THE CHALLENGES AND OPPORTUNITIES IN CONSERVING WIDE-RANGING CROSS-BORDER SPECIES: A CASE STUDY OF THE GREATER MAPUNGUBWE TRANSFRONTIER CONSERVATION AREA ELEPHANT POPULATION

by

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ABSTRACT

Transfrontier conservation areas potentially play a key role in conserving biodiversity and promoting socioeconomic development. However, socio-political factors often affect their effectiveness in achieving biodiversity conservation and sustainable development objectives. Following a transdisciplinary approach, I assessed the challenges and opportunities in conserving and managing the African elephant (*Loxodonta africana*) population within the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA) in Botswana, South Africa and Zimbabwe, southern Africa.

The results showed that the current rate of offtake of bull elephant in the GMTFCA is unsustainable. At current rates of hunting, in fact, trophy bulls were predicted to disappear from the population in less than 10 years. Elephant densities were higher in South Africa and Botswana where the gross domestic product is higher. In addition, elephant densities were higher at sites where the proportion of agricultural land around them was the lowest and where vegetation productivity was the highest. Trophy hunting, as well as other localised human activities, also affected the distribution of elephant within sites, forcing them to trade-off between disturbance avoidance and the availability of food and water. While at the international level, a significant body of law and policy relevant to elephant conservation exists, I found that there was little cooperation among Botswana, South Africa and Zimbabwe, and a lack of implementation of these provisions on a national and trilateral level.

Overall, this study confirmed that poverty was an important factor affecting elephant abundance at the country level, but highlighted that, at the site level, anthropogenic disturbance played a crucial role. A revision of the current hunting quotas within each country and the establishment of a single multi-jurisdictional (cross-border) management authority regulating the hunting of elephant is needed. Further, to reduce the impact of increasing human populations and agricultural expansion, the development of coordinated legislation and policies to improve land use planning, and the development of conservation corridors to link current protected areas, is needed.

The issues regarding the management of this elephant population illustrate the significant challenges involved in achieving a comprehensive, consistent and effective implementation of a transboundary population approach. Southern African countries make an important contribution to elephant conservation and could soon become the last stronghold of elephant conservation in sub-Saharan Africa. Therefore, immediate actions are needed to reduce pressures from human activities in order to enhance the long-term persistence of the species.

Keywords: anthropogenic disturbances; legal framework; *Loxodonta africana*; protected area management; socioeconomics; sustainable utilisation; transdisciplinary approach; trophy hunting

PREFACE

The work described in this thesis was carried out in the Greater Mapungubwe Transfrontier Conservation Area, through the Central Limpopo River Valley Elephant Research project, NOTUGRE and the School of Life Sciences, University of KwaZulu-Natal, Westville Campus, Durban, from January 2000 to July 2015, under the supervision of Professor Rob Slotow (University of KwaZulu-Natal), Mr. Bruce Page (University of KwaZulu-Natal) and Dr. Enrico Di Minin (University of Helsinki).

The author of the thesis collected data in the field from January 2000 to December 2012. These studies represent original work by the author and have not otherwise been submitted in any form for any degree or diploma to any tertiary institution. Where use has been made of the work of others it is duly acknowledged in the text.

Sarah-Anne Jeanetta Selier

14 December 2015

DECLARATION 1 - PLAGIARISM

I, Sarah-Anne Jeanetta Selier declare that:

- The research reported in this thesis, except where otherwise indicated, is my original research.
- This thesis has not been submitted for any degree or examination at any other university.
- This thesis does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged as being sourced from other persons.
- This thesis does not contain other persons' writing, unless specifically acknowledged as being sourced from other researchers. Where other written sources have been quoted, then:
- Their words have been re-written but the general information attributed to them has been referenced
- Where their exact words have been used, then their writing has been placed in italics and inside quotation marks, and referenced.
- This thesis does not contain text, graphics or tables copied and pasted from the Internet, unless specifically acknowledged, and the source being detailed in the thesis and in the References sections.
- Dr A. Trouwborst wrote the first draft of section 5.2 (The transboundary population approach) of chapter 5.

Signed:

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Sarah-Anne Jeanetta Selier

DECLARATION 2 - PUBLICATIONS

DETAILS OF CONTRIBUTION TO PUBLICATIONS that from part and/or include research presented in this thesis (include publications in preparation, submitted, *in press* and published and give details of the contributions of each author to the experimental work and writing of each publication).

Publication 1

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Author contributions:

JS conducted all fieldwork, processed and analysed the data, and designed and wrote the paper. AV and BP contributed to the analyses and provided valuable comments on the manuscript. BP and RS contributed to the design of the paper and provided valuable comments on the manuscript.

Publication 2

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JS conducted all fieldwork, processed and analysed the data, and designed and wrote the paper. EDM contributed to the analyses and provided valuable comments on the manuscript. EDM and RS contributed to the design of the paper and provided valuable comments on the manuscript.

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Author contributions:

JS conducted all fieldwork, processed and analysed the data, and designed and wrote the paper. EDM contributed to the analyses and provided valuable comments on the manuscript. EDM and RS contributed to the design of the paper and provided valuable comments on the manuscript.

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Author contributions:

JS conducted all fieldwork, processed and analysed the data, and designed and wrote the paper. AT contributed to the design of the paper and wrote section 5.2 (The transboundary population approach). AB, RS and AT provided valuable comments on the manuscript.

Signed:

Sarah-Anne Jeanetta Selier

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CHAPTER 1: GENERAL INTRODUCTION

1.1 INTRODUCTION

Current rates of biodiversity loss are unprecedented and on-going (Butchart et al. 2010). This global biodiversity crisis is a response to several human-induced changes in the global environment (Sala et al. 2000). The main threats to biodiversity are human population explosion and over exploitation of natural resources (Vitousek et al. 1997, Foley et al. 2005), human development and subsequent land use changes (Vitousek et al. 1997, Foley et al. 2005), invasive species (Gurevitch and Padilla 2004, Didham et al. 2005), climate change, and global warming (Thomas et al. 2004, Botkin et al. 2007). Of these threats, land use change is likely to have the largest effect on terrestrial ecosystems (Sala et al. 2000, Foley et al. 2005). More than a third of the global land surface has already been transformed by human action (Vitousek et al. 1997). This transformation is ongoing and bound to increase with greater demands from an increasing global human population (Vitousek et al. 1997). Globally, it is expected that terrestrial vertebrates could lose, on average, approximately 12 – 16% of their current effective range by 2040 (Montesino Pouzols et al. 2014). Increasing human populations have led to an increased demand to develop land for human settlements, crops and livestock, and other ecosystem goods essential for human sustenance, subsequently reducing the size of natural ecosystems and impairing their integrity (DeFries et al. 2007, Butchart et al. 2010, Yackulic et al. 2011).

The magnitude of negative impacts of anthropogenic activities on global biodiversity has been documented at several levels of biological organisation (Gaston et al. 2003). Land use changes have not only caused irreversible losses of biodiversity (Vitousek et al. 1997), but have also reduced the carrying capacity of the environment in terms of the numbers of organisms that it can sustain (Donald et al. 2001). This leads to further habitat loss, fragmentation and other anthropogenic pressures such as the unsustainable exploitation of natural resources (Rodrigues et al. 2004, Butchart et al. 2010, Rands et al. 2010). It is estimated that the current rates of species extinction due to human actions is 1000 times greater than background or natural rates of extinction (Pimm and Raven 2000).

The development of protected areas has been used as a key strategy to reduce the loss of biodiversity and to conserve species globally (DeFries et al. 2007). This strategy, however, has had limited success in the developing world, where most of the world’s biodiversity is located (Montesino Pouzols et al. 2014). Human populations in the developing world and particularly Africa, continue to grow rapidly (United Nations 2013). As a result, these countries often find it difficult to maintain their biodiversity due to increasing land use conflicts and insufficient funds for conservation (Krug 2001; Leader-Williams 2005). Even though protected areas are supposed to reduce the risk of extinctions of endangered species, most developing countries do not have the resources to protect large areas, and, subsequently, protect economically valuable species from illegal exploitation (Leader-Williams and Albon 1988, Leader-Williams and Hutton 2005). Many species’ ranges are now restricted to protected areas (Newmark 2008, Karanth et al. 2010a), and within these areas the abundance of large mammals have declined by 59% between 1970 and 2005 (Craigie et al. 2010). The overarching causes of this decline is thought to be over-hunting and habitat conversion, both driven by rapid growth in human population and resource consumption (Baillie et al. 2004, Caro and Scholte 2007). Conservation agencies thus face the challenge of managing trade-offs between the immediate needs of humans, and maintaining the capacity of the biosphere to provide goods and services in the long term (Foley et al. 2005).

While protected areas are critical necessities for biodiversity persistence in increasingly human-dominated landscapes (Baeza and Estades 2010, Stokes et al. 2010, Montesino Pouzols et al.

2014), they are becoming more and more isolated and therefore increasingly vulnerable to anthropogenic activities and other environmental stressors (Sinclair and Byrom 2006, Laurance et al. 2012). This subsequently leads to further fragmentation of habitats (Fischer and Lindenmayer 2007), that, combined with patch sizes, degree of connectivity, habitat quality, and the level of human disturbances, affect biodiversity persistence within protected areas (Michalski and Peres 2005, Chazdon et al. 2008). Increasing human populations near protected area boundaries (Harcourt et al. 2001), and elsewhere, have further resulted in land use conversions that prevent free movement of wildlife (Newmark 2008, Wittemyer et al. 2008). As a result, protected areas become embedded within a mosaic of different land uses such as agriculture, cattle grazing and mining (DeFries et al. 2007, Chazdon et al. 2008, Di Minin et al. 2013c), and are subjected to a range of management strategies (Loveridge et al. 2007). In addition, protected areas are often too small to sustain viable populations of large mammals (Graham et al. 2009, Di Minin et al. 2013b, Packer et al. 2013), as they cannot meet the space requirements of wide-ranging or migratory species (Woodroffe and Ginsberg 1998, Graham et al. 2009, Stokes et al. 2010, Di Minin et al. 2013b). In many instances, this has led to increased human-wildlife conflict (Ogutu et al. 2011, Packer et al. 2013). Not only do those that live with dangerous species incur costs through human-wildlife conflict, but governments also incur the cost of protecting these species, for example, the current cost of anti-poaching measures as seen in the attempts to conserve and protect rhino in South Africa (Di Minin et al. 2015). For communities living with wildlife, the costs can be high. In Gokwe communal land bordering the Sengwa Wildlife Research Area in Zimbabwe, predation by large carnivores on livestock drains about 12% of each household's net annual income (Butler 2000). Similarly, where elephant (*Loxodonta africana*) coexist with humans in eastern and southern Africa, they account for more than 75% of the crop-raiding incidents attributed to large mammals (Hoare and Du Toit 1999).

Thus, in the increasingly human-dominated matrix that surrounds most reserves, the primary drivers that isolate protected areas are habitat loss, fences and roads, overhunting, and disease (Newmark 2008). These drivers restrict the movement of wildlife into and out of reserves and create sinks in the surrounding matrix (Newmark 2008, Balme et al. 2010b).

For the conservation of species whose habitat lies mainly outside protected areas, maintaining landscape diversity, connectivity, and compatibility of wildlife habitats with human land uses is paramount (Karanth et al. 2010a). For this reason, the inclusion of private and communal lands within conservation plans, are integral to the persistence of species (Cousins et al. 2008, Child et al. 2012). There are several other advantages to the inclusion of private reserves, conservancies and game ranches. Privately-owned land can play an important role in the protection of highly endangered species including black (*Diceros bicornis*) and white rhino (*Ceratotherium simum*), since these properties have much higher budgets and are more flexible and efficient in managing complex systems and reducing cost (Di Minin et al. 2013b, Packer et al. 2013, Di Minin et al. 2015). While there is little scope for the expansion of state-owned parks, the privately protected estate is growing (Gallo et al. 2009). These areas have been highly profitable (60%–80% over operational costs) and have been direct contributors to community wealth (Norton-Griffiths 2007, Naidoo et al. 2011, Packer et al. 2013). Beside the economic benefits accruing to landowners, private reserves and game ranches provide the public good 'biodiversity' at no cost to the tax-payer (Krug 2001). Thus, incorporating land uses and management practices which are compatible with biodiversity conservation not only helps protecting critical habitats for a variety of species (Gardner et al. 2007), but also contributes to maintaining landscape connectivity (DeFries et al. 2007). However, land use is also important for providing food, fibre and other ecosystem goods and services essential for human sustenance, and, globally, there is a concern that efforts in maintaining biodiversity are in conflict with those to reduce poverty (Sanderson and Redford 2003). An appropriate balance between land use to improve human well-being, and protected areas to conserve other ecosystem services, is thus needed, but is ultimately a societal decision (Bookbinder et al. 1998, DeFries et al. 2007). The conservation and protection of high value species, thus, does not rely solely on increasing the area under conservation or improving ecological conditions, but also in considering the socioeconomic needs of people (Adams et al. 2004, DeFries et al. 2007, de Boer et al. 2013).

The successful integration of biodiversity conservation and local economic development requires the identification of economic incentives that provide immediate benefits to local people, and that are appropriate in space and time to the scale of threats to biodiversity (Bookbinder et al. 1998). Where wildlife has no or limited value outside protected areas, it dwindles and disappears either through active persecution, loss of habitat, competition with livestock, or over utilisation (Prins and Grootenhuis 2000). In many instances where rural communities receive revenue from a species, they are more likely to conserve it, and will be more tolerant of negative impacts from such species (Barnes 1996, Hurt and Ravn 2000, Lindsey et al. 2007, Blignaut et al. 2008). In Africa, wildlife use, involving both consumptive (i.e. trophy hunting) and non-consumptive use (i.e. photographic ecotourism), is commonly associated with community-based natural resource management (CBNRM) programs (Hurt and Ravn 2000, Du Toit 2002, Blignaut et al. 2008), which could include formal wildlife ranching (Earnshaw and Emerton 2000, Hurt and Ravn 2000, Kreuter et al. 2010). Such CBNRM programs are run in Botswana, Namibia, and Zimbabwe. In both Botswana and Namibia, the CBNRM involve both non-consumptive and consumptive tourism, whereas over 80% of income derived through the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe is from trophy hunting (Lindsey et al. 2007, Blignaut et al. 2008).

For several species, sustainable use activities such as trophy or sport hunting is a valuable conservation tool, and could benefit the conservation of the species, especially outside of formally protected areas (Lindsey et al. 2006, Buckley and Mossaz 2015). Hunters are more likely to venture into areas with little or no infrastructure (Lindsey et al. 2007). Trophy hunting is thus of major importance to conservation in Africa by creating economic incentives for conservation over vast areas, including areas which may be unsuitable for alternative wildlife-based land uses such as photographic ecotourism (Lindsey et al. 2007). Trophy hunting is a key component of community conservation schemes in several countries (Lindsey et al. 2007, Jorge et al. 2013). In parts of Zambia, Zimbabwe, Botswana, Namibia and Tanzania, revenues from trophy hunting have resulted in improved attitudes towards wildlife among local communities, increased involvement of communities in CBNRM programs, requests to have land included in wildlife management projects, and, in some cases increasing wildlife populations (Lewis and Alpert 1997, Child and Chitsike 2000, Naidoo et al. 2015). However, unmanaged trophy hunting could have several detrimental effects on the hunted species (Treves and Karanth 2003, Lindsey et al. 2007, Fa and Brown 2009). In countries with poorly defined or mixed objectives, institutional failure, lack of management capacity, and corruption, the benefits to conservation through trophy hunting may be limited (Smith et al. 2003, Loveridge et al. 2007). A major challenge is, thus, to define appropriate quantifiable indicators to measure sustainability of wildlife offtakes, especially in situations where there are increasing human densities and expectations from households to receive undiminished benefits from the community's wildlife harvest, that could easily lead to unsustainable offtakes (Du Toit 2002). Quotas are critical to the continued existence of wildlife populations on government and communal lands (Hurt and Ravn 2000). At the same time, improved monitoring of wildlife populations is needed to ensure that quotas are sustainable (Selier and Di Minin 2015).

The fact that the home ranges of wide-ranging species, such as elephant and many carnivore species, frequently span administrative and political boundaries such as domestic and, in particular, international borders complicate the monitoring and management of these species (Delsink et al. 2013, Fattebert et al. 2013, Trouwborst 2015). The movement of these species across these administrative boundaries, especially international boundaries, result in an ad hoc approach to their management, particularly within a context of increasing human-wildlife conflict, and an inconsistent consideration of ecological requirements of the species at a cross-border level. The natural consequence is a mismanagement of these populations at a cross-border level (Delsink et al. 2013, Selier et al. 2015, Trouwborst 2015). For this reason, the development and expansion of international cross-border or transboundary conservation areas are essential (Scovronick and Turpie 2009). The term "Transfrontier Conservation Area (TFCA)" is defined by Hanks (2003) as relatively large tracks of land, straddling frontiers between two or more countries, and which embraces natural systems encompassing one or more protected areas. This means that TFCAs can encompass varying mosaics of land use and can incorporate private land, communal land, forest reserves and wildlife management areas, including, where appropriate, consumptive use of wildlife (Hanks 2003). The objective of transboundary conservation areas is, however, not only the conservation of biodiversity, but also the economic development of communities within these border regions (Hanks 2003, Scovronick and Turpie 2009), with one of the aims to open boundaries to encourage relatively unrestricted movement of tourists and large mammals (Hanks 2003). This is especially important for regions with large bodied, valuable mammals, such as African elephant, which could negatively affect both ecosystems (Skarpe et al. 2004, Kerley and Landman 2006, Makhabu et al. 2006, Guldemond and van Aarde 2008, Helm and Witkowski 2012) and local communities surrounding protected areas (Naughton et al. 1999, Hoare 2000, von Gerhardt-Weber 2011). However, several political, legislative and implementation challenges exist that could impede the effective management of such species across international borders (Plumptre et al. 2007).

This study falls within the 'Conservation Biology Domain' and follows a transdisciplinary approach looking at the biology, exploitation, socioeconomic factors and legislation and policy to address the challenges and opportunities of managing high-value, wide-ranging species, subjected to various land uses in human-dominated landscapes that span administrative and political boundaries. The understanding gained through this study of elephant within the Greater Mapungubwe Transfontier Conservation Area (GMTFCA), shared among Botswana, South Africa and Zimbabwe (Selier et al. 2014), can be used not only to improve the conservation practice and management of this specific species, but of cross-border species in human-dominated landscapes in general.

We use the African elephant population in the GMTFCA in southern Africa as a case study, because elephant are a wide-ranging species exposed to different land uses and management practices which differ across various national and international administrative boundaries (van Aarde and Jackson 2007, Delsink et al. 2013, Selier et al. 2015). The African elephant is a high value species both from a consumptive and non-consumptive perspective, and attracts high numbers of visitors to conservation areas in sub-Saharan Africa as a flagship species (Di Minin et al. 2013a). Elephant conservation also benefits other species through an umbrella effect (Di Minin and Moilanen 2012, Di Minin et al. 2013b), and the elephant is a keystone species, that affects ecosystem structure and function (Holdo 2006). Elephant in the study area are transboundary, moving freely among three southern African countries, and are subsequently exposed to different legislation and a range of management practices, which prevent them from using the landscape freely (Selier et al. 2015).

The African elephant was once widespread in the southern African sub-region, occurring in high numbers in most areas until the twentieth century, when large-scale hunting and ivory trade reduced numbers significantly throughout their range (Plug and Badenhorst 2001). Currently the southern African elephant population constitutes 55% of the total African elephant population (Blanc et al. 2007). Within southern Africa, Botswana holds by far the largest population in the sub-region and on the continent (approximately 150 000 animals), while Mozambique, Namibia, South Africa, Zambia and Zimbabwe still hold large elephant populations (Blanc et al. 2007). While elephant numbers appear to be increasing in Botswana and South Africa, there appear to be some initial declines in some of the populations in Mozambique, Zambia and Zimbabwe (CITES et al. 2013). Globally, the African elephant is listed as 'Vulnerable' (A2a) (IUCN Red List (2008 assessment; www.iucnredlist.org), fitting a worrying pattern applicable to many large herbivores across the globe (Ripple et al. 2015). However, the species is considered 'Least Concern' in the southern African region which includes Botswana, South Africa and Zimbabwe (IUCN Red List (2008 assessment); www.iucnredlist.org). Within all three of these countries, the elephant status can be considered a conservation success. However, elephant in the region are the primary agents of ecological change across their range, and are also one of the primary causes of human-wildlife conflict and the source of international controversy (Couzens 2013).

1.2 AIMS AND OBJECTIVES

The aim of this study was to assess the challenges and opportunities for conserving and managing a wide-ranging species in a human-modified landscape that spans administrative and political boundaries using a transdisciplinary approach. The African elephant was used as a case study and the following objectives and sub-objectives were addressed:

1.2.1 To determine the sustainability of trophy hunting on a cross-border elephant population.

The sub-objectives were:

- To describe the current trophy hunting quotas for the respective range countries;
- To examine the effects of trophy hunting under different ecological and hunting scenarios on population sustainability using population models;
- To assess the effects of the spatial patterning of elephant removal by trophy hunting on the population;
- Provide recommendations towards the management of elephant and other cross-border species.

1.2.2 To understand how anthropogenic activities can affect the distribution of a large, wide-ranging, mammal species at different spatial scales.

The sub-objectives were:

- To investigate the relative importance of environmental and anthropogenic factors affecting the distribution of the African elephant population in the Greater Mapungubwe Transfrontier Conservation Area;
- To understand whether elephant trade-off resource selection against avoidance of anthropogenic disturbance, at different spatial scales.

1.2.3 To understand which ecological and socioeconomic factors affect elephant densities in a transfrontier conservation landscape.

The sub-objectives were:

- To describe trends in the abundance of elephant over time;
- To identify site- and country-level factors affecting the abundance of elephant.

1.2.4 To determine the legal challenges of transboundary wildlife management at the population level.

The sub-objectives were:

- To explore the essential elements of organising wildlife law and policy at the transboundary level, drawing on European experiences regarding the management of populations of gray wolf (*Canis lupus*), as well as other large carnivore species, and of pink-footed goose (*Anser brachyrhynchus*);
- To analyse to what degree the transboundary population level approach is incorporated in the applicable law and policy at the global and regional level, the trilateral level, and the national level in the three countries concerned;
- Provide recommendations to the adjustments of laws and policies that are required to ensure the ecological stability of transboundary populations and to provide for their collective management.

1.3 STUDY OUTLINE

This study is presented in six chapters, of which Chapters 2 to 5 are all written in the format for publication in peer reviewed journals:

Chapter 2: Sustainability of elephant hunting across international borders in southern Africa: A case study of the Greater Mapungubwe Transfrontier Conservation Area. This part of the study addressed Objective 1.2.1, to determine the sustainability of trophy hunting on a cross-border elephant population. The GMTFCA elephant population was used as a case study. In this chapter, I presented the current status of the GMTFCA elephant population with regards to trophy hunting and offtake levels within Botswana, South Africa and Zimbabwe. Using VORTEX, I determined how different trophy hunting offtakes under three different environmental conditions will influence the sustainability of this population, and I made recommendations towards a sustainable offtake quota for the population. In addition, using distribution data from six aerial surveys and hunting data per region, I determined the disturbance effects of trophy hunting on bulls and breeding herds separately, and how these influenced the movements of both bulls and breeding herds across the landscape.

Chapter 3: Large mammal distribution in a transfrontier landscape: Trade-offs between resource availability and human disturbance. Objective 1.2.2, was to understand how anthropogenic activities can affect the distribution of a large, wide-ranging, mammal species at different spatial scales. This chapter investigated the environmental and anthropogenic factors that were most important in determining the distribution of elephant at different spatial scales within the GMTFCA. I combined distribution data from six aerial counts over a 12-year period with fourteen variables, representing food availability, landscape, and anthropogenic effects, into generalised linear models. Using these generalised linear models, I investigated what predictors, at a broad scale, as well as within three separate management units within the broader landscape, namely ecotourism, trophy hunting and a combination of hunting and trophy hunting, affected the distribution of elephant. Furthermore, I determined whether human activities within different management units forced elephant to avoid disturbance rather than risk accessing preferable resources i.e. good food and water availability.

Chapter 4: The influence of socioeconomic factors on the effective management of high-value cross-border species. The chapter addressed Objective 1.2.3, to understand which ecological and socioeconomic factors affect elephant abundance in a transfrontier conservation landscape. In this chapter I used elephant abundance data from six aerial surveys conducted in the GMTFCA over a 12-year period in generalised linear models to investigate the effect of seven socioeconomic variables on the abundance of elephant at different spatial scales. Specifically, we ran models at the country and site level. Further to this, I made some recommendations to ensure the persistence of wide-ranging cross-border species within human-dominated landscapes, based on the results provided.

Chapter 5: The legal challenges of transboundary management at the population level: The case of a trilateral elephant population in southern Africa. Chapter 5 addressed Objective 1.2.4, to determine the legal challenges of transboundary wildlife management at the population level. In this chapter, I explained the transboundary population level approach and explored the essential elements of organising wildlife law and policy at the transboundary level, by drawing on European experiences regarding the management of populations of gray wolf (*Canis lupus*) and of pink-footed goose (*Anser brachyrhynchus*). This was followed by an analysis of applicable global, regional, trilateral and national law and policy as pertaining to Botswana, South Africa and Zimbabwe, and to what degree the transboundary population level approach has been incorporated in respective laws and policies. In the final section, I made recommendations to the adjustment of laws and policies that are required to ensure the ecological stability of transboundary populations, and to provide for their collective management. During the course of this study it became apparent that the local and international legal framework was of critical importance for achieving conservation goals. We therefore set out to review and evaluate current legislation and make recommendations about how it might be modified. Whilst the approach to this was scientific in that I conducted a rigorous review and evaluated the effectiveness of current legislation on a scientific basis, the reporting does not follow the normal format for a scientific paper. In publishing law it is important to avoid ambiguity that occurs when data are presented in tables or argued in a comparative context. The layout therefore follows the format for publication in a legal journal rather than a scientific one.

Chapter 6: Conclusion. The concluding chapter highlights the main research findings and how these have addressed the research aim and objectives. I further provide conservation management recommendations, and discuss limitation within the study.

CHAPTER 2: SUSTAINABILITY OF ELEPHANT HUNTING ACROSS INTERNATIONAL BORDERS IN SOUTHERN AFRICA: A CASE STUDY OF THE GREATER MAPUNGUBWE TRANSFRONTIER CONSERVATION AREA

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ABSTRACT

Trophy hunting of African elephant is often implemented as an income generator for communities surrounding protected areas. However, the sustainability of hunting on elephant populations, especially with regards to international cross-border populations has not previously been evaluated. We assessed the effects of trophy hunting on the population dynamics and movements of elephant in the Greater Mapungubwe Transfrontier Conservation Area, which is spread across the junction of Botswana, South Africa, and Zimbabwe. Currently, no common policy exits in quota setting for cross-border species, and each country determines their own quota based on limited data. Using VORTEX, we determined the sustainability of current quotas of elephant off-take under different ecological and hunting scenarios. We used distribution data from six aerial surveys and hunting data per region to determine the disturbance effect of hunting on bulls and breeding herds separately. Hunting of bulls had a direct effect in reducing bull numbers but also an indirect effect due to disturbance that resulted in movement of elephant out of the areas in which hunting occurred. The return interval was short for bulls but longer for females. Only a small number of bulls (<10/year) could be hunted sustainably. At current rates of hunting, under average ecological conditions, trophy bulls will disappear from the population in less than 10 years. We recommend a revision of the current quotas within each country for the Greater Mapungubwe elephant population, and the establishment of a single multi-jurisdictional (cross-border) management authority regulating the hunting of elephant and other cross-border species.

2.1 INTRODUCTION

For the conservation of species whose habitat lies mainly outside protected areas, maintaining landscape diversity, connectivity, and compatibility of wildlife habitats with human land uses is important (Karanth et al. 2010b). For this reason, the development and expansion of international cross-border or transboundary conservation areas are essential (Scovronick and Turpie 2009). The objective of transboundary conservation areas is, however, not only the conservation of biodiversity but also economic development of communities within these border regions (Hanks 2003, Scovronick and Turpie 2009). This is especially important for regions with large bodied, valuable mammals, such as African elephant (*Loxodonta africana*), which could negatively affect both ecosystems (Skarpe et al. 2004, Kerley and Landman 2006, Makhabu et al. 2006, Guldemond and van Aarde 2008, Helm and Witkowski 2012) and local communities surrounding protected areas (Naughton et al. 1999, Hoare 2000, von Gerhardt-Weber 2011). However several challenges impede the effective management of such species across international borders (Plumptre et al. 2007).

Where wildlife has no or limited value outside protected areas, it dwindles and disappears either through active persecution, loss of habitat, competition with livestock, or over utilization (Prins and Grootenhuis 2000). In many instances where rural communities receive revenue from a species, they are more likely to conserve it, and will be more tolerant of negative impacts from such species (Barnes 1996, Hurt and Ravn 2000, Lindsey et al. 2007, Blignaut et al. 2008). In such instances, sustainable consumptive use through trophy hunting may benefit conservation (World Tourism Organisation 1997).

In Africa, wildlife use, involving both consumptive and non-consumptive use, is commonly associated with community-based natural resource management (CBNRM) programs (Hurt and Ravn 2000, Du Toit 2002, Blignaut et al. 2008), which could include formal wildlife ranching (Earnshaw and Emerton 2000, Hurt and Ravn 2000, Kreuter et al. 2010). Such CBNRM programs are run in Botswana, Namibia, and Zimbabwe. At present, approximately 80% of the current African elephant range in southern Africa is outside formally protected areas (Blanc et al. 2007, Abensperg-Traun 2009), leading to increased conflict with local communities (Hoare 2000, Jackson et al. 2008, Riddle et al. 2010). In both Botswana and Namibia, the CBNRM involve both non-consumptive and consumptive tourism, whereas over 80% of income derived through the Communal Areas Management Program for Indigenous Resources (CAMPFIRE) in Zimbabwe is from trophy hunting (Lindsey et al. 2007, Blignaut et al. 2008). Within the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA), during the period 1999 to 2010 between two and 43 elephant were hunted from the population annually. Current quotas are set at 33 for Botswana and 14 for Zimbabwe. Although a zero hunting quota was set for elephant in South Africa, a total of 18 elephant were shot as problem elephant during 2006 to 2010. South Africa has been party to the Convention of the Conservation of Migratory Species of Wild Animals (CMS) since 1991 (CMS 2012a). Zimbabwe joined only in 2012 and Botswana is not party to this convention (CMS 2012b). This leads to little or no consultation among the three countries in setting quotas for hunting elephant, and each country determines their own quota based on restricted subsets of population data.

If the profits realized from harvesting a few individuals are sufficient incentive for people to tolerate the larger population, the goals of trophy hunting and conservation are compatible (Treves and Karanth 2003, Balme et al. 2010a). However, where large mammal species, such as ungulates and carnivores, are targeted, excessive sustained harvesting can lead to extirpation (Treves and Karanth 2003, Lindsey et al. 2007, Fa and Brown 2009), or selective harvesting may have negative evolutionary consequences (Harris et al. 2002, Coltman et al. 2003, Balme et al. 2010a). In addition, studies on carnivore and antelope populations have shown that the selective removal of a few large trophy or the oldest males can potentially lead to the destabilization of social structures and the dominance hierarchy and a loss of social knowledge (Milner et al. 2007). Other possible consequences of the selective removal of large trophy males are infanticide, reproductive females using sub-optimal habitats, and changes in offspring sex ratio (Milner et al. 2007). Little is known about the disruptive effects of hunting on the dominance hierarchy through changes in stress levels, movements, and other behaviour (but see Burke et al. 2008). In situations with high hunting pressure, these effects may be significant and negative (Archie et al. 2008).

Although a few studies have investigated the potential social and demographic effects of hunting adult bulls (Archie et al. 2008, Burke et al. 2008), the results are not directly applicable when attempting to evaluate the merits of hunting cross-border populations. In these studies only the fine-scale genetic implications and stress levels as a result of poaching and hunting respectively were studied. We assessed the effects of hunting on the population dynamics and movements of elephant using data from the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA). Specifically, we describe the current hunting quotas, examine the effects of hunting under different ecological and hunting scenarios on population sustainability using population models, and assess the effects of the spatial patterning of elephant removal by hunting on the population. We conclude by providing some suggestions towards the management of elephant and other cross-border species.

2.2 STUDY AREA

The GMTFCA is situated at the confluence of the Shashe and Limpopo Rivers and includes areas from Botswana, South Africa, and Zimbabwe (Figure 2.1). It is a semi-arid area with low, unpredictable rainfall. The long-term average annual rainfall was 369 mm (1966–2001) with peak rainfall years receiving 917 mm (Jul 1999 to Jun 2000) and low rainfall years receiving as little as 136 mm (Jul 1997 to Jun 1998). Most rain fell between November and February, usually in the form of thunderstorms (Mckenzie 1990). Summer maximum temperatures exceeded 42º C and winter minimum temperatures were as low as −5º C. An elephant population of approximately 1,224 ±72.4 estimated from six aerial counts over a 10-year period roamed freely across the three countries (Selier 2010).

