Competition between people and elephants in the Okavango Delta Panhandle, Botswana

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Declaration of Originality

This thesis is the result of my own work and all else is appropriately referenced. No part of this thesis has been submitted for a degree or diploma or other qualification at any other university. The text does not exceed 100,000 words.

Abstract

The general objective of this study was to gain a greater understanding of the complexities of the competition between people and elephants, focusing on elements that can be investigated in the short term and could aid in devising effective mitigation and management strategies. Specifically, I aimed to a) determine the current elephant population numbers and growth rate in the study area and investigate how reliable aerial survey estimates are; b) monitor the extent of human-elephant conflict (HEC) incidents and compare community based monitoring techniques to a top-down government approach; c) determine key drivers of elephant crop-raiding and explore how spatial autocorrelation affects such data; d) investigate how elephant movements are affected by human habitat modifications, and; e) investigate rural farmers' attitudes towards elephants and compare perceived human-elephant conflict to actual measurable levels of elephant crop damage. My findings show that combinations of social and ecological factors are involved in shaping competition between people and elephants. A multi-disciplinary approach to investigations is, therefore, needed to fully understand such competition and resulting conflicts. Contributory factors to HEC identified in this study include: actual and perceived conflict levels; farmer vulnerability to risk and available coping strategies; susceptibility of crops to elephant foraging, which affects both actual and perceived conflict levels; methods used to measure damage; natural and modified behaviour of people and elephants affecting resource and spatial use as well as how each species reacts to living in close proximity to each other; and human feelings and perception towards elephants and the situation, which are influenced by an array of socio-economic factors. To be successful, effective conflict resolution and management strategies will, therefore, require consideration of short and long term dynamics, as well as a combination of mitigation approaches that consider all elements affecting conflict extent.

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Abbreviations

AFESG - African Elephant Specialist Group
AIC – Akaike Information Criterion
ANOVA – Analysis of Variance
CAMPFIRE – Communal Areas Management Programme for Indigenous Resources
CHA – Controlled Hunting Area
CI – Confidence Interval
CSO – Central Statistics Office
DWNP – Department of Wildlife and National Parks
GIS – Geographical Information System
GLM – Generalised Linear Model
GLMM – Generalised Linear Mixed Effect Model
GPS – Global Positioning Unit
HACIS – Human Animal Conflict Insurance Scheme
HEC – Human-Elephant Conflict
HECTF – Human-Elephant Conflict Task Force
HECWG – Human-Elephant Conflict Working Group
HWC – Human-Wildlife Conflict
IRDNC – Integrated Rural Development and Nature Conservation
IUCN – International Union for the Conservation of Nature
KAZA TFCA – Kavango Zambezi Transfrontier Conservation Area
MET – Ministry of Environment and Tourism (Namibia)
MEWT - Ministry of Environment Wildlife and Tourism (Botswana)
ML – Maximum Likelihood
NGO - Non-governmental Organisation
PAC – Problem Animal Control
PCL – Perceived Conflict Level
RMSE – Root Mean Square Error
SADC – Southern African Development Community
SBSNR - Samburu and Buffalo Springs National Reserve
WAC – Wildlife Acceptance Capacity
WMA – Wildlife Management Area

Introduction

Background

This thesis investigates the complexity of the competition arising between humans and elephants. Elephants and humans require similar resources (food, water and space) and, therefore, in areas where the two species live in close proximity, competition often arises and conflicts can occur. Throughout the range of African (*Loxodonta africana sp.*) and Asian (*Elephas maximus*) elephants, conflict between people and elephants appears to be escalating (Blanc *et al.* 2007; Choudhury *et al.* 2008) for a variety of reasons, which are complex, hard to quantify/determine and very often difficult to resolve (Osborn 1998a). The purpose of this study was to examine the ecological and social aspects of human-elephant conflicts (HEC) in the Okavango Delta panhandle, a HEC 'hotspot' in northern Botswana, in order to gain a greater understanding of this important conservation issue and provide relevant stakeholders with information that will facilitate future management of elephants and HEC in the region.

Elephant range appears to be expanding in Botswana, with elephants frequently occurring in areas inhabited by humans (Chase & Griffin 2007). In addition to this elephant range expansion, the human population in Botswana is also increasing (CSO 2001) and more land is being converted for agricultural uses each year (CSO 2006). The consequent spatial overlap between people and elephants has given rise to incidents of competition between people and elephants, which are thought to be increasing (Natural Resources & People 2006b). Human-elephant conflict is now one of the most serious and challenging wildlife management and conservation issues in Botswana. As such, HEC research and mitigation have been highlighted as important activities by the Government of Botswana in the country's Elephant Management Plan (MEWT 2000) and essential for the successful implementation of regional elephant management strategies such as the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA) (Hanks 2006).

In 2006, the Okavango Delta Management Plan identified HEC 'hotspot' areas in Botswana where future HEC research and mitigation efforts should be concentrated (Natural Resources & People 2006b). One major 'hotspot' identified was the eastern Okavango Delta panhandle in the north west of Botswana, an area on the periphery of the proposed KAZA TFCA boundary. Here, over 15,000 people (CSO 2001) are living along the banks of the river utilising the land for floodplain and dry land subsistence farming, with elephants ranging throughout the area (Jackson *et al.* 2008). It is thought that elephant movements leaving the area may be restricted due to the Namibian border fence to the north and the northern buffalo veterinary fences to the east and south of the area (Jackson *et al.* 2008; Chase & Griffin 2009; Loarie, van Aarde & Pimm 2009a), causing concern of possible increasing elephant numbers and consequent exacerbation of HEC incidents. Evidently, a greater understanding of the patterns and underlying processes of HEC is needed in the Okavango Delta panhandle in order to determine whether HEC is indeed increasing and, if so, what factors are contributing to the severity of the problem here, and ultimately help predict where future incidents may occur so that management strategies can be devised to reduce the problem.

Competition between humans and wildlife is complex, involving contributing factors from an array of disciplines such as ecology, sociology and economics (Hoare 2001; Treves *et al.* 2006; Warren, Buba & Ross 2007). Human-Wildlife Conflict (HWC) situations are also always changing as wildlife and human populations increase, more land is converted for agricultural and residential use, politics change and, even in the broadest sense, climate change affects the ecology of the system and consequently the movements and resource use of wildlife and people. In order to fully understand such a dynamic system it is, therefore, necessary to consider HWC in both the short and long term, using an interdisciplinary approach (Treves *et al.* 2006).

This project has examined the dynamic interaction between elephants and people in the Okavango Delta panhandle over a ‘snap shot’ time period of this long existing relationship. To truly predict what might happen in this area in the future, we would need an elephant based model that would draw on all factors affecting elephant movement, habitat use and behaviour, which ultimately drives the dynamics of any HEC situation. However, such a model requires data, which was unavailable during this study. Therefore, my research focused on factors that could be addressed in the short term and could aid in devising effective mitigation and management strategies, such as developing a crop vulnerability model, offering improvements to crop-damage monitoring techniques and evaluating farmer perceptions and concerns and identifying how best to account for and improve these perceptions during mitigation strategy implementation. Localised elephant movements were also monitored to establish the effect of human habitat modifications on such movements and aerial surveys revealed the current status of the elephant population in the area. I used a multidisciplinary approach to my investigation, addressing both ecological and social issues and, thereby, accounting for the complexities of the situation in my explorations.

Objectives of this study

The general objective of this study was to gain a greater understanding of the complexities of the competition between people and elephants, specifically focusing on elements that can be investigated in the short term and could aid in devising effective mitigation and management strategies.

The specific goals of the project were:

i) to determine the current elephant population numbers and growth rate in the study area and investigate how reliable aerial survey estimates are;

ii) to monitor the extent of HEC incidents and compare community based HWC monitoring techniques to a top-down approach currently used by the Government of Botswana;

iii) to determine key drivers of elephant crop-raiding and explore how spatial autocorrelation affects wildlife crop-raiding data;

iv) to investigate how elephant movements are affected by human habitat modifications, and;

v) to investigate how rural farmers perceive elephants and the competition they are experiencing with them, and compare perceived conflict to actual measurable levels of crop-raiding.

Structure and approach of this thesis

This thesis is written as a series of papers. The first chapter delivers a background to the complexity of human-wildlife conflict issues, highlighting ecological and social factors warranting consideration, while focusing on negative interactions between humans and elephants. Specifically, the aim of this chapter is to a) provide background information to the topic of human-wildlife conflict, elephants and human-elephant conflict scenarios, and; b) review literature on previous studies to illustrate the relevance of topics being investigated in the consequent chapters of this thesis.

Chapter two derives estimates of abundance for the elephant population in the study area using aerial transect survey data. Computer simulation techniques are then used to explore how reliable these estimates are. The aims of this chapter were to a) estimate current population numbers and densities of elephant in the panhandle region; b) determine the elephant population growth rate in the area; c) use simulation to explore the reliability of past and present survey results; and d) explore the effect of spatial scale in the precision of aerial surveys.

The third chapter addresses the monitoring aspect of HEC incidents, because this gives us our understanding of the current extent of HEC in the area. I established a community enumerator monitoring programme for HEC incidents over a three year period in 12 villages along the eastern side of the Okavango Delta panhandle, following the International Union for the Conservation of Nature (IUCN) data collection protocol. Results using this community based HEC monitoring system are compared with data collated from the Government of Botswana Problem Animal Control (PAC) monitoring system. This comparison gives valuable insight into the two systems, highlighting advantages and disadvantages of both. Specifically, this chapter aims to a) record the current status (frequency and extent) of HEC; b) quantify crop-damage by elephants; c) assess temporal patterns of HEC; and d) compare insights from the IUCN protocol and a government approach, not only in data collection efficiency, but also to explore the implications of the effectiveness and impact each has on HEC mitigation and elephant conservation.

In chapter four, I collected data on attributes and position of fields, crop types grown, farmer characteristics, and mitigation measures used, in an attempt to elucidate key drivers of crop raiding by elephants in Botswana using generalised linear models. However, because elephant crop-raiding is a spatial phenomenon I realised that spatial autocorrelation was present in my data, which could lead to spurious results. I therefore, explore the effect of spatial autocorrelation on wildlife crop-raiding data in more detail and make recommendations on how to minimise the effect of this whilst retaining valuable biological information in results. The objectives of this chapter were to a) examine whether raided fields were distributed randomly in the study area, b) identify characteristics of elephant crop raiding in the study area, c) investigate factors affecting the susceptibility of a field to elephant raiding, and d) determine the effect of spatial autocorrelation on HEC data.

Chapter five investigates patterns in localised elephant movements, specifically investigating a) what human infrastructure influences elephant distribution and b) how human habitat modifications affect the frequency of elephant movement and path use. Human distribution and settlements, as well as elephant distribution and movement pathways are mapped using GIS techniques and generalised linear mixed effect models are used to explore what environmental and social factors affect elephant movements.

The sixth chapter addresses the social aspect of HWC by exploring social drivers of HEC. An investigation into how HEC influences human perception and attitudes of rural farmers towards elephants, conservation initiatives and HEC mitigation strategies is conducted using generalised linear models and ordered probit model analysis. Four main questions are addressed: a) What social factors affect attitudes towards wildlife and elephants? b) How severe is the perceived level of conflict with elephants? c) What are the varying coping strategies of people experiencing negative interactions with elephants? and d) do the views of the farming community indicate possibilities for coexistence? Widespread structured interviews with rural subsistence farmers who had either been raided or not-raided within the year of questioning were used to gain an overview of opinions from farmers directly and indirectly affected by living close to elephants.

The final chapter brings together findings from the previous five chapters and discusses possible strategies to address HEC, thereby providing solutions and promoting the future harmonious coexistence of elephants and people in the Okavango Delta panhandle and other similar HEC hotspots.

Chapter 1

Overview

This chapter provides an overview of the current literature concerned with issues surrounding the competition between humans and wildlife, specifically focusing on the negative interactions or conflicts that occur as a result of such competition. It highlights ecological and social factors warranting consideration when investigating human-wildlife conflict (HWC) scenarios, with particular reference to situations involving humans and elephants. Relevant details of elephant biology and ecology are, therefore, provided to facilitate an understanding of the elephant perspective in such human-elephant conflict (HEC) scenarios. Competition for space is a major element of competition between people and elephants, thus, a review of land use planning options is given. In order to devise appropriate management strategies for human-wildlife conflict issues an understanding of spatial and temporal patterns of HWC incidents is needed which ultimately requires effective monitoring of such incidents. A summary of HWC quantifying techniques is, therefore, provided along with a review of findings from previous studies investigating spatial and temporal patterns of HWC incidents. The influence of peoples' attitudes and perceptions adds greater complexity to HWC situations. As a result, a background to the human element of HWC is presented. In conclusion, a brief description of mitigation and management strategies available for addressing HWC is given, to highlight possibilities for co-existence between people and wildlife in the future. In summary, the aim of this chapter is to a) provide background information to the topic of human-wildlife conflict, elephants and human-elephant conflict scenarios, and b) review literature on previous studies to illustrate the relevance of topics being investigated in the consequent chapters of this thesis.

1.1. Defining the ecological relationship between humans and elephants

Where elephants and people co-exist, humans can destroy the natural forage resources of elephants and aggressive interactions between the two species over access to certain resources (i.e. crops, natural forage, water and space) often occur. In addition, elephant densities are often much lower in areas where human densities are high and vice versa, as both people and elephants have been known to emigrate from an area when population numbers of the other species increase. This section provides evidence and historical observations that indicate competition between humans and elephants over resources such as water, space, vegetation and crops does occur and often results in conflict.

1.1.1. Competition

The distribution of species in an ecosystem is largely determined by their adaptations to abiotic environmental factors as well as by biotic interactions with other organisms in their immediate vicinity. Four types of interactions between individuals or populations within the same species or different species have been identified, namely predation (including parasitism), competition, commensalism, and mutualism. Competition is defined as:

> "an interaction between two (or more) organisms (or species), in which, for each, the birth and/or growth rates are depressed and/or the death rate increased by the other organisms (or species)" (Begon, Harper & Townsend 1986).

Competition occurs when a number of organisms need to utilise similar resources in their ecosystem (i.e. they have similar niche requirements) and such resources are in short supply or when organisms seeking the same resources harm one another in the process (Connell 1985).

Competition between two or more species within a community is known as interspecific competition, whereas competition between individuals or populations of the same species is known as intraspecific competition. Interspecific competition ultimately results in individuals of one or all competing species suffering a reduction in fecundity, survivorship or growth and has been shown to ultimately lead to local extinction of the less efficient species (Gause 1934). As a result, a species can be defined as a superior competitor for a particular suite of environmental conditions if, given sufficient time for competitive interactions to go to completion, it displaces another species (Tilman 1987). Such effects of competition on population dynamics of competing species can, therefore, ultimately influence the species' distributions and evolution.

Competition can be classified in three categories, namely interference competition, exploitative competition and apparent competition (Connell 1985). Interference, or contest, competition occurs directly between organisms via aggression or production of toxins etc., when an organism interferes with foraging, survival, reproduction of others or by preventing their physical establishment in a portion of habitat (e.g. Gause 1932). Exploitative competition occurs indirectly between organisms through the consumption or use of similar limiting resources. The use of resources by one organism depletes the amount available to others, or they compete for space (e.g. Gause 1934; Connell 1961; Tilman 1981). Apparent competition occurs indirectly between two species which are both preyed upon by the same predator. For example, species A and species B are both prey of predator C. The increase of species A will cause the decrease of species B because the increase of A would increase the number of predator Cs which in turn will hunt more of species B (Holt 1977).

1.1.2. A historical summary of the development of competition theory

The theory of competition has developed over many years through an array of studies. A combination of field experiments and indirect evidence from observations of how species use resources in the presence and absence of potential competitors, have facilitated a greater understanding of how competition between organisms occurs and the effects it can have in structuring natural populations. Much of the early investigations into competition were influenced by the mathematical models of Lotka and Volterra (see below), which describe how the number of individuals in a population, carrying capacity of the habitat, and competitive effect of one species on another, influence population growth over time (Volterra 1926; Lotka 1932).

The Lotka-Volterra model

The Lotka-Voltera model of interspecific competition (Volterra 1926; Lotka 1932) is an extension of the logistic equation model of intraspecific competition:

$$\frac{dN}{dt} = rN\frac{(K - N)}{K}$$

Where N = Number of individuals in population, K = Carrying capacity and t = time.

Interspecific competition, however, involves two species therefore the population size of one species can be denoted by N_1, and that of a second species by N_2. Their carrying capacities and intrinsic rates of increase are K_1, K_2, r_1 and r_2, respectively. The competition coefficient α_{12} measures the per capita competitive effect on species 1 of species 2. Two simultaneous differential equations constitute the model for interspecific competition:

$$\frac{dN_1}{dt} = r_1N_1\frac{(K_1 - N_1 - \propto_{12} N_2)}{K_1}$$

$$\frac{dN_2}{dt} = r_2N_2\frac{(K_2 - N_2 - \propto_{21} N_1)}{K_2}$$

This model is able to generate a range of possible outcomes: the predictable exclusion of one species by another ($K_1\alpha_{21}>K_2$); exclusion dependent on initial densities ($K_2\alpha_{12}>K_1$ and $K_1\alpha_{21}>K_2$); and stable coexistence ($K_1>K_2\alpha_{12}$ and $K_2>K_1\alpha_{21}$).

In the 1930s, Gause used logistic curves and the Lotka - Volterra model to derive parameters of intrinsic growth rates, carrying capacities and coefficients characterising the influence of one species of yeast (*Saccharomyces cerevisiae* (species *a*)) over the growth of another (*Schizosaccharomyces kefir* (species *b*)) in a mixed population (Gause 1932). In these experiments, he showed that species using the same food supply did not coexist in the laboratory. Gause confirmed that the Lotka-Volterra model for interspecific competition adequately described the growth of each species in a mixed population and showed that species *b* had a greater influence over the growth of species *a,* than vice versa. He further explained this example of interspecific competition by determining that the alcohol waste products from the yeast were the most important factor in inhibiting growth of both species. Species *b* excreted twice as much waste as species *a*, indicating that this was a primary factor behind competition between these two species. Further work of Gause in 1934 and 1935 on three species of protozoan *Paramecium* revealed that competing species may either exclude one another from particular habitats or may coexist, perhaps by utilizing the habitat in slightly different ways (Gause 1934, 1935).

These experiments of Gause gave rise to the principle known as the competitive exclusion principle (Hardin 1960). This states that no two species with similar ecological niches can coexist in a stable equilibrium, meaning that when two species compete for exactly the same requirements, one will be slightly more efficient than the other and will reproduce at a higher rate as a result. The fate of the less efficient species is local extinction (Gause 1932, 1934, 1935; Hardin 1960). Interspecific competition, however, does not always lead to competitive exclusion, because less efficient competing species can coexist albeit at reduced densities.

During the 50s and 60s another theory began to emerge that explained how species can coexist. In 1959, Hutchinson attempted to answer why there are so many kinds of animals? He believed that the process of natural selection, coupled with isolation and later mutual invasion of ranges leads to the evolution of sympatric species, which at equilibrium occupy distinct niches, according to the Lotka-Volterra model and competitive exclusion principle (Hutchinson 1959). An organism’s ecological niche describes how it fits into an ecosystem, including all biotic and abiotic resources the organism uses in its environment. In 1969, Macarthur and Levins went on to propose that there is a maximum level of niche overlap between two given species that will allow continued coexistence (Macarthur & Levins 1967), which is now known as the limiting similarity theory. This theory was developed using quantitative methods, namely the Lotka-Volterra model and probability theory. Tilman further added that other factors, such as competitive ability, colonization ability, and longevity also play a role in the potential for species with limiting similarity to co-exist (Tilman 1994).

During the 70s and 80s numerous models of interspecific competition for resources were developed (see Tilman 1987). Tilman tested such models and theoretical principles of competition by measuring the resource requirements (R*) of plants in a variety of experiments and developed the theory of resource competition (Tilman 1981, 1985, 1988; Tilman & Wedin 1991). In 1981, Tilman performed nutrient competition experiments for all pair wise combinations of four species of freshwater algae (*Ragilaria crotonensis, Asterionella formosa,* *Synedra filiformis and Tahellaria flocculosa*). The results showed that, as predicted by theory, the species with the significantly lower resource requirement was the superior competitor when both species were limited by the same resource. Two species were observed to coexist only if either (1) each was limited by a different resource and met the theoretical criteria for coexistence or (2) the species were limited by the same resource and did not differ significantly in their resource requirements (Tilman 1981). Tilman's model of competition arises solely from the process of acquisition and utilization of limiting resources. He found that disturbance rates and nutrient supply rates do not change the intensity of the competitive interaction but rather influence the dynamics and outcome of the competitive process (Tilman 1987, 1988).

Interspecific competition does not only occur over limited food supplies, (e.g. Gause (1934; 1935) and Tilman (1981)) or from a build up of toxins in the environment from one species affecting another (e.g. Gause (1932). It can also occur over limiting resources such as space, as Connell's work on Scottish barnacles showed in 1961 (Connell 1961, 1985). In his study, Connell conducted direct observations of barnacle growth at varying intertidal intervals around Scotland and showed that interspecific competition over space was an important process in reducing the abundance and reproductive output of the barnacle *Chthamalus* in its natural environment. Competition between *Chthamalus* and *Balanus barnacles* was the principal factor in determining the local distribution of *Chthamalus*, which thrived at lower levels when *Balanus* was absent, but did not survive at these levels when *Balanus* was present. When Connell moved his focus to barnacles in Washington, however, he illustrated the importance of evaluating competition along with other interactions in a particular community and found that intense predation prevented the occurrence of competition (Connell 1983).

Despite extensive research investigating interspecific competition it is, evidently, very difficult to produce a clear and unambiguous demonstration of interspecific competition in either laboratory or natural conditions (Connell 1983). As Hutchinson (1959) surmised, "Although animal communities appear qualitatively to be constructed as if competition were regulating their structure, even in the best studied cases there are nearly always difficulties and unexplored possibilities". A species may vary in at least two ways in the degree to which it competes. It may compete with some species in the community but not others, and it may compete with a particular species sometimes or in certain places but not in others (Schoener 1983). Field experiments studying interspecific competition usually involve changing the abundance of one possible competitor, species A, and comparing the response of the other, species B, to its behaviour in an unmanipulated control. The response measured is usually (i) a change in density; (ii) a change in some rate that could affect density, e.g., fecundity, growth, physiological condition, mortality, etc.; or (iii) a niche shift, e.g., a change in type of resource used or microhabitat occupied (Schoener 1983). However, sometimes interspecific competition may not be detected, even if it is occurring between studied species due to unforeseen circumstances. For example, in some years or at some sites resources may not be limiting or predation may reduce the populations to such low levels that they are not competing for their resources; or the process of competition may be interrupted by physical disturbances. Tilman (1987) concluded that in order to determine if a species is a superior competitor, it is necessary to perform experiments over a long period of time so that competitive interactions can be observed.

