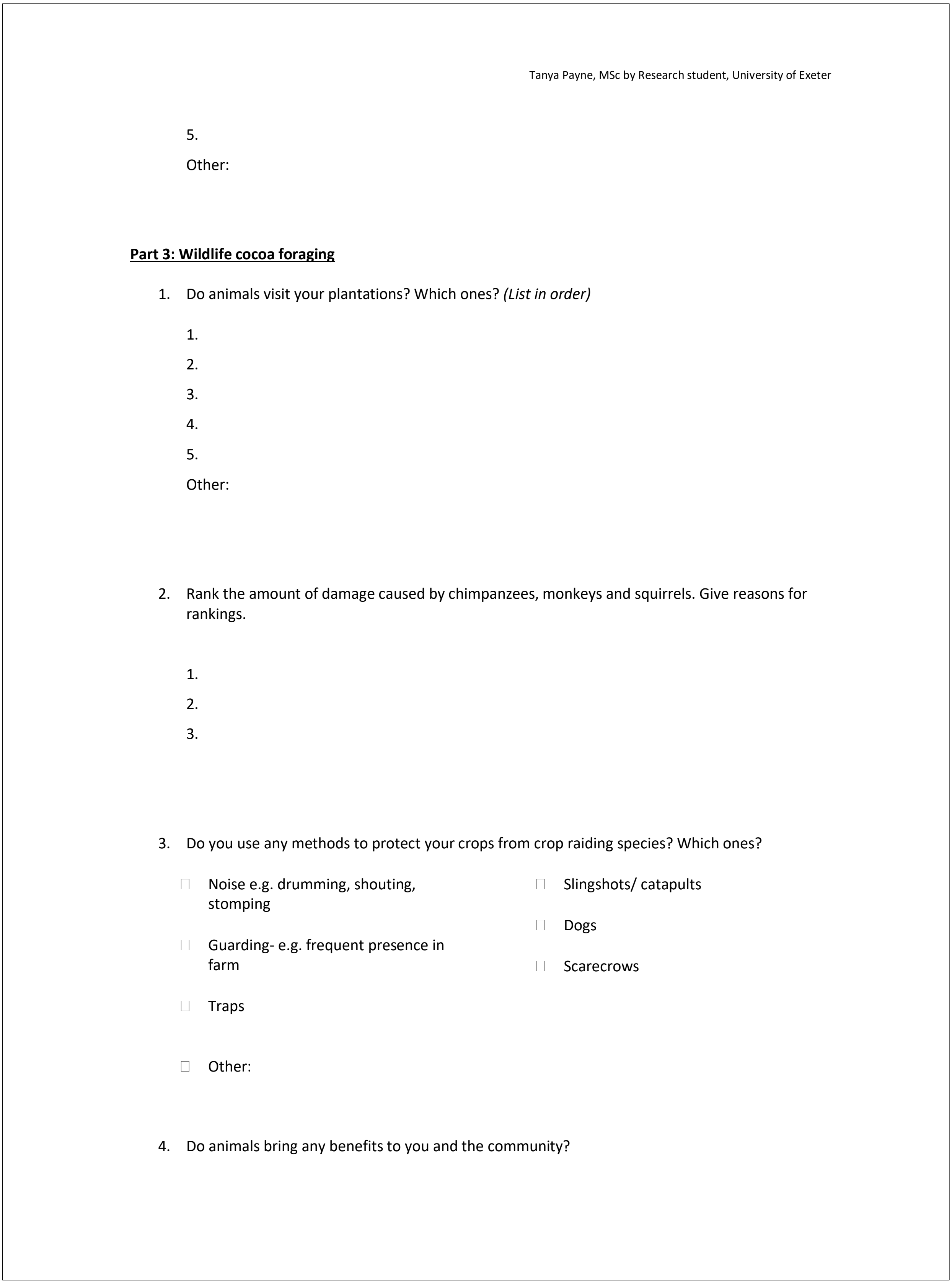
   ☐ Yes ☐ No

3. Has being a member of GCPO benefitted your cocoa harvest?

   ☐ Yes ☐ No

4. What problems do you experience with cocoa farming? *(List in order)*

   1.
   2.
   3.
   4.



5.

Other:

Part 3: Wildlife cocoa foraging

1. Do animals visit your plantations? Which ones? *(List in order)*

   1.
   2.
   3.
   4.
   5.

   Other:

2. Rank the amount of damage caused by chimpanzees, monkeys and squirrels. Give reasons for rankings.

   1.
   2.
   3.

3. Do you use any methods to protect your crops from crop raiding species? Which ones?

   ☐ Noise e.g. drumming, shouting, stomping

   ☐ Guarding- e.g. frequent presence in farm

   ☐ Traps

   ☐ Other:

