Spatial Pattern of Illegal Activities and the Impact on Wildlife Populations in Protected Areas in the Serengeti Ecosystem

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Abstract

Illegal activities in protected areas (PAs) are a major conservation problem linked to biodiversity loss. However, the scale of the problem at a global and local scale is unclear. There is a lack of understanding of the factors driving illegal activities and how law enforcement is targeted to reduce the impact of illegal activities. These information gaps limit the improvement of conservation, making tackling the problem difficult. I use an analytical approach, quantitative field surveys and field experiments in the Serengeti ecosystem to improve our understanding of this problem and how it could be reduced in protected areas.

At a global scale, I found that illegal activities are present in more PAs than previously thought. Population of large wild mammals are more likely to decline in less-strict PAs in countries with limited conservation resources and where illegal hunting is conducted for commercial benefits rather than for subsistence. The probability of the mammal decline increases in countries where land use change is driven by illegal plant exploitation.

At a local scale, in the Serengeti ecosystem, illegal activities are wide-spread, suggesting the problem is bigger than previously perceived. These are driven by poaching decisions made at various scales influenced by local habitats and environmental characteristics. I estimate there could be 137000 wire snares set at any one point across the Serengeti ecosystem, resulting in killing of approximately 14% of the animal population available each year. Despite this, I found current anti-poaching strategies ineffective at detecting and removing wire snares, increasing the risks of animal mortality and potential population declines, and fuelling the illegal wildlife trade.

Any comprehensive strategy towards curbing poaching and other illegal activities in PAs must improve the deterrent effects of law- enforcement patrols through increasing conservation resources and improving their ability to detect and remove existing threats.

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Uvumilivu wa familia yangu katika kipindi chote cha masomo ikiwa mbali nami katika mabara mawili tofauti ulinipa faraja na nguvu katika kipindi chote nikiwa nasoma hapa Uingereza. Namshukuru mke wangu Aishath Adinan kwa upendo, uvumilivu pamoja na kunitia moyo ili kuendelea na masomo mpaka wakati huu napoweka kalamu chini. Mpendwa mwanangu Allytza uvumilivu wake katika kipindi chote nikiwa siko naye umeleta ushindi huu ambao ni ushindi wake pia. Nawashukuru wazazi wangu wote, mama yangu Khadija Mgugwa na baba yangu Abeid Rija kwa upendo wao usiokoma kwangu. Swala zao kwa ajili yangu na ushauri kwangu kwa muda vyote yamekuwa ndio chachu ya mafanikio haya. Napenda pia kuwashukuru ndugu zangu wote katika familia yetu kwa kunipa moyo na ushauri katika kipindi chote ambacho nilihitaji msaada wao nikiwa naendelea na masomo haya].

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…………………. to all the wildlife species faced with pressures of illegal activities within protected areas globally……..and to the park rangers working hard to ensuring the wildlife survival amid a relentless wave of poachers!

Author’s Declaration

I declare that this thesis is a presentation of original work and I am the sole author.

This work has not previously been presented for an award at this, or any other University. All sources are acknowledged as references.

CHAPTER 1 – General Introduction

Sunset on the Serengeti plains with African buffalo and plain zebra

1.0. Introduction

Protected areas are a cornerstone of global conservation strategy and are generally increasing in number and spatial extent both on terrestrial land and water (Watson et al., 2014). Global terrestrial protected area coverage has increased to at least 14% within the past three decades (Zimmerer et al., 2004, Soutullo et al., 2008, Butchart et al., 2015) and is expected to cover 17% of the total global land area by 2020 (CBD, 2015). In addition, the coverage of marine protected areas expected to increase to 10% of the oceans over a similar time (O'Leary et al., 2016). These increases aim at halting the continuing loss of biodiversity and species extinctions facing the globe today (CBD, 2015). However, recent assessments of biodiversity targets indicate that, despite increasing conservation efforts, global biodiversity loss is increasing (Butchart et al., 2010, Tittensor et al., 2014). This loss is especially pronounced in some protected areas within a number of tropical countries (Craigie et al., 2010, Laurance et al., 2012), where biodiversity is the highest (Hillebrand, 2004, Adams & Hadly, 2013, Ripple et al., 2015), conservation resources are limiting (Albers 2010, Tranquilli et al., 2012), and pressures from the illegal exploitation of wildlife are rising (Brashares et al., 2011, Shova & Hubacek, 2011, Critchlow et al., 2015). Consequently, a call has been made to develop new strategies to improving species conservation in protected areas (Butchart et al., 2015, Watson et al., 2016). Understanding the spatial and temporal trends in illegal activities in protected areas and their impacts on long-term population persistence can be a useful strategy to reducing the impacts of illegal activities on the species, thereby safeguarding the integrity of protected ecosystems. This thesis aims to investigate spatial and temporal patterns of illegal activities, especially poaching, at global and local scales, and to evaluate socio-ecological and geographic factors associated with these patterns in protected areas. Further, it examines efficiency of existing anti-poaching effort in tackling snare poaching and assesses the overall impact of illegal hunting on population viability of wild ungulates in protected areas.

Knowing the spatial patterns of illegal activities can be useful in various ways. First, knowledge may be readily used by the park rangers and guards to improve the efficiency of existing patrol activities and the policing of protected areas (Keane et al., 2008, Critchlow et al., 2015). Second it can be used to prioritize areas and guide the appropriate allocation of conservation budgets (Waldron et al., 2013, Plumptre et al., 2014, Critchlow et al., 2016). Third, it can be used to inform intelligence-led anti-poaching strategies to prevent poaching effectively (Moreto, 2015, Ratcliffe, 2016), and inform the strategies to raise community awareness and build capacity to support conservation (Challender & MacMillan, 2014, Steinmetz et al., 2014).

1.1. Protected areas and conservation approaches

Several different categories of protected areas (PAs) have been created to protect biodiversity, with varying degrees of restriction of the human activities permitted within their boundaries (Chape et al. 2005, Dudley 2008). The International Union for Conservation of Nature (IUCN) places protected areas into categories I to VI with conservation approaches within different PAs varying widely from strict prohibition of nearly all human activities (e.g. category I&III) to less strict protection allowing humans and wildlife to coexist alongside each other (e.g. categories IV-VI), the differences potentially defining their overall success in protecting the species that PAs seek to conserve (Dudley, 2008). In the mid-1990s and early 2000s, interests emerged in assessing the performance of protected areas. Studies by Green et al., (1996) and Hocking et al., (2000) examined conservation in tropical PAs, and highlighted the problems facing them; and suggested requirements for improving PA design and management. Shortly afterwards, Bruner et al., (2001) expanded upon such assessments globally and reported that, overall, many PAs have largely been successful in preventing threats such as land clearing. They identified, however, that exploitation pressures on species are less well controlled and that the PAs need improvements at management levels to achieve long-term conservation effectiveness. More recently, several researchers have reported that strict PAs (IUCN category I-III) are more likely to support more biodiversity than protected areas accorded less strict levels of protection (Naughton-Treves et al., 2005, Stoner et al., 2007, Nelson and

Chomitz, 2011). However, it should be noted that this may reflect the designation process more than the subsequent protection status: richer biodiversity areas may be more likely assigned to higher levels of protection (Waldron et al., 2013). Several additional assessments link various factors such as threat level (e.g. deforestation, illegal exploitation, etc.), geographic location of individual PAs and protected area type to the effectiveness of PAs at protecting biodiversity (DeFries et al., 2005, Butchart et al., 2010, Geldmann et al., 2013).

1.2. The human dimensions of protected areas

Species continue to face serious threats, driven mostly by anthropogenic activities, both inside and outside of protected areas. Outside PAs, human-wildlife conflicts represent a growing conservation challenge, incurring PAs substantial additional costs for managing conflicts through compensation schemes: wild animals such as lions, wild dog, elephants with large ranger areas move in and outside protected areas (Ravenelle & Nyhus, 2017). These conflicts have the potential to increase mortality of the wildlife species being killed legally or illegally, to reduce human costs (Woodroffe, 2000, Dickman et al., 2014, Kahler & Gore, 2015). In the West Kilimanjaro ecosystem for example, Mariki et al. (2015) reported that a herd of six elephants was killed by angry villagers who chased them over a cliff, after the elephants had raided crops on their farms. Similarly, over 70% of felid species (e.g. jaguar, caracal, snow leopard, lion, wild dog etc.) globally have been affected by retaliation killings due to human-wildlife conflicts (Inskip & Zimmermann, 2009). Human-wildlife conflicts may have indirect negative effects on species by undermining local support for conservation, (Kissui, 2008, Dickman, 2010). Wildlife declines also have the potential to alter ecosystem functions, such as changing geochemical cycles, pollination and seed dispersal, and carbon sequestration (Harrison, 2011, Wilkie et al., 2011, Duffy et al., 2017). Further, unsustainable human activities outside protected areas, such as deforestation, agricultural farm expansion, and severing wildlife corridor may increase pressures on the wildlife within protected areas; through reduced landscape level habitat availability and increased isolation (DeFries et al., 2005, Newmark, 2008, Seiferling et al., 2012). This can interrupt species movements and reduce the viability of both resident and migratory animals (Bolger et al., 2008, Ogutu et al., 2011). In Africa, and globally, there is evidence that these threats are responsible for almost 50% decline in the populations of several large mammal herbivores in protected areas (Craigie et al., 2010, Ripple et al., 2015). Elephant and rhino populations (Maisels et al. 2013, Kretzschmar et al., 2016), giraffe population (Strauss et al., 2015) and carnivores such as African wild dog, tiger, dhole, snow leopard, and giant otter (Ripple et al., 2014) are a few example species currently experiencing severe population decline within PAs or as a consequence of deleterious activities outside. Within protected areas, the decline of charismatic species can greatly undermine the social and economic opportunities of local communities, where these species provide commercial opportunities, such as tourism (Naidoo et al., 2016).

There is also a growing trade in bushmeat and live animals, derived from both inside PAs, and from the surrounding landscapes. For example, a recent analysis of illegal wildlife use and trade data in Venezuela found that over 85% of the species traded outside its borders originated from within (Sánchez-Mercado et al., 2016) suggesting that increased demands for the bushmeat and live species from other countries can drive the dynamics of illegal exploitation in PAs locally. In addition to having a huge impact on the biodiversity, illegal wildlife trade can also negatively affect human livelihoods and security (Warchol, 2004, Douglas & Alie, 2014). Essentially, the trade in illegally extracted high value animal products, such as ivory, rhino horn and timber, is worth some $10-35 billion annually (Wyler & Sheikh, 2008). It is believed that some of the profits have been financing organized crime and fuelling insurgency in conflict-prone regions (Warchol, 2004, Wyler and Sheikh 2008); though recent studies dispute such claims as being based on political motives rather than substantive evidence (White, 2014, Duffy, 2016). Further, many globally threatening diseases such as Ebola fever (Leroy et al., 2004) and SARS- associated coronavirus (Bell et al., 2004) are linked to human consumption of contaminated primate bushmeat and illegal live trade in wild carnivores respectively. These diseases are of global health concern and cost countries several billion dollars annually (Karesh et al. 2005). These studies may suggest that targeting the illegal exploitations of species from protected areas may reduce these economic and social ills, as well as protect the species themselves. Additionally, although PAs continue to act as centres of species conservation (Gaines et al., 2010, Thomas & Gillingham, 2015), the current threats affecting them require mitigation and improvement for them to continue in future to protect species and the many ecosystem services that they support.

1.3. Strategies to target illegal activities in protected areas

In the countries that lack sufficient conservation resources, PAs rely on the premise that there will be compliance with existing restrictive regulations and legislation, with minimal enforcement applied to them (Rauset et al., 2016; Rowcliffe et al., 2004). For the PAs that do benefit from enforcement, ranger patrols represent the first strategy commonly used to prevent illegal activities (Keane et al., 2008). However, legislation normally works best when citizens choose voluntarily to abide by its rules (Rowcliffe et al., 2004). Historically (i.e. prior to and in the 1980s), law enforcement of PAs in Africa was characterised by the exclusion of people (potentially alienating them) from wildlife areas (popularly known as 'fences and fines' or the 'fortress conservation' approach). Since then, the complexity of dealing with illegal activities in PAs by law enforcement alone has shifted towards a more inclusive approach that recognizes local people as important in PA conservation endeavours (Songorwa, 1999). From the 1990s, many PAs have adopted a 'carrot and stick' approach to conservation as a strategy to reduce exploitation pressures, encouraging local communities to become partners in conservation activities, as well as retaining the option for enforcement. This was characterized by community conservation approaches and community outreach programs adopted by protected areas in many developing countries, recognizing that local communities are both key players and beneficiaries of conservation (McShane et al., 2011; Songorwa, 1999). Despite some success stories on the performance of this conservation model, e.g. reduction of rhino poaching by over 80% in a reserve in Thailand (Steinmetz et al., 2014) and a doubling of the lion population within a decade in the Mara conservancies, Kenya (Blackburn et al., 2016), poor implementation of this model has been blamed for its failing to realize conservation goals in many regions (Berkes, 2004).

Analytical models have been applied to understand the drivers of poaching within protected areas, enabling improvement of existing conservation strategies. For example, simple models of economic incentives associated with the poaching of high value species, such as rhino and elephant (Milner-Gulland & Leader-Williams, 1992) in South Luangwa National Park in Zambia; and the decisions on prey choice by commercial hunters in the Congo basin have been investigated (Rowcliffe et al., 2004). These all suggest that the effective conservation of species that face high risks of poaching is only likely to be achieved when law enforcement is strict effective. However, the clandestine nature of illegal poaching and the secretive activities of those involved make law enforcement difficult. Thus, controlling wildlife crime within PAs requires ranger patrols that are able to detect, and potentially deter, illegal activities on the ground (Keane et al., 2008; Leader-Williams & Milner-Gulland, 1993; Rowcliffe et al., 2004).

Recently, the use of crime models (e.g. (Andresen, 2006; Chainey et al., 2008), has been gaining use in conservation science to address the poaching crisis. Spatial models are fitted to long-term ranger patrol data to derive spatial crime patterns, revealing poaching hotspots and environmental correlates of poaching. For example, retrospective correlates between poaching data (i.e. wildlife snaring data at least five years) with anthropogenic and landscape features in Ruma National Parks in Kenya (Kimanzi et al., 2014), Zambia's South Luangwa National Park (Watson et al., 2013), Nyungwe National Park in Rwanda (Moore et al., 2017) and Queen Elizabeth in Uganda (Critchlow et al., 2015), have shed light on where future patrols could be directed to target poaching hotspots. Additional studies have also applied these spatial models to data collected over short periods (up to two years; (Wato et al., 2006; Wilfred & MacColl, 2014). These models have been useful in improving patrol strategies and allocating budgets (e.g. (Critchlow et al., 2016; Plumptre et al., 2014). However, many of the studies describing the spatial patterns of poaching show variations in the drivers of poaching between individual protected areas and geographic regions where a PA is located, suggesting site-specific data may be required to improve law enforcement in any particular PA. This is a major challenge. Illegal activities are a common problem in many protected areas across the world, but systematic information on the extent of this problem at a global scale is scarce and site-specific information on the locations and timing of poaching and other illegal activities is often lacking.

1.4. Illegal activities in the Serengeti Ecosystem

The Serengeti ecosystem is an iconic conservation landscape, extending across the borders of Tanzania and Kenya in East Africa. It harbours one of the earth's remaining mass wildlife migrations, with over 2 million wild ungulates trekking over the vast Serengeti plains. The largest part of this ecosystem is in Tanzania, where it covers five contiguous protected areas: a strict Serengeti National Park, three less-strict Game Reserves of Maswa, Ikorongo and Grumeti, and a multiple land use Ngorongoro Conservation Area (Figure 1). It also includes several community wildlife management areas in neighbouring districts. In Kenya, the ecosystem extends into the Masai Mara National Wildlife Reserve and borders several community conservancies. Throughout this thesis, I use Serengeti ecosystem (SE) to refer to the four protected areas (PAs) in Tanzania: the Serengeti National Park and the three game reserves where this research was conducted (Fig 1).

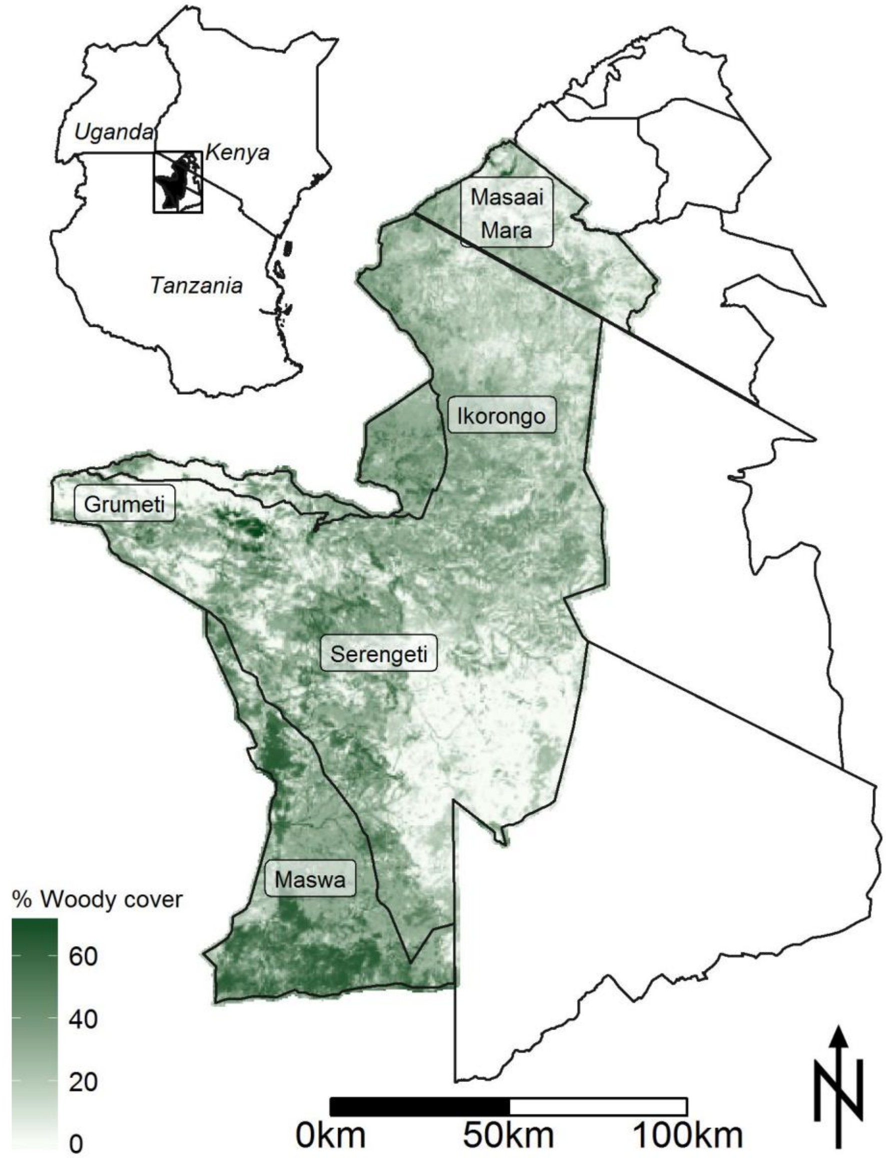

Figure 1. Location of the Serengeti ecosystem in Tanzania (map insert) with the four protected areas, Serengeti National Park, and three Game Reserves; Ikorongo, Maswa and Grumeti where this research was conducted.

The borders of the Serengeti ecosystem abut a growing human population, which influences land use change and exploitation of the wildlife (Estes et al., 2012; Hilborn et al., 2006). With 60 years ecological research in the Serengeti ecosystem (Dublin et al., 1990; Hilborn et al., 2006), several previous studies have looked at the illegal offtake of animals, also known as poaching. The latter has mainly focussed on understanding the drivers of illegal bushmeat consumption within local communities in the ecosystem's western corridor, (Campbell & Hofer, 1995; Loibooki et al., 2002; Ndibalema & Songorwa, 2008; Nuno et al., 2013) and estimating the likely number of wild animals harvested (2-10.5% of Serengeti large herbivore population annually), using poacher encounters with ranger patrols and data from interviews and questionnaire surveys of bushmeat consumptions in the villages bordering Serengeti (Campbell & Hofer, 1995; Mduma et al., 1998; Rentsch & Packer, 2014). The Campbell & Hofer study used information about animals killed by 102 arrested hunters (collected through questionnaires administered by rangers between 1988 and 1992), distance to the home villages of poachers, human population size and they estimated annual offtake of 210000 (10.5%) animals of various species from Serengeti National Park. Mduma and others, on the other hand, estimated that 40000 wildebeest (2%) are illegally killed annually by poachers in the national park. The authors used population models that incorporated long-term wildebeest population census and species demography data (birth rate, mortality rate etc.), and then modelled the wildebeest population as a function of human population increase adjacent to the national park. Mduma et al. (1998) assumed that wire snaring which is used by poachers to kill animals had increased in the park since 1977 as a function of human population increase because antipoaching effort has been minimal: there were often no patrol vehicles or fuel, or insufficient numbers of rangers to conduct patrols (Arcese et al., 1995; Sinclair, 1995b). More recently, Rentsch and Packer (2014) surveyed bushmeat consumption in villages in the western Serengeti corridor, estimating that about 10% (i.e. 100000) of the wildebeest population is illegally hunted each year. Altogether, these studies suggest that the impact of wire snaring (i.e. poaching by wire snares) in the Serengeti can be extraordinarily high, but the long-term effect on these species is poorly documented.

It has been suggested that poaching reduced the population of buffalo in the Serengeti by almost 90%, elephant by 80% between 1970s and early 1990s, and caused the local extinction of rhinos (Dublin et al., 1990). Populations of some species increasing afterwards, for example rhinos have been reintroduced in the recent years. During this period, buffalo population had low rates of increase or failed to increase, especially in areas near human settlements (Metzger et al., 2010). Hilborn et al. (2006) concluded that the reduction in law enforcement patrols (caused by a decrease in conservation funds) during the 1970s resulted in an increase in the poaching of wildlife in the Serengeti, and led to population declines of buffalo and other ungulates. Despite the large existing literature, poaching levels remain high; suggesting that simply knowing how much poaching occurs is insufficient to generate solutions. One explanation for this limitation could be that much of the previous research investigated only the consumer end of poaching, rather than identifying where and when poaching is happening; information which might help the park rangers curb it. More than a decade ago, a modelling study of the costs and benefits of hunting (based on information from interviews of 571 individual arrested hunters between 1980s and 1990s) found that the cost of weapons used in hunting and costs of logistics to travel to the hunting sites were important determinants of the spatial distribution of the hunting activities (Hofer et al., 2000). The authors suggested that poaching in Serengeti could effectively be fought by understanding its spatial distribution within the PAs, but to date this information is still missing. Another explanation for why changes are slow may be that there is a lack of understanding of how the poachers work when they are in the parks trapping animals, and whether existing patrol strategies in the PAs are likely to be efficient in countering illegal activities.

From a research viewpoint, Serengeti is an interesting system because its lack of fences means that the wildlife moves freely between the individual protected areas across the ecosystem. These PAs are also managed through different conservation models (i.e. strictly no take (national park) versus regulated legal offtake (Game reserves)), which have access to differing financial and human resources for conservation. This makes it an appropriate system to examine how different management strategies interact with the dynamics of illegal exploitation of the wildlife. It is important to note that the current conservation strategies implemented in the Serengeti ecosystem are broadly similar to those being adopted in other savanna protected areas in Africa and globally, and thus any results from this ecosystem are relevant to savanna protected areas elsewhere in Africa, and in other continental regions that face similar poaching threats.

The aim of this project is to understand the spatial patterns and trends in illegal activities across a network of protected areas globally and in the Serengeti ecosystem, in particular to provide a deeper understanding of their spatial extent, their impacts and potential mitigations to foster biodiversity conservation in protected areas.

1.5. The thesis objectives and structure of presentation

This study was conceived with four objectives to:

i. Assess the spatial distribution and trends in research on illegal activities in protected areas to identify important factors influencing species decline and to identify the research gaps. This objective is covered in Chapter 2 of this thesis.
ii. Investigate the spatial distribution of illegal activities at a finer-scale, concentrating on landscape correlates of poaching in protected areas within the Serengeti ecosystem. This objective is presented in Chapter 3.
iii. Evaluate ranger patrol efficiency in detecting wire snares used in illegal hunting in protected areas, and to estimate the influence of snares on animal mortality in the Serengeti ecosystem. This forms Chapter 4 of this thesis.
iv. Estimate the sensitivity of the populations of key herbivores in Serengeti to poaching. This information is presented in Chapter 5.

Finally, I discuss the results presented in the foregoing data chapters and their implications for conservation, illegal activity prevention in protected areas, and tackling the illegal wildlife trade in chapter 6.

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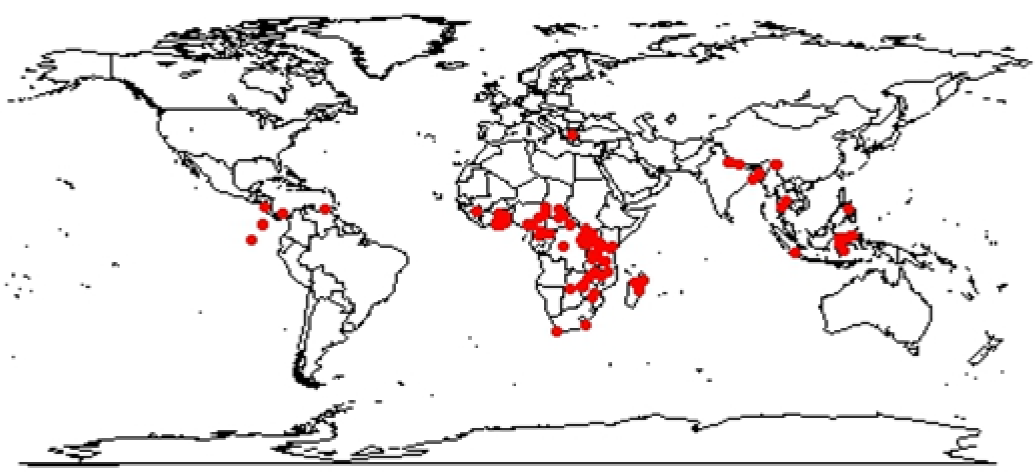
CHAPTER 2 - Spatial distribution and trend in research on illegal activities and influences on wild mammal population declines in protected areas



Spatial distribution of research on illegal activities 1980-2014

2.1. Abstract

Illegal activities are a persistent problem in many protected areas, but an overview of the extent of this problem and its impact is lacking. I review 35 years of research across the globe to examine the spatial distribution of research and socio-ecological factors influencing population decline within protected areas under illegal activities pressure. From 92 papers reporting 1048 species/site combinations, more than 350 species comprising mammals, reptiles, birds, fishes and molluscs were reported to have been extracted illegally from 146 protected areas across four continents. Research in illegal activities has increased substantially during the review period but also shows strong taxonomic and geographic biases towards large wild mammals and African continent respectively, suggesting persistent poaching pressures on wild mammals in African protected areas. Population declines were most frequent i) where there was commercial poaching as opposed to subsistence poaching alone, ii) in countries with a low human development index particularly in strict protected areas and iii) for species with a body mass over 100 kg. Habitat loss associated with greater land use change had an additional significant impact on population decline, particularly in the less-strict categories (IUCN III-VI) of protected area across the continents. Overall, these findings provide evidence that illegal activities are most likely to cause species declines of large-bodied animals in protected areas in resource-poor countries regardless of protected area conservation status (i.e. IUCN category). Given the mounting pressures of illegal activities, additional conservation effort such as improving anti-poaching strategies and conservation resources in terms of improving funding and personnel directed at this problem is a growing priority.

Keywords: human development index, illegal harvest, land use change index, megafauna, population decline, protected areas

2.2. Introduction

Improving the spatial coverage of the protected area (PA) network is increasingly viewed as a global biodiversity conservation priority (Chape et al., 2005; UNEP-WCMC, 2008). The global land area with legal protection for conservation has increased from 3.5% in 1985 (Zimmerer et al., 2004) to 13% recently (Soutullo et al., 2008) and looks set to increase further as countries aim to fulfil the Aichi target of protecting 17% of terrestrial land by 2020 (CBD, 2015). Despite this increase and the investment made in protected areas (Balmford et al., 2003), biodiversity loss is perceived to be continuing even in PAs. For example, Craigie et al. (2010) reported continent-wide population declines of large mammals across several of Africa's protected areas. To allocate conservation resources efficiently, it is important to understand the scale and trends of illegal activities. Most studies, e.g. Leader-Williams et al. (1990); Mitchell (1980); Wright et al. (2000) have focused on a single threat, on a single protected area type or region, and/or over short time periods. These provide crucial PA-specific data, but information on illegal activities and their impact at broader spatial and temporal scales is sparse, making tackling illegal activities difficult. Here I review the site-specific literature to assess the global and regional patterns and impacts of illegal activities on species in protected areas and to provide information on the factors associated with population decline that may help improve the conservation of existing protected areas.

Anthropogenic threats reported from within PAs include hunting, logging, settlement, cultivation, livestock grazing etc. Each can reduce the ability of PAs to preserve biodiversity effectively (Butchart et al., 2010). Moreover, the ability of PAs to protect species can be influenced by factors beyond their boundaries. For example, human-animal conflict has caused widespread mortality of carnivore species at and outside reserve borders, jeopardizing the persistence of populations inside protected areas, particularly for species with large home ranges that encompass both protected and unprotected land (Woodroffe & Ginsberg, 1998). At a larger scale, poaching of migratory species during periods when they are outside PAs has been reported as among the major threats imperiling the long-term conservation of species such as saiga (*Saiga tatarica*) and wildebeest (*Connochaetes taurinus*) antelopes, in Asia and East Africa respectively(Milner-Gulland et al., 2003; Thirgood et al., 2004). The ability of populations to sustain and recover from illegal activities within and surrounding PAs will depend on the type and level of activity, combined with the biological characteristics of the species affected by those activities. For example, selective poaching of males has been associated with the reproductive collapse and population decline of saiga (Milner-Gulland et al., 2003), and species with low rates of reproduction and growth may be particularly prone to population decline and extinction (Cardillo et al., 2005).

While the studies above provide useful accounts of the threats facing PAs, relatively few of them quantify the relative contribution of individual threats to the overall pattern of population change and decline in PAs (Geldmann et al., 2013). Such an assessment is required to identify strategies to improve PA performance, such as where to target additional resources and which actions are most effective at enforcing existing regulations (Bruner et al., 2001). Here, I review research published over 35 years on illegal activities in PAs to understand the global and regional patterns and to assess their impacts on species in protected areas. I evaluate what factors determine the likelihood that illegal activities lead to the decline in the populations of targeted animals. In particular, I assess whether the different legal status in different PAs affects their ability to prevent population declines, and whether attributes of the species (i.e. body mass) and socio-economic context of a country (i.e. human development and agricultural land use change statuses) account for differences in PA success. Finally, I draw on these results to propose recommendations for reducing impacts of illegal activities in protected areas.

2.3. Methods

2.3.1. Data collation

I searched Web of Science, Science Direct, Google Scholar and Scopus for all papers since 1950 using the search terms: illegal activit*AND protected area OR region name (e.g. Europe, Asia, North/South America, Africa and Australia, New Zealand) OR reserve OR biodiversity outcome. Further search was performed incorporating poach / wildlife poaching AND protected areas OR reserve OR region name (as above). All online searches for the publications were conducted between 15th November 2014 and 10th March 2015. I screened the returns based on criteria (see below) that ensured the results from each paper were related to both PAs and illegal extraction of biodiversity;

i. Whether the research was done in a protected area and addressed issues of illegal extraction of biodiversity; animals, plants or both.
ii. Whether the research was on illegal activities by a human population and a mention was clear that the extraction is from the named protected area.
iii. If the research showed impact on species being extracted, decline or not declined
iv. Only used primary data papers not meta-analyses or reviews.
v. Studies covering similar sites and year of data collection were examined for relevance and only one that satisfied all criteria (i-iii) was included in the analysis.

I found no publications that fully satisfied the review criteria for the papers published between 1950 and 1980. Where two or more papers were published for the same protected area during a similar period of data collection, only one that satisfied all the criteria was used for the analysis. From each paper I extracted information on PA location, threat types, study species, perceived impacts of illegal resource use and purpose of research, continent (i.e. Africa, Asia, Europe, South America) and geographic region within continent (i.e. east, central, south and west) (see Table S3 for full details). I recorded population trend (i.e. decline, no decline or unstated) for each PA/species combination from each paper as reported by the paper and the reasons mentioned for such outcome (e.g. illegal hunting, logging) to examine their effect on population status in the PA. For example, if a paper investigated protected areas X and Y and reported impacts of illegal activities on various species i, j, k…, then each species / site combination became one row in the dataset, including the impact scores (1= species decline or 0 = no decline or NA = no reported impact for that species) and any covariate information for PA or paper (author, year of study, PA name, etc.). Where present, the method used to estimate population trend status was recorded. However, as most studies did not describe the data used to arrive at a species outcome (e.g. decline) it was difficult to assess whether the species decline was causal or correlative and I was therefore unable to analyse these data, a common problem in many meta-analysis studies (Taylor et al., 2015).

Further, I extracted body mass data for each species. For mammals, I used the mammal database PanTHERIA (www.pantheria.org) and the current IUCN species red book data (IUCN, 2015). I used body mass of closely-related species for two species that were not located in the mammal database (see Appendix AS1). Body masses for reptile, amphibian and fish species were either extracted from the original papers (if provided) or from credible online material (Appendix AS1). For bird body mass I used Dunning (2007). To identify the geographical location of the study site, I cross-referenced the papers with the WCMC IUCN Protected Planet database (WCMCPP, 2015) to identify the coordinates of the centroid of each PA. I also searched from this PA database for each name of the researched protected area and recorded its appropriate category under the current IUCN-PA categories.

To assess whether population change reported was related to wider scale economic or social change, I extracted country-level human development (HDI) and agricultural land use change (ALC) indices from the UNDP and World Bank databases (UNDP, 2015; WB, 2015). ALC is an index of the amount of land converted to agriculture and other human activities such as settlement. I calculated the ALC over a decade period encompassing the times when research for the reviewed papers were conducted as most papers did not report the exact dates of data collection. I used HDI as a predictor rather than measures of governance (the two are correlated) because HDI is a more direct indicator of development (WB, 2015). Further, to understand the effect of different legal status on species decline I grouped the PAs into two levels of protection: strict PAs (for PAs under IUCN category I & II) and less strict PAs (categories III-VI). Furthermore, I categorized species into two broad groups of mammal and non-mammal for analysis to examine any differences between groups in the way they are threatened by illegal activities. I placed reptile, bird, fish and mollusc into one group: “non-mammal” as data were too sparse for each of these taxa to be tested individually in the model. There was no reported status for any of the plant species in the dataset; therefore, I excluded records of plants from the analysis.