Studies on spiders in the 70s and 80s illustrate some of the difficulties in demonstrating interspecific competition experimentally. In 1979, Wise found that prey abundance was limiting egg production in two orb-weaving spider species, namely the basilica spider *Mecynogea lemniscata* and the labyrinth spider *Metepeira labyrinthea* (Wise 1979). This study indicated that some females of these species of spider were competing but the extent and mechanism of possible competitive interactions were unclear. In 1981, Wise established mixed and single-species populations of young adult females of these same spider species at a range of densities, to determine whether or not high densities of either conspecifics had negative effects upon survival and fecundity. However, this experiment showed no significant negative interspecific effects of density upon survival or reproduction, which indicated that interspecific competition was not evident during this experiment (Wise 1981). These studies indicate that the significance of resource limitation and competition may vary temporally for these species of spider, making investigations into such mechanisms challenging.

In summary, competition is important in regulating the relative abundance of many species and perhaps the species richness of many communities, when population sizes are near carrying capacity and resources are limiting (Schoener 1983; Tilman 1988). Where species are “too” similar in the resources they use they cannot coexist “for long” as one will competitively exclude the other (Gause 1932). Species that coexist in nature, therefore, do so by virtue of “sufficient” differences in ecological niche (Hutchinson 1959; Macarthur & Levins 1967). Geographic distributions of species appear to be influenced by competitive pressures (Connell 1961). Given enough time, such competitive pressures may determine how many and which species coexist in a community and ultimately be a powerful evolutionary force, selecting for adaptations that result in species differing in the type of resources used (i.e. beak size for different sizes of seed) (Hutchinson 1959; Schoener 1983). Experiments performed on species with “substantial” overlap in their use of resources should detect interspecific competition. For example, the introduction of individuals of species B into a place inhabited by species A should depress the latter’s population, or affect individuals in species A i.e. decrease growth or body size, which will depress its population. Removal of individuals of species B should have the reverse effect (Schoener 1983).

1.1.3. Evidence of competition between people and elephants

The interaction between humans and elephants living in the same ecological community could be considered an example of interspecific competition. Both species utilise similar resources within their environment, whether it be water for drinking and bathing, trees and grasses for food or building, and space for home ranges, fields and settlements. In addition, as Parker and Graham (1989a) highlighted, crop cultivation by humans can only take place at the expense of elephant range, where the two species coexist, and elephants find virtually all human food crops palatable leading to potential competition over both natural and cultivated forage resources. Past field observations have shown that where one species persists in high densities the population density of the other is often reduced. For example, Laws, Parker and Johnstone (1975), observed an incident in Uganda where elephants moved out of an area once it became occupied by people. A similar situation was also described by Kingdon near Lake Rukwa, Tanzania in the 1960s (Kingdon 1979). Elephants have also been found to avoid unprotected areas with high human population density (Parker & Graham 1989b; Barnes *et al.* 1991; Eltringham 1991; Happold 1995; Hoare & du Toit 1999; Wittemyer *et al.* 2007; Harris *et al.* 2008; Jackson *et al.* 2008; Graham *et al.* 2009), human settlements (Harris *et al.* 2008) and areas of high human activity (Graham *et al.* 2009). Humans have also been recorded leaving settlements when elephant numbers are deemed too high in the area (e.g. Osborn 1998a). Such emigration and avoidance of areas with high densities of humans could be elephants responding to a competitive pressure on resources by humans. In other words, exploitative competition, where both species use and consume the same resources (space, vegetation and water), may be occurring between people and elephants.

In addition, aggressive interactions that occur between elephants and people over access to resources, which occasionally leads to the death of individuals of either species, could be described as an example of interference competition. Elephants and people have, perhaps as a result of this competitive interference, been found to utilise resources at different times of the day (i.e. water) in areas where the two coexist (Hoare 1999b; Hoare & du Toit 1999; e.g. Graham *et al.* 2009; Loarie, van Aarde & Pimm 2009a) as well as avoid certain areas when the other species is present, as described above. There is evidence, therefore, that people and elephants are avoiding each other, not necessarily over direct competition for resources, but more to avoid incidents of direct interference competition. This would infer that the costs associated with interference competition between elephants and people are actually a driving constraint on resource and spatial use in areas where the two species co-exist. This is consistent with a similar phenomenon recorded as a result of interference competition between elephants and other herbivore species at waterholes in Hwange National Park, Zimbabwe (Valeix, Chamaille-Jammes & Fritz 2007).

To clearly define the interaction between people and elephants as a form of ecological interspecific competition is, however, difficult because current evidence is based purely on observations. Behaviour and movement patterns of elephants are influenced by an array of abiotic and biotic factors and it is, therefore, difficult to explicitly conclude that emigration out of or avoidance of areas with high human densities is a result of a competitive interaction between the two species. In theory, the population growth of people and elephants should be reduced once resources become sufficiently limited by the other species. However, this will only occur once population numbers reach a maximum level (carrying capacity within a specific environment). With people and elephants both being highly mobile species, it is unlikely that this scenario would occur without either species leaving an area. In order, however, to confirm if interspecific competition is indeed taking place, controlled field experiments like those described above are needed, which would be very challenging. Single species populations and mixed species populations would need to be established to determine the density dependent effects of one species on the other. Such enclosure experiments, however, would be logistically challenging (if not impossible) to establish in practise due to the size and home ranges of the study animals. There are also many other driving factors that could affect the population growth of people and elephants, but which are difficult to control for in a natural system (e.g. physical factors such as fire and drought, other animal species which could be competing or predating on the study species, and economic or political factors in the human population).

Even though it is difficult to prove unequivocally that interspecific competition is definitely occurring between people and elephants, current evidence does suggest people and elephants do compete for limited resources where the two species co-exist. It is clear that both species utilise and require similar resources in the ecosystem. Humans do destroy (through utilisation and development) the natural forage resources of elephants and elephants and people do aggressively interact over access to certain resources (i.e. crops, natural forage, water and space). In addition, elephant densities are often much lower in areas where human densities are high and vice versa, as both people and elephants have been known to emigrate from an area when population numbers of the other species increase. Based on the evidence and historical observations outlined above, one can, therefore, infer that competition between humans and elephants over resources such as water, space, vegetation and crops does occur and often results in conflict.

1.2. Human – Wildlife Conflict (HWC)

1.2.1. What is HWC?

When two parties compete for a shared resource there is always potential for disagreement to arise. This can happen between individuals, populations or communities, both within species and between species. It is not surprising, therefore, that when humans and wildlife live in close proximity and share resources, negative interactions or conflict may occur. Messmer, (2000) defines the commonly used term human-wildlife conflict (HWC) as:

> A situation that involves negative interactions between humans and wildlife

In ecological terms HWC could be described as a form of interspecific competition, where two different species in a community rely on similar limiting resources and compete for these. However, because humans are involved the situation becomes more complex and sociological, political and economic aspects of the situation also need to be considered (Treves *et al.* 2006). Human-wildlife conflict scenarios also frequently involve an element of conflict between people who have different goals, attitudes, values, feelings, levels of empowerment, and wealth (Madden 2004), complicating the situation again. For example, a fundamental question in many wildlife conservation issues including human-wildlife interactions, in developed as well as developing countries, is “who owns wildlife?” (Brown & Decker 2005) and therefore whose responsibility is it to reduce HWC? HWC can also become quite controversial where conservation policy and practice have prevented or discouraged people from taking direct action against wildlife species or when the resources concerned (i.e. crops or livestock) have economic value (Thirgood *et al.* 2000; Graham, Beckman & Thirgood 2005). The expectation may be for government agencies to assume responsibility for providing adequate protection against wildlife in such cases, rather than people in direct conflict with wildlife (Naughton-Treves 1999; Campbell 2000; Knight 2000; Hill 2004). As Treves, *et al.*, (2006) argue “the enforcement of environmental protections and non-utilitarian views of wildlife have changed what was once a simple competitive relationship between people and wildlife into a political conflict between people and between institutions”. Careful consideration of ecological, social and political aspects is therefore needed and solutions for addressing HWC should be designed accordingly (Hoare 2001; Warren, Buba & Ross 2007).

The use of the term human-wildlife conflict can also be problematic as Peterson *et al.*, (2010) investigated. This term automatically insights a negative attitude towards the situation where humans and wildlife are interacting and due to traditional definitions of “conflict”, it infers wildlife as conscious human antagonists. Peterson *et al.*, (2010) argue that the phrase human-wildlife conflict is detrimental to coexistence between humans and wildlife as it dichotomizes humans and nature, framing wildlife as something that threatens human existence, rather than contributing to human welfare. They suggest that the term human-wildlife conflict be replaced with something akin to human-wildlife coexistence.

1.2.2. Why does HWC occur?

People interact with wildlife on many different levels, both positively and negatively. For example, Conover, (2002), identified six different attitudes of people towards wildlife in the USA, ranging from negativistic to naturalistic and human attitudes towards animals can be very complex (Dickman 2010). When the interests and behaviour of humans and wildlife clash, this can lead to negative interactions between people and wildlife, which eventually insights more negative attitudes towards wildlife and conflict between the two parties can arise. The underlying cause of HWC is therefore ultimately due to people and wildlife having to live in close proximity in certain environments.

Negative human-wildlife interactions occur across all continents and are not a new phenomenon. Recently, however, it appears that HWC cases are increasing in either frequency or intensity in many places (e.g. Kiiru 1995; Hoare 2000b; Conover 2002; Linkie *et al.* 2003; Weladji & Tchamba 2003; Bagchi & Mishra 2006; Agetsuma 2007; Blanc *et al.* 2007; Linkie 2007; Jackson *et al.* 2008), and factors contributing to these increases depend on the type and location of conflict as well as the perception of different stakeholders (Weladji & Tchamba 2003). For example, in the Benoue Wildlife Conservation Area in Cameroon it was found that some stakeholders considered the involvement of local people in illegal activities as the main cause of conflict, while others thought it was due to their lack of access to natural resources, and others believed the main cause was wildlife damaging property (Weladji & Tchamba 2003). Kolowiski & Holekamp (2006) believe that the human-carnivore conflict occurring in pastoral villages on the edge of the Maasai Mara, Kenya is a result of an exponential increase in the human population, while other studies have identified an increase in wildlife numbers or density as an exacerbating factor (Hoare 1999b). The cause of a conflict or increase in intensity of conflict evidently depends on the wildlife species involved, the type of conflict occurring, the location, the status of prey/natural resource availability, socio-economics, and politics.

A common factor attributed to causing many conflicts between humans and wildlife is land use change. When natural ecosystems are converted to agricultural land or human settlements, wildlife habitats can become reduced and fragmented, which in turn lead to more frequent interactions between people and wildlife (Linkie *et al.* 2003; Agetsuma 2007). For example, a study in Japan examined crop raiding by Japanese deer (*Cervus nippon*) and monkeys (*Macaca fuscata*) on the island of Yakushima and highlighted the main cause of crop raiding to be the change in land use from diverse forest to coniferous plantations (Agetsuma 2007). Linkie *et al.*, (2003) investigated threats to the Sumatran tiger (*Panthera tigris sumatrae*) and found that habitat loss from illegal and commercial logging, oil palm production, pioneer farming, mining operations and forest fires are a major threat to tigers and exacerbate HWC. This could be considered a direct cause of conflict between people and tigers, which then leads on to further conflicts like hunting and poaching by humans, livestock depredation and loss of human lives by tigers. Nhyus & Tilson, (2004a) highlighted that recent history has shown that wildlife conflict with humans can increase as existing habitat becomes progressively fragmented and the pressure of human encroachment becomes more severe. Increasing human and/or wildlife populations and home-ranges have also been identified as causes of increasing HWC (e.g. Conover & Decker 1991; Hoare 2000a; Treves *et al.* 2004), as well as cropping regimes and livestock husbandry techniques (e.g. Ogada *et al.* 2003; Woodroffe *et al.* 2007b). The socioeconomic factors of people living in close proximity to wildlife may also contribute to increasing levels of HWC, for example if the main mode of transport in an area is walking then human encounters with potentially dangerous wildlife species could be a lot more severe than if people could afford to travel in a vehicle.

1.2.3. Types of HWC

Numerous wildlife species have been reported conflicting with humans all over the world, ranging from insects and birds to large mammalian herbivores and carnivores. Negative interactions can occur at various scales and different intensities depending on an array of factors, including the wildlife species involved, mitigation techniques available, and socioeconomic status of the human population.

A common form of HWC is wildlife crop damage, with a variety of wildlife species perpetrators recorded, (Lahm 1996; Naughton-Treves 1998; Hoare 1999b; Tourenq *et al.* 2001; Gillingham & Lee 2003; Agetsuma 2007; Amano *et al.* 2007; Hegel, Gates & Eslinger 2007). Wildlife crop-raiders include baboons e.g. Chacma baboons (*Papio cynocephalus ursinus*); monkeys e.g. Vervet monkeys (*Cercopithecus aethiops*) and Japenese macaque (*Macaca fuscata*); suids e.g. warthogs (*Phacochoerus aethiopicus*) and Japanese serow (*Capricornis crispus*); rodents e.g. porcupines (*Hystrix africaeaustralis*); African elephants (*Loxodonta africana*); Asian elephants (*Elephas maximus*); Japanese deer (*Cervus nippon*); elk (*Cervus elaphus canadensis*); birds e.g. greater flamingo (*Phoenicopterus ruber roseus*) and geese (*Anser albifrons*); and insects. Each wildlife species appears to raid with different frequencies and intensities, highlighting the need to be adaptive in mitigation strategies applied. For example, crop damage is often inflicted by small animals including rodents, birds and insects (e.g. Tait & Napompeth 1987; Naughton-Treves 1998), yet such incidents may be considered small scale problems with low intensity because they are manageable as long as one can afford the mitigation measures. However, an example of a higher scale and greater intensity type of HWC is that of elephants raiding a farmer's crop in a rural African village. Here the wildlife species is large and potentially dangerous to humans, there is no unique mitigation technique that appears to work on its own and elephants can destroy a whole field of crop in one night. The extent and impact of damage incurred from wildlife crop-raiding incidents may be influenced by the type and magnitude of mitigation efforts applied (Hill 2000; Sitati, Walpole & Leader-Williams 2005; Warren, Buba & Ross 2007), as well as cultural and socio-economic aspects of the farmer.

Another widespread form of negative interactions between humans and wildlife worldwide is livestock depredation, with a variety of carnivores recorded killing animals that humans farm, hunt or harvest for consumption or recreation (Conforti & Cascelli-de Azevedo 2003; Treves & Karanth 2003; Frank, Woodroffe & Ogada 2005; Graham, Beckman & Thirgood 2005; Rabinowitz 2005; Johnson *et al.* 2006). In Asia, species such as snow leopard (*Uncia uncial*), tigers (*Panthera tigris*), leopard (*Panthera pardus*), and black bear (*Ursus thibetanus*) have been recorded preying on livestock (Sekhar 1998; Tilson & Nyhus 1998; Hussain 2003; Mishra *et al.* 2003; Nyhus & Tilson 2004b; Bagchi & Mishra 2006; Johnson *et al.* 2006; Ngoprasert, Lynam & Gale 2007; Naim & Chauhan 2010). While in Africa, lions (*Panthera leo*), spotted hyaena (*Crocuta crocuta*), leopards (*Panthera pardus*), cheetah (*Acinonyx jubatus*), wild dogs (*Lycaon pictus*), brown hyaenas (*Hyaena brunnea*) and crocodiles (*Crocodylus niloticus*) are known to attack livestock (Linnell *et al.* 1999; Hemson 2003; Marker & Dickman 2004; Patterson *et al.* 2004; McGregor 2005; Hazzah 2006; Kolowski & Holekamp 2006; Muntifering *et al.* 2006; Begg, Begg & Muemedi 2007; Schiess-Meier *et al.* 2007; Aust *et al.* 2009; Dunham *et al.* 2010). In the USA, wolves (*Canis lupus*), coyotes (*Canis latrans*), grizzly bears (*Ursus arctos*) and black bears (*Ursus americanus*) have been found to predate on sheep and cattle (Mladenoff, Sickley & Wydeven 1999; Conover 2002; Musiani *et al.* 2003; Naughton-Treves, Grossberg & Treves 2003; Gore *et al.* 2006; Wilson *et al.* 2006; Baruch-Mordo *et al.* 2008), wolves in Europe predate on sheep (Bisi *et al.* 2007), and in South America pumas (*Puma concolor*) and Jaguars (*Panthera* onca) cause problems for livestock owners, (Polisar *et al.* 2003; Altrichter, Boaglio & Perovic 2006; Palmeira *et al.* 2008).

Attacks on humans by wildlife, either by large carnivores (i.e. lion, leopard, tiger, bears), (Saberwal 1997; Conover 2002; Nyhus & Tilson 2004b; Johnson *et al.* 2006), large herbivores (i.e. African elephant, Asiatic elephant (*Elephas maximus*), buffalo (*Syncerus caffer*), hippopotamus (*Hippopotamus amphibious*)), (Sukumar 1991; Conover 2002; Sitati *et al.* 2003) or other animals i.e. snakes, spiders etc. (Conover 2002) are recorded occasionally. Birds have also been recorded attacking and injuring humans, for example in Australia magpies attack humans when they are close to magpie nesting sites and as such are considered to be a serious urban wildlife issue (Jones 2004). When humans are seriously injured or killed by such wildlife attacks it can intensify the conflict (Conover 2002). Fear can have a big influence on peoples' attitudes toward wildlife (Kaltenborn, Bjerke & Nyahongo 2006) and greater negative perceptions of wildlife can lead to a decrease in tolerance for conservation of wildlife (Conover 2002). Therefore, it seems imperative to understand whether human encounters with wildlife in an area are experienced as positive or negative in order to form effective management strategies to reduce HWC (Kaltenborn, Bjerke & Nyahongo 2006).

The impact of HWC can also be severe for many wildlife species. Lethal problem animal control methods, along with hunting and poaching pressure present a direct threat to lives of animals involved in negative human-wildlife interactions. Woodroffe, Thirgood & Rabinowitz, (2005) indicate that the deliberate killing of animals that are considered pests has driven several species to extinction, and has contributed to the endangerment of many others. Wildlife are also subjected to indirect risks when they live in close proximity to people, including habitat destruction or fragmentation, reduction in home range, and loss of natural resources i.e. food, water and shelter, (Hussain 2003; Linkie *et al.* 2003; Hedges *et al.* 2005; Johnson *et al.* 2006; Agetsuma 2007).

A less obvious type of HWC occurs when wildlife are assumed to be alternative hosts for diseases (Happold 1995; Conover 2002; Gummow 2010). There are between 100 – 3000 diseases which can be classified as zoonoses, ranging from the bubonic plague hosted and vectored by rodents to Lyme disease hosted by rodents and deer, and vectored by ticks, (Conover 2002). Some diseases can also be transmitted from wild animals to domesticated animals e.g. foot and mouth disease, causing an indirect form of HWC, as well as from domestic animals to wild animals, which seriously threatens certain wildlife species. For example, domestic dogs can transmit diseases e.g. rabies, to wild dogs (*Lycaon pictus*) (Davies & duToit 2004; Woodroffe *et al.* 2007b) and Ethiopian wolves (*Canis simensis*) (Sillero-Zubiri, King & Macdonald 1996; Stephens *et al.* 2001), which are thought to be contributing to declines in these endangered mammal populations (Stephens *et al.* 2001).

Not all HWC incidents result in direct impacts to humans or wildlife, some result in opportunity costs, where people forgo economic or lifestyle choices due to impositions placed upon them by the presence of wild animals or conservation areas (Woodroffe, Thirgood & Rabinowitz 2005). Whatever form of HWC is occurring, it is evident that to aid wildlife managers, rural farmers or other stakeholders involved, information is required on all aspects of the situation to enable informed decisions to be made and effective strategies for management and mitigation developed.

1.3. Competition between elephants and humans

1.3.1. Human-Elephant Conflict (HEC)

Elephants, both in Africa (*Loxodonta sp.*) and Asia (*Elephas maximus sp.*) are a prime example of a wildlife species which frequently competes with people for limited resources (Sukumar & Gadgil 1988; Lahm 1996; Hoare 2000b; Osborn 2004; Chiyo *et al.* 2005; Sitati, Walpole & Leader-Williams 2005). Negative interactions between people and elephants are most frequently referred to as human-elephant conflict (HEC), with the broad definition of HEC adopted by the IUCN African Elephant Specialist Group (AFESG) being:

> Any human-elephant interaction which results in negative effects on human social, economic or cultural life, on elephant conservation or on the environment

Elephants require a large amount of space (Owen-Smith 1988; Chase & Griffin 2007) and due to their physiology and energy requirements, also need to consume large quantities of food and water each day (Laws & Johnstone 1970; Owen-Smith 1988; Sukumar 1990; Ruggiero 1992; Choudhury *et al.* 2008). Humans also require large quantities of these limited resources which leads to competition between the two species when they utilise the same habitats (Hoare 2000b; Conover 2002; Hegel, Gates & Eslinger 2007).

There are four main categories of HEC which have been identified in the literature, including those which have:

a) A direct impact on humans such as crop raiding (destruction and damage to crops), property damage (water installations, fencing etc), injury and death of livestock, injury to humans and loss of human life;
b) Indirect impacts on humans such as restriction on people's movement (especially at night), loss of sleep or reduced school attendance while guarding crops or stores, and early crop harvesting (AFESG 1999; Naughton, Rose & Treves 1999);
c) A direct impact on elephants such as loss of habitat as a result of human encroachment, injury to elephants and elephant death through hunting, poaching or problem animal control;
d) Indirect impacts on elephants such as restriction on elephant movement to certain resources.

One of the most frequent clashes between elephant and people occurs in agricultural environments, where elephants damage or destroy subsistence or commercial farmers' crops. Although elephants are usually not the only perpetrators of crop-raiding in these systems, due to their size and potential danger to humans, elephants are often the species most readily complained about (Naughton-Treves 1999; O'Connell-Rodwell *et al.* 2000; Hoare 2001). Studies comparing wildlife crop-raiding species have shown that smaller wildlife species (i.e. insects, birds, rodents and primates) often cause the most frequent damage to crops (e.g. Naughton-Treves 1998) and result in the greatest widespread economic loss in agriculture (Conover 2002), whereas elephant damage is often localized and infrequent (Naughton-Treves 1998). However, one elephant crop-raiding incident can be very severe to the individual farmer, especially in cases where a whole field of crops is destroyed by one or multiple elephant forays (Naughton-Treves 1998).

Crop-raiding by elephants is not a new phenomenon, even as early as 300BC-AD 300, an account of Indian political and governmental strategy (the *Arthashastra*) recognised that wild elephant populations and human settlements cannot coexist without some conflicts (Eltringham 1991). The relationship between people and elephants has even been described as a form of competitive exclusion (Parker & Graham 1989b). There have been numerous reports of fields and villages even being abandoned due to crop-raiding elephants both in the past (e.g. Bell & McShane-Caluzi 1984; Osborn 1998a) and recently (e.g. see newspaper quote from Zimbabwe below).

> A HERD of elephants from Hwange National Park and Debshan Sanctuary is wreaking havoc in the Vungu area of Gweru District where the jumbos are destroying crops....Villagers in the affected areas ..... are reportedly living in fear and most have deserted their fields....
>
> (Anon 2010) Chronicle, Zimbabwe: 9th April 2010.