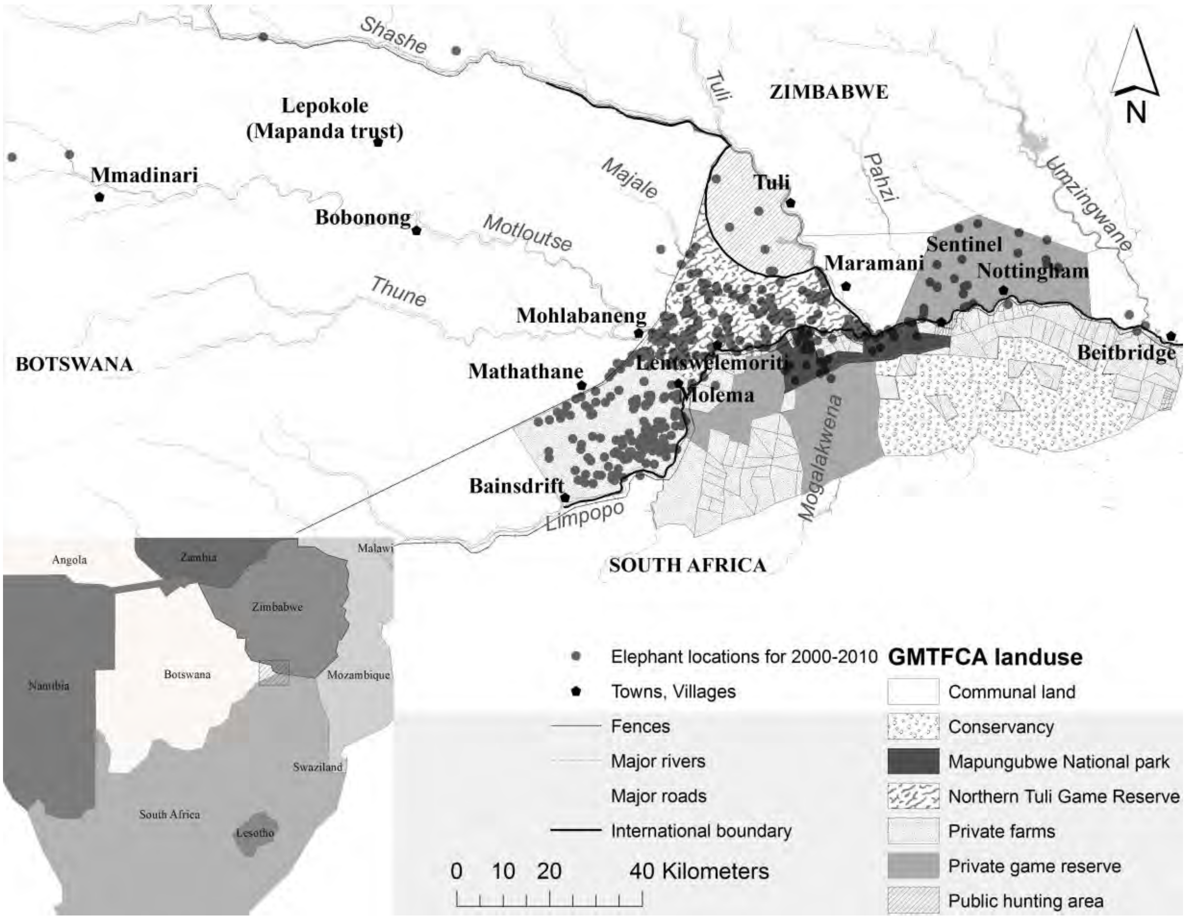


Figure 2.1: The Central Limpopo River study area with different land use practices and the elephant locations for 2000–2010.

Botswana is divided into administrative blocks called Controlled Hunting Areas (CHAs; (Abensperg-Traun 2009) that each have a wildlife off-take quota designated by the Department of Wildlife and National Parks. Some CHAs, such as protected areas, have a hunting quota of zero, whereas other CHAs are designated for community use (Abensperg-Traun 2009). A community with a legally recognized trust and a land use plan can apply for a lease over the CHA from the Tribal Land Board. This will allow the Trust to sub-lease use of their land and their quota to a tourism company for photographic or hunting safaris (Abensperg-Traun 2009). Within the Central Bobonong District in Botswana (Figure 2.1), three community trusts, Mmadinari, Mapanda, and Molema Trusts, have been developed since 2000. Each trust is allocated an annual elephant-hunting quota, and the trusts are responsible for marketing and managing hunting safaris within their community. The allocation of hunting quotas in terms of problem animal control laws have been used to deter elephant from entering communal areas, to compensate local communities for wildlife-related losses, and to improve the tolerance of communities towards elephant (M. Mamani, personal communication)[1].

In the Beitbridge Rural District in Zimbabwe (Figure 2.1), the CBNRM program is run as a CAMPFIRE Project (CESVI 2001). Three hunting concessions occur within this area, of which two (Sentinel Limpopo Safaris and Nengasha Safaris) operate west of Beitbridge. Elephant

[1] Malatsi Mamani, Department of Wildlife and National Park, Bobonong, GMTFCA Elephant Conservation Policy and Management Planning workshop, Duncan Macfadyen Research Centre, De Beers Venetia Limpopo Nature Reserve, 5-6 May 2010.

trophy hunting is also offered within the Tuli Circle Safari Area, Zimbabwe. In South Africa, all elephant crossing out of reserves are considered problem animals, and with the acquisition of a hunting permit from the provincial conservation agency, can be hunted by the farmer or a paying client (Hopkinson et al. 2008).

Several tourism operations operate within the current boundaries of the GMTFCA. Even though photographic tourism is the main economic driver within the area at present (Evans 2010), several operations rely on a combination of trophy hunting and photographic tourism. The Northern Tuli Game Reserve (NTGR) is a private game reserve within Botswana (Figure 2.1), focusing purely on photographic tourism. Within the Zimbabwean section, two private commercial tourism operations focus on a combination of trophy hunting and photographic tourism (Nottingham Estate and Sentinel Ranch), whereas the Mapungubwe National Park (MNP) in South Africa (Figure 2.1) is solely a photographic tourism destination. All of the tourism destinations use, either for viewing or trophy hunting, a single cross-border elephant population that moves freely between the three countries.

2.3 METHODS

2.3.1 Current hunting quotas

We obtained data on quotas for trophy hunting and kill rate of elephant for all three countries from the respective wildlife departments, private landowners, and reserve and farm managers within the three countries. We obtained data on population numbers and the distribution of elephant within the GMTFCA from six aerial surveys conducted within the study area over the period 2000–2010 (Selier 2010). We divided numbers and distribution of elephant into the following regions: 1) Botswana, which included two separate sections, the Northern Tuli Game Reserve (NTGR) and the Tuli Block from the Motloutse River to Baines Drift (TLBL); 2) the Zimbabwean section along the Limpopo River, including Maramani, Sentinel Ranch, and Nottingham Estate up to the Umzingwane River (ZIM); and 3) the South African section, including Mapungubwe National Park and private properties bordering the Limpopo River where elephant had access (MNP)(Figure 2.1). Major rivers (Limpopo, Shashe, and Motloutse rivers) form natural boundaries between the above-mentioned regions (Figure 2.1). Hunting occurred in all of the regions other than the Northern Tuli Game Reserve and all but two of the Tuli Block properties in Botswana.

2.3.2 Population projection

We used VORTEX 9.50 (Lacy et al. 2005) population simulation software to determine the impact of different harvest rates of mature elephant bulls on the viability of the elephant population in the GMTFCA. The VORTEX model has been used extensively and the internal logic and the assumptions inherent in it have been evaluated in terms of its ability to emulate the known behaviour of the populations (Armbruster et al. 1999, Brook et al. 2000, Lindenmayer et al. 2000, Nilsson 2004, Rija 2009).

Van Aarde *et al*. (2008) summarized the annual survival rate of elephant in different age classes, the length of calving intervals, and the age at first calving for different African populations from published data. We used these values to determine the above average, average, and below average demographic parameters (age at first calving and mortality; Table 2.1). We used the maximum values obtained by populations outside South Africa as the parameters for above average conditions. Likewise, we obtained the average demographic parameters by selecting the below average parameters by selecting the population outside South Africa with the lowest values. We did not use data from Luangwa National Park, Bugongo, or Murchison North and South, which we considered outliers because their values were significantly lower or higher than the other areas presented in their table (van Aarde et al. 2008). Luangwa suffered extreme poaching and this could possibly be the reason for the lower survival rates within the population (Owens and Owens 2009). Calving intervals for elephant populations outside South Africa vary between 2.1–9.1 years, with the majority of the populations falling between a 3–5-year calving interval (van Aarde et al. 2008). We thus used a 3-year calving interval or 0.33 breeding rate for above average conditions, a 4-year calving interval or 0.25 breeding rate for average conditions, and a 5-year calving interval or 0.20 breeding rate for below average conditions. We designated 15 years as the age at maturity for bulls in the model (Poole 1994). According to age structure data from Amboseli National Park, Kenya, the percent of males >15 years within the population is approximately 48% of the total male population (Moss 2001). We used this value as the percent of males within the breeding pool of the initial population. To determine mortality rate, the model required the percent of females and males dying for each year from birth to age of first offspring. To model natural mortalities, we used the age-specific survival rates given in van Aarde *et al*. (2008) and divided them by the number of years within each of the categories to get an age-specific mortality (up to 60 years of age). We used the average mortality rate for the age groups 20–29, 30–44, and 45–60 as the adult mortality (van Aarde et al. 2008). The maximum age of reproduction for both males and females was 45 years. Moss (2001) showed that female fecundity declines after 40 years with a rapid drop after 50 years of age.

Table 2.1: Parameters used within the VORTEX analysis for above average, average, and below average environmental conditions with seven different harvesting rates of elephant bulls in the Greater Mapungubwe Transfrontier Conservation Area. Data are the low, median, and high values of mean annual survival rates of elephant in different age classes and populations from published data. Actual is the average observed over the past 10 years in our population.

Conditions	Above average	Average	Below average	Actual data
Age females at first calving	10	13	15	15
Age bulls sexual maturity	15	15	15	15
Max. age reproduction	45	45	45	45
% females breeding/year	33	25	18	25
Mortality rate				
<1 yr	2%	8%	18%	5%
1–9 yr	1%	5%	13%	4%
10–15 yr	2%	5%	10%	4%
>15 yr (adult)	2%	5%	10%	5%
% males in breeding pool	48%	48%	48%	48%
Initial population size	1,224	1,224	1,224	1,224
Upper population limit	3,740	2,299	1,134	2,299

We set the upper and lower limits for the elephant population based on rainfall variation in the area. We calculated average annual rainfall from four rainfall stations: Mashatu Main Camp, Mashatu Tented Camp, Pont Drift border post, and Platjan border post. Over the period, mean rainfall was 324 mm with a 20% variation, which is 8% below the long-term mean of 351 mm with a variance of 11% for the period July 1989 to June 2010. To use these limits for average, below average, and above average conditions we used the following logic. The Northern Tuli Game Reserve was an open system with very few barriers to movement. The South African border was fenced in places but open in others, and the backline of the Tuli Block farms was fenced with a non-elephant proof fence. No barriers prevented movement to the north, east, or southwest. No hunting occurred within the reserve (Selier 2007). We therefore calculated the density of elephant within the Northern Tuli Game Reserve for each year between 1988 and

2010. We then used the maximum, average, and minimum densities of elephant for these years as estimates of the limits during above average, average, and below average conditions, respectively. The maximum density observed in the Northern Tuli Game Reserve since 1988 was 1.22 elephant/km^2 (2001, above average rainfall conditions; (Selier 2010)). Extrapolating this density across the total area available to elephant within the proposed GMTFCA (3,065 km^2) resulted in a maximum population of 3,740 elephant. We determined average density (0.74) and low density (0.37, September 1996, below average rainfall conditions) in the same manner and, when extrapolated to the GMTFCA, gave an average population of 2,299, and a low population of 1,134 elephant.

Current evidence indicates that the GMTFCA population is stable (Selier 2007), and, until 2009, the population was not controlled and levels of hunting and poaching were relatively low. For the 10 years between 2000 and 2010, the elephant population ranged in an area of 2,016 km^2 at an average density of 0.61 elephant/km^2, with biennial aerial censuses varying between 1,080 and 1,294 elephant with a mean count of 1,224 (±72.4 SD) elephant (Selier 2010). If the population is currently at biological carrying capacity and the birth death rate that we observing reflect the population density we can validate these numbers by running the model at average conditions. We thus assumed that the observed birth and death rates were those that determined the population densities at biological carrying capacity.

A sensitivity analysis was run to determine the sensitivity of the model output to changes in parameter values. To evaluate whether the birth and death processes kept the population below the limit we raised the upper limit to a very high level (35,000 elephant/km^2) in the sensitivity analysis.

We ran the model for 10 years under three scenarios of above average, average, and below average rainfall to determine the total elephant numbers and number of bulls in different age classes at the end of the simulation. Because VORTEX is a stochastic model, we used the average of 100 iterations as the final output. We evaluated different annual harvest rates (0, 5, 10, 20, 30, 40, 50) of bulls >15 years in the analysis. In order to calculate the Mean Sustainable Yield (MSY) we determined harvest rates based on starting population numbers for the total GMTFCA set at 68% below (current population), 47% below (average conditions), and 8% above (below average conditions) the upper, high-resource limit (Table 2.1) only because at biological carrying capacity, any harvesting would result in a decline in the population numbers. Given that the existing population showed no trend in population numbers in the past 10 years, but fluctuated around a mean of 1,224 ±72.4 SD, we assumed that the population had a stable age distribution. In the model, we ignored inbreeding depression and incorporated environmental variability under the three environmental scenarios described above.

We assessed the effect of hunting at different levels on the population trend. Even though bulls 15–35 years are unlikely to mate under natural conditions, bulls at this age are sexually active and, in the absence of older bulls, they are capable of reproducing (Poole 1989, Slotow et al. 2000, Moss 2001). Adult bulls >35 years are favoured by trophy hunters (Archie et al. 2008). For this reason, we ran the VORTEX model with bull breeding age at 35 years for all three environmental scenarios (above average, average, and below average), with different harvest rates (0–50 bulls/yr) to determine the impact of harvest rates on bulls >35 years. From the VORTEX output, we determined population size, the total number of bulls, number of bulls >15 years, number of bulls >35 years, and the sex ratio between breeding age females and adult bulls >35 years for the three environmental scenarios. As a comparison and evaluation of the VORTEX output, we also used the formulae of Caughley (1993) and Martin *et al*. (1997) to determine maximum sustainable yield. Caughley (1993) determined that the MSY for elephant was approximately half the population's maximum rate of increase, multiplied by half the size of the population when not harvested. The MSY of a growing population is therefore greater than that of a stable population. On the other hand, Martin *et al*. (1997) recommended that, to sustain good quality trophy elephant hunting, quotas ideally should not exceed 0.7% of the total population.

2.3.3 Effect of hunting on movement

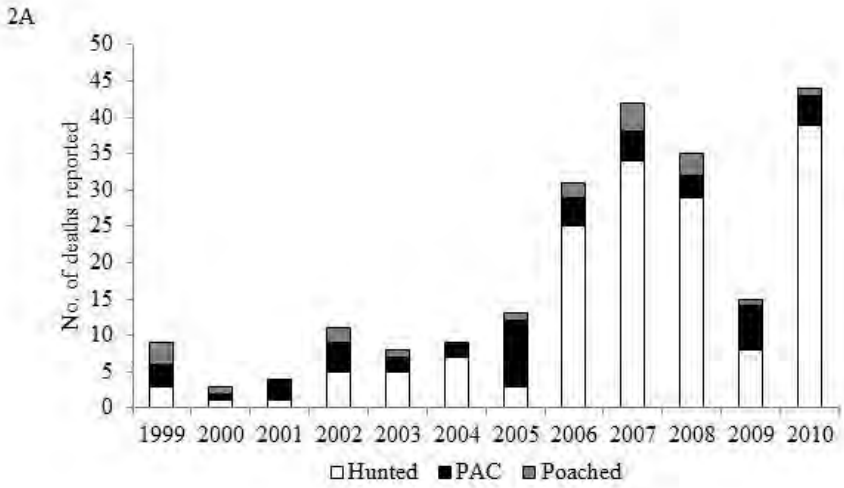
To assess whether hunting influenced movements and visibility of adult bulls, we used regression analysis (Sokal and Rohlf 1995) to relate the number of bulls observed during the aerial surveys in each region to the number of bulls hunted during the same year and in the previous year. We used population numbers from the six biennial aerial surveys between 2000 and 2010 (Selier 2010). The hunting season extended between May and October and most of the hunts occurred prior to the annual count survey of a given year. We also related the number of females observed during the surveys to the number of bulls hunted in the same year and in the previous year.

2.4 RESULTS

2.4.1 Current hunting quotas

For the 2010 hunting season, 40 bulls >35 years old were killed within Botswana and Zimbabwe combined (Figure 2.2A). Since 2006, within Botswana between 16 and 36 trophy bulls annually were allocated to various community trusts within the Central Bobonong District (Figure 2.2B). Since 2006, within Zimbabwe the quota remained steady at approximately 11 elephant per year (six bulls and five tuskless cows) for the Beitbridge District, and additional three elephant bulls for the Tuli Circle Safari Area (Figure 2.2B). Despite no annual quota being allocated since 2006, 18 elephant were shot on the South African side as problem elephant. Eleven of these were mature breeding bulls hunted by paying clients (L. De Jager, personal communication)[2].

[2] L. De Jager, Permitting, Vhembe District, Limpopo Nature Conservation.



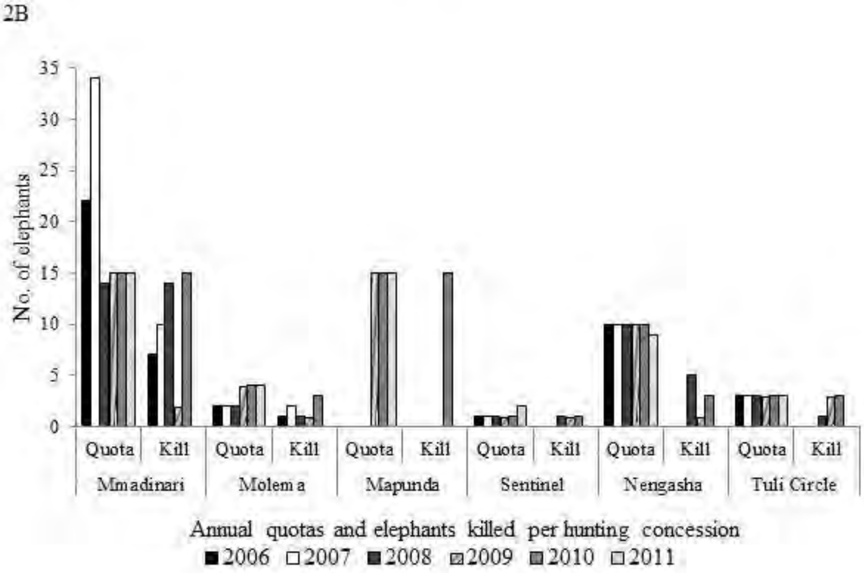


Figure 2.2: Number of elephant killed per annum through unnatural causes within the Greater Mapungubwe population between 1999 and 2010 (A), and a comparison of the annual hunting quota and kill rate for each of the hunting concessions from 2006 to 2011 (B). PAC = problem animal control.

2.4.2 Population projection

The model evaluation showed that the average values we used were able to reproduce the numbers counted in the six biennial censuses over the period 2000 to 2010. Model output using the average current harvesting rate over the past 10 years (14 bulls/yr), a starting population of 1,102 elephant, which was 10% below current density, and a upper limit for average conditions of 2,299 elephant gave a population with a low rate of increase (r = 1.47% SD ± 0.80), which started at an average of 1,102 and ended at 1,277 (SD ± 30.83) animals over the 10-year period. The range in counts over the past 10 years was between 1,080 (2007) and 1,294 (2001). The simulation over 10 years was therefore in agreement with the aerial counts over the 2000 to 2010 period. When we ran the model with observed birth and death rates determining population density under average conditions, the population increased marginally to 1,539.

Sensitivity analysis indicated that the model was most sensitive to changes in the percent of females breeding in any one year (which is equivalent to calving interval). A change from 10% females breeding in any one year to 45% breeding resulted in an increase in the population after 10 years from 1,057 to 2,013 (with all other variables kept constant at average values), and, after 50 years, from 587 to 14,503. The model was also sensitive to adult mortality rate, for which a decrease from 13% to 1% resulted in a population change after 10 years from 1,096 to 1,851 and from 699 to 9,739 after 50 years, for the respective mortality rates. To test the effect of the upper limit of the model the upper limit was raised to 35,000 elephant/km^2. The population did not reach this limit for average conditions of 2,299 after 10 years, but after 50 years the population exceeded the upper limit of 2,299 (3,837±123.65 SD, 39.10 SE). Because our simulations were for 10 years, errors in the estimate of the upper limit did not affect the results of the simulation.

For all three rainfall scenarios, modelled population growth rate declined steadily as the number of bulls harvested per year increased. With 10 years of above average conditions, the population growth rate increased 6.34 ± 0.59% without harvest but only increased by 1.76 ± 3.24% when 50 adult bulls were harvested/yr; Figure 2.3A). Under average conditions, the population growth rate increased 2.29 ± 0.69% without harvest and decreased by 1.51 ± 1.92% when 50 bulls were harvested per annum. With below average conditions after 10 years, the population growth rate decreased by 2.10 ± 0.92% with no harvesting and decreased by approximately 6.76 ± 1.30% when 50 bulls per year were harvested. Under all three conditions, as harvest rate increased, the number of bulls >15 years in the final population declined steadily (Figure 2.3B). After 10 years of harvesting under below average conditions, 18 bulls >15 years remained at the harvest rate of 30 bulls per annum, and under both average and above average conditions, no bulls >15 years were left at a harvest rate of 50 bulls per annum.

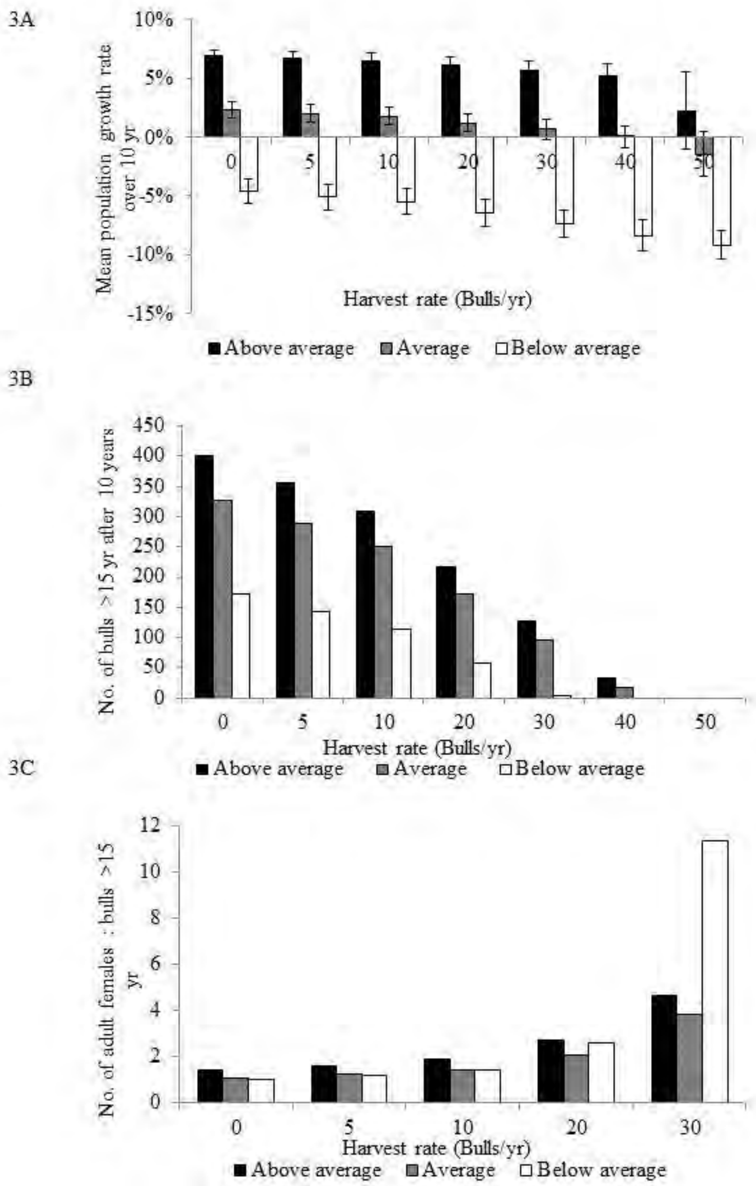


Figure 2.3: Comparison of mean population growth rate (A), number of bull >15 years old (B), and changes in sex ratio of females to bulls >15 years old (C) of elephant in the Greater Mapungubwe population under three environmental scenarios (above average, average, and below average) and seven different adult bull harvesting rates after 10 years of harvesting at the specific rate. Outputs are from VORTEX modelling with 100 iterations run for 10 years. The ratio of breeding age females to bulls >15 years old for harvest rates of 40 and 50 bulls per annum are too large and are not reflected in the graph.

Assuming average conditions, the MSY would be below an annual harvest rate of approximately 40 bulls >15 years old (Figure 2.4). This was approximately 2.60% of the total population or 12% of the initial adult bull population of 321. If hunter preference and the social structure of elephant bulls were taken into account (bulls 15–35 years are unlikely to mate under natural conditions), the maximum sustainable harvest rate of bulls was approximately 10 bulls >35 years per annum. Based on the recommendation of Martin *et al*. (1997), hunting quotas should ideally not exceed 0.7% of the total population and thus only nine bulls >35 years could be hunted from the current population of 1,224 elephant. Based on Caughley (1993) approach, only six bull elephant per annum could be hunted.

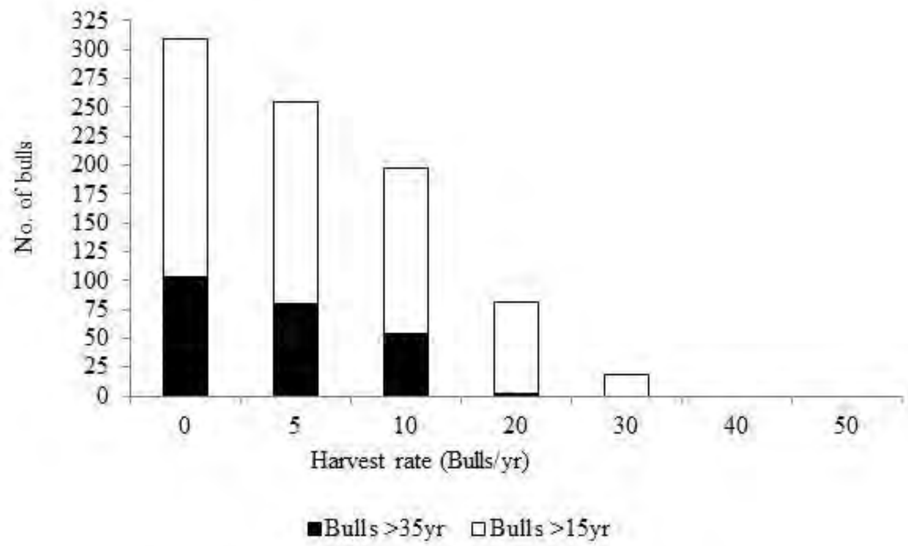


Figure 2.4: Predicted number of elephant bulls >35 years old and the number of bulls >15 years old remaining within the population after 10 years of hunting at different harvesting rates under average conditions in the Greater Mapungubwe population.

Even though relatively high hunting quotas were allocated to the three community trusts within the Central Bobonong District, Botswana, the annual kill rate within these hunting concessions was low. Of the 144 elephant on quota from 2006 to 2010, only 71 elephant were hunted (Figure 2.2B). A similar trend was observed in Zimbabwe, where of the 54 elephant on quota since 2008, fewer than half the quota was harvested (Figure 2.2B). Despite this low off-take, hunting in 2010 was 75% above the MSY levels suggested by the VORTEX model, 77.5% above MSY calculated using the formula of Martin *et al.* (1997), and 85% above that calculated using the formula of Caughley (1993).

Assuming a stable age structure, an initial population of 1,224 elephant should have included approximately 200–300 bulls >15 years old. Under average conditions, the model predicted 321 bulls >15 years old with no hunting. Harvesting led to an increasingly skewed sex ratio. Under no harvest, the model predicted a 1:1 bull >15 years old to breeding age female ratio but dropped to 1:25 bull >15 years old to breeding age females ratio at a removal rate of 40 bulls per annum under average conditions (Figure 2.3C). Under average conditions, the number of bulls >35 years old dropped steadily with an increase in the harvest rate (Figure 2.4). At a harvest rate of more than 20 bulls >35 years old per annum, no bulls would be left in this age class left after 10 years of harvesting regardless of environmental conditions.

2.4.3 Effects of hunting on movement

In all but one region, increasing or constant numbers of bulls were hunted each year between 2006 and 2011 (Figure 2.2B). In those regions where more than three bulls were hunted each year, the numbers counted ranged between zero and 11 animals (Zimbabwe and Mapungubwe; Figure 2.5A). However, where six or more animals were shot in a year, the number in the census was never greater than 13, whereas when hunting occurred less often, up to 25 bulls occurred in the region and where hunting was absent, up to 35 bulls were counted. Female numbers and bull numbers were not correlated in any region for the six years counted (South

Africa: $r^2 = 0.674$, $P = 0.142$; Northern Tuli: $r^2 = 0.137$, $P = 0.796$; Tuli Block: $r^2 = 0.714$, $P = 0.111$; Zimbabwe: $r^2 = 0.137$, $P = 0.796$; $n = 6$). The number of bulls hunted in a particular year negatively affected the number of females observed within each region in the same year ($F_{1,22} = 5.564$; $n = 24$; 1-tailed $P = 0.015$). Where one or no animals were shot in a locality per year, more females (>320) occurred, and conversely where hunting of bulls was greater, fewer females occurred (Figure 2.5B). Since habitats were almost identical, and no other factors limited distribution, these differences can be ascribed to the disturbance effects of hunting bulls on females.

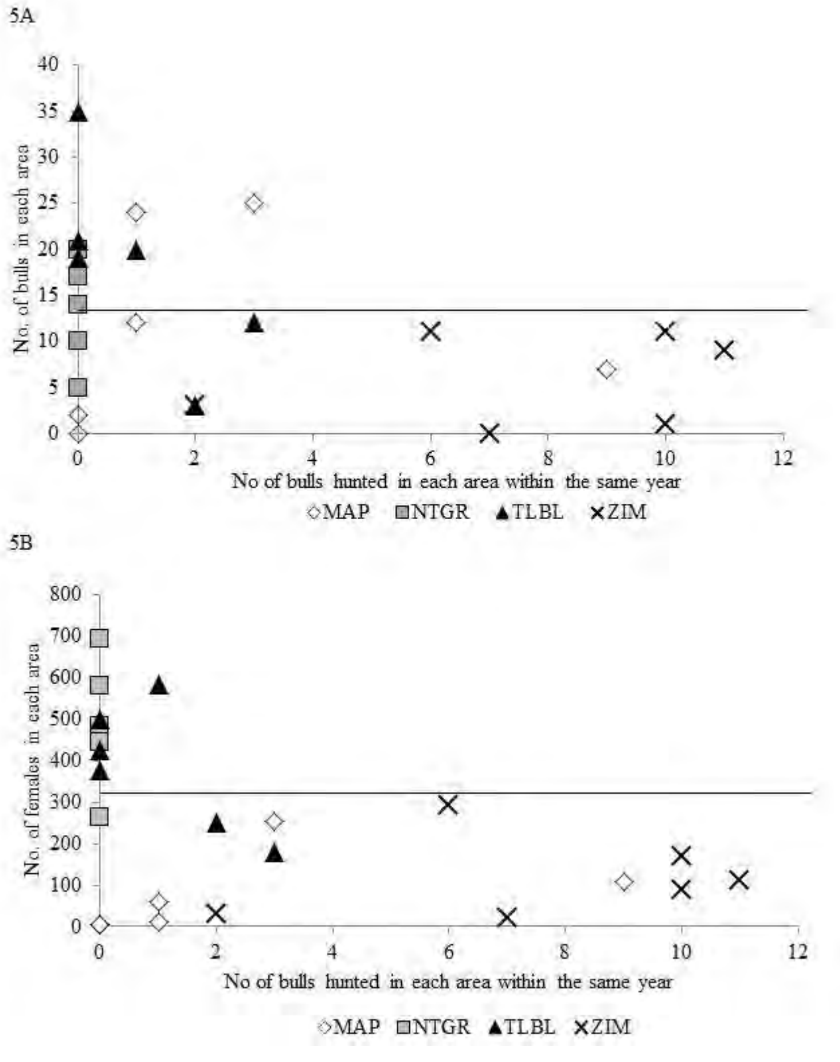


Figure 2.5: Effect of hunting of bulls in the hunting season immediately prior to the aerial survey in each of the four regions of the Greater Mapungubwe Transfrontier Conservation Area on the number of bulls counted (A) and the number of females in each region (B). Each point represents a count in a different year and the symbols indicate different localities (MAP – the South African section, including Mapungubwe National Park and private properties bordering the Limpopo River where elephant had access; NTGR – Northern Tuli Game Reserve; TLBL – Tuli Block from the Motloutse River to Baines Drift and ZIM – the Zimbabwean section along the Limpopo River, including Maramani, Sentinel Ranch, and Nottingham Estate up to the Umzingwane River). The horizontal line in A indicates the maximum number of bulls counted (13) when six or more animals were shot in a year. The horizontal line in B indicates the maximum number of females counted (320) when ≥2 bulls were hunted in a year.

2.5 DISCUSSION

Wildlife resources in Africa have long been hunted for sport, subsistence, and to control population size (Festa-Bianchet 2003). Trophy hunting targets the largest males or those with impressive horns, tusks, or antlers (Ginsberg and Milner-Gulland 1994, Milner et al. 2007). Even though it is generally restricted to a few individuals, where controls are lacking a high proportion of those individuals that qualify can be removed annually (Coltman et al. 2003, Crosmary et al. 2013). High levels of hunting are thus often not a sustainable use of wildlife resources (Baker 1997, Milner et al. 2007). This is especially true of long-lived or large species, such as elephant, with low intrinsic rates of increase (Archie et al. 2008, Fa and Brown 2009).