2.3.2. Data analysis

Several papers reported results for many species, and/or information for multiple protected areas. Here I refer to each unique combination of species within an individual PA (i.e. species × PA) reported in a paper as a study, with therefore potentially several studies per paper (see data extraction in methods). For modelling, I filtered all records where the population outcome was unknown. I analysed spatial and temporal trends in research effort of the previous work on illegal activities and identified existing gaps. To assess whether population declines are associated with generic factors such as PA level of protection, type of poaching, species and a country's socio-economic status, I used generalized linear mixed models (GLMM) with a binomial error term and logit link function using the statistical software R (version 3.2 R Development Core Team, 2015).

I built an initial global model incorporating seven fixed factors: human development index (HDI), percentage land use change index (ALC), log species body mass, poaching type (commercial, subsistence, or a mixture), PA level of protection (i.e. IUCN categories classified as strict and less strict), continent (Africa/Asia/America/Europe) and species group (i.e. mammal and non-mammal). Because different species could relate to the same PA and country as studies from other papers at different times, I accounted for this by including country, paper and PA as random effects in all models and fitted the datasets using the 'glmer' function implemented in the R-package 'lme4'(Bolker et al., 2009). I used a backwards stepwise removal of non-significant terms (with Chi-test) to evaluate the relative effect of each factor on the population decline. I obtained model confidence intervals around variables showing statistical significance in the minimum adequate model using the Wald-method (Bolker et al., 2009). Furthermore, in each model I evaluated whether the likelihood of finding an impact on a species' population was due to the PA level of protection and level of hunting (subsistence, commercial or both) and species group and geographic regions. I also examined whether log species body size, country's human development index and agricultural land use change index were consistently correlated with the population decline.

African elephants (i.e. mainly the African savanna elephant *Loxodonta africana*, but with some data for African forest elephant, *Loxodonta cyclotis*; no data on Indian elephants were included in the dataset), hereafter referred to as ‘elephant’, had large numbers of studies in the dataset (contributing 22 % of all records), probably reflecting the increasing concerns for its poaching (Wasser et al., 2015). To check the generality of my results, I repeated the analysis using two subsets of the data: once for elephants alone, and once for all animal species except African elephants. Similarly, because there were sufficient data to analyse some components separately I repeated all the analyses with and without the strict PAs (IUCN categories I&II) as well as on separate subsets of data including Africa and Asia continents to examine their impact on the whole data set and the relative effects of different predictors between the two continents.

2.4. Results

2.4.1. Spatial distribution and trends in research on poaching in PAs

I identified 1598 papers that met the initial search criteria, of which 92 (reporting 1048 species x PA results = ‘studies’) met all the inclusion requirements and were used for analysis. The 92 published papers researched 146 protected areas from four continents, with the largest number of studies from Africa and Madagascar (819 studies), followed by Asia (162), South and Central America (66), and Europe (1) (Figure 2.1).

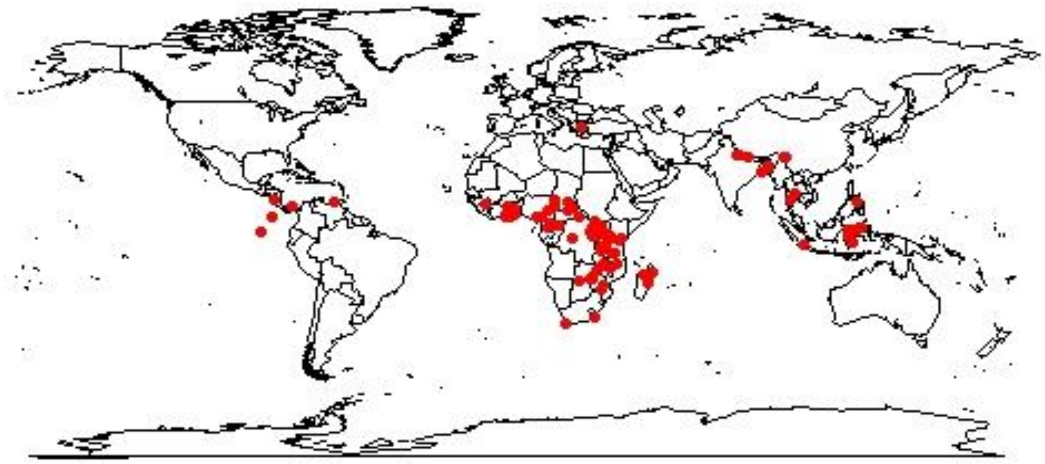


Figure 2.1. The locations of all PAs investigated (red dots correspond to centroids of protected areas). Numbers of studies (species x PA combinations) were: Africa and Madagascar (819), Asia (162), South and Central America (66), and Europe (1).

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Most papers focused on single PA (i.e. local scale, n = 54 papers), or few PAs existing as one contiguous ecosystem and landscape (n = 39 papers). All protected area types were investigated but the IUCN category II level of protection was researched the most (57.35%, n = 65 papers; Figure 2.2.). There was no paper published between 1950 and 1979 that satisfied my search criteria: all relevant papers were published in 1980 onwards.

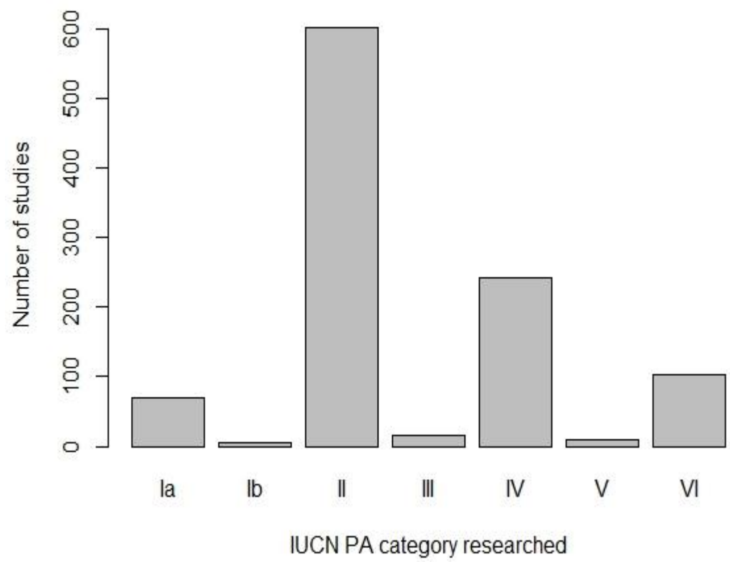

Figure 2.2. The number of studies (species x PA combinations) on illegal activities carried out in PAs with different levels of IUCN protection (I, greatest protection; VI, least). This is based on the 92 papers published between 1980 and 2014, encompassing four continents. There is a strong bias towards the PA category II. The IUCN PA categories are used to facilitate comparisons of different PA 'entities' in different countries (e.g. conservation area, forest reserve, game controlled area, game reserve, game sanctuary, marine reserve, national park and nature reserve).

The research had varying purposes: investigating impacts (71 papers); conservation rationale (e.g. providing new methods for investigating illegal activities; 17 papers) and management/control of illegal activities (5 papers). Further, research has increased substantially during the last 35 years with greater number of published papers since 2005 (Figure S2.1); most of this increase was in Africa.

Furthermore, I found a temporal increase in the variety of research methods throughout the period (Figure 2.3). Interviews (n = 22 papers), animal counts (n = 28) and patrols (n = 9) were the most commonly used methods in the literature. Several research projects (n = 34) combined methods, while two used snare surveys. Animal count was the dominant research method during the first decade (1980-1990) and has increased in use since then.

Use of interviewing and patrolling methods to investigate illegal activities in PAs has been used between 1990 and the present (i.e. 2014).

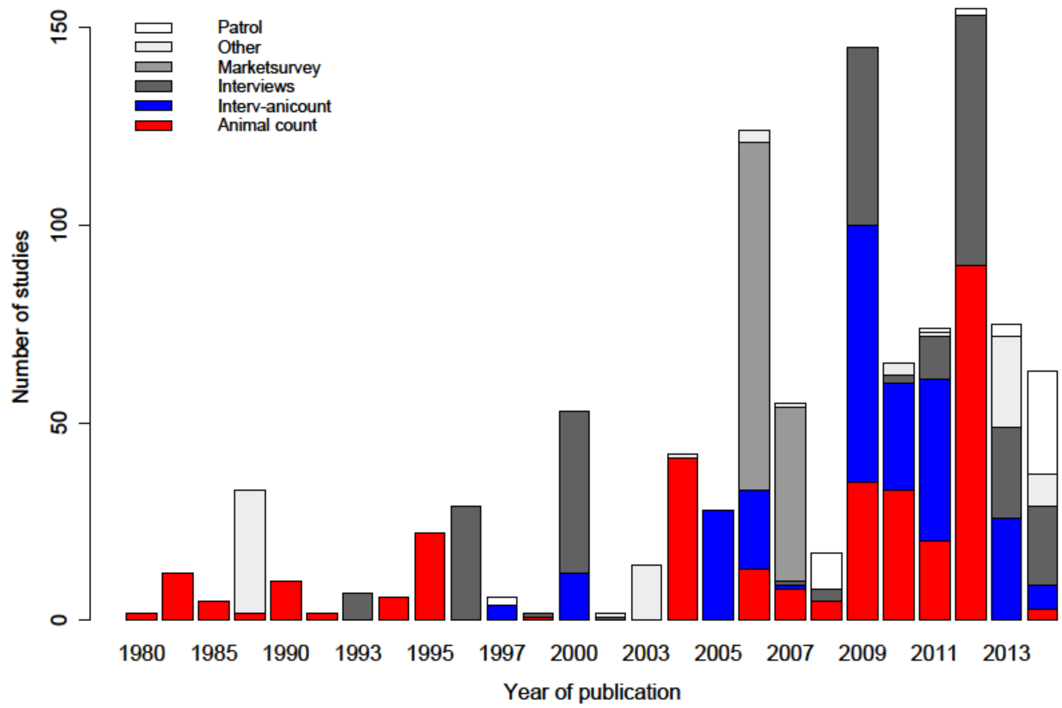

Figure 2.3. Methods used in the 92 publications considered. Recent studies show an increasing diversity of methodological approaches. 'Other' includes bone collection from poacher camps and sporadic field observations.

2.4.2. Patterns of species extraction and country socio-economic status

Three hundred and fifty-three species, comprising mammals (220), reptiles (18), birds (17), fishes (12), molluscs (2) and plants (84) were reportedly harvested illegally in the 146 protected areas (Table S2.1). These species were extracted for subsistence use (10.8%, N = 221), commercial use (19.9%) or both (60.6%). Almost nine percent of the studies did not report a reason of illegal resource extraction, and none of the plant studies included all the data that were required for this analysis. The countries where research was conducted showed varying levels of development (Figure 2.4) and land use change (Figure 2.5).

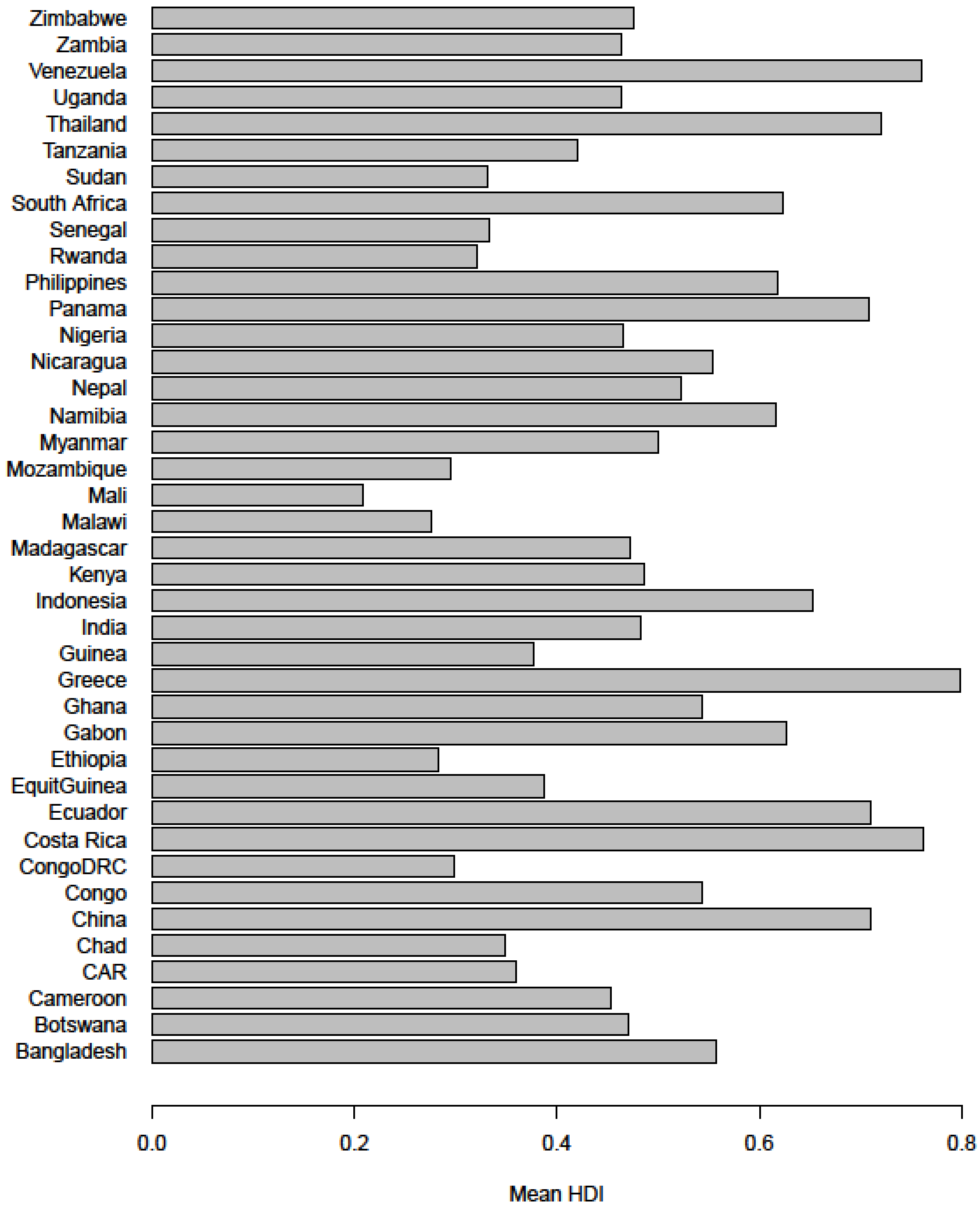

Figure 2.4. Human development index (HDI) for the countries where research was conducted. HDI values shown are the mean scores for the total number of years that a country was researched.

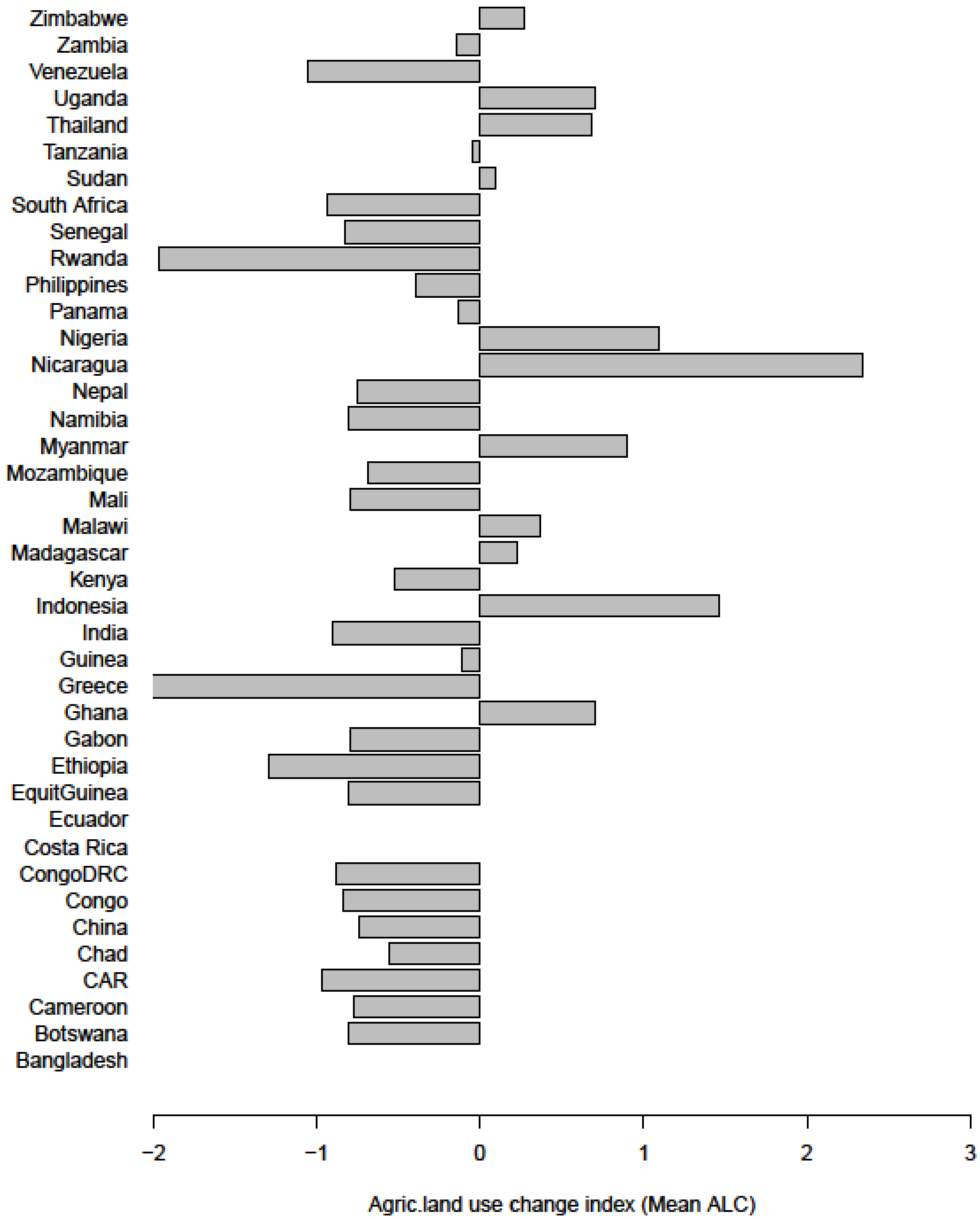

Figure 2.5. Agricultural land use (ALC) change index of a country where research was conducted. Negative change infers to loss of the natural habitats to agricultural activities and settlement. Mean ALC includes the total number of years a particular country PAs were researched during the last 35 years.

2.4.3. Socio-ecological and geographic factors influencing population decline in PAs threatened with illegal activities

Species body mass and species group (mammal or non-mammal) strongly influenced probability of species decline in all PA types with species with greater body mass especially mammals experiencing larger population decline (Model 1 in Table 2.1, Figure 2.6a). There was an effect of the dominant species in my dataset. When I removed the African elephants, I found that species declines were greater in PAs faced with commercial rather than subsistence poaching alone (Figure 2.6b); as before, mammals and species with greater body mass also exhibited the greatest declines in these models (Model 2 in Table 2.1).

Table 2.1. Results of the GLMM best models showing probability of species decline and the explanatory factors in PAs across four continents. Coefficient sign (+/-) indicates size of effect of covariables i.e. increase for plus and decrease for minus, non-mammals in the table (with minus sign-had the lowest probability of decline.

	Type of model and effect size											
	Model1-all PAs/animal species			Model2-all species/PAs excl. elephants			Model3-strict PAs			Model4-less strict PAs		
Factor in final best model	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value
Species body mass	0.225 ± 0.072	1	0.0013	0.243 ± 0.081	1	0.0018	0.379 ± 0.929	1	0.0001			
Human development index							-8.324 ± 3.526	1	0.0168			
Poaching level (subsist)				-0.221 ± 0.924	3	0.0474						
Poaching level (not given)				0.773 ± 1.496								
Poaching level (commercial)				1.879 ± 1.022								
Agric. land use change index										-2.078 ± 0.739	1	0.0006
Species group (non-mammal)	-1.975 ± 0.910	1	0.0119	-1.792 ± 0.873	1	0.0188	-2.384± 1.231	1	0.0161			

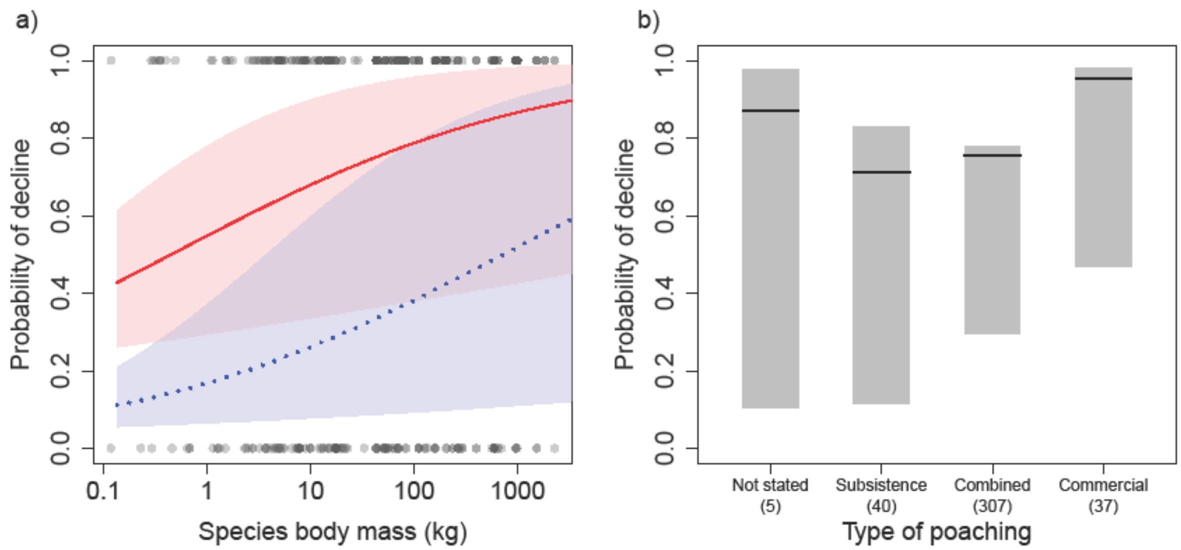

Figure 2.6. Effect of species body mass (a) and level of poaching (b) on the probability of decline of animal species (mammal-upper solid line, and non-mammal-lower dashed line) excluding the dominant species (African elephants) across all PAs (IUCN I-VI). Large bodied species had higher risk of decline when threatened with illegal activities. Shaded area shows 95% CI around the estimates of effect size for the mammal and non-mammal species.

Separate analysis on PAs according to protection level revealed variable results. Human development index (HDI), species body mass and species groups strongly influenced the probability that species would decline in strict PAs (Model 3 in Table 2.1, Figure 2.7). Strict PAs in low human development index countries were associated with increased species decline, of mammal species with greater body mass. In contrast, in less strict PAs (IUCN category III-VI), I found species decline was best explained by agricultural land use change (ALC) in the wider area; i.e. species in all PAs with greater habitat loss were more likely to have experienced population decline (Model 4 in Table 2.1).

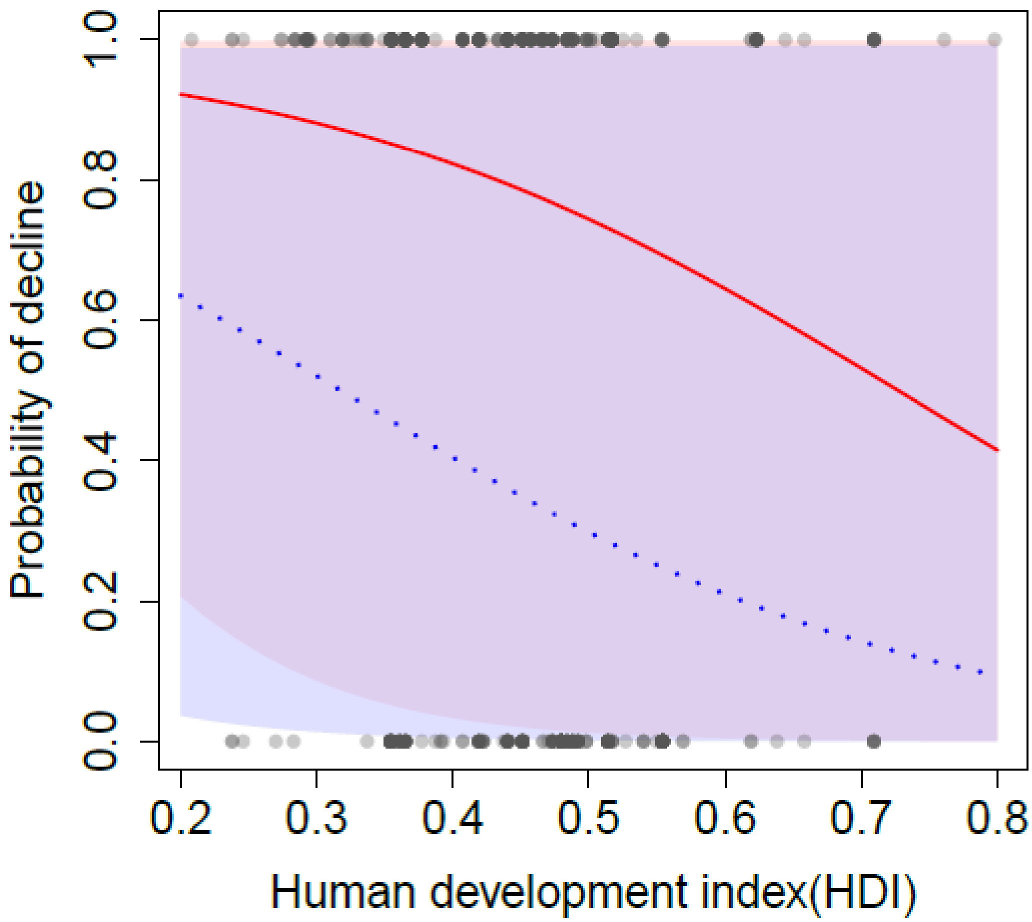

Figure 2.7. HDI influence on population decline of animal species (upper solid line and lower dotted line for mammal and non-mammal respectively) within strict PAs (IUCN category I&II). Least developed countries had the highest probability of their PAs experiencing species decline. Shaded areas show 95% CI around the estimates of effect size of this factor.

The impact of illegal activities on African elephants was variable. Across all PAs (IUCN I-VI), decline was highly associated with lower human development index countries, high habitat loss; i.e. negative agricultural land use change index and with the geographic region of PA location (Model-All PAs in Table 2.2, Figure 2.8). This decline was greatest in central Africa, followed by the east and west; and least in southern Africa. In an analysis of strict PAs alone I found two factors (i.e. human development index and agricultural land use change index) associated with the increase in elephant decline (Model-strict PAs in Table 2.2). On the other hand, I found geographic region significantly associated with increased elephant decline within less strict PAs in central and east Africa (Model-less strict PAs in Table 2.2).

Table 2.2. Results from GLMM models with various factors explaining the probability of decline of African elephants across all PA types and separately between strict and less strict PAs.

	Type of model and effect size								
	All PAs types (cat. I-VI)			Strict PAs (cat. I & II)			Less strict PAs (cat. III-VI)		
Factor in final best model	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value
Human development index	-12.350 ± 4.207	1	0.0005	-19.459 ± 6.496	1	0.0004			
Agri. land use change index	-1.461 ± 0.699	1	0.0116	-1.173 ± 0.826	1	0.0371			
Africa zone (East)	0.723 ± 1.116	3	0.0100				0.406 ± 1.307	3	0.0282
Africa zone (South)	-2.186 ± 1.041						-2.773 ± 1.275		
Africa zone (West)	0.075 ± 2.749						0.981 ± 0.677		

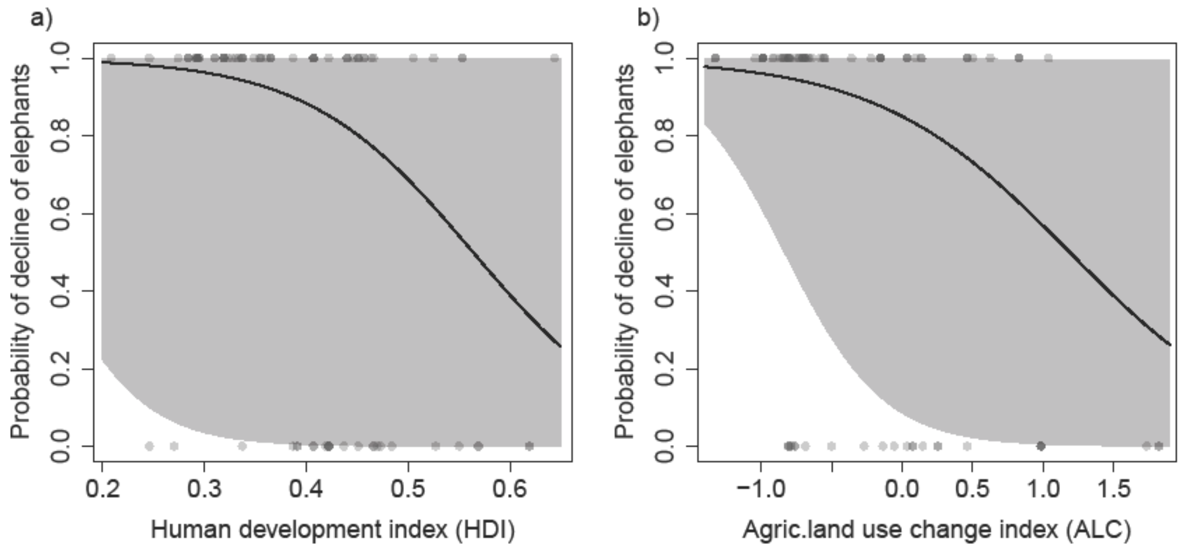

Figure 2.8. The country's human development index (a) and agricultural land use change index (b) as best predictors of decline in the African elephant within in all PAs (category I-VI) across the African continent. High negative ALC values are associated with severe habitat loss and a high risk of population decline of African elephants. Gray area shows 95% CI around the estimates in the minimum adequate GLMM model.

Furthermore, the probability that illegal activities were associated with population declines also varied regionally across PAs in Africa, Asia and Latin America. Lower human development index of countries, greater species body mass and species group (i.e. mammal) were the strongest predictors of increased species decline in PAs across Africa ((Model -Africa in Table 2.3). In contrast, species decline in Asia was strongly positively associated with PA strictness (Model-Asia in Table 2.3). On the combined data for Asia and Latin America, I found that increased probability of species decline in PAs across these continents was correlated with greater agricultural land use change (Model combined in Table 2.3).

Table 2.3. Results from GLMM models with influence of the various factors on the probability of species decline in PAs separately for Africa and Asia continents and on combined data for Asia and Latin America.

	Type of model and effect size								
	Africa-all animals/PAs			Asia-all animals/PAs			Asia + L. America		
Factor in final best model	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value	Coefficients + SE	DF	p-value
Human development index	-8.657 ± 3.708	1	0.00386						
Body mass	0.338 ± 0.084	1	0.00022						
Species group (non-mammal)	-2.269 ± 1.095	1	0.01223						
PA strictness				1.974 ± 0.684	1	0.0237			
Agric. land use change index							-13.946 ± 4.741	1	0.0077

2.5. Discussion

Biodiversity in protected areas faces numerous anthropogenic threats (Butchart et al., 2010). I analysed data published since the 1980's to understand impacts of illegal activities on species decline in protected areas. There were a strong taxonomic and geographic biases in research on illegal activities within protected areas during the review period. I found that population declines were more likely consequences of illegal activities in countries with low human development index (HDI). Different groups of species declined at different rates, with large bodied mammals mostly likely to show population decline. As well as poaching, I also found that habitat loss had an additional impact on population decline of animal species in less strict PAs, particularly in countries experiencing greater agricultural land use change. Further, species decline was also associated with geographic region of PA location being greater in Africa than Asia or Latin America.

The identification of correlations between human development indices and illegal activities in this study supports a widely held view, e.g. Adams et al. (2004); Bennett et al. (2007), but one that is often based on limited data (Nellemann et al., 2014): that biodiversity decline is greater in relatively poor regions. Low human development scores could impact illegal activities in two ways: firstly, poor people may tend to exploit species illegally from PAs because they have limited alternatives (Brashares et al., 2011). Secondly, poor countries have fewer resources to invest in PA conservation. Underfunding may result in increased illegal activities in PAs due to insufficient law enforcement (Keane et al., 2008). This is supported by my model encompassing the most strictly protected areas alone, which indicates high probability of species decline associated with low human development index countries (Table 2.1). Hilborn et al. (2006) demonstrated that increased funding budgets for anti-poaching activities in the Serengeti National park greatly reduced poaching pressures and lead to the recovery of the buffalo population. However, increasing conservation funding may not necessarily result into improved conservation particularly when social and political constraints exist. For example, social and political unrest may increase rates of illegal activities, reduce wildlife populations and thwart conservation efforts altogether (Bouche et al., 2012; de Merode et al., 2007). My results provide evidence that poverty, in as much as it is measured by the HDI, may have significant negative impacts on species due to accelerated poaching, whether that be because of increased external pressures on PAs or decreased policing and protection within strict PAs. Further, illegal activities increased species vulnerability to decline through increased habitat loss mostly in less strictly protected areas (Table 1). This could be because these PAs are often afforded minimal protection (Joppa & Pfaff, 2011) and therefore this exposes them to intense illegal activities making them less reliable for the effective conservation of large and medium-size mammals (Caro, 1999). These findings also provide evidence that supports Craigie et al.'s (2010) assertions that illegal hunting and habitat loss are the major causes of continental-wide declines in megafauna in PAs across Africa. My study highlights the need to consider human development issues more seriously to ensure effective conservation of biodiversity within existing protected areas.

Large bodied species are likely highly susceptible to decline because they have slow growth rates and so overharvesting is likely to cause population decline (Purvis et al., 2000). Low population growth rates in combination with multiple threats from poaching and diseases are known to have significant impact on population persistence (Cardillo et al., 2005; Woodroffe & Ginsberg, 1998). By contrast, smaller mammals (with higher reproductive and growth rates) showed fewer declines and appeared to sustain harvest, though relatively few small species are the specific targets of poaching in the PAs. My model that excluded elephant suggested that commercial poaching has the greatest potential to cause population decline. Further, the propensity for elephants to decline in low human development index countries and geographic regions, and where habitat loss was taking place (Table 2. 3), is consistent with recent analyses that this species is threatened with poaching and habitat loss across its habitat range in Africa (Maisels et al., 2013; Wittemyer et al., 2014).