   ☐ Slingshots/ catapults

   ☐ Dogs

   ☐ Scarecrows

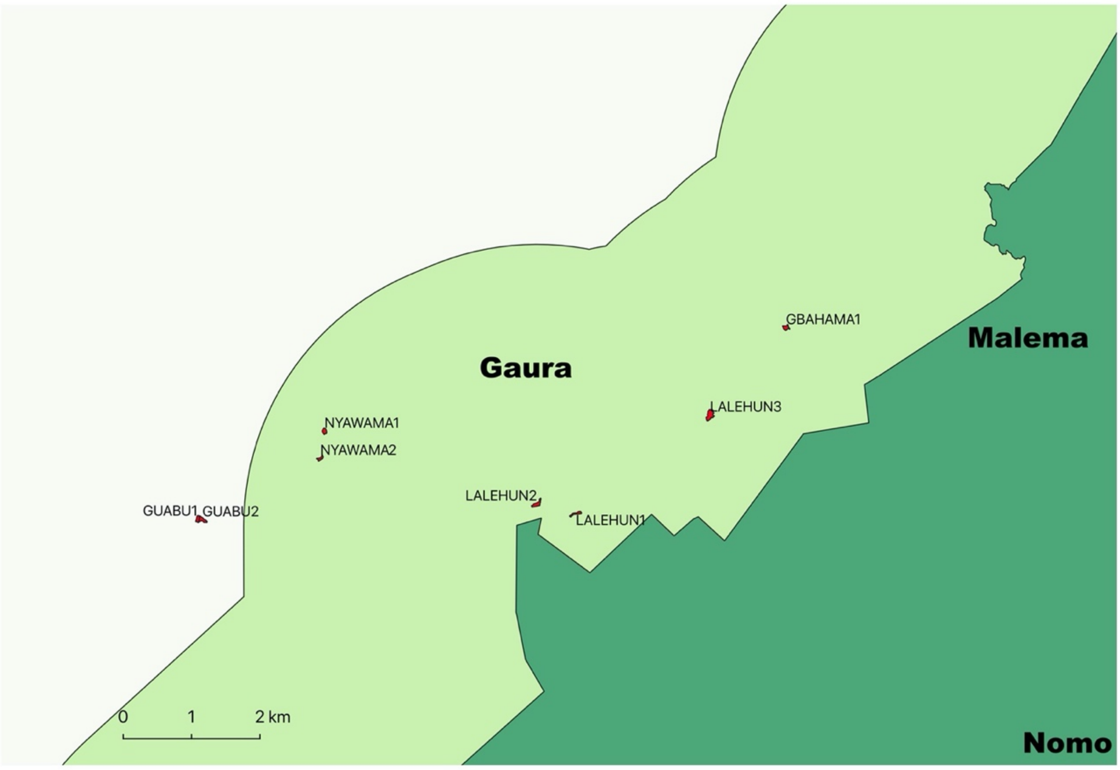
4. Do animals bring any benefits to you and the community?

Figure S1: Example semi-structured interview conducted with GRNP cocoa farmers (n=71)

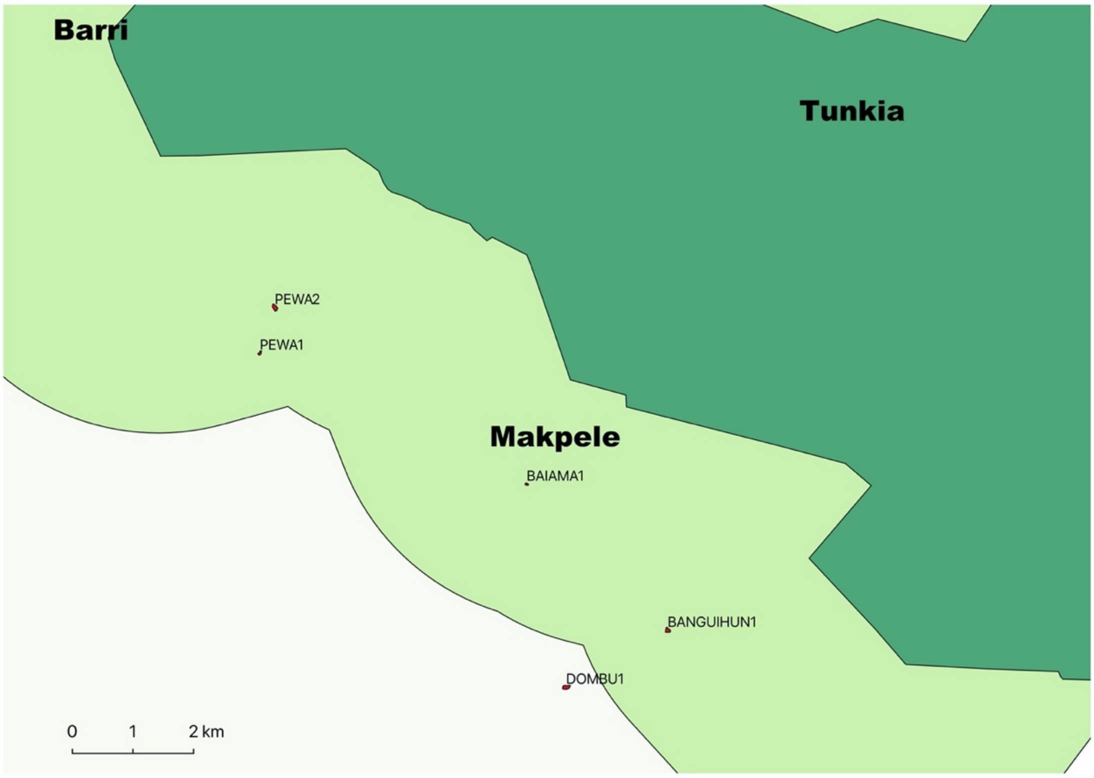
Table S2: Descriptive statistics of cocoa plantations surveyed for cocoa foraging traces by chimpanzees, monkeys and squirrels in the leakage belt of Gola Rainforest National Park.