Several factors, including environmental conditions, influence the number of elephant that can be hunted per annum. Our model showed that selective hunting under all three environmental conditions tested, not only can have a direct effect on reducing population size (Milner et al. 2007, Allendorf et al. 2008) but also can bias the sex-ratio in favour of females and heavily skew the age structure towards younger animals. Undisturbed elephant populations have only a slightly skewed sex ratio favouring females (Poole and Thomsen 1989, Wittemyer 2001). Thus selective hunting consequently could have an effect on reproduction (Ginsberg and Milner-Gulland 1994, Milner et al. 2007, Allendorf et al. 2008). In several species, including saiga antelope (*Saiga tatarica*; (Milner-Gulland and Bennett 2003)) and elephant (Dobson and Poole 1998), a sex-ratio threshold may exist (77 females per male for elephant), below which fecundity decreases as a result of insufficient male breeding capacity.

For a harvest system to be sustainable, consideration of its effect on age-dependent or size-dependent fecundity, growth, and survival rates of individuals, and the growth rate and age structure of the population is warranted (Fa and Brown 2009). For most species, older, high-value trophy animals are past breeding, and form approximately 10% of the total male population (Hurt and Ravn 2000). When considering that the majority of mating in an elephant population is done by bulls >35 years old (Poole 1989, Hollister-Smith et al. 2007), and that loosing these bulls can lead to social problems (Slotow et al. 2000), the maximum sustainable yield for social stability within the study area was predicted by the VORTEX model to be about 10 bulls >35 years old per annum. The models of Caughley (1993) and Martin *et al*. (1997) both suggest even more conservative rates. Harvesting quotas should further take into account other sources of mortality, such as problem animal control and natural mortalities (Baker 1997). These are rarely included in the calculation of quotas (Baker 1997, Caro 1999). The current levels of trophy hunting are thus unsustainable and far exceed the MSY of the population. Even though hunting brings in revenue to local communities, the overall value to the area is relatively low, and at the present harvest rate, hunting is not sustainable.

During the six biennial aerial surveys, the greatest number of bulls counted was 88 in 2001 (Selier 2010). Several factors could contribute to the fewer number of bulls counted. Lone bulls are difficult to spot and a few were likely missed during the counts. Further younger bulls associated with breeding herds might have been assumed to be part of the breeding herd. These errors, however, do not account for the large difference between what the model predicted and what was observed during the aerial counts. Therefore, hunting over the past 10 years had likely already depressed bull numbers.

Old bull elephant also have greater reproductive success, and their longevity may further reflect greater fitness (Hollister-Smith et al. 2007, Ishengoma et al. 2008). In populations recovering from poaching, the lifetime reproductive output of dominant male elephant in the population increased (Ishengoma et al. 2008). Whether this has an impact on reducing genetic diversity in the population is still unclear (Ishengoma et al. 2008). Removing the primary male breeders in a population not only hampers reproduction and recruitment but could also disrupt the social organization (Milner et al. 2007, Whitman et al. 2007). In elephant, older bulls have a social network with high centrality and strong bonds (Archie and Chiyo 2012). Consequently, the elimination of older bulls from elephant populations may negatively affect social cohesion in bull elephant groups (Ishengoma et al. 2008, Archie and Chiyo 2012). Further, the selective removal of adult males over an extended period could result in a greater proportion of younger males (Milner et al. 2007), which may increase the reproductive tenure of these males (Poole 1989, Archie and Chiyo 2012). In the absence of older bulls, young bulls increase the frequency and duration of their musth period (Slotow et al. 2000). Abnormal behaviours in these young males, such as elevated aggression, killing people, and killing white rhino (*Ceratotherium sinum sinum*) have been the result of distorted male age hierarchies (Slotow et al. 2000, Slotow et al. 2001, Slotow and van Dyk 2001, Bradshaw et al. 2005).

Where communal areas occur on the periphery of protected areas, the open borders between protected and communal lands create a source-sink effect with animals constantly being removed from the periphery of the protected areas (Hoare 2000, Balme et al. 2010b). High human densities and conflicting land use practices (crop farming) draw elephant, especially bulls, towards community areas, primarily during periods of low natural food availability, thereby creating an ecological trap (Hoare and Du Toit 1999, Chiyo et al. 2005). Elephant can move from tourism areas where they may be wanted, to areas where they are unwanted such as community crop fields. Ongoing killing of problem animals on the periphery of protected areas erodes the quality of the remaining animals in terms of trophy quality (Hoare 1995) and genetic diversity (Archie and Chiyo 2012).

The aim of allocating hunting quotas in terms of problem animal control laws are to deter elephant from entering communal areas and to compensate local communities for damage to crops and property with the aim to improve the tolerance of communities towards elephant (M. Mamani, personal communication)[3]. We show, however, that hunting bulls is not an effective deterrent, as elephant return to the region within a year of the hunts. Similar results have been reported from Kasungu (Malawi) where high levels of poaching, of mainly bulls, led to additional males continuously moving into the area (Bell 1981). Younger bulls are more often responsible for crop raiding (Chiyo et al. 2005, Ahlering et al. 2011), but the older bulls are required for a good trophy income (Hurt and Ravn 2000, Festa-Bianchet 2003, Milner et al. 2007, Slotow et al. 2008). Regulations on damage causing animals differ between Botswana and South Africa. Botswana applies a clear and systematic process in dealing with damage causing animals (Wildlife conservation and National Parks Act no 28 of 1992), whereas South Africa at present, uses a more ad hoc and less systematic approach, which does not deal sufficiently with migratory cross-border movements of elephant (National Environmental Act 1998 and the National Norms and Standards for the Management of Elephant, 2008).

A further consequence of hunting bulls, not taken into account in the model is the disturbance factor. For levels of hunting much greater than or close to the numbers counted in each year to be sustained over the 5-year period, immigration into the areas where greater hunting pressure occurs is necessary. These results indicate that although bulls do not completely avoid areas with greater hunting pressure, fewer bulls entered these areas, than areas where less or no hunting occurred. Thus, sustained high levels of hunting in a region do not appear to cause bulls to avoid that region. Thus shooting trophy bulls does not alleviate the problem of conflict within Botswana. Given the systematic approach of Botswana towards problem causing animals, this approach would not likely be effective within South Africa. However, high levels of hunting of bulls caused a disturbance effect within breeding herds, possibly because of the high stress levels observed throughout the population during hunting disturbances (Burke et al.

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[3] Malatsi Mamani, Department of Wildlife and National Park, Bobonong, GMTFCA Elephant Conservation Policy and Management Planning workshop, Duncan Macfadyen Research Centre, De Beers Venetia Limpopo Nature Reserve, 5-6 May 2010.

2008). In a system where both consumptive and non-consumptive use is made of elephant, disturbance has major ramifications.

Photographic tourism is at present the main economic driver within the region (Evans 2010) and elephant are one of the main draws. Habituated viewable elephant, including large bulls with trophy size tusks, are important to the tourism industry (Blignaut et al. 2008, Slotow et al. 2008). Excessive hunting will therefore affect photographic tourism within the Limpopo Valley through significantly reduced numbers of big bulls, and could affect the chances of viewing elephant in general (Slotow et al. 2008, Di Minin et al. 2013a). Furthermore, because of its selective nature, trophy hunting may decrease the number of large-tusked individuals (Festa-Bianchet 2003). However, it is important that all stakeholders within transboundary areas benefit from wildlife, and a consultative process, which includes all stakeholders, is required to develop a sustainable non-consumptive and consumptive use plan with cost-benefit sharing for the area.

2.6 MANAGEMENT IMPLICATIONS

Current hunting quotas within the GMTFCA are unsustainable, and an urgent revision is required within each country with the establishment of a single multi-jurisdictional (cross-border) management authority regulating the hunting of elephant (and other cross-border species). The allocation of hunting quotas should be based on current data, taking into consideration the environmental conditions as well as the population dynamics and social structure of the species under consideration. Based on our results taking into consideration the social stability of the population and the current environmental conditions, the maximum sustainable yield is 10 bulls >35 years old per annum. Cooperation between countries, increased landscape connectivity, and the ability to generate income from tourism have been shown to work successfully in increasing wildlife numbers elsewhere (Plumptre et al. 2007).

A conservation planning assessment with the objective of enhancing biodiversity protection, while promoting sustainable development and improved quality of life for communities within the GMTFCA, is urgently required and should include all stakeholders. Where communities are included in the process and directly benefit from wildlife, either through consumptive or non-consumptive means, they are more likely to take ownership and the incentives to develop the land for arable purposes or livestock herds will be removed, thus benefiting biodiversity conservation (Hanks 2003).

2.7 ACKNOWLEDGMENTS

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CHAPTER 3: LARGE MAMMAL DISTRIBUTION IN A TRANSFRONTIER LANDSCAPE: TRADE OFFS BETWEEN RESOURCE AVAILABILITY AND HUMAN DISTURBANCE

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ABSTRACT

Understanding factors that affect the persistence of charismatic megafauna in human-dominated landscapes is crucial to inform conservation decision-making and reduce human-wildlife conflict. We assessed the effect of environmental and anthropogenic factors at different landscape and management scales in predicting the distribution of African elephant (*Loxodonta africana*) within the Greater Mapungubwe Transfrontier Conservation Area in Southern Africa. We combined aerial distribution counts over a 12-year period with fourteen variables, representing food availability, landscape, and anthropogenic effects, into generalised linear models. Generalised linear models were run for the broader landscape, as well as three separate management units within the broader landscape, namely ecotourism, trophy hunting, and a combination of hunting and ecotourism. Human activities within different management units forced elephant to trade-off between disturbance avoidance, and good food and water availability. In addition, the important predictors of elephant distribution within each of the management units differed from the predictors at the broader landscape. Overall, our results suggest that at the fine scale, elephant are constrained by factors that may be masked at the broader landscape scale. We suggest that accounting for anthropogenic disturbance is important in determining the distribution of large, wide-ranging, mammal species in increasingly human-dominated landscapes, and that modelling needs to be done at the spatial scales at which conservation decisions are made.

3.1 INTRODUCTION

Current rates of biodiversity loss and habitat transformation are unprecedented and on-going (Butchart et al. 2010). The magnitude of the negative impacts of anthropogenic activities on global biodiversity has been documented at several levels of biological organisation (Gaston et al. 2003). Large-bodied, wide-ranging, mammals are particularly sensitive to anthropogenic activities because they require large and well-connected patches to persist (Di Minin et al. 2013b). However, in some areas large mammals are also able to persist in human-dominated landscapes with a high degree of habitat fragmentation (Athreya et al. 2013). Ignoring anthropogenic factors when modelling species distribution may lead to an incorrect assessment of the habitat requirements that affect their persistence (Pearson and Dawson 2003). Hence, it is strategic to understand the relative importance of anthropogenic factors in affecting species distributions in human-dominated landscapes in order to address and alleviate the risk of local extinction (Cabeza and Moilanen 2001).

Species distribution models are extensively used in spatial conservation prioritisation (Araújo and Guisan 2006, Kremen et al. 2008, Blach-Overgaard et al. 2010, Drummond et al. 2010). Climate exerts a dominant control over the natural distribution of species (for reviews see Hughes 2000, Walther et al. 2002), and studies have often focused on the characterization of a species' bioclimatic envelope in order to predict distribution (Araújo and New 2007, Blach-Overgaard et al. 2010, Araújo and Peterson 2012). However, human activities can influence species distribution and behaviour, and may outweigh the sole influence of climate on species distribution (Erb et al. 2012, Llaneza et al. 2012, Murai et al. 2013). Great apes (Michalski and Peres 2005, Murai et al. 2013), as well as large carnivores and herbivores (Hoare 1999, Michalski and Peres 2005, Kinnaird and O'Brien 2012), for instance, are highly sensitive to anthropogenic activities. Legal and illegal harvesting of biodiversity can have direct effects through reduction of numbers and indirect disturbance effects (Llaneza et al. 2012, Selier et al. 2014) that ultimately influence species distribution (Basille et al. 2009, Graham et al. 2009, Llaneza et al. 2012). Hence, more studies are required that can investigate the role of anthropogenic disturbance on biodiversity distribution at different spatial scales (Guisan and Thuiller 2005).

While protected areas are fundamental for biodiversity persistence in increasingly human-dominated landscapes (Baeza and Estades 2010, Stokes et al. 2010), they are often too small to sustain viable populations of large mammals (Graham et al. 2009, Di Minin et al. 2013b), as they cannot meet the space requirements of wide-ranging or migratory species (Woodroffe and Ginsberg 1998, Graham et al. 2009, Stokes et al. 2010, Di Minin et al. 2013b). In addition, protected areas are becoming more and more vulnerable to anthropogenic activities and other environmental stressors (Sinclair and Byrom 2006, Laurance et al. 2012), leading to further fragmentation of habitats (Fischer and Lindenmayer 2007) that, and combined with, patch sizes, degree of connectivity, habitat quality, and the level of human disturbances, affect biodiversity persistence within protected areas (Michalski and Peres 2005, Chazdon et al. 2008). Conservation areas are often embedded within a mosaic of different land uses such as agriculture, cattle grazing, commercial forestry and mining (DeFries et al. 2007, Chazdon et al. 2008, Di Minin et al. 2013c), as well as falling under a range of management strategies (Loveridge et al. 2007). Incorporating management practices, which are compatible with biodiversity conservation, not only helps protecting critical habitats for a variety of species (Gardner et al. 2007), but also contributes to maintaining landscape connectivity (DeFries et al. 2007). In addition, understanding which anthropogenic activities affect the spatial distribution of large mammals can help prevent human-wildlife conflict (Athreya et al. 2011).

The goal of this study was to understand how anthropogenic activities can affect the distribution of a large, wide-ranging, mammal species at different spatial scales. Specifically, we investigated the relative importance of environmental and anthropogenic factors in affecting the distribution of African elephant (*Loxodonta africana*), in the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA) in Southern Africa. The African elephant attracts high numbers of visitors to conservation areas in sub-Saharan Africa as a flagship species (Di Minin et al. 2013a). Elephant conservation also benefits other species through an umbrella effect (Di Minin et al. 2013a, Di Minin and Moilanen 2014), and elephant are a keystone species that affects ecosystem structure and function. As elephant in the study area are exposed to a range of management practices, which prevent them from using the landscape freely, we also wanted to understand whether elephant trade-off between resource selection and avoidance of anthropogenic disturbance, at different spatial scales.

3.2 METHODS

3.2.1 Study area

This study was undertaken within the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA), in Botswana, South Africa and Zimbabwe (Figure 3.1). The GMTFCA covers 3650 km^2 centred on the confluence of the Shashe and Limpopo Rivers. The region is semi-arid with low, unpredictable, rainfall (Harrison 1984) that averaged 365 mm annually between 1966-2001 (Selier 2007). Summer maximum temperatures can exceed 42 ºC, while winter minimum temperatures can be as low as -5 ºC (Mckenzie 1990). The elephant population in the GMTFCA consists of approximately 1224 ± 72.4 individuals (Selier et al. 2014).

The study area is characterized by a human-dominated landscape with a range of land use and management practices (Table 3.1). For the purpose of this paper, three main conservation orientated management units within the GMTFCA were selected: i) ecotourism (Northern Tuli Game Reserve and Mapungubwe National Park), ii) trophy hunting (Tuli Circle, and Mapungubwe Private Reserve) and iii) management units in which a combination of ecotourism and hunting is practiced (Sentinel Ranch, Nottingham Estate and the Tuli Block to Baines Drift) (Figure 3.1). Electric fences restrict the movement of elephant and other wildlife in certain sections. These fences extend along the western boundary of the Northern Tuli Game Reserve (NTGR), the northern boundary of the Tuli Block and along the Limpopo River on the South African side, with a gap in the fence known as the Vhembe gap around the confluence of the Limpopo and Shashe rivers (Figure 3.1). Water provisioning differs across sections of the study area, with the highest density of artificial waterholes (roughly a waterhole every 3-5 km) within the NTGR and Mapungubwe National Park. Across southern Africa water provisioning is a standard management procedure used by many wildlife managers (Smit et al. 2007a). Water provisioning can influence the movement patterns, home range utilisation and size and the impact that elephant have on local vegetation (van Aarde et al. 2008).

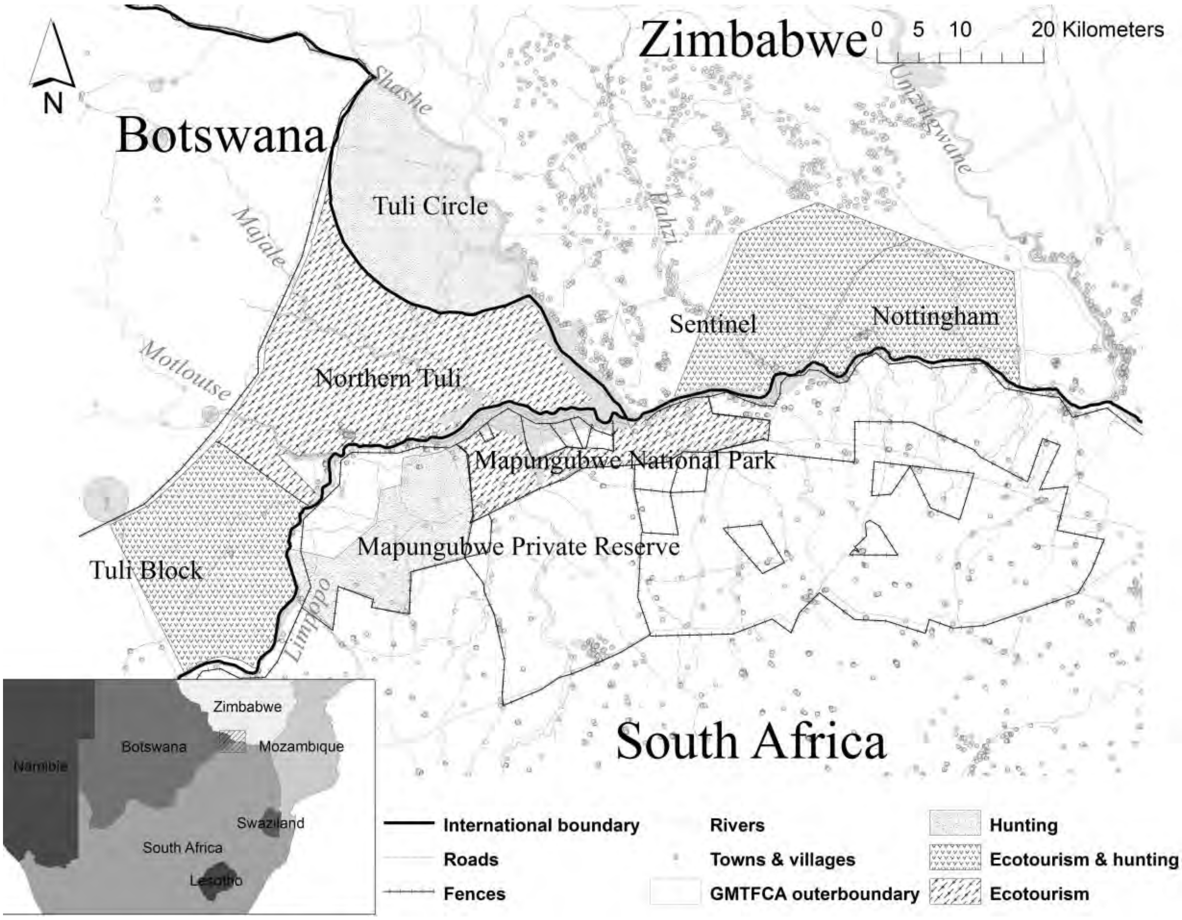

Figure 3.1: The Greater Mapungubwe Transfrontier Conservation Area and surrounding areas illustrating the locations of the three different management units.

Table 3.1: Land use areas within the three countries forming the Greater Mapungubwe Transfrontier Conservation area, the size of each area, approximate people densities and main land use.

Area	Tenure	Area (km^2)	Approx. people density (persons/km^2)	Main land use
Botswana				
Northern Tuli Game Reserve	Private game reserve	720	<1	Ecotourism
Talana Farms	Private land	18	2-3	Commercial agriculture
Tuli Block	Private land	739	<1	Ecotourism, hunting, subsistence agriculture, subsistence livestock farming
Bobirwa sub district	Communal land	13205	2-7	Subsistence agriculture, subsistence livestock farming, hunting
South Africa				
Mapungubwe National Park	State land	215	<1	Ecotourism
Mapungubwe Private Reserve	Private game reserve	232	<1	Hunting
Venetia	Private game reserve	345	<1	Hunting, ecotourism

SA farms	Private land	64.78		Commercial agriculture, hunting
Zimbabwe				
Tuli Circle	State land	550	<1	CAMPFIRE hunting
Maramani	Communal land	490	10.61	Subsistence agriculture, Irrigation scheme, subsistence livestock farming, CAMPFIRE hunting
Sentinel Ranch	Private land/reclaimed land	320	<1	Ecotourism, hunting, wildlife ranching
Nottingham Estate	Private land/reclaimed land	250	<1	Commercial citrus farming, ecotourism, hunting
River Ranch	Resettled state land	170	1.47	Subsistence agriculture, Irrigation scheme, subsistence livestock farming, CAMPFIRE hunting
Machachuta	Communal land	760	6.38	Subsistence agriculture, Irrigation scheme, subsistence livestock farming, CAMPFIRE hunting
Masera	Communal land	340	7.76	Subsistence agriculture, Irrigation scheme, subsistence livestock farming, CAMPFIRE hunting

3.2.2 Elephant distribution data

Data on the distribution of elephant within the GMTFCA were obtained from aerial surveys conducted within the study area over the period 2000-2012 (Selier 2012). We combined location point data from all counts to describe the overall presence (and absence) of elephant based on a 427.5 m x 427.5 m grid. The number of presence points within the whole GMTFCA and ecotourism, hunting, and mixed management units were 561, 278, 16 and 233 respectively. An equal number of absence to presence data (except at presence locations) were randomly sampled for the GMTFCA, as well as in each of the three management units (Franklin and Miller 2009).

3.2.3 Explanatory variables

We selected 14 environmental and anthropogenic variables expected to affect elephant distribution (Table 3.2). Landscape composition and structure (Murwira and Skidmore 2005), food and water availability (De Beer and van Aarde 2008), rainfall-related change in food quality and water availability (Chamaillé-Jammes et al. 2008), elephant social structures (Harris et al. 2008, Young et al. 2009a), management activities and distance to human settlements (Hoare 1999, Selier et al. 2014) are all thought to influence the local distribution of elephant (van Aarde et al. 2008).

The topographic attributes of the landscape such as elevation, slope and aspect also influence the distribution of elephant through direct or indirect effects on resource availability and habitat suitability. Elephant tend to avoid steep slopes (Nellemann et al. 2002), and prefer certain habitat types such as riparian environments and wetlands (Kinahan et al. 2007, Smit et al. 2007b, Harris et al. 2008). We used a digital elevation model to describe the primary topographic attributes of the landscape (Hof et al. 2012). Elevation data were extracted from Aster Global Digital Elevation Model v002 (ASTG TM) at 30 m resolution (Table 3.2). The digital elevation model was used as an indirect measure for other non-climatic related factors that may restrict species geographically (e.g. food species distribution) (Ngene et al. 2009, Smith et al. 2012), and as a surrogate for spatial variation in temperature and precipitation (Hof et al. 2012). Slope angle is related to overland and subsurface flow of water, and, therefore, affects potential soil moisture and soil characteristics (Franklin and Miller 2009). Aspect also has an influence on soil moisture availability (Franklin and Miller 2009). Slope and aspect were constructed from the digital elevation model using Spatial analyst in ArcGIS (version 9.3; Environmental Systems Research Institute (ESRI), Inc., CA, USA).

Food and water availability and distribution are key ecological drivers affecting elephant distribution (Chamaille-Jammes et al. 2007b, Ngene et al. 2009, Shannon et al. 2009). To determine the water availability within the study area we calculated the shortest distance to water, combining all rivers, dams and artificial waterholes, using the Euclidean distance function in the packages *Raster* (Hijmans and van Etten 2012) and *SP* (Pebesma and Bivand 2005) in R 2.15.2 (R Development Core Team 2012) at a 427.5 m resolution. The Enhanced Vegetation Index (EVI) was used as a measure of vegetation productivity, and thus the amount of forage available to elephant (Pettorelli et al. 2005, Young et al. 2009b). The EVI data were downloaded for the period January 2000 to December 2012 (Table 3.2). The EVI time series was produced from the NASA 500 m, 8-day, BRDF-corrected, surface reflectance data (MCD43A4) (CSIR-Meraka Institute 2011). The geometric mean of the 8-day composites for the end of the wet seasons for all years and the dry season for all years of the aerial counts respectively were used as a measure of the vegetation productivity for the specific season. Wet and dry season EVI were found to be highly correlated. Hence, we only kept the average wet season EVI values. A vegetation map (Table 3.2), describing the broad vegetation types was included, as forage biomass and quality can differ amongst vegetation types (Harris et al. 2008), thereby influencing elephant distribution (Chamaille-Jammes et al. 2007b, Loarie et al. 2009a). Soil characteristics may also determine forage quality for elephant (Fritz et al. 2002). Hence, a soil classification map (Table 3.2) was also included in the analysis. The original rasters were converted to coarser resolution by aggregating the data up to 427.5m resolution and summing up the pixel values in blocks of four cells using Spatial Analyst in ArcGIS (version 9.3; Environmental Systems Research Institute (ESRI), Inc., CA, USA).

Table 3.2: Different variables used in the generalised linear models for the broader landscape and different management units within the Greater Mapungubwe Transfrontier Conservation Area.

Variable	Data type	Data origin	Source
Elevation	ASTG TM, Raster, Continuous data	Aster Global Digital Elevation Model v002 (ASTG TM)	http://asterweb.jpl.nasa.gov/gdem.asp
Slope	Raster, Continuous data	Calculated from Global DEM using ArcGIS 9.3	
Aspect	Raster, Dummy variables, north; east; south; west	Calculated from Global DEM using ArcGIS 9.3	
Soil types	Categorical data	Peace Park Foundation	http://www.peaceparks.org
Vegetation types	Categorical data	Peace Park Foundation	http://www.peaceparks.org
Hydrology	Raster, Euclidean distance from	Peace Park Foundation	http://www.peaceparks.org
Resource availability	EVI (MCD43A4), Raster, Continuous data	CSIR-Meraka Institute 2011, 8 -day composites	http://wamis.meraka.org.za/products/long-term-time-series

Roads	Raster, Euclidean distance from	Peace Park Foundation	http://www.peaceparks.org
Fences	Raster, Euclidean distance from	Peace Park Foundation	http://www.peaceparks.org
Villages, agriculture	Raster, Euclidean distance from	Peace Park Foundation	http://www.peaceparks.org
Trophy hunting	Raster, Euclidean distance from	Point data obtained from respective wildlife departments, reserve managers & landowners	

In areas where human and elephant range overlap, spatial factors such as human density, land transformation, agriculture, roads and proximity to protected areas also influence the distribution of elephant (Hoare and Du Toit 1999, Parker and Osborn 2001, Sitati et al. 2003, van Aarde and Jackson 2007). Human activities such as the erection of fences (Boone and Hobbs 2004, Vanak et al. 2010) and hunting (Burke et al. 2008, Selier et al. 2014) can further influence the distribution of elephant. Since 2006, trophy hunting of elephant within and on the periphery of the GMTFCA has steadily increased (Selier et al. 2014), and this increase in hunting could influence the distribution of the population. Distance from human settlements, fences, roads and hunting locations were thus included as human disturbance variables that could possibly influence the distribution of elephant within the GMTFCA (Table 3.2). Euclidean distance surfaces from these variables with a resolution of 427.5 m were calculated in R 2.15.2 (R Development Core Team 2012) using the packages *Raster* (Hijmans and van Etten 2012) and *SP* (Pebesma and Bivand 2005). Data on trophy hunting quotas and kill rates were obtained from the respective wildlife departments, private landowners and reserve and farm managers from the three countries for the period 2000-2012. A Euclidean distance surface with a resolution of 427.5 m was calculated in R 2.15.2 (R Development Core Team 2012) using all hunting points for the entire region and extracted for each of the management units.

3.2.4 Statistical analysis

The statistical analyses were conducted in R (version 2.15.2) (R Development Core Team 2012). We used an information theoretic approach (Burnham and Anderson 2002) and Akaike's information criterion (AIC) to calculate statistical models. We modelled elephant distribution in the broader GMTFCA landscape, and in each of three different management units (ecotourism, hunting and a combination of ecotourism and hunting) within the GMTFCA using a generalised linear model (GLM) with logit-link and binomial distribution of errors. The presence/absence data for elephant was used as the response variable. We determined the magnitude and direction of the coefficients for the independent variables with multimodel averaging implemented in the *R* (version 2.15.2) (R Development Core Team 2012) package glmulti (Calcagno 2010). The relative importance of each predictor variable was measured as the sum of the Akaike weights over the six top-ranked models containing the parameter of interest (Conroy and Brook 2003). We also assessed each model's structural goodness of fit using the percentage of deviance explained by the model. The top-ranked models for each scenario were validated by using leave-one out cross validation, which is used to estimate the mean model-predictor error by successively omitting one observation from the training data set and using it for validation. Finally, we controlled for spatial autocorrelation in the model residuals by using Moran's I (Dormann et al. 2007), which can be considered a spatial equivalent to Pearson's correlation coefficient and normally varies between 1 (positive autocorrelation) and -1 (negative autocorrelation). The expected Moran's I value for lacking spatial autocorrelation is close to 0.

To avoid multicolinearity among explanatory variables, we retained the variables with the greatest explanatory effect on elephant presence that were not strongly correlated (Franklin and Miller 2009). We tested for correlation using the *Corrgram* package in R 2.15.2 (Wright 2012), with a cut-off of $r = 0.80$. As explained above, wet and dry season vegetation productivity variables (EVI's) were strongly correlated. As a result, we only used wet season EVI (averaged over all counting years) in the analyses.

The resource selection probability function in the package *Raster* (Hijmans and van Etten 2012) in R (R Development Core Team 2012) was then used in combination with the variables with the highest relative importance (≥ 0.8) in the top-ranked models, to predict the distribution of elephant within each management unit and the broader landscape. The distribution models for each management unit were then merged into a single map to calculate the differences in predicted distribution. The difference between the predicted distribution models was calculated, by subtracting the grid cell value of the broad scale distribution model from the grid cell value of the distribution model developed at the management unit level. As a result, grid cells with a value of 0 represent areas where the broad landscape scale distribution model and the distribution model developed at the management unit level matched. Grid cells with a negative value represented areas where the distribution model developed at the management unit level under-predicted elephant distribution compared to the broad-scale model. Grid cells with a positive value were areas where the distribution model developed at the management unit level over-predicted elephant distribution compared to the broad scale model.

3.3 RESULTS

The results of the generalised linear models for the broader landscape and for each of the management units are summarised in Table 3.3. For the broader landscape, eight variables affected elephant distribution. These were distance from water, vegetation, soil, northern aspect and the human disturbance variables (distance from fences, distance from human settlements and distance from hunting). Both vegetation and soils are categorical variables. In total eight vegetation types and eleven soil types affected elephant distribution (Table 3.4). The selected model had an AIC weight of 0.22 (Table 3.3) and had a mean prediction error of approximately 20% using leave-one-out cross validation. For the ecotourism management unit, elevation, distance from hunting, and wet season vegetation productivity (EVI) affected elephant distribution (Table 3.3). The model had an AIC weight of 0.23 (Table 3.3) and a mean prediction error of approximately 23%. Only two variables were contained in the best model for the hunting management unit (Table 3.3), namely vegetation (five vegetation types) (Table 3.4) and distance from hunting. The selected model had an AIC weight of 0.40 (Table 3.3) and a mean prediction error of approximately 28%. For the mixed management unit, distance from water, vegetation (seven vegetation types) (Table 3.4), as well as wet season vegetation productivity (EVI) affected elephant distribution (Table 3.3). The model had an AIC weight of 0.23 (Table 3.3) and a mean prediction error of approximately 19%. Overall, the explanatory power of our models (the percentage of deviance explained) increased from the ecotourism model (33.3%) to the broader landscape model (41.1%) (Table 3.3). All top-ranked models had relatively low spatial autocorrelation in the residuals (Table 3.3).

The distribution of elephant, which was predicted based on the significant variables in the top-ranked models (Figure 3.2), varied between the broader landscape model (Figure 3.3A) and each management unit (Figure 3.3B). Specifically, we found that 30% of the grid cells in the fine scale model had a higher value, 33% had the same value and 37% had a lower value compared to the broad scale model (Figure 3.4).

Table 3.3: Top-ranked predictors of elephant distribution within the Greater Mapungubwe Transfrontier Conservation Area.