The pattern of species declines across the network of protected areas is worrying and suggests that PA policing (including access to appropriate conservation information) and resources need to be improved. PA-specific information is important for understanding how illegal activities vary spatially and across time and there is a need to be able to predict future trends and thereby possible future management strategies e.g. Critchlow et al. (2015). Furthermore, land use change poses additional risks of species decline in PAs across Asia and Latin America, relative to Africa, and its impact was more severe in less strict than strict PAs (Table 2.3). This could be attributable to the habitat loss and poaching occurring inside these protected (Corlett, 2007; Curran et al., 2004; Leisher et al., 2013), or the wider effects on animals that roam outside protected areas for parts of the year. These results are consistent with previous studies that have reported biodiversity decline and loss within PAs in these regions e.g. Geldmann et al. (2013); Harrison (2011); Laurance et al. (2012). My findings suggest that land use change is a major threat that requires urgent attention to improve PA conservation across Latin America and continental Asia.

The geographical bias in the spatial distribution of research observed in these data is likely a consequence of interests among the researchers rather than being driven solely by the levels of illegal activities in particular PAs or countries. However, the temporal and spatial patterns of research observed in this study provide insight into the extent of the problem of illegal activities in PAs and therefore suggest that PAs are currently in need of new strategies to minimize impacts of illegal activities and to improve their conservation effectiveness (Watson. et al., 2016). To date, research effort has concentrated on quantifying the extent and impact of illegal activities on focal species; in other words, documenting the problem. Far less information is available on which conservation management strategies (including human development and preventing illegal international trade, as well as within-PA activities) are most effective at reducing illegal activities. Only 5 out of 92 publications reviewed here explicitly considered how the management of illegal activities might affect population declines. New research should focus on developing and testing new methods for reducing levels and impacts of illegal activities on species in PAs.

2.5.1. Conservation implications

Tackling illegal activities within protected areas remains a high conservation priority. My results suggest that a combination of strategies may be required that simultaneously reduce the frequency with which illegal activities are attempted and reduce the likelihood that such attempts will be successful (from the perspective of the perpetrators). Regarding the former, I found that illegal activities in poor countries often lead to population declines within PAs.

These findings suggest that poverty alleviation may be an appropriate conservation strategy to reduce illegal activity pressures (Bennett et al., 2007). The implication of this for local and national policies is that more effort needs to be invested to improve the social and economic status of the human populations. This needs to work in tandem with increasing the effectiveness of traditional conservation activities to prevent illegal activities; which may itself reduce the inclination of people to attempt future illegal activities. At the international level, these results may imply that PAs in low HDI countries may need more international support to curb pressures of illegal activities, particularly those driven by trans-boundary forces. Sound strategies to stop elephant poaching and ivory trade, and trade in bushmeat, as well as strengthening collaboration in conservation and research, are necessary to improve PA effectiveness in these countries.

2.6. Acknowledgements

I am grateful to R. Critchlow for the comments on earlier drafts of this chapter and for cross-checking the papers I retrieved from online databases.

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2.8. Supplementary Results

Table S2.1. The reasons recorded by the studies to explain the status of biodiversity reported in the reviewed papers. Plus (+) sign indicates the reason was mentioned together with the poaching in the PA. Studies (43.2%, n = 453) that sum up to the totals did not indicate the status of species being investigated and are not shown here.

Reasons cited for species decline	All studies	reported decline	reported no decline
Poaching	884	255	201
+Drought	1	1	0
+Habitat loss and grazing by domestic animals	43	16	3
+Predation by hyena	1	1	0
+Civil conflicts/war	2	1	1
+Floods	1	1	0
+Legal hunting	23	13	10
+Water scarcity	37	17	20
+Disease	36	26	9
+Logging	12	2	10
Not mentioned	8	0	8
Total	1048	333	262

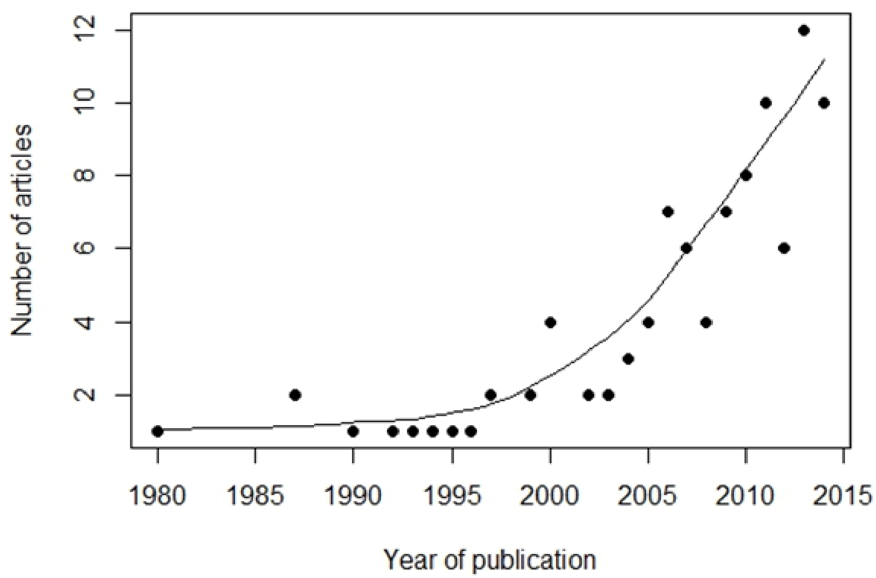

Figure S2.1. Increasing publication trend for illegal activities in PAs during the last 35 years.

Table S2.2. 353 species extracted and threatened with illegal activities in 146 PAs across four continents as published in the last 35 years (1980-2014)

Species	Taxon	IUCN- PA category	Cited threat	# of assessments	Publication period	Continent
Artocarpus chaplasha	Plant	II	Poaching	2	2014	Asia
Catopuma temminckii	mammal	II	Poaching	1	2005	Asia
Cephalophus rufilatus	mammal	II	Poaching	1	2009	Africa
Cephalophus silvicultor	mammal	II	Poaching	1	2009	Africa
Giraffa camelopardalis	mammal	VI	poaching and legal hunting	1	2011	Africa
Hystrix cristata	mammal	II	Poaching	1	2009	Africa
Lophura leucomelanos	bird	II	Poaching	1	2005	Asia
Macaca mulatta	mammal	II	Poaching	1	2005	Asia
Martes flavigula	mammal	II	Poaching	1	2005	Asia
Muntiacus muntjak	mammal	II	Poaching	1	2005	Asia
Paguma larvata	mammal	II	Poaching	1	2005	Asia
Acacia auriculiformis	Plant	II	Poaching	2	2014	Asia
Acacia mangium	Plant	II	Poaching	2	2014	Asia
Accipiter francesii	bird	II	Poaching	1	2003	Africa
Aceros nipalensis	bird	IV	Poaching	1	2010	Asia
Aceros nipalensis	bird	II	Poaching	1	2005	Asia
Acinonyx jubatus	mammal	II, IV	poaching and legal hunting	4	1995-2013	Africa
Acrocar Pearsonus sp.	Plant	II	poaching	1	2012	Africa
Aepyceros melampus	mammal	II, IV, VI	poaching, water scarcity, legal hunting	13	1995-2013	Africa
Ailurus fulgens	mammal	II, IV	poaching	2	2010-2011	Asia, Africa
Albizia grandibracteata	Plant	II	poaching	1	2012	Africa
Albizzia procera	Plant	II	poaching	2	2014	Asia
Albizzia saman	Plant	II	poaching	2	2014	Asia
Alcelaphus buselaphus	mammal	Ia,II, IV, VI	poaching, livestock grazing, water scarcity, disease	15	1995-2013	Africa
Allophylus oxidentalis	Plant	IV	poaching	1	2004	America
Alopias superciliosus	Fish	VI	poaching	1	2013	America
Alouatta palliata	mammal	III	poaching	1	2000	America
Amblysomus hottentotus	mammal	IV	poaching and grazing	2	2009	Africa
Anas platyrhynchos	bird	II	poaching	1	1996	Africa
Apeiba tibourbou	Plant	IV	poaching	1	2004	America
Aphania sp.	Plant	II	poaching	1	2012	Africa
Arctictis binturong	mammal	II	poaching	2	2005-2012	Asia
Arctocebus calabarensis	mammal	IV	poaching	1	2007	Africa
Arctonyx collaris	mammal	II	poaching	2	2012	Asia
Argus sp	bird	IV	poaching	1	2010	Asia
Artocarpus heterophyllus	Plant	II	poaching	2	2014	Asia
Atelerix albiventris	mammal	II, IV	poaching	2	1996-2000	Africa
Atherurus africanus	mammal	II, IV, V	poaching and legal hunting	5	2000-2009	Africa

Atherurus macrourus	mammal	II	poaching	1	2005	Asia
Atilax paludinosus	mammal	IV	poaching and grazing	3	2007-2009	Africa
Avahi occidentalis	mammal	II	poaching	1	2003	Africa
Aves spp.	bird	Ia, II, IV	poaching	10	2006-2009	Africa, Asia
Axis axis	mammal	V	poaching	1	2013	Asia
Bat spp	mammal	Ia, II, IV	poaching	10	2007	Africa, Asia
Batoid spp	Fish	II	poaching	1	2014	America
Bdeogale nigripes	mammal	IV	poaching	1	2007	Africa
Bitis arientans	reptile	II	poaching	1	2009	Africa
Bitis gabonica	reptile	II, IV	poaching	2	2007-2009	Africa
Blighia spp.	Plant	II	poaching	1	2012	Africa
Boa constrictor	reptile	II	poaching	1	1996	Africa
Bos gaurus	mammal	II	poaching	2	2012	Asia
Bridelia micrantha	Plant	II	poaching	1	2012	Africa
Budorcas taxicolor	mammal	II, IV	poaching	2	2010-2011	Asia, Africa
Bunopithecus hoolock	mammal	II, IV	poaching	2	2005-2010	Asia
Calycophyllum candidissimum	Plant	IV	poaching	1	2004	America
Canis adustus	mammal	IV	poaching	1	2000	Africa
Canis aureus	mammal	II	poaching	1	2012	Asia
Canis spp	mammal	IV	poaching and grazing	2	2009	Africa
Capricornis sumatraensis	mammal	IV	poaching	4	2010	Asia, Africa
Carcharhinus falciformis	Fish	II, VI	poaching	2	2013-2014	America
Carcharhinus galapagensis	Fish	VI	poaching	1	2013	America
Catopuma temminckii	mammal	II, IV	poaching	3	2010-2012	Asia, Africa
Cebus capucinus	mammal	III	poaching	1	2000	America
Celtis iguanaea	Plant	IV	poaching	1	2004	America
Celtis spp.	Plant	II	poaching	1	2012	Africa
Cephalophus callipygus	mammal	II	poaching and logging	1	2011	Africa
Cephalophus dorsalis	mammal	II, IV, V	poaching, logging and legal hunting	6	2000-2011	Africa
Cephalophus leucogaster	mammal	II	poaching and logging	1	2011	Africa
Cephalophus natalensis	mammal	IV	poaching	2	2006	Africa
Cephalophus nigrifrons	mammal	II	poaching and logging	2	1997-2011	Africa
Cephalophus ogilbyi	mammal	II, IV	poaching	3	2006-2007	Africa
Cephalophus rufilatus	mammal	Ia, II	poaching	5	1996-2012	Africa
Cephalophus silvicultor	mammal	Ia, II, IV	poaching and logging	5	2007-2012	Africa
Cephalophus spadix	mammal	IV	poaching	1	2006	Africa
Cephalophus spp	mammal	Ib, IV-VI	poaching	4	2009	Africa
Ceratogymna atrata	bird	IV	poaching	1	2007	Africa
Ceratotherium simum	mammal	IV, VI	poaching, livestock grazing, legal hunting	3	2009-2011	Africa
Cercocebus agilis	mammal	II	poaching	1	2006	Africa
Cercocebus torquatus	mammal	II, IV	poaching, habitat loss	4	2006-2011	Africa
Cercopithecus ascanius	mammal	II	poaching	1	2006	Africa
Cercopithecus erythrotis	mammal	II, IV	poaching, habitat loss	4	2006-2011	Africa
Cercopithecus mitis	mammal	Ib, IV, VI	poaching, livestock grazing	6	2006-2009	Africa

Cercopithecus mona	mammal	II, VI	poaching, habitat loss	5	2006-2011	Africa
Cercopithecus neglectus	mammal	II	poaching	1	2006	Africa
Cercopithecus nictitans	mammal	II, IV	poaching, habitat loss	4	2006-2011	Africa
Cercopithecus pogonias	mammal	II, IV, V	poaching, legal hunting, habitat loss	3	2000-2011	Africa
Cercopithecus spp	mammal	II	poaching and logging	1	2011	Africa
Chelonia spp.	reptile	II, IV	poaching	2	2007-2014	Africa, America
Chlorocebus aethiops	mammal	Ia, II, IV	poaching, livestock grazing	9	2000-2012	Africa
Chomelia speciosa	Plant	IV	poaching	1	2004	America
Citrus limon	Plant	IV	poaching	1	2004	America
Civettictis civetta	mammal	II, IV	poaching	5	1996-2007	Africa
Coccoloba caracasana	Plant	IV	poaching	1	2004	America
Colobus guereza	mammal	Ia, II	poaching	4	2006-2012	Africa
Colobus polykomos	mammal	II	poaching	1	2009	Africa
Connochaetes taurinus	mammal	II, IV, VI	poaching, livestock grazing, legal hunting	6	1995-2013	Africa
Cricetomys emini	mammal	II, IV	poaching	3	2006-2007	Africa
Cricetomys gambianus	mammal	II, IV	poaching	3	2006-2009	Africa
Crocodylus niloticus	reptile	II	poaching	1	2013	Africa
Crocodylus porosus	reptile	II	poaching	2	1996	Africa
Crocuta crocuta	mammal	II	poaching	3	1996-2013	Africa
Crossarchus obscurus	mammal	IV	poaching	1	2007	Africa
Cryptoprocta ferox	mammal	II, VI	poaching	3	2003-2012	Africa
Cuon alpinus	mammal	II	poaching	2	2005-2012	Asia
Cupania dentata	Plant	IV	poaching	1	2004	America
Damaliscus korrigum	mammal	II, VI	poaching	2	1995-2009	Africa
Damaliscus lunatus	mammal	II, VI	poaching and water scarcity	2	2012-2013	Africa
Dasyprocta punctata	mammal	III	poaching	1	2000	America
Delphinid spp	Fish	II	poaching	1	2014	America
Dendroaspis jamensoni	reptile	IV	poaching	1	2007	Africa
Dendrohyrax dorsalis	mammal	IV	poaching	1	2007	Africa
Diceros bicornis	mammal	II, IV-VI	poaching, habitat loss, disease, legal hunting	28	1981-2014	Africa
Didelphis marsupialis	mammal	III	poaching	1	2000	America
Diospyros abyssinica	Plant	II	poaching	1	2012	Africa
Dombeya mukole	Plant	II	poaching	1	2012	Africa
Dracaena steudneri	Plant	II	poaching	1	2012	Africa
Dremomys lokriah	mammal	II	poaching	1	2005	Asia
Egretta alba	bird	II	poaching	1	2003	Africa
Elephas maximus	mammal	II	poaching	1	2012	Asia
Equus burchellii	mammal	II	poaching	2	1995-2006	Africa
Equus quagga	mammal	IV	poaching, water scarcity, livestock grazing, legal hunting	9	2009-2013	Africa
Erythrina abyssinica	Plant	II	poaching	1	2012	Africa
Erythrocebus patas	mammal	Ia, II	poaching	5	2006-2012	Africa
Eucalyptus spp.	Plant	II	poaching	1	2012	Africa
Eudorcas rufifrons	mammal	II	poaching	1	1996	Africa
Eudorcas thomsonii	mammal	II	poaching	2	1995-2005	Africa

Eugenia salamensis	Plant	IV	poaching	1	2004	America
Eulemur fulvus	mammal	II	poaching	1	2003	Africa
Eulemur rufifrons	mammal	II, VI	poaching	1	2012	Africa
Fagara angolensis	Plant	II	poaching	1	2012	Africa
Felis chaus	mammal	II	poaching	1	2005	Asia
Ficus insipida	Plant	IV	poaching	1	2004	America
Ficus obtusifolia	Plant	IV	poaching	1	2004	America
Ficus spp.	Plant	II	poaching	1	2012	Africa
Funisciurus pyrropus	mammal	IV	poaching	1	2007	Africa
Funtumia spp.	Plant	II	poaching	1	2012	Africa
Galago spp.	mammal	IV	poaching	2	2006-2007	Africa
Galeocerdo cuvier	Fish	VI	poaching	1	2013	America
Gallus gallus	bird	II	poaching	1	2005	Asia
Genetta genetta	mammal	II	poaching	1	2013	Africa
Genetta sp.	mammal	II, IV	poaching	5	2000-2007	Africa
Genetta tigrina	mammal	IV	poaching and grazing	2	2009	Africa
Giraffa camelopardalis	mammal	Ia, II, VI	poaching, disease, water scarcity	14	1995-2013	Africa
Gmelina arborea	Plant	II	poaching	2	2014	Asia
Gorilla gorilla	mammal	II	poaching, logging	4	1996-2011	Africa
Guazuma ulmifolia	Plant	IV	poaching	1	2004	America
Gutera plumifera	bird	IV	poaching	1	2007	Africa
Gypohierax angolensis	bird	IV	poaching	1	2007	Africa
Haliotis midae	Mollusc	II, IV	poaching	3	1999-2013	Africa, Asia
Hamelia patens	Plant	IV	poaching	1	2004	America
Helarctos malayanus	mammal	II, IV	poaching	4	2005 2011	Asia, Africa
Helogale parvula	mammal	IV	poaching	1	2000	Africa
Herpestes urva	mammal	II	poaching	1	2012	Asia
Heterohyrax brucei	mammal	IV	poaching	1	2000	Africa
Hipotragus equinus	mammal	II, IV	poaching	2	2013-2014	Africa
Hippopotamus amphibius	mammal	Ia, II, IV, VI	poaching, disease, water scarcity	11	1995-2013	Africa
Hippotragus equinus	mammal	Ia, IV, VI	poaching, disease, water scarcity	9	2000-2013	Africa
Hippotragus niger	mammal	II, IV, VI	poaching, legal hunting, water scarcity	6	2000-2013	Africa
Hyemoschus aquaticus	mammal	II, IV	poaching	3	2006-2007	Africa
Hylochoerus meinertzhageni	mammal	Ia, II	poaching	4	2006-2012	Africa
Hylopetes spadiceus	mammal	II, IV	poaching	2	2010-2011	Asia, Africa
Hyracoidea spp.	mammal	IV	poaching	1	2006	Africa
Hystrix africaeaustralis	mammal	IV	poaching, livestock grazing	3	2000-2009	Africa
Hystrix brachyura	mammal	II, IV	poaching	5	2005-2014	Asia, Africa
Hystrix cristata	mammal	II	poaching	3	1996-2013	Africa
Ichneumia albicauda	mammal	IV	poaching	1	2000	Africa
Ictonyx striatus	mammal	IV	poaching	1	2000	Africa
Istiophorid spp	Fish	II	poaching	1	2014	America
Isurus oxyrinchus	Fish	VI	poaching	1	2013	America
Jacaranda sp.	Plant	II	poaching	1	2012	Africa

Jacquinia aurantiaca	Plant	IV	poaching	1	2004	America
Jasus lalandii	Mollusc	II	poaching	1	2013	Africa
Karwinskia calderonii	Plant	IV	poaching	1	2004	America
Kinixys erosa	reptile	II	poaching	2	2006	Africa
Kobus ellipsiprymnus	mammal	Ia, II, IV, VI	poaching, disease, water scarcity	13	2000-2013	Africa
Kobus kob	mammal	Ia, II	poaching and floods, disease	8	1996-2012	Africa
Kobus vardonii	mammal	II, VI	poaching	3	2009-2013	Africa
Lepilemur edwardsi	mammal	II	poaching	1	2003	Africa
Leptosomus discolor	bird	II	poaching	1	2003	Africa
Lepus saxatilis	mammal	II	poaching	1	2009	Africa
Lepus sp.	mammal	IV	poaching	1	2000	Africa
Lepus victoriae	mammal	II	poaching	1	1996	Africa
Litocranius walleri	mammal	II	poaching	1	2006	Africa
Lophocebus albigena	mammal	II	poaching and logging	1	2011	Africa
Loxodonta africana	mammal	Ia, Ib, II, IV-VI	poaching, drought, disease, water scarcity	73	1980-2014	Africa
Loxodonta cyclotis	mammal	Ia, Ib, II, IV-VI	poaching, drought, disease, water scarcity	28	1980-2014	Africa
Luehea candida	Plant	IV	poaching	1	2004	America
Lutra lutra	mammal	II, IV	poaching	2	2010-2011	Asia, Africa
Lycaon pictus	mammal	II, VI	poaching and legal hunting	3	2011-2013	Africa
Macaca arctoides	mammal	II, IV	poaching	3	2005-2010	Asia, Africa
Macaca assamensis	mammal	II, IV	poaching	1	2010-2011	Asia, Africa
Macaca fascicularis	mammal	II	poaching	1	1996	Africa
Macaca nemestrina	mammal	II	poaching	1	2012	Asia
Macaranga schweinfurthii	Plant	II	poaching	1	2012	Africa
Madoqua kirkii	mammal	II, IV	poaching	1	2000-2006	Africa
Maesa lanceolata	Plant	II	poaching	1	2012	Africa
Mammal sp	mammal	Ia, II, IV, VI	poaching	25	2007-2014	Africa
Mandrillus leucophaeus	mammal	II, IV	poaching, habitat loss	4	2006-2011	Africa
Manilkara chicle	Plant	IV	poaching	1	2004	America
Manis gigantea	mammal	II	poaching	2	1996-2009	Africa
Manis javanica	mammal	II, IV	poaching	4	2005-2010	Asia, Africa
Manis temminckii	mammal	IV	poaching	1	2000	Africa
Manis tetradactyla	mammal	IV	poaching	1	2007	Africa
Manis tricuspis	mammal	II, IV	poaching	3	2006-2007	Africa
Manouria impressa	reptile	II, IV	poaching	2	2005-2010	Asia
Markhamia spp.	Plant	II	poaching	1	2012	Africa
Martes flavigula	mammal	II, IV	poaching	2	2010-2011	Asia
Mastichodendron capiri var. tempisque	Plant	IV	poaching	1	2004	America
Mazama americana	mammal	III	poaching	1	2000	America
Melicocca bijugatus	Plant	IV	poaching	1	2004	America
Mellivora capensis	mammal	II, IV	poaching	2	1995-2000	Africa
Millettia dura	Plant	II	poaching	1	2012	Africa
Monachus monachus	mammal	II	poaching	1	2004	Europe

Moschus sp	mammal	IV	poaching	2	2010-2011	Asia, Africa
Mungos mungo	mammal	IV	poaching	1	2000	Africa
Muntiacus feae	mammal	II	poaching	1	2014	Asia
Muntiacus muntjak	mammal	IV	poaching	3	2010-2014	Asia
Muntiacus putaoensis	mammal	II	poaching	1	2005	Asia
Muntjac sp	mammal	II	poaching	1	2011	Africa
Mustela strigidorsa	mammal	IV	poaching	1	2010	Asia
Mustelidae spp.	mammal	IV	poaching	1	2006	Africa
Naemorhedus baileyi	mammal	II	poaching	1	2011	Africa
Naja spp.	reptile	IV	poaching	1	2007	Africa
Nandinia binotata	mammal	II, IV	poaching	3	2006-2007	Africa
Nanger granti	mammal	II	poaching	2	1995-2006	Africa
Nasua narica	mammal	III	poaching	1	2000	America
Neoboutonia sp.	Plant	II	poaching	1	2012	Africa
Neofelis nebulosa	mammal	II, IV	poaching	4	2005, 2010	Asia, Africa
Neotragus moschatus	mammal	Ib, II, IV, VI	poaching	5	2000-2009	Africa
Newtonia buchananii	Plant	II	poaching	1	2012	Africa
Numida meleagris	bird	II	poaching	2	1996-2009	Africa
Odocoileus virginianus	mammal	III	poaching	1	2000	America
Olea welwitschii	Plant	II	poaching	1	2012	Africa
Oreotragus oreotragus	mammal	II, IV	poaching	2	2000- 2013	Africa
Orycteropus afer	mammal	II	poaching	2	2006-2009	Africa
Oryx beisa	mammal	II	poaching	1	2006	Africa
Osteoaemus tetraspis	reptile	II, IV	poaching	3	2006-2007	Africa
Otolemur crassicaudatus	mammal	IV	poaching	1	2000	Africa
Ourebia ourebi	mammal	Ia, II, IV, VI	poaching, disease	7	2000-2012	Africa
Paguma larvata	mammal	II, IV	poaching	2	2010-2011	Asia, Africa
Pan troglodytes	mammal	II, IV	poaching, logging	7	1996-2011	Africa
Pancovia turbinata	Plant	II	poaching	1	2012	Africa
Panthera leo	mammal	II, IV, VI	Poaching, legal hunting	8	1995-2013	Africa
Panthera pardus	mammal	II, IV, VI	poaching, legal hunting	6	1996-2013	Africa
Panthera tigris	mammal	IV, VI	poaching	2	2002-2010	Asia
Papio anubis	mammal	Ia, II	poaching	7	1996-2012	Africa
Papio hamadryas	mammal	II, IV	poaching	2	2000-2009	Africa
Papio ursinus	mammal	II	poaching	1	2013	Africa
Paradoxurus hermaphroditus	mammal	II	poaching	1	2012	Asia
Pardofelis marmorata	mammal	II	poaching	1	2012	Asia
Parinari excelsa	Plant	II	poaching	1	2012	Africa
Pecari tajacu	mammal	III	poaching	1	2000	America
Pedetes capensis	mammal	IV	poaching	1	2000	Africa
Petrea volubilis	Plant	IV	poaching	1	2004	America
Phacochoerus aethiopicus	mammal	IV, VI	poaching, legal hunting	2	2000-2011	Africa
Phacochoerus africanus	mammal	Ia, II, VI	poaching, disease, water scarcity	13	1995-2013	Africa
Philantomba monticola	mammal	II, IV	poaching, livestock grazing, logging	12	2006-2012	Africa

Philatomba maxwelli	mammal	II	poaching	1	2009	Africa
Phyllostylon brasiliensis	Plant	IV	poaching	1	2004	America
Piliocolobus preussi	mammal	II	poaching	2	2006	Africa
Piper tuberculatum	Plant	IV	poaching	1	2004	America
Pisonia macranthocrapa	Plant	IV	poaching	1	2004	America
Plant sp	Plant	Ia, II-VI	poaching	25	1999	Africa, Asia
Poelagus marjorita	mammal	II	poaching	1	2006	Africa
Polyscias fulva	Plant	II	poaching	1	2012	Africa
Potamochoerus larvatus	mammal	Ib, II, IV, VI	poaching, livestock grazing, legal hunting	11	2000-2011	Africa
Potamochoerus porcus	mammal	Ia, II, IV, VI	poaching, water scarcity	10	2006- 2013	Africa
Pouteria sapota	Plant	IV	poaching	1	2004	America
Prionace glauca	Fish	VI	poaching	1	2013	America
Prionailurus bengalensis	mammal	II	poaching	2	2005- 2012	Asia
Prionodon linsang	mammal	II	poaching	2	2010-2011	Asia, Africa
Prionodon pardicolor	mammal	II	poaching	1	2005	Asia
Procolobus pennantii	mammal	II	poaching and habitat loss	1	2011	Africa
Proechimys semispinosus	mammal	III	poaching	1	2000	America
Propithecus diadema	mammal	II, VI	poaching	2	2012	Africa
Propithecus verreauxi	mammal	II, VI	poaching	2	2012	Africa
Propithecus verreauxi coquereli	mammal	II	poaching	1	2003	Africa
Protoxerini spp.	mammal	IV	poaching	1	2006	Africa
Protoxerus stangeri	mammal	IV	poaching	2	2000-2007	Africa
Prunus spp.	Plant	II	poaching	1	2012	Africa
Pseudospondias microcarapa	Plant	II	poaching	1	2012	Africa
Psidium guajava	Plant	IV	poaching	1	2004	America
Psidium sp.	Plant	II	poaching	1	2012	Africa
Psittacine spp	bird	II	poaching	1	2011	America
Pternistis leucoscepus	bird	II	poaching	1	1996	Africa
Python regius	reptile	II	poaching	1	2009	Africa
Python sebae	reptile	II, IV	poaching	4	2006-2009	Africa
Quassia amara	Plant	IV	poaching	1	2004	America
Raphicerus campestris	mammal	II	poaching	1	2013	Africa
Raphicerus sharpei	mammal	IV	poaching	1	2000	Africa
Rat spp	mammal	IV	poaching	1	2007	Africa
Redunca redunca	mammal	Ia, II, IV, VI	Poaching, predation, water scarcity, disease	13	1985- 2013	Africa
Rhinoceros unicornis	mammal	II, IV	poaching	6	2008-2013	Asia
Rhynchocyon spp.	mammal	IV	poaching	1	2006	Africa
Rondent spp	mammal	Ia, II	poaching	9	2009	Asia
Rusa unicolor	mammal	II, IV	poaching	5	2005- 2014	Asia
Saguinus geoffroyi	mammal	III	poaching	1	2000	America
Sapindus saponaria	Plant	IV	poaching	1	2004	America
Sapium spp.	Plant	II	poaching	1	2012	Africa
Schoepfia schreberi	Plant	IV	poaching	1	2004	America
Sciurid spp	mammal	II	poaching	3	1996- 2006	Africa

Sciurus granatensis	mammal	III	poaching	1	2000	America
Senna spectabilis	Plant	II	poaching	1	2012	Africa
Simarouba glauca	Plant	IV	poaching	1	2004	America
Snail spp	Mollusc	II, IV	poaching	3	2006-2007	Africa
Solanum erianthum	Plant	IV	poaching	1	2004	America
Spathodea campanulata	Plant	II	poaching	1	2012	Africa
Sphyrna zygaena	Fish	VI	poaching	1	2013	America
Sterculia apetala	Plant	IV	poaching	1	2004	America
Strombosia scheffleri	Plant	II	poaching	1	2012	Africa
Struthio camelus	bird	II	poaching	3	1995-2009	Africa
Sus scrofa	mammal	Ia, II, IV	poaching	14	2009-2014	Asia, Africa
Swietenia sp.	Plant	II	poaching	2	2014	Asia
Sylvicapra grimmia	mammal	Ia, II, IV, VI	poaching, livestock grazing, disease, water scarcity	13	2000-2013	Africa
Syncerus caffer	mammal	Iab, II, IV, VI	poaching, livestock grazing, disease, water scarcity	27	1995- 2013	Africa
Syzygium sp.	Plant	II	poaching	2	2014	Asia
Tabernaemontana spp.	Plant	II	poaching	1	2012	Africa
Tamandua mexicana	mammal	III	poaching	1	2000	America
Taurotragus derbianus	mammal	II	poaching and disease	5	2010-2012	Africa
Taurotragus oryx	mammal	II, IV, VI	poaching, livestock grazing, water scarcity	9	1995- 2013	Africa
Teclea nobilis	Plant	II	poaching	1	2012	Africa
Tectona grandis	Plant	II	poaching	2	2014	Asia
Tenrec ecaudatus	mammal	II	poaching	1	2003	Africa
Thouinidium decandrum	Plant	IV	poaching	1	2004	America
Thryonomys swinderianus	mammal	II, IV	poaching	7	1996- 2009	Africa
Thunnus albacares	Fish	II	poaching	1	2014	America
Trachypithecus phayrei	mammal	II, IV	poaching	2	2010-2011	Asia, Africa
Trachypithecus pileatus	mammal	II	poaching	1	2005	Asia
Tragelaphus angasii	mammal	II	poaching	1	2013	Africa
Tragelaphus eurycerus	mammal	II	poaching, logging	2	2006-2011	Africa
Tragelaphus imberbis	mammal	II	poaching	1	2006	Africa
Tragelaphus scriptus	mammal	Ia, II, IV, VI	poaching, disease, livestock grazing, water scarcity	18	1995-2013	Africa
Tragelaphus spekii	mammal	II, IV	poaching	2	2000- 2006	Africa
Tragelaphus spp	mammal	II	poaching	2	2006	Africa
Tragelaphus strepsiceros	mammal	II, IV, VI	poaching, legal hunting, water scarcity	8	2000-2013	Africa
Tragulus kanchil	mammal	II	poaching	1	2012	Asia
Trema micrantha	Plant	IV	poaching	1	2004	America
Trema spp.	Plant	II	poaching	1	2012	Africa
Trichospermum mexicanum	Plant	IV	poaching	1	2004	America
Trionyx triunguis	reptile	II	poaching	2	2006	Africa
Triplaris melaenodendron	Plant	IV	poaching	1	2004	America
Trophis racemosa	Plant	IV	poaching	1	2004	America
Urera caracassana	Plant	IV	poaching	1	2004	America
Ursus thibetanus	mammal	II, IV, VI	poaching	4	2010-2012	Asia, Africa
Uvariopsis congensis	Plant	II	poaching	1	2012	Africa

Vanga curvirostris	bird	II	poaching	1	2003	Africa
Varanus nilocticus	reptile	II, IV	poaching	3	2006-2007	Africa
Varanus spp	reptile	II	poaching	1	1995	Africa
Viverra megaspila	mammal	II	poaching	1	2012	Asia
Viverra zibetha	mammal	II	poaching	1	2012	Asia
Ximenia americana	Plant	IV	poaching	1	2004	America
Zanthoxylum belizense	Plant	IV	poaching	1	2004	America
Zizyphus guatemalensis	Plant	IV	poaching	1	2004	America

2. 9. Appendices

2.9.1. List of Additional Sources of data for Species body mass: MAMMALS

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2.9.2. List of Additional Sources of data for Species body mass: REPTILES

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CHAPTER 3 - Correlates of spatial variation in illegal activities in the Serengeti ecosystem

A morning view of the Serengeti with balloon tourism in the central grass plains

3.1. Abstract

Pressures from illegal activities continue to threaten the Serengeti ecosystem in East Africa. I found that different classes of illegal activities (animal poaching signs, wire snares, tree harvesting and illegal cattle grazing) occurred in different areas of the ecosystem. Animal poaching and wire snaring were more widely distributed than previously reported, while plant extractions and cattle incursions were mostly clustered within few kilometres of PA borders. Fine-scale habitat features including water pools, animal tracks and paired trees predicted animal poaching and wire snares. At the landscape scale, illegal activities were associated with high net primary productivity, areas that are close to rivers and areas of intermediate distance from villages where poachers may live, suggesting that poaching decisions are made at varying scales based on the local and landscape features within the ecosystem. The presence of wide-spread illegal activities, even in the remotest areas of the PAs, suggests that poachers operate with limited concern for detection. These results are useful for improving anti-poaching activities to reducing threats in protected areas.