In the past, farmers in many parts of Africa formed large villages with highly defended (i.e. collectively many men were responsible for actively protecting crops) cultivated areas to reduce crop loss by animals such as elephants (Parker & Graham 1989b). Then with the arrival of Europeans, elephants seen as a threat to human life or property were commonly controlled by shooting (Carrington 1958; Hanks 1979). Selective shooting of elephants by wildlife authorities has been widely employed throughout Africa since, as the main method of control (Hoare 1995), however, Bell & McShane-Caluzi, (1984) showed empirically that it actually has little effect on deterring crop-raiding elephants. Today, wildlife management authorities and community members commonly use a variety of short term mitigation measures alongside controlled shooting, such as barrier methods (i.e. bush, wire and electric fences); olfactory repellents (i.e. chilli peppers (Osborn 2002; Parker & Osborn 2006; Hedges & Gunaryadi 2009)); acoustic deterrents (i.e. drums and bangers) (Hoare 1995; Conover 2002) and more recently bees (King *et al.* 2009; King *et al.* 2010; King, Douglas-Hamilton & Vollrath 2011). However, as elephant behaviour is relatively adaptable, animals may habituate over time to loud noises, fires, throwing of objects and torches (Osborn & Parker 2003) and therefore many active Problem Animal Control (PAC) methods tend to diminish in effectiveness after repeated use (Taylor 1993). Long-term solutions to reducing conflict, including compensation schemes, improved land-use and settlement planning and problem animal translocation programmes (Hoare 1995; Massei *et al.* 2010) have also been used, however the effect of such schemes is harder to assess as they require long term monitoring. Integrated management solutions, such as community based conflict management described by Osborn & Parker, (2003) are currently being trialled, and appear to be effective, in countries such as Namibia, Zambia and Botswana (Songhurst 2010) and India (Davies *et al.* 2011).

1.3.2. HEC in Botswana

Botswana lies in the southern region of Africa, covering an area of 600, 370km^2 comprised of predominantly deep Kalahari soils. The elephant population of Botswana ranges freely over approximately 100, 265km^2 of the country, but with only 19% of the range inside protected areas, (Blanc *et al.* 2007). It is part of the largest contiguous population of elephants in Africa, with a current population estimate of 128,000 elephants and average density of 1.75 elephants/km^2 (Chase 2011). The population was estimated to be growing at a rate of 5-6% per annum in 1998 (Gibson, Craig & Masogo 1998), however, a recent study reviewing the population estimates and densities of elephant in Botswana argue that numbers have begun to stabilize (Junker, van Aarde & Ferreira 2008). Whether or not the population of elephants is increasing, it is evident that the elephant range is expanding with home ranges as large as 24,828 km^2 recorded (Chase & Griffin 2007). Comparably, the human population in Botswana is increasing nationally at around 1.2% per annum (CSO 2001) and people are requiring more land for agricultural uses every year. The most recent national agricultural survey shows that, over 3 years (2003-2006), the area of land planted for traditional agriculture increased by 46%, while the number of subsistence farmers planting crops increased by 51% (CSO 2006). Interactions between people and elephants are becoming more frequent and consequently there appears to be an increase in HEC incidents (Natural Resources & People 2006a).

The Government of Botswana states that the overall goal for elephant conservation and management in Botswana is to:

> Conserve and optimise elephant populations while ensuring the maintenance of habitats and biodiversity, promoting the contribution of elephants to national development and to the communities within their range at the same time minimising their negative impacts on rural livelihoods (MEWT 2000)

However, it is feared that high concentrations of elephants (especially around permanent water sources) are causing detrimental effects on the vegetation and consequently affecting other herbivore species, as well as affecting the livelihoods of people living on the periphery of the expanding elephant range. The Government of Botswana therefore identified ‘reducing HEC to an acceptable level’ as an important management activity in the 2000-2001 Botswana Elephant Management Plan (MEWT 2000), and in 2006 a component of the Okavango Delta Management Plan (ODMP) was specifically designated to addressing human elephant conflict (Natural Resources & People 2006a). Recently, the Government of Botswana has been awarded a Global Environment Fund grant from The World Bank to address ‘Human-Wildlife Conflict and Improve Rural Livelihoods in Northern Botswana’ (World Bank 2010), a main element of which is addressing HEC. It appears that the reduction of HEC is considered to be a highly important factor in the future management and conservation of elephants in Botswana.

Currently, HEC in Botswana is being addressed through a number of strategies. At the government level, the Department of Wildlife and National Parks (DWNP) collect data on HWC incidents and have a PAC unit, whose role is to scare problem animals away from people’s property sometimes using lethal action. To alleviate economic pressure of HEC on individuals the government introduced a solely state funded compensation scheme in 1996 (DWNP 2010). Compensation systems are based upon paying reparation to property owners for losses incurred from wildlife damage and aim to raise farmer’s tolerance for losses by minimising the economic impact of these losses (Nyhus, Osofsky & Ferraro 2005). Crop damage inflicted by animal species listed as dangerous in Schedule 9 of the Wildlife Conservation and National

Parks Act of 1992 (hippopotamus, rhinoceros, elephant, and buffalo) are compensated for. Compensation rates vary depending on the crop damaged, and 50% of the replacement value is given for other types of property e.g. fences damaged (DWNP 2010). Compensation, however, is a method for increasing tolerance for a problem rather than preventing it from occurring (Macdonald & Sillero-Zubiri 2004).

At a community level, people in Botswana use traditional methods including, fences, shouting, drums and occasionally guns to scare elephants, however people feel many of these are becoming ineffective. In 2006, the government introduced a chilli mitigation strategy to try and empower farmers to actively protect their own crops and devolve more responsibility to communities for crop protection. Then in 2010, the compensation rules became stricter specifying that farmers need to show they are actively trying to protect their fields before compensation will be given (DWNP 2010). All these actions are coming from a top-down approach and have been introduced through organizations outside of the affected community. As Osborn & Parker (2003) highlighted, interventions introduced in this way can incite an expectation from farmers for these external organisations to resolve the conflict, and therefore there is a risk that communities may still rely heavily on the government to resolve HEC for them.

Ultimately, HEC prevention in both the short and long term requires an understanding and identification of conditions promoting HEC, knowledge of the elephant and human populations involved, and an ability to focus assistance and mitigation accordingly (Treves *et al.* 2004). As discussed above, careful consideration of ecological, social and political aspects is therefore needed in order to address and hopefully reduce the conflict (Hoare 2001; Warren, Buba & Ross 2007).

1.4. Elephant biology

Elephants have a prominent role in ecosystems and are consequently referred to as ecosystem engineers (Western 1989; Ruggiero 1992). These animals are also a significant natural resource and through effective sustainable management may provide a major source of income for many African and Asian countries (Eltringham 1991; Happold 1995; Goodwin 1996). Elephants are highly intelligent and socially sophisticated creatures, which have adapted to a wide variety of different habitats (Laws & Johnstone 1970; Estes 1991). Preliminary genetic evidence suggests that there may be two distinct living species of African elephant identified namely the savannah elephant *Loxondonta africana* (Blumenbach, 1797) and the forest elephant *Loxodonta cyclotis* (Matschie, 1900), and a possible third species, the west African elephant has been postulated (Eggert, Rasner & Woodruff 2002; Blanc *et al.* 2007; Blanc 2008). Only one living species of Asian elephant *Elephas maximus* (Linnaeus, 1758) has been identified, which is sub divided into three distinct subspecies, namely the Indian elephant *Elephas maximus indicus* (Cuvier, 1798), the Sri Lankan elephant *Elephas maximus maximus* (Linneaus, 1758) and the Sumatran elephant *Elephas maximus sumatranus* (Temminck, 1847), (Choudhury *et al.* 2008), with a possible fourth subspecies suggested for the Bornean elephants (Fernando *et al.* 2003). Across the wide range of extant elephants the biggest threat to their survival and conservation is competition with people, either through poaching, reduction of habitat or problem animal control (PAC).

1.4.1 Population numbers

The estimated total number of elephants in Africa is 472, 269 animals with an estimated range of 3,335,827km^2 (Blanc *et al.* 2007), and 58% occurring in southern Africa. On a regional scale it is estimated that populations in southern and eastern Africa increased at an annual rate of 4% (95% CI of rate 1.14% to 6.58%) between 2002 -2007, based on relevant total numbers, (Blanc *et al.* 2007), however this does not reflect the situation in central or western Africa.

Population growth rates are variable between study sites. The Amboseli Elephant Research Project in Kenya has collected data on individually known free-ranging elephants for the past 36 consecutive years. In Moss (2001) it is recorded that the average population increase in Amboseli was 2.17% per annum (range = 3.75% - 11.28%) between 1974 and 1999, with no immigration or emigration observed. After a decrease in numbers due to poaching and drought, the population recovered at a rate of 3.75% per annum. Taylor (1993) estimated the population growth rate in Nyaminyami District, Zimbabwe to be 3.4% and in Samburu and Buffalo Springs National Reserve (SBSNR), Kenya the elephant population increased at a rate of 4.6% per annum over the 6 year study period they were monitored (Wittemyer *et al.* 2005). In Addo National Park, South Africa the average population growth rate was a lot higher at 5.53%, (Whitehouse & Hall-Martin 2000) and Gibson, Craig & Masogo (1998) estimated the Botswana population increasing at 5-6% per annum. The maximum rate of increase for elephant populations was calculated as 7% per annum (Calef 1988), based on first ovulation age of females in ideal conditions (8-10 years) and the mean calving interval in rapidly expanding populations (3-4.5 years), which agrees with the highest rate of increase reported in the wild in Addo National Park, (Hall-Martin 1980).

Death rates in elephant populations not only vary throughout their range but also within any one area or population as a result of local environmental conditions (Hanks 1979). For example, mean annual mortality in Addo National Park was estimated at 1.43% (Whitehouse & Hall-Martin 2000), yet in SBSNR it was higher at 2.6%, (Wittemyer *et al.* 2005). In 1969, Laws (1969) found death rate varied between 2-3% in Tsavo and Douglas-Hamilton, (1972) recorded a 3-4% death rate in Lake Manyara, Tanzania. Maximum rate of population decrease in Amboseli was 7.42%.

Elephant population density also appears to vary across their range and between seasons (Laws & Johnstone 1970; Douglas-Hamilton 1973; Sommerlatte 1976; Hanks 1979; Taylor 1993; Whitehouse & Hall-Martin 2000; Hien 2005). For example in Africa, densities as low as 0.09 elephants/km^2 have been recorded in Burkino Faso (Hien 2005), and as high as 5/km^2 in Lake Manyara, Tanzania, (Douglas-Hamilton 1973). In Chobe National Park, Botswana higher densities were recorded in the dry season (4.58 elephants/km^2) than in the wet season (0.32-0.52/km^2) (Sommerlatte 1976). Environmental factors affecting local elephant densities appear to be water (Verlinden & Gavor 1998; Stokke & duToit 2002; Redfern *et al.* 2003; Grainger, van Aarde & Whyte 2005; Chamaille-Jammes, Valeix & Fritz 2007), vegetation heterogeneity (Murwira & Skidmore 2005; Chamaille-Jammes, Valeix & Fritz 2007) and sodium availability (Hanks 1979).

It is evident from the current literature available on elephant abundance that there are certain limitations in collecting precise data on densities and numbers. These are often practical difficulties relating to the remoteness of some elephant populations, and the habitats where elephants are found i.e. in dense vegetation or in difficult terrain (Blanc *et al.* 2007; Choudhury *et al.* 2008). A number of different survey techniques are used to estimate numbers and this can make comparisons between studies difficult. Possible sources of bias in surveys include the choice of survey technique, surveyor skill, quality and availability of equipment, financial constraints, climatic conditions and vegetative cover (e.g. Norton-Griffiths 1978; Barnes 1993; Douglas-Hamilton 1996; Kangwana 1996; Craig 2004; Hedges & Lawson 2006)

In chapter two of this thesis, I determine elephant abundance in the study area and explore issues of bias and precision in aerial transect surveys for monitoring elephant populations.

1.4.2. Social organisation

Elephants have highly complex social structures and elephant society is fundamentally matrilineal (Eltringham 1982). Elephants are generally found in three types of groups: a) all male groups, b) cow/calf groups, and c) mixed groups - cow/calf groups with adult males present, (Moss 1996). In female led groups, there appear to be six hierarchical tiers of organisation in elephant society, namely mother-calf units; family units; bond/kinship groups; clans; subpopulations; and populations (Douglas-Hamilton 1972; Moss & Poole 1983; Wittemyer, Douglas-Hamilton & Getz 2005). The average family unit size of African elephants is between 8-9 animals (ranging from 2-25), (e.g. Moss & Poole 1983; Wittemyer 2001) whereas Asian elephants are usually found in smaller groups of 4-8 animals although some larger groups of 10-20 have been recorded, (Eltringham 1991). The family unit is usually comprised of a matriarch (old female), her daughters and their calves, as well as a number of female and male adolescent and juvenile offspring. Adult bulls often associate with family groups, especially when females are in oestrous, however, bulls generally tend to be solitary or associate in all male herds (range 1-30), (Moss 1988; Eltringham 1991; Moss 1996). Aggregations of elephant groups can range from 1-500 elephants, (Eltringham 1991), with an average of 20 animals recorded in Amboseli, Kenya, (Moss 1988).

1.4.3. Feeding ecology

The elephant is one of the last few mega-herbivores (i.e. plant-eating mammals that reach an adult body weight in excess of 1,000 kg) still existing on earth (Owen-Smith 1988). Both the African and Asian species are generalist feeders, browsing and grazing on a variety of different plants and plant types. Given their physiology and energy requirements, elephants need to consume large quantities of food (~4-7% body weight) and water per day (Laws & Johnstone 1970; Ruggiero 1992; Choudhury *et al.* 2008), e.g. an adult African elephant requires an average of 100-300kg food and 220L water per day (Eltringham 1991; Estes 1991), and spend between 70-90% of their time foraging (Ruggiero 1992; Osborn 2004). The proportions of different plant types in the diet appear to vary depending on habitat and season, (e.g. Barnes 1982), but evidently they select food that offers the highest nutrient intake at a given place or time (Osborn 2004).

A variety of different plant types have been recorded in the elephant diet, consisting of different botanical taxa (McKay 1973; Barnes 1982; Sukumar 1989; Ruggiero 1992). Grass appears to be predominant in the diet of African elephants in the wet season and browse during the dry (Laws, Parker & Johnstone 1975; Field & Ross 1976; Guy 1976; Barnes 1982; Eltringham 1982; Ruggiero 1992). Conversely, Sukumar, (1989) found that in the Asian elephant in southern India, 69% of the elephants diet in the dry season was made up of browse, then in the first wet season grass made up a higher proportion of the diet (54%), and in the second wet season browse returned to being the higher proportion of plants in the diet (56%). Guy, (1976) found that over the whole year elephants in Zimbabwe consumed more browse than grass in the proportion of 1:18, but in the wet season more time was spent grazing than browsing.

Osborn, (2004) found that there was a link between the onset of crop-raiding in Zimbabwe and the reduction in quality of wild grass toward the end of the wet season. He surmised that the decline in quality of wild grass could be a trigger for crop-raiding, and showed that crops could be favourable to elephants due to them being low in fibre, high in moisture and high in crude protein.

1.4.4. Habitat use and home ranges

Elephants can subsist in a variety of habitats including dense forest, open and closed savannah, grassland and, at considerably lower densities, in the arid deserts of Namibia and Mali (Blanc *et al.* 2007), provided there are sufficient quantities of food and water. In all these ecosystems elephants have the ability to significantly modify their habitat, and thus play an important role in structuring and reshaping vegetation (Laws & Johnstone 1970), facilitating browsing and grazing of smaller herbivores (Owen-Smith 1988), dispersing seeds and fruits (Laws, Parker & Johnstone 1975; Western, Moss & Georgiadis 1983; Ruggiero 1992), and in nutrient cycling (Ruggiero 1992) . At high densities elephants can convert woodland to shrub land by eating from, and killing, large numbers of trees and bushes, (Ben-Shahar 1993; Styles & Skinner 2000; Mosulego *et al.* 2002), however, when elephant numbers are reduced in an area this can lead to an increase in bush cover, as occurred in Queen Elizabeth National Park in Uganda (Lock 1993). At low densities elephants are thought to promote plant species richness, restrict bush encroachment (Western 1989) and have an important effect on nutrient cycling, making nutrients found in woody material available to other species (Anderson & Walker 1974).

A combination of biotic and abiotic factors may determine the distribution patterns of large herbivores in savannah ecosystems, (Redfern *et al.* 2003). For elephants, seasonal differences in habitat use have been attributed to water availability, as well as palatability, availability and nutritional quality of food (Sommerlatte 1976; Jachman 1984; Sukumar 1990; Viljoen & Bothma 1990; Leggett 2006a; Shannon *et al.* 2006) and mineral availability (Weir 1972). In Pongola National Park, South Africa for example, elephants strongly selected *Acacia*-Marula woodland in the wet season, whereas during the dry season habitats in lower – lying regions were selected, reflecting the nutritional quality of forage. Elephants have been found to avoid cultivated land and unprotected areas, while favouring rivers, bushed grassland, forest and woodland (e.g. Foley 2002; Hien 2005; Wittemyer *et al.* 2007; Mpanduji, East & Hofer 2008). It is also thought that during the dry season dominance hierarchy between families affect habitat use, with dominant groups disproportionately using preferred habitats i.e. avoiding unprotected areas and human habitations, however, such behaviours were not recorded during the wet season when resource availability was higher, (Wittemyer *et al.* 2007).

Home range sizes of elephant vary throughout the elephant range, and appear to differ for female and bull herds. Bull and cow herds typically have larger home range sizes in the wet season than in the dry season (e.g. Moss & Poole 1983; Stokke & duToit 2002; Sukumar 2003; Charif *et al.* 2005; Chase & Griffin 2007; Young, Ferreira & van Aarde 2009), however, some studies have found home range sizes increase in the dry season (e.g. deVilliers & Kok 1997; Chase & Griffin 2008) and others have found no apparent seasonal variation (e.g. Fernando *et al.* 2008).

Home range sizes can also vary considerably within a region, for example Chase & Griffin, (2007) found that bull elephant home ranges in northern Botswana, ranged from <185km^2 to as large as 24,828 km^2. It appears that elephants in more arid regions, like Botswana, Mali and Namibia exhibit larger home ranges, (e.g. Blake *et al.* 2002; Leggett 2006b; Chase & Griffin 2008), which could be a reflection of quantity and quality of vegetation. For example, cow home ranges have been recorded as small as 15km^2 in Tanzania (Douglas-Hamilton 1972) to as large as 5,860km^2 in arid regions of Namibia (Lindeque & Lindeque 1991). It is also apparent that bull home ranges are typically larger than females (Sukumar 1989; Chase & Griffin 2007; Fernando *et al.* 2008), for example in northern Botswana the average cow home range was estimated at 4,653km^2 compared to the bull average of 9,075km^2, (Chase & Griffin 2008). Home ranges for African forest elephants appear to have a smaller range (785-2,534km^2), (Tchamba 1995; Blake *et al.* 2002) than savannah elephants (15-24,828km^2), (Douglas-Hamilton 1972; Leuthold 1977; Dunham 1986; Viljoen & Bothma 1990; Lindeque & Lindeque 1991; Thouless 1996; Blake *et al.* 2002; Charif *et al.* 2005; Douglas-Hamilton, Krink & Vollrath 2005; Galanti *et al.* 2005; Leggett 2006b; Chase & Griffin 2007; Dolmia *et al.* 2007; Ngene *et al.* 2009), which is probably a reflection of the area of available habitat.

1.4.5. Movements

Seasonal movements of elephants appear to be strongly influenced by water availability, with daily movements in the dry season being larger than in the wet seasons, 6km/day and 3km/day respectively, (Loarie, van Aarde & Pimm 2009a; Loarie, van Aarde & Pimm 2009b). In certain areas, seasonal movements are predictable, while in others, movement patterns are far more difficult to decipher, however, it has been found that elephants can move large distances in search of resources such as food, water and minerals or in response to disturbance (Blanc *et al.*

2007). For example, one cow herd moved 80km in 24-36hr in the Kunene region, Namibia to access resources, and in the same study one elephant's cold dry season movement distance was approximately 625 km, reflecting its large movement path to find sufficient habitat requirements (Leggett 2006b). Chase & Griffin, (2007) recorded movements of 350km in areas around artificial waterholes in Botswana, and Chamaillé-Jammes, Valeix & Fritz, (2007), also found that surface-water availability in Hwange National Park, Zimbabwe is a key driver of elephant distribution and abundance. Several studies have linked elephant movements with vegetation type and density (Merz 1986; Barnes *et al.* 1991; Verlinden & Gavor 1998), and elephants have even been found to travel over 50km to exploit seasonal fruits (White 1994). Fences appear to restrict elephant movement, especially family groups, (Albertson 1998; Chase & Griffin 2007; Loarie, van Aarde & Pimm 2009a), causing animals to bunch up along fences.

Elephants tend to be more active at night, when temperatures are lower and often frequent water points at night time (Foley 2002; Loarie, van Aarde & Pimm 2009a). Interestingly, it has been found that the speed of elephant movement also varies in relation to resource use and behaviour. In their study in Kenya, Ngene *et al.*, (2009) found that speeds <0.2km/hr were associated with foraging or resting; faster speeds of 0.2-1km/h were associated with areas of limited forage or movements between habitat types; and consistently high speeds of >1km/hr were associated with movements from water to foraging sites through elephant corridors. Foley, (2002) found similar patterns in elephant movement in Tanzania, where elephants moved four times faster through migration corridors outside the park than inside the protected area. This difference in elephant activity could be due to presence of human activities outside the park.

It is clear from reviewing the literature on elephant ecology that elephants can move vast distances in search of food and water, potentially requiring a large home range subject to the availability of such resources. One of the main underlying causes of conflict between people and elephants is consequently competition for space (Hoare 2000b) and therefore one long term mitigation strategy to reduce conflict could be to develop more effective land use planning, e.g. zoning and/or fencing certain areas (e.g. Mishra 1971; Taylor 1993). However, to enable such strategies to be developed and successfully implemented a greater understanding of the spatial use by elephants and people is required.

1.5. Land use

Two main approaches have been used by land use planners to try and minimise potential conflicts between people and wildlife over land use, namely i) the multi-use concept where compatible land uses can occur in the same area and ii) zoning. Zoning has been widely used in biodiversity conservation in the creation of national parks, nature reserves and other protected areas (Linnell *et al.* 2005). Theoretically though, large mammal populations are best conserved in landscapes where large protected areas are surrounded by buffer zones, connected by corridors and integrated into a greater ecosystem (Nyhus & Tilson 2004a). Land use planning for conservation landscapes, to take conservation beyond the boundaries of protected areas is therefore important for the conservation of wide ranging mammals like elephants (Noss *et al.* 1996; Wikramanayake *et al.* 1998; Hoare 2000a; Fernando *et al.* 2005), especially as a large portion (~69%) of known African elephant range exists outside of protected areas (Blanc *et al.* 2007). As a result, correctly managed buffer zones or adaptive management areas around protected areas may be as important as wildlife reserves to the long-term viability of wide-ranging species (Noss *et al.* 1996). Such a conservation strategy, however, would require creating zones within a multi-use landscape to try and minimise negative interactions between humans and wildlife (Fernando *et al.* 2005; Linnell *et al.* 2005).