	Model	No of variables	Delta	AIC Weight	% dev. expl.	Moran's I	Cross validation
All landscape	Soil*+Veg*+Fen+Hun+Asp N+Vill+Wat+EVI	8	0.00	0.22	0.411	0.083 ± 0.005	20%
	Soil*+Veg*+Fen+Hun+Asp N+Slo+Vill+Wat+EVI	9	0.09	0.21	0.408		
	Soil*+Veg*+Fen+Hun+Asp N+Road+Vill+Wat+EVI	9	0.11	0.21	0.414		
	Soil*+Veg*+Fen+Hun+Asp N+Road+Slo+Vill+Wat +EVI	10	0.35	0.19	0.411		
	Soil*+Veg*+Fen+Hun+Asp N+Slo+Vill+Asp W+Wat +EVI	10	1.97	0.08	0.411		
	Soil*+Veg*+Ele+Fen+Hun+Asp N+Slo+Vill+Wat +EVI	10	2.03	0.08	0.411		
Ecotourism	Ele+Hun+Slo+EVI	4	0.00	0.24	0.333	0.044 ± 0.008	23%
	Ele+Hun+EVI	3	0.06	0.23	0.291		
	Ele+Hun+Slo+Wat+EVI	5	0.39	0.20	0.344		
	Ele+Fen+Hun+Slo+EVI	5	1.42	0.12	0.341		
	Ele+Hun+Slo+Vill+EVI	5	1.58	0.11	0.338		
	Ele+Hun+Asp S+Slo+EVI	5	1.68	0.10	0.334		

Hunting	Veg*+Hun	2	0.00	0.40	0.346	0.071 ± 0.015	28%
	Veg*+Hun+Wat	3	0.16	0.37	0.333		
	Veg*+Asp E+Hun+Wat	4	3.12	0.08	0.344		
	Veg*+Hun+Asp N+Wat	4	3.27	0.08	0.349		
	Veg*+Hun+Asp W+Wat	4	3.75	0.06	0.354		
	Veg*+Asp E+Hun+Asp W+Wat	5	7.06	0.01	0.265		
Mixed land use	Veg*+Ele+Wat+EVI	4	0.00	0.23	0.374	0.098 ± 0.013	19%
	Ele+Road+Wat+EVI	4	0.06	0.22	0.370		
	Veg*+Ele+Road+Wat+EVI	5	0.27	0.20	0.367		
	Veg*+Ele+Road+Vill+Wat+EVI	6	0.83	0.15	0.371		
	Veg*+Ele+Road+Slo+Wat+EVI	6	1.65	0.10	0.359		
	Veg*+Asp E+Ele+Road+Wat+EVI	6	1.69	0.10	0.369		

Note: Veg, vegetation (categorical variable*); soil, different soil types (categorical variable*); Fen, distance from fences; Hun, distance from hunting; Asp N, northern aspect; Vill, distance from human settlements; Wat, distance from nearest water source; EVI, average end of wet season resource availability; Slo, slope; Road, distance from the nearest road; Ele, elevation; Asp S, southern Aspect; Asp E, eastern aspect; Asp W, western aspect. Plus signs imply additive terms in the model.

Table 3.4: Important soil and vegetation types predicting elephant distribution in the top-ranked models.

	Soil types	Vegetation types
All landscape	Calcaric cambisols (Very deep coarse loamy soils)	Guibourtia mixed woodland
	Calcaric regosols (Shallow to moderate shallow fine loamy to clayey soils	Jubernardia woodland (Granophyre)
	Chromic Cambisols (Very deep coarse loamy soils)	Limpopo Ridge bushveld
	Chromic luvisols (moderate depth clayey soils)	Mopani woodland (Granophyre)
	Cutani-Profondic Luvisols	Riparian woodland (Alluvium)
	Eutri-Arenic regosols (Shallow to moderate shallow fine loamy to clayey	Riverbed
	Eutric Arenosols (Very deep sandy to coarse loamy soils)	Sandbanks
	Eutric leptosols (Shallow to moderate shallow fine loamy soils)	Thicket woodland (Mopane dominant)
	Haplic luvisols (Moderate depth clayey soils)	
	Leptic regosols (Shallow to moderate shallow fine loamy soils)	
	Rhodic Cambisols (Very deep coarse loamy soils)	
	Rubic arenosols (Very deep sandy to coarse loamy soils)	
Hunting		Guibourtia mixed woodland
		Limpopo Ridge bushveld
		Mopani woodland (Granophyre)
		Riparian woodland (Alluvium)
		Riverbed
Mixed land		Guibourtia mixed woodland
		Jubernardia woodland (Granophyre)
		Mopani woodland (Granophyre)
		Riparian woodland (Alluvium)
		Riverbed

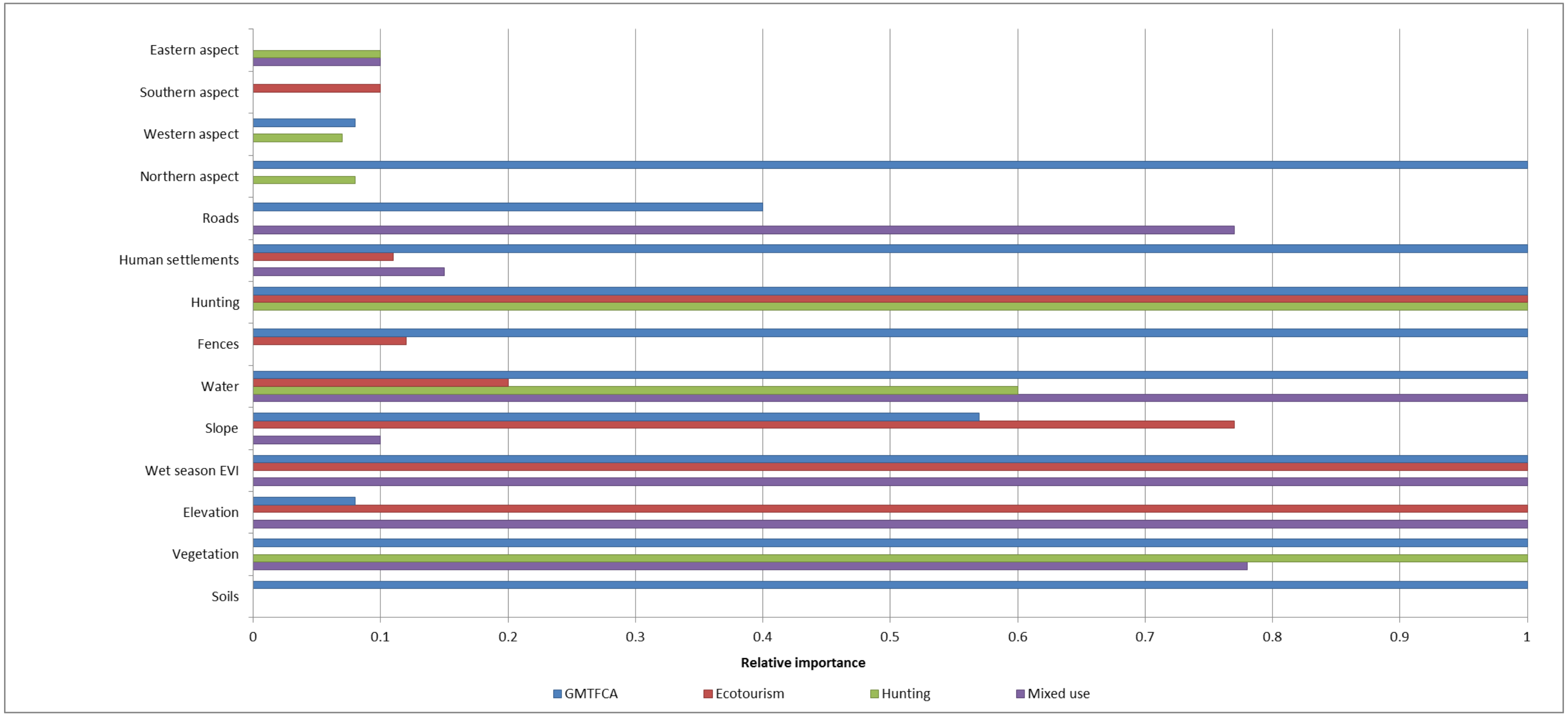

Figure 3.2: The relative importance of the most important variables affecting elephant distribution within the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA) and in each of the management units (ecotourism, hunting, and mixed use). The response variable is the elephant distribution data collected between 2000–2012.

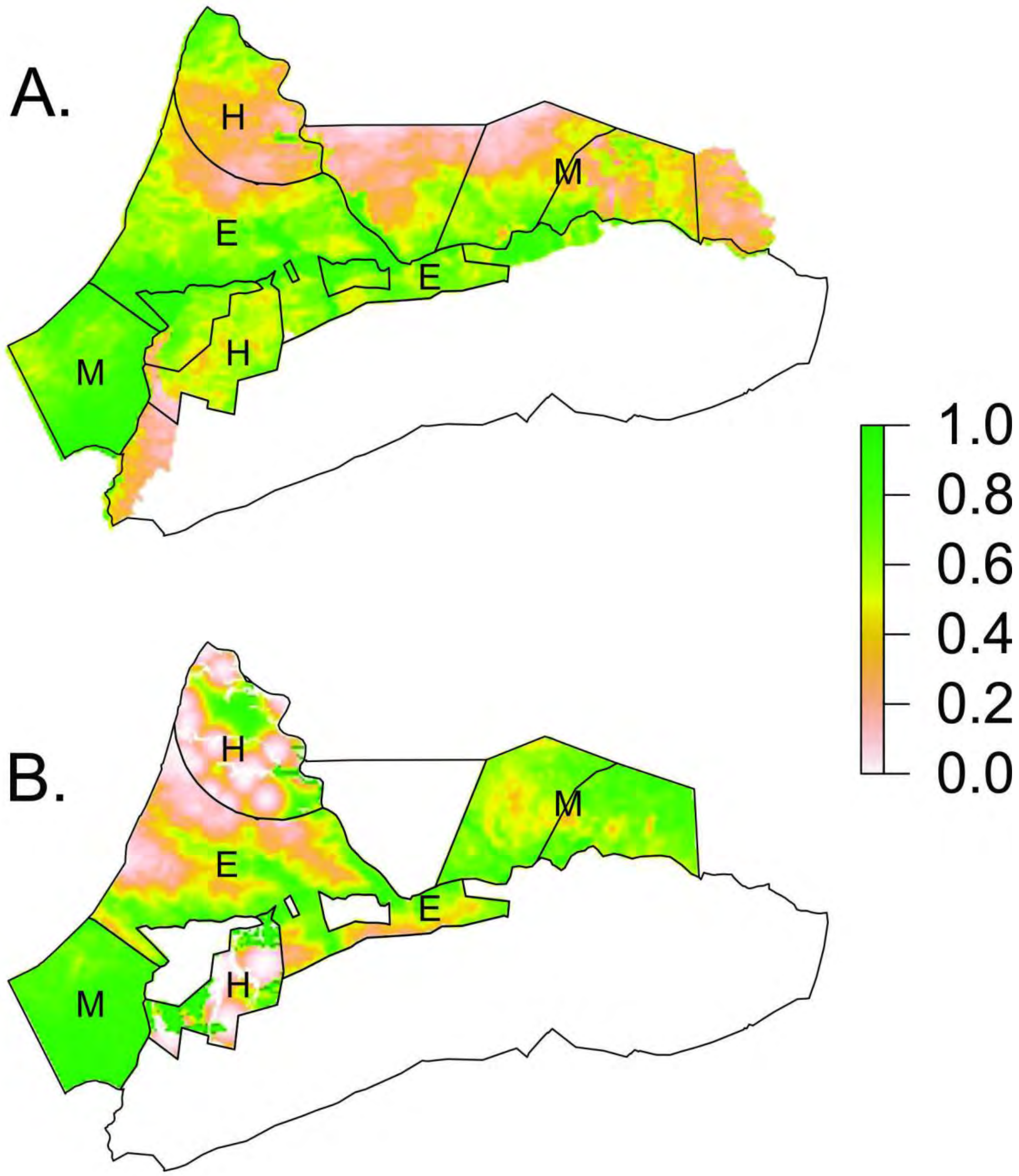

Figure 3.3: A comparison of the predictive distribution of elephant on a broad landscape scale compared to the fine scale management units. The predictive distribution of elephant modelled based on the significant variables in the final models for the broader landscape (3.3A), and for each land use type (Fig. 3.3B). Green indicates a high probability for elephant to occur in the area and white a very low probability. E indicates the areas where ecotourism is the main land use, H where trophy hunting is the main land use and M where both trophy hunting and ecotourism are carried out.

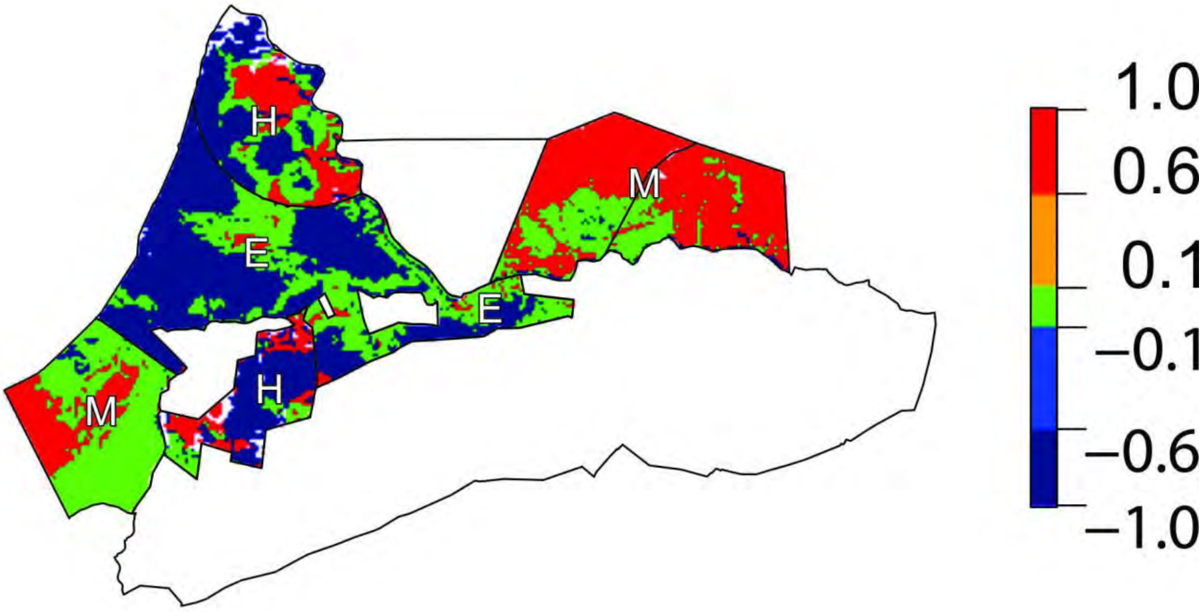

Figure 3.4: The difference in predictive distribution between the broad scale model and the combined fine scale management units. Green corresponds to areas predicted as suitable by both the broad and fine scale models, thus areas where there is a match between the fine and broad scale predictions, dark blue and blue correspond to where the fine scale model under estimated and orange and red where the fine scale model overestimated the probability of elephant presence. E indicates the areas where ecotourism is the main land use, H where trophy hunting is the main land use and M where both trophy hunting and ecotourism is offered.

3.4 DISCUSSION

We used generalised linear models to investigate how different environmental and anthropogenic variables affected the distribution of elephant at different spatial scales. We ran models for the broader landscape and three separate management units (ecotourism, hunting and a combination of ecotourism and hunting). Overall, our results suggest that accounting for anthropogenic disturbance is important in determining the distribution of a large, wide-ranging, mammal species such as elephant. Anthropogenic activities within different management units, in fact, forced elephant to trade-off between disturbance avoidance and good food and water availability. Remarkably, elephant distribution was affected by trophy hunting carried out in neighbouring management units, even when ecotourism was the main conservation land use.

In human-dominated landscapes, individuals constantly trade-off resource availability and risk avoidance, adapting their ranging and foraging behaviour to avoid unexplored areas (Druce et al. 2008) and human-induced disturbances (Hernández and Laundré 2005). Elephant have been found to use space in a manner that reduces contact with humans, for example, by altering their drinking behaviour (Jackson et al. 2008), avoiding areas close to human settlements (Graham et al. 2009), adopting different day-time and night-time behaviour (Douglas-Hamilton et al. 2005, Graham et al. 2009), increasing their rate of movement (Douglas-Hamilton et al. 2005), and leaving areas entirely when human densities reach a certain threshold (Hoare and Du Toit 1999). Our results confirm these findings, but also suggest that anthropogenic activities at the management unit level prevent them from avoiding disturbance. While elephant avoided hunting on the broad scale and within the ecotourism and hunting management units, they did not avoid hunting within the mixed management unit. The electric fences and, possibly, the lower hunting intensity (Selier et al. 2014) in the area can potentially explain this result.

However, we have to caution against generalising results for the hunting management unit due to the small sample size within this management unit, which can influence the accuracy of the predictive distribution of the species (Cumming 2000, Stockwell and Peterson 2002, McPherson et al. 2004). While the few data points within the management unit might be a reflection of the landscape scale decisions made by elephant on where to be within the broader landscape (Selier et al. 2014), they might not reflect effectively the fine-scale selections made by elephant. Regardless, future studies should focus more on hunting areas to better understand how different hunting intensities can influence elephant distribution (Slotow et al. 2008, Burton et al. 2012, Selier et al. 2014).

The distribution of resources such as food and water were also key ecological drivers determining the distribution of elephant in the landscape (Chamaillé-Jammes et al. 2008, van Aarde et al. 2008, Shannon et al. 2010). In two of the four models, elephant were attracted to water. In dry savannahs, water is the main driver determining the spatial use of elephant (Chamaille-Jammes et al. 2007b, Smit et al. 2007b, Young et al. 2009a). In the broader landscape, soils along the Limpopo River such as Eutric arenosols and Haplic luvisols were selected for by elephant. Furthermore, elephant also selected for riparian woodland in the broader landscape model and the mixed management unit (Chamaille-Jammes et al. 2007a, Loarie et al. 2009b, a, Shannon et al. 2009). Because water and nutrients accumulate in valley slopes and smaller depressions (Ben-Shahar 1996), these areas can serve as nutrient hotspots attracting a variety of herbivores due to the higher biomass (Bergman et al. 2001), and higher quality, of forage available in these areas (Nellemann et al. 2002, Grant and Scholes 2006).

The way elephant use space is likely to depend on a combination of biological, behavioural and ecological processes that may work at various scales (Douglas-Hamilton et al. 2005, Jachowski et al. 2013). At the landscape scale, the decision is made on where to be within the landscape, and then on the finer spatial scale, the decision is on how to utilise the local resources (Murwira and Skidmore 2005). This study has shown that anthropogenic activities such as hunting can influence the decision making processes of species at both the landscape and fine spatial scales, resulting in changes on where the species is distributed and how the species utilises its environment. These changes in how a species, particularly megafauna such as elephant, utilise their landscape can have significant effects on ecosystem structure, resulting in cascading effects such as the loss of large trees (Shannon et al. 2011). When management decisions are not made at the appropriate decision level for the species or system under consideration, a mismatch in spatial scale occurs with implications for the effective management and conservation of the species (Cumming et al. 2006). This has far reaching implications for cross-border species, where the stress effects of hunting could be transmitted to ecotourism areas within neighbouring countries (Delsink et al. 2013). The mismatch of spatial scales further has implications for highly-managed threatened species where management actions might affect the persistence of a species, either as a result of the species utilising lower quality resources due to disturbance effects or through an increase in human-wildlife conflict (Ciuti et al. 2012). This study further suggests that management regimes within, and neighbouring, proposed wildlife corridors should be taken into consideration as these corridors are unlikely to be used if disturbance effects from management actions, for example such as hunting, are high.

In conclusion, this study supports the idea that localised human activities strongly affect the distribution of wide-ranging species in human-dominated landscapes (Baeza and Estades 2010, Erb et al. 2012). Hence, it is important that future studies will consider human disturbance at both a regional- and local scale when modelling species distribution (Erb et al. 2012). Particularly, these results have important implications for the effective allocation of conservation actions that can enhance the long-term persistence of wide-ranging species in human-dominated landscapes. Factors such as human disturbance need to be taken into account when modelling the distribution of large mammal species in increasingly human-dominated landscapes, and modelling needs to be done at the spatial scales at which decisions are made, as a mismatch may have important implications for conservation planning.

3.5 ACKNOWLEDGMENTS

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CHAPTER 4: THE INFLUENCE OF SOCIOECONOMIC FACTORS ON THE DENSITIES OF HIGH-VALUE CROSS-BORDER SPECIES

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ABSTRACT

Unprecedented poaching levels triggered by demand for ivory in Far East Asia are threatening the persistence of elephant *Loxodonta africana*. Southern African countries make an important contribution to elephant conservation and could soon become the last stronghold of elephant conservation in Africa. While the ecological factors affecting elephant distribution and densities have extensively been accounted for, there is a need to understand which socioeconomic factors affect elephant densities in order to prevent conflict over limited space and resources with humans. We used elephant density data from aerial surveys conducted in the Greater Mapungubwe Transfrontier Conservation Area over a 12-year period in generalized linear models to investigate the effect of eight socioeconomic variables on the densities of elephant at the site and country levels. Important factors in predicting elephant densities were gross domestic product, the proportion of total land surface under agriculture around sites where elephant were present, and the vegetation productivity. Specifically, elephant density was higher in countries where the gross domestic product was higher, in areas where the proportion of total land surface under agriculture was the lowest; and sites where vegetation productivity was the highest. Our results confirm that poverty is an important factor affecting elephant distribution at the country level, but highlight that at a local scale human disturbance and food availability plays an important role. To reduce the impact of increasing human populations and agriculture, the development of coordinated legislation and policies to improve land use planning, and the development of conservation corridors to link current protected areas between range countries, are needed.

4.1 INTRODUCTION

Growing human populations and rural poverty in Africa have led to an increasing demand for agricultural land (Krug 2001). Between 1970 and 2005, wildlife abundance in African protected areas declined by 50% (Craigie et al. 2010), and many species' ranges are now restricted to protected areas (Newmark 2008, Karanth et al. 2010a). While protected areas are fundamental for biodiversity persistence in increasingly human-dominated landscapes (Baeza and Estades 2010, Stokes et al. 2010, Montesino Pouzols et al. 2014), they are often too small to sustain viable populations of large mammals (Graham et al. 2009, Di Minin et al. 2013b, Packer et al. 2013), as they cannot meet the space requirements of wide-ranging or migratory species (Woodroffe and Ginsberg 1998, Graham et al. 2009, Stokes et al. 2010, Di Minin et al. 2013b). Increasing human populations near protected area boundaries (Harcourt et al. 2001) and elsewhere have further resulted in land use conversions that prevents free movement of wildlife (Newmark 2008, Wittemyer et al. 2008), embedding protected areas within a mosaic of different land uses such as agriculture, cattle grazing and mining (DeFries et al. 2007, Chazdon et al. 2008, Di Minin et al. 2013c). In many instances, this has led to increased human-wildlife conflict (Ogutu et al. 2011, Packer et al. 2013). Not only do those that live with dangerous species incur costs through human-wildlife conflict, but also governments incur the cost of protecting these species, for example the cost of anti-poaching measures as seen in the attempts to conserve and protect black (*Diceros bicornis*) and white rhinoceros (*Ceratotherium simum*) in South Africa at the moment (Di Minin et al. 2015).

Protected areas fall under a range of management strategies (Loveridge et al. 2007), and the resources allocated to, or generated within, these areas will directly relate to their ultimate success (Leader-Williams and Albon 1988, Di Minin and Toivonen 2015). There is, however, a marked underinvestment in state-protected areas, especially in developing countries. According to Balmford *et al*. (2002) the world spent approximately US$ 6.5 billion each year on the existing reserve network, yet half of this was spent in the United States alone. Effective elephant *Loxodonta africana* conservation has been estimated to cost US$365-930/km^2/year (Leader-Williams and Albon 1988), while in unfenced reserves, such as in Kenya, the cost of protecting lion *Panthera leo* requires budgets in excess of US$2000/km^2 per annum (Packer et al. 2013). Within national conservation departments across Africa there is a shortage of manpower and ultimately resources (Leader-Williams and Albon 1988, Selier and Di Minin 2015), that may lead to the mismanagement of protected areas and a failure to protect species within these areas (Krug 2001).

Illegal hunting of iconic species has drastically increased over the past years in range countries with high poverty levels and bad governance (Burn et al. 2011, Gandiwa et al. 2013, Bennett 2015). Of the 12 countries in Africa estimated to have elephant populations larger than 15 000 individuals, eight are among the bottom 40% of the world's most corrupt countries, and three are among the bottom 11% (Bennett 2015). On the other hand, elephant range states in southern Africa have contributed positively to the conservation of elephant and holds more than 55% of the total elephant population on the continent (Blanc et al. 2007, CITES et al. 2013). Outside of protected areas, pressure on wild animals is often higher, as elevated human densities around conservation areas can explain local species extinction (Brashares et al. 2001). Effective protection is only achievable with the support of society at large, as success in protecting wild animals may depend not only on protection status or law enforcement efforts, but also on the desire of people to respect the law, to put the law into effect, and to tolerate or even admire wildlife (Stern et al. 2001). Thus, merely setting aside protected areas for the protection of species is not enough. In areas where elephant are present, human variables might better explain the present-day densities of elephant in Africa than ecological variables (de Boer et al. 2013). Human factors are thus becoming dominant in determining the quality of the Earth's ecosystems (Vitousek et al. 1997), and therefore need to be included in policy-relevant analyses.

Southern Africa represents the stronghold of elephant conservation (Blanc et al. 2007). While the ecological factors affecting elephant distribution and densities have extensively been accounted for, there is a need to understand which socioeconomic factors affect elephant densities in those countries that have a positive contribution to elephant conservation. In this paper, we used the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA) elephant population as a case study to assess the effect of socioeconomic factors on the densities of elephant within a cross-border landscape. We use the GMTFCA elephant population because the population, like many others (van Aarde and Jackson 2007, Chase 2009) are transboundary, meaning their range extends across international boundaries and range beyond designated protected areas, which makes this population ideal to test whether different socioeconomic factors, such as different levels of governance in different countries, are important in affecting elephant densities. The general goal of this paper was to understand which ecological and socioeconomic factors affect elephant densities in a transfrontier conservation landscape. The objectives were (i) to describe trends in the densities of elephant over time; and (ii) to identify socioeconomic factors affecting the densities of elephant.

4.2 METHODS

4.2.1 Study area

This study was undertaken within the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA), in Botswana, South Africa and Zimbabwe (Figure 4.1). The GMTFCA covers 3,650 km^2 centred on the confluence of the Shashe and Limpopo Rivers. The region is semi-arid with low, unpredictable, rainfall (Harrison 1984) that averaged 365 mm annually between 1966-2001 (Selier 2007). Summer maximum temperatures can exceed 42 ºC, while winter minimum temperatures can be as low as -5 ºC (Mckenzie 1990). The elephant population in the GMTFCA consists of approximately 1224 ± 72.4 individuals (Selier et al. 2014). Electric fences restrict the movement of elephant and other wildlife in certain sections. These fences extend along the western boundary of the Northern Tuli Game Reserve (NTGR), the northern boundary of the Tuli Block and along the Limpopo River on the South African side, with a gap in the fence known as the Vhembe gap around the confluence of the Limpopo and Shashe rivers (Figure 4.1) (Selier et al. 2014).

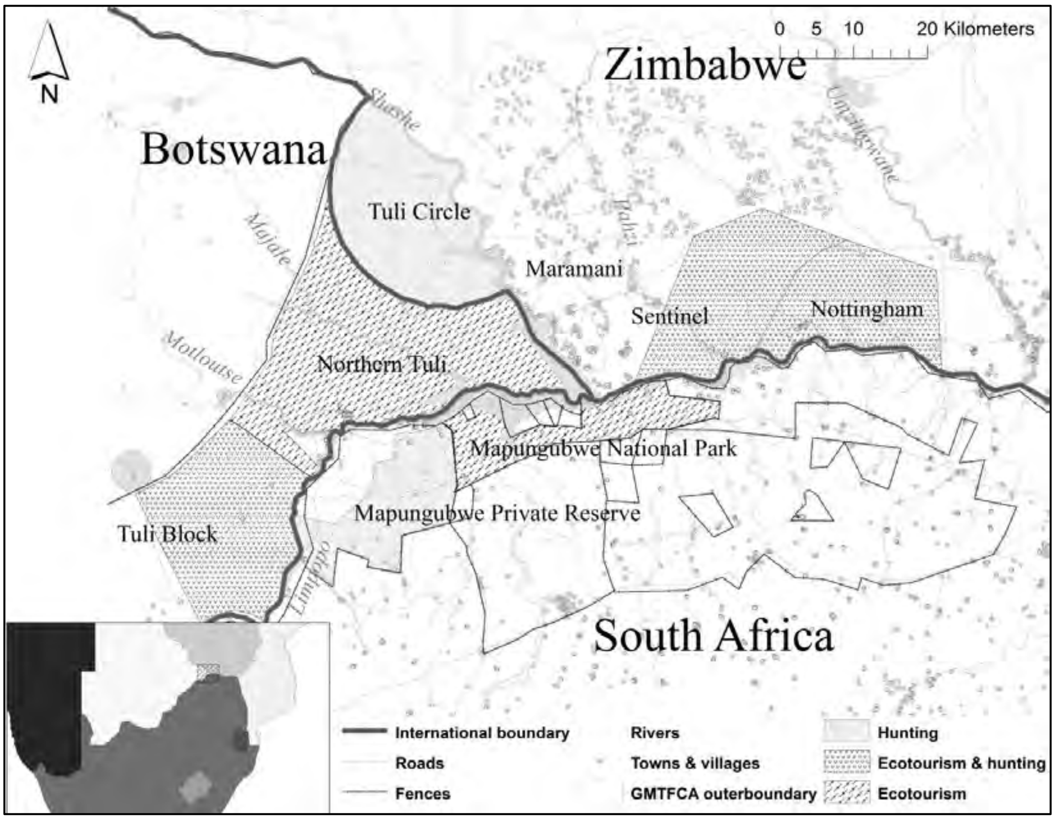

Figure 4.1: The Greater Mapungubwe Transfrontier Conservation Area and surrounding areas illustrating the borders between the three countries and the different sites within the countries used in the analysis.

The study area is characterised by a human-dominated landscape with a range of land use and management practices (Figure 4.1) (Selier et al. 2015). Land use, and ownership within and surrounding the GMTFCA, are diverse, and include contractual partners, private and communal landowners, land claimants, private tourism operations, game farms and subsistence and commercial farmers (GMTFCA TTC 2011). The following sites were included in the study: the Northern Tuli Game Reserve, and Tuli Block in Botswana, Tuli Safari Area, Maramani and Nottingham Estate and Sentinel Ranch complex in Zimbabwe and Mapungubwe National Park and Mapungubwe Private Nature Reserve in South Africa (Figure 4.1). Ownership and land use practices for each site are summarized in

Table 4.1. Several commercial operations operate within the current boundaries of the GMTFCA, all of which use, either for photographic tourism or trophy hunting, this single cross-border elephant population that can move freely between the three countries. Photographic tourism is the main economic driver within the area at present (Evans 2010), but several operations rely on a combination of trophy hunting and photographic tourism (Table 4.1).

Table 4.1: Different sites within the three countries encompassing the Greater Mapungubwe Transfrontier Conservation Area assessed indicating ownership status and main activities offered within each site.

Country	Site	Ownership	Land use practices
Botswana	Northern Tuli Game Reserve	Privately owned	Photographic tourism
	Tuli Block to Bains Drift	Privately owned	Photographic tourism and trophy hunting
South Africa	Mapungubwe National Park	State-owned	Photographic tourism
	Mapungubwe Private Nature Reserve	Privately owned	Trophy hunting
Zimbabwe	Tuli Safari Area	State-owned	Trophy hunting
	Nottingham Estate - Sentinel Ranch Complex	Privately owned	Photographic tourism and trophy hunting
	Maramani	Communal	Trophy hunting

4.2.2 Statistical analysis

Data on the distribution and abundance of elephant within the GMTFCA were obtained from total aerial surveys conducted within the study area at the end of the dry season (July – September) over the period 2000-2012 (Selier 2012, Selier et al. 2015). Three fixed-wing aircraft were used to count the study area simultaneously and the same method was used during all counts. The GMTFCA was divided into different sites based on ownership and land use practices (

Table 4.1). Elephant densities per site per year (2000, 2001, 2004, 2007, 2008, 2010 and 2012) were used as the response variable.

We were guided in the choice of candidate covariates by the aims of the analysis, in particular to enable characterisation of sites and countries. After a correlation analysis using the cor() function in R 2.15.2 (R Development Core Team 2012), with a cut-off of r = 0.80, we retained the eight variables with the greatest explanatory effect on elephant density that were not strongly correlated (Franklin and Miller 2009). The final variables used are summarised in Table 4.2.

Table 4.2: Socioeconomic variables included in the generalised linear models with country included as a fixed effect to determine the variables that best explain elephant densities in the Greater Mapungubwe Transfrontier Conservation Area.