Keywords: Animal poaching, Bayesian hierarchical modelling, environmental covariates, illegal activities, Serengeti ecosystem, wire snares

3.2. Introduction

Although protected areas (PAs) are designed to prevent biodiversity loss, threats from illegal activities are rising for many species, leading to widespread population declines and species loss (Butchart et al., 2010; Laurance et al., 2012). Law enforcement is an important tool for controlling illegal activities within PAs (Keane et al., 2008; Rauset et al., 2016) but its effectiveness can be limited, especially in PAs within developing countries where conservation budgets are small (Tranquilli et al., 2012) and where impact of illegal activities on species is increasingly apparent (Harrison, 2011; Ripple et al., 2015a). Consequently, new conservation strategies such as establishing ecological targets and their performance metrics for PAs are needed to improve effectiveness of protected areas (Watson. et al., 2016). Understanding the local contexts under which illegal activities occur and their trends across time and space may enable us to better understand how perpetrators of illegal activities operate and thus help inform conservation decisions to prevent them (Critchlow et al., 2015; Watson et al., 2013). Here I examine the spatial patterns and drivers of illegal activities in four protected areas within the Serengeti ecosystem, assessing how poaching decisions at various scales may drive the distribution of illegal activities across the ecosystem with the aim of improving efficiency of ranger patrols.

Illegal activities such as land conversion, firewood harvesting and bushmeat poaching can be common even within protected areas, reducing their ability to preserve biodiversity (Laurance et al., 2012). Across Africa, Craigie et al. (2010) report around 50% declines in the abundances of wild mammal populations in 78 protected areas, caused by a combination of legal and illegal activities. Together, these studies provide a broad regional picture of the extent and impacts of illegal activities in protected areas, but there is limited information on how illegal activities are spatially distributed within individual protected areas, information that may enable rangers to improve their detection and prevention activities. For example, recent work has shown that knowing the poaching history of an area can increase detection of illegal activities by over 250% in Queen Elizabeth National Park, Uganda (Critchlow et al., 2016), and it could reduce anti-poaching budgets by at least 60% in the Virunga Conservation landscape in Central Africa (Plumptre et al., 2014). Furthermore, studies elsewhere suggest that poaching rates correlates with animal abundance, proximity to water sources, roads and reserve borders (Kimanzi et al., 2015; Moore et al., 2017; Wato et al., 2006; Watson et al., 2013) but the importance of these variables vary widely between protected areas in different regions. For example, in East Africa, illegal activities occur close to ranger posts in Nyungwe National Park, Rwanda (Moore et al. 2017) while they occur away from ranger posts in Queen Elizabeth National Park in Uganda (Critchlow et al. 2015). On another hand , proximity to ranger posts influences illegal activity occurrence in Khao Yai National Park in Thailand (Jenks et al., 2012). This suggests that PA-specific ecological information may be necessary when tackling illegal activities.

The Serengeti ecosystem in East Africa holds one of the largest remaining assemblages of wild ungulates on earth, but continues to face pressures from illegal activities. Illegal hunting limits populations of several species including African buffalo (Metzger et al., 2010), giraffe (Strauss et al., 2015), impala and gazelles (Setsaas et al., 2007), and black rhino and elephant (Metzger et al., 2010) in this ecosystem. To prevent poaching, existing conservation strategies need to be improved using information of where and at what rates poaching occurs in the protected areas (Hofer et al., 2000). Previous work in the Serengeti ecosystem shows that poachers hunt bushmeat both to meet their family's protein requirements and to sell for cash benefit, and they are more likely to poach when law enforcement is perceived to be weak (Fischer et al., 2014; Rentsch & Damon, 2013). These studies can be useful, however, they do not inform us about where poaching occurs inside PAs nor do they provide information about other threats, such as illegal livestock grazing and tree cutting, for the rangers to be able to target and prevent the full range of threats impacting species in the Serengeti ecosystem.

Essentially, the location of poaching and other illegal activities is likely to be the product of several decisions made by poachers, which can be structured at three levels: the general area to visit (landscape scale), precisely which sites are chosen (fine scale) and where within that area to operate (local scale, Figure 3.1). Each of these decisions will likely be made with the intention of maximising the success of the activity (e.g. capture rates of animals, or extraction of suitable trees) while simultaneously minimising the chances of being caught by the park guards. Exactly how this balance is resolved depends on the perceived costs and benefits in each area, for each activity. Here, I detail the range of illegal activities in the greater Serengeti ecosystem, and build spatially explicit models to identify correlates of activities at three spatial scales. I predicted that (1) non-commercial (i.e. wood cutting, thatch harvesting and livestock grazing) and commercial (animal poaching-related signs) illegal activities will show differing spatial distribution pattern. I expected non-commercial activities to be restricted to the peripheries of the PAs (i.e., constrained by convenience and feasibility), while animal poaching will occur anywhere in the ecosystem where the net commercial profit is maximised, i.e. taking into account perceived risks of capture, as well as financial gain (Hofer et al., 2000). (2) Animal poaching will be highest in sites with the highest potential for catch success, and consequently should increase in locations with high abundance of target animals, animal forage (food), water availability and high tree cover (where snares and poachers may be hidden) at the local scales, (3) poaching levels should increase near roads, rivers and villages, decline close to ranger stations and increase in high altitude areas (above sea level) owing to the perceived safety for poachers.

3.3. Materials and methods

3.3.1. Study area

The Serengeti-Mara Ecosystem (SE) in East Africa encompasses five contiguous protected areas; the Serengeti National Park, (strict PA), Ngorongoro Conservation Area (multiple use PA) and three game reserves; Maswa, Ikorongo and Grumeti (within which tourist hunting is the only permitted activity) in Tanzania and the Masai Mara reserve in Kenya. Most of the ecosystem in Tanzania is covered by extensive plains of semi-arid savanna with smaller areas of riverine forest. Mixed *Acacia* and *Commiphora* woodlands extend over much of the central and northern regions with some occasional large open grasslands (Reed et al., 2009). The South and Eastern areas receive an average of 500 mm annual rainfall, increasing to >1200 mm in the north and west peaking mainly during wet season between November and May (Sinclair & Arcese, 1995). Rainfall is among the factors determining vegetation characteristics in this ecosystem, the animal migration, and consequently poaching (ibid). I conducted this study in the Serengeti National Park and Ikorongo, Maswa and Grumeti Game Reserves, referred to here as Serengeti ecosystem.

3.3.2. Field surveys

To map the spatial distribution of illegal activities in the protected areas, we walked standard transects spaced at least 2 km apart within the existing PA management zones across the ecosystem. We started each survey at least 80 m from the road network and walked rectangular transects measuring at least 8 km (except for one transect of 4 km) across various habitats within each protected area. Along each transect, I together with two experienced national park rangers who received three days specific training searched an 80 m-wide strip, walking 15 m apart, and identifying all signs of illegal activities (wire snares and snare prints, pit-traps, poached animal carcasses, tree cuts, poacher camps, human footprints or other signs such as shoes, wrist watch, bicycle, cloth, etc. and livestock incursion). On finding an event, we recorded the precise location using a hand-held GPS (Garmin eTrex 20). The detection of active snares, or signs of recent illegal activity is relatively easy in the open habitats of Serengeti, (Hofer et al., 2000), I therefore assumed that within the 80m strip of transect traversed by three field personnel all illegal activities present were detected. We assigned each sign to one of four broad groups of illegal activities (animal poaching, tree cutting, livestock grazing and other) for further analysis. For each carcass encountered, we assessed signs indicating poaching following standard protocols developed in this ecosystem (Mduma et al., 1999; Sinclair, 1995a). For freshly killed animal carcass, we inspected the body for signs of injuries caused by humans, such as knife cuts on the neck, spine or flank, or wire snares on the leg or neck. We also recorded poaching when we found fresh body parts, such as heads, lower parts of legs and stomachs or any animal other remains which provided evidence of poacher activity, as these are often left by poachers in the field on poaching success (Mduma et al., 1999). We also recorded poaching when skeletons up to three months old (for larger animals) were found with signs such as wire snares and axe/knife cuts.

To understand the fine scale habitat characteristics associated with the exact locations of illegal activities, we measured habitat characteristics at the point where evidence of illegal activities was found. As a control comparison, we also measured similar characteristics at a location (dummy) 50 m away from the edge of the transect in a perpendicular direction and a randomly assigned direction. At both real and dummy locations, we measured ground cover and herb height within a 1x1 m plot and number of animal trails, water pools, paired and single-growing trees (i.e. potentially used for suspending wire snares by poachers) within a 20m radius area. Fieldwork was undertaken between July and November 2015 and 2016. To assess the differences between years I repeated a subset of the 2015 transects (n = 6) in 2016. Overall, we surveyed 56 transects in 2015 and 32 transects in 2016 covering 920.25 km across the ecosystem.

3.3.3. Extraction of covariates of illegal activity occurrence

I collated information on environmental covariates associated with the occurrence of illegal activities in Serengeti from various databases. Firstly, I used the online MODIS (product-MOD17A3) and ASTER (https://reverb.echo.nasa.gov/reverb/users/new) databases to obtain net primary productivity (NPP) and digital elevation model (DEM), respectively for the entire Serengeti ecosystem. For NPP, I collated biomass productivity tiles for each of 15 years (2000-2014) and used the mean for further analysis. I chose to use mean NPP rather than the year specific values because poachers tend to return to their own patches over long periods (Chritchlow et al. 2015) and the long-term average conditions are likely more important than differences between years, moreover, data were not available beyond 2014. NPP was used as proxy for food abundance because it is a strong predictor of animal densities in African ungulates, and elephant in protected areas including the Serengeti (Duffy & Pettorelli, 2012; Pettorelli et al., 2009). I accessed the Serengeti GIS database (accessed from Tanzania National Parks (TANAPA) and the Frankfurt Zoological Society-FZS office in Serengeti) and obtained layers for the rivers and permanent water bodies, park roads, ranger stations and villages surrounding the PAs. Roads data were collated by FZS by driving all roads in the ecosystem with a GPS. Location of ranger stations and villages data were collated by the protected areas by conducting surveys. Rivers data were extracted from databases collated by FZS and Tanzania Wildlife Research Institute using existing topographic map and FAO Africover map. Because wildebeest and buffalo are two of the commonest targets for poachers in the area (Campbell & Hofer, 1995), I extracted their densities from the Tanzania Wildlife Research Institute (TAWIRI) systematic aerial census data collected in 2010 and 2014. Trees are important to poachers both because they provide cover from rangers, and because they provide physical support for wire snares. Consequently I extracted information on tree cover from Serengeti habitat map provided by Reed et al. (2009) because this is considered more accurate representation than maps created at a global scale (Reed et al. 2009).

To understand the local scale habitat characteristics, I extracted each environmental covariate at the level of transect subunits. A subunit is defined as a section of transect that falls within a $500m^2$ grid cell (Figure 3.1). I calculated mean NPP, altitude, tree cover, and animal density within each 500m grid cell. Next, I measured distance of each grid cell to the nearest road, river, ranger station and village (where poachers may live) using R (R core Team, 2016). Finally, to understand the landscape scale habitat characteristics, I collated the environmental covariates as above but aggregating all data across the transect and computing the mean tree cover, animal density, NPP, the total number of snares found, etc. for each transect. To test the influence of the different ranger management zones within the protected areas, I used the locations of ranger stations and administrative units (i.e. PA zones) to identify the nearest ranger station with jurisdiction in the respective management unit, and the distance to that ranger station.

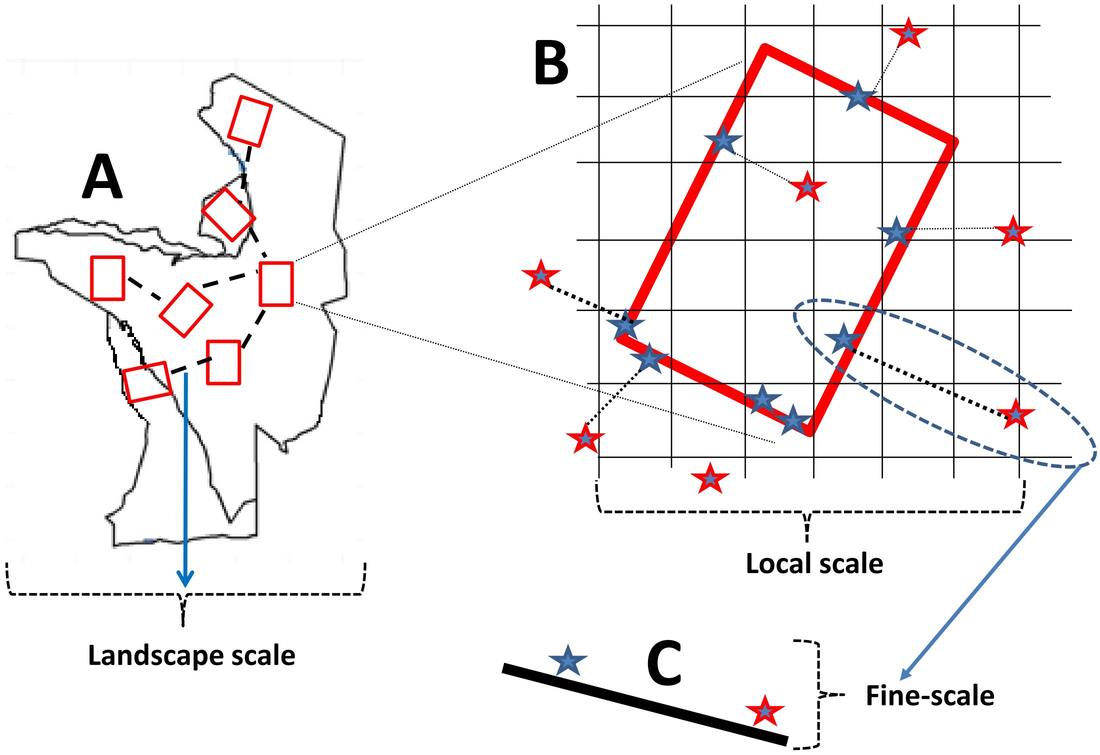

Figure 3.1. Schematic illustration of the three scales of poaching decisions (A = Landscape, B = local (within transect), C= fine-scale). At the landscape level, a poacher will decide on which site of the ecosystem to go for hunting (e.g. East, West, Central etc.). At C, poachers will decide where to locate a snare trap (at either blue or red star), and lastly, a poacher will decide how far to distribute all the snares from the initial point, creating a trapping pattern like B. This hierarchy of decisions was used to structure the analysis to understand the drivers responsible at each decision level. At A, data were compared between transects across the landscape. At B, comparison was made between different grid cells (i.e. subunits) where a transect (i.e. red rectangular block) crossed while at fine-scale C, paired data were compared to test which location was the actual location of an illegal activity.

3.3.4. Statistical analysis

I analysed these data based on the drivers of poaching at each of the three levels of poaching decisions: fine-scale, local and landscape scale (Figure 3.1). Firstly, to disentangle what drives poaching decisions at the exact locations where illegal activities occurred (fine-scale) based on independently recorded paired- real and dummy data points along the transect, I built a generalised linear mixed model (GLMM) with a binomial error term and logit link (within the 'glmer' function implemented under R-package 'lme4' Bolker et al. (2009) to predict whether each point was an actual location of illegal activity, or its paired absence point. I built models for each illegal activity type including eight covariates (ground cover, herb height, number of trees, animal tracks, water pools, bushes and tree canopy height) and point ID identifier that acted to pair real and dummy locations of illegal activities as random effects to account for potential spatial autocorrelation. Next, I used a backwards stepwise removal of non-significant term (with Chi test) to identify covariates significantly associated with the occurrence of illegal activities. I estimated confidence intervals for the variables showing significant effect on illegal activities using the Wald method (Bolker et al., 2009). In each model, I examined the effect of each covariate on the probability of occurrence of poaching.

Secondly, to understand what environmental covariates influence poaching decisions at a local scale (i.e. within transect level), I fitted spatially explicit Bayesian models on each class of illegal activity using data collated from the 500 m cells. To prepare data for the local level analysis, I first computed the counts of each type of illegal activity within each 500m grid cell and centred and scaled the covariates to the transect mean. To account for spatial autocorrelation, I identified cells within 30km of one another and used the resulting neighbourhood to fit an intrinsic Conditional Autoregressive Models using Integrated Nested Laplace Approximation (INLA: (Rue et al., 2009) in the R-INLA package (Rue et al., 2013). I fitted a Poisson GLMM (with prior = loggamma) with log transect subunit length as an offset to account for variability in the lengths of transects walked within each 500m cell. For each model covering each class of illegal activity, I included ten covariates as linear fixed factors: distance to rivers, roads, villages, ranger stations and wildebeest and buffalo densities, tree cover, NPP, altitude and sampling year.

Thirdly, to test the hypothesis about drivers of poaching decisions at the landscape scale, I aggregated data from the subunits for each of transect surveyed (N =88). In addition to the fixed factors investigated in the local scale analysis, I added tree cover as a quadratic term and management units as fixed factors to the model in the landscape scale analysis to examine their influences on the observed patterns of illegal activities within individual protected areas. I expected intermediate tree cover to be associated with higher snaring than low or high tree cover areas, because low tree cover does not provide shade for animals or appropriate cover for poachers while high tree cover tends to be avoided by wild ungulates perhaps due to perceived predation risks and are therefore not suitable for animal snaring. As before, I used INLA to fit Poison GLMM models with a Continuous Autoregressive model of spatial autocorrelation as above. In the latter two analyses, the effect of covariates was considered significant if the model confidence intervals were not overlapping zero. Further, at each scale of analysis, I examined each category of illegal activity signs separately to understand variation in the drivers of these activities.

3.5. Results

3.5.1. Illegal activity patterns at fine-scale

At the fine-scale the occurrence of poached animal carcass was associated with more water pools, high ground cover and more animal trails (Figure 3.2,Table 3.1).

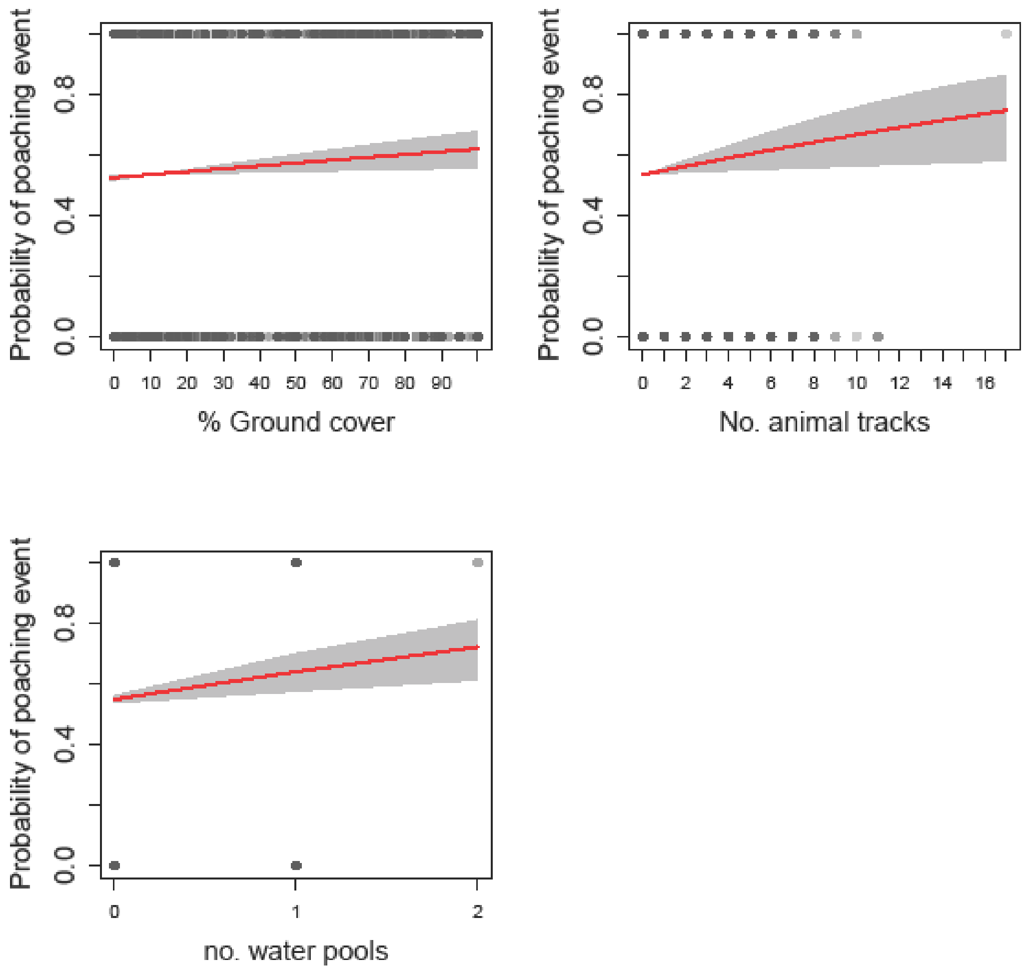

Figure 3.2. The influence of fine-scale habitat characteristics on the probability of animal poaching in the Serengeti Ecosystem. The presence of water pools, animal tracks and high ground cover were the strongest predictors of animal poached in the area, suggesting that ranger patrols targeting sites with these covariates may improve detection of illegal activities. Darker grey points indicate more observations in the covariate.

Table 3.1. Results for the final best GLMM models for fine-scale analysis indicating influence of various covariates on the probability of illegal activities (animal poaching, wire snaring and plant extraction) in the Serengeti Ecosystem. Bold text indicates significance of the variable in the model.

Illegal activity/Covariate name	Mean effect	DF	$\chi 2$	p-value
(a) Animal poaching				
Ground cover (%)	0.134 ± 0.045	1	2.95	0.0036
Number of animal tracks	0.111 ± 0.046	1	2.41	0.0167
Number of water pools	0.0167 ± 0.047	1	3.52	0.0004
(b) Wire snaring				
Number of paired trees	0.558 ± 0.139	1	4	0.00001
(c) Plant extraction				
Herb height (cm)	**-0.505 ± 0.224**	**1**	**-2.25**	**0.024**
Ground cover (%)	0.219 ± 0.0189	1	1.16	0.248
Number of trees	0.138 ± 0.165	1	-0.84	0.403
(d) Cow grazing				
Ground cover (%)	-0.0352±0.165	1	-0.214	0.87
Number of paired trees	0.032 ± 0.166	1	0.194	0.846
(e) Other signs				
Herb height (cm)	-0.312 ± 0.237	1	-1.314	0.189

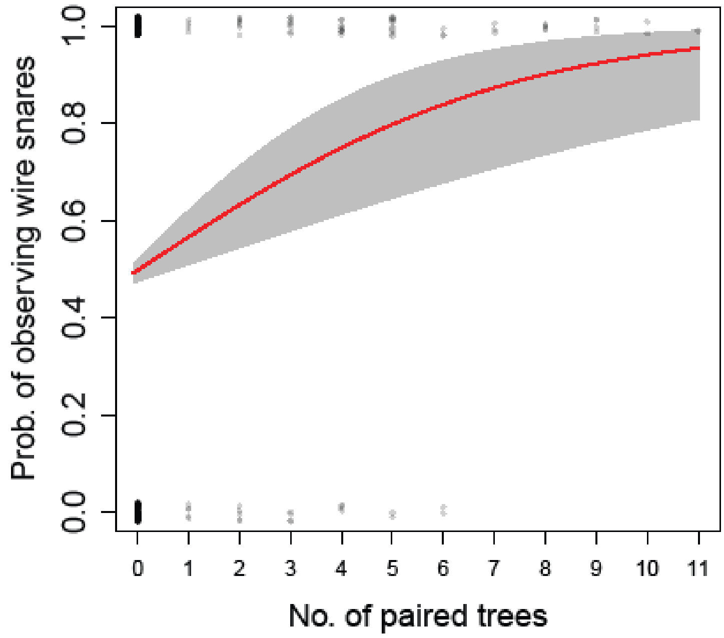

Figure 3.3. The influence of the number of paired-growing trees on the probability of occurrence of wire snares in the Serengeti ecosystem. There is clear evidence that poachers most often target paired trees for setting wire snares to catch animals. Anti-poaching teams may need to target treed areas to recovers wire snares.

Wire snare occurrence was strongly associated with the number of paired growing trees in the locality (Figure 3.3). The probability of plant extraction increased in areas with low tree cover, shorter herbs and slightly high ground cover (Figure 3.4, Table 3.1). No covariate was important in the observed occurrence patterns of cattle incursion or other signs of illegal activities (Table 3.1).

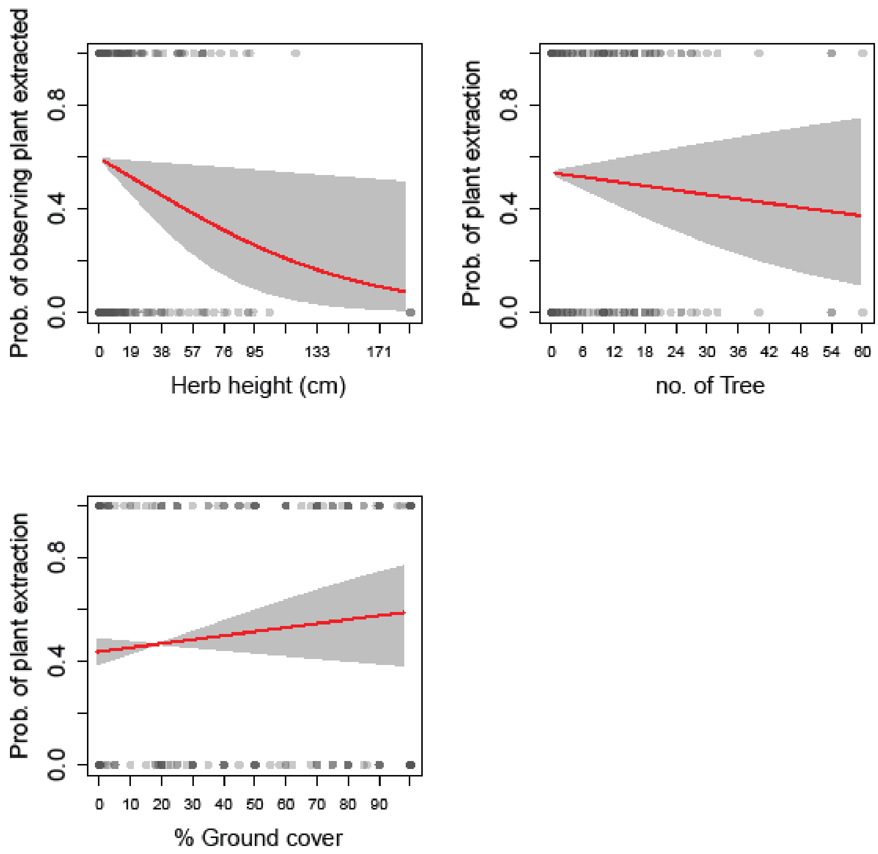

Figure 3.4. The influence of fine-scale habitat characteristics on the probability of plant extraction in the Serengeti Ecosystem. Plant extraction occurred in areas with relatively short grass herbs and sparse trees, and with slightly high ground cover suggesting that poachers may be selecting areas with ensured maximum visibility to avoid being caught by patrol rangers.

3.5.2. Illegal activity patterns at local scale

At the local scale, poached carcass sightings were highest in areas with relatively high wildebeest density and at high altitude and decreased away from rivers (Figure 3.5,Table 3.2).

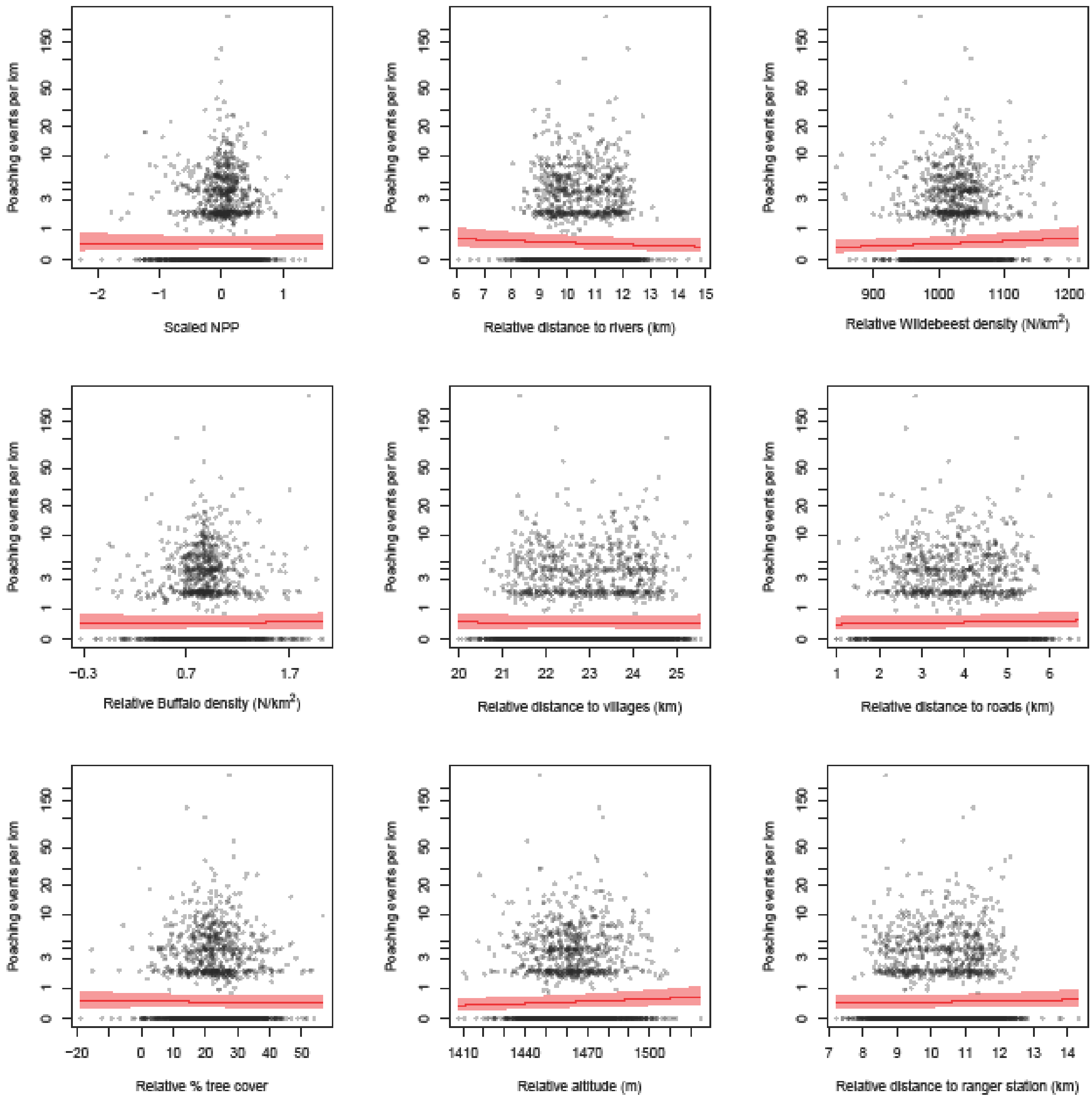

Figure 3.5. Influence of environmental covariates on the abundance of poached animals at the local scale. Illegal activities slightly increased only in areas with high altitude and wildebeest density and occurred closer to rivers.

Table 3.2. Results from local scale analysis with INLA model indicating influence of various covariates on the abundance of poaching at a local scale. There was significant (**bold text**) effect of high wildebeest density, high altitude and shorter distance from rivers.

Model covariates(**CARCASS**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-0.869±0.220	-1.310	-0.866	-0.444
NPP	-0.046±0.098	-0.236	-0.046	0.148
Distance to rivers	**-1.133±0.424**	**-1.966**	**-1.133**	**-0.303**
Buffalo density	-0.012±0.129	-0.267	-0.012	0.241
Wildebeest density	**0.999±0.484**	**0.051**	**0.999**	**1.951**
Woody cover	-0.030±0.065	-0.159	-0.030	0.098
Altitude	**0.981±0.417**	**0.163**	**0.981**	**1.800**
Village distance	-0.394±0.633	-1.637	-0.394	0.847
Distance to park roads	0.122±0.173	-0.217	0.122	0.461
Distance to ranger station	0.091±0.188	-0.281	0.092	0.459
Sampling year (2016)	0.139±0.353	-0.570	0.144	0.820

TableS3.1. Results from local scale analysis with INLA model indicating influence of various covariates on the abundance of wire snares at a local scale. There was no covariate indicating significant effect on the abundance of wire snares.

Model Covariates(**SNARES**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-4.335 ± 0.404	-5.199	-4.310	-3.615
NPP	-0.062 ± 0.336	-0.708	-0.067	0.613
Distance to rivers	2.579 ± 1.406	-0.152	2.568	5.373
Buffalo density	0.054 ± 0.466	-0.864	0.055	0.966
Wildebeest density	0.800 ± 1.720	-2.559	0.793	4.198
Woody cover	0.309 ± 0.251	-0.180	0.307	0.807
Altitude	-0.578 ± 1.450	-3.448	-0.572	2.253
Village distance	2.547 ± 2.153	-1.668	2.541	6.789
Distance to park roads	0.920 ± 0.592	-0.230	0.916	2.094
Distance to ranger station	-0.021 ± 0.583	-1.196	-0.011	1.099
Sampling year (2016)	-0.470 ± 0.338	-1.147	-0.466	0.183

No covariate was important on the number of wire snares recorded at this scale (TableS3.1). Both illegal cattle grazing, and tree extraction occurred closer to villages but further away from the ranger stations (Table 3.3, Table 3.4). Sightings of other signs of illegal activities were highest where wildebeest density was low and varied between the sampling years (Table 3.5).

Table 3.3. Results from local scale analysis with INLA model indicating influence of various covariates on the abundance of illegal cattle grazing at a local scale. There was significant (**bold text**) effect of distance to the villages, away from the ranger station and between the sampling periods.

Model Covariates(**COW**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-6.756 ± 1.317	-9.922	-6.556	-4.738
NPP	0.595 ± 0.436	-0.232	0.585	1.482
Distance to rivers	2.132 ± 1.668	-1.106	2.119	5.446
Buffalo density	-1.084 ± 0.747	-2.642	-1.052	0.296
Wildebeest density	-1.078 ± 3.125	-7.284	-1.056	5.005
Woody cover	0.374 ± 0.275	-0.155	0.370	0.926
Altitude	-0.608 ± 1.388	-3.364	-0.598	2.096
Village distance	**-6.705 ± 3.127**	**-12.994**	**-6.655**	**-0.699**
Distance to park roads	0.507 ± 0.931	-1.320	0.507	2.336
Distance to ranger st.	**2.017 ± 0.908**	**0.259**	**2.007**	**3.830**
Sampling year (2016)	**-3.507 ± 1.510**	**-7.020**	**-3.319**	**-1.078**

Table 3.4. Results from local scale analysis with INLA model indicating influence of various covariates on the abundance of tree cutting at a local scale. Illegal plant harvesting occurred closer to the villages and differed significantly (**bold text**) between the sampling periods.