To design effective land use planning to try and minimise HWC, it is imperative to understand how both animals and people utilise space and resources. For example, Treves *et al.*, (2004) found that wolves appeared to prey on livestock where there were high proportions of pasture but low proportions of crop land, coniferous forest, herbaceous wetlands and open water. Therefore management actions to reduce human-wolf conflicts could be targeted in areas with low proportions of crop-land, forest, wetlands, open water or high proportions of pasture where human-wolf conflict was likely to be high. Similarly, Fernando *et al.*, (2005) found that land use patterns and agricultural practices influenced intensity of human–elephant conflict in Sri Lanka. With a fragmented mosaic of small forest patches (protected areas) utilized by elephants scattered throughout a human-dominated landscape of irrigated agriculture, exacerbating HEC, yet, adaptive management (common-use) areas managed according to traditional agricultural practices provided essential resources to elephants, and allowed coexistence of humans and elephants through temporal and spatial resource partitioning.

Wittemyer, *et al.,* (2007) showed that elephant spatial organization appears to be dynamic and subject to the unpredictability of their food distribution. Spatial use strategies by elephants are therefore adaptable to local conditions, as variations in home range size illustrates, which is a characteristic of migratory species. A variety of local factors have been identified that appear to influence home range size, movements and distribution of elephants, including: water, food and mineral availability; habitat heterogeneity; sex; age; kinship; social behaviour; and reproductive status (Hanks 1979; Sukumar 1989; Lindeque & Lindeque 1991; Datye & Bhagwat 1995; Thouless 1996; deVilliers & Kok 1997; Sukumar 2003; Osborn 2004; Charif *et al.* 2005; Grainger, van Aarde & Whyte 2005; Murwira & Skidmore 2005; Shannon *et al.* 2006; Wittemyer *et al.* 2007; Chase & Griffin 2008; Fernando *et al.* 2008). Ultimately though, home range and distribution of elephants are affected by the quality and distribution of resources associated with seasonal fluctuations in the ecosystem, for example rainfall (Sukumar 2003; Osborn 2004; Leggett 2006b; Woodroffe *et al.* 2007a; Chase & Griffin 2008). The fragmentation of large contiguous elephant habitats, as a result of developments such as human settlements, fences, and roads, have also been found to greatly influence elephant movements and home range sizes (Datye & Bhagwat 1995; Hoare 1999a; Guldemond & van Aarde 2008; Chase & Griffin 2009; Loarie, van Aarde & Pimm 2009a), highlighting one effect human-elephant coexistence can have on elephant populations whilst simultaneously illustrating the adaptability of these animals.

Allocation of land for human use is typically determined by soil fertility, with the most fertile soils being dedicated to agriculture and livestock production, and the least fertile soils to non-agricultural uses (Martin & Taylor 1983; Happold 1995). A full understanding of how people choose land and utilise resources in an area experiencing human-wildlife conflict is essential to determine how land use planning can be improved in the future to minimise conflicts and increase the likelihood of coexistence. In Botswana, land is divided into three land tenure systems namely, Customary (72%), State-land (23%) and Freehold (5%). Customary land is allocated under the Tribal Land Act (Government of Botswana 1968) and every Batswana citizen is entitled to a residential and agricultural plot. State land and freehold land regulations are governed by the President under the State Land Act (Government of Botswana 1966), (Mathuba 2003). Land is zoned into different land use categories including communal areas, game reserve and national parks, and wildlife management areas (WMA). The communal areas are further subdivided into settlements, arable lands and grazing areas and the WMAs can be classified as either consumptive or non-consumptive wildlife utilization areas (Magole 2009).

Agricultural land is chosen by a farmer based on their own preferences such as soil quality, vegetation, distance to water or distance to a village. The Land Board then assesses applications, conducts a site visit and has the authority to recommend or reject customary land applications. On site visits, plot boundaries are demarcated and the area of land measured. The distance of the plot to the nearest river is also measured, all plots are required to be <200m from the river. A consultation process is an important part of the land allocation procedure in Botswana to ensure agreement from neighbouring land owners and community members (Mathuba 2003). Recently, Land Board have started to consider wildlife migratory pathways when assessing land, and applications can be refused if such paths are present on or close to the land to try and minimise conflicts between people and wildlife (e.g. Tawana Land Board 2005). Options for zoning within a multi-use area, such as a WMA, to reduce HWC, however, does not appear to have been explored.

In chapter five of this thesis I explore how people and elephants are currently using the land in the Okavango Delta panhandle and how human habitat modifications affect elephant movements, in order to propose future ways of improving land use planning to try and minimise HEC.

1.6. Quantifying HWC

To gain a comprehensive understanding of human wildlife conflict situations it is imperative to a) know what the current HWC situation is and be able to monitor conflict incidents efficiently; and b) be able to identify patterns (spatial and temporal) or key drivers influencing patterns, which can help in predicting when and where conflict incidents may occur in the future.

Studies investigating human-wildlife conflict incidents have been widely documented, yet methods to collect information range from descriptive reporting (Waithaka 1993; Kiiru 1995; Ngure 1995) to measuring and quantifying actual damage (e.g. Naughton-Treves 1998; Osborn 1998a; Hoare 1999b, a; Smith & Kasiki 1999; O'Connell-Rodwell *et al.* 2000; Tourenq *et al.* 2001; Sitati *et al.* 2003; Barnes 2005; Chiyo *et al.* 2005; Sam *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Barnes 2007; Warren, Buba & Ross 2007). Several studies have focused on the human element of HWC and used questionnaire surveys to estimate stakeholder perceptions towards HWC and estimate perceived damage (e.g. Conover & Decker 1991; Lahm 1996; O'Connell-Rodwell *et al.* 2000; Saj, Sicotte & Paterson 2001; Marker, Mills & Macdonald 2003; Arlet & Molleman 2007; Bisi *et al.* 2007; Hegel, Gates & Eslinger 2007; Hill, Carbery & Deane 2007; Karlsson & Sjostrom 2007; Cai *et al.* 2008) while others have attempted to quantify damage in economic terms (Conover 1997, 2002; Nyhus, Osofsky & Ferraro 2005; Holmern, Nyahongo & Roskaft 2007; Rondeau & Bulte 2007). In many countries monitoring of HWC is undertaken by government wildlife management authorities, however, more recently emphasis is being placed on involving community members from areas where HWC is occurring in monitoring activities. This takes the form of training community enumerators or wildlife scouts to collect data on conflict incidents (e.g. Hoare 1999a; Hockings & Humle 2009). Such systems, where community members are involved in wildlife management and HWC monitoring appear to be effective, and have been used in previous studies (Osborn 1998a; Hoare 1999b; Sitati, Walpole & Leader-Williams 2005; Hockings & Humle 2009) as well as implemented in community conservancies in Namibia through the support of a non-governmental organisation (NGO) Integrated Rural Development & Nature Conservation (IRDNC) (IRDNC 2004).

In chapter three of this thesis I explore and compare insights from a government "top-down" approach to HEC monitoring with a community based approach.

1.7. Identifying patterns and contributing factors of HWC

Numerous studies have been conducted to explore temporal and spatial patterns of conflict incidents, which can help in identifying high intensity ‘hotspot’ areas of conflict and aid in predicting where and when future incidents of HWC are likely to occur. Research into patterns of livestock depredation by carnivores, for example, has identified factors such as husbandry practices, human activities, habitat type and predator behaviour increasing the risk of conflict between people and carnivores (Linnell *et al.* 1999; Ogada *et al.* 2003; Treves *et al.* 2004; Woodroffe *et al.* 2007b). From a comparison of 28 studies however, Graham, Beckman & Thirgood, (2005) identified that livestock losses to predators were not affected by predator density, methods of husbandry or human population density but were negatively associated with net primary productivity and predator home range sizes. These studies illustrate that there appear to be no identifiable patterns in livestock depredation by carnivores on a general scale, but rather patterns can be identified for specific species or areas.

Studies conducted on a variety of wildlife species causing crop destruction have investigated factors and determined patterns which may help in predicting such cases (Tourenq *et al.* 2001; Treves *et al.* 2004; Chiyo *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Woodroffe, Thirgood & Rabinowitz 2005; Cai *et al.* 2008). Common themes arising from these studies indicate that temporal and spatial patterns of wildlife crop-raiding appear to vary across space and time (Goldman 1986; Dudley, Mensah-Ntiamoah & Kpelle 1992; Hillman-Smith *et al.* 1995; Lahm 1996; Naughton-Treves 1998; Weladji & Tchamba 2003; Warren, Buba & Ross 2007) and may be influenced by animal density (Naughton-Treves 1998; Hegel, Gates & Eslinger 2007), area of cultivated land (Tourenq *et al.* 2001), and/or the location of fields in relation to landscape features such as water availability, protected areas, and wildlife habitat, (Lahm 1996; Hill 1997; Naughton-Treves 1998; Smith & Kasiki 1999; Hill 2000; Saj, Sicotte & Paterson 2001; Linkie 2007; Cai *et al.* 2008). Other factors, such as, season, amount of guarding, human density, hunting, isolation of fields and crop types available, have also been found to effect the degree of damage a field receives (Atteh 1984; Newmark *et al.* 1994; Naughton-Treves 1998; Hill 2000; Warren, Buba & Ross 2007).

Foraging strategies or preferences of wildlife may also influence patterns of crop damage incidents, (Dudley, Mensah-Ntiamoah & Kpelle 1992; Lahm 1996; Naughton-Treves 1998; Naughton, Rose & Treves 1999; Weladji & Tchamba 2003; Warren, Buba & Ross 2007), with certain animal species appearing to favour different crop types and thus making these crops more susceptible to being raided (Sekhar 1998). A study in Gabon, identified that elephants damaged and destroyed banana plants more often than other crop types, (Lahm 1996). Bush pigs appear to prefer cassava and yams, and rodents are significant pests of rice (Goldman 1986; Adesina, Johnson & Heinrichs 1994; Hill 1997; Naughton-Treves 1998). Common staple food crops and crops with economic value are often said to receive the greatest losses (Putman 1989; Conover & Decker 1991). It has also been found that some species raid crops when the quantity or quality of natural forage is low (Naughton-Treves 1998; Sekhar 1998; Osborn 2004), and raiding often occurs when crop availability is high and may be determined by the seasonal ripening of nutritious crops (Sukumar 1989; Naughton-Treves 1998; Sekhar 1998; Gillingham & Lee 2003; Chiyo *et al.* 2005; Linkie 2007).

Male elephant herds have been found to more frequently crop raid than female led groups (Bell 1983; Sukumar & Gadgil 1988; Sukumar 1991; Thouless 1994; Osborn 1998a; Hoare 1999b; Sitati *et al.* 2003; Chiyo & Cochrane 2005; Chiyo *et al.* 2005). Sukumar, (1991) surmised that one explanation for this could be that male elephants have a greater inclination to take risks to derive better nutrition for enhancing reproductive success (male behaviour hypothesis), and such behaviour is therefore consistent with predictions of the optimal foraging theory (Krebs & Davies 1991; Hoare 1999b). Other crop-raiding animals have also been found to raid in accordance with optimal foraging models, and Amano *et al.*, (2007) propose using models that consider foraging goals and constraints for identifying measures for minimizing agricultural damage problems, particularly those caused by grazing bird species such as geese. Sitati *et al.*, (2003) showed that male elephant behaviour in crop raiding incidents is less easy to predict than female led family groups, which could be attributed to this male behaviour hypothesis or may be due to elephant ecology in specific areas as Arlet & Molleman, (2007) surmised. Elephants appear to select forage based on nutritional value, palatability and mineral content (Sukumar 1990; Osborn 1998a). In a study in Zimbabwe, Osborn (1998b) identified that elephants started to crop raid when the quality of wild grasses began to decline in their natural habitats. He surmised that when grass quality and availability begins to decline, it becomes worth the risks associated with raiding crops and this is the trigger for crop-raiding.

In chapter four, I conduct a comparative analysis of raided and non-raided fields and use generalised linear models to explore key drivers of elephant crop raiding.

1.8. Human element

Ultimately, there are two sides to the story in all situations of HWC, and it is therefore essential to evaluate both sides (wildlife and human) in order to devise successful and acceptable solutions. Many studies addressing situations have concentrated on the human side of HWC, investigating perceptions and attitudes of people, effects on humans and the socio-economic implications of the conflict (Gillingham & Lee 2003; Hussain 2003; Hill 2004; Okello 2005; Warren 2005; Bagchi & Mishra 2006; Linkie 2007). From a human perspective HWC can include crop destruction, livestock depredation, property destruction and threat to human lives. Such situations can be complex and may also include an element of human-human conflict, encompassing damages to the individual that result from international, national, or local wildlife legislation, regulations, or policies that are designed to protect or conserve wildlife, public benefits, and individual property rights (Messmer 2000). As well as there being direct "conflicts" (real) as outlined above some negative human-wildlife interactions may be perceived (or at least the extent of the situation exaggerated), others may be economic or aesthetic, and some social or political (Messmer 2000).

1.8.1. Perceptions

It is important to understand whether encounters with wildlife are experienced as positive or negative by different people in an area in order to formulate effective management strategies for wildlife species (Warren, Buba & Ross 2007). For example, people living in close proximity to wild elephants often view them as a threat to their livelihoods and indeed their lives (Smith & Kasiki 1999), many people advocate elephant numbers being reduced (Smith & Kasiki 1999) and unless direct benefits are apparent may also resent protected areas and conservation policies (Western & Wright 1994; Smith & Kasiki 1999; Songorwa 1999; O'Connell-Rodwell *et al.* 2000). An increase in HWC can then lead to an increase in negative perceptions towards wildlife and conservation initiatives, and thereby hinder management and conservation strategies. Peoples' attitudes towards wildlife are complex, with social factors as diverse as religious affiliation, ethnicity and cultural beliefs all shaping conflict intensity (Knight 2000; Dickman 2010).

It is imperative therefore, as Hill, (Hill 2004) outlines, to explore people's perspectives of human–wildlife conflicts because perceptions and expectations shape attitudes and responses to damage by wildlife. For example fear of certain wildlife species can often influence attitudes towards them, (Kaltenborn, Bjerke & Nyahongo 2006), with some animals e.g. livestock considered less of a threat or more acceptable when inflicting damage than others (Warren, Buba & Ross 2007). Understanding how people perceive risks is valuable in examining the complex nature of human–wildlife interactions and can help inform our understanding of conflict (Dickman 2010). Tolerance of wildlife species and support for wildlife conservation can also be influenced by benefits received from living close to wildlife (Okello 2005; Holmern, Nyahongo & Roskaft 2007), and it was found that tolerance of livestock depredation increased with level of education in Tanzanian livestock owners (Holmern, Nyahongo & Roskaft 2007). However, factors such as the extent of losses incurred from problem animals, lack of compensation for these losses, and lack of community involvement in wildlife conservation can increase local resentment (Okello 2005). Less tolerance/greater resentment may result in direct action towards human-wildlife conflict including retaliatory killings by farmers. This in turn can increase pressure on wildlife populations especially those which are already threatened, such as snow leopards (Hussain 2003).

It has also been found that people often over-report the scale of an HWC problem, leading to a mismatch between local perceptions of wildlife damage and its actual occurrence (Gillingham & Lee 2003; Linkie 2007). For example, crop damage estimates based on farmers' perceptions in Tanzania over reported the scale of the problem compared to the independent monitoring data collected (Gillingham & Lee 2003). Such exaggeration can therefore lead to a biased view of the situation and blow the extent of the conflict out of proportion (Sekhar 1998; Siex & Struhsaker 1999; Gillingham & Lee 2003; Linkie 2007). Nevertheless, even if results show actual damage from HWC incidents is low in an area, as Warren, Buba & Ross, (2007) highlight this does not automatically provide evidence that people and wild animals should be able to co-exist harmoniously. It is important to consider that individual losses to people can be very serious even in “low level” HWC areas and this issue also needs to be addressed when forming conservation policy and education initiatives.

Evidently, there is a complex interaction between cultural, social and personal factors which ultimately determine how costly living close to certain wildlife species is perceived to be and what level of antagonism is felt towards them, and consequently what actions people take against these animals (Dickman 2010). A lack of understanding of the ecological and social issues of human–wildlife conflicts, however, often hampers the formulation of effective conflict resolution and conservation management strategies (Bagchi & Mishra 2006). It appears that it is, therefore, vital that an interdisciplinary approach encompassing both ecological and social aspects (e.g. Hotte & Bereznuck 2001; Ogada *et al.* 2003; Nyhus & Tilson 2004a), be used to develop appropriate conflict management strategies (Treves & Karanth 2003).

1.8.2. Costs of HWC

Protecting wildlife habitats or conservation areas has the potential to bring benefits to local communities and help diversify economies. However, where human-wildlife conflict occurs there are also costs involved. Such costs can be classified into two categories: Direct effects, including loss of crops, damage to property, loss of human life, injuries to humans, loss of livestock; and indirect effects, such as competition for natural resources, sleepless nights, restricted movements, and disruption to daily routine (Conover 1997).

Ideally, the costs of HWC can be reduced by mitigating the amount of damage accrued through HWC incidents. However, another approach to decreasing the costs involved with living close to wildlife is to increase the benefits people receive from living close to protected areas or wildlife (Thouless 1994). This could be through community based conservation projects, whereby communities receive income from tourism or other wildlife related activities in the area (Newmark & Hough 2000; Johannesen & Skonhoft 2005) or through incentive schemes (Mishra *et al.* 2003) such as government (DWNP 2010), or NGO compensation (IRDNC 2004) or insurance (Stewart & Diggle 2004). Support for wildlife conservation is often dependent on benefits received by local stakeholders, (Okello 2005), therefore, as Johannesen (2007) demonstrated, it is crucial for conservation programmes to forge a link between benefit levels and conservation friendly behaviour in order to improve both wildlife conservation and human welfare.

Experience from community-based conservation projects, however, show that distribution of benefits can be problematic and such strategies do not necessarily reduce HWC or improve conservation (Newmark & Hough 2000; Johannesen & Skonhoft 2005). Often, benefits from wildlife are far less than the economic loss of HWC to people. For example a study investigating livestock depredation in Serengeti National Park, Tanzania, found that benefits accrued to people from predators in the national park were greatly inadequate compared to the losses incurred through livestock depredation (Holmern, Nyahongo & Roskaft 2007). However, implementing incentive schemes aimed at conserving endangered animals can work, as encouraging results reported by Mishra *et al.*, (2003) for snow leopard show. Compensation schemes though, unfortunately, have a number of problems associated with them as well. For example they may decrease the motivation to defend crops or livestock (Hoare 2001; Nyhus, Osofsky & Ferraro 2005; Warren, Buba & Ross 2007), or people may overstate damages to gain additional income (Schwerdtner & Gruber 2007), or they may give an incentive to people to convert land for agriculture use, leading to the migration of people into wildlife areas and thereby increasing potential for HWC to arise (Bulte & Rondeau 2005; Rondeau & Bulte 2007).

Where human-wildlife conflict occurs in areas with low income or where people are dependent upon a single livelihood strategy, potential consequences of HWC incidents are intensified by a lack of alternative assets or income strategies, which can exacerbate the antagonism of people towards conflict causing animals (Naughton-Treves & Treves 2005; Dickman 2010). If a person is wealthy, has alternative sources of income and/or engages in social reciprocity with their family and community, then it is surmised that they could be less vulnerable than other people (Naughton-Treves & Treves 2005).

The current African elephant range falls within countries classified in the low to upper-middle income classification categories and as such are often referred to as “developing countries” or “nations with a low level of material well-being”, (UNSD 2010). Livelihoods of people living within the African elephant range are often restricted, with poorer people frequently relying on natural resources (Newmark *et al.* 1994; DeBoer & Baquette 1998). Such communities are therefore likely to be more vulnerable to the consequences of conflicts with elephants and therefore require adequate coping strategies (either improved mitigation to decrease costs or increased benefits of living close to elephants) to reduce such vulnerability.

In chapter six I explore the complex social drivers of human-elephant conflict through structured interviews and ordered probit model analysis.

1.9. How to address HWC

It is evident that one of the most significant challenges for conservation in the modern world is managing situations where people and wildlife utilise the same space and compete for similar resources (Balmford *et al.* 2001; Sitati, Walpole & Leader-Williams 2005; Woodroffe, Thirgood & Rabinowitz 2005; Dickman 2010). Fall & Jackson, (2002) envisioned a strategy to deal with wildlife conflicts in which the first step involves assessing the animal, human, and environmental aspects of wildlife damage problems, followed by development of knowledge based decision models, implementing management actions, and then monitoring outcomes to facilitate ongoing problem management. They suggest that multi-scale, multi-factor analyses should allow for more complex and competent management and better coordination of effort between agriculturalists, wildlife, land and public refuge managers, and managers of damage mitigation and compensation programmes. Dickman, (2010) reiterates the need to use a multidisciplinary approach to conflict management due to the complexities of human attitudes towards animals, and cautions against making assumptions about human attitudes and behaviour when deciding how to tackle conflict.

Current approaches used to address human-wildlife conflicts can be grouped into three broad categories: habitat modification, altering the size or behaviour of wildlife populations, and altering human attitudes or behaviours (Conover 2002; Hegel, Gates & Eslinger 2007). Often, wildlife management authorities and community members tackle wildlife crop damage by using short term mitigation measures, (Conover 2002). Such methods are useful as immediate defensive measures to protect fields from attack (Osborn & Parker 2003) and in reducing negative perceptions of wildlife (Hill 2000), however, they do not address the underlying causes of the problem (Jackson *et al.* 2008). As discussed above, the extent of negative interactions between humans and wildlife are affected by different elements: perceived and actual costs; human response to these costs; and consequences to wildlife of these responses, with elements affecting the intensity of HWC situations involving both environmental and social risk factors (Dickman 2010). Three key factors were highlighted by Dickman, (2010) to consider on a local level before deciding which mitigation strategies are likely to be most successful in any HWC situation: perceptions of risk, disproportionate responses and social influences.

Chapter 2

Abundance and growth in a potentially “trapped” elephant population in the Okavango Delta Panhandle, Botswana

2.1. Introduction

The ability to reliably estimate population numbers and densities of animals is a fundamental component of many ecological studies (Elphick 2008) and has a prominent role in the conservation and management of wildlife. In general, it is not possible to count all the individuals in a population with certainty, especially when animals are free ranging over a vast area. It is, therefore, necessary to sample the population using methodology that allows robust inferences to be made about the entire population from the observed sample (Milner-Gulland & Rowcliffe 2007). Wildlife surveyors need to decide how much bias is acceptable and how precise the estimates of population abundance and density need to be for the relevant wildlife management applications (Hone 2008). Simulations could provide a cost effective method to do this.