Variable	Data description	Source
Enhanced Vegetation Index (EVI)	Resource availability at end of the dry season, Raster, Continuous data	CSIR-Meraka Institute 2011, 8-day composites; http://wamis.meraka.org.za/products/long-term-time-series
Water	Water availability, Raster, Average Euclidean distance from water	www.peaceparks.org
Agri	Proportion of total land surface under agriculture	http://data.worldbank.org/indicator
CPI	Corruption perception index (CPI score)	http://www.transparency.org/country
GDP	Per capita gross domestic product (current US$)	http://data.worldbank.org/indicator
Human densities	People per km^2 within each country	http://data.worldbank.org/indicator
Livestock	Cattle, goat and sheep densities per km^2 rasters summed to create a single raster (cell size 0.0083) for livestock densities	http://www.fao.org/geonetwork/srv/en/main.home
Owner	1. Privately owned	www.peaceparks.org
	2. State-owned	www.peaceparks.org
	3. Communal land	www.peaceparks.org

Site-level covariates included in the analysis were food and water availability, livestock and human density, proportion of agricultural land and ownership. Food and water availability and distribution are key ecological drivers affecting elephant distribution (Chamaille-Jammes et al. 2007b, Ngene et al. 2009, Shannon et al. 2009). To determine the water availability within the study area we calculated the shortest distance to water, combining all rivers, dams and artificial waterholes, using the Euclidean distance function in the packages *Raster* (Hijmans and van Etten 2012) and *SP* (Pebesma and Bivand 2005) in R 2.15.2 (R Development Core Team 2012) at a 536.7 m resolution. The average distance to water per site was calculated in ArcGIS (version 9.3; Environmental Systems Research Institute (ESRI), Inc., CA, USA) using cell statistics. Elephant are bulk feeders and thus occur in lower densities in areas with lower plant biomass (Olff et al. 2002). We used the Enhanced Vegetation Index (EVI) as a measure of vegetation productivity, and thus the amount of forage available to elephant (Pettorelli et al. 2005, Young et al. 2009b). The EVI data were downloaded for the period January 2000 to December 2012 (Table 4.2). The EVI time series was produced from the NASA 500 m, 8-day, BRDF-corrected, surface reflectance data (MCD43A4) (CSIR-Meraka Institute 2011). The log-transformed geometric mean of the 8-day composites for the end of each dry season of each of the count years were calculated with a grid cell size of 536.7 m x 536.7 m, and used as a measure of the vegetation productivity per site per count year. Human densities are negatively correlated with elephant densities (van Aarde and Jackson 2007, de Boer et al. 2013, Selier et al. 2015). Hoare and du Toit (1999) further suggested that elephant densities are negatively related to human densities below a specific threshold. The number of people per km^2 within each country was thus included as a variable that may influence elephant densities within the GMTFCA (Table 4.2). The number of livestock also reflects human presence, and we included livestock densities (livestock/km^2), calculated by adding up the densities per grid cell for cattle, goats and sheep (http://www.fao.org/geonetwork/srv/en/main.home) in ArcGIS (version 9.3; Environmental Systems Research Institute (ESRI), Inc., CA, USA) to create a single raster for livestock densities (Table 4.2). The average value within each site was then calculated and used as a measure of the livestock density per site. The proportion of the total land area under agriculture (http://data.worldbank.org/indicator) also reflects human presence and may be used as a proxy for land fragmentation and the proportion of people that may be impacted on by wildlife through human-wildlife conflict (Abensperg-Traun 2009). Ownership (owner) of each site within the three range countries was defined as whether the area was privately owned (1), state-owned (2) or communal land (3) (Table 4.2).

Country-level covariates included in the analysis were the per capita gross domestic product (GDP/cap) and the corruption perception index (CPI). The GDP/cap values per country per year were obtained from the World Bank's World Governance Indicators (WGI) project (http://info.worldbank.org/governance/wgi/) (Table 4.2). The differences in the GDP/cap between countries are expected to influence the efficacy of conservation policies (Wittemyer et al. 2008, Burn et al. 2011), and may also positively influence people's attitudes towards conservation (Burn et al. 2011). It was expected that higher elephant densities would occur in areas with higher levels of education, income and life expectancy, and where more resources are invested in conservation (Leader-Williams and Albon 1988). Country or regional policies, level of corruption and the capacity of a country to successfully implement policies further influence the level of protection provided. We therefore included the Corruption Perceptions Index (CPI) from Transparency International (http://www.transparency.org/) as an index of the level of corruption for each country as predictor variables (Table 4.2). We included CPI because it was extensively used in previous studies (Smith et al. 2003, Burn et al. 2011, de Boer et al. 2013). Values for humandens, CPI, GDP, water and EVI were not uniformly distributed and were log-transformed (Franklin and Miller 2009).

We used an information theoretic approach (Burnham and Anderson 2002) and Akaike's information criterion corrected for finite sample size (AICc) to calculate statistical models in R

v. 2.15.2 (R Development Core Team 2012), using package glmulti (Calcagno 2010). Generalised linear models with a negative-binomial error distribution and a log-link function were used to examine the socioeconomic drivers of elephant densities within the GMTFCA. We included countries and sites as random effects in the analysis, following Burn *et al*. (2011). All covariates were fitted as fixed effects – i.e. with constant regression coefficients across sites and countries. The countries under consideration were Botswana, South Africa and Zimbabwe. We determined the magnitude and direction of the coefficients for each independent variable using multi-model averaging across 100 models. The relative importance of each predictor variable was measured as the sum of the Akaike weights over the six top-ranked models containing the parameter of interest (Conroy and Brook 2003). We also assessed each model's structural goodness of fit using the percentage of deviance explained by the model. Finally, we validated the top-ranked models by using leave-one-out cross validation, which is used to estimate the mean model-predictor error by successively omitting one observation from the training data set and using it for validation.

4.3 RESULTS

According to the generalised linear models (Table 4.3), the best predictors for elephant densities were GDP/cap, proportion of agricultural land, and the amount of forage available (EVI) (Figure 4.2). The selected model had an AICc weight of 0.1 (Table 4.3) and had a mean prediction error of approximately 10% using leave-one-out cross validation. The explanatory power of our model (the percentage of deviance explained) was 46% (Table 4.3).

Table 4.3: Top-ranked predictors of elephant densities within the Greater Mapungubwe Transfrontier Conservation Area

Model	Variables	# of variables	AICc	weights	% of deviance
1	EleDens ~ Agri + GDP + EVI	3	18.011	0.112	0.455
2	EleDens ~ Agri + GDP	2	18.948	0.099	0.416
3	EleDens ~ Agri + GDP + EVI + Water	4	19.497	0.076	0.512
4	EleDens ~ Agri + GDP + Livestock + EVI	4	20.130	0.055	0.504
5	EleDens ~ Agri + GDP + Water	3	20.845	0.039	0.431
6	EleDens ~ Agri + GDP + Livestock + EVI	4	21.060	0.035	0.433
7	EleDens ~ GDP	1	21.263	0.031	0.280
8	EleDens ~ Agri + CPI + GDP	3	21.305	0.031	0.422
9	EleDens ~ HumDens + Agri + GDP + EVI	4	21.532	0.027	0.460
10	EleDens ~ HumDens + GDP	2	21.580	0.027	0.389

The magnitude and direction of the coefficients for each independent variable averaged across 100 models are presented in Table 4.4. The coefficient for proportion of land under agriculture had – as expected a negative sign, indicating that an increase in agricultural land was expected to result in a decrease in elephant densities (Table 4.4). As expected, the coefficient for EVI was positive, indicating that elephant occur in areas with higher forage availability. Similar, the coefficient for GDP/cap had a positive sign, indicating that countries with a higher GDP/cap will have higher elephant densities. The relative importance of each predictor variable is presented in figure 4.2. Human densities, livestock densities, the Corruption Perception Index, distance from water and ownership had a smaller effect on elephant densities (Figure 4.2). The coefficients for human and livestock densities, as well as distance from water, were negative, meaning that elephant densities were higher at sites with lower human and livestock densities and that elephant densities were higher in proximity to water (Table 4.4).The coefficient for public protected areas and communal lands were negative, indicating that there were fewer elephant on state-owned land and communal land than on privately owned land within the GMTFCA (Table 4.4). The coefficient for CPI was positive so that elephant densities were higher in countries where governance was better (Table 4.4).

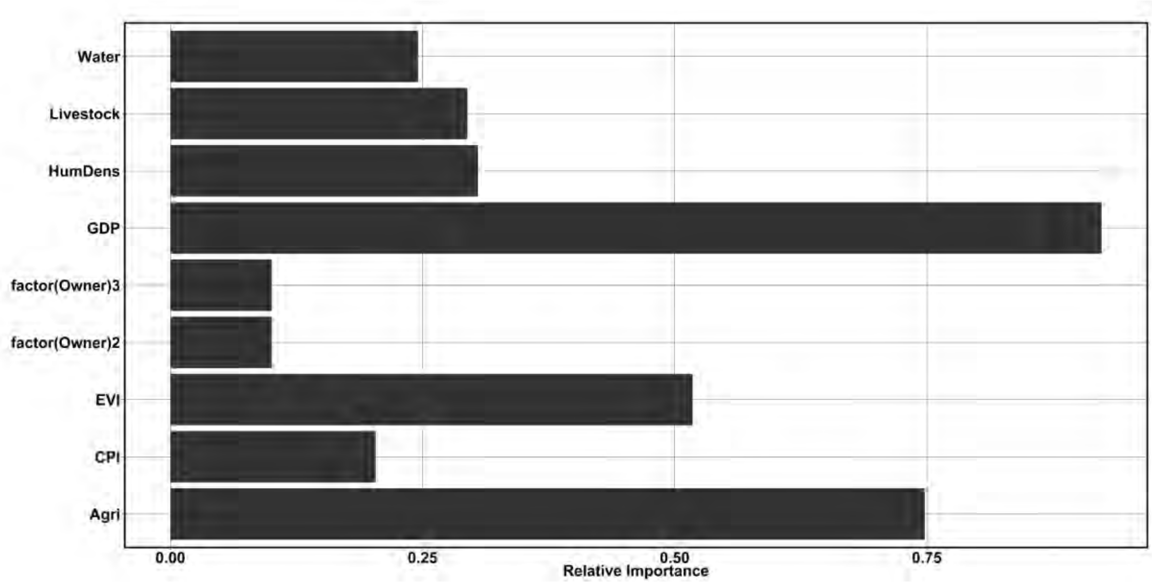

Figure 4.2: Relative importance of the most important variables affecting the densities of elephant within the Greater Mapungubwe Transfrontier Conservation Area at the site level.

Table 4.4: The magnitude and direction of the coefficients for the predictor variables used in the models.

Variable	Estimate	Uncond. variance	Nb models
factor(Owner)2	-0.0119	0.001	32
factor(Owner)3	-0.0213	0.002	32
CPI	0.077	0.0338	41
Water	-0.0746	0.0263	28
Livestock	-0.007	0.0002	45
HumDens	-0.056	0.015	51
EVI	0.5565	0.4502	42
Agri	-0.9117	0.4671	47
GDP	0.4621	0.0402	75

4.4 DISCUSSION

In this study, we used generalised linear models to investigate the effect of socioeconomic variables on the densities of elephant at the site and country levels within the GMTFCA. We found that GDP/cap, EVI and the proportion of land under agriculture were important in predicting elephant densities. Particularly, elephant densities were higher in countries with higher GDP/cap and at sites that had higher food availability and lower proportion of land under agriculture around them. Data on GDP/cap was only available at a country level. Therefore, future studies should include more detailed information at a smaller administrative scale to account for more localised poverty levels. The distribution of resources such as food and water are key ecological drivers determining the distribution and densities of elephants in the landscape (Young et al. 2009a, Selier et al. 2015). Our results confirm these findings, but also suggest that future agriculture expansion will likely negatively influence the densities of elephant.

Factors such as GDP/cap, poverty and the encroachment of humans on protected areas, are more important factors in affecting elephant conservation (Blanc et al. 2007, de Boer et al. 2013). The GDP/cap is a reliable indicator of a country's investment in protected area management (Wittemyer et al. 2008, Burn et al. 2011) and is positively correlated with human welfare which in turn influences attitudes towards conservation (Burn et al. 2011). The average GDP/cap for Botswana and South Africa over the duration of the study was similar (US$5271.29 vs US$ 5167.07). However the average GDP/cap for Zimbabwe for the same period was much lower compared to the other two countries (US$ 526.53). Countries, such as Zimbabwe, with lower GDP/cap seldom have the capacity or funds to maintain state-owned parks (James et al. 1999, Krug 2001, Craigie et al. 2010) let alone enlarge their system of protected areas (Leader-Williams and Hutton 2005, Gallo et al. 2009). An increased investment in the management of protected areas and law enforcement were shown to be important predictors of Great ape survival in Africa (Tranquilli et al. 2012). A lack of resources will not only lead to poorly managed parks with insufficient protection systems in place, but could also lead to increased corruption especially where wildlife officials are poorly paid and conservation agencies deal with high value species (Smith et al. 2003). Further, multiple consecutive declines in the GDP/cap increase the number of poor (Jalan and Ravallion 2000, Suryahadi and Sumarto 2010) depending on natural resources, increasing bushmeat consumption and the illegal killing of wildlife that ultimately affects species persistence (Gandiwa et al. 2013, Brashares et al. 2014).

Our study has further shown that an increase in the proportion of land under agriculture will negatively influence elephant densities. Land use changes, such as an increase in agricultural land, reduce the size of natural ecosystems, and increase fragmentation of the landscape restricting the movement of wide-ranging species (Woodroffe and Ginsberg 1998, Di Minin et al. 2013b, Selier et al. 2015). In addition, agricultural expansion can isolate protected areas from their surrounding landscapes, leading to an island effect where no or limited connectivity exists between protected areas (DeFries et al. 2007, Butchart et al. 2010, Yackulic et al. 2011). Hoare and du Toit (1999) showed that when agriculturally transformed land becomes spatially dominant over natural woodland elephants disappear from the system. In this human-dominated landscape the size and connectivity of the remaining patches of elephant habitat will determine whether or not elephants remain as residents or move away (Hoare and Du Toit 1999). Increased fragmentation of the landscape restricting the movement of wide-ranging species could further lead to increased human-wildlife conflict. Conflicting land use practices (crop farming) draw elephant and other conflict species towards community areas, primarily during periods of low natural food availability, thereby creating an ecological trap (Hoare and Du Toit 1999, Nyhus and Tilson 2004, Chiyo et al. 2005). Therefore, coordinated land use planning to maintain protected areas of sufficient size and maintain connectivity between protected areas will be required to maintain elephant in the study area and potentially limit human-elephant conflict.

Overall, the results suggested that accounting for socioeconomic factors is important in determining the abundance of a large, wide-ranging, mammal species such as elephant. Where these populations are transboundary the joint management of these species on a population level is imperative for their continued persistence in a human-dominated landscape (Linnell et al. 2008, Trouwborst 2015). This will require the development of coordinated legislation and policies to improve land use planning (Chapron et al. 2014, Montesino Pouzols et al. 2014, Trouwborst 2015), the development of multi-use zones around protected areas (Wittemyer et al.

2008), and conservation corridors to link current protected areas between range countries (van Aarde and Jackson 2007). Montesino Pousols *et al*. (2014) demonstrated that if coordinated international action is not taken quickly further biodiversity loss is unavoidable. Thus, to maximize conservation benefits and to alleviate the impacts of future human growth and land use changes on wildlife action should be taken quickly. Immediate action is needed in countries that are already making a positive contribution in Africa to the conservation and protection of elephant to prevent future human population increases and activities from negatively impacting on the persistence of elephant in these countries. Effective protection of source populations in a well-connected system of protected areas that buffers them from anthropogenic threats remains the key action to ensure the future persistence of wide-ranging species, such as elephant, in the developing world (Di Minin et al. 2013b, Di Minin and Toivonen 2015).

4.5 ACKNOWLEDGMENTS

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CHAPTER 5: THE LEGAL CHALLENGES OF TRANSBOUNDARY WILDLIFE MANAGEMENT AT THE POPULATION LEVEL: THE CASE OF A TRILATERAL ELEPHANT POPULATION IN SOUTHERN AFRICA

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ABSTRACT

More than 80% of the African elephant (*Loxodonta africana*) range in Africa still exists outside of formal protected areas, and sits across many administrative and political boundaries. These ranges comprise a matrix of multi-use landscapes of potentially divergent administrative, legal and political systems. It is further recognised that the evolution of the various human focused administrative systems, from an elephant conservation perspective, has been ad hoc and without integration. This has resulted in or facilitated a progressive encroachment on the natural range by human settlement and agricultural activities. The movement of elephant across international boundaries results in a parochial approach to their management, particularly within a context of increasing human-wildlife conflict, and an inconsistent consideration of ecological requirements of the elephant at a transboundary level, leading to a mismanagement of these populations, at a transboundary level. This study investigates the conservation and management of the Central Limpopo River Valley (CLRV) elephant population roaming between Botswana, South Africa and Zimbabwe, and utilising the trilateral Greater Mapungubwe Transfrontier Conservation Area (GMTFCA). The desirability of conserving and managing wildlife at the level of the biological unit of the transboundary population is taken as a starting-point, following insights from transboundary large carnivore and waterbird species in Europe. The many legal and policy frameworks applicable to the CLRV elephant population are identified and tested against this approach. The current approach taken in respect of the CRLV elephant population meets the benchmark to a substantial degree, although some essential steps remain to be taken. We discuss potential adjustments to laws and policies that are required to ensure the ecological stability of transboundary elephant populations, and to provide for their collective management.

5.1 INTRODUCTION

What do geese (*Anser* spp., *Branta* spp.) and wolves (*Canis lupus*) in Europe have in common with elephants (*Loxodonta africana*) in southern Africa? In fact, quite a lot. All three enjoy protected status under multiple international legal instruments (Hopkinson et al. 2008, Fleurke and Trouwborst 2014, Trouwborst 2015). At the same time, all three have a high potential for so-called 'human-wildlife conflict' (Peterson et al. 2010, Redpath et al. 2013), and are subject to smaller or larger degrees of lethal control (Johnson and Madsen 2013, Trouwborst 2014). These traits, in turn, are linked to the fact that the life histories of geese, wolves and elephants require populations of these animals to range beyond designated protected areas into the wider landscape (Trouwborst 2012, Di Minin et al. 2013b). Last but not least, many populations of geese, wolves and elephants – and many other species besides – are transboundary, overlapping the territories of several countries (Chase 2009, Trouwborst 2012, Selier et al. 2015, Trouwborst 2015). These traits, however, can lead to a potential mismanagement of transboundary populations, because of a mismatch between the scales at which these animal populations operate and the scale at which administrations operate (Linnell et al. 2008, Linnell and Boitani 2012, Delsink et al. 2013, Pitman et al. 2015).

Whereas this article addresses all of the aforementioned shared characteristics, the main focus is on the latter, that is, the transboundary nature of many wildlife populations. In particular, it explores the notion of adjusting relevant law and policy to the spatial scale of each animal population, including where this population is transboundary. This notion, which makes evident biological sense, is at the forefront of current thinking regarding the conservation and management (including sustainable use) of cross-border species (Trouwborst 2015). Despite its simplicity at a conceptual level, however, the actual implementation of conservation and management at the transboundary population level is a complex and challenging affair (Trouwborst 2015). This paper explores the theory and practice of transboundary population level management primarily from the perspective of one particular wildlife population, namely the population of African elephant inhabiting the Central Limpopo River Valley (CLRV) in Botswana, South Africa and Zimbabwe. By focusing on the emblematic African elephant, this article builds on a rich tradition of international law scholarship (Glennon, Couzens 2013, Adam 2014, Nollkaemper 2014), adding the perspective of transboundary population level conservation.

The methodology employed has multidisciplinary features. Whereas it chiefly concerns the identification, interpretation and comparison of legally relevant documents, it also draws on data from the biological and other pertinent disciplines. Concretely, the approach taken is as follows. First, the essential elements of organising wildlife law and policy at the transboundary population level are explored (in Section 2), drawing on European experiences regarding the management of populations of gray wolf and other large carnivore species and of pink-footed goose (*Anser brachyrhynchus*). This is followed (in Section 3) by an introduction of the general situation regarding elephants in southern Africa and the CLRV elephant population in particular. Subsequent sections then analyse to what degree the transboundary population level approach (as described in Section 2) is incorporated into the applicable law and policy at the global and regional level (Section 4), the trilateral level (Section 5), and the national level, in the three countries concerned (Section 6). Conclusions and recommendations are presented in a final section (Section 7).

5.2 THE TRANSBOUNDARY POPULATION APPROACH

From a conservation perspective, it is preferable to adjust relevant law and policy to the spatial scale of a wildlife population – even where this population straddles the territories of various countries – rather than adjusting it to biologically meaningless political and administrative boundaries.

5.2.1 Wolf, bear, wolverine and lynx populations in Europe

An instructive example where this approach has been developed in a comparatively consistent and comprehensive way concerns the four largest terrestrial carnivore species occurring in Europe, i.e., gray wolf, brown bear (*Ursus arctos*), wolverine (*Gulo gulo*) and Eurasian lynx (*Lynx lynx*). Given that Europe, like Africa, is composed of multiple countries, the fact that conservation areas often occur on international borders (Montesino Pouzols et al. 2014, Di Minin et al. 2015), and given the low densities at which the large carnivore species occur, the need for transboundary coordination is especially strong in this context to effectively manage these wide-ranging species at a population level (Kaczensky et al. 2013, Chapron et al. 2014, Trouwborst 2014). Some basic elements of the envisioned cross-border approach are described in the following statement in a paper regarding wolves:

"*The first step that is required is to move away from viewing wolf distribution within the arbitrary lines on maps that national or provincial borders represent and to look at the actual distribution. The resulting view is one of a 'meta-population like' structure where demographic viability is achievable in many regional units that have a more or less continuous distribution of wolves (populations). It is crucial that these populations are managed as biological units – with the administrative bodies (be they intra- or inter-national) that share a population coordinating their activities to ensure that their independent actions enhance rather than hinder each other*" (Linnell and Boitani 2012).

The approximately 12,000 wolves living in Europe are spread across ten distinct populations, eight of which are transboundary (Kaczensky et al. 2013, Chapron et al. 2014). Roughly comparable situations exist for bears (ten populations, eight of which transboundary), lynx (ten populations, eleven of which transboundary) and wolverines (two populations, both of which transboundary) (Kaczensky et al. 2013, Chapron et al. 2014).

The four species are covered by two important European legal instruments for wildlife conservation. The first is the 1979 Convention on European Wildlife and Natural Habitats (Bern Convention) (Convention on the Conservation of European Wildlife and Natural Habitats, Sept. 19, 1979, E.T.S. 104), to which virtually all European countries are contracting parties. The second is the 1992 European Union (EU) Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (Habitats Directive) (Council Directive 92/43/EEC of 21 May 1992 on the Conservation of Natural Habitats and of Wild Fauna and Flora, O.J. L 206, 7 (1992)), which binds the 28 EU member states. Both instruments set out obligations concerning the generic protection of the four large carnivore species involved, and the protection of their habitat (Fleurke and Trouwborst 2014). However, these obligations target the countries concerned individually. No provision is made for concerted conservation actions tailored to transboundary wildlife populations, notwithstanding a generally phrased obligation in the Bern Convention for contracting parties to 'cooperate whenever appropriate and in particular where this would enhance the effectiveness of measures taken under other articles of this Convention' (Article 11(1)(a)).' Moreover, the specific legal regimes applicable to the various species under these instruments vary from country to country, due to reservations submitted by several parties to the Bern Convention and country-specific differences established under the Habitats Directive (Trouwborst 2010, 2012). For instance, under the Bern Convention, depending on the party concerned, the wolf is a 'strictly protected fauna species' under Appendix II, a 'protected fauna species' under Appendix III, or neither (Fleurke and Trouwborst 2014). Comparable differences apply under the Habitats Directive, and to some of the other species involved. The situation is compounded further by the fact that not all Bern Convention parties are also EU member states. The resultant fragmentation of the European legal landscape in respect of the four large carnivores adds to the urgency of transboundary cooperation at the population level (Linnell et al. 2008, Linnell and Boitani 2012, Epstein 2013, Trouwborst 2014, 2015).

To remedy these shortcomings, both the Standing Committee of the Bern Convention (the principal body established under the Convention) (Standing Committee Recommendation No. 115 (2005) on the Conservation and Management of Transboundary Populations of Large Carnivores), and the European Commission (charged with supervising the implementation of the Habitats Directive) (Standing Committee Recommendation No. 137 (2008) on Population Level Management of Large Carnivore Populations) have expressly advocated a transboundary population level approach to large carnivore conservation and management. Of particular interest is the development of a detailed guidance document on the issue by the Large Carnivore Initiative for Europe (LCIE) (http://www.lcie.org), under contract from the European Commission. These 'Guidelines for Population Level Management Plans for Large Carnivores in Europe' (Carnivore Guidelines) were finalized, and endorsed by the Commission, in 2008 (Linnell et al. 2008). The Guidelines call for the adoption of a population level management plan, by the competent authorities of all countries involved, for each large carnivore population, and set out detailed instructions in this regard (Linnell et al. 2008). Upon the Carnivore Guidelines' adoption, the European Commission submitted that 'it is difficult, if not impossible, for one Member State to manage and protect its large carnivores in the absence of concerted and convergent actions being taken by its neighbours' (European Commission 2008). In particular, it held that 'effective management of large carnivore populations which are shared between Member States can only be achieved through shared and coordinated management plans as described in the[se] guidelines.' The Commission considers these Carnivore Guidelines to represent 'best practice' when it comes to the application of the Habitats Directive to large carnivores (European Commission 2008). The Standing Committee of the Bern Convention has similarly called on parties to the Convention 'to reinforce cooperation with neighbouring states in view of adopting harmonized policies towards management of shared populations of large carnivores, taking into account the best practice in the field of management of populations of large carnivores' (Standing Committee Recommendation No. 137 (2008)). The Carnivore Guidelines are expressly referred to in the Recommendation in question (Standing Committee Recommendation No. 137 (2008)).

Especially significant for present purposes is a template provided in the Carnivore Guidelines, setting out the ingredients that each transboundary management plan should contain (Linnell et al. 2008). Even if the template is focussed on European large carnivores, it does appear to represent a relatively comprehensive catalogue of elements to be included in transboundary population level conservation generally. The template is reproduced in its entirety in the first two columns of Table 5.1. Most of the elements mentioned in the template are clearly conducive, and some of them imperative, to the achievement of meaningful transboundary population level cooperation. To avoid undue repetition, however, the analysis here is limited to highlighting a few of the most essential ones, concerning objectives and specific actions. As regards the former, according to the Carnivore Guidelines' template the objectives for the population concerned should be 'specific and measurable', encompassing concrete goals in terms of numbers, range, and other parameters such as harvest rates, damage levels, poaching levels, 'that can be used to measure the success of management actions.' (Linnell et al. 2008) These goals ought to be 'distributed in space' between the various administrative units involved 'such as countries, states, counties, wildlife management units or protected areas' (Linnell et al. 2008). As regards specific actions, the template stresses that it is 'crucial' that the removal of animals be 'coordinated between all management units that share a population,' based on a pre-determined 'population level limit for the number of individuals that can be removed per year' (or, arguably, any other coherent time unit employed) (Linnell et al. 2008). Significant attention should, furthermore, be paid to ensuring connectivity within the population as well as with neighbouring populations (Linnell et al. 2008). A final point singled out here is that each plan should indicate any ‘changes in legislation that are needed to bring about the population level management plan’ (Linnell et al. 2008).

Table 5.1: Template for transboundary population level management plans (1st and 2nd columns, replicated directly from J.D.C. LINNELL *ET AL.*, GUIDELINES FOR POPULATION LEVEL MANAGEMENT PLANS FOR LARGE CARNIVORES IN EUROPE (2008) at 35-37); corresponding elements and caveats regarding the current management approach to the CLRV elephant population (3rd column); and recommendations concerning the latter's improvement (4th column).

Template: Items	**Template: Explanatory Notes**	**CLRV Elephant Population: Current Status**	**CLRV Elephant Population: Recommendations**
1. Background	This section summarizes the background information about the specific population and its metapopulation context. It is intended to serve as a reference for justifying the objectives and associated actions that come later in the document, and to increase the transparency, credibility and robustness of the overall plan. Outlining the similarities and differences in circumstances between different management units is important. It will include the following sub-sections.	This section has been summarized in the draft GMTFCA Elephant Management Plan and in papers published on the elephant population.	
1.1. Population definition	Describes the geographic limits of the population, where possible separating between (1) the distribution of the reproductive portion of the population, (2) the total area of regular occurrence of resident individuals and (3) the areas where individuals, such as dispersers, occasionally occur. If the distribution of animals within a population is clumped, then these population segments need to be described.		
1.2. Management units	Describes the existing management units – such as national, state or county		

	borders, wildlife management unit borders, or PA borders that overlay this distribution.		
1.3. Population description	Describes the history, status, trend, and ecology of the population. If any data are available on demographic parameters (reproduction or mortality) they should be gathered and presented. Likewise, as detailed as possible time series data on population trends and eventual human harvest should be gathered on as dine a spatial scale as possible. Special emphasis should be placed on describing the survey/monitoring/census methods that have been used such that the quality of the data can be evaluated.	Through the CLRV elephant research project, data are available on the distribution, numbers and trends of this population. Some demographic data is available. Biennial counts from 2000-2014 have been done by the CLRV elephant research project. Data on offtakes (DCAs and trophy hunting) are collected on national levels, and are not shared amongst the three range countries. Information on elephant history, distribution, numbers and demographics are summarized in the draft GMTFCA Elephant Management Plan.	
1.4 Habitat description	Describes the quality of the habitat within the geographic limits of the populations and in surrounding areas where expansion is possible. Presents data on anthropogenic (human population, infrastructure, agriculture, land use) and biological (forest cover, prey distribution) parameters.	Within the GMTFCA some work has been done on evaluating habitat quality. The Mapungubwe Elephant Management Plan includes actions to manage elephant impact. The CLRV elephant research project has collected data on anthropogenic disturbances and impacts on elephant. Peace Park Foundation has collected data on human densities, infrastructure etc. and broad vegetation classification. Information on biophysical features of the TFCA is included in the draft GMTFCA Elephant Management Plan.	
1.5. Continental context	Describes the existing and potential connections to neighbouring populations	The development of the GMTFCA has improved connectivity for the population	

	within the metapopulation. Evaluates the importance of this population inside the European context – both in terms of numbers and connectivity.	between the three range states. However the GMTFCA only covers part of the population's range. There is the potential to connect this population with neighbouring transfrontier populations in all three range states.	
1.6. Current management			
1.6.1. Legal status and management regime	Describes the current management practices within each of the management units.	The legal frameworks for the management of elephant in Botswana, South Africa and Zimbabwe have much in common, but some marked differences exist. All three countries within their legal framework have made provision for ownership of elephant and the non-consumptive and consumptive utilization of elephant. However, the conditions under which use is allowed are different. Some information is provided within the draft GMTFCA Elephant Management Plan.	
1.6.2. Damage and conflicts	Summarises data on the different conflicts that occur and on ways in which these have been mitigated.	Human-elephant conflict is a concern in all three range states. Data on DCA offtakes are collected differently in all three countries. Limited data available and currently not analysed to inform offtake quotas. Botswana is the only country that maintains a Problem Animal Control Unit. A further concern is the potential impact elephants might have on biodiversity. Provisions have been made in national law in all three states but manners to deal with this issue differ between range states and between administrative units within	DCA offtake quotas should be calculated on a population level taking the natural mortality rate and all other offtakes such as trophy hunting into consideration. There should be a mechanism through which the different management units could report back on conflicts and offtakes. A monitoring program that collects data consistently across administrative units should be developed to measure potential impacts of elephant on biodiversity.

		countries. No monitoring program has been put in place to measure potential impacts.	
1.6.3. Obstacles to conservation	Identifies the major threats, limiting factors and obstacles to successful conservation in the region. A SWOT or DSPIR method could be used to structure this debate.	No SWOT or DSPIR analysis has been done. Information available through the CLRV elephant research project.	This could be done through a research project,;CLRV elephant project has already identified some threats facing the population
1.6.4. Conservation status	Summarise the conservation status of the population and any conservation measures that have been taken recently to improve this status.	Regional Red List status is Least Concern, and elephant is considered Least Concern in all three countries. In Zimbabwe, elephants are not protected and control over wildlife has fully been transferred to landowners, which in the case of communities are transferred to the RDCs. These Councils were further empowered to adopt bylaws addressing natural resource management. In Botswana, elephants are partially protected and can only be utilized under permit. Ownership and control of elephant is claimed by the state. In South Africa, elephant is a protected species and in the case of the elephant crossing into South Africa from neighbouring countries considered *res nullius*. However ownership can be established through control and constraint e.g. suitable fences. The status of elephant within the GMTFCA will likely be that of *res nullius* with the state as overall custodian.	A summary of the conservation status and conservation measures taken is included in the draft GMTFCA Elephant Management Plan.
2. Definition of goals and	This section develops both the overall vision and temporally- and spatially-		

objectives	specific, measurable, objectives and targets that the plan seeks to reach. It contains the following sub-sections.		
2.1. Statement of overall vision	Develops a common overall vision for large carnivore conservation in the region. It could also include statements about large carnivore conservation and should relate to other conservation and social economic objectives for the same region.	A common vision has been developed within the draft GMTFCA Elephant Management Plan. The national visions for elephant conservation align between the three range states and focus on conserving elephant populations while ensuring the maintenance of habitats and biodiversity and promoting the contribution of elephants to development.	
2.2. Measurable objectives	This is the section where specific and measurable objectives are developed within the frames of the overall vision. These objectives should be impact-orientated (represent desired end points), measurable, time-limited, specific and credible. These objectives should be based on the best available science, be tailored to the specific species and region, include both short-term and long-term objectives, and make uncertainties transparent (Tear *et al*. 2005).	Specific and measurable objectives have been developed for the GMTFCA through the draft GMTFCA Elephant Management Plan. The Plan, however, has not yet been approved by range states or implemented. The Plan, furthermore, only addresses a section of the CLRV elephant population's range and does not address issues outside of the boundaries of the GMTFCA. Objectives on a national level pertain to the national elephant population, not specific subpopulations. There is, however, a mismatch between the individual countries' elephant management objectives. A Southern Africa Regional Elephant Conservation and Management Strategy has been developed but only partially implemented by range states. Again, this strategy speaks to the regional elephant population and not to specific populations.	Establish measurable objectives for CLRV elephant population, and incorporate these in GMTFCA and national contexts.