Model Covariates (**TREE)**	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-5.826 ± 0.915	-7.933	-5.716	-4.346
NPP	0.176 ± 0.391	-0.567	0.167	0.969
Distance to rivers	-0.935 ± 1.871	-4.666	-0.916	2.689
Buffalo density	0.485 ± 0.776	-1.038	0.484	2.008
Wildebeest density	-3.555 ± 2.622	-8.774	-3.533	1.538
Woody cover	0.017 ± 0.270	-0.511	0.017	0.548
Altitude	0.551 ± 1.606	-2.640	0.562	3.675
Village distance	**-11.874 ± 3.411**	**-18.787**	**-11.799**	**-5.375**
Distance to park roads	-0.248 ± 0.950	-2.130	-0.243	1.603
Distance to ranger station	0.190 ± 0.956	-1.708	0.196	2.052
Sampling year (2016)	**-2.775 ± 1.164**	**-5.387**	**-2.665**	**-0.792**

Table 3.5. Results from local scale analysis with INLA model indicating influence (**bold text**) of wildebeest density and sampling year on Other signs of illegal activities at a local scale. Illegal plant harvesting occurred closer to the villages and differed significantly between the sampling periods.

Model Covariates(**OTHER**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-3.820 ± 0.429	-4.775	-3.782	-3.085
NPP	-0.163 ± 0.398	-0.914	-0.175	0.655
Distance to rivers	2.589 ± 1.689	-0.693	2.577	5.942
Buffalo density	-0.756 ± 0.561	-1.859	-0.756	0.347
Wildebeest density	**-4.962 ± 2.243**	**-9.432**	**-4.9433**	**-0.608**
Woody cover	0.313 ± 0.306	-0.286	0.313	0.916
Altitude	-0.497 ± 1.702	-3.866	-0.489	2.824
Village distance	-0.155 ± 2.648	-5.367	-0.152	5.039
Distance to park roads	0.625 ± 0.708	-0.759	0.623	2.022
Distance to ranger station	0.084 ± 0.665	-1.266	0.099	1.352
Sampling year (2016)	**-1.535 ± 0.451**	**-2.477**	**-1.516**	**-0.700**

3.5.3. Illegal activity distribution and drivers at landscape scale

Animal poaching was the most common illegal activity and was distributed widely across the ecosystem, though levels differed between protected areas (Figure 3.6). Illegal activity types were correlated with different environmental covariates. The abundance of poached animal carcasses was highest in areas with high NPP, at lower altitude closer to rivers and varied significantly between the sampling period (Figure 3.7, Table 3.6). Overall, I found strong evidence of impacts of PA management units on the observed patterns of animal poaching across the landscape, where poaching levels differed greatly between ranger zones (Figure 3.8). Wire snaring was mostly associated with high NPP and tended to occur at low altitude and in areas with low abundant wildebeest. However, snaring was strongly associated with some ranger management zones (Table 3.7). Like in the local scale, illegal cow grazing concentrated closer to villages and were distant away from rivers, at low altitude and varied between different ranger management zones (Table 3.8). Plant extraction correlated negatively with wildebeest abundance, but other signs tended to concentrate in areas with high NPP and low wildebeest abundance and were away from ranger stations (Table 3.9,Table 3.10).

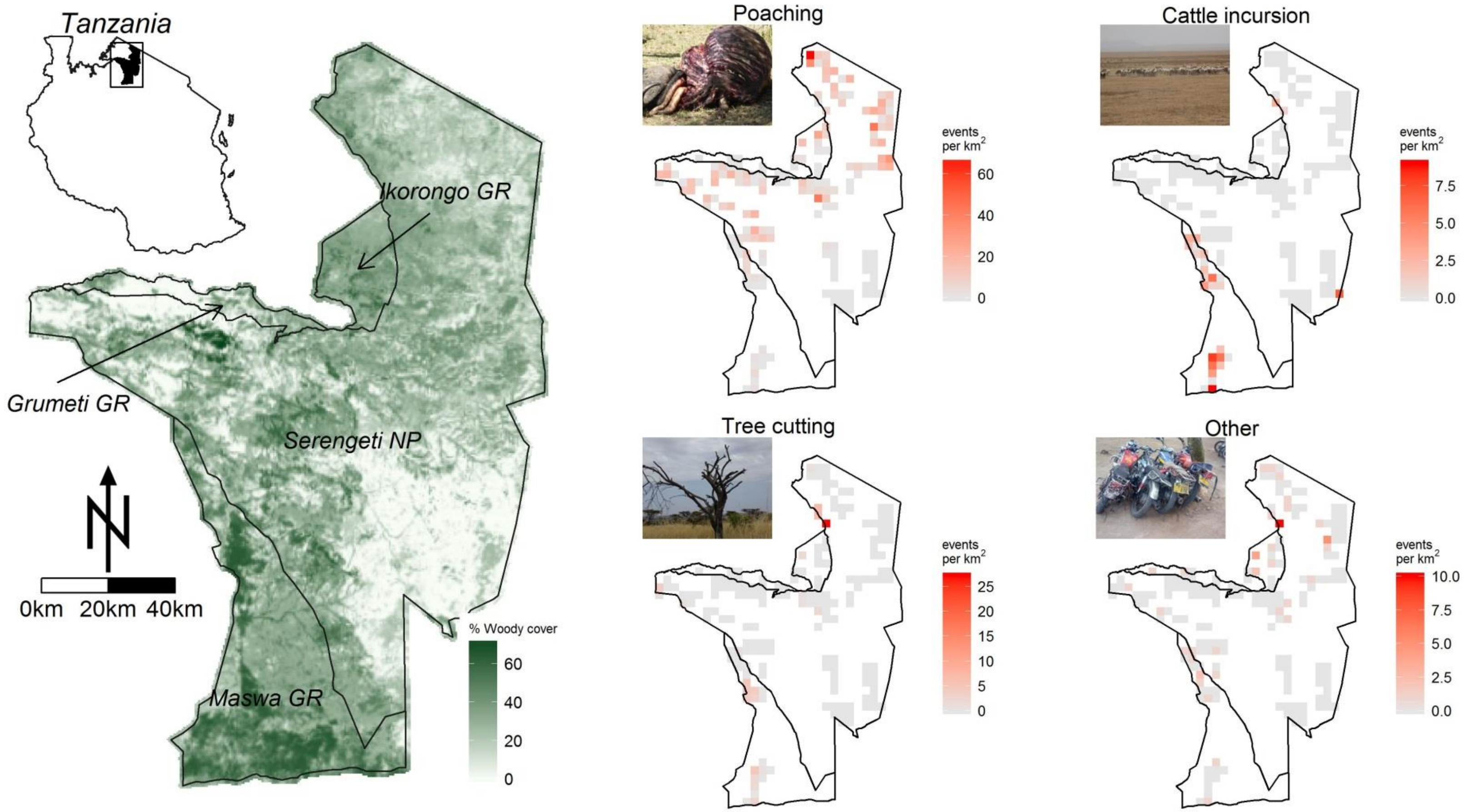

Figure 3.6. Location of the Serengeti Ecosystem (Serengeti National Park and Grumeti, Ikorongo and Maswa Game Reserves) in Tanzania and the spatial distribution of illegal activities: animal poaching, cattle incursion, tree cutting and other signs of illegal activities such as motorcycle tracks, poacher camps etc. The blocks in the maps show where field surveys were conducted and the locations of illegal activities during the two years of fieldwork. Grey indicates transect location where there was no record of an illegal event and white indicates areas without any transects.

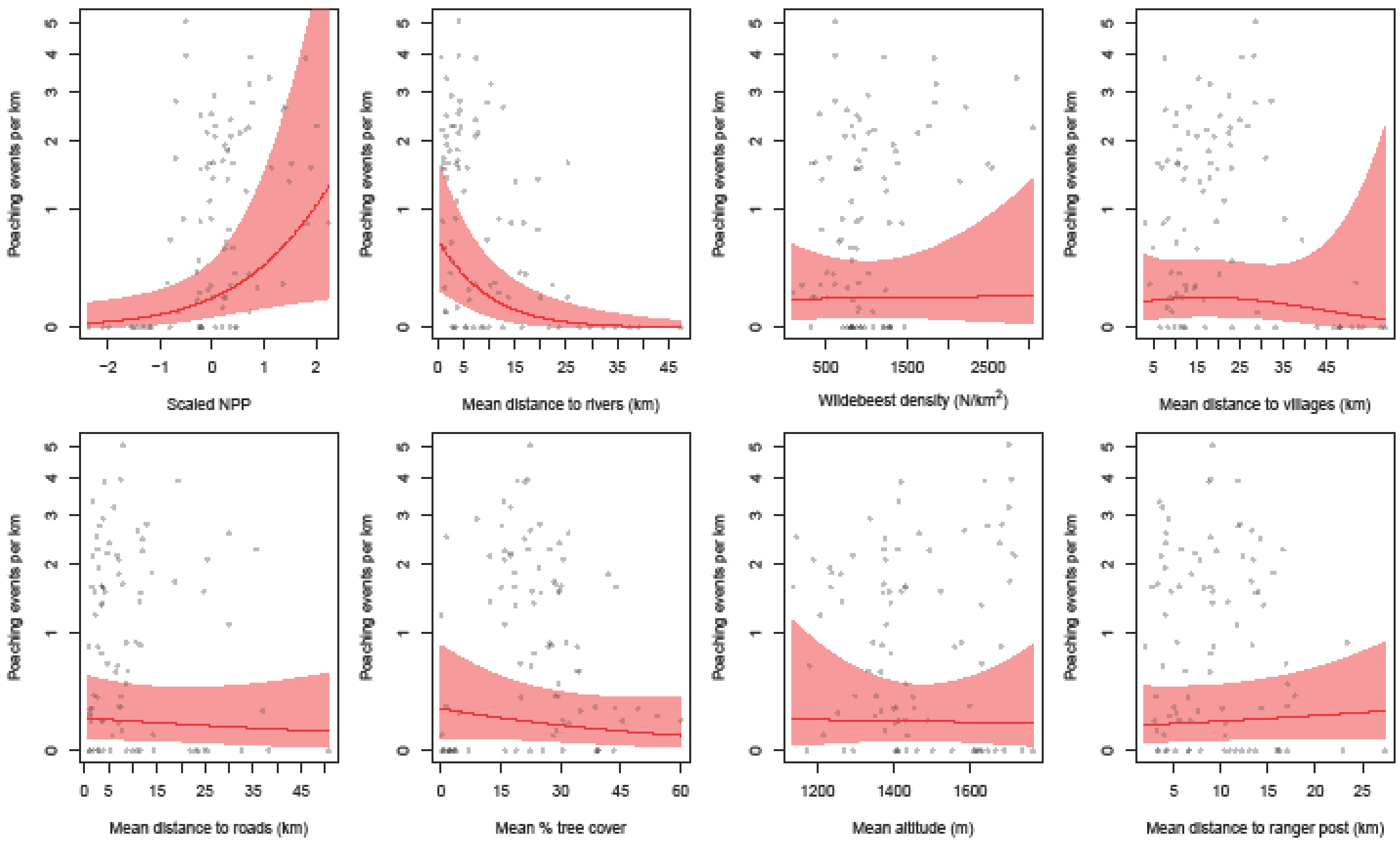

Figure 3.7. The influence of environmental covariates on the patterns of animal poaching at the landscape scale in the Serengeti Ecosystem. Animal poaching (carcass abundance) was associated with high NPP and lower altitude and occurred mostly closer to rivers and park roads. There was evidence for the poaching peaking at locations 25 km away from the villages of poacher residence.

Table 3.6. Results from the landscape scale analysis indicating significant (**bold text**) influence of various covariates on the abundance of animal poaching in the Serengeti ecosystem. Animal poaching increased in areas with high NPP, closer to rivers, across the management zones and between the sampling period.

Model Covariates (**CARCASS**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-3.412 ± 0.812	-5.097	-3.381	-1.904
NPP	**0.861 ± 0.367**	**0.159**	**0.854**	**1.607**
Distance to rivers	**-1.301 ± 0.366**	**-2.061**	**-1.288**	**-0.617**
Wildebeest density	0.019 ± 0.240	-0.460	0.021	0.488
Buffalo density	-0.011 ± 0.125	-0.255	-0.012	0.237
Woody cover	-0.098 ± 0.221	-0.533	-0.099	0.340
Woody cover (quadr)	-0.195 ± 0.186	-0.561	-0.195	0.172
Altitude	-0.040 ± 0.379	-0.803	-0.034	0.693
Village distance	-0.135 ± 0.398	-0.922	-0.135	0.650
Village distance(quadr)	-0.182 ± 0.292	-0.759	-0.181	0.393
Distance to park roads	-0.124 ± 0.176	-0.478	-0.122	0.215
Distance to ranger st.	0.106 ± 0.147	-0.183	0.105	0.398
Sampling year (2016)	**1.197 ± 0.313**	**0.593**	**1.192**	**1.827**
Ranger zone 2	1.467 ± 0.842	-0.161	1.456	3.154
Ranger zone 3	**2.908 ± 0.985**	**1.029**	**2.887**	**4.911**
Ranger zone 4	**1.718 ± 0.800**	**0.194**	**1.700**	**3.346**
Ranger zone 5	**3.872 ± 1.152**	**1.687**	**3.842**	**6.226**
Ranger zone 6	2.339 ± 1.228	-0.025	2.319	4.818
Ranger zone 7	1.409 ± 0.954	-0.444	1.399	3.313
Ranger zone 8	**1.452 ± 0.739**	**0.044**	**1.435**	**2.955**

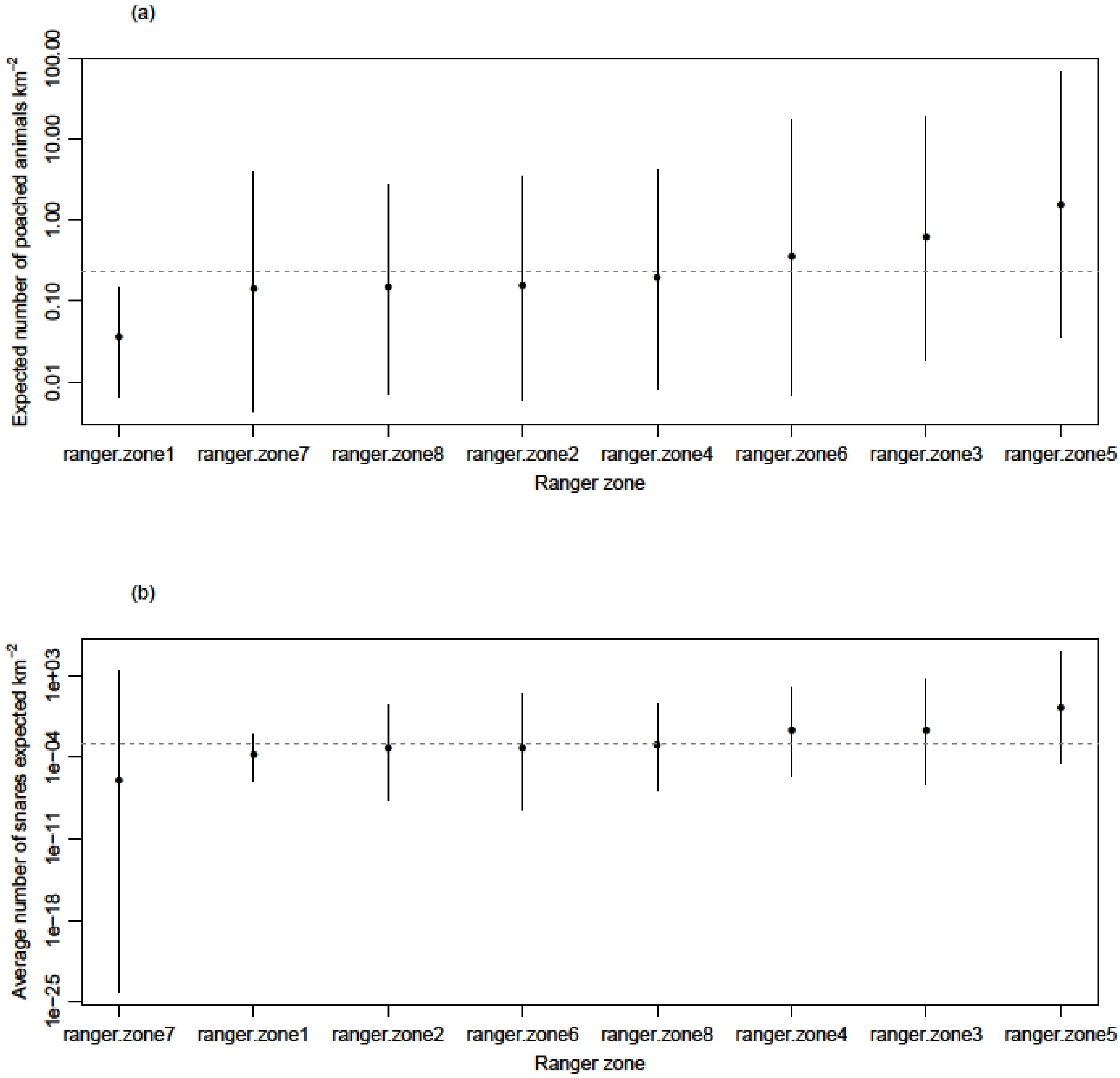

Figure 3.8. Relative effect of individual ranger zones on illegal activity deterrence within the Serengeti Ecosystem showing some ranger zones were more effective in combating poaching than others. The dotted line indicates the median effect separating the better (below the line) and worse (above it) zones in anti-poaching effectiveness.

Table 3.7. Results from landscape scale analysis showing influence of various covariates on the abundance of wire snaring at the landscape scale. There was evidence for snaring (**bold text**) mostly in areas with high NPP and low wildebeest density and altitude. Wire snaring also showed differing pattern at different ranger zones suggesting variation in poaching deterrence between zones in the Serengeti Ecosystem.

Model Covariates (**SNARES**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-8.839 ±2.357	-13.978	-8.668	-4.681
NPP	**4.368 ± 1.384**	**1.992**	**4.245**	**7.450**
Distance to rivers	-1.286 ± 1.327	-4.055	-1.239	1.209
Wildebeest density	**-1.594 ± 0.842**	**-3.411**	**-1.542**	**-0.077**
Buffalo density	-0.137 ± 0.372	-0.896	-0.129	0.580
Woody cover	0.024 ± 0.653	-1.258	0.019	1.332
Woody cover (quadr)	-0.793 ± 0.573	-1.990	-0.772	0.282
Altitude	**-2.694 ± 1.130**	**-5.067**	**-2.647**	**-0.593**
Village distance	0.651 ± 1.248	-1.873	0.670	3.076
Village distance(quadr)	-0.617 ± 1.150	-3.123	-0.530	1.421
Distance to park roads	-0.934 ± 0.633	-2.270	-0.905	0.237
Distance to ranger st.	0.391 ± 0.450	-0.485	0.385	1.299
Sampling year (2016)	-0.953 ± 0.972	-3.021	-0.903	0.826
Ranger zone 2	1.131 ± 2.453	-3.851	1.173	5.867
ranger zone 3	4.991 ± 2.916	-0.521	4.899	11.032
ranger zone 4	**4.818 ± 2.127**	**0.942**	**4.701**	**9.355**
ranger zone 5	**9.394 ± 3.263**	**3.462**	**9.214**	**16.360**
ranger zone 6	1.291 ± 3.506	-5.818	1.345	8.086
ranger zone 7	-7.911 ± 14.308	-41.636	-5.341	12.599
ranger zone 8	1.945 ± 1.959	-1.714	1.870	6.034

Table 3.8. Results from landscape scale analysis indicating significant (**bold text**) influence of various covariates on the abundance of illegal cattle grazing at the landscape scale. There was evidence for ranger deterrence between management zones, but illegal grazing was high mostly in areas closer to rivers and lower altitude, closer to villages and park roads.

Model Covariates(**COW**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-165.29 ± 62.29	-314.65	-154.07	-74.07
NPP	12.94 ± 11.91	-7.60	11.65	39.70
Distance to rivers	**38.32 ± 23.74**	**7.64**	**33.32**	**96.07**
Wildebeest density	-2.66 ± 8.94	-19.88	-3.00	18.01
Buffalo density	-5.38 ± 6.24	-18.97	-4.87	7.28
Woody cover	-22.23 ± 12.88	-45.65	-22.34	3.67
Woody cover (quadr)	9.43 ± 4.92	-0.17	9.60	18.03
Altitude	**-14.23 ± 8.57**	**-35.02**	**-12.74**	**-1.90**
Village distance	**-25.72 ± 19.89**	**-76.10**	**-21.21**	**-0.80**
Village distance(quadr)	-3.83 ± 11.35	-23.00	-5.32	26.00
Distance to park roads	**-44.64 ± 22.23**	**-94.06**	**-42.67**	**-9.43**
Distance to ranger st.	1.52 ± 3.20	-4.52	1.28	8.59
Sampling year (2016)	-9.73 ± 8.17	-31.15	-7.72	0.28
Ranger zone 2	13.69 ± 23.55	-41.42	17.68	48.31
Ranger zone 3	**83.50 ± 24.49**	**41.63**	**81.89**	**136.77**
Ranger zone 4	**63.32 ± 20.19**	**17.67**	**64.04**	**100.09**
Ranger zone 5	-14.96 ± 27.19	-77.51	-11.05	26.62
Ranger zone 6	**61.92 ± 19.10**	**31.85**	**59.32**	**106.03**
Ranger zone 7	-43.88 ± 26.36	-103.68	-40.47	-0.14
Ranger zone 8	**-57.53 ± 34.00**	**-129.94**	**-55.88**	**2.37**

Table 3.9. Results from landscape scale analysis indicating significant (**bold text**) influence of various covariates on the abundance of illegal tree cutting at the landscape scale in the Serengeti. There was evidence for ranger deterrence between management zones but tree cutting was high mostly in areas closer to villages and lower wildebeest density.

Model Covariates(**TREES**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-55.349 ± 26.857	-128.899	-49.599	-19.807
NPP	5.218 ± 6.414	-2.627	3.602	22.226
Distance to rivers	3.272 ± 3.854	-3.347	2.700	12.482
Wildebeest density	**-12.912 ± 8.640**	**-35.756**	**-10.845**	**-2.449**
Buffalo density	-0.220 ± 1.545	-3.735	-0.126	2.711
Woody cover	4.022 ± 5.068	-1.845	2.661	18.314
Woody cover (quadr)	-1.749 ± 2.582	-9.005	-1.078	1.334
Altitude	-5.563 ± 5.762	-20.404	-4.211	1.746
Village distance	**-14.397 ± 13.365**	**-52.795**	**-10.348**	**-0.828**
Village distance(quadr)	-2.162 ± 4.914	-15.206	-0.926	4.011
Distance to park roads	0.083 ± 2.730	-6.747	0.428	4.398
Distance to ranger st.	3.420 ± 3.226	-0.224	2.562	12.158
Sampling year (2016)	-2.124 ± 3.464	-10.898	-1.450	3.012
Ranger zone 2	16.188 ± 9.733	-0.235	15.197	37.601
Ranger zone 3	**23.070 ± 11.274**	**6.109**	**21.361**	**49.631**
Ranger zone 4	**34.992 ± 14.726**	**12.275**	**32.901**	**69.646**
Ranger zone 5	-5.495 ± 24.363	-62.888	-1.176	29.686
Ranger zone 6	**42.974 ± 16.729**	**16.926**	**40.702**	**81.885**
Ranger zone 7	-11.731 ± 21.769	-62.846	-8.157	20.210
Ranger zone 8	**19.355 ± 9.445**	**5.076**	**17.921**	**41.479**

Table 3.10. Results from landscape scale analysis indicating significant (**bold text**) influence of various covariates on the abundance of Other signs of illegal activities at the landscape scale in the Serengeti. There was evidence for ranger deterrence between management zones, but more signs were recorded mostly in areas with high NPP, lower wildebeest density and away from ranger stations.

Model Covariates (**OTHER**)	Mean effect	Lower quantile (0.025)	Median quantile (0.5)	Upper quantile (0.975)
Model intercept	-8.901 ± 2.635	-14.879	-8.621	-4.521
NPP	**3.144 ± 1.092**	**1.294**	**3.041**	**5.607**
Distance to rivers	-0.456 ± 0.946	-2.400	-0.433	1.359
Wildebeest density	**-1.433 ± 0.708**	**-2.990**	**-1.383**	**-0.179**
Buffalo density	-0.213 ± 0.353	-0.979	-0.191	0.424
Woody cover	-1.473 ± 0.886	-3.377	-1.421	0.133
Woody cover (quadr)	0.271 ± 0.490	-0.656	0.256	1.289
Altitude	-0.663 ± 0.912	-2.661	-0.599	0.957
Village distance	-1.057 ± 1.253	-3.521	-1.065	1.459
Village distance(quadr)	-0.094 ± 0.833	-1.938	-0.025	1.358
Distance to park roads	0.018 ± 0.470	-0.956	0.032	0.904
Distance to ranger st.	**1.365 ± 0.498**	**0.517**	**1.320**	**2.479**
Sampling year (2016)	-0.944 ± 1.099	-3.422	-0.840	0.923
Ranger zone 2	3.188 ± 2.485	-0.997	2.942	8.786
Ranger zone 3	**4.787 ± 2.565**	**0.362**	**4.560**	**10.504**
Ranger zone 4	3.663 ± 2.636	-0.936	3.449	9.474
Ranger zone 5	5.226 ± 3.245	-0.487	4.982	12.332
Ranger zone 6	3.904 ± 3.234	-1.657	3.616	11.109
Ranger zone 7	3.819 ± 2.698	-0.987	3.637	9.684
Ranger zone 8	-0.075 ± 2.007	-4.046	-0.081	3.914

3.6. Discussion

I quantified the spatial patterns of all illegal activities and analysed their drivers in the Serengeti ecosystem. Although I found significant correlates of illegal activities at each scale of analysis, the strongest effects were identified at the fine and landscape scales. For all classes of activity, the significant effects were differences between management areas, with consistently higher levels of illegal activity in Serengeti National Park and Maswa Game Reserve than in the Grumeti and Ikorongo Game Reserves. This result may seem surprising but counters an earlier argument by Caro (1999) that game reserves were ineffective conservation areas for large African ungulates, and suggests that improved enforcement may be able to reduce illegal activities in protected areas regardless of the level of protection. Several illegal activities were either associated with water, wildebeest density and NPP (wire snaring and poached carcasses) or distance to villages (poached carcasses, plant extraction: grasses and trees) and ranger station (other signs), consistent with previous studies elsewhere in east and southern Africa (Critchlow et al., 2015; Watson et al., 2013). Further, fine-scale patterns of illegal activities tended to correlate significantly with water, ground cover and abundant animal tracks (for poached carcasses and plant extraction). I found a strong effect of paired trees on wire snaring pattern in the Serengeti ecosystem which has not been reported elsewhere previously.

3.6.1. Illegal activity distribution and drivers

Previous work in Serengeti has suggested poaching hotspots exist in the western corridor and parts of northern Serengeti (Campbell & Hofer, 1995; Metzger et al., 2010). As expected, I found signs of heavy animal poaching in these areas, but I also identified hotspots of poaching, mostly in wooded parts of the eastern and central areas not reported previously. I also found illegal cattle grazing and plant extraction (trees and grass for thatch) tended to be concentrated in the peripheries of Serengeti National Park and Maswa Game Reserve. The finding of poaching hotspots in the eastern corridors farther away from areas with the highest human populations and known high poaching pressure (Campbell & Hofer, 1995; Metzger et al., 2010) is unusual and may suggest four things. First, there may be a depletion of potential prey sought by the poachers in western borders of Serengeti National Park, i.e. the common hunting zones (Campbell

& Hofer, 1995); second, it may reflect improved anti-poaching effort in the western corridors, which may have displaced poaching farther into the east; third, it may indicate improved equipment (e.g., motorbikes) and organisation of poaching activities, facilitating exploitation of game in more distant locations; or, fourth, it might suggest that such activities have simply been under-reported in the past. Poaching pressure is known to respond to anti-poaching activities by either increasing during times of low enforcement effort (Hilborn et al., 2006) or in areas less frequently patrolled (Moore et al., 2017).

The wide-spread nature of animal poaching, but more marginal distribution of cattle grazing and plant harvesting in Serengeti is similar to the patterns of illegal activities reported in Queen Elizabeth National Park, Uganda (Critchlow et al., 2015), but differs from the more restricted poaching activities reported from the protected area borders in Kenya national parks (Kimanzi et al., 2015; Wato et al., 2006), South Luangwa National Park, Zambia (Watson et al., 2013) and Sundarbans Reserves in India and Bangladesh (Aziz et al., 2017). Two possible reasons for these differences are the distribution and type of resources being sought and the balance between the cost and benefits of acquiring that resource. As risk of being caught increases, poachers are more likely to operate around the edges of PAs. Where that risk is low, they will operate where the chance of finding animals is highest. On the other hand, poachers may take more risks and hunt in the middle of PAs, if benefit is higher. In the Queen Elizabeth National Park for example, commercial poaching occurs widely in the interior of the park, while non-commercial activities are restricted within the PA peripheries (Critchlow et al., 2015). Thus, the distribution of illegal activities relative to park peripheries could provide useful insights into the effectiveness of ranger patrols as a deterrent and of the value of products being poached: viewed this way, changes in locations of illegal activities could be a useful indicator of changes in the costs and benefits of poaching.

Whereas poaching occurred farther away from ranger stations in the Sundabarns Reserved Forest in India/Bangladesh (Aziz et al., 2017), or closer in Khai Yao National Park, in Thailand, due to high animal density near ranger stations (Jenks et al., 2012), the current study did not find this effect on animal poaching except for other signs of illegal activities, suggesting poachers operate regardless of ranger activity in this ecosystem. Notwithstanding the lack of direct effect, I did find significance differences between management areas. Despite bordering areas of high human density, illegal activities were scarcer in the Ikorongo and Grumeti reserves than the Serengeti National Park or Maswa Game Reserve. Ikorongo and Grumeti have proportionally more resources (8.2 km^2 per ranger area) for conducting anti-poaching activities and are managed jointly between the government and a private business sector, whereas Serengeti NP (43.5 km^2 per ranger area) and Maswa (72.1km^2 per ranger area) are managed by the government alone, echoing Hugo Jachmann (2008) study of resourcing and poaching rates in Ghana. Thus, it seems plausible that poachers totally avoid highly patrolled areas, but are indifferent to ranger posts where densities are lower. Although game reserves are sometimes considered ineffective in conserving wild ungulates in Tanzania, due to high anthropogenic pressures on them (Caro, 1999), these results for Grumeti and Ikorongo Game Reserves demonstrate an example where sufficient resources invested in protecting the wildlife, coupled with good management, can generate effective conservation in these areas.

Not finding a significant association between poaching and distance to the park roads is surprising because roads are known to influence poaching patterns elsewhere (Aziz et al., 2017; Wato et al., 2006; Watson et al., 2013). Roads facilitate access to hunting areas (Haines et al., 2012), and are used for transporting bushmeat to market (Lindsey et al., 2013a). The vastness of the Serengeti ecosystem, the low ranger activities in some management areas (i.e. Serengeti National Park and Maswa Game Reserve), and the relative ease of riding motorbikes off-road within the ecosystem, may have contributed to the widespread poaching activities regardless of the roads network. Further work into how such poaching behaviours explain the spatial patterns of poaching in Serengeti will be necessary to enable effective targeting of law enforcement strategies to reduce poaching.

The pattern of cattle incursion and tree cutting are likely driven by demands from people living close to the PAs and because they can find what they want relatively easily (Mackenzie & Hartter, 2013). Additionally, the existing penalty imposed against these activities (ca. US$23 per cow at the time of fieldwork) may incline grazers to concentrate closer to borders to escape more easily to their villages. Anecdotal evidence suggests that illegal grazing in some areas bordering the villages is conducted during night to avoid law enforcement rangers. Illegal cattle grazing may impact native wildlife species negatively through increased resource competition (Madhusudan, 2004), increased risk of disease transmission from domestic animals (Wiethoelter et al., 2015) and coupled with tree cutting may increase habitat loss potentially exposing species to edge effects such as reducing wildlife abundance along borders and pushing animals further towards the interior (Brodie et al., 2015).

3.6.2. Conservation Implications

Illegal activities remain widespread and frequent in many protected areas. Understanding their spatial drivers provides insights into fighting and reducing illegal activities impact on the wildlife. Many of the patterns of illegal activities observed in the Serengeti ecosystem mirror some existing situations in several other protected areas faced with poaching across the tropics. These results could thus be useful to improving law enforcement strategies in protected areas elsewhere. The variation in the effectiveness of different PAs, suggest that increased investment in rangers and patrols could reduce the levels of these activities and that managing large conservation landscapes in isolation may not offer long-term conservation effectiveness because of the risk associated with the redistribution of poaching events in PAs which are relatively weakly protected. Further, in conserving migratory wildlife, effectiveness could greatly improve by adopting a co-management model for the wildlife, or at least collaborate in anti-poaching activities, such as sharing intelligence information about potential poaching incidences, ranger training opportunities etc. This could help redress some conservation limitations (e.g. resources) especially in protected areas where resources may be relatively thinly spread.

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CHAPTER 4 - Snare detection and the mortality risks to large wild mammals in the Serengeti ecosystem: a field experiment

Photograph of poacher wire snare set to trap animals taken in the Western corridor of the Serengeti National Park during field survey

4.1. Abstract

Law enforcement is an important component of protected area management, and identifying the effectiveness of law enforcement in the field can help target conservation resources. Despite many protected areas being faced with persistent poaching using wire snares, there are very few empirical studies evaluating the efficiency of the anti-poaching efforts in tackling animal snaring problem. Hence, the capacity of ranger patrols to reduce the mortality of wild mammals is unclear. I set 2316 dummy wire snares across the Serengeti ecosystem to test ranger detection efficiency, and evaluated the habitat and environmental factors influencing snare detectability and animal capture. Monthly snare detection rate was low overall (0.033), but differed significantly between management zones. Snares were more likely to be detected where the density of bushes was low and when dummy snares were in large groups. The median expected daily 'potential capture rate' of animals in the dummy snares was estimated to be 0.025 animals per snare per day, increasing significantly in sites with tall trees. I estimated that eighteen species of ungulate could have been caught in the snares, and several of these species are known to be in decline in the ecosystem. These results indicate that effective control of poaching (by direct location of set snares) in this ecosystem is currently lacking. However, even the best patrolled areas still had low detection probabilities, suggesting that ranger patrols to remove snares are unlikely to significantly decrease animal mortality unless they also deter poachers. These data suggest that deterring poachers is likely to be as important as snare detection (or more so) in protected areas that are subject to poaching with snares.

Keywords: Animal capture rate, dummy wire snare, poaching simulation, Serengeti ecosystem, snare detection probability.