Chiefdom	Plantation	Area (m^2)	Perimeter length (m)
Gaura	GBAHAMA1	4689.741	347.906
	GUABU1	3978.094	285.175
	GUABU2	5574.589	379.974
	LALEHUN1	4457.342	433.802
	LALEHUN2	6663.392	463.581
	LALEHUN3	11811.155	558.394
	NYAWAMA1	4363.167	253.555
	NYAWAMA2	3254.382	312.159
Makpele	BAIAMA1	1512.996	158.427
	BANGUIHUN1	4636.163	293.105
	DOMBU1	6338.365	350.587
	PEWA1	2228.72	208.501
	PEWA2	5916.836	354.064
Malema	BANDAJUMA1	2867.946	311.123
	BANDAJUMA2	2101.602	202.814
	MADINA1	4897.721	285.046
	MADINA2	8984.963	527.309
	MADINA3	2372.983	305.532
	NJABAMA1	5175.254	305.432
	NJABAMA2	4298.18	255.827
	TAKPOIMA1	3444.748	232.101
	TAKPOIMA2	2552.483	236.871
	TAYAMA1	11847.037	489.446
	TAYAMA2	14492.128	627.100

a)


Gaura
Malema
Nomo
GBAHAMA1
LALEHUN3
NYAWAMA1
NYAWAMA2
LALEHUN2
LALEHUN1
GUABU1
GUABU2
0
1
2 km

b)


Barri
Tunkia
PEWA2
PEWA1
Makpele
BAIAMA1
BANGUIHUN1
DOMBU1
0
1
2 km

c)

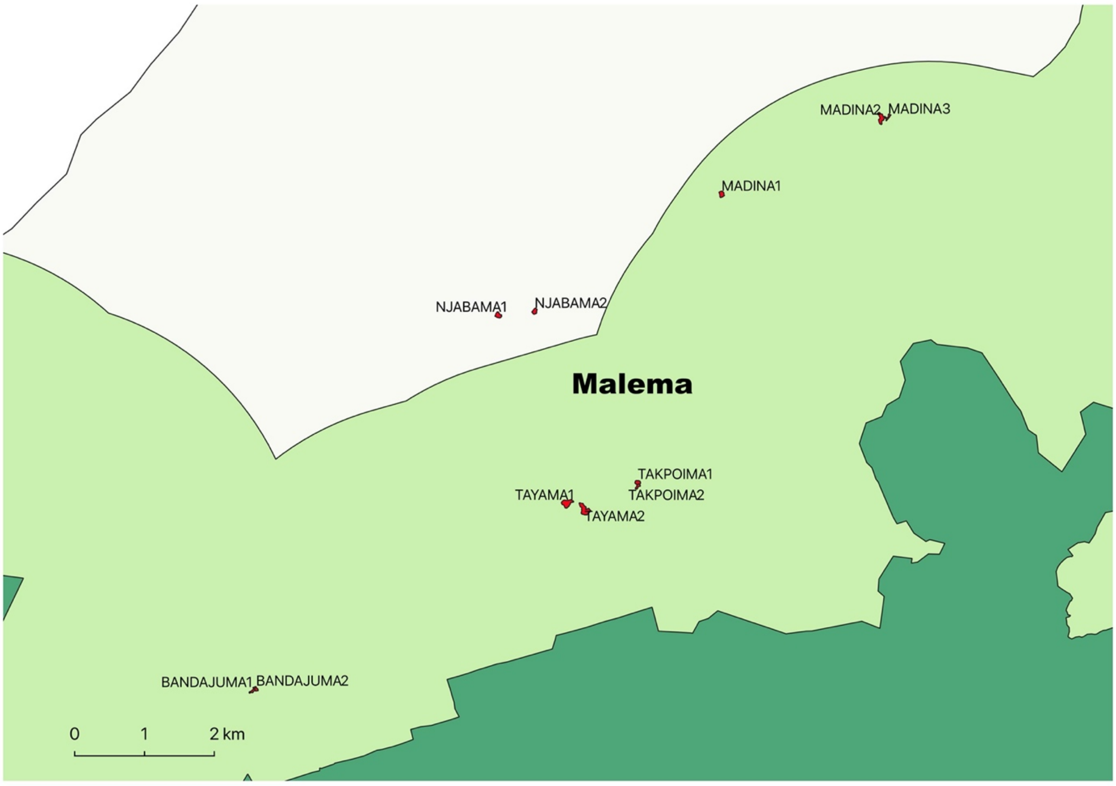
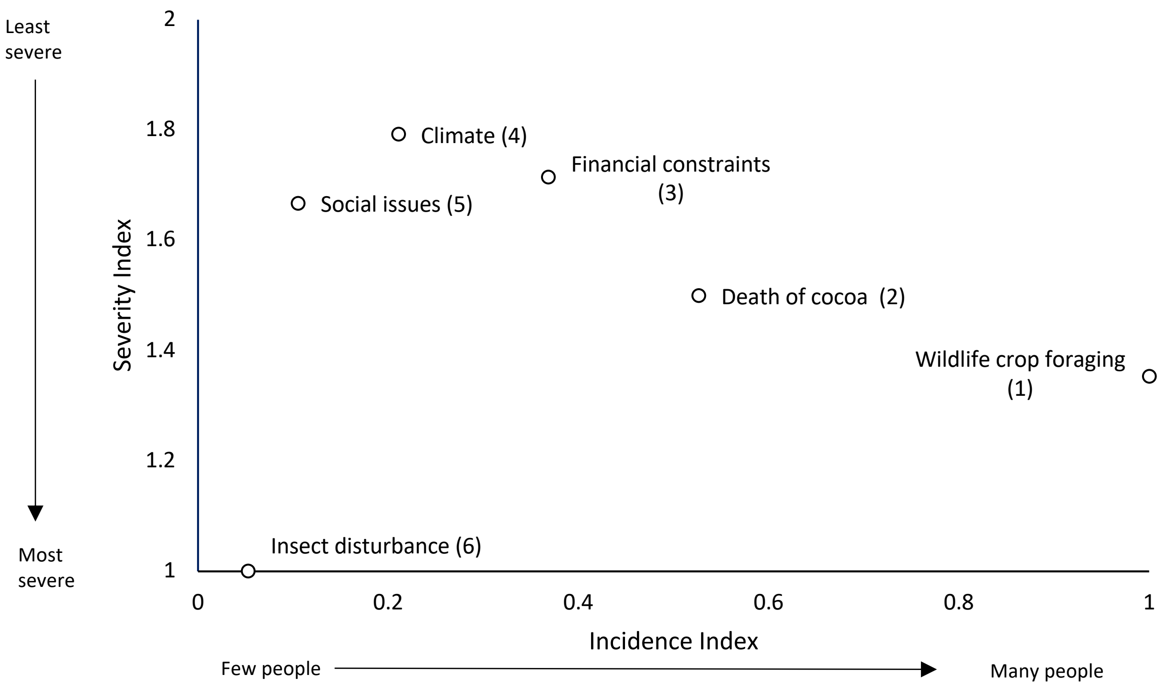
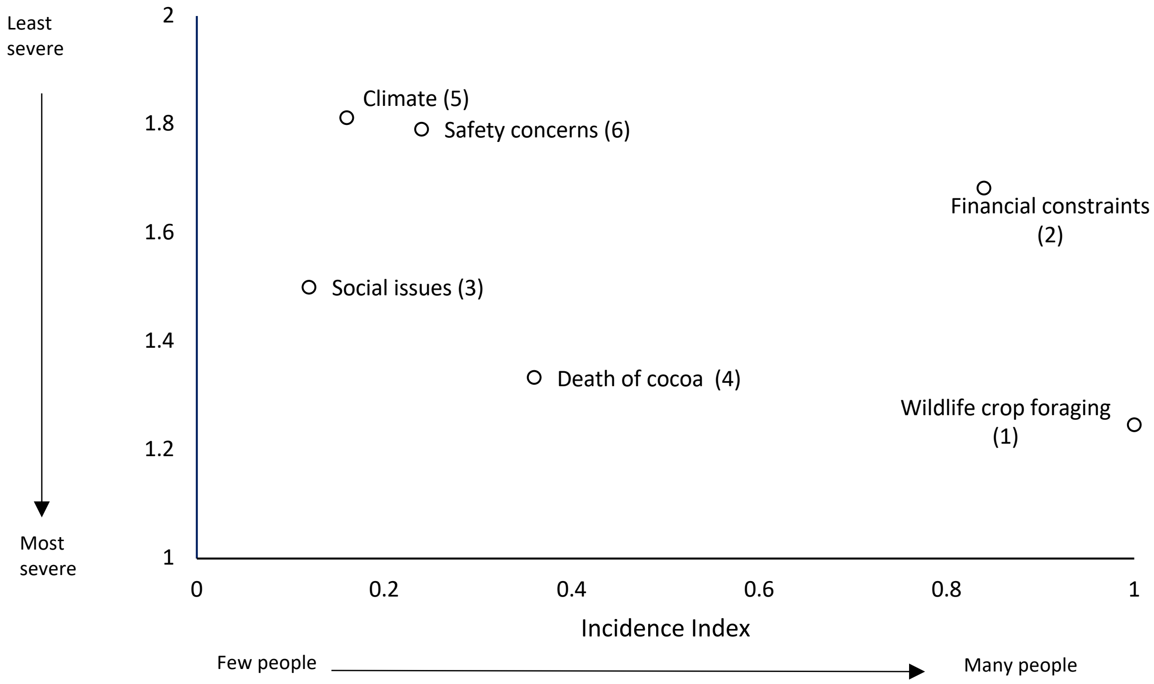