2.2.1. Favourable reference population	Develops a common understanding of what the threshold favourable reference population value will be for this population.	Currently the population is estimated at 1400 elephants; no favourable reference population value has been determined for this population.	An agreement between all stakeholders should be reached on what the threshold favourable reference population value should be for the CLRV elephant population.
2.2.2. Favourable reference range	Develops a common understanding of what the threshold favourable reference range distribution will be for this population.	The boundaries of the GMTFCA have been determined but elephant range is wider than current boundaries. No favourable reference range has been agreed upon on a national or trilateral level.	An agreement between all stakeholders should be reached on what the threshold favourable reference range distribution should be for the CLRV elephant population.
2.2.3. Population goals	Explores how far beyond the threshold levels required to satisfy community obligations it is desirable to go for this population.	This has not yet been determined. The overall goal on a national and trilateral level is to conserve and sustainably use elephant to contribute to the economy of the region, but no specific goals in how this will be achieved within the GMTFCA or broader area/population has been determined.	An agreement should be reached by all stakeholders. Require stakeholder agreement on how single quota for trophy hunting should be distributed amongst stakeholders and how revenue gained through the sustainable utilisation of elephant and cost of management will be shared across the range states and between all stakeholders.
2.2.4. Success criteria	Develops a set of measurable parameters, such as population size or trend, harvest rates, damage levels, poaching levels that can be used to measure the success of management actions.	No measurable parameters have been set to measure success of management actions.	Develop measureable parameters to measure success of management actions. If elephant population is to be utilized, a single quota should be set based on population estimates. Harvest rates should be recorded and must include all offtakes i.e. trophy hunting and DCAs and measured against population trends. The data obtained should be used to inform the annual off take quota.
2.2.5. Connectivity and	Specifically develops a plan to maintain or enhance the connectivity both within	Development of GMTFCA to maintain and enhance connectivity between three	Approval and implementation of the GMTFCA Elephant Management Plan.

expansion	this population and with neighbouring populations. Areas where expansion is to be encouraged or favoured and corridors crucial for connectivity should be identified.	range states, but only MoU between three countries. Treaty has not yet been signed. No current plan to improve connectivity between neighbouring populations.	Develop broader plan to establish connectivity between populations through the identification of corridors.
2.2.6. Spatial aspects of management	The overall objectives developed in the previous sections should be distributed in space between various management units such as countries, states, counties, wildlife management units or PAs. The relationship between this plan and any PAs, especially Natura2000 sites, should be considered in detail. Particular attention should be paid to integration of the needs for population connectivity in the national infrastructure and industrial development plans.	Within the GMTFCA various draft plans for the management of species and economic development exist, but have not yet been approved or implemented. Currently, the management plan for elephants within Mapungubwe National Park aligns well with the draft GMTFCA Elephant Management Plan. Land use plans are developed on a national level with little coordination between range states when dealing with transboundary areas or developing corridors to improve connectivity between populations.	Approval and implementation of the GMTFCA Elephant Management Plan. Coordinated land use planning and the development of a strategy to improve connectivity between neighbouring populations.
3. Actions	These are specific action points that need to be considered. They focus on the actions that mainly apply to population level management planning – other national actions may also exist but not all need to be repeated. It is not automatic that the actions should be identical in all management units – but they should be coordinated and compatible with each other. Sharp boundaries between widely different actions should be avoided.		
3.1. Maintaining range and population size	Outlines concrete actions that will act on the population to ensure that its conservation status is maintained or	Actions taken within GMTFCA include removal of internal fences to improve connectivity within the population.	Set favourable population size value and develop actions to maintain or improve conservation status accordingly.

	enhanced (as appropriate). Outlines steps that will be made to maintain or enhance internal connectivity within the population, especially if there are a number of population segments.	However, no favourable population size has been determined. In the absence of a clear objective it is difficult to develop specific actions. Management actions do not align between different management units or between range states. No coordination of elephant offtakes between range states. Individual countries are implementing unsustainable trophy hunting quotas based on restricted subsets of population data. DCA offtakes not considered in annual offtakes. No adaptive management of offtakes to ensure sustainability. Current offtakes too high and unsustainable.	
3.2. Maintaining and enhancing connectivity	Outlines any specific actions that will be taken to maintain or enhance external connectivity to neighbouring populations. Develops clear land use plans for crucial corridors. If translocation or reintroduction is to be considered, these need to be described in detail.	Land use planning done on provincial level in South Africa. Agreements in place between Limpopo and SANParks but no specific plans with regards to elephant and improving connectivity for this population. Land use planning in Botswana and Zimbabwe done on national level; no specific plans in place to improve connectivity for this elephant population. Botswana highlights importance of corridors within draft management plan but no actions identified.	Suggest development of cross-border strategy looking at best placement for corridors to link neighbouring populations.
3.3. Adapting legislation	Describes any changes in legislation that are needed to bring about the population level management plan. Sharp boundaries between management units with widely different legislations should	Currently national legislation not used effectively to further transboundary population level management.	Align national legislation to population level management needs.

	be avoided.		
3.4. Ensuring adequate wild prey base, natural food supply and habitat quality	Describes measures that will be taken to ensure that adequate prey and habitat are available for large carnivores. For bears it is important that forestry maintains food trees and that presence of hunting and forestry practices do not disturb denning bears during winter. For lynx and wolf it is crucial that wild ungulate harvest takes into account the presence of predators when setting quotas.	Some provisions on national level to prevent impact of elephant on biodiversity; some actions implemented on management unit level, i.e., exclusion of elephants from riverine section in Mapungubwe National Park.	Develop monitoring program that collects data consistently across administrative units to measure potential impacts of elephant on biodiversity and agreed upon actions to be taken.
3.5. Damage control and conflict resolution	Describes how various conflicts will be mitigated and how this mitigation will be funded. In order to foster a sense of fairness and justice it would be beneficial if the same, or at least similar, incentive measures and levels of support could be obtained in all management units sharing a population.	DCAs are managed on a national level with little communication between range countries. Offtakes not measured against population estimates; no adaptive framework. In Botswana compensation scheme to mitigate elephant human conflict. Good records of DCA incidents are kept. Botswana has DCA team. In South Africa DCAs managed on provincial level; no compensation is paid but local hunters can hunt DCA elephant. Records of DCA hunts kept at provincial level. The Elephant Norms and Standards and provincial ordinance provide legislative framework for DCAs. Zimbabwe has no clear/specific management policy for problem elephant management, but RDCs allocate resources to problem animal management. No framework has been developed as to how conflict will be resolved within the boundaries of the GMTFCA; no	Suggest the establishment of a cross-border management authority that meets regularly and manages offtakes of elephant as well as the quota setting per management unit and range state. Further suggests the development of a framework for the coordination and funding of conflict mitigation.

		coordinated strategy between range states for (funding) conflict resolution.	
3.6. Coordinating harvest / control of carnivores	It is crucial that the removal of large carnivores be coordinated between all management units that share a population. A population level limit for the number of individuals that can be removed per year should be set. Development of the logic behind the application of derogations is based on a consistent, but locally relevant, logic. Ensure that evaluation of 'no detrimental effect' when applying for derogations is conducted on the population level.	No coordination of elephant offtakes between three range countries. Individual countries implement unsustainable trophy hunting quotas based on restricted population data subsets. DCA offtakes not considered in annual offtakes. CITES trophy hunting quotas determined on national level and not based on individual population estimates.	Establish cross-border management authority that meets regularly and manages elephant offtakes and quota setting per management unit and range state. Actions should include development of a single offtake quota for the population, how quota would be allocated between management units and range states, how revenue gained from elephant will be used to further management and conservation of the specific population.
3.7. Enforcement	Reports that enforcement (anti-poaching) is seriously planned and coordinated between management units to ensure that poaching in one unit cannot be passed off as legal harvest in another.	No coordinated anti-poaching efforts between three range states. Botswana conducts anti-poaching patrols on border. South Africa and Zimbabwe no enforcement along international border. SANParks conducts anti-poaching patrols within Mapungubwe National Park.	Establish cross-border management authority to coordinate law enforcement. Align national legislation to ensure that illegal activities in one range state are not considered a legal activity within a neighbouring range state.
3.8. Cross-border exchange of experience among stakeholders and interest groups	Establishes a forum for stakeholders and interest groups from all management units to meet and discuss large carnivore management related issues together.	Trilateral Technical Committee (TTC) with supporting working groups established for GMTFCA to coordinate functions between countries until Treaty is signed and enters into force and a joint management structure is formalized. Treaty has not yet been signed. Not all stakeholders are present on the current TTC; some capacity and financial constraints prevent participation on TTC.	Signing and entry into force of Treaty and development of cross-border management authority that includes all stakeholders in the area as well as scientists to inform management.
3.9. Institutional coordination of	Establishes a contact forum for all management authorities sharing a	A high-level committee of senior officials meets, as part of SADC Ministers	Establish cross-border management authority meeting on regular basis which

management authorities	population to exchange information and meet periodically.	Responsible for Environment and Natural Resources, once every two years. However, decisions taken do not always translate into actions on the ground. Multilateral agreements have set in place a cooperative framework, but may have different officials associated with their implementation to those operating within the framework of trilateral or bilateral agreements. This complicates coordination and integration of elephant management, particularly on matters relating to sovereignty, i.e., setting offtake quotas and DCA management.	could inform the high-level committee.
3.10. Coordination of monitoring and scientific research programs	It is crucial that population monitoring be conducted in a comparable and coordinated manner. Different management units may use some different methods and focus on different parameters, but there must be a minimum overlap in data collected to permit population level evaluation of population status and trend. Describes how transboundary research cooperation will be stimulated.	Currently monitoring done on national levels; the only monitoring done trilaterally is through the CLRV elephant research program.	Establish cross-border management authority and develop and allocate adequate funding to monitor the population on a population level, also to inform sustainable offtake quotas.
3.11. Ensuring sectorial coordination within and between countries	Establishes a contact forum for coordination between sectorial interests (e.g. environment, tourism, agriculture, forestry, infrastructure) between all management authorities within the relevant region. This forum should ensure that planning of other sectorial activities does not increase conflicts in	GMTFCA TTC in place. However, TTC does not include all stakeholders and has shown limited progress. No joint management structure in place.	Establish cross-border management authority including all relevant stakeholders and meeting regularly to discuss management issues. This authority could inform the high-level committee on progress and problems encountered.

	carnivore range or fragment habitat within carnivore range or in connectivity corridors.		
3.12. Monitoring efficacy of implemented management measures	A system for assessing the effects of management measures adopted must be in place in order to allow revision of the management plan and its eventual adaptation/modification.	No system in place either on a national or trilateral level to assess the effects of management measures.	Establish cross-border management authority meeting regularly and informing high-level committee.

Whereas the Carnivore Guidelines generally refer to population level management ‘plans’, it is made clear that the transboundary cooperation concerned may take any of various shapes, as long as it adequately serves its purpose. It could involve a legally binding agreement, but this is not a strict requirement. The arrangement involved needs to be sufficiently flexible to adjust to future developments regarding the population concerned, but also sufficiently formal and high-profile to warrant its actual observation by the governmental actors involved (Trouwborst 2010). In the words of Beyerlin (2014), any governmental transboundary wildlife regime ‘must fail unless it contains tailored, detailed rules on the conditions, targets, and modalities of cooperation.’

It should be noted that, unfortunately, the speed with which this population level approach is actually being implemented by European countries in respect of large carnivores still leaves much to be desired. Notwithstanding a number of promising initiatives, the first fully-fledged transboundary population level management plan is still to be formalized (Blanco 2013). This tardiness might be partly accounted for by the tenacious nature of the challenges associated with large carnivore conservation in particular (Linnell and Boitani 2012).

Be that as it may, the approach to transboundary cooperation at the population level as outlined in the Carnivore Guidelines is of significant interest for present purposes, because of its comprehensiveness and detail, and because of the way it is embedded within applicable international legal frameworks. More than anything, it provides a benchmark as to what transboundary cooperation at the population level should ideally look like (Trouwborst 2015). This benchmark will be employed in the in-depth review below of the transboundary cooperation concerning the Central Limpopo River Valley elephant population.

5.2.2 A Goose Population in Northwestern Europe

The next example to consider is the population of pink-footed goose that breeds on Svalbard (Spitsbergen) in the Arctic region, and seasonally migrates through Norway to wintering grounds in Denmark, the Netherlands and Belgium (Adam 2008). The steady increase of this goose population in the recent past has also increased conflicts with agricultural interests affected by the grazing geese, and raised concerns over the degradation of tundra vegetation in Svalbard (Adam 2008, Johnson and Madsen 2013). The pink-footed goose provides an illustrative example, especially as it involves the actual implementation of distinct elements of the transboundary population level management approach as detailed above.

In 2012, the Meeting of the Parties to the African-Eurasian Waterbirds Agreement (AEWA) (Agreement on the Conservation of African-Eurasian Migratory Waterbirds, June 16, 1995) (Adam 2008), a subsidiary treaty under the Bonn Convention on Migratory Species (CMS) (Convention on the Conservation of Migratory Species of Wild Animals, June 23, 1979, 10 I.L.M. 15) which covers the pink-footed goose, adopted a denominated ‘International Species Management Plan’ (ISMP) for the pink-footed goose population in question (Madsen and Williams 2012). The overarching objectives of the ISMP are to (i) ‘[m]aintain a sustainable and stable Pink-footed Goose population and its range’; (ii) ‘[k]eep agricultural conflicts to an acceptable level’; (iii) ‘[a]void increase in tundra vegetation degradation in the breeding range’; and (iv) ‘[a]llow for recreational use [i.e., hunting] that does not jeopardize the population’ (Madsen and Williams 2012).

The ISMP incorporates a good number of the essential elements of a transboundary population level approach as outlined in the current section above. For instance, the Plan is adjusted to a distinct and well-defined biological unit extending across various countries, namely the Svalbard-breeding population of pink-footed goose. Furthermore, the Plan’s overarching objectives have been translated into specific and measurable targets, including a ‘population size of around 60,000’ geese (Madsen and Williams 2012). The various objectives are pursued through a series of detailed, coordinated conservation and management measures, *inter alia* concerning the reduction of human-goose conflict, the maintenance of the populations' range and connectivity, and the grazing impact on tundra vegetation (Madsen and Williams 2012). An International Working Group has been set up as a central coordinating body, and is composed of one government representative and one expert from each of the four range states (Norway, Denmark, the Netherlands and Belgium) (http://www.pinkfootedgoose.aewa.info). An especially significant feature for present purposes is the approach developed under the ISMP for the control of goose numbers, whereby the overall goose removal target is periodically determined at the transboundary population level, and then translated into recommended hunting bag quotas for the countries involved (Madsen and Williams 2012).

5.3 THE CENTRAL LIMPOPO RIVER VALLEY ELEPHANT POPULATION

The African elephant was once widespread in the southern African sub-region, occurring in high numbers in most areas until the twentieth century when large-scale hunting and ivory trade reduced numbers significantly throughout their range (Plug and Badenhorst 2001, Carruthers 2010). Currently the southern African elephant population constitutes 55% of the total African elephant population(Blanc et al. 2007). Within southern Africa, Botswana holds by far the largest population in the sub-region and on the continent (approximately 150 000 animals), while Mozambique, Namibia, South Africa, Zambia and Zimbabwe still hold large elephant populations (CITES et al. 2013). While elephant numbers appear to be increasing in Botswana and South Africa, there seem to be declines in some of the populations in Mozambique, Zambia and Zimbabwe. Globally, African elephant is listed as 'Vulnerable' (A2a) (IUCN Red List (2008 assessment; www.iucnredlist.org), fitting a worrying pattern applicable to many large herbivores across the globe (Ripple et al. 2015). However, the species is considered 'Least Concern' in the southern African region which includes Botswana, South Africa and Zimbabwe (IUCN Red List (2008 assessment; www.iucnredlist.org). Within all three of these countries the elephant status can be considered a conservation success, but at the same time elephants in the region are the primary agents of ecological change across their range (Kerley et al. 2008), are one of the major causes of human-wildlife conflict (Hoare 1999, 2000) and a source of international controversy (Couzens 2013).

Increasing human population numbers and the concomitant demands on land and natural resources, however, have resulted in a fragmented landscape with protected areas imbedded in a human-dominated landscape (Baeza and Estades 2010, Di Minin et al. 2013b). Several species including large carnivore species and mega-herbivores such as elephant depend on large intact natural areas to accommodate their extensive home ranges and to a certain extent enable regulation of population numbers through natural processes(Woodroffe and Ginsberg 1998, Di Minin et al. 2013b). The majority of protected areas in southern Africa are significantly smaller than what is required for the home ranges of large and, certainly, mega-herbivores (Di Minin et al. 2013b, Packer et al. 2013). As a consequence, and in the absence of population management (Kerley and Shrader 2007, Bertschinger et al. 2008), populations of these species rapidly approach and can exceed the carrying capacity of the protected area which places pressure on the vegetation as well as the boundary fences as the species attempt to migrate or disperse to low density areas (Kerley et al. 2008). More than 80% of the elephant range in Africa still exists outside of proclaimed (state and private) protected areas (Blanc et al. 2007, Abensperg-Traun 2009) and these areas often span administrative and political boundaries such as municipalities and provinces, and in particular international borders (Delsink et al. 2013). Only 20-30% of Botswana's elephant population occurs within formally protected areas. Van Aarde and Ferreira (2009) suggested that there are currently eight elephant conservation clusters in southern Africa. The CLRV elephant population could be considered as the ninth cluster. Of the nine clusters, five span international boundaries. These areas, therefore, are likely to comprise a matrix of multi-use landscapes of potentially divergent administrative, legal and political systems. It is further recognised that the development of the human landscape has been ad hoc which has allowed a continual encroachment by human settlement and agricultural activities (Lindsey et al. 2014). The occurrence of elephant in close proximity to people often results in human-elephant conflict (Hoare 2000). This conflict is naturally exacerbated outside of protected areas particularly in those areas, of southern Africa, of increasing human and elephant densities (Hoare 2000, Jackson et al. 2008, Riddle et al. 2010).

The CLRV elephant population's current distribution spans three southern African countries, namely Botswana, South Africa and Zimbabwe, and includes an area of some 180 km along the Limpopo River between Zanzibar Border Control in the west and Beitbridge in the east, in a belt of about 20 km on either side of the river (Figure 5.1). The elephant population consists of approximately 1,224 ±72.4 individuals and is increasing at <2% per annum (Selier et al. 2014). Historically, however, elephants roamed freely across the CLRV until approximately the start of the twentieth century from when hunting and increased human densities and agricultural activities led to the near extinction of elephants in the Limpopo Valley (Selier 2007, Forssman et al. 2014). With the establishment of the Northern Tuli Game Reserve (NTGR) in Botswana in the early 1970s and its presidential declaration as a private game reserve under the Wildlife and National Parks Act (Wildlife Conservation and National Parks Act, Nov. 10, 1992, sec. 13), elephants started increasing within the region and slowly expanded their range moving east across the Shashe River into Zimbabwe, and further west along the Tuli Block in Botswana (Figure 5.1).

In 2006, the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA) was established with the signing of a Memorandum of Understanding (MoU) by the Governments of the three partner countries (GMTFCA TTC 2011). The GMTFCA is a transboundary park between Botswana, South Africa and Zimbabwe with the present core area covering 2573 km^2 centred on the confluence of the Shashe and Limpopo rivers and including the NTGR (Botswana), Mapungubwe National Park (MPNP) (South Africa) and the Tuli Safari Area (TSA) (Zimbabwe). The park however has the potential to double to 5638 km^2 with the inclusion of additional properties within all three countries (Figure 5.1) (GMTFCA TTC 2011).

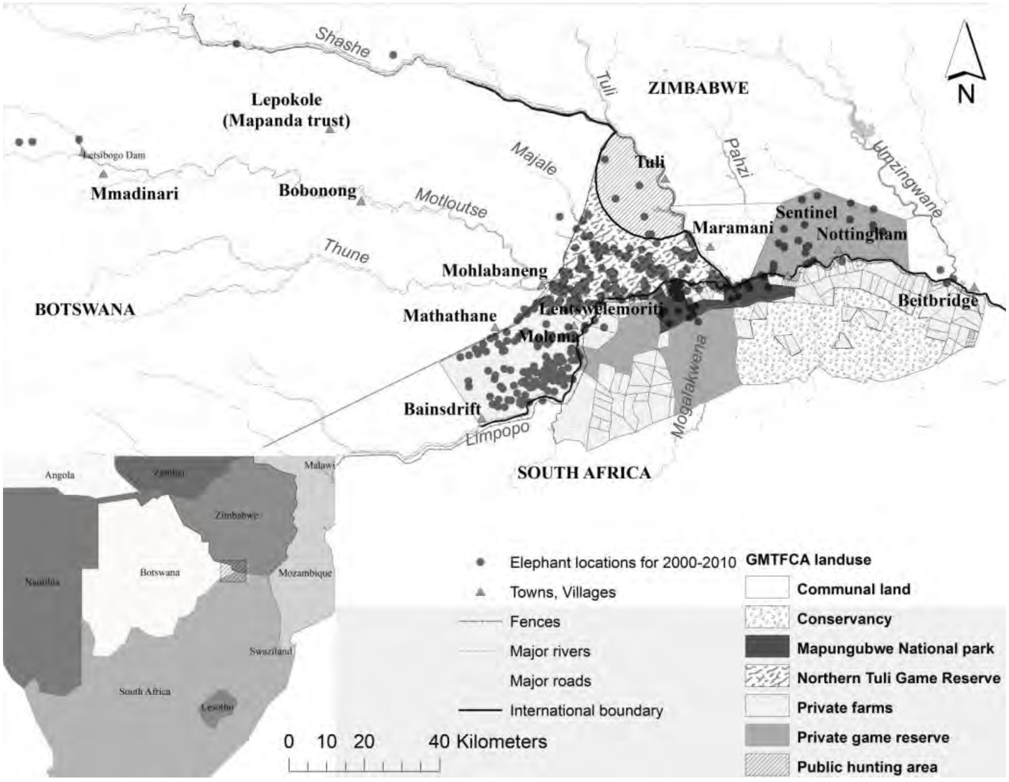

Figure 5.1: The Central Limpopo River study area with different land use practices and the elephant locations for 2000–2010.

Land use and ownership within and surrounding the GMTFCA are unusually diverse, and include contractual partners, private and communal landowners, land claimants, private tourism operations, game farms and subsistence and commercial farmers (GMTFCA TTC 2011). The administrative and governance structures for the conservation areas in the GMTFCA are presented in Table 5.2. Several tourism operations run within the current boundaries of the GMTFCA, all of which use either for viewing or trophy hunting, this single cross-border elephant population that moves freely between the three countries. Photographic tourism is the main economic driver within the area at present (Evans 2010), but several operations rely on a combination of trophy hunting and photographic tourism (Selier et al. 2015).

Table 5.2: Administrative and governance structures for conservation areas within the GMTFCA (replicated directly from GMTFCA TTC, Collaborative Policy and Planning Framework for the Management of Elephants in the Greater Mapungubwe Transfrontier Conservation Area, 2011-2020 (2011) at 46).

Country	**Province**	**District**	**State Land**	**Communal Land**	**Private Sector**
Botswana	Central	Bobonong			NOTUGRE Central Tuli Farm Bock
South Africa	Limpopo	Capricorn Vhembe Waterberg	Mapungubwe National Park	Vhembe Game Reserve Mogalakwena Nature Reserve	Venetia Limpopo Nature Reserve Limpopo Valley Conservancy
Zimbabwe	Matabeleland South	Beitbridge Gwanda	Tuli Safari Area	Maramani Machuchuta Masera Halisupi	Nottingham Sentinel River Ranch

The Northern Tuli Game Reserve forms the original core of the elephant distribution. This is an area of 770 km^2 that lies north of the Limpopo River and west of the Shashe and Motloutse Rivers (Figure 5.1). The farms are privately owned and used for commercial photographic tourism. To the southwest of the NTGR, the Tuli Block extends westwards for approximately 350 km. These farms are used for game ranching, hunting, cattle farming and commercial agricultural production. Movement by game (including elephants) between the NTGR and the remainder of the Tuli Block is relatively unrestricted. West of the NTGR is the communal land of the Batswana that is mainly used for subsistence crop and cattle farming. The number of people varies from around 3000 in towns like Mathathane and Selebi-Phikwe to as few as 10 people in the cattle posts spread out over a large section of the area (Selier 2007). Movements of game between the NTGR and the communal land and between the Tuli Block and the communal areas are partially restricted by a 2 m high electrified game fence. A double 3 m high electrified military fence runs along the Limpopo River on the South African bank opposite Botswana and Zimbabwe, which in places has been removed. North of the NTGR is the Tuli Safari Area (TSA), a 416 km^2 state-owned controlled hunting area managed by the Zimbabwean National Parks and Wildlife Authority. On the eastern side of the Shashe River is a 6 km strip of communal land called Maramani. The area of Maramani covers about 490 km^2 and is inhabited by about 5,200 people and an unknown number of livestock. Sentinel Ranch (300 km^2) is situated east of Maramani. Nottingham Estate, comprising some 250 km^2, is situated east of Sentinel Ranch (CESVI 2001). The main commercial activity on this ranch is citrus farming. Hunting (including elephants) occurs on both farms and within the communal areas to the east, west and north through the CAMPFIRE (Communal Areas Management Programme for Indigenous Resources) program (Selier et al. 2014). The northern borders of both Sentinel Ranch and Nottingham Estate are fenced with a 1.5 m high cattle fence. River Ranch occurs to the east of Nottingham Estate. This is a resettled farm of about 170 km^2. About 60 families have settled within the southern part of the ranch, and use it for livestock grazing (Selier et al. 2015).

The process of establishing Mapungubwe National Park has a long and complex history dating back as far as 1922. In 1983 and 1984 respectively the archaeological sites K2 and Mapungubwe Hill and its southern terrace were declared national monuments in terms of the former National Monuments Act (National Monuments Act (Act no. 28 of 1969)). According to an agreement signed in June 1995 between the provincial government of the Northern Province (renamed the Limpopo Province in 2002) and the South African National Parks (SANParks), the Northern Province would make available the property Greeffswald, then part of the Vhembe nature reserve, to be declared a national park in terms of the National Parks Act (National Parks Act (Act no. 57 of 1976 as amended)). The park was provisionally known as Vhembe/Dongola National Park, but was later renamed as Mapungubwe National Park (MPNP) (National Parks Act (Act no. 57 of 1976 as amended)). In 2003, the Mapungubwe Cultural Landscape, synonymous with Mapungubwe National Park and National Heritage Site was designated as National and World Heritage sites (SANParks 2013). The current national park consists of land managed by SANParks under contract with the landowners (SANParks 2013). The total surface area of the park declared in terms of South African legislation (National Environmental Management: Protected Areas Act (NEM:PAA) (Act no. 57 of 2003)) is 153 km^2 which includes seven privately owned contracted properties, with an additional 490 km^2 in the process of being designated (SANParks 2013). A further 45 km^2 privately owned land managed under contract by SANParks but not designated and 127 km^2 privately owned land that is not managed by SANParks are present within the core area of the World Heritage site (SANParks 2013).

Due to the establishment of the national park and the development of the GMTFCA some fences between Botswana and South Africa and between Zimbabwe and South Africa were removed allowing elephant access to Mapungubwe National Park and large sections along the Limpopo River within South Africa. As a result elephants have been expanding their range east and west along the Limpopo River. However, movement of elephants further into South Africa are restricted by electrified game fencing, and, thus is limited to those properties bordering the Limpopo River. The expansion of the elephant's range, and the inclusion of areas outside of formally PAs, have brought elephant into conflict with both commercial farmers on the South African side, as well as local communities within Botswana and Zimbabwe (Selier et al. 2014). Elephants are usually associated with a wide range of conflicts. Most common are conflicts associated with their impact on agricultural crops and infrastructure such as wells (Sitati et al. 2003, Sitati et al. 2005). A second conflict specifically within southern Africa is the possible impact elephants can have on riverine habitat, through the removal of spectacular large trees with high aesthetic and ecological value (Kerley et al. 2008). Beyond these conflicts (which have a physical, material and economic basis), are a wide range of social conflicts that range from a direct fear for personal safety in the presence of elephants to a fear of the socio-economic changes that elephants often come to symbolise (Sitati et al. 2012). These conflicts, when combined often lead to a very low tolerance of elephants among rural communities with whom they have to share living space (Twine and Magome 2008).

Where wildlife, in particular the elephant, has no direct benefit to landholders, it is bound to disappear in the dispersal areas surrounding protected areas, and when there are no dispersal areas, the protected areas will become islands within which wildlife is likely to disappear sooner or later (Prins and Grootenhuis 2000). In contrast, however, where communities in dispersal areas receive revenue from a species, they are more likely to conserve it, and be more tolerant of negative impacts arising from the dispersing species (Hurt and Ravn 2000, Lindsey et al. 2007, Blignaut et al. 2008). Within the CLRV, only a part of the elephant population's range is currently protected, within the boundaries of the GMTFCA. As a result human-elephant conflict is a concern in both agricultural and rural communities bordering the GMTFCA in all three countries, with elephants causing extensive damage to crops and wells (Selier 2007). Apart from trophy hunting, elephants (mainly bulls) are destroyed as damage-causing animals (DCAs). Depending on local policy and practice, DCAs may be professionally hunted or destroyed by the conservation agency (Hopkinson et al. 2008). In South Africa alone 19 bulls were destroyed in 2011 as DCAs on properties bordering the Limpopo River (Selier et al. 2014).

5.4 GLOBAL AND REGIONAL LAW AND POLICY

5.4.1 Global instruments

Wildlife management has long been regulated at the international level (Bowman et al. 2010). A key global agreement regulating the use of elephant is the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) (Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES), March 3, 1973, 12 I.L.M. 1085), which has 181 parties (Couzens 2013). CITES provides a legal framework to regulate the international trade in specimens of wild animals and plants and their derivatives, listed in three appendices, through export and import permit systems. The aim of the Convention is to protect species against overexploitation as a result of international trade. The latter poses a significant threat to elephant. Article III of the Convention deals with species that are threatened with extinction included in Appendix I, and prohibits, with few exceptions, international commercial trade in these species. Trade in Appendix I species is further subject to strict requirements. Article IV of the Convention deals with species that are not yet threatened, but which may become so unless trade is controlled, and these species are listed in Appendix II. Appendix III concerns species subject to national regulation and requiring international cooperation for trade control. The Convention requires states to adopt legislation that i) designates at least one management authority and one scientific authority; ii) prohibits trade in specimens in violation of the convention; and iii) penalizes such trade, calling *inter alia* for the confiscation of specimens illegally traded or possessed.

In 1977, all populations of the African elephant were listed on Appendix II of the convention limiting the international trade in elephants and their products (Van Aarde and Ferreira 2009). In 1989, due to increased poaching levels and illegal trade in ivory and a resultant rapid decline in elephant numbers as derived from data in the Elephant Trade Information System (ETIS) and Monitoring the Illegal Killing of Elephants Programme (MIKE), all African elephant populations were uplisted to Appendix I, effectively banning all international trade in elephant (Van Aarde and Ferreira 2009). Many southern African countries disagreed with the African elephant trade ban, and continued to argue against it indicating that international trade in ivory from their countries is justified (Stiles 2004, Couzens 2013). In 1997, at the 10th CITES Conference of the Parties (COP), the populations of African elephant in Botswana, Namibia and Zimbabwe were downlisted to Appendix II (Stiles 2004, Couzens 2013), with the following annotation:

"*Populations of Botswana, Namibia and Zimbabwe: For the exclusive purpose of allowing: 1) export of hunting trophies for non-commercial purposes; 2) export of live animals to appropriate and acceptable destinations (Namibia: for non-commercial purposes only); 3) export of hides (Zimbabwe only); 4) export of leather goods and ivory carvings for non-commercial purposes (Zimbabwe only). No international trade in ivory is permitted before 18 months after the transfer to Appendix II comes into effect (i.e. 18 March 1999). Thereafter, under experimental quotas for raw ivory not exceeding 25.3 tonnes (Botswana), 13.8 tonnes (Namibia) and 20 tonnes (Zimbabwe), raw ivory may be exported to Japan subject to the conditions established in Decision of the Conference of the Parties regarding ivory No. 10.1. All other specimens shall be deemed to be specimens of species included in Appendix I and the trade in them shall be regulated accordingly*."

In 2000, the South African elephant population followed those of the other three southern African countries and was downlisted to Appendix II with the same annotation (Stiles 2004, Couzens 2013). Botswana, South Africa and Zimbabwe are parties to this Convention; all three countries subscribe to the sustainable use concept and have pleaded on more than one occasion for the sale of stockpiled ivory. Botswana has a CITES export quota of 800 tusks as hunting trophies (400 elephant), South Africa 300 tusks as trophies (150 elephant) and Zimbabwe a 1000 tusks as trophies (500 elephant).

The Convention on the Conservation of Migratory Species of Wild Animals (CMS or Bonn Convention) (Convention on the Conservation of Migratory Species of Wild Animals (CMS), June 23, 1979, 19 I.L.M. 15), similar to CITES, is a species-based agreement focusing on the immediate protection of certain species included in lists, differentiating according to the degree of threat. The CMS aims to conserve terrestrial, marine and avian migratory species throughout their ranges, requiring cooperation among 'range states' host to migratory species regularly crossing international boundaries. Migratory species can be included in one or both of the Appendices. The Convention defines 'migratory species' as species 'whose members cyclically and predictably cross one or more national jurisdictional boundaries'(Article 1(1)(a)), but this has subsequently been interpreted by the CMS COP in a flexible manner, as encompassing any species whose range extends across more than one country (Trouwborst 2012). This approach has enabled the inclusion of species and populations that can hardly be considered migratory in the classical sense – as in the case of the CLRV elephant population. As such the CMS has evolved into an instrument that focusses on the conservation of transboundary rather than purely migratory wildlife (Trouwborst 2012). The African elephant is included in Appendix II (species with an unfavourable conservation status). CMS parties that are range states of Appendix II species are required to conclude global or regional agreements to maintain or restore the species concerned to a favourable conservation status (Article IV(3)). These agreements can either be in the form of 'AGREEMENTS' under Article IV(3) or less formal 'agreements' under Article IV(4). Such subsidiary instruments can take the shape of treaties or non-binding Memoranda of Understanding (MoU). With respect to AGREEMENTS under Article IV(3), these should, 'where appropriate and feasible,' *inter alia* provide for:

1. 'Conservation and, where required and feasible, restoration of the habitats of importance in maintaining a favourable conservation status, and protection of such habitats from disturbances;
2. Maintenance of a network of suitable habitats appropriately disposed in relation to the migration routes;
3. Where it appears desirable, the provision of new habitats favourable to the migratory species;
4. Elimination of, to the maximum extent possible, or compensation for activities and obstacles which hinder or impede migration;
5. Measures based on sound ecological principles to control and manage the taking of the migratory species' (Article V(5)).