4.2. Introduction

Protected areas (PAs) are losing wildlife often from poaching (Harrison, 2011; Laurance et al., 2012), potentially emptying reserves and undermining the ecological and biological functions they were designed to protect (Wilkie et al., 2011).While some forms of poaching are high profile (e.g. elephant and rhinos), others, such as snaring for bushmeat, are widespread but their effects are underappreciated by management authorities in many PAs (Bruner et al., 2001; Harrison, 2011). Animal snaring is a persistent conservation problem in PAs across the tropics, noted in west and central (Fa & Brown, 2009), east (Wato et al., 2006), and southern Africa (Becker et al., 2013), and also in Southeast Asia (Aziz et al., 2017; Corlett, 2007). However, although the ability of rangers to detect and remove snares is a crucial part of reducing poaching pressure, the effectiveness of rangers has received much less attention. This could be attributable to the difficulty associated with surveying snares leading to poor knowledge of the full size of the wire snaring problem in the field and the assumptions by many PAs management authorities that snare poaching and its impact are minimal (Harrison, 2011; O'Kelly, 2013). Understanding the ability of field rangers to detect and remove snares can lead to a better understanding of the magnitude of the snaring problem, its impact on the target animals and could provide the appropriate information to devise strategies to help field rangers find and remove wire snares more effectively. Here I seek to assess the detection efficiency of dummy snares and the likely rates at which animals are captured in snares in an iconic conservation landscape in East Africa.

A recent increase in research on snare locations in PAs has led to useful insights into poacher activities. For example, field-based snare surveys show that poachers are more likely to set snares in areas near to transport roads and permanent water bodies (Wato et al., 2006; Watson et al., 2013), away from patrol ranger stations and park boundaries (Aziz et al., 2017; Watson et al., 2013) and in areas with a high abundance of target animals (Kimanzi et al., 2015). Further, retrospective analyses of ranger-collected poaching information in protected areas also indicate that poachers are likely to return to hunt in areas where they have been successful on previous hunting trips (Critchlow et al., 2015). Altogether, these studies improve our knowledge that could be useful in fighting poaching in protected areas (Moore et al., 2017).

Snares are a potent hunting tool, mostly used to catch animals in PAs where there is strict law enforcement (Fa & Brown, 2009). Compared with other hunting tools, such as guns, snares are silent, cheap and can be placed and checked efficiently, reducing the risks of being caught by park rangers (Kümpel et al., 2009). In Serengeti National Park, where ranger activity is high, poachers often set snares quickly, then remain hidden in secluded hunting camps emerging to inspect their snares and collect any catch once or twice daily, at times when ranger activities are perceived to be low (Kaltenborn et al., 2005). Snares can have significant direct and indirect effects on the target species in both the short and long-term. In the short-term, snares directly kill, cause injuries or maim animals, reducing population density of a target species overall (Aziz et al., 2017; Fa & Brown, 2009; Noss, 1998). In the longer term, the severe decline of hunted animal populations may impact ecosystem functions, such as reduced herbivory associated with biomass collapse, and seed dispersal limitations (Dirzo et al., 2014; Peres et al., 2016; Stokstad, 2014). In addition to the deliberately-caught animals, snares may be left unchecked if poachers are disturbed, or abandoned if they perceive the costs and risks associated with collecting snares at the end of a poaching trip are too high (i.e. higher than the cost of obtaining new ones to set in future). The magnitude of this is unknown, but in the Central African Republic and Equatorial Guinea, Noss (1998) and Kümpel et al. (2009) reported 27% and 9% of animals caught in snares in the Bayanga and Mbo reserves and Monte Mitra forest, respectively, were not available for use by hunters due to scavenging and decomposition. It may be presumed that most of these had been caught in snares that had been left temporarily or permanently untended. This suggests that increased efforts to remove snares could reduce animal mortality in protected areas. Furthermore, although snares are often set with particular species in mind (Coad, 2008), once set, a snare can catch any animal including birds, reptiles and mammals, meaning that species with no value to poachers can be caught and never appear in markets (Campbell & Hofer, 1995; Kümpel et al., 2009; Noss, 1998). Nonetheless, they contribute to animal mortality.

Within PAs, snare removal (desnaring) may help in two ways: by removing a direct threat to the animals and also discouraging poacher's activities, especially when desnaring rates are higher than the ability of the poachers to replace them (Moore et al., 2017). Despite snare poaching being widespread in the tropics, few studies have evaluated the efficiency of desnaring by rangers in protected areas, or its effect on animal mortality. One recent study is by Moore et al. (2017), who analysed retrospective ranger-collected poaching information in Nyungwe National Park to examine ranger efficiency. Although they report decline of snaring in some areas that were frequently patrolled, they were unable to demonstrate a clear ranger deterrence effect. The most extensive study of snare removal is from Cambodia, where O'Kelly (2013) investigated snare detection in Seima forest reserve. This study found only 30% (35 out of 115 set snares) of snares were detected even when rangers were informed of the presence of snares within a restricted 1km study area, suggesting that overall rates of snare detection could be very low. Nonetheless, there remains a lack of knowledge of how rangers work to target snares in a savannah ecosystem in which visibility is relatively higher than in the forests (O'Kelly, 2013).

In the present study, I simulate poaching to assess the rates at which rangers detect dummy wire snares, and then estimate the extent to which snare detection might (or might not) reduce poaching in the Serengeti ecosystem. I expected that snare detection would be highest in the protected areas which have the highest investment in law enforcement (i.e. in game reserves) due to assumed increased patrols (Hilborn et al. 2006). I examine the environmental factors (habitat) which influenced snare detection probability; and estimated the mortality risks to large mammals posed by wire snaring by examining the extent to which species vulnerability to poaching is influenced by the same environmental factors. I discuss the results in light of the existing field situation and provide suggestions for future research to test ranger efficacy in the detection of snares in protected areas. Testing ranger performance in the field enables us precisely to estimate the overall levels of snaring in the Serengeti, and hence its impacts on the species affected. It therefore underpins efforts to improve the ability of ranger patrols to detect and remove snares in protected areas.

4.3. Materials and Methods

4.3.1. Study areas

We conducted experiments in the Serengeti National Park and two neighbouring game reserves in the Serengeti ecosystem, Tanzania. The protected areas are managed by different authorities and differ markedly in the law enforcement investment accorded to them. All areas are strictly protected: Serengeti National Park allows only photographic tourism while regulated trophy hunting occurs in the Grumeti and Ikorongo Game Reserves. Each protected area is enforced by a team of game rangers who conduct regular patrols to prevent activities such as wire snare poaching, livestock grazing and plant extraction. The anti-poaching activities within these protected areas are coordinated from ranger stations scattered in arbitrary management zones (i.e. east, west, north etc.) within each protected area. For analysis, as in the rest of this paper, we refer to Ikorongo and Grumeti reserves as separate management zones, and separate Serengeti National Park into its separate management zones.

Short grass plains characterise the central and southern parts of the ecosystem, whereas the north and western corridors are extensively covered by sparse-tree vegetation dominated by *Acacia* and *Commiphora* species (Reed et al., 2009). Rainfall is bimodal, falling at 500 mm annually in the east and southern corridors peaking up to 1200 mm in November through May in the west and northern corridors during wet season (Sinclair & Arcese, 1995). The west, north and southwest of Serengeti National Park is covered by savanna or woodland, receiving an average annual rainfall of 800 mm (Sinclair & Arcese, 1995). Despite protection, illegal hunting using wire snares, illegal livestock grazing and plant exploitation are common threats to the ecosystem (Loibooki et al., 2002; Metzger et al., 2010).

4.3.2. Data collection and analysis

To test the hypothesis that rangers were more likely to detect snares in PAs where there was high investment in law enforcement (i.e. more rangers per unit area and more resources: vehicle, fuel, etc) and the ecological factors driving snare detection, I simulated poaching by setting dummy snares (see below for a description) in the Serengeti National Park and the two game reserves. In each management zone, I aimed to set at least 30 groups of snares in locations typical of actual poacher activities, accessing locations in all regions of each zone by vehicle or on foot. To site dummy snares effectively, we used experience of how poachers set snares gained over from initial surveys in 2015 (during which 340 snares were located; Rija et al. *in prep*) and from 3.5 years working as a ranger. I set a total of 2316 dummy snares, distributed across 309 separate locations (sites). The dummy snare made of an easy-to break plastic material (hence cannot catch or cause harm to animals), looks very similar to most real wire snares used by poachers in these protected areas (Figure 4. 1). Since poachers typically set groups of snares, we set a random number of between 4 and 20 snares at each of the 309 locations used. Each snare within a group was $\leq$ 200 m from another snare, and each site within a zone was separated by at least 2 km from the next site (Figure 4. 2). We used local habitat characteristics, such as available water pools, abundant green grasses, available ungulates; animal trails etc. to select sites to locate dummy snares. These habitat features have significant correlations with the distribution of illegal activities in the Serengeti ecosystem (Chapter 3) and are frequently used by rangers as cues for detailed searches for signs of poaching (Walsh & White, 1999). Overall, the dummy snare experiments spanned seven management zones (Figure 4. 2); East = 415 snares (in 61 groups), North = 295 (41), South = 236 (23), Central = 376 (67), Ikorongo = 346 (45), Grumeti = 120 (11) and West = 528 (61). We fixed each dummy snare on a tree with a loop positioned mostly between paired growing trees. Whenever possible, we constructed small bush fences similar to those used by poachers (Rija pers. obs.) to guide animals through the intended paths and snare and to ensure equivalent detectability of dummy and real snares.

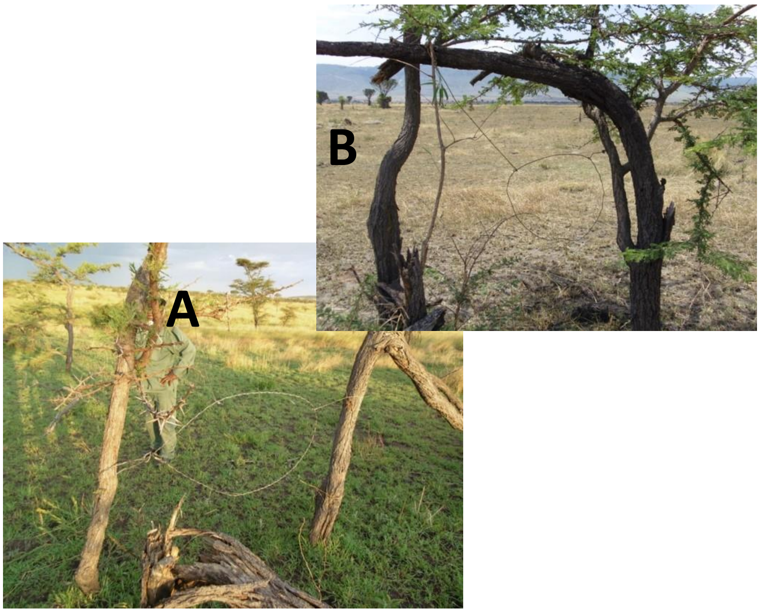

Figure 4. 1. Photographs of real poacher snare (A) and a dummy snare (B) used in poaching simulation experiment to understand ranger detection efficiency in the Serengeti ecosystem. Photograph courtesy by SCCri team 2015/2016.

To test the hypothesis that habitat structures (e.g. available water pools, bushes, animal tracks etc.) and characteristics (e.g. herb height, tree height etc.) influence detection of the dummy snares, we recorded the exact GPS coordinates of each snare set, as well as its proximity to the nearest snare, and the number of trees, bushes and animal trails within a radius of 20 m of the snare. Available water pools (within visible distance from set dummy snares) and whether the snare was set on an animal trail was recorded. We also measured the ground cover, herb height, and tree height within a radius 20 m of the snare as these characteristics have been shown to significantly influence the distribution of wire snares in Serengeti (Chapter 3). I expected these to influence snare detection by the rangers. I also separated ‘trees’ into those that were single or paired, but these numbers were highly correlated with tree number (r >0.5), so I used tree number in subsequent analysis.

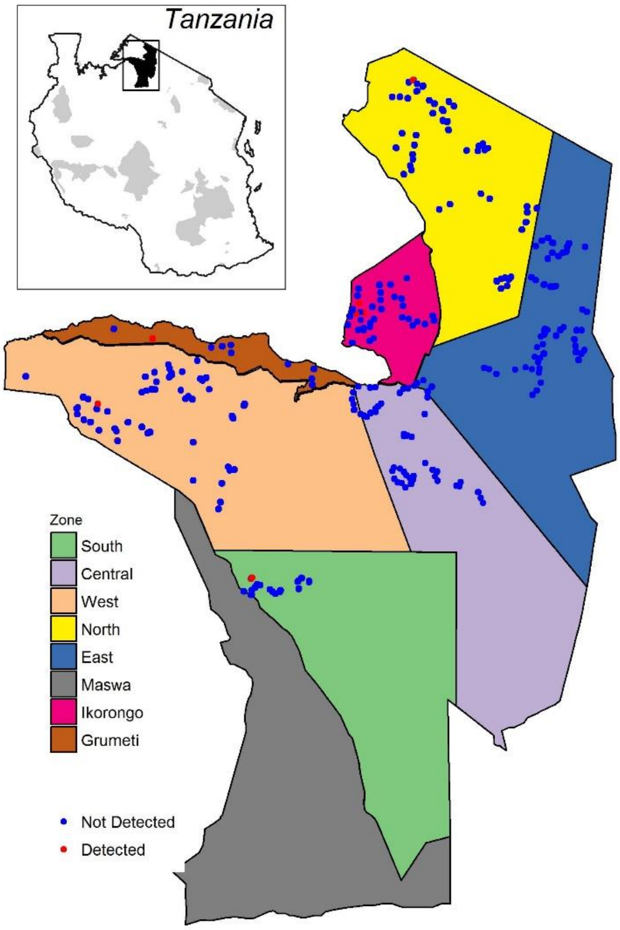

Figure 4. 2. Location of the three protected areas in Serengeti Ecosystem (Serengeti National Park and Grumeti and Ikorongo Game Reserves) in Tanzania and the spatial distribution of the dummy snare experiments across different management zones. There was low snare detection by the rangers during the three months of field testing.

To estimate the daily detection rate of snares and the mortality risks posed to the animals, we left the snares in the field for a period of up to twelve weeks (range 30-84 days) before returning to remove unfound snares, recording the length of time each snare was in the field and undetected. Throughout this period, we recorded the recovery of dummy snares by the rangers, who had been requested to remove every dummy snare they encountered during routine patrols. For each ranger-recovered snare, we recorded the date recovered. To minimize errors resulting from the rangers being informed of the location of the dummy snares, we set dummy snares without ranger presence, but rangers were informed of the experiment when the snaring within a particular management zone was completed. Our discussions with rangers during field work showed they were positive about these experiments and rangers were honest to report that they did or did not detect any dummy snare or simply they did not perform any patrols (e.g. in East and Central zones of Serengeti National Park) due to resource limitations. This suggests that ranger effort may have not been particularly biased by their perception of these experiments, though we acknowledge the possibility.

To assess species mortality risks from wire snares, we recorded the status of each snare during removal. If the snare was intact and undisturbed we assumed that no animals would have been captured, but if the snare loop was broken open and confirmed the cause to be an animal walked into it, we assumed a capture would have been made. In reality, not all disturbed snares will result in capture, so our estimates of capture rates could be biased upwards. At disturbed snares, we recorded the species that would have been captured, based on fresh or recent signs at the location, such as animal dung/pellets, spoor and hairs of animals.

4.3.3. Data analysis

I modelled the daily survival (i.e. non-discovery) of dummy snares to obtain the detection probability following Aebischer (1999). Essentially, daily snare survival can be modelled as the binomial proportion of days not discovered over the total days in the field. For snares that were not recovered by rangers, both values are the length of time between setting and recovery; for those discovered, this was the number of days between setting and the day before recovery, over the total number of days in the field. To examine the effect of different habitat characteristics and animal density (Wildebeest data as used in Chapter 3) on snare detection probability, I fitted a mixed effect model to these data, taking status of individual snare (i.e. detected or not) as the dependent variable and measured habitat parameters as explanatory variables. To account for the confounding effect of snare grouping on the detection probability, I modelled snare group ID as random factor. To examine the effect of individual variables on the detection, I built twelve models, each with a different combination of variables, examining the best model fitting the data based on Akaike's Information Criterion (AIC) values: several models showed similar predictive power, thus required model averaging (Grueber et al., 2011). I performed model selection and averaging of the best competing models using AIC values and Akaike's weights (delta ≤ 2) (Burnham & Anderson, 1998; Grueber et al., 2011). I used predictions from the final averaged model to visualize the effect of the covariates most strongly influencing detection probability. Further, I used result from this mixed model to explore the relative contribution of individual management zones on the detection probability of dummy snares.

To estimate the mortality risk presented by an individual snare, I calculated the overall proportion of snares that were not disturbed on removal, and using the mean number of days in the field converted this to the daily probability of capture (P_d) using the equation:

$$P_d = 1 - \exp(\log(N_{nd} / N_s) / T)$$

Where N_{nd} is the number of snares that were not disturbed during the experiment, N_s is the total number of snares and T is the mean time (in days) snares were present in the field. Because very few (<1%) snare trapped animals manage to escape which may end up dyeing of severe injuries (Kümpel et al., 2009; Noss, 1998), I assumed that all animals that get trapped would eventually be removed from the ecosystem.

I used this as a dependent variable in a binomial Generalized Linear Model (GLM) with logit link function to examine the effect of the measured covariates on the species catch risks, setting snare days in field as an offset to account for the variability in the duration of the experiments. I used the deletion of nonsignificant terms, in turn, and the Chi- test to evaluate the relative effect of each covariate in the model (Bolker et al., 2009). Finally, to examine how the various habitat characteristics influenced the probability that individual species would be caught in dummy snares, I built models for four species (which were frequent enough to model, but considered at risk from poaching) using a similar procedure to that described for the overall data on 'captures'.

4.4. Results

Overall, I found snare detection efficiency was very low, with only 50 (~2%) of the dummy snares recovered by rangers during the experiment, with a corresponding average monthly detection probability of 0.033 (CI: 0.025-0.043). There was clear evidence for variation in detection of snares between management zones ($\chi 2 = 0.00$, df

= 6, p = 0.0001). Figure 4.3 shows that game reserves had highest and third highest detection probabilities, and hence higher than four of the five zones in the national park.

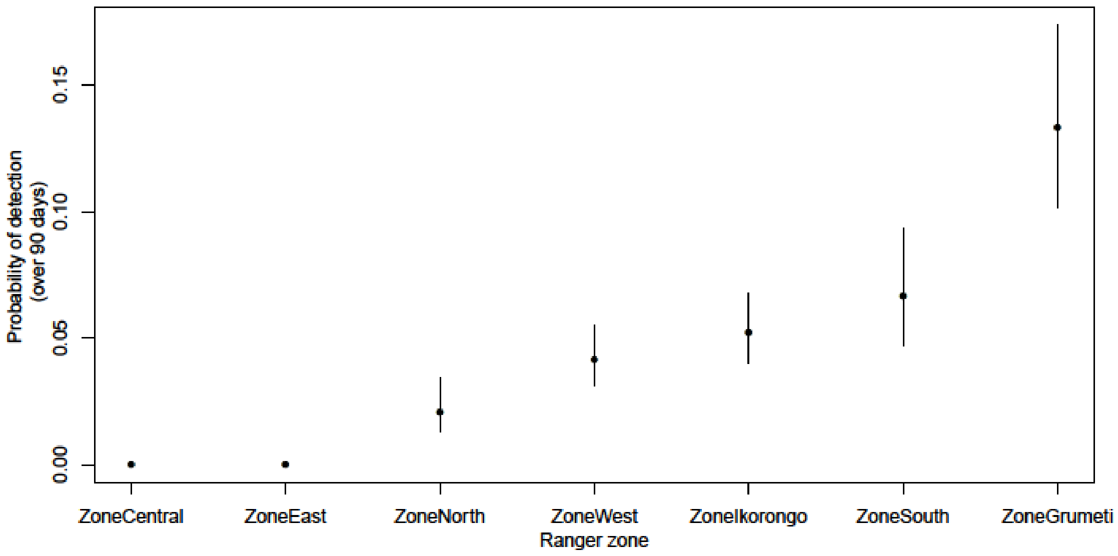

Figure 4. 3. Variation in ranger performance of anti-poaching activities on the detection probability of dummy snare across different management zones with Grumeti zone showing relatively higher detections (+ confidence interval) than other zones. Central and East zones show excessive intervals because there was zero dummy snares recovered in these zones suggesting zero patrols were conducted during the study period.

I also found strong evidence that the presence of any bushes reduced the likelihood of snare detection ($\chi 2$ = 2.54, df = 1, p = 0.003, Figure 4.4). However, I did not find bushes important in snare detection when I considered how many are in an area, instead the number of water pools had a strong effect on the snare detection ($\chi 2$ = 0.0, df = 1, p = 0.045). Further, highest snare detection was associated with larger snare size χ^2 = 2.02, df = 1, p = 0.005; Figure 4.5). I found no evidence that other local scale covariates influenced dummy snare detection rates.

I found that 1760 snares (76%) could have caught an animal whilst set, representing a daily animal capture rate of 2.4%. Eighteen species would have been caught in our dummy snares (Figure 4.6). Overall, the probability of catching an animal decreased significantly as tree height increased ($\chi2$ = 2.077, df = 1, p = 0.0378: Table 4.1) but tended to increase in areas with abundant animal trails ($\chi2$ = 2.323, df= 1, p = 0.020) and high animal density ($\chi2$ = 4.366, df =1, p = 0.0001), and varied significantly across the management zones (Table 4.1). Significantly more snares (97.6%) would have caught an animal in the national park than in the game reserves (20.4%).

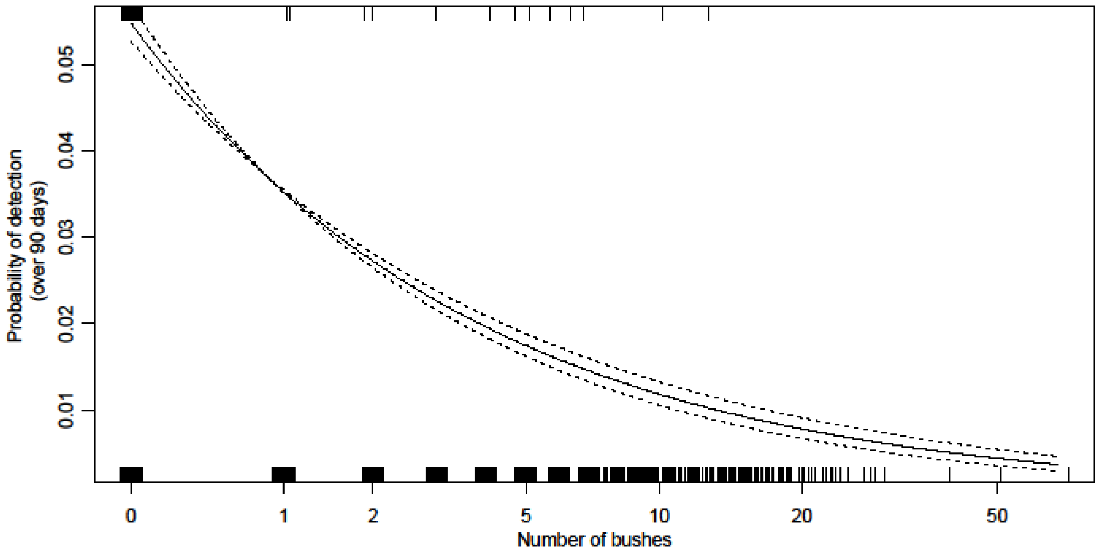

Figure 4. 4. Effect of bush density at snare sites on the probability of detecting dummy snares in the Serengeti ecosystem. Detection was higher in sites with no bushes, but after excluding these sites no further significant declines in detection were noted associated with increased bush cover. Low detection rates suggest weak enforcement by anti-poaching patrols in these protected areas.

When I examined individual species separately, I found species-specific capture risk associations (Table 4.1). For buffalo (*Syncerus caffer*), a high risk of capture in snares was strongly positively associated with high bush density ($\chi2$ = 3.027, df = 1, p = 0.0024). For zebra (*Equus burchelli*), high ground cover ($\chi2$ = 1.988, df = 1, p = 0.046) and snares set over animal tracks ($\chi2$ = 2.104, df = 1, p = 0.0354) strongly increased the high risk of capture probability while, high canopy height of trees ($\chi2$ = 1.222, df = 1, p = 0.039) and animal density ($\chi2$ = 3.116, df = 1, p = 0.001) were the strongest risk factors for wildebeest (*Connochaetes taurinus*) capture. Further, high capture risk of

Topi (*Damaliscus lunatus*) was strongly negatively associated with shorter grasses ($\chi2$ = -2.382, df = 1, p = 0.017) but tall herbs increased capture risk for impala, *Aepyceros melampus* ($\chi2$ = 4.119, df = 1, p = 0.0001). All the five species analysed were at high risk (at least 10% in Figure 4.6) of being caught in all the zones, reflecting the spatial distribution of these species across the ecosystem where our dummy snares were located. No further analysis was conducted for other species which 'were caught' in relatively low numbers i.e. below 10%.

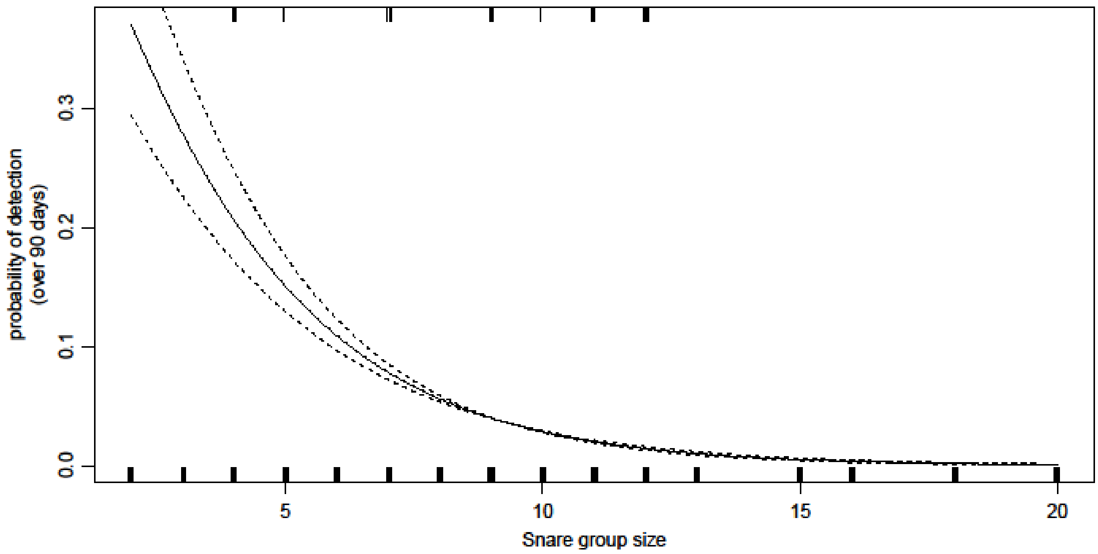

Figure 4. 5. The influence of snare group size at particular locality on ranger detection probability of dummy snares indicating high detection was likely in small snare cohort size. This result contradicts our expectation for many snares being found in large groups size and may suggest limited search effort is performed when poaching sign is encountered, thereby risking species being poached as more snares are likely to be left in the reserves.

Table 4.1. Results from binomial GLM analysis based on model averaging with the probability of catching animals in snares in the Serengeti ecosystem. The propensity for the animal to capture in dummy snares was associated with shorter trees, high animal density and was highest in the Eastern corridor of Serengeti ecosystem. Models with (*) show the variable had significant effect on the catch probability.

Model parameter	Mean	Adjusted SE	z-value	Pr(>|z|)
(Intercept)	-3.16601	0.37431	8.458	< 2e-16***
Ground cover (%)	-0.17051	0.09894	1.723	0.0848
Herb height	0.23013	0.14274	1.612	0.1069
Snare on track (Yes/No)	0.38857	0.21151	1.837	0.0662
No. of animal track	0.23328	0.1004	2.323	0.0202*
Tree height	-0.19865	0.09564	2.077	0.0378*
Number of bushes	-0.10681	0.1018	1.049	0.2941
Zone-East	0.78092	0.54205	1.441	0.1497
-Grumeti	0.51	0.47679	1.07	0.2848
-Ikorongo	0.53603	0.4047	1.325	0.1853
-North	-1.14296	0.58753	1.945	0.0517*
-South	0.29777	0.45926	0.648	0.5167
-West	0.07018	0.36663	0.191	0.8482
Wildebeest density	0.89335	0.20463	4.366	0.0001**
Number of trees	0.02294	0.06676	0.344	0.7312

4.5. Discussion

I found snare detection was low overall, but this differed between management zones. Dummy snare detection was influenced by local habitat characteristics particularly bushes and snare group size set in a particular locality. Further, I also found that eighteen species would have been caught by the dummy snares and the species capture probability was associated with different habitat characteristics and between the management zones.

Although there are no equivalent estimates from elsewhere in savanna ecosystems to provide wider context, the detection rate in Serengeti (3.3%) appears rather low. This detection result cannot directly be compared to the 30% snare detection rate in Seima forest ecosystem (O'Kelly, 2013), where visibility is relatively low and rangers were directed to search areas (in 1 km^2 searched by seven rangers) where snares had been set. In contrast, in my study system rangers patrolled a bigger area (e.g. western zone where a few dummy snares were detected is about 5200 km^2 patrolled by about 37 rangers) than Seima forest and we had not informed rangers about the locations of snares.

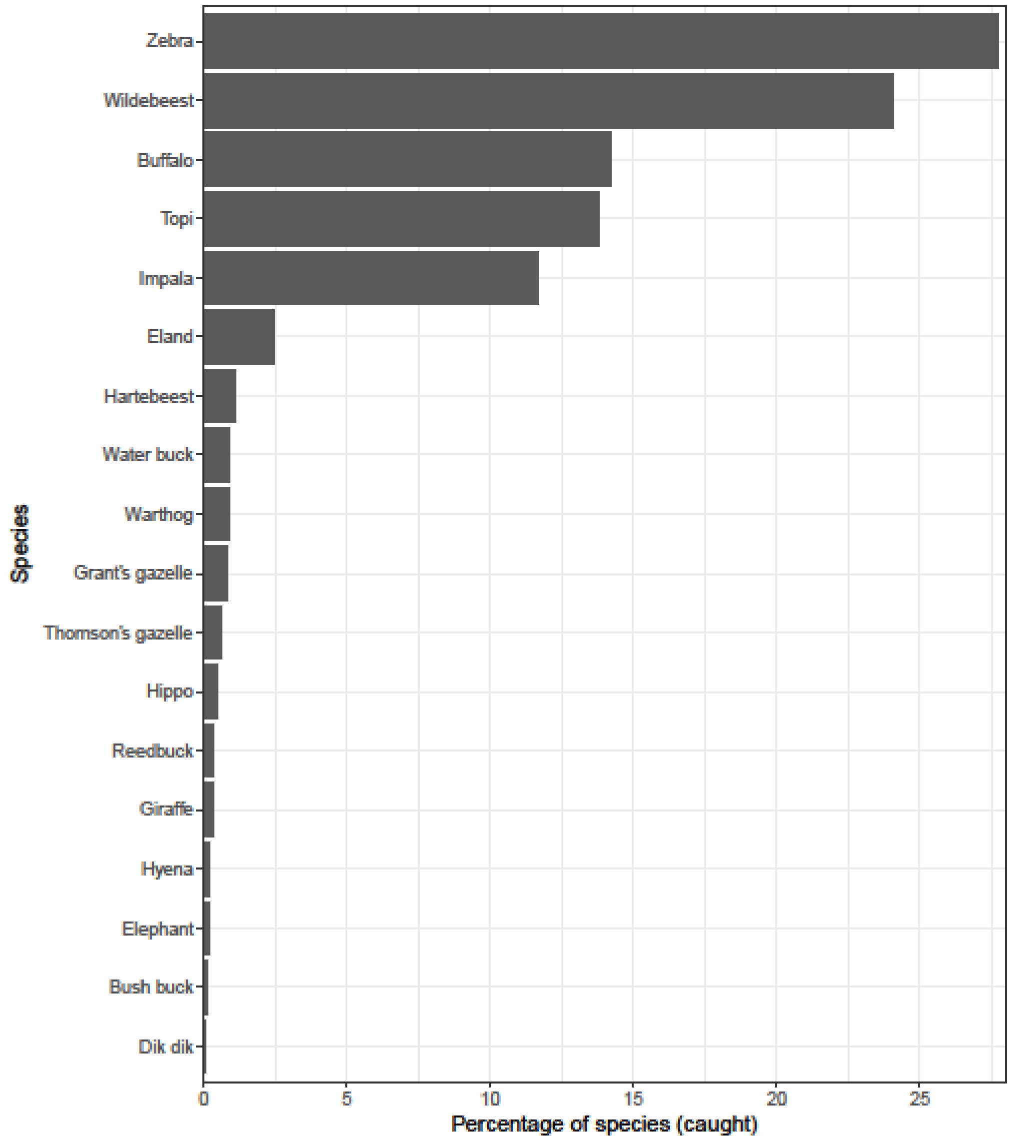

Figure 4. 6. The species ‘caught’ in dummy wire snares set in the protected areas in Serengeti. Overall, these data indicate some species may be at higher risks of being killed by poachers than others.

Unsurprisingly, I found that snare detection was lower in areas with more bushes. Poachers in Serengeti are known to use various techniques when setting snares including selecting suitable sites with bushes and high herb height (Chapter 3) which have the highest chance of catching animals. Such sites are also used to avoid easy detection of wire snares by park rangers. Improving detection in such areas may require more frequent use of foot than vehicle patrols. An influence of snare group size on detection rates was also not surprising and confirms similar finding in Cambodian Seima forest reserve where snares in drift lines are detected more than single snares

(O'Kelly, 2013). Although snare group size influenced detection, even within groups where at least one snare was found, relatively few snares were recovered (4% in Ikorongo Game Reserve) overall, probably because being unaware of the exact numbers set in a particular area it may have been difficult to decide on the search effort required recovering all the snares after the first encounter. This finding suggests that improving ranger search strategies is possible.