An array of sampling techniques have been used to estimate wildlife population density, abundance and distribution (e.g. Caughley 1976; Norton-Griffiths 1978; Buckland *et al.* 1993; Khaemba *et al.* 2001; Redfern *et al.* 2002; Pollock *et al.* 2006; Jackson *et al.* 2008; Chase & Griffin 2009; Pearse *et al.* 2009) varying from transect (strip, point or line) surveys (e.g. Norton-Griffiths 1978; Buckland *et al.* 1993) to mark-recapture methods (e.g. Jolly 1965; Hargrove & Borland 1994; Southwell & Low 2009). Finite population sampling methods such as total counts and strip transects, are often used to survey wildlife populations (e.g. Caughley & Grigg 1981; Marsh, Harris & Lawler 1997; Redfern *et al.* 2002; DWNP 2006; Jackson *et al.* 2008; Chase & Griffin 2009). Aerial transect surveys are the most commonly used method for surveying large mammals where the terrain is difficult or inaccessible, or where large areas need to be surveyed. Such surveys are used over a range of habitat types, including African savannah (e.g. Redfern *et al.* 2002; Chase & Griffin 2009; Ferreira & van Aarde 2009), oceans (e.g. Baumgartener 1997; Marsh, Harris & Lawler 1997; Wright *et al.* 2002; Marsh *et al.* 2004; Pollock *et al.* 2006), the Australian pastoral zones (e.g. Caughley & Grigg 1981) and the arctic tundra (e.g. Rivest *et al.* 1995). All these types of surveys encompass a significant economic cost and, therefore, verification of sampling designs by repeat surveys is often not feasible.

Even when sound survey methodology is adhered to, it is generally accepted that most aerial surveys are subject to a number of potential sources of bias (Caughley 1974; Marsh & Sinclair 1989; Jachman 2002; Elphick 2008; Laake, Dawson & Hone 2008). Such sources of bias include: Survey method bias (controllable by correcting for height variation and transect width variation); observer effects (minimized by training observers and using photo-corrected observations); environmental variables i.e. rainfall (uncontrollable); species-specific characteristics i.e. habitat preference, herd cohesion, herd size, species colouration (uncontrollable), and; undercounting bias (Caughley 1974; Norton-Griffiths 1978; Samuel & Pollock 1981; Milner-Gulland & Rowcliffe 2007; Elphick 2008; McIntosh *et al.* 2009). For example, in transect survey methods, population variance is often used as a measure of precision of the density estimate in such surveys, (Norton-Griffiths 1978). Animal populations are, however, often aggregated due to environmental and behavioural factors, which contradict the fundamental assumption of this precision estimate that all individuals are independently and randomly distributed within the sampling plot (Milner-Gulland & Rowcliffe 2007).

An accurate estimate is one that is near to the true total but may have wide confidence limits. Alternatively, a precise estimate has narrow confidence limits but the population estimate itself may be biased (Norton-Griffiths 1978). Precise censuses are needed to follow population trends, but the repeatability must be high (i.e. the degree of bias remains constant from census to census). On the other hand, accurate estimates are required, for example, if a population is to be reduced or if biomass estimates are being calculated (Norton-Griffiths 1978; Ferreira & van Aarde 2009). Choosing an appropriate sampling strategy is, therefore, critical because it influences precision and bias of estimated parameters and, therefore, may determine whether objectives are attained (Pearse *et al.* 2009). Assessing the efficiency of sample designs can be difficult in field studies because the tests themselves are subject to similar biases and direct empirical comparisons can be costly (Khaemba *et al.* 2001; Jackson *et al.* 2008; Pearse *et al.* 2009). Simulation is, therefore, a viable alternative to compare and validate different survey techniques applicable to aerial surveys of animals and has proven useful for such surveys (e.g. Khaemba *et al.* 2001; Ferreira & van Aarde 2009; Pearse *et al.* 2009).

The largest contiguous population of African elephants (*Loxodonta africana*) is estimated at 218,091 elephants (Blanc *et al.* 2007), occurring across five countries, Angola, Botswana, Namibia, Zambia and Zimbabwe (Chase & Griffin 2006), with a total range of approximately

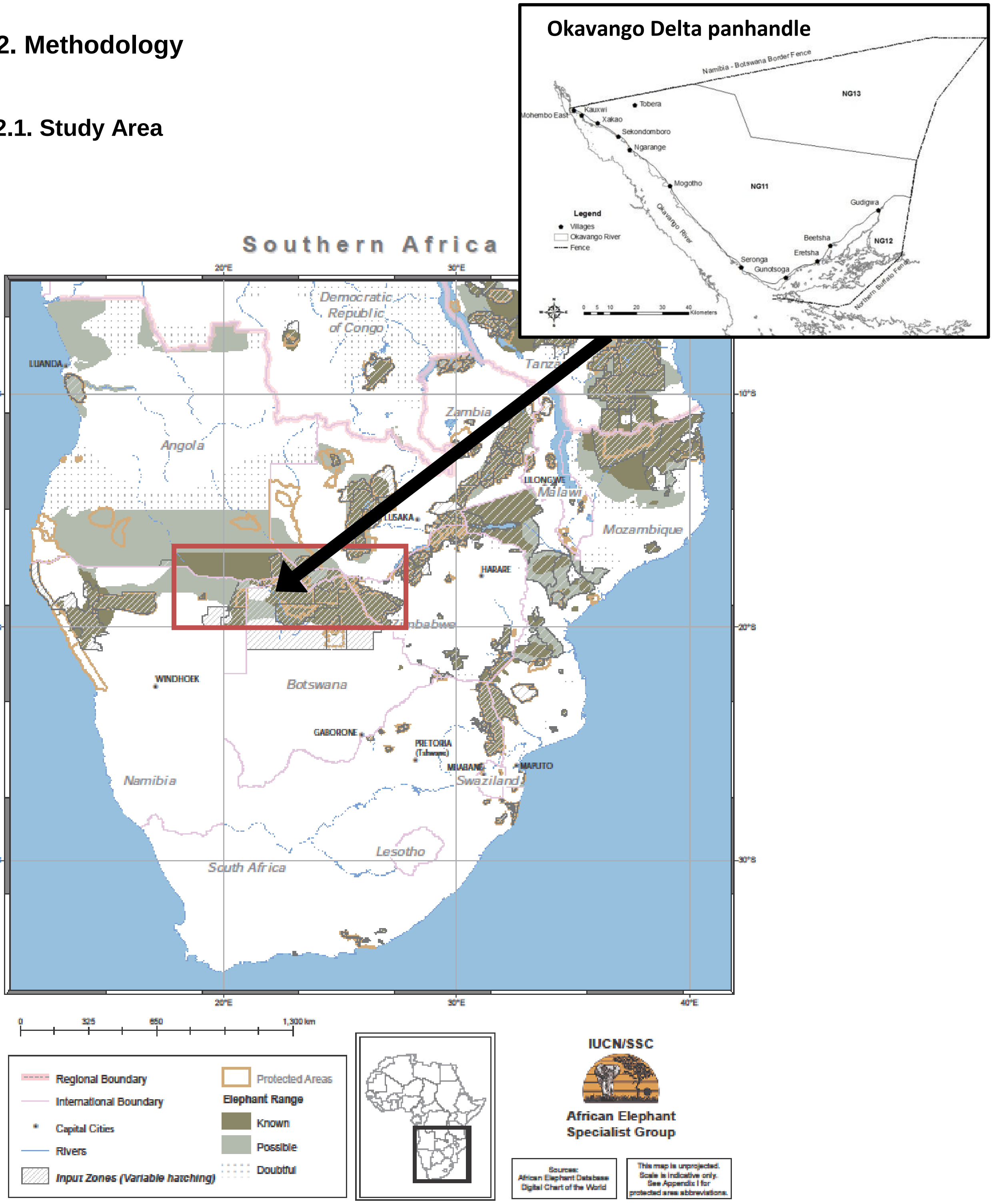
187, 220km^2, (Blanc *et al.* 2007). This total estimate consists of results from twenty-one separate surveys, which were estimated from sampled sub populations living in separately defined geographic areas. Extrapolating survey results to larger areas is a useful tool in wildlife management and conservation initiatives, but it is important to understand the bias involved with such results.

This study took place in the Okavango Panhandle, Botswana, an area that forms a central part of the contiguous elephant population range across the Kavango region. The eastern panhandle study site boundaries (i.e. Namibian border fence, northern buffalo fence and the permanent Okavango River) are potential barriers to elephant movements (Chase & Griffin 2007, 2009; Ferreira & van Aarde 2009), however, indicating that elephants could effectively be trapped in this area. Over fifteen thousand people are also living in the eastern panhandle, (CSO 2001), sharing and competing for resources with this elephant population. Yet, it is unclear how many elephants occur here or how fast the population is growing. Previous aerial surveys have been conducted in the area by Department of Wildlife and National Parks (DWNP) during 1996- 2004 and Jackson *et al*. (2008) in the dry season 2003 and wet season 2004. Population numbers and density estimates that have resulted from these surveys show large fluctuations, which are not biologically possible. Reasons for such variations could be attributed to either survey/observer/sampling bias or because only part of the elephant range is being surveyed.

This paper aims to a) estimate current population numbers and densities of elephant in the panhandle region; b) determine the elephant population growth rate in the area; c) use simulation to explore the reliability of past and present survey results; and d) explore the effect of spatial scale in the precision of aerial surveys.

2.2. Methodology

2.2.1. Study Area



Source: African Elephant Specialist Group, IUCN, Gland, Switzerland

Figure 1. Elephant range in southern Africa with the Kavango region highlighted in red box. The Okavango panhandle survey areas (NG11, NG12 and NG13) are enlarged and fences potentially restricting elephant movement are marked with dotted line

The study was conducted within the Kavango region of southern Africa, on the eastern side of the Okavango Panhandle, where the Okavango River reaches the Okavango Delta in Botswana. The area encompasses three controlled hunting areas (CHAs), namely NG11, NG12 and NG13. The Namibian border marks the northern boundary, while the northern buffalo fence marks the southern and eastern boundary, and the Okavango River the western boundary (UTM Zone 34 7910000 – 7990000 South and 580000 – 710000 East) (figure 1).

Deep Kalahari sands dominate throughout NG11 and NG13, and main vegetation types include shrub land towards dune crests with Burkea (*Burkea Africana*) and shrubbed woodland with mixed mopane (*Colophospermum mopane),* (Mendelsohn & Obeid 2004; Mosojane 2004). NG12 comprises predominantly seasonal floodplain. Fertile soils that support subsistence agriculture are confined to lower depressions on land near the Okavango River and floodplains (Tawana Land Board 2005). Protected areas occur north of the Namibian border fence, namely Babwata National Park and Mahango Game Reserve. South and east of the northern buffalo fence are wildlife management areas (WMAs), which are utilized by photographic and hunting tourism operations.

The estimated elephant population in the eastern panhandle was 3,782 in 1996 (DWNP 1996) and 15,429 in 2010 (this study), with elephants ranging throughout the study area. Telemetry studies by Jackson *et al.* (2008) in the Okavango Panhandle region indicated that the north-south buffalo fence blocks elephant movements from the Okavango River east to the Kwando River (see figure 1) and it is reported that the Namibian border fence (see figure 1), poses a significant barrier to elephant movements between Namibia and Botswana, (Chase & Griffin 2006, 2007, 2009). This information suggests that elephant movement is restricted out of the eastern panhandle.

2.2.2. Aerial surveys

Two aerial surveys were conducted over NG11, NG12 and NG13. The first took place in August 2008 over six days (August 22 – 25 and Aug 27-28, 2008) and the second in June/July 2010 over six days (June 30 – July 5, 2010). Surveys were conducted during morning hours (~0730 - ~1130 hrs) and corresponded with the dry season when vegetation cover is sparse and therefore visibility of herds is increased. Transect sampling was used rather than block or quadrat sampling, to minimise sampling error from the effect of animals not being distributed evenly.

Aerial surveys were conducted along flight transects using a Cessna 206 in 2008 and a Cessna 182 in 2010. All transects were flown at 100 knots and altitude was maintained between 90-92m (300-308ft) using a radar altimeter. Prior to flying, all transects were incorporated into a digital map of the study area, using ArcView 3.3 (ESRI), with their beginning and end point coordinates. All flight transects were systematically flown along generally north/south axes (figure 2). A north-south orientation was flown so that transects traversed the shorter dimension of the study area making the transect lengths shorter and hence the sample unit smaller. Transects were also aligned perpendicular to the Okavango River, to reduce sampling error (Norton-Griffiths 1978). GPS receivers (Garmin 12 xl, Garmin 176c) and DNR Garmin software (Minnesota Department of Natural Resources, MIS Bureau, GIS Section) were used to navigate along transects.

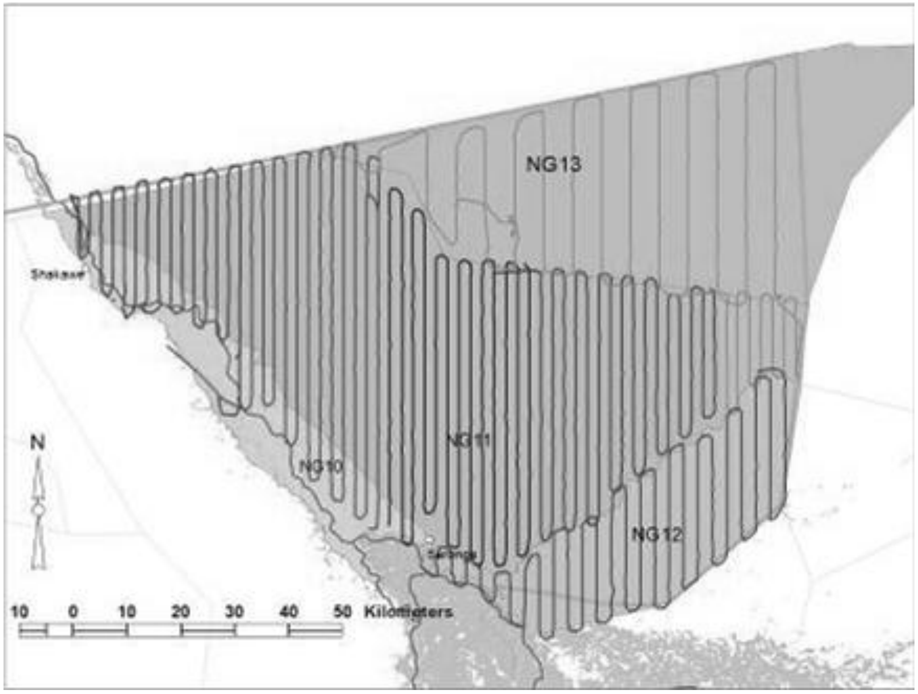


Figure 2. Map of transects flown over the survey area

The standard methodology for strip transect sampling developed by Norton-Griffiths (1978) was used. Two wands were attached to the wing struts of the plane to delineate a 250m interval for recording elephant observations at an altitude of 90m (300ft). Additionally, a mark was put on the plane window to help observers keep their eyes at a consistent height to maintain the same sighting angle for each observation. This helped keep consistent interval widths for each observation. Each interval width on each side of the plane was calibrated and confirmed prior to initiating the first survey by placing markers at measured distances on the ground and conducting flyover tests (Norton-Griffiths 1978). Where necessary the wands were adjusted to provide a 250m-wide strip at 90m (300ft) altitude and struts were attached for the duration of each survey.

The census zone was divided into three strata (see table 1). These strata were delineated according to wildlife management areas, and expected distribution and abundance of elephants from prior surveys (DWNP 1996, 1999, 2001, 2002, 2003, 2004, 2005, 2006; Jackson *et al.* 2008). Three levels of sampling intensity were used. In areas designated for high intensity sampling, NG11, transects were spaced 2 km apart, providing a ~25% sampling coverage. Transects were spaced 2.5 km apart in NG12 that was designated for moderate sampling intensity, providing a sampling coverage of ~20%. Transect spacing of 5 km was used for low intensity sampling in NG13, providing ~10% sampling coverage (figure 2).

Using the standard methodology for strip transect sampling developed by Norton-Griffith (1978), only elephants that were observed within the interval were counted and recorded. For each elephant seen within the transect interval, the observer called out the numbers of elephants and herd type (bull or breeding herd). The same two observers (A. Songhurst (R) and K. Landen (L)) were used throughout the survey, one on each side of the plane. The front seat recorder (M. Chase) logged all elephant observations made by the observers and assisted the pilot with navigation along the pre-determined transect lines, recording the start and end times for each transect. With each herd observation, the GPS waypoint, waypoint number, time of observation, altitude, and number of elephants observed were recorded.

To verify herd size and the sighting of herds within the interval defined by the wands, two Canon EOS 10D digital cameras with 20 mm wide-angle lenses, camera backs with time code generators and window camera mounts were used. A camera was mounted on each side of the plane. The cameras provided high-resolution images to verify counts in subsequent analyses. Observers took a photo with each elephant observation of >10 animals and a GPS time code was recorded to the minute for every frame exposed. The digital images of each herd were interpreted and compared to the observers' counts. This enabled me to correct for counting bias following methods outlined in Norton-Griffiths (1978) and determine whether elephants recorded actually occurred within the strip interval.

2.2.3. Population growth

Data were collated from past aerial surveys undertaken in the eastern Okavango panhandle. All past surveys used the standard methodology for strip transect sampling developed by Norton-Griffith (1978) and calculated population estimates using Jolly's Method II (Jolly 1969). Surveys were conducted by the Department of Wildlife and National Parks (DWNP) in eight dry seasons (1996, 1999, 2001, and 2002-2006) over a survey area incorporating NG11, NG12 and NG13. Areas varied slightly between years (see appendix I) ranging from 9835km^2 to 9919km^2. Strip widths of ~400m (200m each side of the plane) were used in DWNP surveys and six nautical miles (12km) were left between transects, giving a mean sampling intensity of 3.46% (DWNP 1996, 1999, 2001, 2002, 2003, 2004, 2005, 2006). In the dry season of 2003, Jackson *et al.*, (2008) conducted a survey in NG11 covering an area of 5952km^2, using strip width of 800m (400m each side of plane) and one nautical mile (2km) between transects, giving a sampling intensity of 40%.

Population estimates were plotted from past and current surveys to investigate the population growth rate in the study area. Generalised linear models with normal error structure were conducted for a) the total study area survey population estimates and b) for the management area NG11 survey population estimates, with year of survey fitted as the explanatory variable and significance tested at $p < 0.05$. The maximum growth rate for an elephant population was calculated at 7% by Calef, (1988), using a minimum inter-calving period of three years (ranges between 3 and 4.7 years) and a mean age of first birth of a calf at eleven years (ranges between 8-14 years). The estimated population growth rate was compared to this predicted maximum rate of increase. The death rate of natural elephant populations vary between different populations but appear to range between 1.43 – 7.4%, while population growth rates average 2.17% (ranging between 3.75-11.28%).

2.2.4. Carcass ratio

The number of dead elephants (carcasses and skeletons) were recorded using Norton-Griffith's (1978) standard methodology for strip transect sampling in 2008 and 2010 aerial surveys. Only elephant carcasses that were observed within the interval were counted and recorded. For each carcass seen within the transect interval, the observer called out the numbers of carcasses and the level of decay (bones only or skin present). The number of dead elephants were estimated in the same way as live elephants for 2008 and 2010 using the Jolly method II (Jolly 1969). Carcass ratios (No. dead/(No. dead + No. alive) were calculated for both years and estimated population change calculated following methods in Douglas-Hamilton and Burrill, (1991).

2.2.5. Simulation surveys

I used simulations to compare precision of population estimates and estimated population growth rates with different sample designs, varying spatial distributions of elephants and a range of spatial scales.

Distribution of elephants were simulated as spatial point patterns following a clustered distribution, using similar approaches to other studies, such as Khaemba *et al*. (2001) and Stein and Georgiadis, (Stein & Georgiadis 2006). The dynamics of an elephant population are simulated so as to reflect a real life distribution. In my study the Mat\’ern Cluster process was used to generate spatial patterns of elephants (Mat\'ern 1986; Baddeley & Turner 2010). The Mat\’ern's cluster process is constructed by first generating a Poisson point process of “parent” points with intensity (kappa). Then each parent point is replaced by a random cluster of points, the number of points in each cluster being random with a Poisson (mu) distribution, and the points being placed independently and uniformly inside a disc of radius (r) centred on the parent point (Baddeley & Turner 2010).

Simulations used the total census zone (8732km^2) and 1000 simulated survey samples were taken at varying sampling intensities (3%, 5%, 10%, 20%, 40%, 50% and 80%). The simulations were repeated using five different spatial distributions of elephants by varying the value of r in the Mat\’ern Cluster process, ranging from very clumped (r = 0.006) to very dispersed (r = 1). These simulations represented the radius of an elephant herd being six metres (approximate length of one adult elephant) up to 1000m. The root mean square error (RMSE), defined as $\sum(\hat{Y}-Y)^2$ was used to measure the precision of simulated surveys and compare sampling intensities and elephant distribution (Khaemba *et al.* 2001).

The average family unit size of African elephants is between eight to nine animals (ranging from 2-25), (Moss & Poole 1983; Wittemyer 2001) and bulls generally tend to be solitary or associate in all male herds (range 1-30) (Moss 1988; Eltringham 1991; Moss 1996). My survey results showed that the average herd size in the Okavango panhandle (combining family group and bull herd results) was eight animals, therefore the mean cluster (herd) size (mu) used in my simulations was eight.

To decipher how real my calculated population growth rates were (with respect to them occurring by chance alone), I simulated ten (number of actual surveys conducted in total study area) surveys and explored the probability of observing the change in elephant population numbers (slope) actually observed in the data. First, I generated ten random population estimates with a normal distribution using the simulated mean and standard deviation of population estimates derived from methods above and regressed these against a vector (1:10) representing time. This was repeated 1000 times for each combination of elephant distributions and sampling intensities. I then looked to see where the slope actually observed in the data lay in the frequency distributions of these simulated slopes.

To investigate the effect of spatial scale on the precision of aerial surveys, the whole simulation process was repeated, using a census zone covering the whole of the estimated elephant range in the Kavango region (187,220km^2).

2.2.6. Population growth in relation to HEC incidents

The yearly totals of elephant crop raiding incidents were collated from Department of Wildlife and National Parks (DWNP) Problem Animal Control (PAC) records from 1998-2010, as well as past population survey estimates of elephant numbers and densities in the panhandle over the past twelve years.

To determine whether the number of elephants or elephant density affects the number of elephant crop raiding incidents in the area, estimated elephant population numbers and densities were plotted against the total number of crop raiding incidents recorded per year over the past twelve years. A linear regression was conducted for the yearly number of crop raiding incidents, with the estimated elephant population per year or elephant density per year fitted as continuous variables.

2.2.7. Data analysis

R 2.11.1 was used for all statistical analyses (R Development Core Team 2010) and R language verified using Crawley (2007).

Generalised linear models (GLM) with normal error structures were conducted to explore elephant population growth. For each GLM used, the maximum model was fitted and simplified by stepwise deletion of non-significant interactions, quadratic terms, and main effects. Model-checking plots were drawn to check for constancy of variance and normality of errors. Model fit was then checked using F-test for normal errors and significance determined for all analyses at $p < 0.05$ (Crawley 2007).

Following the guidelines developed by Norton-Griffiths, (1978) abundance and variance estimates for strip transect counts were calculated from observation data. I calculated actual strip width observed and adjusted for altitude following Norton-Griffiths (1978) and used the traditional Jolly’s Method II (see appendix II) for unequal sized sampling units (Jolly 1969). The Jolly’s Method II ‘ratio method’ is based on the calculation of the ratio between animals counted and area searched. The population estimate is based on the density of animals per sample unit (transect) rather than number of animals per sample unit. I calculated population estimates for each block and summed these estimates to obtain an estimate for the entire survey area.