Figure S2: Location, size and shape of plantations (n= 24) where cocoa foraging surveys carried out within each study chiefdom at Gola Rainforest National Park, (a) Guara, (b) Makpele and (c) Malema.

a)


Least severe
Most severe
Severity Index
2
1.8
1.6
1.4
1.2
1
Climate (4)
Financial constraints (3)
Social issues (5)
Death of cocoa (2)
Wildlife crop foraging (1)
Insect disturbance (6)
0
0.2
0.4
0.6
0.8
1
Incidence Index
Few people
Many people

b)


Least severe
Most severe
2
1.8
1.6
1.4
1.2
1
Climate (5)
Safety concerns (6)
Financial constraints (2)
Social issues (3)
Death of cocoa (4)
Wildlife crop foraging (1)
0
0.2
0.4
0.6
0.8
1
Incidence Index
Few people
Many people

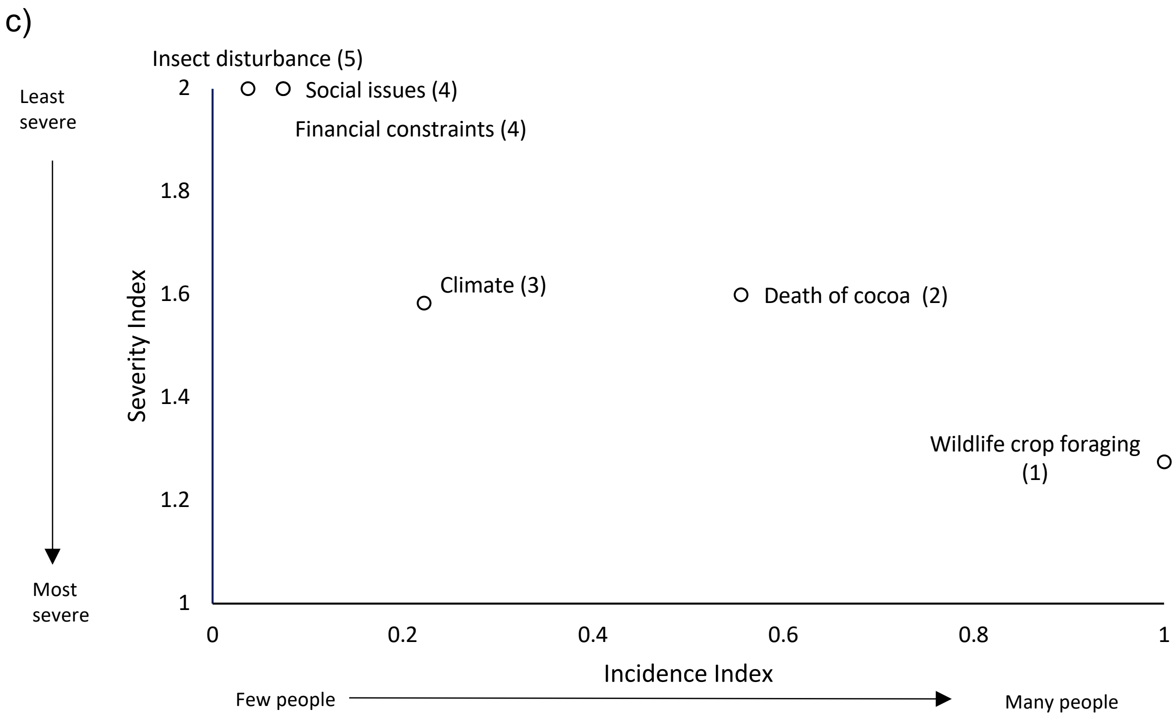
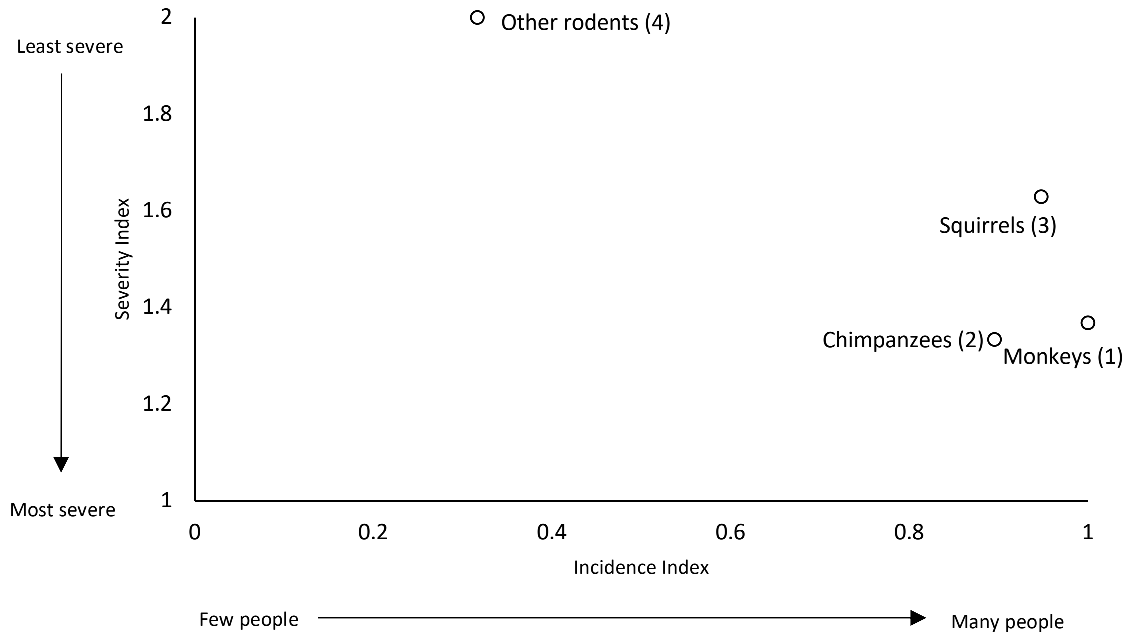
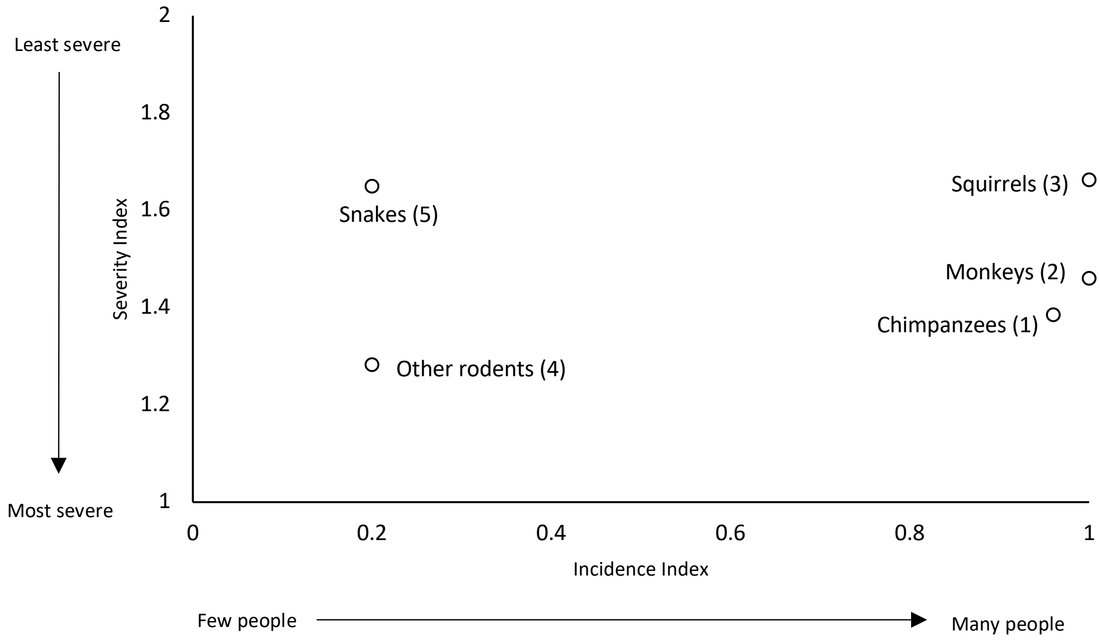