Successful snare detection requires two things to happen: firstly, rangers must actually patrol an area to detect anything; secondly, patrols must actually find the snares in the area they search. In our study we noted very few or zero patrols in two zones during the study period where there no dummy snares were returned and the rangers informed us that no patrols were conducted during this time. Most ranger stations in these zones had no rangers available or there was no vehicle or fuel for conducting road patrols or transporting foot patrols to locations away from the ranger post. In my experience such problems are not atypical (Arcese et al., 1995; Sinclair, 1995b), although our surveys did coincide with an ongoing ranger training programme that drew several rangers from protected areas for at least one month. This detection result cannot be linked to ineffectiveness of the existing patrol efforts alone rather the available patrol effort (i.e. rangers) seem far too lower than is probably required to effectively patrol the whole area. In my experience from a previous study (Chapter 3), a team of three rangers patrolling a 15 km transect of 0.08 km width in one day can effectively patrol an area of 1.2km^2/day (i.e. 36 km^2 in one month). This means that in one management zone with 5200km^2 and 37 rangers (10 teams) in Serengeti, if half this area is suitable for snaring, only 13.8% of the area can be effectively patrolled each month. To achieve 100% of this area effectively patrolled each month, it would require an increase of 28 patrol teams (i.e. 84 rangers) which is more than twice the number of rangers currently available and it is unlikely that this number could be achieved. This means that a desnaring strategy relying on the current number of rangers available in the parks will not effectively remove the snaring problem, rather rangers should be targeting to catch poachers to enhance poaching deterrence. On the other hand, even within game reserves with greater number of rangers, the detection was low (4% in Ikorongo) suggesting that preventing animal mortality will requires use of other multiplying effects to improve deterrence in these protected areas. Not conducting patrols frequently may risk the increase of illegal activities particularly when poachers perceive ranger activities to be low (Fischer et al., 2014). In the Serengeti National Park for example, cessation of anti-poaching patrols during the 1970s led to increased poaching pressure, reducing buffalo density by 90% (Dublin et al., 1990), with population recovery starting when anti-poaching resources were increased again after 1989 (Hilborn et al., 2006). Similarly, in some areas in Nyungwe National Park Rwanda, decline of snaring pressure was positively correlated with increased frequency of patrols (Moore et al., 2017).

Snare detection was higher in the game reserves than in national park probably because game reserves have higher investment in antipoaching resources (i.e. more rangers and patrol vehicles) and we observed more foot patrols than in a national park (N. Ngowi-Reserves Manager pers. communication, 2016). This finding may suggest that the antipoaching patrols are more effective in these reserves than in national park and supports our previous observation of few illegal activities in these reserves compared to the national park (Chapter 3) and are not poaching hotspot areas (Chapter 5). Despite this, however, the apparently low detection rates everywhere across the ecosystem (e.g. zero detection in east and central zones) is unlikely to act as significant disruptors to poacher activities, and suggests that ranger activity in these areas has perhaps displaced poachers from one part of the ecosystem. This process has previously been recorded elsewhere where illegal exploitation pressure on bordering unprotected areas peaks when law enforcement is increased inside protected areas (Ewers & Rodrigues, 2008). Such 'leakage' of poaching means effective patrolling in one area may currently have no overall impact on poaching rates across the ecosystem, and argues strongly for a coordinated ecosystem-level approach to tackling law enforcement. Besides patrolling, rangers in Serengeti National Park also guard hotels and tourist camps to ensure security for the tourists: as tourist operations have increased in recent years (Díez Gutiérrez et al., 2017), more rangers have been diverted to these activities potentially reducing the anti-poaching patrols in the area particularly during high tourism season (June to October) when poaching also appears to peak (Arcese et al., 1995).

I found eighteen mammal species at risk of being caught in snares (Figure 4. 6), including three species with declining populations in the Serengeti ecosystem: giraffe (Strauss et al., 2015), eland and buffalo (Metzger et al., 2010) which raises concerns over the sustainability of the illegal harvests. In the study, more zebra than wildebeest were at high risk of being caught in snares. This does not correlate with the current populations of wildebeest (about 1 million) and zebra (0.25 million) in the Serengeti ecosystem (Mduma et al., 1999). Also both are migratory species and share similar ecological requirements (Grange et al., 2004; Sinclair, 1995a), but instead there are more resident zebra than wildebeest in the woodland areas (Sinclair, 1995a), which may have increased its risks to catch in the dummy snares overall. Three species (buffalo, giraffe and eland) that were at risks of catch in our snares are also experiencing population decline suggesting that improving law enforcement effort is urgently needed in the Serengeti ecosystem (Metzger et al., 2010; Strauss et al., 2015).

Importantly, directly measuring ranger detection efficiency in the field is challenging. In situations where the number of rangers in the field is low (i.e. 1 ranger per approx.42 km^2 patrol area) and where poaching is high as is in Serengeti (Hilborn et al., 2006), a study design that integrates both ranger movement and search efficiency gives a more accurate picture of the overall effectiveness of desnaring activities. In this study I chose to measure detection efficiency based on routine ranger patrol activities within the PAs which avoided disrupting the rangers and also captured the actual situation in the study areas. I assumed that the recovered snares did not present any catch risk to the animals, and although I had requested rangers to record the status of a snare (i.e. loop open or snare intact) it was difficult to record these data. Consequently, we excluded these 2.8% of snares and assume they have very minimal impact on the estimates. Also, although I did not reveal information to the rangers until after snare setting was completed within any ranger zone it is possible that the ranger's knowledge that they were being studied may have increased their search effort in the field and consequently increased snare detection. The size of this effect on the detection probability reported in this study cannot be quantified without additional field data. However elsewhere in Ghanaian protected areas Jachmann (2008) reported increased detection of poaching activities when rangers were being monitored. Furthermore, estimated capture probability should be treated with caution because it was estimated over the three months of the study period; this may have lifted upward slightly as a result of the high population density of animal in the trapping zone during the study period. The superabundant wildebeest and zebra migrate widely across the ecosystem (Sinclair & Arcese, 1995), and this greatly reduces their risks to catch in snares during times when they are outside the areas with snares, though some resident species such as impala, Topi, waterbuck etc. may still be caught.

4.5.1. Conservation implications and future research on detectability

I found that snare detection rates from ranger patrols in the Serengeti Ecosystem are far from being adequate to achieve the primary goals of snare removal: reduction of animal mortality and disruption of poacher activities. Indeed, approximate calculations of the number of patrols needed to effectively cover the Serengeti ecosystem for desnaring demonstrates the futility of such a goal. However, current low detection rates likely increase the risk of animal poaching and may explain the widely spatial distribution of illegal activities observed in this ecosystem, with poachers avoiding the few higher detection areas and relocating to less-well patrolled areas. I found that snares were most likely to capture zebra, wildebeest, topi and impala, reflecting their relative abundance in the ecosystem (Grange et al., 2004; Mduma et al., 1999). The two areas with the highest snare detection, and with lowest overall levels of illegal activity have substantially more resource invested in law enforcement, suggesting that training and motivating additional rangers across the ecosystem could reduce poaching in the ecosystem rather than simply relocating it: previous ecosystem level improvements in patrol effort have resulted in increased populations of target species (Hilborn et al., 2006). More research is needed to explore effect of other ranger multiplying activities on poaching deterrence as the current law enforcement effort is unlikely to provide positive long-term conservation impacts in reducing the mortality of wildlife due to snaring.

4.6. Acknowledgements

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CHAPTER 5 - Poaching continues to threaten large mammal populations in the Serengeti ecosystem

A wildebeest severely injured by poachers in the Serengeti ecosystem

5.1. Abstract

Assessing the population dynamics of species faced with threats such as poaching is a conservation priority, yet many illegally hunted species in protected areas experience decline without detailed understanding of the role of different threats. Here I combine information of illegal offtake of four large wild mammal species in Serengeti ecosystem (i.e. Giraffe, Buffalo, Zebra and Wildebeest) and their demographic and environmental stochastic events to examine their vulnerability to population decline under different illegal offtake scenarios. Population models show high probabilities of declines in two large herbivores: giraffe and buffalo while zebra population remains relatively stable. In contrast, wildebeest numbers show increase despite hunting levels suggesting that its large population size and migratory behaviour helps this species to withstand current harvesting pressure. Further, current law enforcement effort detects negligible proportions of animal snaring, but detection rates correlate with overall number of snares: the primary tool used to kill animals in Serengeti; suggesting increasing effective ranger patrols can reduce poaching in this conservation landscape. Overall, current illegal harvests of 29,100 tons (300000 animals) per year of bushmeat for subsistence and trade threaten viability of several species, generating an existential threat to the function of the Serengeti ecosystem and the livelihoods of the local human populations dependent on wildlife. Swift improvements in funding conservation, protected areas management, anti-poaching strategies and rural development are needed to reduce poaching impacts.

Keywords: poaching, population viability, snare capture rates, wire snares, population vulnerability, Serengeti ecosystem

5.2. Introduction

The illegal harvest (poaching) of wildlife threatens 17.3 % of vertebrates (IUCN, 2017a; Ripple et al., 2016), and is implicated in the extinction of <300 vertebrate species since the 15th century (Dirzo et al., 2014; Milner-Gulland & Bennett, 2003; Ripple et al., 2016). Within protected areas, poaching of high value products such as ivory and rhino horn has received widespread attention but most illegal harvests are of lower value products (Ripple et al., 2015b). Bushmeat poaching silently drains animals from reserves (Dirzo et al., 2014; Harrison, 2011; Young et al., 2016) and undermines the ability of protected areas to protect the ecological services for which they were designated (Ripple et al., 2016). Further, many species faced with poaching pressure in most protected areas lack rigorous assessment of their population trends, potentially risking declines occurring unnoticed by PAs management authorities. Such assessments are important to inform conservation decisions to curb illegal hunting. Here, I combine surveys of wire snares, experimental estimates of snare detection and animal capture rates to estimate ecosystem-wide harvest levels and build population models to assess vulnerability of four most hunted species to population decline in the Serengeti ecosystem.

Assessing population vulnerability to decline due to poaching is challenging because illegal harvesting is often difficult to estimate due to methodological limitations as poaching is conducted in secrecy. This can be even more challenging in protected areas where illegal killing of animals is conducted silently using wire snares (Becker et al., 2013). Identifying the scale of the problem requires quantifying both the numbers of snares and the capture rates of those snares, and then estimating the rates at which animals are killed. Previous population assessments of wildebeest, buffalo and rhinoceros populations in Serengeti (e.g. Metzger et al. 2010) used indirect methods (i.e. hunter population estimates, or household surveys of local consumption) to derive hunting rates that were incorporated into population projection models. Although useful, such methods underestimate the full scale of poaching and because animal mortality by poaching on some species such as wildebeest is assumed to be of minimal effect (Mduma et al., 1998) managers may underestimate its impact on wildlife populations. Ignoring important poaching could lead to ineffective conservation decisions with negative consequences for species that are sensitive to hunting pressure due to inherent demographic factors such as slow growth and reproductive rates and small population size. In this chapter, I seek to assess vulnerability to population declines driven by poaching for four herbivores in Serengeti. To do this I (i) estimate the number of wire snares that are likely to be available within the protected Serengeti ecosystem and to illustrate hotspots of wire snaring and examine their relation to current law enforcement effort, (ii) estimate snare capture rates of animals and (iii) make projections of populations given estimates of poaching induced mortality.

5.3. Methods and Analysis

5.3.1. Spatial patterns of wire snaring in Serengeti

I collected poaching data along 88 transects of 4-14 km, a total of 920.25 km of walked transects in four protected areas (i.e. Serengeti National Park, Maswa, Ikorongo and Grumeti Game Reserves) in the Serengeti ecosystem (Chapter 3), recording the exact locations of wire snares over the period of two years (2015 & 2016). Along each transect, a team of three experienced (with formal training) national park rangers who also received three days training about the sampling methods prior to this fieldwork searched an 80 m strip, seeking all signs of illegal activities (e.g. wire snares and other signs not reported here) and recorded the precise location of each wire snare using a hand held GPS (Garmin eTrex 20).To understand environmental correlates of wire snaring, I used the online MODIS (product-MOD17A3) database to extract Net primary productivity (NPP) averaged over 15 years (2000-2014) within 500 m grids of the location of wire snares. Also, I estimated woody tree cover from an existing Serengeti vegetation map (Reed et al. 2009) and measured distances of wire snare location to nearest, rivers, villages and ranger stations from topographic shape files obtained from the Serengeti GIS databases (accessed from Tanzania National Parks (TANAPA) and Frankfurt Zoological Society- FZS office in Serengeti). To incorporate any deterrent effect of ranger stations, I identified the nearest ranger station within each management zone area and computed the distance. To obtain smooth interpolated maps of wire snares, I rasterised the raw survey data on the same 500m resolution grid used for each covariate, counted the number of snares found in each grid cell and computed the length of transect walked in each cell. Before analysis, I scaled and centred all covariates. To account for spatial autocorrelation, I fitted an intrinsic Conditional Autoregressive model (iCAR) (Besag & Kooperberg, 1995) using a queens case neighbourhood matrix. iCARs have been shown to perform well under similar conditions (Beale et al., 2014). I fitted spatially-explicit regression models (a zero-inflated Poisson model) using Integrated Nested Laplace Approximation -INLA (Rue et al., 2009), predicting the number of snares in all cells where no surveys were undertaken using the covariates and spatial random effects. To assess model fit, I used a leave-one-out cross-validation test, sequentially dropping data from each of the 88 transects in turn and comparing predicted numbers of snares against those actually observed. Because there was very strong spatial clustering of snares and strong spatial correlation in the residuals I expected predictions to average higher than observed at low densities and lower at high observed densities: the estimates are a long-term expectation of snare density, while observations are a single snapshot of a highly aggregated pattern. As a further check of the snare density map, I assessed whether the predicted hotspots of snaring correlate with locations where illegal killing is known to occur by overlaying the snare density estimates with the last known locations of GPS-collared individuals (wildebeest and zebra) that have been reportedly killed by poachers in the Serengeti ecosystem (Figure 5.1).

5.3.2. Snare detection

I measured snare detection by park rangers using a simulated poaching experiment with dummy wire snares set in three protected areas, the Serengeti National Park and Ikorongo and Grumeti Game Reserves (Chapter 4, Figure 5.1b). I used dummy snare detection probability to estimate the number of snares available in the protected areas (Figure 5.1c).

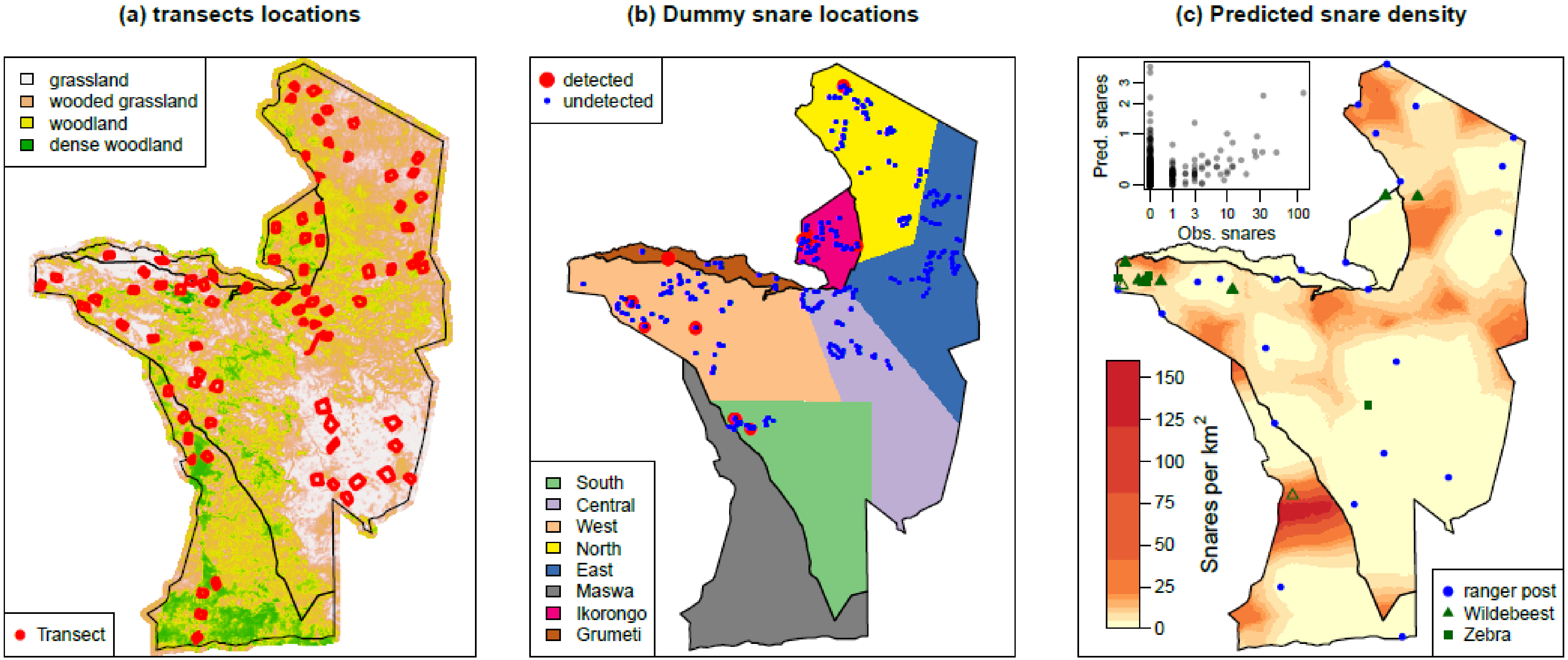

Figure 5. 1.The spatial distribution of wire snares in the Serengeti ecosystem (c) based on interpolation of wire snare data collected along 920.25 km of walked transects (a) and the location of dummy snare experiments (b) used to calculate snare detection and animal capture rates. The inset figure (c) shows fit of the snare prediction model while green squares and triangles are the overlaid locations of poached GPS-collared animals (based on 54 collared individuals) that appear to match well with the predicted snare hotspots in Serengeti, indicating usefulness of the predicted density map.

5.3.3. Snare estimates, capture rate and animals killed

To estimate animal capture I defined open grassland as <5% woody cover, a definition that includes >14% of the total study area because wire snares can only be set in areas where there are trees. I used two independent methods to estimate the number of wire snares available in the Serengeti ecosystem at any point in time. Firstly, the interpolation of the survey data above using zero inflated Poisson models and secondly, using the probability of finding dummy snares and the actual number of real poacher snares collected by the rangers during the period of the field experiments. Essentially, the number of snares for each ranger zone was calculated by dividing the number of poacher snares found in the zone by the detection probability over the study period, summing estimates for all zones to estimate the snare total for the Serengeti ecosystem. However, because two management zones recovered zero poacher snares during the experimental study period which may suggest limited patrols were conducted (indeed confirmed so by the rangers themselves), I used two methods to calculate the number of wire snares. Firstly, I analysed snares only for the zones with known non-zero snare returns and corrected the total estimate for the area of zones with unknown data. Secondly, because data on spatial distribution of poaching (Figure 5.1a&c, also see Figure 3.6 in Chapter 3) indicate higher poaching in the zones with missing data (Figure 5.2), I assumed snares density in these areas were twice the value in the known zones.

To understand how many animals may be killed in snares, I first estimated the probability of snares capturing an animal using both dummy snare and snare survey datasets. As before, I used two independent routes to calculate animal capture rates. For the dummy snares, I first computed the overall average proportion of snares (excluding those few snares recovered by rangers) that were disturbed (i.e. an animal entered and broke the dummy snare loop). Subtracting from one gives the overall probability of not catching an animal during the field experiment. The daily probability of capture can then be calculated using the equation in chapter 4 (Figure 5.3b). To estimate daily capture rate from snare survey data, I divided the number of live animals (live captured, n=2) encountered trapped in wire snares and also in combination with still trapped in wire snare but dead animals (fresh capture n = 8+2) found during fieldwork by the total number of poacher wire snares encountered during the survey period (Figure 5.3a). To estimate overall animals killed in wire snares, I resampled (i.e. probability sampling) the snare estimates and capture rates from the methods (Figure 5.3a&b) drawing 1000 samples to compute median estimate for capture rate and snare estimate which were then used to compute an annual estimate of overall harvest (Figure 5.3c). Secondly, to do this and to account for the spatial distribution of animals in the animal capture rates, (i) I computed density dependent capture rates based on density of animals in a cell (i.e. within animal density map) and estimates of capture rates above. (ii) I then multiplied this by snare densities which gives the number of deaths each day. (iii) From this, I checked whether the number of deaths is more than the number of animals in the cell, and if so, I recomputed using the number of animals present in that cell and removed animals to give (iv) a new map of animal density.

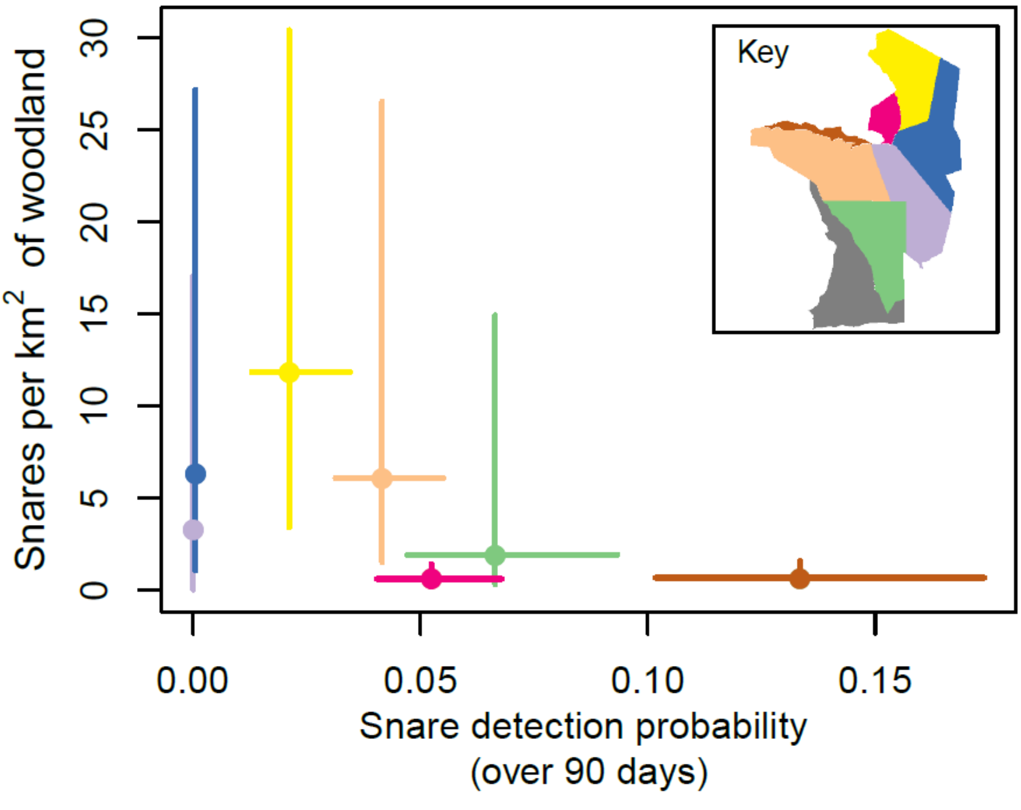

Figure 5. 2. Snare detection probability across different ranger management zones (inset map key) in the Serengeti ecosystem. High animal snaring (snare density) was associated with management zones where ranger detection of the dummy snares was low, suggesting improved anti-poaching strategy could greatly reduce wildlife mortality in this conservation landscape.

To get estimates of deaths per year, I repeated steps (i) to (iv) 365 times, each time using the generated new density map.

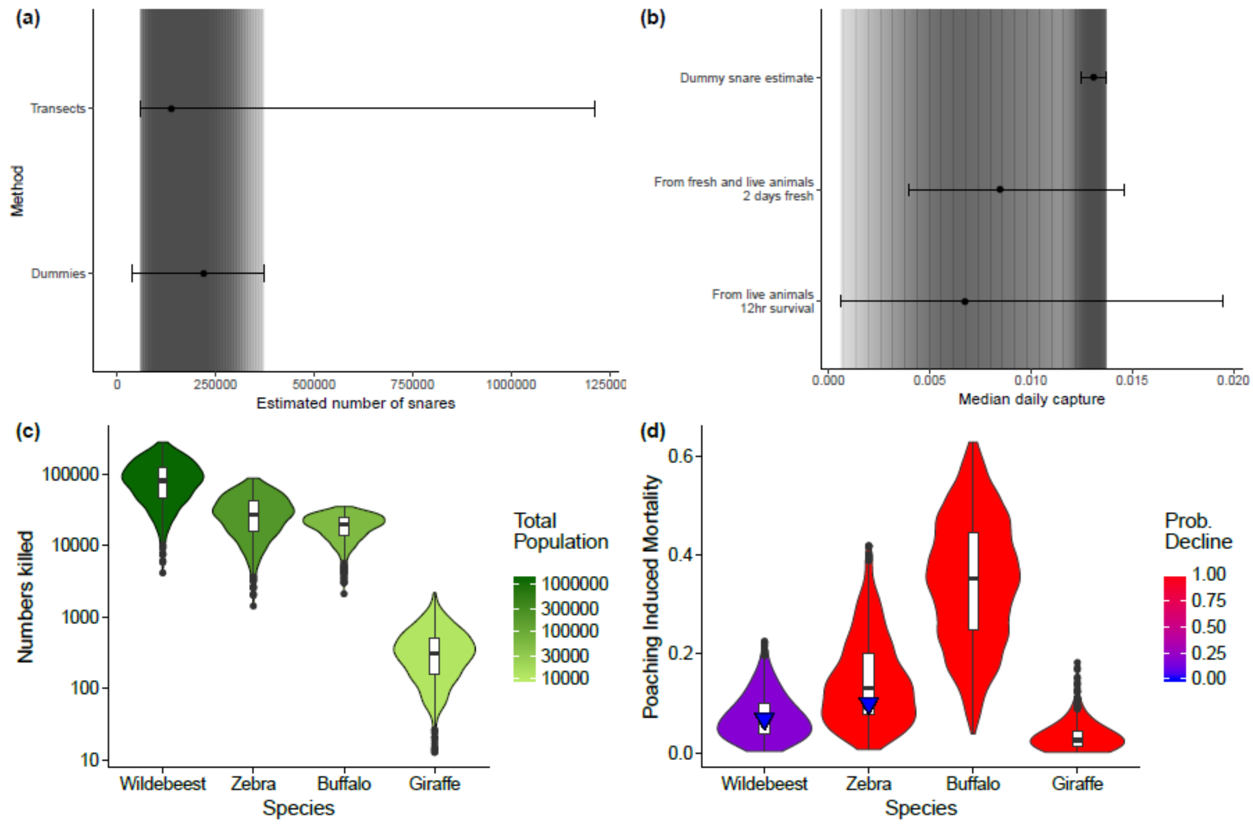

Figure 5. 3. Effects of wire snare poaching on wild mammal herbivore mortality in the Serengeti ecosystem indicating high vulnerability to decline of giraffe, buffalo and zebra (d) due to current illegal harvests of these animals (c) calculated from wire snares (a) and snare capture rates (b) estimated from the surveys (see text for details). The inset blue triangles in (d) is the mean poaching-related mortality of poached GPS-collared animals that match well with the probability of decline of these species. All estimates are expressed with 95% CL.

5.3.4. Modelling impact of poaching on harvested populations

To examine the effect of illegal harvest on wild herbivore population viability, I built stochastic, age-structured matrix models (Caswell, 2001; Tuljapurkar & Caswell, 1997) using the popbio- R package (Stubben & Milligan, 2007). I built three models for each of four species for which basic demographic data were available for Serengeti (Table S5.1, Table S5.2, Table S5.3, Table S5.4): the first model used natural conditions, assuming estimates of mortality for the population did not include poaching. Secondly, I built a model that assumed poaching offtake is completely compensatory: for every animal killed, natural mortality is reduced by one animal and thus annual mortality is estimated as the minimum of natural and poaching mortality. Thirdly, I built a model that assumed poaching induced mortality is additive: poaching mortality is added to natural mortality to generate an overall mortality rate. These two poaching scenarios bracket the best- and worse-case possibilities for poaching impacts on mortality, with the truth lying between. Using the first model I checked whether field data were plausible, expecting matrix models to increase in population. Using the second two models I estimated sensitivity to poaching. I simulated four herbivore (i.e. giraffe, African buffalo, zebra and wildebeest) population trajectories over various generation times (up to 40 years). Because some published species vital parameters (survival rates, birth rates, etc.) do not indicate variance around mean estimates for these species, I introduced 10% variability on these parameters (Mduma et al., 1998), creating normal distribution of the variance estimates for the modelling purpose and to account for the influence of environmental perturbation on the species demography (Kretzschmar et al., 2016; Mduma et al., 1998). For each species, I built deterministic age and stage-structured population matrix model using species demographic information and population size without harvest, running models for 10,000 iterations each to exploring the vulnerability of wild mammal herbivore populations to decline due to illegal hunting (Figure 5.3d).

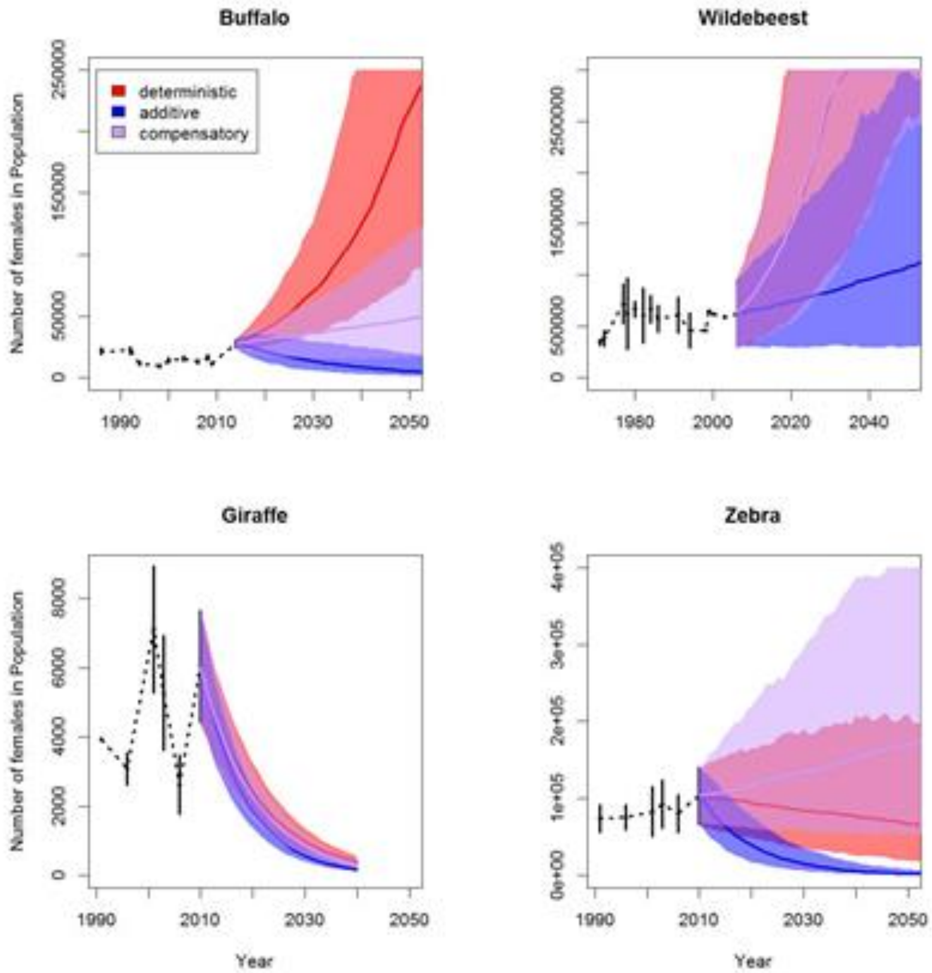

Figure 5. 4. Initial population projection exploring trends of four wild herbivore populations over three decades depicting effect of various scenarios of illegal harvest in Serengeti, i.e. no offtake (deterministic), when poaching is additive and when it is compensatory (see details in Methods). Population models for giraffe and zebra show decline even without imposed hunting pressure suggesting that their populations could already be limited by factors driving species reproduction and survival. The dotted lines are the population trends based on census surveys of these species plotted with 95%CL.

5.4. Results and Discussion

The Serengeti ecosystem is a conservation icon with a migration involving nearly 2 million ungulates (Sinclair & Arcese, 1995). Despite research suggesting bushmeat consumption around Serengeti is widespread, possibly accounting for up to c. 118,000 wildebeest per year (Campbell & Hofer, 1995; Rentsch & Packer, 2014), and that data showing local impacts of poaching on buffalo populations (Metzger et al., 2010), it is easy to assume that since law enforcement increased in the 1980s harvests have been largely sustainable (Mduma et al., 1998; Sinclair, 1995a). Although wildebeest numbers are indeed stable (Mduma et al., 1999; Sinclair & Arcese, 1995) which is consistent with my results that show increasing wildebeest population under most scenarios (Figure 5.4), this view may overlook impacts of poaching on other species for which Serengeti is internationally important. Moreover, current estimates of harvests stem from household consumption surveys that cannot capture information about bushmeat exported to other communities, nor do they report on likely non-target captures and uncollected captures decomposing in the wild (Becker et al., 2013; Lindsey et al., 2011; Newing, 2001; Noss, 1998). A systematic survey of the scale, impact and control of bushmeat harvests from Serengeti could provide important insights into the process of bushmeat harvesting in savannahs.

The challenge posed by illegal activities for effective conservation within protected areas is widely appreciated (Butchart et al., 2010; Hilborn et al., 2006), but the scale of snare poaching for bushmeat in the park and reserves in the Serengeti ecosystem (Arcese et al., 1995) has remained a subject of debate. Bushmeat consumption is widespread in local villages surrounding Serengeti and is presumed to be unsustainable (Hofer et al., 1996; Rentsch & Damon, 2013), however inside protected areas previous single species assessments report poaching has minimal impact on the key stone wildebeest (Mduma et al., 1998), although poaching reduced populations of African buffalo, elephant and rhino before these species recovered in the mid 1980s after law enforcement was improved (Hilborn et al., 2006). I argue that this debate has continued partly because of the difficulty with assessing the full size and impact of the animal snaring that risks continued defaunation inside protected areas (Young et al., 2016) and also due to the previous research were biased on bushmeat consumption in villages outside of Serengeti in the western corridor (Campbell & Hofer, 1995; Loibooki et al., 2002; Rentsch & Damon, 2013).

Here I estimated that a total of 137,226 (95% CI: 58,618-1,212,940) snares were present across the Serengeti ecosystem at any one time, almost all in the woodlands (only 6 snares (0.9%) were encountered in open wooded-grasslands). Snare densities in woodlands were lowest in management zones where rangers found more dummy snares, suggesting investment in ranger resources can locally reduce poaching activity despite apparently low detection rates. Detection rates of my dummy snares were remarkably low: over three months I estimate only 3.3% (CI: 2.5-4.3%) of snares would be discovered, but there were differences in detection rate between ranger management zones (Figure 5. 2). The low detection efficiency of snares implies that patrol effort targeted at snare removal alone is beyond the capacity of current rangers available in Serengeti (Hilborn et al., 2006; Sinclair, 1995b).

As an independent estimate of snare abundance, I combined the rate at which rangers detected dummy snares and the number of poacher snares collected by the rangers during the same period for 3 ranger zones where reliable numbers were available (21% of the total area) and extrapolated this to all areas assuming higher densities in poorly covered areas. Combining the two estimates, I calculate that between 58,600 and 371,500 wire snares are present across the Serengeti ecosystem (Figure 5. 3a). I used two different methods to estimate the rate at which snares capture animals: the number of live or freshly dead animals found in poacher snares during our surveys (and estimates of how long animals may survive in snares), and the rate at which the dummy snares were disturbed by animals (an overestimate of the actual capture rate). These computations suggested each snare has a 0.8 – 2.5% chance of capturing an animal every day (Figure 5. 3b). Combining snare density estimates, daily capture rates of each species group and maps of animal density from aerial surveys and GPS collar data allows to estimate overall numbers of animals poached from Serengeti each year as 300000 (CI: 109117- 403089) with zebra and large antelopes most frequently harvested (Figure 5.3c).