Confidence Intervals were used as non-parametric measures of precision $CI(x) = x \pm t_{2,\alpha} * SE(x)$, (Milner-Gulland & Rowcliffe 2007). The 95% confidence intervals (CI) were calculated and the CI expressed as a percentage of the population estimate as a measure of precision.

Two sample t-tests were used to compare mean bull and family group sizes per observer and between surveys. Chi-square χ^2 goodness-of-fit tests were used to compare density estimates between years, and numbers of total herds, bull herds, and family groups seen per observer.

2.3. Results

2.3.1. Transect data

For the entire 8,732 km^2 survey area, a total of 101 transects were flown in both 2008 and 2010: 63 in NG11, 25 in NG12; and 13 in NG13, totalling a distance of 3,294.92 km. Sampling intensity and search rate were calculated for the total survey and per strata (see table 1). The average transect length was 38.7km (range 4 – 67km), and the average time to fly one transect was 13.63 minutes (range 2 – 24mins).

Table 1. Aerial survey transects flown in the eastern Okavango Panhandle by block/strata

Strata Name	**Area (km^2)**	**Year**	**Actual Strip Width (km)**	**Area Covered (km^2)**	**Total Time (Min)**	**Transect Spacing (km)**	**Sampling Intensity (%)**	**Search Rate (km^2/min)**
NG11	5140	2008	0.513	1252.71	766	2.0	23.7	1.64
		2010	0.4998	1220.48	819	2.0	23.1	1.49
NG12	1092	2008	0.513	226.31	144	2.5	18.6	1.57
		2010	0.4998	220.5	170	2.5	18.1	1.30
NG13	2500	2008	0.513	211.27	132	5.0	10.3	1.60
		2010	0.4998	205.83	137	5.0	10	1.50
Total	8732	2008	0.513	1690.29	1042	-	19.7	1.62
		2010	0.4998	1646.8	1126	-	19.2	1.46

2.3.2. Elephant population survey

2.3.2.1. Population size and density

A significantly larger number of elephants were observed in 2010 (n = 2834) than in 2008 (n = 1927), ($\chi^2 = 172.8$, df = 1, $p < 0.001$) for the whole study area (see appendix I). There were also significantly large differences in the number of elephants sighted in each survey block between 2008 than 2010: NG11 ($\chi^2 = 6.258$, df = 1, $p = 0.01$); NG12 ($\chi^2 = 106.4$, df = 1, $p < 0.01$); and NG13 ($\chi^2 = 160.19$, df = 1, $p < 0.01$).

Combining herd observations for both observers and accounting for average flight altitudes, strip transect sampling from the August 2008 survey provided an estimated total of 8,905 elephants for the 8732km^2 area (1.05 elephants/km^2) compared to 15,429 elephants in the same area (1.77 elephants/km^2) in 2010 (see table 2), indicating a finite rate of population change of 1.3 over the past two years or a population increase of 30% per year. The density estimates for the total study area were not significantly different between years ($\chi^2 = 0.18$, df = 1, $p = 0.67$).

Population estimates and elephant densities for NG11, NG12 and NG13 showed increases between 2008 and 2010 (see table 1). Densities were not significantly different between years for each survey block NG11 ($\chi^2 = 0.01$, df = 1, $p = 0.91$), NG12 ($\chi^2 = 0.51$, df = 1, $p = 0.48$) and NG13 ($\chi^2 = 0.82$, df = 1, $p = 0.36$). Considering the population estimates and their standard errors for each survey block, estimates were not statistically different between years ($p < 0.05$).

2.3.2.2. Carcass ratio

A larger number of dead elephants were observed in the total study area in 2010 (n = 44) than in 2008 (n = 41), and estimated number of dead elephants were consequently higher in 2010 than 2008. In NG11, a larger number of dead elephants were estimated for 2010, but for NG12 and NG13 estimated numbers of dead elephants decreased in 2010 (see table 2).

Carcass ratios were calculated for the total study area and per survey block (see table 2) and percentage of population change per year estimated based on carcass ratios (Douglas-Hamilton

& Burrill 1991). For carcass ratios of 1% a yearly population change of 27.8% (95% CI:15.5% - 41.5%) is estimated, for those of 2% a yearly population change of 17.9% (95% CI: 7.4 – 29.4%) is estimated and for carcass ratios of 4% a yearly population change of 8.7% (95% CI:-0.2-18.5) is estimated (see Douglas-Hamilton & Burrill 1991). My carcass ratios for the whole study area ranged from 1% to 2%, indicating a population increase ranging between 17.9% and 27.8% would be predicted in the panhandle from this model.

Table 2. Estimates of elephant numbers for the August 2008 and July 2010 dry season aerial surveys, Okavango Panhandle, Botswana

	Dry season August 2008								Dry season July 2010						
Area	**Size (km^2)**	**N (Alive)**	**SE**	**95% CI**	**CI as % of N**	**Density (ele/km^2**	**N (Dead)**	**Carcass Ratio**	**N (Alive)**	**SE**	**95% CI**	**CI as % of N**	**Density (ele/km^2**	**N (Dead)**	**Carcass Ratio**
NG11	5140	6383	846	4692-8074	26.5	1.32	127	2	7816	1222	5373-10259	29.5	1.52	160	2
NG12	1092	1303	292	700-1906	46.3	1.27	29	2	2937	1054	762-5112	74.1	2.69	15	1
NG13	2500	1219	425	293-2145	76	0.48	47	4	4676	1288	1869-7483	59.5	1.87	36	1
Cumulative Total	8732	8905	-	-		1.02	203	2	15429	-	-		1.77	211	1
Calculated Total	8732	9963	1170	7642-12284	23.3	1.14	212	2	15027	2008	11043-19011	25.7	1.72	233	2

2.3.2.3. Herd observations and abundance

In August 2008, 127 bull herds, and 131 family groups were observed, while 7 herds were unclassified. Bull herd size averaged 2.8 elephants (range 1-18), while family group size averaged 11.8 (range 2-41).

For the July 2010 survey, 152 bull herds and 179 family groups were observed. Average bull herd size was 2.23 elephants (range 1-16), while family group size averaged 13.59 elephants (range 2-70) (see appendix III).

In 2008 and 2010 surveys, most herds were seen in NG11 (n = 208 and n=213 respectively) with more family groups than bull herds observed. Bull herd size averaged 2.7 elephants (range 1-19) in 2008 and 1.95 elephants (range 1-9) in 2010, while family group size averaged 11.9 elephants (range 2-50) in 2008 and 13.3 elephants (range 2-70) in 2010. In NG12, 44 herds were observed in 2008 and 72 in 2010. Average bull herd size was larger in 2010 (3.02 elephants (range 1-16)) than 2008 (2.4 elephants (range 1-7)) as well as average family group size (18.04 elephants (range 2-70) rather than 11.6 elephants (range 3-32)). The fewest number of herds were observed in NG13 in 2008 (n = 13) and 2010 (n = 46). Average bull herd size was larger in 2008 (5.7 elephants (range 1-10)) than in 2010 (1.43 elephants (range 1-4)), while family group size averaged 8.6 elephants (range 2-22).

2.3.3. Observer and count bias

There were no differences between the number of herds observed by each observer in 2008 ($\chi^2 = 3.24$, df = 1, $p = 0.07$) or 2010 ($\chi^2 = 0.87$, df = 1, $p = 0.35$). Likewise, there was no significant difference between the number of bull herds observed by each observer in 2008 ($\chi^2 = 0.07$, df = 1, $p = 0.79$) or 2010 ($\chi^2 = 0.95$, df = 1, $p = 0.33$), or the number of family groups observed in 2010 ($\chi^2 = 0.05$, df = 1, $p = 0.82$). However, the number of family groups observed by each observer differed in 2008 ($\chi^2 = 5.2$, df = 1, $p = 0.02$), with observer L reporting more herds (n = 78) than observer R (n = 52).

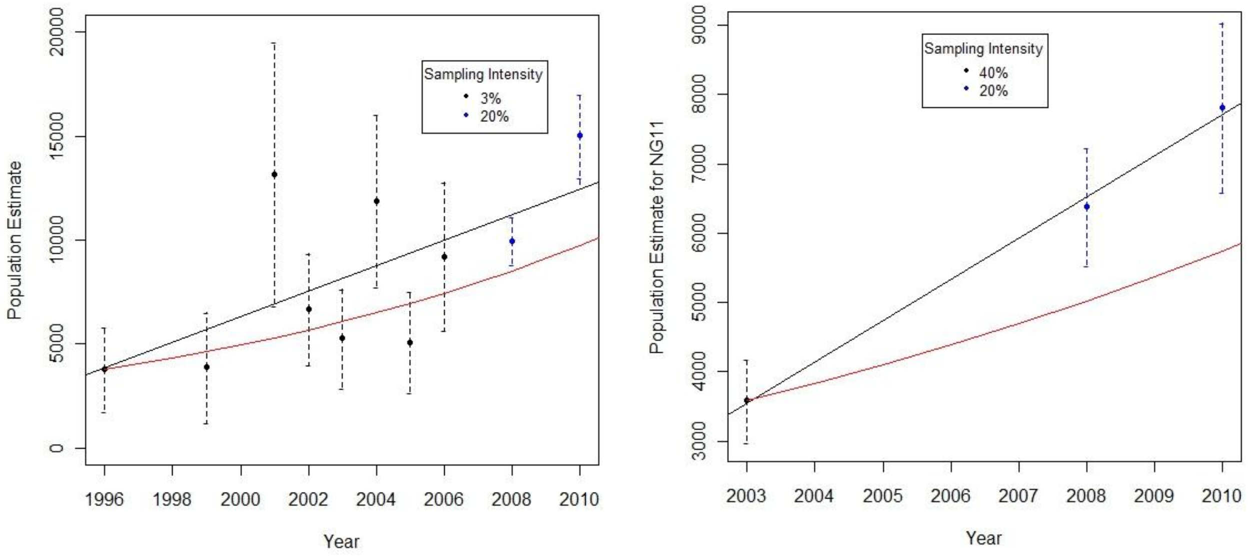
There were no significant differences between the two observers for average bull herd size in 2008 ($t = 0.27$, df = 95, $p = 0.79$) or 2010 ($t = 0.32$, df = 143, $p = 0.75$), or family group size in 2008 ($t = 0.95$, df = 100, $p = 0.34$) or 2010 ($t = 1.22$, df = 177, $p = 0.23$), (see appendix III).

There was no significant difference between the elephant numbers seen by observers compared to numbers in photographs in 2008 ($t = 0.26$, df = 195, $p = 0.79$) or 2010 ($t = 0.1$, df = 199, $p = 0.91$). The overall count bias calculated from photo-corrections was 1.02 in 2008 and 0.98 in 2010, this ranged from 0.94 in NG13 in 2008 to 1.1 in NG12 in 2008.

2.3.4. Population growth

Survey estimates from the past 14 years in the whole study area and for the past 7 years in NG11, indicate that the elephant population is increasing faster than the calculated maximum rate of increase (see figure 3 and appendix I). The finite rate of change over 14 years in the total study area using past and recent raw data is 1.1, (increase of 9.5% per year) and the regression line indicates a growth rate of 8.6% a year. Severe drought occurred in the mid 1990s, which could explain low population estimates in 1996-1999 and may therefore bias the finite rate of change. However, when these data were removed from analysis the finite rate of change remained the same (1.1). In NG11 the finite rate of change over 7 years is 1.1, also indicating a 9.5% increase per year.

Generalised linear models for survey elephant population estimates in the whole study area and in NG11, with normal error structure, retained year as having a statistically significant positive effect ($F = 5.3$, $df_M = 1$, $df_R = 9$, $p < 0.05$) and ($F = 298$, $df_M = 1$, $df_R = 2$, $p < 0.01$) respectively, confirming that the population has significantly increased.



a

b

Figure 3. Population growth for a) total study area (NG11, NG12 and NG13) and b) NG11. Red line is projected growth using maximum rate of increase (7%) for an elephant population (Calef 1988) and black is predicted growth from surveys.

2.3.5. Simulations

Simulations showed that as sampling intensity increased the root mean square errors (RMSE) decreased and the survey estimate is more precise. The interquartile range (IQR) decreases with increasing sampling intensity from 3% (IQR = 2600) to 20% (IQR = 1005) to 40% (IQR = 743), however, it is still relatively large for all sampling intensities.

As sampling intensity increases the RMSE decrease, yet the RMSE are not highly affected by the spatial distribution of animals. However, at lower sampling intensities, the RMSE increases slightly with more clustered distributions of elephants, indicating that surveys with lower sampling intensity are less precise when elephants have a clustered distribution (see appendix IV).

I am confident in the observed trend that elephant numbers are increasing (i.e. slope of regression) in the study area. The estimated slope from these data lay outside the 95% confidence limits of the frequency distribution of randomly generated slopes (calculated using simulated means and standard deviations of population estimates for different sampling intensities and spatial distributions of elephants), indicating that the observed trend was not random (see figure 4). However, the interquartile range for population estimates calculated from simulations was relatively large for all sampling intensities and, therefore, my population estimates cannot be considered highly accurate. Thus, I cannot be certain what the true actual population of elephants is in this area.

Consistent patterns were seen when I sampled at a broader spatial scale. The simulations for the whole Kavango region showed RMSE increasing as sampling intensities decreased and the interquartile range of population estimates decreasing as sampling intensity increased. At low sampling intensity the spatial distribution of elephants affected the precision of surveys over the larger study area, with more dispersed distributions resulting in larger RMSE.

2.3.6. Population growth in relation to HEC incidents

There was no significant relationship between number of elephants or elephant density and number of crop raiding incidents per year.

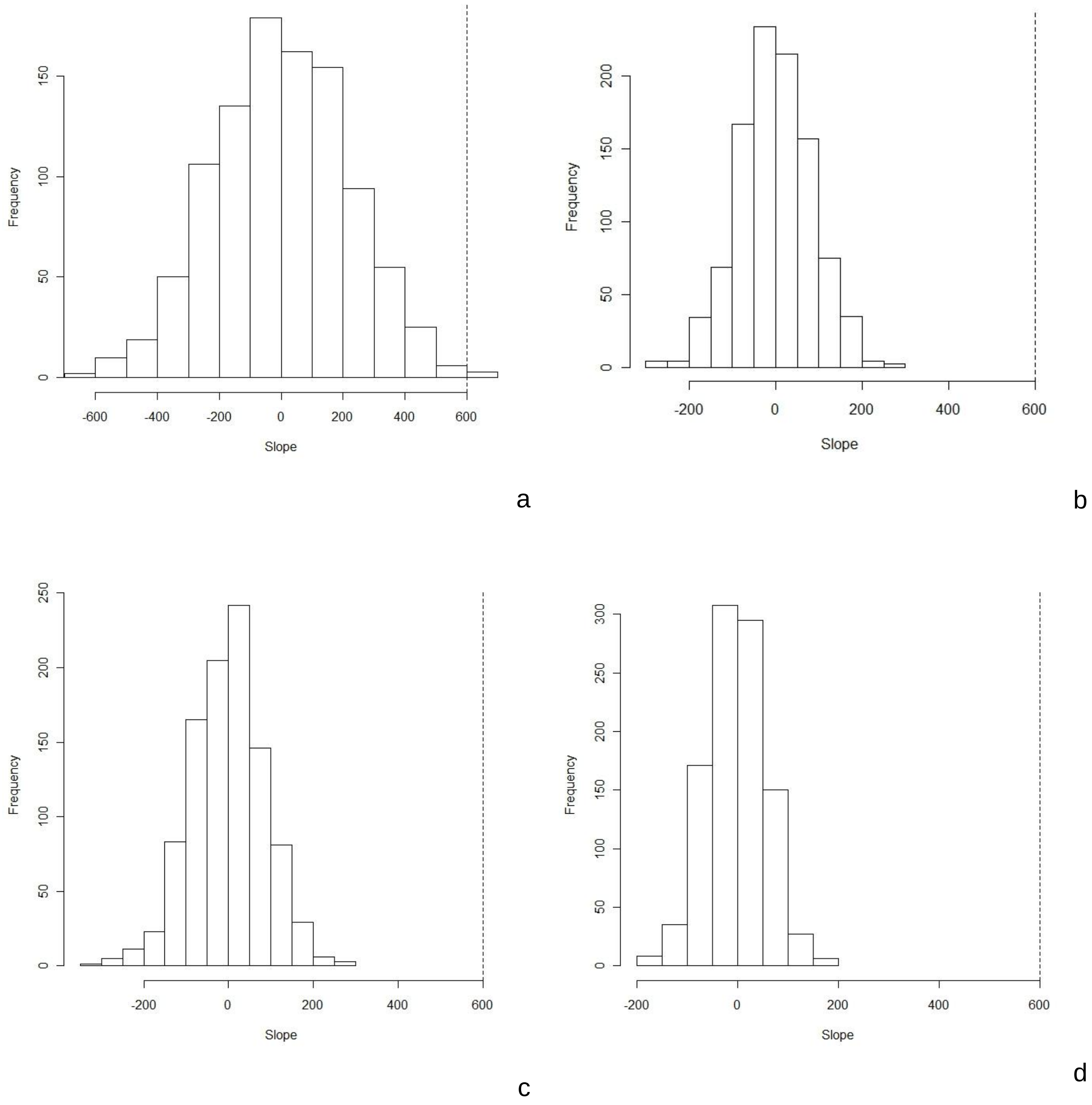


Figure 4. Frequency distributions of randomly distributed slopes for a simulated clustered distribution of elephants (r = 0.006) at different sampling intensities a) 3%, b) 10%, c) 20% and d) 40%. Dotted line represents actual slope calculated for aerial survey over total study area.

2.4. Discussion

2.4.1. Population growth

My study has shown that although I cannot consider past and present elephant population estimates to be highly accurate in the Okavango Delta panhandle (as evident from the large standard errors around the mean), I can be certain about the estimated trend in population numbers. From past and current surveys in this area it appears that the elephant population is increasing at a rate of 9.5% per year, a rate higher than the theoretical maximum intrinsic rate of increase of 7% per year calculated for a closed population of elephants by Calef (1988). Rapid population increases have been found in elephant populations throughout Africa. For example, population growth rates as high as 13.3% per year have been reported in Addo National Park, South Africa (Whitehouse & Hall-Martin 2000), >11% in Amboseli National Park, Kenya (Moss 2001), 10% in Hwange National Park, Zimbabwe (Dudley *et al.* 2001) and 7.1% in Tarangire National Park, Tanzania (Foley & Faust 2010). Rapid population growth has been attributed to a number of environmental and social factors such as high rainfall, low population density, high resource availability and release from stresses such as poaching and drought (Dudley *et al.* 2001; Moss 2001; Trimble, Ferreira & van Aarde 2009; Foley & Faust 2010). Severe droughts were recorded throughout much of southern Africa during 1981-84 and 1992-5 (Walker *et al.* 1987), which caused a lot of drought related mortality of elephant in the region (Dudley *et al.* 2001). Therefore, it is possible that the high population growth rates we are seeing in the Okavango Delta panhandle could be a response to such declines before 1996.

Alternatively, such large annual increases in elephant numbers in the panhandle could indicate that elephants are entering the population through immigration (as well as birth) from other areas of the elephant range with little emigration out of the area (and few deaths). For example, Calef’s (1988) calculated maximum rate of increase is based on a population with a stable age structure and does not include other demographic rates that also contribute to changes in population size, such as immigration and emigration (Milner-Gulland & Rowcliffe 2007). If, however, elephants are indeed migrating into the Okavango Delta panhandle, then they must be crossing geological features that previously were considered barriers to movement, such as the Namibian border fence, the northern buffalo fence or the main Okavango River (e.g. Albertson 1998; Jackson *et al.* 2008; Chase & Griffin 2009).

The fences marking the northern, eastern and southern boundaries of my study area (see figure 1) were constructed in response to an outbreak of contagious bovine pleuropneumonia in 1995 (Amanfu *et al.* 1998; Scott Wilson Resource Consultants 2000; Martin 2005). The Namibian border fence was completed in 1997 and the Northern Buffalo fence in 1996. Prior to the completion of these fences, satellite collared elephants were recorded moving between Namibia and Botswana (Rodwell 1995), but subsequent to the fence construction, Albertson (1998) reported that the Namibian border fence had ‘terminated’ all wildlife movements, including elephants. Telemetry studies conducted in 2004 supported Albertson’s argument, indicating no movement of elephant across either the Namibian border fence or the northern buffalo fence from the panhandle (Jackson *et al.* 2008). Chase and Griffin, (2009) surmise that these fences may have severed the seasonal dispersal pattern of elephants and elephants may have become trapped in Botswana, consequently leading to a decline in elephant numbers in the Caprivi after the border fence was complete (Craig 1998). Loarie, Van Aarde, and Pimm, (2009a) confirmed that fences appear to restrict elephant movements in southern Africa and in the wet season elephant dispersal is constrained and elephants tend to congregate along fences. Chase and Griffin, (2006) did show, however, that elephants were dispersing through damaged sections of the border fence in 2005.

The proportion of dead elephants to all dead and alive elephants (carcass ratio) has been used as an index of relative elephant mortality (Douglas-Hamilton & Hillman 1981). Carcass ratios were low for the total study area in the Okavango panhandle (2%) indicating relative elephant mortality is low. Douglas-Hamilton and Burrill’s, (1991) model for determining population trends, therefore, predicts that a population with such low mortality could be increasing at a rate of 17.9% (95% CI: 7.4 – 29.4%) per year. Our estimated population increase falls within the 95% confidence interval of this predicted range. A low carcass ratio may indicate a low mortality rate, or it may be low as a result of low emigration rates and more live elephants entering the population through immigration (Douglas-Hamilton & Burrill 1991). Survival among adult elephants is high (Owen-Smith 1988) and elephants have few natural predators (apart from young occasionally being hunted by lion (*Panthera leo*) or hyeana (*Crocuta crocuta*)). The highest cause of mortality in elephant populations is, therefore, either human induced (Eltringham 1982) or through malnutrition during periods of drought (Walker *et al.* 1987; Dudley *et al.* 2001; vanAarde & Jackson 2007). Currently, there are no legal elephant hunting quotas (community or commercial) in the wildlife management areas of the eastern Okavango Delta panhandle, so human induced elephant mortality is limited to problem animal control activities in this area. Poaching levels are also considerably lower than neighbouring countries, owing to strict policy and legislation implementation and an effective anti-poaching unit in the Botswana Defence Force. No incidents of drought related mortality have been recorded in the panhandle recently either, which could also explain the low carcass ratios.