Whereas a CMS MoU for the West African elephant population came into effect in 2005, to date no agreements under either Article IV(3) or IV(4) of the CMS has been developed for elephant within the southern African region. South Africa and Zimbabwe are parties to the CMS, but Botswana is not. The fact that Botswana is not yet a party to the CMS, however, would not stand in the way of Botswana becoming a party to any future subsidiary CMS agreement(s) covering elephants (Trouwborst 2015).

Many other international legal instruments are of relevance for present purposes, even if they do not specifically list elephant. One of these is the Convention on Biological Diversity (CBD) (Convention on Biological Diversity (CBD), June 5, 1992, 1760 U.N.T.S. 79) is an overarching agreement specifically addressing biodiversity conservation and sustainable use on an ecosystem, species and genetic level (Article 1). The Convention's 193 contracting parties include Botswana, South Africa and Zimbabwe. Even though the CBD lacks lists of species requiring special attention, many of its obligations are of relevance to elephant. These include duties regarding the *in situ* conservation (Article 8), *ex situ* conservation (Article 9), sustainable use of biodiversity (Article 10), socio-economic measures acting as incentives for conservation and sustainable use (Article 11), and environmental impact assessments (Article 14). The Convention provides guiding principles that should be taken duly into account when developing national policy and laws. The CBD COP has adopted specific principles and operational guidelines on sustainable use, which provide guidance to ensure that the use of the components of biodiversity will not lead to the long-term decline of biological diversity (Addis Ababa Principles and Guidelines for the Sustainable Use of Biodiversity, CBD COP Decision VII/12, February 20, 2004).

The World Heritage Convention (UNESCO Convention Concerning the Protection of the World Cultural and Natural Heritage, Nov. 16, 1972, 11 I.L.M. 1358) is also relevant, in particular due to the listing of Mapungubwe National Park as a cultural World Heritage site (GMTFCA TTC 2011). As parties to the Convention, Botswana, South Africa and Zimbabwe are as far as possible to identify, protect, conserve, present and transfer heritage sites within their territories (Article 4). Article 5 of the Convention stipulates that each party 'shall endeavour, in so far as possible, and as appropriate for each country,' to 'integrate the protection of that heritage into comprehensive planning programmes' and to 'take the appropriate legal, scientific, technical, administrative and financial measures necessary for the identification, protection, conservation, presentation and rehabilitation of this heritage'. In general, those species whose habitat is situated within a listed World Heritage site are likely to benefit from the protection regime imposed by the Convention (Trouwborst 2015). In some cases, however, conflict might arise between the conflicting objectives set out to conserve a cultural landscape, and those species occupying the landscape. This is the situation with elephants occupying the Mapungubwe Cultural Landscape. The gallery forest within the park is considered part of the ambience of the cultural heritage (SANParks 2013). At the same time, these forest areas are also favoured by elephants (Chamaille-Jammes et al. 2007b, Shannon et al. 2009). Over time the impact of elephants on the forest has been significant and has become a bone of great contention. In an attempt to reduce the elephant impact, a section of the gallery forest in proximity to Mapungubwe hill has been fenced to exclude elephants from this part of the park (SANParks 2013).

5.4.2 Regional instruments

In addition to these four global treaties, many regional legal instruments are of relevance for present purposes. The earliest record, from an international perspective, that African elephant populations were under threat from both hunting and habitat loss can be traced back to the 19th century (Lausche 2008), with the drafting by several colonial powers of the Convention of the Preservation of Wild Animals, Birds and Fish in Africa – the 1900 London Convention (Convention of the Preservation of Wild Animals, Birds and Fish in Africa, May 19, 1900). This Convention set up a mechanism for the protection of 'useful' or 'harmless', or rare and endangered wild animal species and the reduction of pest species (Articles II(1), II(13) and II(15)). The mechanisms included a prohibition of consumptive use of those species that were considered rare or were threatened by extinction (schedule I). For elephants, the Convention prohibited hunting of young animals and specifically young elephants with tusks less than five kilogrammes (Schedule II(11)). This Convention never entered into force as the majority of the signatory states failed to ratify it, although its provisions did exercise an influence on the administration of colonies in southern (and eastern) Africa (IUCN 2006).

The 1900 London Convention was followed by the 1933 London Convention Relative to the Preservation of Fauna and Flora in their Natural State, which entered into force in 1936 (Convention Relative to the Preservation of Fauna and Flora in their Natural State, Nov. 8,

1933) (Hayden 1942). The lack of decision-making institutions and secretariat services proved to be a significant inadequacy of the Convention and consequently afforded little protection of elephants (Hayden 1942). Furthermore, the Convention lacked a general policy for the protection of nature in Africa which embraced the interests and expectation of the African people themselves (IUCN 2006). The correction of this Convention was overtaken by the decolonisation of Africa, resulting in the purpose and benefits of the convention not being applied to either elephant conservation, nor peoples use and management thereof. The first conservation milestone for the newly formed 21 African states was the Arusha Manifesto of 1961 (Watterson 1961). The key driver for the Arusha Conference was the concern that natural resources were deteriorating and this was creating or driving socio-economic problems in Africa (Watterson 1961). The Manifesto also recognized the critical need for co-operative trusteeship between African states as a significant mechanism to conserve and protect dwindling natural resources. The Arusha Manifesto gave rise to the 1968 African Convention on the Conservation of Nature and Natural Resources (the Algiers Convention) (African Convention on the Conservation of Nature and Natural Resources, Sep. 15, 1968) which replaced the 1933 London Convention. In turn, the Algiers Convention will be superseded by the (revised) African Convention on Conservation of Nature and Natural Resources (African Convention on Conservation of Nature and Natural Resources, July 11, 2003), which was adopted in Maputo in 2003 (the Maputo Convention), when it enters into force (IUCN 2006).

As parties to the Algiers Convention, it is incumbent on Botswana, South Africa and Zimbabwe to co-operate with respect to elephant population management and to refrain from taking parochial decisions that may have adverse impacts on this shared wildlife resource. In particular, they are to grant special protection throughout their territories to species such as elephant listed in the Convention's Annex. This includes the prohibition of their 'hunting, killing, capture or collection (Article VIII).' For elephant with tusks over 5 kg each 'Class B' this prohibition may, however, be lifted 'under special authorization' at the discretion of the 'competent authority'. For elephant with tusks under 5 kg each 'Class A', exceptions may be made 'only on the authorization in each case of the highest competent authority and only if required in the national interest or for scientific purposes (Article VIII).' Other relevant provisions *inter alia* address habitat protection (Article X), and the generic restriction of certain means of capture and killing, such as a prohibition on the use of poisoned baits (Article VII). As regards the revised 2003 Maputo Convention, of the three countries under consideration only Botswana is a signatory.

A relevant forum with a more delimited geographical scope is the Southern African Development Community (SADC). The SADC's Regional Indicative Strategy Development Plan (RISDP), adopted in 2003, is a 15-year regional integration framework, setting the priorities, policies and strategies for achieving the long term goals of SADC, and providing guidance to member states, regional stakeholders and international partners in achieving these goals (SADC 2003). The RISDP contains a section specifically addressing wildlife (SADC 2003). The promotion of community-based natural resource management (CBNRM) programmes, TFCAs, common management practices, sustainable wildlife utilization, and capacity building are some of the strategies set out in the RISDP which are of relevance to elephant management (SADC 2003).

The principal legally binding instrument of the SADC for present purposes is the SADC Protocol on Wildlife Conservation and Law Enforcement (Protocol to the SADC Treaty on Wildlife Conservation and Law Enforcement, Aug. 18, 1999), to which Botswana, South Africa and Zimbabwe are parties. The Protocol seeks to establish a framework for, *inter alia*, the conservation and sustainable use of wildlife resources in the SADC Region (Cirelli and Morgera 2010). Whilst recognizing the sovereign rights of the parties, this framework includes recognition that biodiversity, and particularly transboundary biodiversity – e.g., a transboundary elephant population – is most efficiently safeguarded through international cooperation. Furthermore, the Protocol directs state parties from 'causing damage to the wildlife resources of other states or in areas beyond the limits of national jurisdiction.' The management of the transnational elephant population outside of a joint management agreement by one state party may be prejudicial not only to the elephant population's wellbeing, but to the other state parties' legitimate access to and use of this wildlife resource. In such circumstances, the Protocol provides for interstate co-operation particularly on matters where a decision taken by one state is 'likely to affect the natural resources of any other State (Article XVI(1)(b)).' Thus the removal of what is deemed to be excess or damage causing animals (DCAs) by one state may negatively impact on another's opportunity to do the same and hence the need for co-operative management agreement of the transnational elephant population by the state parties. Such agreement would provide for and adjustment of the allocation of resources (if required) when a state party is required to undertake extraordinary action. This includes *force majeure*, defence of human life, or defence of property (Article XVI (2)). When such circumstance arises, the cause and action taken must be shown to be unique and in accordance with the purpose of the action taken (Article XVI (2)).

Finally, each state party must implement and interpret its domestic legislation, policies and biodiversity management for the conservation and persistence of the shared or transnational biodiversity (See, e.g., Articles 3, 5 and 6). Article 6 makes provision for parties to cooperate to, *inter alia*, achieve a framework for the management and use (including removal) of wildlife, as well as enforcing compliance with multilateral agreements and applicable domestic laws providing for its protection and conservation, and preventing overexploitation and extinction of species and habitats (Article 7). As a mechanism to jointly achieve the necessary level of protection and use of wildlife/elephants, the cooperating state parties are required to collect information (i.e. monitor), and share information with each other, and from that sharing provide for the joint management of the species (Article 8). This joint management function is operationalized through a 'Wildlife Sector Technical Coordinating Unit' (Article 5).

A regional instrument specific to elephant, albeit not legally binding, is the Southern Africa Regional Elephant Conservation and Management Strategy drafted in 2005 (SADC 2007). It highlights that most key elephant populations in the region are shared and move across international boundaries, that populations are not evenly distributed across the different range states and that there is a set of issues and concerns common to all range states (SADC 2007). The purpose of the Strategy is to facilitate coordination, collaboration and communication in the management of elephant populations across the region so as to conserve elephants and expand their range within historic limits, forming as contiguous a population as possible across southern Africa, and, in so doing, realizing their full potential as a component of wildlife-based land use for the benefit of the region and its people. The Strategy has a strong emphasis on sustainable utilization. It strives to foster appropriate coordination at a transboundary level regarding land use planning, human-elephant conflict mitigation measures, law enforcement, management of trophy hunting, other management offtake exercises, and understanding and accommodating cross-border elephant movement. It expressly aims for the harmonization of policies in these regards, and for the development and implementation of 'agreements/protocols on management of cross border populations (SADC 2007).'

5.4.3 Appraisal

It is thus clear that a significant body of international law and policy of importance for elephant conservation already exists, at varying levels and with varying degrees of detail. Principles uniform to all the relevant overarching instruments include the conservation and sustainable use of biodiversity, the need for cooperation between range states sharing wildlife populations, and the need to harmonise wildlife legislation where dealing with resources or species that straddle across countries' borders. The SADC Protocol provides the necessary omnibus for the harmonisation of wildlife legislation across SADC country boundaries (Cirelli and Morgera 2010).

However, even though the idea of managing species at a population level rather than within administrative boundaries is gaining momentum at the international level, within international treaties regulating the sustainable use of species, such as CITES, the prevailing unit continues to be the range state rather than the biological population entity. For instance, decisions on trade in wildlife/elephant products depend on country-specific information, mostly of limited precision, provided by MIKE and ETIS. However, most major populations span several countries and elephant move freely across borders (Mpanduji 2009). Decisions targeting one country therefore may be undermined by factors affecting elephants in another country (Frank and Maurseth 2006). This further highlights the need for transboundary cooperation between range states.

Both the CMS and the SADC Protocol place emphasis on the need for transboundary cooperation. The CMS expressly calls for cooperation among range states, promoting the development of common provisions for the proper management of transboundary areas or species. The significance of the CMS for elephant is, however, curtailed because some important range states, in this case Botswana, are missing as contracting parties. Cooperation on a regional level is, at any rate, also important for the effective implementation of global and regional international agreements. To illustrate, implementing uniform penalties by neighbouring countries will assist in preventing the bypassing of CITES rules, which could result from choosing to trade wildlife in certain countries rather than others (Cirelli and Morgera 2010).

The SADC Protocol provides for a collective conservation framework for, in particular, protection and sustainable use of wildlife populations that extend and fulfil the lifecycles between two or more counties. The fulcrum of the SADC Protocol is that it restrains each country, when making decisions on a shared wildlife resource, from 'causing damage to the wildlife resources of other states or in areas beyond the limits of national jurisdiction (SADC Protocol Article 3).' Thus when the removal of animals in one country has the potential to render an equivalent removal in another unstainable, then it may be argued that the first removal would be in contrary to this provision of the Protocol. As a means to prevent such circumstances arising, the key objectives of the Protocol include provisions that promote sustainable use of wildlife, the exchange of information concerning wildlife management and use, and the fostering of the conservation of shared wildlife resources through the establishment of transfrontier conservation areas (Article 4). Further a series of operational governance structures (e.g. Wildlife Sector Technical Coordinating Unit, Committee of Senior Officials and the Technical Committee) exist to ensure that the objectives of the Protocol are achieved. Within these structures, the cooperating countries must *inter alia*, establish co-operative management programmes for the conservation of transboundary wildlife that prevents over-exploitation and extinction of species exploited. Finally, the Protocol provides a mechanism for the sharing of information that harmonises the monitoring and control of transboundary wildlife. In such, it would be incumbent on each country to ensure that the removal of animals in excess of the jointly agreed quota has no significant consequence for the population overall or the interests of another country.

The SADC Protocol, therefore, represents one of the most advanced efforts towards regional harmonisation of wildlife legislation that is being experimented with around the world (Article 4). SADC countries have already formed a 'Community' and have institutionalised cooperation in numerous sectors (Article 4). A high level committee of senior officials meets, as part of the SADC Ministers Responsible for Environment and Natural Resources, once every two years (Article III). However, decisions taken do not always translate into actions on the ground. This is a clear example of a mismatch in temporal scales as well as where social processes lead to a scale mismatch as a result of fragmentation of responsibilities and the lag effect of bureaucracies in dealing with ecological issues (Cumming et al. 2006).

Specific policy for transboundary elephant conservation in Southern Africa exists in the form of the Regional Elephant Conservation and Management Strategy, which however still awaits formal adoption and implementation.

This touches on another general issue, namely that, in order for international wildlife treaties and other instruments to play an effective role in the conservation of transboundary species, compliance by range states with their international obligations and related commitments is required. States do, however, at times seem to neglect wildlife conservation obligations, especially where these might have considerable socioeconomic consequences. Even so, it seems fair to assume that international wildlife instruments play a significant role and that the conservation status of several species would have deteriorated (further) without them (Trouwborst 2015).

5.5 TRILATERAL AGREEMENTS AND POLICY

The GMTFCA MoU was signed by the three participating countries in 2006. It is significant for present purposes that one of the MoU's objectives is to 'enhance ecosystem integrity and natural ecological processes by harmonising wildlife management procedures across international boundaries and striving to remove artificial barriers impeding the natural movement of animals (GMTFCA MoU, Article 6(1)(c)).' A Trilateral Technical Committee (GMTFCA TTC) and several working groups were established to deal with the formulation of sectoral plans aimed at the adoption of an integrated Development Plan for the GMTFCA, and the signing of the Treaty in which the operational procedures for managing the GMTFCA will be established. In 2011, the GMTFCA resource management committee was formed to deal with cross-border challenges at an operational level. The Treaty between Botswana, South Africa and Zimbabwe on the establishment of the GMTFCA has to date not been signed. Several draft joint management plans have been developed, but have not yet been approved or implemented. These include a GMTFCA Large Predator Management Plan (SLPRG 2010) and, significantly, a Collaborative Policy and Planning Framework for the Management of Elephants in the GMTFCA 2011-2020 (hereinafter 'GMTFCA Elephant Management Plan') (GMTFCA TTC 2011).

The GMTFCA Elephant Management Plan envisages the presence of elephants as integrated drivers of ecosystem integrity, benefiting all stakeholders and enhancing the livelihoods of people, thereby contributing to the social, cultural, ecological and economic development of the Transfrontier Conservation Area (GMTFCA TTC 2011). In addressing identified issues of conservation, protection and ecological management, including veterinary disease control, together with human-elephant conflict minimisation and livelihood improvements of local people, the strategic goal is to maintain and adaptively manage variable elephant use of cultural and biological landscapes, enhance rural livelihoods and improve wildlife benefits, whilst reducing conflict and engaging stakeholders through effective communication. Five specific objectives have been formulated to achieve the above: elephant populations will be (1) conserved, and (2) protected; (3) elephant impacts will be managed; (4) populations will be sustainably used across the GMTFCA landscape in collaboration with local stakeholders; and, (5) human-elephant conflict will be reduced through spatial planning, mitigation measures and increased benefits. Accompanying each of these objectives is a set of strategies and actions.

The development of the GMTFCA and its Elephant Management Plan form a positive start towards the management of the CLRV elephant on a population level. However, the process of developing the GMTFCA has been very slow, and nearly 10 years after the signing of the MoU the Treaty between the three countries has not yet been signed, hampering efforts to implement management plans and collaborative law enforcement for the conservation of elephant. This is an example of a mismatch in temporal scales, where bureaucracies are slow in dealing with rapid ecological changes that require quick management actions. Furthermore, the draft Elephant Management Plan only applies to part of the elephant's range, albeit a significant part of its overall range (Selier et al. 2014).

When evaluating the draft GMTFCA Elephant Management Plan against the template provided in the Carnivore Guidelines, which sets out the ingredients that each transboundary management plan should contain, the degree of conformity with this blueprint is striking. Table 5.1 provides for a detailed comparison in this regard. The GMTFCA Elephant Management Plan includes many of the measurable objectives required, although it fails to include certain others. The draft GMTFCA Elephant Management Plan does not define or include a favourable reference population value or the favourable reference range. The Plan suggests regular aerial surveys to monitor elephant population numbers, distribution and trends, but lacks population goals and a set of measurable parameters to measure the success of management actions. It further indicates that a single quota should be developed for the GMTFCA elephant population, but no attempt has been made towards discussions between range states to develop a single overall offtake quota for the population, let alone the division of this overall quota between the countries involved. The lack of cross-border cooperation in the management of elephant and the implementation of a single offtake quota shared by all three countries has resulted in individual countries implementing unsustainable trophy hunting quotas based on restricted subsets of population data (Selier et al. 2014). Excessive hunting can lead not only to a reduction in numbers (Lindsey et al. 2007), but also to disturbance effects that force species, such as elephant, to trade-off between disturbance avoidance and good food and water availability (Selier et al. 2015). Current management decisions are thus not made at the appropriate decision level for the species under consideration, with a resultant mismatch in spatial scales (Cumming et al. 2006). This has far-reaching implications for cross-border species, where the stress effects of hunting could be transmitted to ecotourism areas within neighbouring countries (Delsink et al. 2013, Selier et al. 2015). In order to understand the consequences of management activities such as trophy hunting and to implement an adaptive quota system based on population trends, long-term monitoring is essential (Selier and Di Minin 2015). Where an effective monitoring system with clear objectives is in place, consumptive utilisation is sustainable. The GMTFCA Elephant Management Plan further includes objectives to maintain and enhance connectivity within the population, but lacks objectives and actions to enhance connectivity with neighbouring populations. The legal framework as it pertains to elephant for each country is highlighted in the management plan but no attempt is made to describe any changes in legislation that are needed to bring about population level management.

The effective implementation of the GMTFCA draft Elephant Management Plan will depend on whether a legal framework can be established within which collaborative planning and law enforcement relating to elephant management in the GMTFCA can be practiced. This will require harmonisation of wildlife legislation among the three countries where dealing with resources or species that straddle countries' borders. Multilateral agreements have set in place a cooperative framework, but may have different officials associated with their implementation to those operating within the framework of trilateral or bilateral agreements. This makes coordination and integration of elephant management extremely difficult, particularly on matters relating to sovereignty, i.e., the setting of offtake quotas, and the management of DCAs.

5.6 NATIONAL LAW AND POLICY

5.6.1 Botswana

The Botswana Wildlife Conservation Policy (Wildlife Conservation Policy of 1986) deals with utilisation of wildlife resources outside of protected areas. Hunting is, in principle, allowed outside protected areas, and the policy aims at sustainable harvesting of wildlife resources and an equitable distribution of the benefits, while also encouraging the development of a commercial wildlife industry that is viable in the long term. The Policy further deals with the zoning and protection of wildlife areas, land use planning and zoning for wildlife and the protection of wildlife migration. Land use planning must accord wildlife resources a position that reflects their considerable economic significance through protected areas (preservation), Wildlife Management Areas (WMAs; conservation and sustainable utilisation) and Controlled Hunting Areas (CHA; licensed hunting) (Wildlife Conservation Policy of 1986).

The Wildlife Conservation and National Parks Act (Act No. 28 of 1992) is still the main piece of legislation concerning wild animals and will be superseded by the Botswana Wildlife Act of 2008, once it enters into force. It resulted from the merger of the Fauna Conservation and Parks Acts (Cirelli and Morgera 2010). The objective of the Act is to make provision for the management, utilisation and conservation of the country's wildlife resources so as to generate development benefits for current and future generations of Botswana; to maintain the country's biodiversity; to give effect to CITES and any other international conventions for the protection of fauna and flora to which Botswana is a party; and to provide for the establishment, control and management of wildlife areas. Numerous regulations have been adopted to operationalise the Act (E.g., Wildlife Conservation (CITES) Amendment Order of 1999; National Parks and Game Reserve Regulations (SI 28 of 2000); Wildlife Conservation (Hunting and Licensing) Regulations (SI 35 of 2001); Wildlife Conservation and National Parks (Amendment) Regulations of 2006). The competent ministry and government wildlife agency in the present subject area is the Department of Wildlife and National Parks (DWNP). Its director also acts as the CITES Management and Scientific Authority. The Wildlife Conservation and National Parks Act expressly grants ownership of wild animals to the owner of land on which animals are kept or confined within a game proof fence (Section 83). Elephant is listed as a partially protected game animal which can be hunted under license (Section 18, part 1). However, in January 2014 a temporary hunting ban was introduced and no quotas, licenses or permits will be issued for the hunting of part 1 and part 2 schedule game animals listed under the Act (eNCA August 1, 2015, BBC News November 29, 2012).

Private Game Reserves and Game Ranches are subject to the Game Ranching Policy. This policy complements the Wildlife Conservation Policy by increasing economic returns from wildlife outside of protected areas and WMAs; developing an environmentally friendly game ranching industry; promoting species conservation through game farming; ensuring a viable and healthy wildlife game population for stocking of ranches; promoting Batswana participation; creating jobs and income and economic diversification. The principal resource management objective for Private Game Reserves, used mainly for ecotourism purposes, is biodiversity conservation as determined by the owner and endorsed by government. The primary objective for Game Ranches is the sustainable utilisation of wildlife resources, maintaining biodiversity and economic use of wildlife which includes consumptive utilisation through hunting, cropping for meat production and captive breeding, translocation and restocking.

The Botswana Department of Wildlife and National Parks developed a draft management plan for elephants in 1991 which was never implemented. In 2003, the government carried out a review of the 1991 plan and drafted the National Policy and Strategy for the Conservation and Management of Elephants in Botswana ('Botswana Elephant Plan') (2003). The Plan aims to conserve and optimise elephant populations while ensuring the maintenance of habitats and biodiversity, promoting the contribution of elephants to national development and to human communities within their range, while minimizing their negative impacts on rural livelihoods. With regard to limiting risks to human life and property, management actions are to be considered that pose the least risk, are feasible, practical, and economically and aesthetically acceptable.

The four primary objectives identified in the Botswana Elephant Plan are to (1) reduce human-elephant conflicts to acceptable levels; (2) prevent, reduce or reverse unacceptable elephant-induced environmental changes; (3) maximise the benefits from sustainable utilisation of elephants; and (4) protect elephants through law enforcement. Because of varying land uses, the Plan breaks down activities to tailor the specific objectives to geographic units. In the Bobirwa sub-district of the country, the intention is to maximise benefits through both consumptive (trophy hunting) and non-consumptive (photographic tourism) utilisation (GMTFCA TTC 2011). The Plan's provisions mostly target northern Botswana, with no management prescriptions provided for the Tuli area of interest to the GMTFCA. The Plan does, however, emphasise the importance of cooperation between neighbouring countries in elephant management. Where TFCAs are developed, the Plan encourages the harmonisation of elephant management amongst participating countries; setting specific targets in this regard (Objective 5). These include the setting up of an inter-governmental committee to deal with cross-border issues. Measures to reduce human-elephant conflict include elephant-free zones; reduction of elephants through translocation, culling or attracting animals away from areas of concern; and training and empowerment of communities within elephant range to carry out control measures, to increase both tolerance and effectiveness of measures. Botswana is the only country that maintains a Problem Animal Control Unit, within the DWNP.

5.6.2 South Africa

According to South African common law, wild animals enjoying a state of natural freedom are considered *res nullius* (Hopkinson et al. 2008). However, if certain requirements are met, ownership of a wild animal can be acquired (Joubert 1999). In particular, ownership can be established through control and constraint, e.g. through suitable fences (Game Theft Act (Act no. 105 of 1991)). Within South Africa, wildlife management occurs separately at the national and provincial levels, and unfortunately uniformity between national and provincial legislation – or indeed between different pieces of provincial legislation – is not always ensured (Hopkinson et al. 2008).

The National Environmental Management Act (NEMA) (Act no. 107 of 1998) provides the primary legislation for the management of natural resources in South Africa. Within that framework, the legal basis for elephant management is provided by the National Environmental Management: Protected Areas Act (NEM:PAA) (Act no. 57 of 2003); the National Environmental Management: Biodiversity Act (NEM:BA) (Act no. 10 of 2004); the NEM:BA Threatened or Protected Species Regulations (ToPS Regulations 2007); and the Norms and Standards for the Management of Elephants in South Africa ('Elephant Norms and Standards'(Government Gazette No. 30833, Feb. 29, 2008). Over the years, the focus of this legislation has shifted from narrow protectionism to sustainable use.

The objective of NEM:PAA is to provide for a national PA system. It requires overall and subsidiary management plans for protected areas (NEM:PAA, section 39 and 41(4)). NEM:BA provides for the management and conservation of South Africa's biodiversity; the protection of species and ecosystems that warrant national protection; and the sustainable use of indigenous biological resources (NEM:BA). The ToPS Regulations were promulgated to operationalise the NEM:BA permit system for restricted activities involving threatened and protected species; to provide for the registration of captive breeding operations, commercial exhibition facilities, game farms, sanctuaries, rehabilitation facilities and the like; to regulate hunting of ToPS species; to completely prohibit the carrying out of certain activities in respect of certain ToPS species; and to provide for operation of the Scientific Authority. Elephant is listed as protected due to its high conservation value and international trade value (NEM:BA, Section 56(1)(d)). All activities regarding elephant, i.e. translocation, hunting, etc., require a prior permit.

The Elephant Norms and Standards, which apply to wild and captive elephants alike, came into effect in 2008 NEM:BA Section 9). They are not themselves legally binding, but assist officials in implementing the applicable laws to elephants. Their purpose is to ensure that elephants are managed in a way that warrants the long-term survival of elephants within the ecosystems in which they occur or may occur in the future; to promote broader biodiversity and socio-economic goals; and to enable achieving specific protected area management objectives. The document provides for three types of areas where elephant could be found, namely (1) a controlled environment; (2) an extensive wildlife system (where elephants are covered by the ToPS Regulations); and (3) a limited wildlife system. The situation in the CLRV does not fit option (1) or (3) above, but could possibly fit option (2) (extensive wildlife system). However, many properties along the Limpopo River do not meet the definition of an extensive wildlife system, as they are not game farms, are not fenced, and self-sustaining wildlife populations cannot be managed on these properties.

In particular, the Elephant Norms and Standards require an elephant management plan to be developed for protected areas, registered game farms, private and communal land where elephants occur (Section 6). Such areas are usually fenced, and the landowners of such areas are generally in control of the elephant populations within them. An elephant management plan shows that the area's managers are capable of managing the elephants on the property concerned. Importantly, such elephant management plans provide the basis for trophy hunting applications. Along the Limpopo River, however, many farms occur which are small, unfenced, and not managed for elephant (or even other game), and landowners are not in control of the elephants, which come and go as they please. These landowners are thus unable to submit elephant management plans, and consequently also unable to apply for trophy hunting. These landowners can only apply for the hunting of a roaming problem elephant or DCA, and may not permit a foreign hunter to do so (Elephant Norms and Standards Part 5(8)-(9)). Finally, a drawback of the Elephant Norms and Standards is that they do not effectively cater for elephant movements between South Africa and neighbouring countries. Given the emphasis on elephant management within fenced areas, the Norms and Standards' implications for the elephant population utilising the GMTFCA are less than clear.

The only provincial legislation relevant to this study is the Limpopo Environmental Management Act (Act no. 7 of 2003, May 1, 2004). The Act essentially prohibits the hunting of wild animals without prior authorization, and provides for the classification of game into categories affording different levels of protection. Elephants are listed as a 'specially protected wild animal' (Schedule 2). No provincial ordinances deal with the question of ownership of wild elephant.

At the level of the Mapungubwe National Park and World Heritage Site, the overarching objectives of the applicable Management Plan for 2013-2018 include promoting and fostering international cooperation, preserving biodiversity across international boundaries, protecting the cultural heritage and geographic landscape of the area, and facilitating socio-economic benefits (SANParks 2013). The latter include managing the provision of benefits of the GMTFCA to the region and its people. However, the development of a sustainable elephant offtake quota is not mentioned or implied anywhere amongst the actions to be taken within this context.

5.6.3 Zimbabwe

The Environmental Management Act (2002 (Cap.20:27)) sets out the general framework for environmental matters in Zimbabwe, addressing environmental institutions, planning, standards, and impact assessment. It is complemented by the Parks and Wildlife Act (1975 (Cap. 20:14)), which provides the main legislation for wildlife management. It makes provision for the establishment of six particular protected area types – National Parks, Safari Areas, Recreational Parks, Sanctuaries, Botanical Reserves and Botanical Gardens – describing the purposes for which each can be used. Other legislation allows for the establishment of Game Areas on communal lands (Communal Lands Act, 1982 (Cap. 20:04)).

Uniquely, Zimbabwe has delegated resource use rights, authority and responsibility for wildlife management, including elephants, to the legally authorized land occupants, enabling the latter to manage and derive full benefit from wildlife on their land. In the case of communities, rural development councils (RDCs) are the competent authority. RDCs are, for instance, empowered to adopt bylaws addressing natural resource management. In 1989, Zimbabwe instituted a benefit-sharing program for wildlife, CAMPFIRE (Fischer et al. 2011). The programme focuses especially on communal areas adjacent to PAs, where human-wildlife conflict tends to be most problematic, bringing human-elephant conflicts to the fore. Although no specific management policy or plan for problem elephant management exists, RDCs allocate resources to problem animal management.

A Policy and Plan for Elephant Management in Zimbabwe were adopted by the competent Ministry 1997 and, although not fully implemented, is still in force (Policy and Plan for Elephant Management in Zimbabwe, 1997). The policy acknowledges elephant as an important component of Zimbabwe's wildlife and cultural heritage and aims to conserve elephant at levels which promote biodiversity conservation, while ensuring their sustainable use and their contribution to national development. This combined objective is to be achieved by (i) maintaining at least four demographically and genetically viable populations; (ii) maintaining numbers and densities below levels which would compromise biodiversity; and (iii) maintaining or increasing elephant range at or above the 1996 level (Policy and Plan for Elephant Management in Zimbabwe, 1997). The accompanying management plan sets out associated management actions to give effect to the policy (Policy and Plan for Elephant Management in Zimbabwe, 1997).

5.6.4 Comparison and Appraisal

Elephants are at the centre of some of the more important wildlife and environmental management decisions having to be made within southern Africa. The legal frameworks for doing so in Botswana, South Africa and Zimbabwe have much in common, but some marked differences exist. Below, a comparison is made on several important counts, namely concerning elephant conservation and management objectives; legal status of elephants; their (consumptive) use; monitoring; population connectivity; and transboundary cooperation.

The overall national visions for elephant conservation of the three countries align well, focussing on conserving elephant populations, while ensuring the maintenance of habitats and biodiversity and promoting the contribution of elephants to national development. Yet, the concrete objectives towards achieving this differ. Zimbabwe's Policy and Plan for Elephant Management focusses on maintaining at least four demographically and genetically viable elephant populations, and managing these at specific ecological carrying capacities through periodic population reductions, either through culling or translocations. Botswana's Elephant Plan, which is still in draft format and has not yet been effectively implemented, is more conservative. Whereas concerns regarding the impact of elephants on biodiversity have been raised, the active removal of elephants through, for instance, culling has not been approved. A good example of the different outlooks is the recent trophy hunting ban in Botswana (eNCA August 1, 2015), as compared with the international public outcry against the capture and sale of wild caught elephant calves in Zimbabwe (Cruise 2015).

All three national legal frameworks make provision for ownership of elephants and their non-consumptive and consumptive utilisation. The conditions attached to such use differ, however. In Zimbabwe, elephants are unprotected and their control transferred to landowners and RDCs. In Botswana, elephants are partially protected and can only be utilised under permit. Ownership and control of elephant is claimed by the state. In South Africa, elephant is a protected species. Elephants crossing into South Africa from neighbouring countries are *res nullius*, but ownership can be established through control and constraint. The status of elephant within the GMTFCA will likely be that of *res nullius* with the state as overall custodian.