Figure S3: Risk map showing perceptions of GRNP farmers from (a) Gaura (n= 19 interviewees), (b) Makpele (n= 25) and (c) Malema (n= 27) chiefdoms towards wildlife cocoa foraging and other risks associated with cocoa farming identified during semi-structured interviews.

a)


Least severe
Most severe
Severity Index
2
1.8
1.6
1.4
1.2
1
0
0.2
0.4
0.6
0.8
1
Incidence Index
Few people
Many people
Other rodents (4)
Squirrels (3)
Chimpanzees (2)
Monkeys (1)

b)


Least severe
Most severe
Severity Index
2
1.8
1.6
1.4
1.2
1
0
0.2
0.4
0.6
0.8
1
Incidence Index
Few people
Many people
Snakes (5)
Other rodents (4)
Squirrels (3)
Monkeys (2)
Chimpanzees (1)

c)

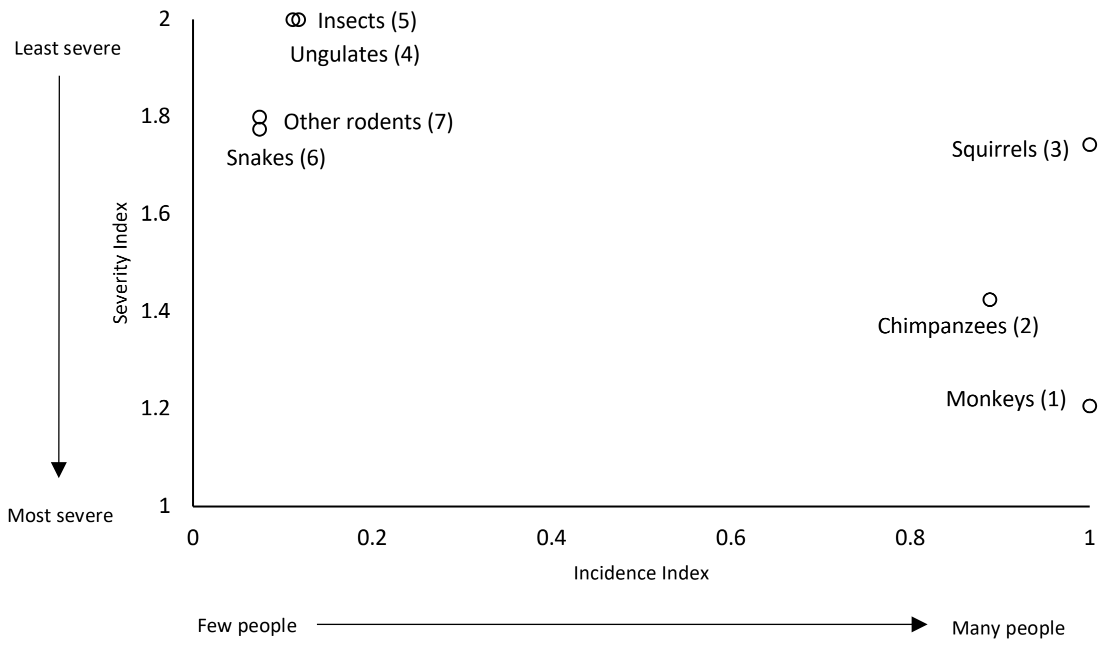


Figure S4: Risk map showing perceptions of GRNP farmers from (a) Gaura (n= 19 interviewees), (b) Makpele (n= 25) and (c) Malema (n= 27) chiefdoms towards cocoa foraging species groups identified during semi-structured interviews.