To fully understand the dynamics of this elephant population I would need to gain a greater understanding of the survival and productivity rates of the population (Milner-Gulland & Rowcliffe 2007). Demographic fluctuations in elephant populations can result from variations in conception rate, prenatal survival, first year survival and cumulative juvenile survivorship (Trimble, Ferreira & van Aarde 2009), which can be affected by environmental conditions and often show intra- and inter-population differences (vanAarde & Jackson 2007). Evidently, further investigations into elephant population dynamics are, therefore, needed in my study area. I also need to be able to incorporate extrinsic parameters such as immigration and emigration into a population dynamical model. However, in such a wide ranging population it would be difficult to make direct observations and distinguish immigrants from established individuals making it difficult to estimate the rate of immigration (Abadi *et al.* 2010). If I had a better understanding of immigration and emigration rates, I would be able to identify what the Okavango Delta panhandle elephant population's role is in the larger population of the Kavango region, such as whether it is a source or a sink population (Pulliam 1988; Thomas & Kunin 1999).

If elephants are moving into the panhandle from surrounding areas, one fundamental question is why? What is attracting elephants to the panhandle or alternatively deterring elephants from other areas? One explanation could be that elephants are coming from Namibia (if they can cross the fence), possibly to move away from human disturbance in the Caprivi region (Chase & Griffin 2009). However, from a review of population estimates in the Caprivi over the past 20 years there does not appear to be a pattern of population fluctuations between the panhandle and the Caprivi. Yet, if I review population estimates from the Okavango Delta (south of the northern buffalo fence), there is a clear pattern of Okavango Delta numbers decreasing where Okavango Panhandle numbers increase and vice versa (see appendices V and VI). From past survey estimates, it appears that elephant numbers decreased in 2002 and 2005 in the panhandle concurrently with population increases in these years in the delta, while in 2004 and 2006 numbers increased in the panhandle and decreased in the delta. Possible explanations for elephants moving from the Delta to the panhandle could be to access certain food, water and mineral resources or in response to disturbance from hunting concessions south of the buffalo fence (Blanc *et al.* 2007). Surface-water availability was found to be a key driver of elephant distribution in Zimbabwe (Chamaille-Jammes, Valeix & Fritz 2007), so elephants may be moving to access the main Okavango River in the panhandle. Several studies have also linked elephant movements with vegetation type and density (Merz 1986; Barnes *et al.* 1991; Verlinden & Gavor 1998), so it may be that elephants are migrating in search of certain food types including seasonal fruits (White 1994).

Despite the increase in elephant numbers in the panhandle, there was no statistically significant relationship between elephant numbers or elephant density and crop raiding incidents per year. In a study in Zimbabwe, however, Hoare (1999b) found that elephant density was significantly (but weakly) correlated to the number of problem elephant incidents. This difference could be attributed to different contributing factors associated with different study sites or it may be due to the spatial scale used for analysis. Hoare (1999b) calculated elephant density and problem elephant incidents per ward rather than for the whole study area. My results found no correlation between crop raiding incidents and elephant numbers or density. This correlation is, however, difficult to make when survey estimate precision is low as a result of low survey intensity.

2.4.2. Precision and bias in aerial surveys

Ideally population abundance surveys should minimise bias and maximise precision, with low bias being the priority in management terms (Milner-Gulland & Rowcliffe 2007). Spotting and counting bias represent the most important source of bias in aerial surveys (Norton-Griffiths 1978), which can be minimised by appropriate survey designs. Past and present aerial surveys conducted in the eastern Okavango Delta panhandle kept such bias to a minimum by using strip transects and stratified sampling, keeping a constant flying height and speed, and using the same observers per survey. Certain aspects of survey designs varied between surveys, however, including strip widths and sampling intensities, which may affect the accuracy and precision of results.

In transect surveys, all individuals within the sampling strip are sampled and there is no requirement to calculate an estimation of detectability (Milner-Gulland & Rowcliffe 2007).

However, it has been found that observers tend to underestimate the number of individuals in large groups (Elphick 2008), and this was apparent in my survey. Observer counts were on average 2% lower than photo-corrected counts, although this difference was not statistically significant. My survey used photo-corrected counts to minimize such counting bias. This underestimate could, however, be a potential source of bias in DWNP surveys where photo-corrected counts were not used. Counting bias is also affected by the strip width in transect surveys, with narrower strip widths avoiding a larger search area, which reduces the chance of observers missing animals in the strip. Narrower strip widths, therefore, reduce counting bias and increase the accuracy of counts (Norton-Griffiths 1978; Jachman 2002). The DWNP and my surveys used narrow strip widths (400m and 500m respectively), however, Jackson et al (2008) used a wider strip width (800m) which may have resulted in less accurate abundance estimates.

Sampling intensities varied between past and current surveys in the study area. DWNP surveys had an average 3% sampling intensity, indicating that they used a smaller number of small sampling units, thereby increasing accuracy but also increasing sampling error. Whereas my survey had an average sampling intensity of 20% for the whole census zone because we had slightly larger sampling units and increased the number of sampling units, thereby slightly decreasing accuracy but also reducing sampling error. For NG11, Jackson *et al.* (2008) had a 40% sampling intensity compared to our 23% coverage, because they used larger sampling units, which decreased the accuracy of their estimates. Evidently, sample error is reduced with larger sample sizes and so, ideally, a large number of smaller sampling units should be used to minimise counting bias and sample error (Norton-Griffiths 1978).

My study reveals the usefulness of using simulations to test the reliability of survey data. Population estimates from past and current surveys in the panhandle showed large fluctuations (see appendix I) that are not biologically possible. However, with the aid of simulations, I was able to determine with certainty trends in population numbers that were predicted from this data. Simulations also revealed that such trends do not appear to be significantly affected by sampling intensity of surveys, spatial distribution of elephants or the spatial scale of the survey, indicating that, as long as bias is minimised during survey design and implementation, aerial survey data can be used reliably for predicting population trends despite variations in the above variables. Patterns were consistent across different spatial scales indicating that surveying smaller or larger areas of elephant populations should not significantly affect the precision of abundance estimates.

The 95% confidence intervals expressed as a percentage of a population estimate are used as a measure of precision. Data from past aerial surveys show that surveys conducted at lower sampling intensities are less precise, because these 95% confidence intervals were larger. My simulations confirmed that precision of estimates increase with higher sampling intensities, as found in previous studies using simulation (e.g. Khaemba *et al.* 2001; Ferreira & van Aarde 2009). My simulations also showed, however, that sampling at lower intensities is sufficient for reliably observing trends in population changes over time (Ferreira & van Aarde 2009). At higher sampling intensity, the precision of survey estimates was slightly lower with more clustered distributions of elephants in our simulations. It has been found in previous simulation surveys that precision of abundance estimates increases marginally with increasing sampling intensity when animal herds are clumped (Khaemba *et al.* 2001) and it has been predicted that sampling intensities as high as 50-70% are actually required to improve precision when herds are clumped (Ferreira & van Aarde 2009). Depending on the purpose of the survey, knowledge of animal distributions may be important prior to survey design to determine whether an increase in sampling intensity to improve precision outweighs the cost of a high intensity survey. If the main purpose of the survey is to identify population trends, however, then our simulations indicate that the distribution of elephants should not affect the ability to identify such trends.

If the intention of monitoring is to gain accurate estimates of population abundance for management strategies such as establishing hunting quotas or identifying sustainable harvesting numbers for population control then both precision and accuracy of population estimates need to be maximised. This requires surveys to be conducted at the highest sampling intensity possible (i.e. 40%) using the narrowest transects width possible (i.e. 400m). However, time and financial constraints would need to be considered to assess the viability of conducting such surveys. If the purpose of the survey is to estimate trends in elephant population abundance, however, then our study shows that longitudinal studies at even the smallest sampling intensity are adequate.

2.4.3. Conclusion

The management of burgeoning elephant populations in some southern African countries (such as Botswana) and resulting conflicts with humans living on the edge of expanding elephant ranges, are important conservation issues (Blanc *et al.* 2007). The ability to reliably estimate population numbers, trends and densities of elephants has, therefore, a very important role in helping to guide their effective and appropriate conservation and management. The design of an aerial survey to monitor wildlife population abundance needs to reflect the level of precision and accuracy required for consequent management actions. Surveys designed to sample a population to estimate abundance, need to minimise bias and maximise precision at levels relevant to desired wildlife management applications (Hone 2008). Simulations provide a viable method to verify the precision of aerial survey abundance estimates and trends in population numbers, when logistics and economics inhibit repeated aerial surveys.

Survey results from the Okavango panhandle reliably indicate that numbers are increasing faster than elephants reproductive potential would predict. However, the accuracy of population estimates are low for this study area and are not adequate to make informed decisions for management actions such as population control.

The increase in elephants in the Okavango panhandle is likely to be a result of a combination of factors. The population has been increasing rapidly since the droughts in 1995, with low mortality rates and probable high immigration rates. If elephants are indeed migrating into the panhandle area then they must be crossing geological features (fences and a river) that were previously thought to restrict movement, most likely at times when the fences are damaged. These findings have major implications for the management of elephants in this area. With an increasing elephant population and an expanding human population in the panhandle requiring more land for agricultural uses, incidents of human-elephant conflict are widespread. It is therefore imperative to understand why the elephant population numbers are increasing in order to design appropriate management strategies. If elephants are migrating into the panhandle when the fences are damaged, then once fences are repaired this could be effectively trapping elephants within this area. In this scenario, an effective management intervention would be to remove the fences to allow free migration. If, however, the population is increasing rapidly because of low mortality in the area, then it may be necessary to consider increasing the off-take of elephants through schemes such as community hunting quotas, which would consequently require more accurate estimates of population numbers. It is also evident that further research on the population dynamics and movements of elephants in this area is needed before informed management decisions can be made.

Chapter 3

Measuring Conflict - comparing insights from a standardised IUCN protocol with those obtained using government approaches

3.1. Introduction

Charismatic mega fauna, including carnivores and large herbivores are important flagship species for broad conservation initiatives (Walpole & Leader-Williams 2002), however, often these same animals may compete with people living in close proximity to their habitats and conservation areas. One common form of negative interactions between people and mega herbivores in agricultural landscapes is crop damage, (Naughton-Treves 1998; Hoare 1999b; Gillingham & Lee 2003; Hegel, Gates & Eslinger 2007), which can hinder agricultural community support for wildlife conservation (Hegel, Gates & Eslinger 2007). Studies investigating crop damage incidents by wildlife have been widely documented, but methods often vary and, consequently results can be difficult to compare across national and international regions (Hoare 1999a). For example, some studies rely on data collected through government departments, while others promote the use of independent data collection systems involving local communities, i.e. The International Union for Conservation of Nature (IUCN) human-elephant conflict (HEC) data collection protocol (Osborn 1998a; Hoare 1999b, a). Patterns of crop damage can vary from one site to another even when both pest species and crop types are similar, (Gill 1992a, b; Siex & Struhsaker 1999; Warren, Buba & Ross 2007), and so, in order to render studies to identify patterns of human-wildlife conflict (HWC) consistent on a regional and international/continent-wide scale, it is essential to promote the use of standardised data collection methods. It is not clear, however, how data collected from such systems compare to ‘top-down’ government compensation driven monitoring programmes.

A variety of methods have been used to collect information on wildlife crop damage. Some studies have been predominantly descriptive reporting the current extent of wildlife damage in an area (Waithaka 1993; Kiiru 1995; Ngure 1995). Several methods have focused on using questionnaire surveys to estimate the perceived damage (e.g. Conover & Decker 1991; Lahm 1996; O’Connell-Rodwell *et al.* 2000; Saj, Sicotte & Paterson 2001; Marker, Mills & Macdonald 2003; Arlet & Molleman 2007; Hegel, Gates & Eslinger 2007; Hill, Carbery & Deane 2007; Cai *et* *al.* 2008), while others have attempted to measure and quantify actual damage (e.g. Naughton-Treves 1998; Osborn 1998a; Hoare 1999b; Hoare & du Toit 1999; Smith & Kasiki 1999; O'Connell-Rodwell *et al.* 2000; Tourenq *et al.* 2001; Sitati *et al.* 2003; Barnes 2005; Chiyo *et al.* 2005; Sam *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Barnes 2007; Warren, Buba & Ross 2007). Relatively recently, a number of studies have compared actual damage estimates to perceived damage estimates (Sekhar 1998; Siex & Struhsaker 1999; Gillingham & Lee 2003; Fernando *et al.* 2005; Linkie 2007). Few studies (e.g. Stewart 2004) have, however, attempted to compare methods to quantify and monitor actual wildlife damage and, indeed, top-down government approaches to community based monitoring programmes.

Elephants, both in Africa (*Loxodonta africana* and *Loxodonta cyclotis*) and Asia (*Elephas maximus*), are a prime example of a mega herbivore that often compete with humans in agricultural landscapes. Yet, as Hoare, (1999a) highlights, many of the studies investigating elephant crop-raiding have been set up independently and conducted using different methods, with results presented in ways that are difficult to compare. In an effort to provide a solution to this problem, the Human-Elephant Conflict Task Force (HECTF) of the IUCN African Elephant Specialist Group (AFESG) established a standard HEC data collection protocol in 1999 (Hoare 1999a). This protocol is based on the reporting system used by Hoare & du Toit, (Hoare & du Toit 1999) and provides a standardized system to allow valid comparisons to be made about levels of HEC both within and across different bio-geographical regions of Africa (Hoare 1999a).

The IUCN data collection protocol focuses on the use of community enumerators to collect data and has three levels of investigation. First, a passive process of measuring the number of events reported is used, (Deodatus & Lipiya 1991; Simons & Chirambo 1991; Sukumar 1991; Kiiru 1995; Hoare 1999a). Then, an estimate of actual loss accrued through elephant damage is obtained (Barnes, Azika & Asamoah-Boateng 1995; Tchamba 1995; Barnes 1996; Bhima 1998; Osborn 1998a; Hoare 1999b; Sitati *et al.* 2003; Barnes 2005; Chiyo & Cochrane 2005; Chiyo *et al.* 2005; Sitati, Walpole & Leader-Williams 2005). Finally, an investigation of community perception through questionnaire surveys is conducted (Bell & McShane-Caluzi 1984; Languy 1996; Wunder 1996; Sam, Haizel & Barnes 1997; Bhima 1998; Blaney *et al.* 1998; DeBoer & Baquette 1998; Naughton-Treves 1998). This protocol, like other IUCN guidelines for addressing HWC (e.g. Hockings & Humle 2009), strongly promotes the involvement of local community enumerators to take an active role in monitoring and mitigating conflict incidents

(Hoare 1999a). Hoare & Mackie, (1993) highlight the need to use problem incident data that has been collected by independent enumerators to assess incidents (numbers and extent) because information sourced by farmers (Hawkes 1991; Newmark *et al.* 1994; Thouless 1994; Tchamba 1995) or government departments may be influenced by socio-economic circumstances, politics and government compensation schemes, and may, therefore be biased (Hoare 1999b; Siex & Struhsaker 1999). This approach, however, differs from the current system of HEC data collection used in Botswana.

Botswana is the only member of the Southern African Development Community (SADC) to employ a solely state funded HWC compensation system, (Hemson 2003). Compensation systems are based upon paying reparation to property owners for losses incurred from wildlife damage and aim to raise farmer's tolerance for losses by minimising the economic impact of these losses (Nyhus, Osofsky & Ferraro 2005). There are certain issues associated with compensation schemes, however, which could bias results of damage incident records, including an amount of moral hazard associated with them. For example, they may decrease the motivation to defend crops (Hoare 2001; Nyhus, Osofsky & Ferraro 2005; Warren, Buba & Ross 2007), or they may give an incentive to people to convert land for agriculture use, leading to the migration of people into wildlife areas, thereby, increasing HWC risk and resulting compensation (Bulte & Rondeau 2005; Rondeau & Bulte 2007; Warren, Buba & Ross 2007). A serious problem associated with ex-post compensation schemes (where compensation is given for damages after they have occurred) is the incentive for people to overstate damages to gain additional income (Schwerdtner & Gruber 2007) or even make bogus claims for damage that either did not occur or was inflicted by non-compensated perpetrators, e.g. domestic livestock (AFESG 2002).

Despite potential problems associated with compensation, an advantage of the scheme in Botswana is that it not only encourages farmers to report incidents, but it has facilitated the development of a comprehensive Department of Wildlife and National Parks (DWNP) Problem Animal Control (PAC) data collection system. The DWNP have records of the number of wildlife damage incidents reported as well as actual assessments and quantifications of damage, which is considerably more than exists in some government monitoring programmes in other countries (Hoare 1999a, 2001; AFESG 2002). So, how does HEC data collected from a 'top-down' approach monitoring programme differ to the IUCN approach? I attempted to identify and compare the advantages and disadvantages of HEC data collection between DWNP and IUCN methods.

This study established a community enumerator monitoring programme for HEC incidents over a three year period in the Okavango Delta panhandle, Botswana, following the IUCN data collection protocol. In addition, the government DWNP PAC monitoring scheme continued in the study area. With these two HEC monitoring systems in place at the same time, I was presented with the opportunity to a) record the current status (frequency and extent) of HEC; b) quantify crop-damage by elephants; c) assess temporal patterns of HEC; and d) compare insights from the IUCN protocol and a government approach, not only in data collection efficiency, but also to explore the implications of the effectiveness and impact each has on HEC mitigation, and elephant conservation.

3.2. Methodology

3.2.1. Study area

The study was conducted between January 2008 and July 2010 on the eastern side of the Okavango Delta Panhandle, where the Okavango River reaches the Okavango Delta in Botswana (see figure 5). The eastern panhandle covers an area of 8,732 km^2 and consists of controlled hunting areas NG11, NG12 and NG13. The Namibian border marks the northern boundary and the northern buffalo fence marks the southern boundary (UTM Zone 34 7910000 – 7990000 South and 580000 – 710000 East). Protected areas occur north of the Namibian border fence, namely Babwata National Park and Mahango Game Reserve. South and east of the northern buffalo fence are wildlife management areas, which are utilized by photographic and hunting tourism operations.

Rainfall averages 360 - 500 mm annually and generally falls between mid October and March (rainy season), (Department of Meteorology, 2010). The mean monthly maximum temperatures in the panhandle range from 26.1 ºC– 35.1ºC, with July being the coldest month and October the hottest (Department of Meteorology, 2010). Deep Kalahari sands dominate throughout the study area, and main vegetation types include mopane woodland (*Colophospermum mopane*), silver terminalia woodland (*Terminalia sericea*), two Acacia vegetation types, namely Candle pod acacia (*Acacia hebaclada*) – Buffalo thorn (*Ziziphus mucronata*) and Umbrella thorn (*Acacia tortillis*) – Combretum (*Combretum imberbe*) valley sandveld, Zambezi Teak (*Baikiaea plurijuga*), False mopane (*Guibourtia coleosperma*), Burkea (*Burkea Africana*), Riverine woodland (e.g. *Diospyros mespiliformis; Croton megalobotrys; Acacia nigrescens; Lonchocarpus capassa; Kigelia africana*), and mixed woodland (*Colophospermum mopane*, *Terminalia sericea*, *Acacia sp.*) (Albertson 1998; Mendelsohn & Obeid 2004). Subsistence agriculture occurs in lower depressions near the Okavango River and floodplains (Tawana Land Board 2005) and in land <14km from the main road where there are fertile soils.

The estimated elephant population in the eastern panhandle is between 9,000 – 14,000 elephants (see chapter two) and elephants range throughout the study area.

The 2001 census recorded 15,718 people living in the eastern panhandle, with Seronga village being the largest settlement with 1,641 inhabitants (CSO 2001). There are 12 main villages (population > 500 people) in the area, extending from Mohembo-East to Gudigwa, including Kauxwi, Xakao, Tobera, Sekondomboro, Ngarange, Mogotho, Seronga, Gunotsoga, Eretsha, and Beetsha, with additional cattle posts and settlements occurring between villages. The economy is quite diverse and includes floodplain and dry land agriculture, livestock farming, wage labour, craft and tourism related enterprises (ODMP 2002; Mendelsohn & Obeid 2004). Depending on annual rainfall the planting of crops occurs between November and January and harvesting occurs between April and June (Mosojane 2004).

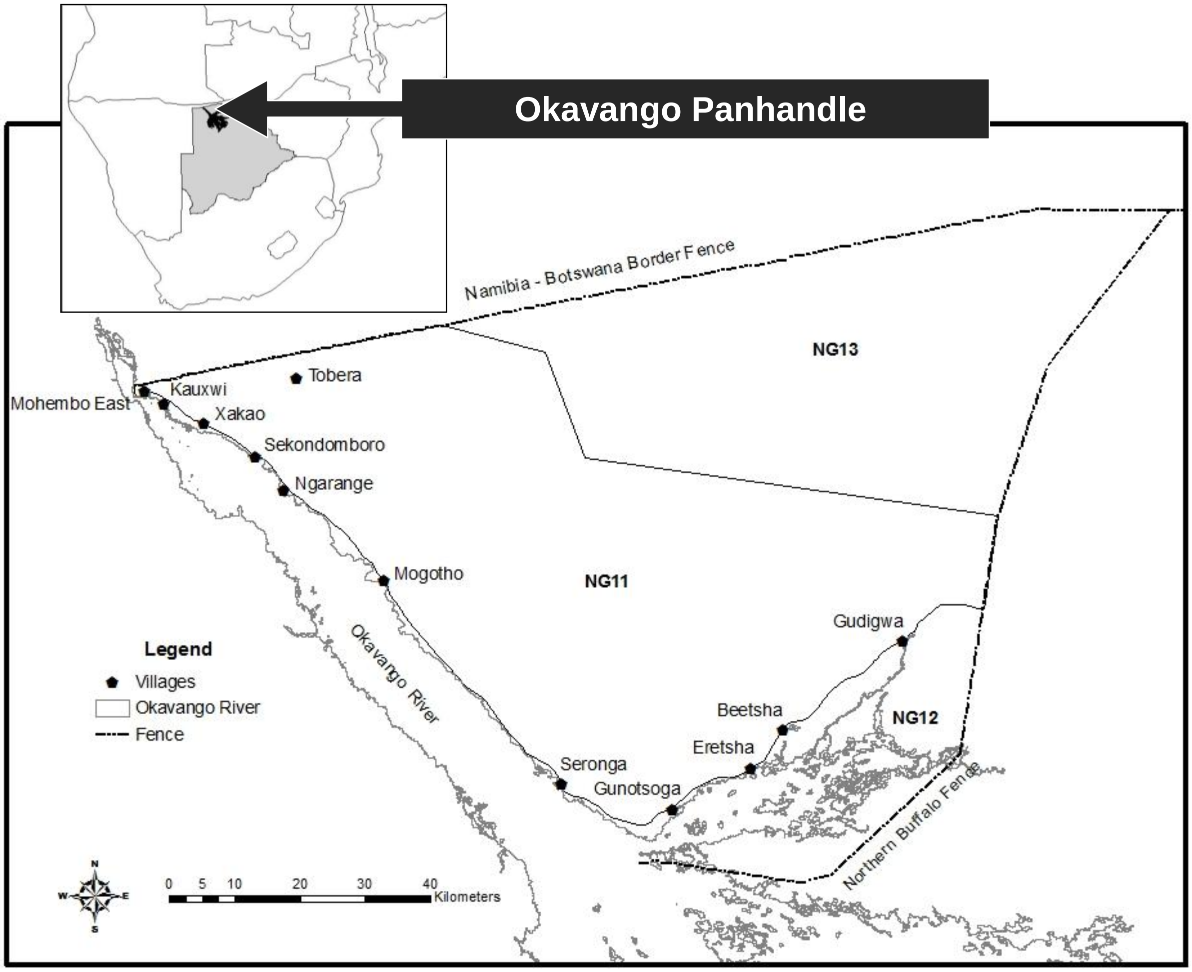


Figure 5. Map of the study area

3.2.2. Data analysis

All data analysis was carried out using R 2.11.1, (R Development Core Team 2010).