Policies in all three countries draw on the notion that the survival of elephant within the country is reliant on its economic value to people, especially in light of increasingly conflicting land uses. Indeed, sustainable utilisation of elephant can generate important benefits for local communities and at the same time assist in expanding the conservation estate (Lindsey et al. 2007, Selier and Di Minin 2015). However, where consumptive utilisation is driven purely by economic incentives it can lead to the extirpation of populations and have negative evolutionary consequences (Lindsey et al. 2007, Selier and Di Minin 2015). It is thus important that the goals of utilisation and conservation are in line and that utilisation is sustainable. Long-term monitoring of offtakes and population numbers is essential in this regard (Selier and Di Minin 2015). In terms of transboundary populations, monitoring is furthermore essential to ensure that the management actions of one country do not have negative repercussions across the border. Elephants belonging to the CLRV population are hunted in all three countries. Yet, there is little or no consultation among the three countries to ensure collaborative monitoring or the coordinated setting of elephant hunting quotas, with each country determining their own national quota based on restricted subsets of population data (Selier and Di Minin 2015). Current quotas are set at 14 for Zimbabwe and 33 for Botswana (Selier et al. 2014). As of 2014, however, no trophy hunting is allowed within Botswana (eNCA August 1, 2015). In South Africa, no hunting quota has been set for the CLRV population. Even so, a total of 47 elephants have been shot as DCAs between 2006 and 2014 (Selier et al. 2014). Data on hunting and DCA offtakes within each country are collected, but not used to feedback into a monitoring framework or shared with neighbouring countries. Importantly, the combined offtake must be considered as unsustainable (Selier et al. 2014). Besides, it appears that this offtake could impact adversely on photographic tourism activities in Botswana in future (Selier et al. 2015). There is thus a mismatch between national quota setting and the fine-scale requirements of individual elephant populations (Cumming et al. 2006).

The legal construction whereby ownership of wildlife can be established through fencing has resulted in the development of a very profitable game industry within South Africa and Zimbabwe, but to a lesser extent in Botswana (Cirelli and Morgera 2010). Game fencing, especially in the case of South Africa, has major implications for connectivity between neighbouring elephant populations, and no provision has been made in national legislation to maintain or enhance connectivity between populations (Vanak et al. 2010). In fact, it could indeed be argued that the current legislation incentivises the fragmentation of landscapes, thus hindering connectivity. In Botswana's draft Elephant Plan, however, provision has been made to allow for connectivity between elephant populations and the natural movements of elephants within the country.

Relevant law and policy in all three countries highlight the importance of transboundary cooperation in general, but in the case of the CLRV elephant population little has been done to put collaborative management into practice.

5.7 CONCLUSIONS AND RECOMMENDATIONS

Conservation challenges facing elephants in southern Africa are similar in crucial respects to those facing many large carnivores, not only in Africa but also in Europe and elsewhere (Trouwborst 2015). Successful conservation and management of these species must take into account both the ecological needs of the animals themselves and the social, cultural, economic and political needs of people (de Boer et al. 2013). Balancing biological realism and anthropogenic pragmatism is as important to wolf management in Europe as it is to elephant management in southern Africa. Likewise, international law and policy regarding such controversial species needs to be interpreted and applied across a diversity of local contexts.

The trilateral CLRV elephant population provides a particularly vivid illustration of the related key challenge that is in the spotlight of the present article, namely the fragmentation of the legal landscape. Encouragingly, the above analysis confirms that the need to cooperate in order to manage transboundary wildlife at the level of their populations rather than the level of countries (or other artificial, administrative units) is receiving increasing recognition in governmental and intergovernmental circles.

At the international level, the preceding analysis attests to the existence of a significant body of international law and policy that is of importance for elephant conservation in general, and potentially conducive to transboundary cooperation at the population level in particular. The SADC Protocol on Wildlife Conservation and Law Enforcement and the Southern Africa Regional Elephant Conservation and Management Strategy are cases in point. Moreover, in terms of international legal instruments, the fragmentation of the southern African landscape is only modest. Almost all international treaties discussed count all three countries involved amongst their contracting parties – a notable exception being Botswana in respect of the CMS.

In terms of national law and policy, however, the degree of fragmentation is significant in the case under consideration. Notable differences between Botswana, South Africa and Zimbabwe exist concerning elephant management objectives; elephants' legal status; the hunting and culling of elephants; cross-border monitoring; and measures to ensure connectivity. For instance, a prominent challenge concerning the CLRV elephant population is the absence of a single offtake quota, shared by the three countries, for the transboundary population as a whole.

As regards actual trilateral cooperation at the level of the CLRV elephant population, the need to remedy the aforementioned mismatch and to coordinate management at the transboundary population level has been duly recognised. What is more, the development of the GMTFCA and the associated draft Elephant Management Plan apparently goes beyond what has been done for many other cross-border populations. It is worth highlighting that the comprehensive and detailed approach developed for this trilateral region ticks many of the boxes of the uniform blueprint for transboundary population level cooperation produced in the European large carnivore context, as documented in Table 5.1.

At the same time, population level management of the CLRV elephants is clearly still a work in progress. In particular, the GMTFCA framework yet remains to be fully endorsed and implemented by the relevant authorities in the three countries. For instance, the GMTFCA Treaty still remains to enter into effect and the associated Collaborative Policy and Planning Framework for the Management of Elephants equally still awaits formal endorsement. (Notably, the same is true of several of the relevant national instruments reviewed above). Crucial implementation steps still missing with respect to the CLRV elephant population include coordinated monitoring and coordinated offtake management. It should also be noted that the spatial focus of many of the cooperation efforts in the region is on the GMTFCA rather than the CLRV elephant population. Despite significant overlap, the match between the two is not exact.

In addition to the endorsement and implementation of the aforementioned instruments, it is recommended that a cross-border management authority for the CLRV region be established, consisting of government representatives, at least one scientific expert per country, and other relevant stakeholders, to assist with the coordination and implementation of management actions pertaining to elephants and other cross-border species. Advice by this authority should be mandatory regarding the allocation and sharing of trophy hunting quotas based on scientific monitoring data, coordination of enforcement activities, and the sharing of information between the management authorities and stakeholders (see Table 5.1).

In sum, the trilateral Central Limpopo River Valley elephant population provides an illustration, first, of what a transboundary population level approach to the conservation and sustainable use of wildlife could – or should – look like in practice. The cooperative instruments devised for this cross-border elephant population are exemplary in many respects, as they tick many of the boxes for the aforementioned approach. Lessons learned from the CLRV elephant situation can be applied to the EU carnivore situation and elsewhere. At the same time, however, the remaining shortcomings regarding the implementation of the common management of the CLRV elephant population clearly illustrate the significant challenges involved in achieving a comprehensive, consistent and effective application of a transboundary population level approach.

CHAPTER 6: GENERAL CONCLUSION AND RECOMMENDATIONS

Growing human populations and rural poverty in Africa have led to increasing demands for agricultural land (Krug 2001). While protected areas are fundamental for biodiversity persistence in increasingly human-dominated landscapes (Baeza and Estades 2010, Stokes et al. 2010, Montesino Pouzols et al. 2014), they are often too small to sustain viable populations of large mammals (Graham et al. 2009, Di Minin et al. 2013b, Packer et al. 2013), as they cannot meet the space requirements of wide-ranging or migratory species (Woodroffe and Ginsberg 1998, Graham et al. 2009, Stokes et al. 2010, Di Minin et al. 2013b). Furthermore, the range of many species such as elephant and many large carnivore species often span administrative and political boundaries such as domestic and, in particular, international borders (Delsink et al. 2013, Fattebert et al. 2013, Trouwborst 2015). For the persistence of these species, management on a population level, even where populations are transboundary, is required (Trouwborst 2015, Selier et al. in review). However, even though this makes evident biological sense at a conceptual level, the actual implementation of conservation and management at the transboundary population level is complex, and poses several challenges.

In this thesis, I followed a transdisciplinary approach, using the African elephant (*Loxodonta africana*) population within the Greater Mapungubwe Transfrontier Conservation Area (GMTFCA) as a case study, to assess the challenges and opportunities facing the conservation and management of a wide-ranging cross-border species, such as elephant, in a human-dominated landscape that spans administrative and political boundaries. I used the GMTFCA elephant population because the population, like many others (van Aarde and Jackson 2007, Chase 2009), is transboundary – overlapping the territories of several countries, and ranges beyond designated protected areas (Selier et al. 2014). As such, these populations are exposed to a range of management practices, which prevent them from using the landscape freely (Selier et al. 2015). In addition, they are likely to come into conflict with communities on the edges of protected areas (Hoare 1999, Sitati et al. 2003, Selier et al. 2014). The African elephant is considered a high value species for both consumptive and non-consumptive activities, and attracts high numbers of visitors to the GMTFCA (Evans 2010), and other conservation areas in sub-Saharan Africa, as a flagship species (Di Minin *et al.* 2013a). While the focus of this thesis is the GMTFCA elephant population, the conservation actions aimed at this population will benefit several other wide-ranging species in transboundary areas, as I explain below.

In this chapter, I synthesise the main findings of the thesis, discuss the conservation and management implications of the study, point out some limitations within the study, and make recommendations towards the effective management of cross-border species in human-dominated landscapes.

6.1 SYNTHESIS OF MAIN RESULTS

In this thesis, following a transdisciplinary approach, I assessed the challenges and opportunities facing the conservation and management of a wide-ranging cross-border species, in a human-dominated landscape that spans administrative and political boundaries. In Chapter 2, I found that, where activities such as trophy hunting were not coordinated among administrative bodies, these activities could have a detrimental impact on the persistence of the species, and that these anthropogenic disturbances caused species to avoid areas of high human disturbances. In Chapter 3, I continued to show that human activities, such as trophy hunting, within different management units forced elephant to trade-off between disturbance avoidance, and good food and water availability. In addition, the important predictors of elephant distribution within each of the management units differed from the predictors at the broader landscape, suggesting that at the fine scale, elephant were constraint by factors that may be masked at the broader landscape scale. In Chapter 4, I showed that the per capita gross domestic product (GDP/cap), the proportion of total land surface under agriculture around sites where elephant were present, and vegetation productivity, were important predictors of elephant densities in those countries that have positively contributed to elephant conservation. Finally, in Chapter 5, I showed that a significant body of international law of importance for elephant conservation already exists, but that there was little cooperation among range states, and a lack of implementation of these provisions on a national and trilateral level. There is, thus, clear potential for enhancing the contribution of international law to cross-border species conservation and management. In the following paragraphs I briefly discuss the main results of the thesis.

Many species are exposed to some form of harvesting either through consumptive utilisation such as trophy hunting or illegal poaching, or through the legal control of damage causing animals (Nyhus and Tilson 2004, Lindsey et al. 2007, Anthony et al. 2010). The killing of large mammals for the sport-hunting industry is considered highly controversial for ethical and welfare reasons (Travers 2015) and is often justified on the basis that it contributes to the conservation of endangered species, decreasing human-wildlife conflict and/or generating funding for local stakeholders (Lindsey et al. 2006, Lindsey et al. 2007, Buckley and Mossaz 2015). In Chapter 2, I examined the effects of trophy hunting on the population dynamics and movements of the GMTFCA elephant population. Few studies have investigated the potential social and demographic effects of hunting adult bulls (Archie et al. 2008, Burke et al. 2008). In these studies, only the fine-scale genetic implications and stress levels deriving from illegal and legal hunting were studied, and are thus not directly applicable when attempting to evaluate the merits of hunting cross-border populations where malpractices in one country could negatively impact on the population as well as on economic ventures relying on the presence of these species. In Chapter 2, instead, I assessed the effects of hunting on the population dynamics and movements of a cross-border elephant population using data from the GMTFCA.

My results indicated that the current hunting quotas within the GMTFCA are unsustainable, and, based on the outcomes of a population viability analysis, a trophy hunting quota of maximum 10 trophy bulls/annum was recommended (Selier et al. 2014). I further showed that, although bulls did not completely avoid areas with greater hunting pressure, fewer bulls entered these areas than areas where less or no hunting occurred (Selier et al. 2014). Thus, sustained high levels of hunting in a region, for example to deter crop raiding by bulls, do not appear to cause bulls to avoid that region. Thus, shooting trophy bulls does not alleviate the problem of human-elephant conflict and cannot be used as a tool to deter bulls from entering community areas (Selier et al. 2014). However, high levels of hunting of bulls caused a disturbance effect within breeding herds, possibly because of the high stress levels observed throughout the population during hunting disturbances (Burke et al. 2008). In a system where both consumptive and non-consumptive use is made of elephant, disturbance may have major ramifications by possibly altering the distribution and behaviour of elephant within the landscape.

Next, I tried to understand how the disturbance effect of anthropogenic activities can affect the distribution of elephant at different spatial scales. Specifically, I assessed the effect of environmental and anthropogenic factors at different landscape and management scales in predicting the distribution of elephant within the GMTFCA. My results suggested that accounting for anthropogenic disturbance is important in determining the distribution of large, wide-ranging, mammal species, forcing such species to trade-off between disturbance avoidance and the availability of resources i.e. food and water availability. This study supports the idea that localised human activities strongly affect the distribution of wide-ranging species in human-dominated landscapes (Baeza and Estades 2010, Erb et al. 2012). Overall, my results suggest that factors such as human disturbance need to be taken into account when modelling the distribution of large mammal species in increasingly human-dominated landscapes, and that ecological modelling needs to be done at the spatial scales at which conservation decisions are made (Erb et al. 2012), as a mismatch may have important implications for conservation planning and species persistence (Selier et al. 2015). Particularly, these results have important implications for the effective allocation of conservation actions that can enhance the long-term persistence of wide-ranging species in human-dominated landscapes.

In Chapter 4, I assessed the effect of socioeconomic factors on the abundance of elephant within a cross-border landscape. The wide-ranging nature of elephant and the fact that they range beyond designated protected areas, are likely to bring them into conflict with communities on the edges of protected areas (Hoare 1999, Sitati et al. 2003). Furthermore, southern Africa represents the stronghold of elephant conservation (Blanc et al. 2007), and studies are needed that assess threats at the site and country level in order to prevent conflict for limited space and resources with humans. I found that elephant density was higher in countries where the gross domestic product was higher, in areas where the proportion of total land surface under agriculture was the lowest; and sites where vegetation productivity was the highest. My results confirm that poverty is an important factor affecting elephant distribution at the country level, but highlight that, at a local scale, human disturbance and food availability plays an important role. Overall, the results suggested that accounting for socioeconomic factors, and at the correct spatial scale, is important in determining the abundance of a large, wide-ranging, mammal species such as elephant.

Finally in Chapter 5, I analysed the applicable global, regional, trilateral and national laws and policies as they pertain to Botswana, South Africa and Zimbabwe. I evaluate the degree to which the transboundary population approach is supported by respective laws and policies and the level to which this approach has been incorporated in these laws and policies. My results indicated that a significant body of international law of importance for elephant conservation already exists, and that advances and provisions have been made within international agreements, and to some degree on a trilateral and national level, for the management of transboundary elephant on a population level. There is, however, a clear potential for enhancing the contribution of international law to cross-border elephant conservation and management. At present there is a mismatch between national legislation and regional and international agreements, and among the national legislation of the individual countries. The inappropriate trophy hunting quota setting for elephant is a consequence of inappropriate legislation and policy and the lack of cooperation between range countries. The resultant fragmentation of the southern African legal landscape in respect of elephant thus requires the formalisation of cooperation between range countries in a manner that takes into account differing legislation or possibly passing new legislation more appropriate for TFCAs.

6.2 CONSERVATION AND MANAGEMENT IMPLICATIONS

Trophy hunting can create incentives for wildlife and habitat protection under diversity of scenarios and over vast areas which may be unsuitable for other forms of wildlife-based land uses such as photographic tourism or within areas of political instability (Lindsey et al. 2006, Jorge et al. 2013, Buckley and Mossaz 2015). For these reasons trophy hunting can be considered an important conservation tool in Africa (Lindsey et al. 2007, Jorge et al. 2013, Buckley and Mossaz 2015). If the profits realised from harvesting a few individuals are sufficient incentive for people to tolerate the larger population, the goals of trophy hunting and conservation are compatible (Treves and Karanth 2003, Balme et al. 2010a). However, where consumptive utilisation is solely driven by economic gain and not by conservation objectives, it can lead to negative impacts on species (Lindsey 2007, Fa and Brown 2009, Selier et al. 2014). In this thesis, I have shown that the current hunting quotas for elephant within the GMTFCA are unsustainable and an urgent revision of the hunting quotas is required within each country

(Selier et al. 2014). The allocation of hunting quotas should be based on actual data, taking into consideration the environmental conditions as well as the population dynamics and social structure of the species under consideration, and be allocated on a population level, i.e. across the three countries and different land uses (Selier et al. 2014). Based on these results, and taking into consideration the social stability of the population and the current environmental conditions, the maximum sustainable yield is 10 bulls >35 years old per annum (Selier et al. 2014). Trophy hunting targets the largest males or those with impressive horns, tusks, or antlers (Ginsberg and Milner-Gulland 1994, Milner et al. 2007). Even though it is generally restricted to a few individuals, where controls are lacking, a high proportion of those individuals that qualify can be removed annually (Coltman et al. 2003, Crosmary et al. 2013). Selective harvesting may also have negative evolutionary consequences (Balme et al. 2010a, Selier et al. 2014). Continuous selection for large-tusked bulls will lead to the extirpation of those individuals in a population, and ultimately could depress the quality of trophies. There is some evidence of trends towards smaller tusks in southern Africa due to trophy hunting, with concern for a temporal shift in heritable traits such as tusk size (Nuzzo and Traill 2013). These large-tusked bulls have a high photographic value and the extirpation of these individuals within in a population could impact photographic tourism operations. It is thus suggested that a trophy hunting protocol, that restricts the number of large-tusked and potentially large-tusked bulls that can be hunted in a population, be developed to ensure the persistence of large-tusked bulls that are important for photographic tourism in the population. It is further suggested that a single multi-jurisdictional (cross-border) management authority regulating the hunting of elephant and other cross-border species be established. This would likely require changes to current legislation and or policy within range countries or the passing of new legislation more appropriate for TFCAs (Chapter 5). A conservation planning assessment with the objective of enhancing biodiversity protection (Di Minin and Toivonen 2015), while promoting sustainable development and improved quality of life for communities within the GMTFCA, is urgently required, and should include all stakeholders (Fischer et al. 2011, Selier et al. 2014, Naidoo et al. 2015).

In the following chapter, I further explored the disturbance effects anthropogenic activities may have on elephant distribution. In this study I was able to show that anthropogenic activities, such as trophy hunting, can influence the distribution of species at both the landscape and finer management scales, causing animals to concentrate in areas with low anthropogenic disturbances (Selier et al. 2015). These changes can have significant impacts on ecosystem structure, potentially resulting in cascading effects such as the loss of large trees (Shannon et al. 2011). More importantly, I demonstrated that when management decisions are not made at the appropriate decision level for the species or system under consideration, a mismatch in spatial scale occurs with implications for the effective management and conservation of the species (Cumming et al. 2006). This has far reaching implications for cross-border species, where the stress effects of hunting could be transmitted to photographic tourism areas within neighbouring countries (Delsink et al. 2013, Selier et al. 2015). These disturbance effects may result in behavioural changes such as elephant moving away from these areas and aggregate in protected areas (van Aarde et al. 1999, Selier et al. 2015), congregating in large groups, shifting to drink more at night (Martin et al. 1996) or becoming more aggressive (Whyte 2001). These behavioural changes could lead to an alteration of the quality of photographic tourism experiences (Burke et al. 2008). The mismatch of spatial scales at which management occurs has implications for highly-managed threatened species where management actions might affect the persistence of a species. This could either be as a result of the species utilising lower quality resources due to disturbance effects or through an increase in human-wildlife conflict (Ciuti et al. 2012).

The effective protection and conservation of high value species does not solely rely on increasing the size of protected areas or improving ecological conditions, but through considering the local socioeconomic conditions within the area (Adams et al. 2004, Burn et al. 2011, de Boer et al. 2013). My study showed that factors such as GDP/cap, poverty and the encroachment of humans on protected areas, are important factors predicting the abundance of elephant in those countries that have positively contributed to elephant conservation (Blanc et al. 2007, de Boer et al. 2013). The GDP/cap is a reliable indicator of a country's investment in protected area management (Wittemyer et al. 2008, Burn et al. 2011) and is positively correlated with human welfare which in turn influences attitudes towards conservation (Burn et al. 2011). Countries with higher GDP/cap have higher investments in the management of protected areas, and have lower levels of corruption (Nyhus and Tilson 2004, Burn et al. 2011). Burn *et al.* (2011) showed that poor governance is an important driver in the illegal killing of elephants. Furthermore, a model developed by Smith *et al.* (2003) predicted that countries with a governance score of less than 3.1 would show population declines. My study has indicated that, where populations are transboundary, high corruption levels and poor governance in one range state (i.e. Zimbabwe) has implications for neighbouring range countries sharing populations, and could negatively impact those populations.

Landscape fragmentation goes hand in hand with an increase in human densities (Baillie et al. 2004, Foley et al. 2005). Our study showed that elephant densities were negatively correlated with an increase in in agricultural land and human densities. Human densities, especially on the edges of protected areas, will continue to increase (Foley et al. 2005, Wittemyer et al. 2008), while it is likely that state budgets will continue to be constrained in trying to balance social demands and the demands of protected areas (Krug 2001, DeFries et al. 2007). Increasing human densities, and the resulting higher demand for land and natural resources, are expected to increase human-wildlife conflict, especially on the edges of protected areas (Wittemyer et al. 2008, Packer et al. 2013) and frequently have significant, negative impacts on biodiversity, such as illegal timber and mineral extraction (Curran et al. 2004), bushmeat hunting (Brashares et al. 2004, Gandiwa et al. 2013), fire frequency (Kodandapani et al. 2004), and increased fragmentation of the landscape restricting the movement of wide-ranging species (Woodroffe and Ginsberg 1998, Di Minin et al. 2013b, Selier et al. 2015). Increased human densities on the edges of protected areas may lead to species extinctions (Brashares et al. 2001) such as the high extinction rates of carnivores in reserves across the world as a result of human induced mortalities (Woodroffe and Ginsberg 1998, Brashares et al. 2001, Packer et al. 2013).

My study emphasised that, in those countries that are already making a positive contribution in Africa to the conservation and protection of elephant, the country's investment in protected area management, increasing human and livestock densities and the encroachment of humans on protected areas, were important factors to mediate for the continued persistence of elephant and other wide-ranging species. Effective protection of source populations in a well-connected system of protected areas that buffers them from anthropogenic threats, thus remains a key action to ensure the future persistence of these species, in the developing world (Di Minin et al. 2013b, Di Minin and Toivonen 2015). The inclusion of multi-use zones around protected areas in land use planning will further assist in the buffering of protected areas against anthropogenic activities (Wittemyer et al. 2008, Di Minin and Toivonen 2015), and provide for economic incentives through the sustainable utilisation of natural resources in these areas (Naidoo et al. 2011, Child et al. 2012), and may reduce human-wildlife conflict on the edges of protected areas (Wittemyer et al. 2008). If we change the ways that we govern private conservation areas, and if we can place landholders and rural communities at the junction of benefit and management through a combination of co-ownership, co-management, and policy changes (Child and Chitsike 2000), wild resources can be profitable and simultaneously address rural poverty and environmental injustice (Bookbinder et al. 1998, Child et al. 2012).

In order to understand the consequences of management activities such as trophy hunting and to implement an adaptive quota system based on population trends, long-term monitoring is essential (Selier and Di Minin 2015). Where an effective monitoring system with clear objectives is in place, consumptive utilisation is defendable (Selier and Di Minin 2015). The cost of monitoring, long-term commitment and planning are often highlighted as constraints to the effective implementation thereof (Western et al. 2009). However great advances have been made in developing more cost-effective methods to monitor offtakes and population trends (Caro 2011, Bunnefeld et al. 2013), and where there is an economic benefit through consumptive utilisation a portion of the revenue should feedback into a monitoring programme to ensure sustainability in the long term (Selier and Di Minin 2015). However, failure to address social issues, such as inequitable distribution of hunting revenues and the involvement of communities, can undermine the success of hunting operations (Selier and Di Minin 2015). Illegal harvesting of species reduces the number of animals available for trophy hunting (Balme et al. 2009, Chiyo et al. 2015) and impacts not only the survival of species but also impacts on the revenue awarded to communities who bear important costs from conservation (Hutton and Leader-Williams 2003, Jorge et al. 2013, Challander and MacMillan 2014). The monitoring of cross-border populations is, however, compounded by the legal landscape in which the species occur (Selier et al. in review), and is reliant on effective cooperation between range states and buy in from politicians within these countries (Di Minin and Toivonen 2015, Selier et al. in review).

Where species ranges cross jurisdictional and administrative boundaries, especially international boundaries, cooperation and joint management of these species on a population level is essential (Chapron et al. 2014, Trouwborst 2015). This requires the development of coordinated legislation and policies to improve land use planning (Chapron et al. 2014, Montesino Pouzols et al. 2014, Trouwborst 2015). At present, the southern African legal landscape in respect of elephant is fragmented and adds to the need for transboundary cooperation at the level of the biological unit of the population involved (Selier et al. in review). There is a mismatch between national legislation and regional and international agreements and among the national legislation of the individual countries. Specific legal regimes applicable to elephant vary among the three countries, and the situation is compounded further by the fact that all three countries are not party to all international treaties (Selier et al. in review). On an international, regional and national level elephant are considered as a natural resource of great economic potential (Blignaut et al. 2008). It is all three countries' formal position that the survival of elephant within the country is reliant on its economic value to people, especially in light of increased conflicting land uses (Selier et al. in review). To this end, the Southern African Development Community protocol (SADC), the Southern African Regional Elephant Conservation and Management Strategy, and national legislation in all three countries, supports community-based natural resource management (CBNRM) programmes such as the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe.

Several instruments exist that govern cross-border cooperation in the conservation and sustainable use of natural resources (Cirelli and Morgera 2010, Trouwborst 2015, Selier et al. in review). However, there seems to be little cooperation between southern African countries to cooperatively manage and utilise these cross-border high value species such as the African elephant on a political level that is necessary for the effective management of these species (Selier et al. in review). Montesino Pousols *et al*. (2014) demonstrated that, if coordinated international action is not taken quickly in the development of coordinated legislation and policies to improve land use planning, further biodiversity loss is unavoidable. Thus, to maximise conservation benefits and to alleviate the impacts of future human growth and land use changes on wildlife, action should be taken quickly. With growing human population numbers, the pressures on the current conservation estate are likely to increase and with that the demand for natural resources. Southern Africa, at present, is considered a safe haven for elephant, considering that 55% of the continents elephant occur in this part of the continent (Blanc et al. 2007). With the demand for ivory increasing and the number of elephant decreasing elsewhere on the continent the pressure from illegal ivory poaching on the southern African populations are likely to increase significantly (CITES et al. 2013). It is therefore important for range states to collaborate in the protection of high value, wide ranging species and coordinate management actions. Since communal lands comprise a large fraction of rural Africa (up to 500% more than state-managed forest reserves and national parks) (Alden Wily 2011), economic incentives to communities that promote or at a minimum tolerate living with wildlife will be an important solution to promote participation of local communities in biodiversity conservation efforts and improved enforcement (Child et al. 2012, Ihwagi et al. 2015, Naidoo et al. 2015). Enhanced enforcement combined with effective engagement with local communities, for example, has enabled Nepal to not lose a single rhinoceros to poaching in 2011 and 2013 (Emslie 2013). Further cooperation between countries, increased landscape connectivity, and the ability to generate income from tourism have been shown to work successfully in increasing wildlife numbers elsewhere (Plumptre et al. 2007).

Several studies (Max-Neef 2005, Hadorn et al. 2006, De Vleeschouwer and Raboy 2013) have linked the current gap between science and action to 1) the complexity of the problems themselves; 2) the compartmentalisation of knowledge and management sectors and 3) the failure to ensure effective collaboration between scientists, managers, decision-makers and other stakeholders. The value of integrating a transdisciplinary approach into research projects is that it increases the ability of projects to solve problems in the real world (Krott 2002) and bridge the current gap between science and action (Reyers et al. 2010). Here, I used a transdisciplinary approach to address some of the challenges facing the conservation and management of a wide-ranging cross-border species in a human-dominated landscape that spans administrative and political boundaries and made recommendations as to how to bridge the current gap between science and action as it pertains to the GMTFCA elephant population. I have shown that the persistence of high-value, wide-ranging species are not only dependent on their ecological needs, but influenced by people and their actions and decisions. This required an understanding of the socioeconomic factors driving decision-making that could influence the persistence of these species in the landscape.

Daily and Ehrlich (1999) argued that failing to see conservation problems as multidimensional, requiring inputs from multiple disciplines, will lead to poor decisions and policies with serious negative consequences for biodiversity and human wellbeing. In this thesis, I have shown that the current unsustainable trophy hunting offtakes of elephant are a direct consequence of inappropriate legislation and policy and the lack of cooperation between range countries. However, the banning of trophy hunting may not be the best solution and could lead to further habitat fragmentation and biodiversity loss (Smith et al. 2003, Di Minin et al. 2016). Economic incentives, such as the development of sustainable trophy hunting quotas and the equitable sharing of benefits from tourism with local communities, could be an important solution to promote participation of local communities in biodiversity conservation efforts and improved enforcement (Child et al. 2012, Naidoo et al. 2015, Di Minin et al. 2016). Further the lack of coordinated land use planning between range states may lead to increased human densities on the edges of protected areas leading to further fragmentation of the landscape, increased human-wildlife conflict and ultimately the extirpation of these species from the landscape. This not only have implications for biodiversity in general, but also socioeconomic implications for tourism operations relying on the presence of these species. The implementation of these conservation plans will require close involvement from the implementing agencies and other stakeholders, the integration of biological analyses with research on the economic and social consequences of conservation and on the institutional landscape for implementation (Pierce et al. 2005). Finally, poor governance in one or more range country could further constrain the effective implementation of conservation management plans. The development and implementation of policies that reduce the effect of corruption is thus urgently required (Smith et al. 2003).

6.3 LIMITATIONS WITHIN THE STUDY

A potential limitation of this study was the seasonality of surveys which were limited to the end of the dry season and only represents a set moment in time. Long term movement data of elephant groups throughout the study area could further inform this study. The limited number of observations of elephant within hunting areas is a further limitation. While the limited number of observations of elephant within the management unit might be a reflection of the landscape scale decisions made by elephant on where to be within the broader landscape (Selier et al. 2014), they might not reflect effectively the fine-scale selections made by elephant. Future studies should focus more on hunting areas to better understand how different hunting intensities can influence elephant distribution (Slotow et al. 2008, Burton et al. 2012, Selier et al. 2014). A business model that compares the economics and social benefits of trophy hunting to that of photographic tourism would be informative and may assist management authorities in developing a benefit sharing model for the Transfrontier Park (Selier et al. 2014). More specifically, such a model could compare the management costs per hectare for each management unit, the income generated through different forms of tourism practices including high density photographic tourism, low density photographic tourism and trophy hunting. Finally a stakeholder questionnaire survey addressing the expectations and fears of the various groups involved in the Transfrontier Park and could further assist authorities in designing management strategies to address stakeholder expectations and fears.

Data on socioeconomic variables used in this analysis is currently only available at a country level. We recognise that the scale at which the human variables measured may influence our results (Wittemyer 2011). We thus suggest that future studies, should consider including these variables at a site level where available. Our study further showed that lower densities of elephant occurred in state-owned protected areas compared to private game reserves. However, as the budgets and an evaluation of the management efficiency of the various sites were not available, the reasons for the higher densities of elephant on private land could not be tested. It is thus suggested that future studies consider these variables, to further explore the mechanisms that result in increased animal densities on private land compared to state-owned land.

6.4 CONCLUSION

Understanding factors that affect the persistence of charismatic megafauna in human-dominated landscapes is crucial to inform conservation decision-making and reduce human-wildlife conflict. In my study, I have demonstrated that the persistence of high value wide-ranging species are not solely depended on ecological factors, but that factors such as a country’s investment in protected areas, poverty, the encroachment of humans and activities such as trophy hunting and the legal landscape can influence the long term persistence of these species.

The effective management of these species on a population level (Linnell et al. 2008, Trouwborst 2015) thus require collaboration between range countries, the development of coordinated legislation and policies to improve land use planning (Chapron et al. 2014, Montesino Pouzols et al. 2014, Trouwborst 2015), the development of multi-use zones around protected areas (Wittemyer et al. 2008), and conservation corridors to link current protected areas between range countries (van Aarde and Jackson 2007). It is further important that the goals of trophy hunting and conservation are compatible, and that an adaptive framework is used to ensure sustainability (Selier and Di Minin 2015). This can only be achieved through the development of a single offtake quota, strict regulations and long-term monitoring of offtakes and population numbers. It is further suggested that a single multi-jurisdictional (cross-border) management authority regulating the management and hunting of elephant and other cross-border species be established. The necessary legislation and policy to enable this has to be created, and tasked to a specific agency, such as the multi-jurisdictional management authority, to give it teeth. A conservation planning assessment with the objective of enhancing biodiversity protection, while promoting sustainable development and improved quality of life for communities within the GMTFCA, is urgently required and should include all stakeholders.

Immediate action is needed in countries that are already making a positive contribution in Africa to the conservation and protection of elephant to prevent future human population increases and activities from negatively impacting on the persistence of elephant in these countries. Effective protection of source populations in a well-connected system of protected areas that buffers them from anthropogenic threats remains the key action to ensure the future persistence of wide-ranging species, such as elephant, in the developing world (Di Minin et al. 2013b, Di Minin and Toivonen 2015). However, poor governance in range countries, such as Zimbabwe, could play a considerable role in determining the success of national and trilateral strategies to conserve high-value, wide-ranging species (Smith et al. 2003). The development and implementation of policies that reduce the effect of corruption is thus urgently required (Smith et al. 2003).

CHAPTER 7: LITERATURE CITED

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