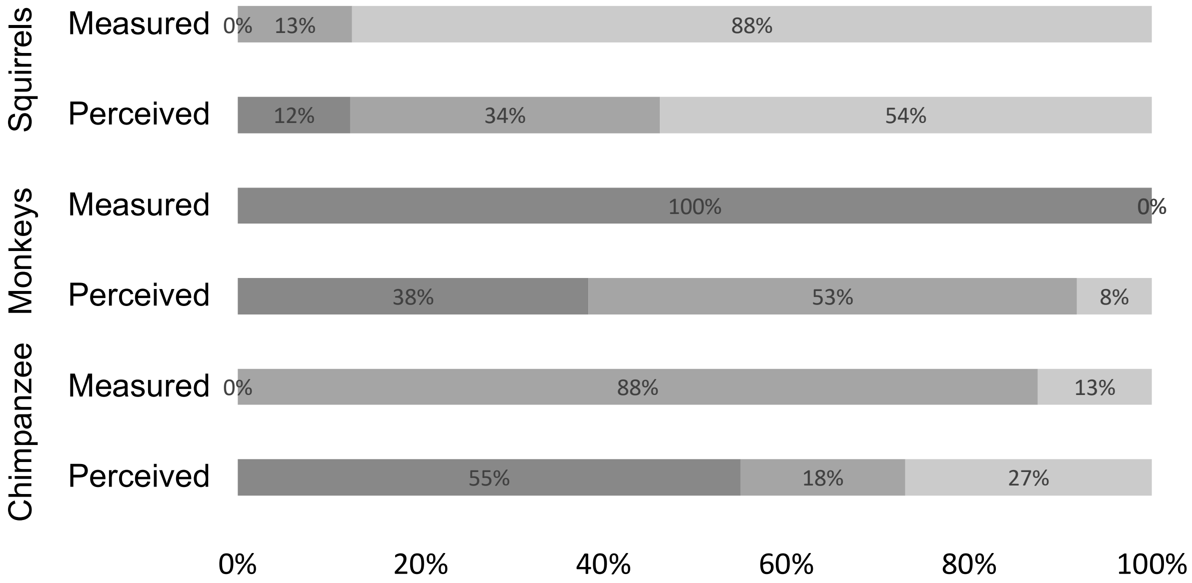
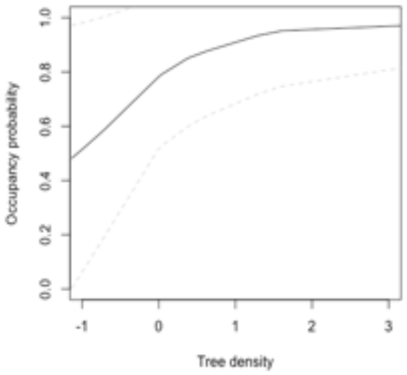


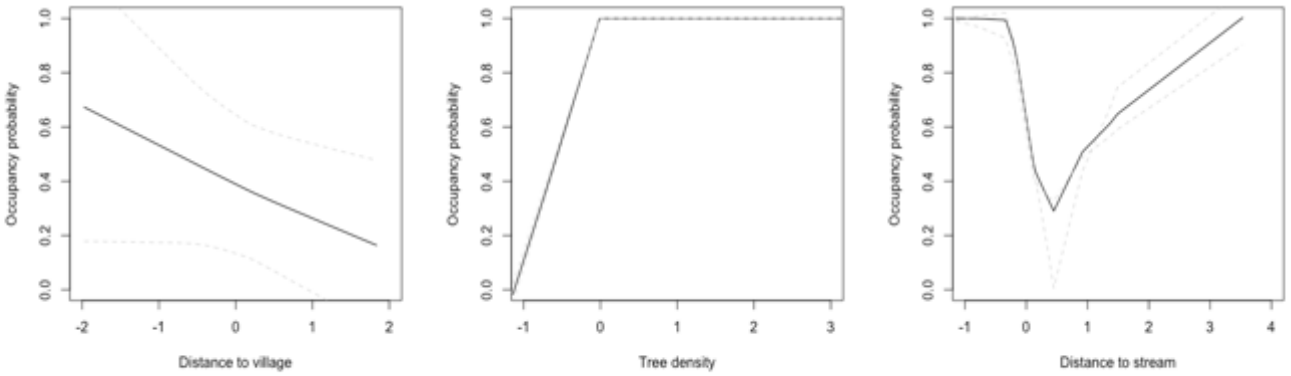
Figure S5: Percentages of perceived rankings from semi-structured interviews (n=71) and measured rankings from plantation surveys (n=24) of the amount of damage caused by chimpanzees, monkeys and squirrels. Rank 1= most damage (dark), Rank 3= least damage (light).

Figure S6: Relative detection frequency (trap rate per 100 days) of each mammal species detected during camera trap survey of Gola Rainforest National Park

a) Maxwell's duiker



b) African brush-tailed porcupine



c) Marsh mongoose

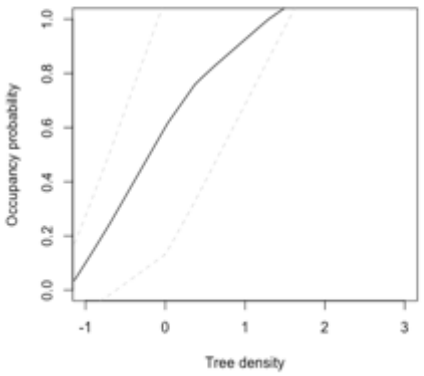


Figure S7: Effects of covariates with summed model weights >0.50 on model-predicted occupancy probability of (a) Maxwell's duiker, (b) African brush-tailed porcupine and (c) marsh mongoose, detected during camera trap survey in Gola Rainforest National Park. All covariates standardised to z scores.