These capture rates imply likely annual mortality rates from poaching of 5.5-7% and 5.3-10.5% for zebra and wildebeest, which compare with estimates of 5.5-7% and 5.3-10.5% for confirmed or suspected losses to poaching among 54 GPS collared individuals (Fig 5.1c-green squares and triangles), most of which occurred in high snare density areas of the ecosystem. The wildebeest mortality rates are similar to previous findings reported by other authors in this ecosystem i.e. 10 % for wildebeest (Rentsch and Packer, 2014) while there are no published comparable estimates of zebra mortality due to poaching. Such high mortality rates obviously suggest populations may be susceptible to decline. Using stochastic matrix models, I confirmed high vulnerability of buffalo and giraffe populations to increased mortality from poaching, and that zebra population is relatively stable while the wildebeest population is currently not susceptible and show increasing population trend (Figure 5.3d), the difference presumably be due to greater sensitivity of resident than migrant wildlife.

My estimates of illegal offtake from Serengeti is larger than previously thought and suggests resident wildlife numbers such as those of Topi, for which Serengeti is internationally important may be supressed by current harvest, while zebra population dynamics may also be approaching tipping point. The current biomass harvest provides substantial economic incentives for continued poaching that earns an estimated equivalent of $9780 per poacher per annum. Reducing poaching in the Serengeti will require improving anti-poaching and management strategies notably, by increasing snare detection efficiency and intelligence-led ranger patrols, but also providing a viable livelihood strategy for the local communities to offset the significant economic impact of halting poaching in this conservation landscape.

5.5. Acknowledgements

I acknowledge and thank Drs Thomas Morrison and Grant Hopcraft for sharing data on movement of GPS-collared wildebeest and zebra that were poached in the Serengeti that I used to validate my snare density map. I also thank Dr Beale and Prof Chris Thomas for the comments on the modelling and calculations of the illegal kill estimates.

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Table S5.1. Zebra demographic data used in the analysis

Age	Male.surv	SE.malesurv	Fem.surv	SE.femsurv	Fe.birth	SE.fe.b.rate	Male rep	inter.b.intv	SE.int.b.interv	Year	Popsize	SE.popsize	Pop(95%CL)	Pop(95%UCL)
0	0.389	0.026	0.389	0.026	0.000	0.000	0	0.00	0.000	1991	147805	18380	111780.2	183829.8
1	0.847	0.057	0.847	0.057	0.000	0.000	0	0.00	0.000	1996	150834	16537	118421.5	183246.5
2	0.979	0.066	0.979	0.066	0.000	0.000	0	0.00	0.000	2001	166303	33368	100901.7	231704.3
3	0.954	0.064	0.954	0.064	0.686	0.046	1	1.08	0.072	2003	185434	31986	122741.4	248126.6
4	0.954	0.064	0.954	0.064	0.686	0.046	1	1.08	0.072	2006	161049	24748	112542.9	209555.1
5	0.875	0.059	0.875	0.059	0.686	0.046	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
6	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
7	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
8	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
9	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
10	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
11	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
12	0.875	0.059	0.875	0.059	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
13	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
14	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
15	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
16	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
17	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
18	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
19	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
20	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
21	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
22	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
23	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
24	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
25	0.768	0.051	0.768	0.051	0.883	0.059	1	1.08	0.072	2010	207166	37638	133395.5	280936.5
26	0.000	0.000	0.000	0.000	0.000	0.000	0	0.00		2010	0	0	0.0	0.0

Table S5.2. Wildebeest demographic data used in the analysis

Age	Male.surv	SE.malesurv	Fem.surv	SE.femsurv	Fe.birth	SE.fem.b.rate	Male rep	inter.b.intv	SE.int.b.interv	Year	Popsize	SE.popsize	Pop(95%CL)	Pop(95%UCL)
0	0.746	0.050	0.746	0.050	0	0	0	0	0	1971	692777	28825	636280	749274
1	0.885	0.059	0.885	0.059	0	0	0	0	0	1972	773014	76694	622693.8	923334.2
2	0.865	0.058	0.865	0.058	0.37	0.025	1	1.75	0.117	1977	1440000	200000	1048000	1832000
3	0.888	0.059	0.888	0.059	0.89	0.060	1	1.75	0.117	1978	1248934	354668	553784.7	1944083
4	0.888	0.059	0.888	0.059	0.89	0.060	1	1.75	0.117	1980	1337979	80000	1181179	1494779
5	0.888	0.059	0.888	0.059	0.89	0.060	1	1.75	0.117	1982	1208711	271935	675718.4	1741704
6	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	1984	1337879	138135	1067134	1608624
7	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	1986	1146340	133862	883970.5	1408710
8	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	1991	1221783	177240	874392.6	1569173
9	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	1994	917204	173632	576885.3	1257523
10	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	1998	923460	198959	533500.4	1313420
11	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	1999	1296944	300072	708802.9	1885085
12	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	2000	1245222	144934	961151.4	1529293
13	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	2003	1183966	128371	932358.8	1435573
14	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	2006	1239164	322017	608010.7	1870317
15	0.792	0.053	0.792	0.053	0.95	0.064	1	1.75	0.117	2006	1239164	322017	608010.7	1870317
16	0.78	0.052	0.78	0.052	0.95	0.064	1	1.75	0.117	2006	1239164	322017	608010.7	1870317
17	0.78	0.052	0.78	0.052	0.95	0.064	1	1.75	0.117	2006	1239164	322017	608010.7	1870317
18	0.78	0.052	0.78	0.052	0.95	0.064	1	1.75	0.117	2006	1239164	322017	608010.7	1870317
19	0.78	0.052	0.78	0.052	0.95	0.064	1	1.75	0.117	2006	1239164	322017	608010.7	1870317
20	0	0	0	0.000	0	0	0	0	0	2006	0	0	0	0

Table S5.3. Giraffe demographic data used in the analysis

Age	Male.surv	SE.malesurv	Fem.surv	SE.femsurv	Fe.birth	SE.fem.b.rate	Male rep	inter.b.intv	SE.int.b.interv	Year	Popsize	SE.popsize	Pop(95%CL)	pop(95%UCL)
0	0.478	0.022	0.478	0.021966	0	0	0	0	0	1991	7853	13	7827.52	7878.48
1	0.793	0.025	0.793	0.024552	0	0	0	0	0	1996	6166	485	5215.4	7116.6
2	0.793	0.025	0.793	0.024552	0	0	0	0	0	2001	14228	1866	10570.64	17885.36
3	0.870	0.031	0.870	0.031225	0	0	0	0	0	2003	10552	1678	7263.12	13840.88
4	0.870	0.031	0.870	0.031225	0.274	0.009	0	19.9	1.556	2006	5246	871	3538.84	6953.16
5	0.870	0.031	0.870	0.031225	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
6	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
7	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
8	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
9	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
10	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
11	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
12	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
13	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
14	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
15	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
16	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
17	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
18	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
19	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
20	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
21	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
22	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
23	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2

24	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
25	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
26	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
27	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
28	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
29	0.715	0.059	0.888	0.010665	0.274	0.009	1	19.9	1.556	2010	12078	1645	8853.8	15302.2
30	0	0	0	0	0	0	0	0	0	2010	0	0	0	0

The demographic data contained in these tables were collated from published literature: (Grange et al., 2004; IUCN, 2017b; D. E. Lee & Strauss, 2016; Mduma et al., 1999; Pellew, 1983; Sinclair, 1977)

Table S5.4. Buffalo demographic data used in the analysis

Age	Male.surv	SE.mal.surv	Fem.surv	SE.femsurv	Fe.birth	SE.fem.b.rate	male rep	inter.b.intv	SE.int.b.interv	Year	Popsize	SE.popsize	Pop(95%CL)	pop(95%UCL)
0	0.67	0.045	0.67	0.045	0	0	0	0	0	1986	43456	2911.55	37749.36	49162.64
1	0.86	0.058	0.86	0.058	0	0	0	0	0	1992	44246	2964.48	38435.62	50056.38
2	0.981	0.066	0.981	0.066	0	0	0	0	0	1994	23601	1581.27	20501.72	26700.28
3	0.971	0.065	0.971	0.065	0.2	0.013	1	1.5	0.101	1998	19156	1283.45	16640.43	21671.57
4	0.971	0.065	0.971	0.065	0.2	0.013	1	1.5	0.101	2000	28564	1913.79	24812.98	32315.02
5	0.971	0.065	0.971	0.065	0.2	0.013	1	1.5	0.101	2003	31026	2078.74	26951.67	35100.33
6	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2006	26001	1742.07	22586.55	29415.45
7	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2008	32919	2205.57	28596.08	37241.92
8	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2009	23041	1543.75	20015.26	26066.74
9	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
10	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
11	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
12	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
13	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
14	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
15	0.927	0.062	0.927	0.062	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
16	0.927	0.062	0.599	0.040	0.82	0.055	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
17	0.599	0.040	0.599	0.040	0.66	0.044	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
18	0.599	0.040	0.599	0.040	0.66	0.044	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
19	0.599	0.040	0.599	0.040	0.66	0.044	1	1.5	0.101	2014	55411	3712.54	48134.43	62687.57
20	0	0	0	0	0	0	0	0	0	2014	0	0	0	0

CHAPTER 6 - General Discussion

6.1. Summary of the thesis findings

In this thesis, I have addressed a major conservation concern; illegal activities in protected areas, focusing on the continuing loss of biodiversity across the globe. In the first chapter, I showed there was a gap in the understanding of the magnitude (i.e. quantity and distribution in space and time) of illegal activities in Protected Areas (PAs) and assessed which socio-economic and ecological factors influence population declines that arise from illegal activity pressures, both globally and regionally. My systematic assessment of the published literature (Chapter 2) found that illegal activities are commonplace in many PAs in Africa, Asia, America and Europe, contributing to the over-exploitation of some plants and animal populations ranging from invertebrates to mammals. I found that, at both the global and regional levels, illegal activities caused population decline mostly in species located in less-strictly protected areas (IUCN category IV-VI) of resource poor countries, and where illegal exploitation is undertaken for commercial purposes, rather than subsistence. The review revealed that most publications over the last four decades focused mostly on the prevalence of a single type (e.g., only poaching, illegal grazing or illegal plant extraction) of illegal activity, and hence there was little information on how the whole problem is currently being tackled. There was also an important lack of understanding of the actual size of this problem within individual protected areas, and the factors driving it; data that could help inform new conservation strategies to reduce poaching. I dealt with this problem in chapter 3 by conducting a detailed investigation in the Serengeti ecosystem encompassing four contiguous protected areas in Tanzania.

In the Serengeti, I found that animal poaching was widespread across the ecosystem, compared to other illegal activity types (i.e. illegal grazing and tree cutting). I showed that the distribution patterns of poaching in the Serengeti are driven by a hierarchy of decisions made by poachers from the landscape to the fine scale. At the large scale, these decisions are driven by proximity to rivers and permanent water bodies, distance to villages, and high net primary productivity (Critchlow et al. 2015, Watson et al. 2013, Chapter 3-this thesis). At the fine scale, these are driven by local habitat characteristics such as presence of water pools, paired-growing trees, high herb height and grass availability (Chapter 3). Poachers presumably make these decisions to optimize their hunting effort and to avoid being caught by the park guards. Further, I quantified wire snaring and estimated that 137000 (CI: 58600-371,500) wire snares could be available in the Serengeti on any one day, thus developing a first poaching risk map for this ecosystem, and revealing hotspots of wire snaring to help park rangers improve future anti-poaching patrols (Chapter 5). Importantly, in chapter 4, I simulated poaching using a field experiment, by setting out dummy snares, to assess ranger patrol efficiency in removing wire snares. I found that the current anti-poaching patrol effort in Serengeti is not sufficient to removing large quantities of wire snares set to catch animals due to substantially low snare detection rate (median 3.3% , CI: 2.5-4.3snares per day). I also revealed that more signs of illegal activities occur in the ranger zones which showed low snare detection probability. Ultimately, any potential strategies for tackling poaching must be justified in the context of its impacts on the target populations. Therefore, I asked in chapter 5 whether or not the current poaching rates are likely to impact the long-term survival of the target species. Pulling data of illegally killed wildlife (estimated using data from Chapter 3 & 4, i.e. 300000, CI: 109117- 403089) animals annually) and species demographics into population projection models, I demonstrated that the populations of some target ungulate species such as giraffe (*Giraffa camelopardalis*) and buffalo (*Syncerus caffer*) are likely more susceptible to current exploitation pressures, while the super abundant wildebeest (*Connochaetes taurinus*) and zebra (*Equus burchelli*) show robust population growth under current levels of exploitation, suggesting that other factors such as food availability for wildebeest (Mduma et al., 1999) and low survival rates and predation for zebra (Grange et al., 2004) may be regulating these populations in the Serengeti. Thus, illegal harvesting of the first two species, which are large, reproduce relatively slowly (Lee et al., 2016; Sinclair, 1977; Strauss et al., 2015), and are often associated with scrub and woodland (where poaching levels are often high), is not sustainable in the Serengeti ecosystem, whereas populations of the more rapidly-reproducing wildebeest, which spend much of the year on the open plains (where snare densities are low), are currently being harvested illegally at sustainable levels. However, the currently stable population growth of zebra which is constrained by predation pressure and low survival rate of foals (Grange et al., 2004) suggests that its population could easily decline if current poaching pressure is not abated.

My research has implications in the three main areas of conservation science; management strategy evaluation and conservation monitoring, sustainability, and management of wildlife crimes, and practical consequences for improvement of ranger antipoaching strategies and allocation of conservation resources in protected areas.

6.2. Management of protected areas under illegal activity threats

The spatially widespread illegal activities in PAs could be linked to three factors: relatively low funding for biodiversity conservation by national governments and the international community (Balmford & Whitten, 2003; Miller et al., 2013), limited conservation monitoring which leads to poor understanding of the extent and effect of illegal activities on species (Danielsen et al., 2005), and poor conservation strategies that do not adequately address the current poaching crisis (Watson. et al., 2016). Limited funding remains a major constraint. The Convention on Biological Diversity's Target 20 for year 2011-2020 strategic plans (CBD, 2011) emphasizes the importance and continued efforts from international organisations to increasing conservation funding to protect species, especially in poor developing countries where biodiversity is highest (Brooks et al., 2006). However, since the Rio summit in 1992, the flow of funds into the developing countries has not improved substantially (Miller et al., 2013), even though threats are mounting due to increasing demands for the burgeoning human populations and changes in consumption behaviour of people towards increased bushmeat consumption in response to improved wealth status (Godoy et al., 2010; Wilkie et al., 2005), potentially increasing species exploitation (McNamara et al., 2016).

Increased funding of conservation in the developing countries may help in two ways. First, it could help improve the livelihoods of the poor local communities through such projects addressing the social-economic constraints that influence illegal and unsustainable biological resource extraction. For example, protected areas reduce poverty in some local communities by 10% and 8% in Costa Rica and Thailand respectively (Andam et al., 2010; Sims, 2010). Second, it could help increase chances for budgets being set aside nationally for conservation, although that may depend on the political willingness of local governments (Wright & Winters, 2010). Furthermore, availability of conservation funds could increase grass root conservation efforts, attracting local communities into conservation business ventures, and establishing tourism-based wildlife conservancies, and outreach programmes to protect threatened species such as lions and rhino (Blackburn et al., 2016; Steinmetz et al., 2014). My findings from both global analysis and Serengeti ecosystem suggest that illegal activities are associated with low conservation resources. In the Serengeti ecosystem in particular, the higher illegal activities found in a low-resourced [i.e. Maswa Game Reserve (72.1 km^2/ranger area) and Serengeti National Park (43.5 km2/ranger area) both wholly state-funded PAs] than in relatively high-resourced PAs [i.e. Ikorongo and Grumeti Game Reserves (8.2 km2/ranger area) and jointly funded by state and private entity] supports this viewpoint and corroborate previous studies elsewhere that also identified weak conservation outcomes in developing countries with poor funding for PAs (Bruner et al., 2004; Mansourian & Dudley, 2008). Further, at the national and PA levels, the underfunding of protected areas and law enforcement sections could lead to fewer rangers recruited in the conservation sectors, fewer patrol resources (e.g. vehicles, salaries, field allowances and other equipment and communication resources) and potentially low morale on the personnel working at the forefront of fighting illegal activities (Sinclair, 1995b). Weak law enforcement, as a consequence of these limitations, may fail to deter illegal activities, hence increasing impacts on the wildlife populations (Hilborn et al., 2006).

The patterns of illegal activities in Serengeti ecosystem also suggest ineffective ranger-based monitoring and allocation of conservation resources in the PAs. Although ranger-based monitoring exists in Serengeti and many other PAs globally, but they are static (i.e. use fixed routes or same method such as patrolling from vehicles etc. which are easy for poachers to learn) and are rarely based on strategic allocations of effort aided by appropriate monitoring (Brashares & Sam, 2005). Further, even in many protected areas where patrols are conducted, they are rarely recorded to inform future management actions (Danielsen et al., 2005), although some monitoring tools such as SMART exist but only few countries (currently 46) have adopted SMART in protected areas. The collection of such information using appropriate monitoring system for illegal activities could help improve efficient allocation of existing conservation resources to target illegal activities more effectively (Dhanjal-Adams et al., 2015; Plumptre et al., 2014) and improve patrol efficiency (Critchlow et al., 2016). Increased risk of detection may also deter poachers, and therefore reduce poaching-associated animal mortality by more than the number of snares found or poachers caught. Besides improving future anti-poaching patrols, long-term spatial data of illegal activities could help detect changes to the ecosystem and its threats, including monitoring deterioration or changes in populations, perhaps identifying the impacts of drivers such as illegal livestock grazing or climate change (Beale et al., 2013; Kideghesho et al., 2013). Although good monitoring systems are expensive, and this may limit implementation in developing country protected areas (Danielsen et al., 2005), the methods I used in chapter 3 and appropriate ranger-based monitoring of illegal activities in the Serengeti ecosystem could easily be implemented more widely without substantial additional funds. This would be easiest in the PAs where patrolling systems are already working. Standardized surveys could be incorporated into the existing anti-poaching plans, where PAs could collect standardised and robust data about illegal activity distribution more regularly through walking randomized standard transects in PAs. Such long-term data could be useful in understanding the spatial and temporal trends in illegal activities, thus helping to prioritize and re-direct anti-poaching efforts to areas where effectiveness could be maximised (Critchlow et al., 2016). Because the locations of poaching can be predicted (Chapters 3&5), my structured survey method could be adopted for use in monitoring illegal activities in PAs that currently lack ranger-based monitoring or where the monitoring of illegal activities is unstructured (such as in the Spatial Monitoring and Reporting Tool, SMART, which is currently being used). Compared to SMART, my stratified randomised transects survey method (but with increased concentration of efforts in strata with high levels of poaching, and where populations of poaching-susceptible species exist) is easier for field rangers to collect data on patrols, and provides high quality data that are relatively easy to analyse. Further, in PAs with existing SMART monitoring (i.e. currently only 389 sites in 46 countries; (SMART, 2016), adoption of my structured survey method could provide a great opportunity to test the efficiency of these methods particularly on the resources (i.e. rangers, time, finances, analytical skills need, etc.) needed to collect similar data on illegal activities.

Well-coordinated systems of managing protected areas could improve conservation effectiveness both in small, isolated and large inter-connected reserves. My results revealed relatively high (but still low) snare detection rates in the joint-managed PAs than in the state-run PA, and this may partly reflect limited conservation resources. However, it may also suggest that coordination of the existing conservation effort in the Serengeti ecosystem could be improved. Any such potential improvements in strategy should be addressed because all the PAs strive to protect same animals, which move seasonally across the ecosystem. Improving anti-poaching programs only in some PAs (e.g. Ikorongo and Grumeti - as it stands now) may not have long-term positive impacts on the Serengeti wildlife populations, as a whole, if the same individual animals are subject to increased poaching levels at some periods of the year, elsewhere in the Serengeti system. Due to potential displacement of poachers in well patrolled areas, improving anti-poaching only in these reserves could simply shift poaching pressure into the Maswa and Serengeti National parks, where fewer resources may be available for conservation (Ewers & Rodrigues, 2008). A collaborative ecosystem conservation approach that ensures shared conservation resources (e.g. vehicles, ranger training, etc.) between the various PAs may be necessary to secure long-term conservation effectiveness in the Serengeti ecosystem.

6.3. Sustainability of poaching and the wildlife populations under increasing human consumption demands in the Serengeti

Current hunting levels (Chapter 5) in the Serengeti study area cannot realistically be driven by the subsistence use alone, and suggest that commercial bushmeat poaching is likely to be a valuable economic activity in the region. An important question is the extent to which illegal hunting is sustainable; i.e. whether the wildlife populations under exploitation can persist (at certain levels of abundance) and whether hunting is generating a renewable protein source that would be sufficient for the ever-growing local human population in the area (or traded further afield). To explore these questions, it is important to consider how the illegal hunting business may be benefitting the perpetrators of poaching economically, and its contribution to the existing protein sources in the region. The economic pull of illegal poaching will depend on the benefits (financial and through exchange for other goods and services) that accrue, the perceived risk of detection and severity of punishment, and how the economic benefits compare with those that the same individuals would be able to obtain from potential alternative livelihoods and economic activities. The latter may be or may not be sufficient incentives for them to stop poaching. The price of bushmeat in a local black market in the local communities surrounding the Serengeti ecosystem is on average \$0.8 per kilogram of fresh meat (Rentsch & Damon, 2013); but this price can increase to \$2 for an equivalent weight of dried bushmeat (Rija, AA. 2015 personal field observation). Fresh bushmeat is cheaper than most alternative animal protein sources (such as chicken (\$2.1), beef (\$1.4), goat (\$1.5), sheep (\$1.5), or the sardine-like fish, Dagaa (\$1.0); all values per kg, fresh weight), and makes up 33% of all consumed proteins sources per week per household in the local communities in Serengeti (Rentsch & Damon, 2013). Bushmeat demand for a household per week costs \$2.2, which represents an average of 25% of all animal protein sources used by a household per week; this is two to eight times higher than for other animal protein alternatives (Ndibalema & Songorwa, 2008; Rentsch & Damon, 2013). This reflects high demand for protein from the bushmeat in these communities, and the demand could potentially increase with the growing human population (Estes et al., 2012; Sinclair, 1995b). What does this protein consumption pattern mean in the context of illegal hunting pressure in Serengeti?

To put this into perspective, it is important to consider how trap effort can drive dynamics of bushmeat poaching and potentially the economy of illegal hunters. In our experience, about 50 snares could be set by a poacher in a day or during a hunting trip in Serengeti (Chapter 3&4). A group of two poachers for example, conducting a two-week hunting trip (14 days), once each month, can set an average of 100 snares in the PAs. With a daily animal capture rate of 0.015 (Chapter 4), they could catch 21 animals (i.e. 14*0.015*100). Because snares are non-selective and we know various species could be caught (Chapter 4), each with an average weighted biomass of 97 kg (Table 6. 1), in one hunting trip, poachers could generate US\$1630. This amounts to US\$9780 per hunter per annum. Calculating total incomes based on the total annual offtakes from this ecosystem (300000 animals per year or 29100 tons of bushmeat, see Chapter 5) means poachers could generate about US\$23,280,000 each year (i.e. 0.8*300000*97). This sum is far higher than any alternative income generating opportunity available for the local people (Fischer et al., 2014; Moro et al., 2013). Indeed, it represents roughly half (or may equal) the amount collected in park gates from tourists visiting the Serengeti. This suggests that it will be difficult or impossible to prevent poaching in the Serengeti until alternative projects that generate equivalent income are established. A combined strategy is likely to be required that: increases the perception of risk and punishment, provides (or facilitates) alternative sources of income, educates communities to avoid poaching-sensitive species (as far as if possible within the indiscriminate constraints imposed by snaring), and undermines the price advantage of bushmeat, relative to meat from domestic animals and fish. Can reductions in the costs of the alternative protein sources provide an answer to stopping the illegal hunting? There is no easy answer to this question as poaching is not driven by the provision of proteins alone.

Assuming that beef could be an alternative meat protein source in the local communities, and assuming all the bushmeat harvested is consumed locally, then an equivalent 29100 tons of beef would be needed each year to feed *ca.* one million people living around the Serengeti ecosystem (Sinclair et al., 2008). This means that significant improvements in cattle husbandry would be needed to produce 72750 beef cattle each year (each of 400 kg weight) to meet the beef proteins needed in these communities. This would imply that more land would be required for grazing cattle. This then requires us to ask whether the increased demand for grazing land could also be accommodated inside the protected areas? What would be the sustainability of this model and impacts to the habitats and wildlife populations in Serengeti, if applied? These pertinent questions are beyond the scope of this thesis, but require further assessment if a long-term reduction in poaching pressure is to be achieved.

This approach could be successful, given that models of alternative protein sources (e.g. (Rentsch & Damon, 2013), suggest that decreasing the price of fish and beef would be likely to reduce the quantity of bushmeat demanded by the human population. However, these authors and others (e.g. (Kaltenborn et al., 2005; Moro et al., 2013) assume that the volume of bushmeat harvested (i.e. 29100 tons per year) is wholly consumed by the hunters and local communities, ignoring the likelihood that some of this quantity will be traded in neighbouring cities or beyond Tanzania's borders. This is yet to be quantified in the context of hunted bushmeat in Serengeti, but the challenge is clearly a very different one if the price of beef (and other protein sources) has to be reduced to a level that is lower than that of bushmeat across the entire African continent (even then, poaching may still be an attractive option at lower bushmeat prices for a substantial fraction of the population). If bushmeat is a necessary protein source within these communities as previous research suggests e.g. Rentsch and Damon (2013), then it would require a model that will provide opportunities for the management authorities for these protected areas to harvest the equivalent amount of bushmeat proteins required by the local communities and sell to them. This could earn the PAs extra source of revenue (US$23,280,000 each year), potentially employ former poachers, and the revenues could help fund conservation activities in these PAs. It would also be possible to ensure that the harvest only exploited those species for which a sustainable yield could be obtained without generating population declines of the harvested species. Even such a model will bring more questions than solutions to the poaching problem. For example, how could the current wildlife conservation policy and conservation laws (URT, 1998, 2009) which prohibit consumptive use in National Park reconcile with such a legal harvesting model? Even within the Ikorongo and Grumeti Game Reserves where regulated hunting is permitted, a previous study shows that commercialization of bushmeat had failed before (Holmern et al., 2002). In 1993, the Tanzania's Department of Wildlife (WD), through the Serengeti Regional Conservation Project (SRCP) launched commercial utilization of bushmeat through legal game cropping in Grumeti and Ikorongo reserves and Ikona wildlife management area (WMA), where the wild meat was sold to the local communities in the villages within the Serengeti's western corridor in order to provide incentives for conservation and reduce poaching pressure. This business proved economically unrealistic, collapsed a decade later due to increased poaching pressure that reduced wildlife population in the hunting areas and because poaching earned more benefits to the villagers than the meat they are sold by SRCP (Dublin et al., 1990; Holmern et al., 2002). The experience from this hunting business may probably make any consideration of such models unwelcome. Even when considered, how sustainable would this model be in terms of providing sufficient quantities of bushmeat proteins, and sustaining wildlife populations, to the ever-growing human population around Serengeti (Estes et al., 2012)? How could such a model ensure sustainability of the ungulate populations under future land use change and climate change impacts? This thesis does not provide answers to these questions. Rather, it emphasizes that future research is necessary to further evaluate the many options for tackling poaching in the Serengeti ecosystem.

Table 6.1. Mean weighted average biomass of animals illegally harvested in the Serengeti ecosystem each year. Species body mass were collated from herbivores database (see also, Chapter 2), percentage usable meat collated from published literature in East and Southern Africa (Blumenschine & Caro, 1986; Marks, 1973; Ndibalema & Songorwa, 2008). Average weighted biomass was calculated as the sum of the product of proportion of each species recorded illegally killed in the surveys (Chapter 3) and its dressed carcass weight collated from the citations above.

Exploited Species	Proportion caught	Species body weight(kg)	Percentage useable	dressed carcass weight (kg)	Ave. weighted biomass (Kg)
Buffalo	0.021	592.6	0.65	385.19	8.166
Bushbuck	0.001	43.2	0.65	28.08	0.026
Dikdik	0.006	4.8	0.85	4.08	0.023
Eland	0.005	562.3	0.8	449.84	2.074
Giraffe	0.005	964.6	0.73	704.158	3.570
Grant's gazelle	0.001	42.5	0.75	31.875	0.044
Hartebeest	0.001	160.9	0.75	120.675	0.167
Hippo	0.001	1536.3	0.65	998.595	0.922
Hyena	0.002	63.3	0.4	25.32	0.057
Impala	0.020	52.5	0.7	36.75	0.728
Reedbuck	0.000	43.2	0.8	34.56	0.016
T. gazelle	0.001	22.5	0.8	18	0.025
Topi	0.021	127.1	0.75	95.325	1.980
Warthog	0.006	75.6	0.65	49.14	0.317
Water buck	0.002	204.3	0.75	153.225	0.282
Wildebeest	0.331	198.6	0.85	168.81	55.944
Zebra	0.096	279.1	0.85	237.235	22.779
TOTAL weighted average biomass					**97.119**

Summing up this, if these simple bushmeat economics are near correct, and given the high consumption of bushmeat protein over alternative sources in the local human populations, the alternative livelihoods approach is very unlikely to solve the problem, at least for commercial poachers. It is worth noting that the bushmeat economy involves different types of poachers (subsistence, commercial), who conduct animal snaring in PAs, middlemen who purchase bushmeat from the hunters (and who may sell to other middlemen when the meat is transported to cities and abroad), and traders who sell bushmeat to ultimate consumers. Further individuals supply wire snares, motorcycles etc. to the hunters for shared benefits. The producers of hunting equipment – hunters – middlemen – traders operate as an economic system, yet it is often perceived that it is only the arrested hunters who are the major drivers of poaching dynamics in protected areas. Designing policies to tackling poaching should take these various groups into account to achieve better conservation effectiveness.

6.4. Wildlife harvest in Serengeti and the illegal wildlife trade

Trade in wildlife products is an increasing conservation problem globally (Cooney et al., 2017; Douglas & Alie, 2014; Lavorgna, 2014). It impacts individual species directly, it alters ecosystems and the wider range of biodiversity they contain, and it also hinders the social and economic development in many communities (Sollund, 2011; Warchol, 2004). Traded species are being over-exploited to extinction (e.g., rhinos are extinct in most protected areas of East Africa and globally) or severe decline (e.g. ivory trade has exterminated elephant from over 60% from its range in Africa (Maisels et al., 2013; Wasser et al., 2015; Wittemyer et al., 2014)). The high-value products (horn, ivory) of these charismatic species are illegally traded internationally, and the plight of these animals captures national and international attention, and hence attracts funding (which is required to prevent further declines and extinction). Illegal bushmeat consumption, on the other hand, is the silent threat to many of the less iconic species, such as buffalo, zebra, impala etc. (Rija AA, this thesis). This challenge attracts far less attention. We do not know what proportion of the bushmeat harvested in the Serengeti (i.e. 300000 animals annually) is consumed locally or traded within and beyond Tanzania's borders, but a recent study suggests that bushmeat trade is a growing problem across the East and southern African region (Lindsey et al., 2013b). This suggests that an integrated regional approach may be required (i.e. taking actions solely in the Serengeti region may not be sufficient) to curb illegal trade that drives poaching in protected areas. The illegal wildlife market (live animals and body parts) is a high profit business estimated at 8 to 10 billion dollars per year (Warchol, 2004) and some of this profit has been linked to increasing conflicts in some regions by financing insurgence forces (Dalberg, 2012; Douglas & Alie, 2014; IFAW, 2008), though more recent studies dispute these claims as being made largely based on political motives rather but substantive evidence (Duffy, 2016; White, 2014). Further, many globally threatening diseases such as Ebola fever (Leroy et al., 2004) and SARS- associated coronavirus (Bell et al., 2004) have been ascribed to the human consumption of contaminated primate bushmeat and illegal live trade in wild carnivores respectively. These diseases pose significant global health concern and cost countries several billion dollars annually (Karesh et al., 2005). Illegal wildlife trade is an increasing threat to both nature and humanity; tackling it is a growing priority (Sánchez-Mercado et al., 2016).

The effective targeting of the illegal wildlife trade is likely to require a multi-pronged approach (Challender & MacMillan, 2014). This will include: improving anti-poaching effectiveness, e.g. boost local patrols and intelligence based strategies to enhance detection to prevent the supply side (McNamara et al., 2016); improving long-term monitoring of illegal harvests to generate data to inform the improvement of anti-poaching strategies (Rija, A.A., this thesis); integrating other locally-based methods such as investing in community-led conservation initiatives that increase local community willingness to conserve the wildlife (Blackburn et al., 2016; Naidoo et al., 2016); and more generally developing approaches to increase benefits of wildlife to the local communities (Cooney et al., 2017). Further, more research effort into the attributes involved at the supply and transportation chain such as understanding the supply chain of wire snares used in poaching (e.g. their sources, transportation & production points including craftsmen) and tracing how bushmeat harvested from Serengeti enters the international market will improve our capacity to fight the illegal wildlife market. Increasing the availability and reducing the costs of culturally acceptable alternative sources of protein, improving income sources of poor communities and promoting sustainable resource use are all likely to be necessary in reducing poaching in threatened protected areas (Moro et al., 2013).

6.5. Conclusions and future research on illegal activities

Illegal activities are a critical threat to the survival of wildlife and the functionality of protected areas and are a key driver of the global illegal wildlife trade. The drivers of illegal activities within protected areas are many and intertwined, thus their tackling requires a multi-layered approach and collaborative conservation efforts between the local communities (including PAs and the surrounding human population), governments and international conservation agencies. On a broad course scale, funding for biodiversity conservation in protected areas needs to be sustained to ensure any long-term biodiversity conservation impacts. At a fine-grained scale within protected areas, anti-poaching patrol strategies require improvement to achieving high detection of infraction and the prevention of poaching. In the Serengeti ecosystem, the spatial maps of wire snares and other illegal activities represent a novel contribution of this work toward improving current anti-poaching patrols. Any comprehensive strategy to reducing poaching and the illegal wildlife trade will need further understanding of: (i) fine-scale snaring rates by poachers in order to improve intelligence-led conservation, (i) how the ranger traits (e.g. education, field experience, age, level of co-ordination etc.) influence snare detection, (iii) how ranger-antipoaching activities deter or displace illegal activities in protected areas, and (iv) economic studies of the wire snare and bushmeat trades in the Serengeti ecosystem and international trade networks from this source.

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