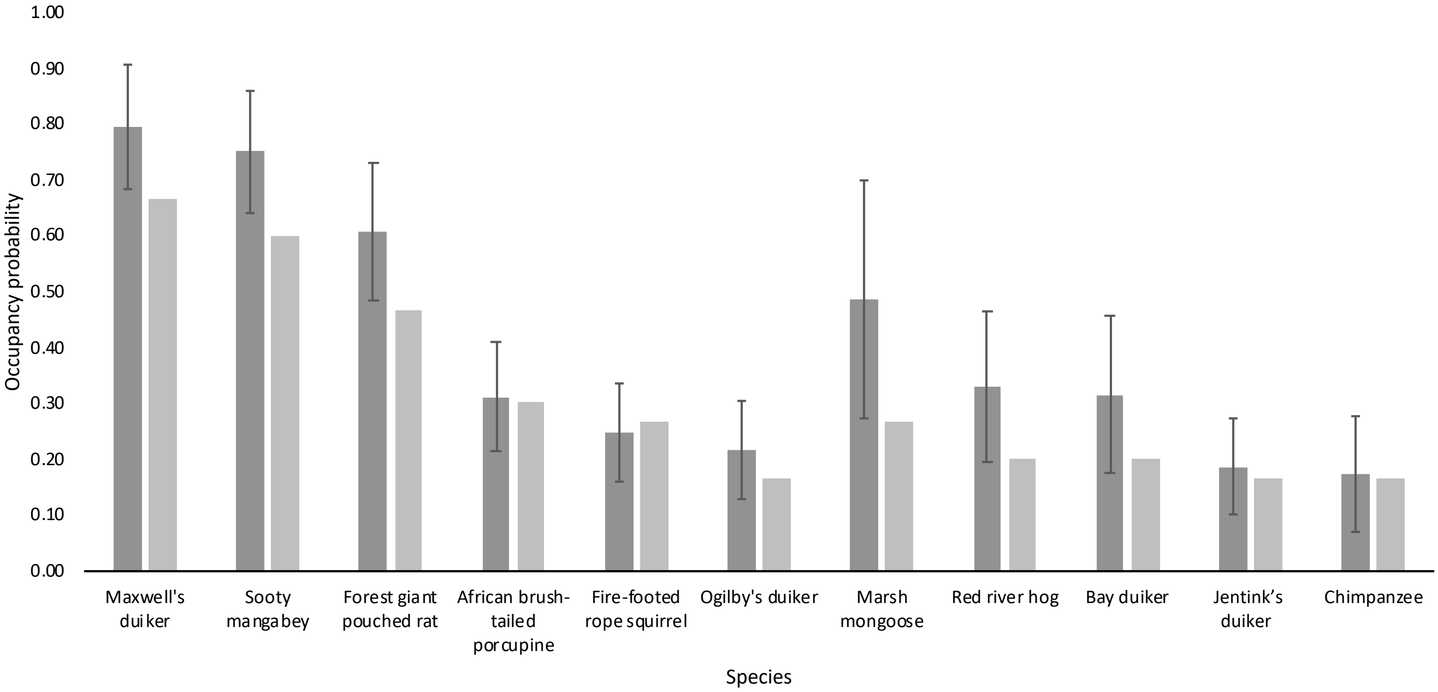


Figure S8: Mean model-derived (dark grey, with standard error bars) and naïve occupancy estimates for the 11 most frequently detected species (≥10 independent detections) during camera trap (n=30) survey of Gola Rainforest National Park.

Table S3: Summary of objectives, findings and management implications from study of wildlife cocoa foraging at Gola Rainforest National Park, Sierra Leone

Objectives	Key findings	Examples of management implications
Determine farmer's perceptions of wildlife cocoa foraging	- Wildlife cocoa foraging most significant issue associated with cocoa farming	- Mitigating negative impacts caused by wildlife cocoa foraging is a conservation and development priority - Improving tolerance by increasing yields through developing solutions to other risks identified
Compare perceived and measured damage attributed to different species groups	- Seven crop foraging species groups identified	- Further study of behaviours and contributions of other crop foraging groups
	- Particularly negative perceptions towards chimpanzees, monkeys and squirrels	- Chimpanzees, monkeys and squirrels are priorities for conflict mitigation
	- Chimpanzees disproportionally blamed for damage	- Chimpanzees are priority species for knowledge sharing activities to improve species-specific tolerance
	- Monkeys most damaging species	- Monkeys are priority species group for plantation management to reduce crop foraging damage
Determine the spatial characteristics and crop defence methods that predict susceptibility of plantations to cocoa foraging	- Variation between species groups in terms of both spatial characteristics and crop defence strategies that are effective	- Monkey cocoa foraging damage may be reduced by land use planning that incorporates plantations closer to roads and longer perimeter lengths - Use of dogs may be effective at reducing chimpanzee and total crop foraging damage
	- Limited effectiveness of all crop defence strategies	- Reduced use of ineffective methods - Further study to increase sample sizes and trial of novel techniques
Evaluate the effectiveness of the GCPO in terms of mitigating conflict over wildlife crop foraging	- Reduced total, chimpanzee and monkey crop foraging at member-owned farms - Membership of GCPO perceived as benefit of living alongside wildlife	- Membership should be extended to other villages, promotes reduced losses and increased tolerance as well as further ecological benefits

Table S4: Summary of objectives, findings and management implications from study of species presence and distribution at Gola Rainforest National Park, Sierra Leone

Objectives	Key findings	Examples of management implications
Provide an updated baseline assessment of the mammal community at GRNP	- 26 mammal species detected, one newly recorded species and five globally threatened	- Further study to ensure populations of undetected and rarely detected species have not declined, likely using alternative and species-targeted survey methods
Determine variation in species richness across and surrounding GRNP	- No difference in mean species richness between surveyed areas	- Monitoring land use changes to important habitat of the leakage belt, and close collaboration with human communities of leakage belt - Further study of species presence and distribution across leakage belt, e.g. increased number of cameras
Assess the fine-scale impacts of anthropogenic and ecological factors on species richness and species-specific occurrence at different sites	- No impact of covariates on species richness	- Importance of species-specific and occupancy approach in future monitoring
	- Varied responses amongst modelled species to proximity to anthropogenic activities	- Human activities should be further quantified and monitored, and negatively affected species should be prioritised for conservation efforts - Sites with closest proximity to human threats may be targets for law enforcement patrol efforts
	- Varied responses amongst modelled species to ecological features	- Species with particular habitat preferences should be prioritised for conservation efforts, and anthropogenic impacts on ecological features should be monitored

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