For all analyses (see details below), only variables that had statistically significant test results ($p<0.05$) in univariate analysis were entered into the multivariate analysis. Continuous variables (i.e. rainfall, area of damage, distance variables) were square root transformed, where appropriate, to approximate better to a normal distribution and factor levels reduced where necessary to simplify the final model without loss of explanatory power.

Generalised linear models (GLM) with specified error structures (see sections below for details) were conducted. For each GLM used, the maximum model for two-way interactions was fitted and simplified by stepwise deletion of non-significant interactions, quadratic terms, and main effects. Model-checking plots were drawn to check for constancy of variance and normality of errors. Model fit was then checked using chi-squared goodness-of-fit test for Poisson and binomial errors and F-test for normal errors and significance determined for all analyses at $p < 0.05$. If the residual deviance was larger than the residual degrees of freedom (over dispersion) then the model was refitted using quasi-Poisson or quasi-binomial errors rather than Poisson or binomial, respectively. Where outlier data points were present, models were fitted with and without these points and compared using chi-squared goodness-of-fit tests. If parameter estimates or their standard errors were greatly affected by such points, they were identified as highly influential points and were removed from the final model (Crawley 2007).

3.2.3. Government of Botswana PAC data collection

Data on HEC incidents in Botswana are collected through the problem animal control (PAC) unit of the Department of Wildlife and National Parks (DWNP). There is one DWNP office situated in Seronga village, which services the whole study area. People are encouraged to report HEC incidents (crop damage; property damage; injury or death of livestock due to elephant; injury or death of a person; injury or death of an elephant) to either the Kgosi (Chief); police department; or DWNP official in their village, or directly to the DWNP office in Seronga within 7 days of the incident occurring. All reports are collected from villages by DWNP personnel and collated in Seronga. DWNP officers visit reported incidents and complete a compensation assessment form, recording details of the farmer, incident location (preferably with global positioning system (GPS) coordinates), crops damaged, mitigation used and measurement of area damaged (using a surveyor's wheel). DWNP aims to verify incidents and compensate within two weeks of the initial report date (DWNP 2010). Locations of raiding incidents were grouped into twelve village categories according to the closest distance between the incident and a main village.

3.2.4. Independent primary HEC data collection using IUCN protocol

In order to establish a reliable independent conflict reporting system for this study, the standardized data collection protocol of the Human Elephant Conflict Working Group (HECWG) in the IUCN (Hoare 1999a) was followed to collect primary data on HEC. Such a protocol has been used in previous studies (Barnes 1996; Hoare 1999b; Hoare & du Toit 1999; Smith & Kasiki 1999; Sitati *et al.* 2003; Chiyo & Cochrane 2005; Chiyo *et al.* 2005; Hedges *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Barnes 2007; Hedges & Gunaryadi 2009) and is supported by other NGOs (e.g. WWF) working on HWC programmes (WWF 2005).

Through consultations with village Kgosi (chiefs), one community member from each of the twelve study villages was selected and trained as an enumerator of elephant crop raiding incidents. This provided widespread coverage of the entire study area where the main human settlements occur and conflict between people and elephants had been previously reported. Farmers reported elephant crop-raiding incidents in the traditional manner (either to the Kgosi, Police or DWNP official in the village), and the local enumerator retrieved the report information from the relevant personnel before visiting farmers.

All HEC incidents occurring between January 2008 and May 2010, covering three crop raiding seasons were assessed. All incidents were visited by the local enumerator, who conducted an assessment of the incident and completed a standardised data collection form. The principal investigator (A. Songhurst) verified all crop damage assessments to ensure consistency and reliability of data collection. The location of incidents were described (area of village) and geo-referenced (GPS taken in centre of field for crop damage) using a Garmin Etrex GPS. Details on the incident, such as date and time of occurrence, characteristics of elephant perpetrators/victims i.e. age (following methods outlined in Laws 1966; Western, Moss & Georgiadis 1983; Lee & Moss 1995; Moss 1996); sex (based on herd classifications in Moss

2001; Chiyo & Cochrane 2005); herd size (similar to methods used in Chiyo & Cochrane 2005), and information on the person reporting the incident, i.e. age, gender, ethnicity, family size and livelihood were recorded for all incidents.

For crop damage incidents, the average pace size of each enumerator was measured, and the area of all fields and damaged portions of the fields were estimated in square metres using enumerator paces (Hoare 1999a). Data were also collected on field characteristics including the year of field establishment, previous years raided, surrounding vegetation type, entrance and exit directions of elephant, nearest available water source, location and direction of nearest elephant movement path to river, and active mitigation techniques in use. Physical barriers surrounding fields were categorized as bush fence, wire fence, tin-can fence, trench or none. The GPS location and number of watch huts were recorded and the distance between these and the damage incident recorded in metres.

3.2.5. Temporal patterns of crop-raiding

Data collected using the IUCN protocol (enumerator method) and from the Government of Botswana records (DWNP method) were used to identify temporal patterns in the number of HEC incidents (including crop-raiding; property damage; injuries and killing of humans; and killing of elephants) occurring. The effects of environmental variables on the number of crop-raiding incidents per month were investigated.

I conducted a generalised linear model (GLM) with a Poisson error structure of number of raiding incidents/month and a GLM with normal errors of the amount of damage measured per month. The following terms were fitted as explanatory variables: rainfall/month and rainfall/previous month for the previous twelve months as continuous variables, as well as month and year as factors.

3.2.6. Comparisons between government and IUCN approaches

To evaluate insights on elephant crop-raiding incidents gained from the IUCN approach with those obtained using government approaches, three comparisons were undertaken. First, the total number of crop-raiding incidents recorded by both approaches, second, the temporal efficiency of the two approaches and finally the measurement of area damaged in a field by each approach.

Total number of raiding incidents and monthly totals of raiding incidents recorded by the two approaches were compared using chi-squared tests and significance determined at $p < 0.05$.

The number of days between the date of an elephant crop-raid and the date the damage was assessed were calculated for the IUCN and government approaches. First, the difference in temporal efficiency between approaches was investigated by conducting a GLM of the number of days between raid and assessment for the IUCN approach, with a quasipoisson error structure. The following terms were fitted as explanatory variables: the number of days between raid date and assessment date for government data, distance from damaged field to main village, and distance of damaged field to DWNP office as continuous variables and village identity as a factor.

A second set of analyses investigated factors affecting the temporal efficiency of the DWNP approach. A GLM of the number of days between raid date and assessment date for government data with quasipoisson error structure was conducted. The following terms were fitted as explanatory variables: distance from damaged field to main village, and distance of damaged field to DWNP office as continuous variables and village identity as a factor.

The measurement of area damaged in a field by each approach was compared. A GLM of the measurement of area damaged by enumerators, with normal error structure, was conducted. The following terms were fitted as explanatory variables: the DWNP measurement of area damaged, the number of days between raid date and assessment date for enumerator data, distance from damaged area of field to main village, and distance of damaged area of field to DWNP office as continuous variables and village identity as a factor.

A second set of analyses investigated variables that may affect the measurements taken by the DWNP approach. A GLM of the DWNP measurement of area damaged, with normal error structure, was conducted. Explanatory variables included: the number of days between raid date and assessment date for DWNP data, distance from damaged field to main village and distance of damaged field to DWNP office as continuous variables and village identity as a factor.

3.2.7. Assessment of factors affecting actual and proportional damage inflicted during an elephant crop-raid

Data collected under the IUCN protocol provided additional information to that collated from government records, which allows further detailed analysis of underlying patterns of crop raiding to be conducted. One example of such an analysis is the assessment of environmental and social factors which may affect the actual or proportional damage inflicted during an elephant crop-raid.

Only data collected using the IUCN approach was used in the subsequent analyses. First, a GLM of area damaged, with a normal error structure was conducted. Second, the percentage of a field damaged was calculated for all raided fields assessed by enumerators and a GLM of a two-vector response variable (proportion of field damaged and proportion not damaged), with binomial error structure, was conducted. The following terms were fitted as explanatory variables for both models: a) distance from damaged area to i) nearest permanent (river) and semi permanent (waterhole) water, ii) next field, iii) main village, iv) main-road, v) main elephant path, and vi) watch huts; b) farmer age; c) farmer family size; d) number of years field established; e) area of field; f) elevation; g) number of watch huts; h) number of guards; i) number of years field previously raided; and j) elephant herd size as continuous variables; and k) mitigation techniques used (coded 1=used, 0=not used); l) crop types grown; m) ethnicity of farmer; n) gender of farmer; o) livelihood of farmer; p) elephant sex; q) elephant estimated age category; r) and predominant vegetation type around the field as factors. Factor levels of the vegetation variable were reduced to simplify the final model without loss of explanatory power.

Initially, all data were used in the analyses, however, this assumes no spatial autocorrelation is present in the data, potentially leading to excessive reductions in standard errors of the parameter estimates (Dormann 2007; Fortin & Dale 2009). Therefore analyses were conducted on a sub sample of data, using methods outlined in chapter four, to remove spatial dependence among samples.

3.3. Results

3.3.1 Characteristics of crop production

A total area of 19,911 ha of land is cultivated for crops in the Okavango Delta panhandle. A variety of crops are grown, including cereal crops (i.e. maize (*Zea mays* L.), millet (*Pennisetum glaucum* L. / *Eleusine coracana* L.) sorghum (*Sorghum bicolour* Pers.) sweet reed (*Sorghum vulgare* Pers.)) and vegetable crops (i.e. watermelon (*Citrullus lanatus* Thunb.), beans (*Vigna aconitifolia* Jacq. / *Phaseolus vulgaris* L.), groundnuts (*Arachis hypogaea* L.), melons (*Citrullus lanatus* Mansf.), pumpkin (*Cucurbita sp*. Duch) and other vegetables).

The mean area of a field assessed in the study area was 3.2 ha (95%CI: 3.02 – 3.32ha), and the predominant crop grown was millet (94% of farmers grew millet). The mean elevation of fields was 992.9m (±19.8m), ranging from 880-1110m above sea level. The median age of a field was 7 years, (range 1 to 89). The age of farmers ranged from 19 – 104 years old (median = 48 years), with 62.8% being female. Five main ethnic groups were identified in the panhandle, 70.4% of farmers were Bahambukushu, 13.4% Bayei, 11.9% Basarwa, 3.2% Baxereku and 0.8% Bakalaghadi, while 0.3% of farmers withheld their ethnicity.

3.3.2. Summary of findings from government and IUCN approaches

3.3.2.1. Crop-raiding

Elephant crop raiding is seasonal in the Okavango Delta panhandle, with most incidents occurring throughout the crop growing season (January – May), although a few incidents of crop damage to vegetable gardens occurred after this season. Using data collected by the IUCN and DWNP approaches combined, a total of 1040 individual fields were identified as raided in the past three years (2008-2010), however the number of incidents and extent of damage recorded by DWNP officers and enumerators differed. Both records show that a greater number of elephant crop-raiding incidents and a greater amount of damage occurred in 2008 crop season compared to 2009 and 2010 (see table 3).

The frequency and intensity of raiding incidents varied across villages in the study area, although results vary between approaches. For example, DWNP records show that Tobera village had the largest number of raids (n=138) and Kauxwi village the least (n = 12), however enumerator data indicate that the village of Seronga had the most cases of raided fields in the study period (n=119) and the village of Gudigwa had the least (n=21). The total amount of crop-land damaged over the three years was significantly less for enumerator records (162.12 ha) than DWNP measurements (850.75 ha), (χ^2 = 468.2, df = 1, p value < 0.01). The percentage of cultivated land in the study area actually damaged by elephant ranged from 0.8% (enumerator records) to 4.3% (DWNP records).

Table 3. Number of raiding incidents, total and mean area damaged from enumerator and DWNP data over the past three years (2008-2010)

Year	Total No. Fields identified as raided	Total No. Raiding Incidents Recorded DWNP	Total No. Raiding Incidents Recorded Enumerator	Total Area Damaged DWNP Measurement (ha)	Total Area Damaged Enumerator Measurement (ha)	Mean Area Damaged per field DWNP in ha (± SD)	Mean Area Damaged per field Enumerator in ha (± SD)
2008	486	309	405	386	119.65	1.3 (±1.0)	0.5 (±0.9)
2009	321	225	198	295	11.08	1.4 (±1.0)	0.1 (±0.4)
2010	233	163	185	170	31.39	1.8 (±1.3)	0.2 (±0.6)
Total	1040	697	788	850.75	162.12	1.4 (±1.1)	0.3 (±0.7)

The IUCN protocol gathered information on both the area of fields and the area of damage, enabling the proportion of damage per field to be calculated. The median proportion of damage per field was 2.02%, with a quarter of raided farms suffering less than 0.4% and a quarter suffering more than 10.1% damage. A small number of fields were totally destroyed (n=23). Millet was the principal crop destroyed by elephants (82.7% of cases), although all crop varieties were affected. I was unable to measure percentage of each crop type taken due to farmers planting numerous crop types interspersed within a field, and hence making it logistically difficult to measure areas of crop type grown. Most raiding incidents were carried out by male only groups (76.2%), while the rest were raided by female led family groups. In fields where footprints were visible (n = 569), perpetrators of crop raiding were classified as adult, adolescent or juvenile. Adult male elephants (77%) were the main perpetrators of raids.

Enumerator data show that elephant groups responsible for crop raiding ranged in size from 1-22 elephants (median = 4) with 97.3% in groups of ≤ 10 animals. Out of 788 raiding incidents that occurred between 2008 - 2010, 161 were carried out by female led family groups (20.7%), 593 by male only groups (76.4%) and 22 by both sexes (2.9%). In 13 raiding incidents elephant group type were not recorded due to lack of evidence (no footprints visible or visual observation made). Over half of the incidents conducted by male only herds involved adult elephants (58%), while 4% involved only adolescent elephants, and 12% involved a mixture. Age classes were not estimated for the remainder of incidents involving male herds.

The DWNP data recorded the amount of compensation paid over the past three years in the eastern Okavango Delta panhandle. A total sum of BWP 238,660.3 (£21,755.72) was paid out between 2008-2010 for elephant damage (either to crops or fence), with a mean amount of BWP378.2 (£34.48) compensation paid per farmer.

3.3.2.2. Human death

DWNP and enumerator records showed that four people have been killed by elephant in the Okavango Delta panhandle in the past five years. Three men near Gunotsoga village and one woman near Beetsha, all victims were over the age of 70 years. All incidents occurred during the day, one incident occurred in a crop field and the others were in the bush. The incident in Beetsha involved a female led herd whereas the most recent incident near Gunotsoga (in 2009) involved a lone male elephant.

3.3.2.3. Elephant death

DWNP records give an overview of the number of elephants killed through problem animal control (PAC). They show that a total of 17 elephants were killed in the past three years in the panhandle (4 in 2008, 6 in 2009 and 7 in 2010). Yet, enumerator records show that a total of 27 elephants were killed through PAC either by farmers or DWNP personnel over the past three years (6 in 2008, 9 in 2009 and 12 in 2010). Enumerator records also showed that all elephants killed in 2010 were male and 69% were >26 years old, whereas in 2009, 27% of elephants killed were female and the rest were males between 10-36+ years.

3.3.2.4. Property damage and injury/killing of livestock

DWNP records show 26 cases of fence damage in 2008 and 2010, and 32 in 2009. A total of BWP1, 268.05 (£115. 59) was given for compensation for fence damage in the past three years.

DWNP reports show that livestock have been attacked and killed by elephant in the panhandle. In 2008, 1 cow and 1 bull were killed, while in 2010, 2 cows and 1 bull were killed and one cow injured.

3.3.3. Temporal patterns of HEC

Crop raiding is the most prevalent type of HEC in the area. The total number of crop raiding incidents (frequency) per month and the amount of actual damage incurred (intensity) per month varied between the three years. From enumerator and DWNP data, it is evident that most raiding incidents occurred in 2008, with a peak in the number of elephant raids in Mar/April 2008 throughout the study area. The largest amount of damage occurred during April and May 2008. In 2009, there were fewer raiding incidents than in 2008 and 2010. Peaks in both numbers of incidents and area of damage occurred in February for both 2009 and 2010. Most raiding incidents took place between 8pm and 6am when it was dark (97%).

Most incidents of elephants being killed through PAC occurred during the crop raiding season (January – April). Fence damage incidents occurred throughout the year, whereas livestock death/injury due to elephants appeared to occur later in the season (August - January). Incidents of people killed by elephant in the last five years occurred towards the end or after the crop raiding season (April-Oct).

The amount of rainfall recorded at the closest reliable Meteorology Department office (Shakawe) show that the 2010 rainy season (Sept/Oct – May/Jun) had the highest rainfall (750.9mm) out of the three years, with 2008 slightly lower (726.2mm) and 2009 the least (470.9mm). The GLM of the total numbers of crop-raiding incidents with Poisson errors retained the month variable as having a significant effect on the number of raiding incidents occurring (χ^2 = 1132.6, df = 11, p value = < 0.01), with number of raiding incidents likely to be less in June and greater in March (mid crop season). Although the variable rainfall/ month did not have a significant effect on the number of crop-raiding incidents, the variable rainfall three months before was retained as having a significant positive effect ($\chi^2 = 142.9$, df = 1, p value < 0.01) (there are likely to be more raids in a month if there was more rain three months previously). However, there are likely to be less raids if there has been higher rainfall one and three months before, as the interaction between rainfall one month before and rainfall three months was retained as having a significant negative effect ($\chi^2 = 102.2$, df = 1, p value = < 0.01).

The GLM of the amount of damage incurred during an elephant raid with normal errors retained the variable rainfall three months before a raid as having statistically significant positive effects on the area damaged, using measurements from both enumerator and DWNP methods (F = 30.1, $df_M = 1$, $df_R = 32$ $p<0.001$ and F = 22.2, $df_M = 1$, $df_R = 35$, $p<0.001$, respectively). The interaction between rainfall three months before and year was retained as having a significant negative effect on area damaged using enumerator records (F = 7.8, $df_M = 2$, $df_R = 32$, $p<0.01$), (indicating there is likely to be less damage if there had been higher rainfall three months before a raid in 2009 and 2010).

3.3.4. Comparisons between insights from IUCN protocol and government records

3.3.4.1. Record of crop-raiding incidents

The total number of raiding incidents recorded per year varied between enumerator and DWNP records (see figure 6). There was a significant difference between the total number of raiding incidents recorded over three years by the enumerator and DWNP systems ($\chi^2= 4.5$, df=1, $p < 0.05$). However, when analysing data per year, there was only a significant difference between DWNP and enumerator records in 2008 ($\chi^2= 15.4$, df =1, $p < 0.05$).

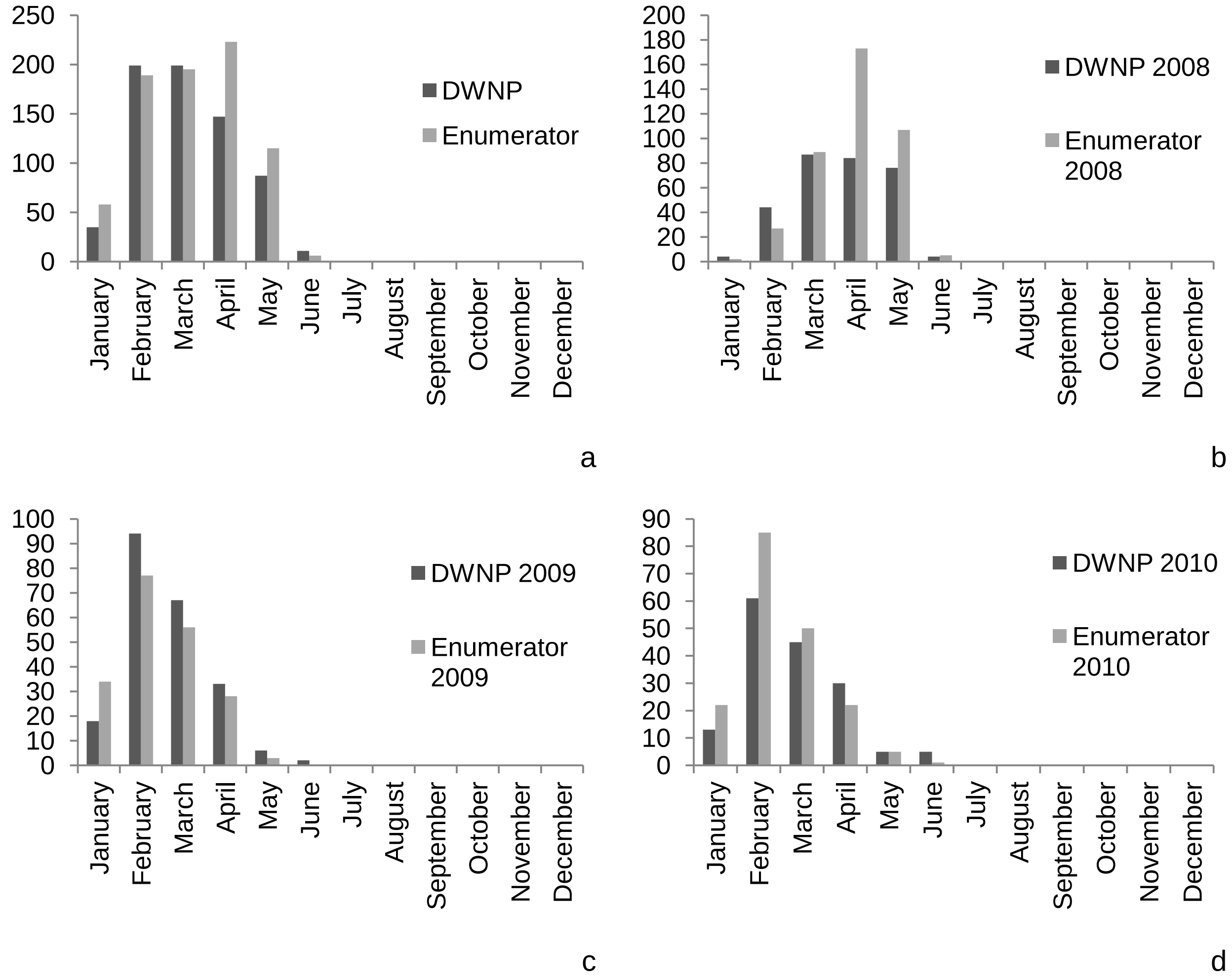


Figure 6. Monthly raiding incidents for a) 3 years 2008-2010, b) 2008, c) 2009 and d) 2010

3.3.4.2. Temporal efficiency

Overall, the mean number of days between the date an incident occurred and the date it was assessed was significantly higher for DWNP records (μ = 30 days), than enumerators (μ = 20 days), (Wilcoxon rank sum test: *W*= 111176.5, df = 1039, p value < 0.05). It is evident that the number of days between raid and assessment date vary between villages for both enumerator and DWNP data (see figure 7).

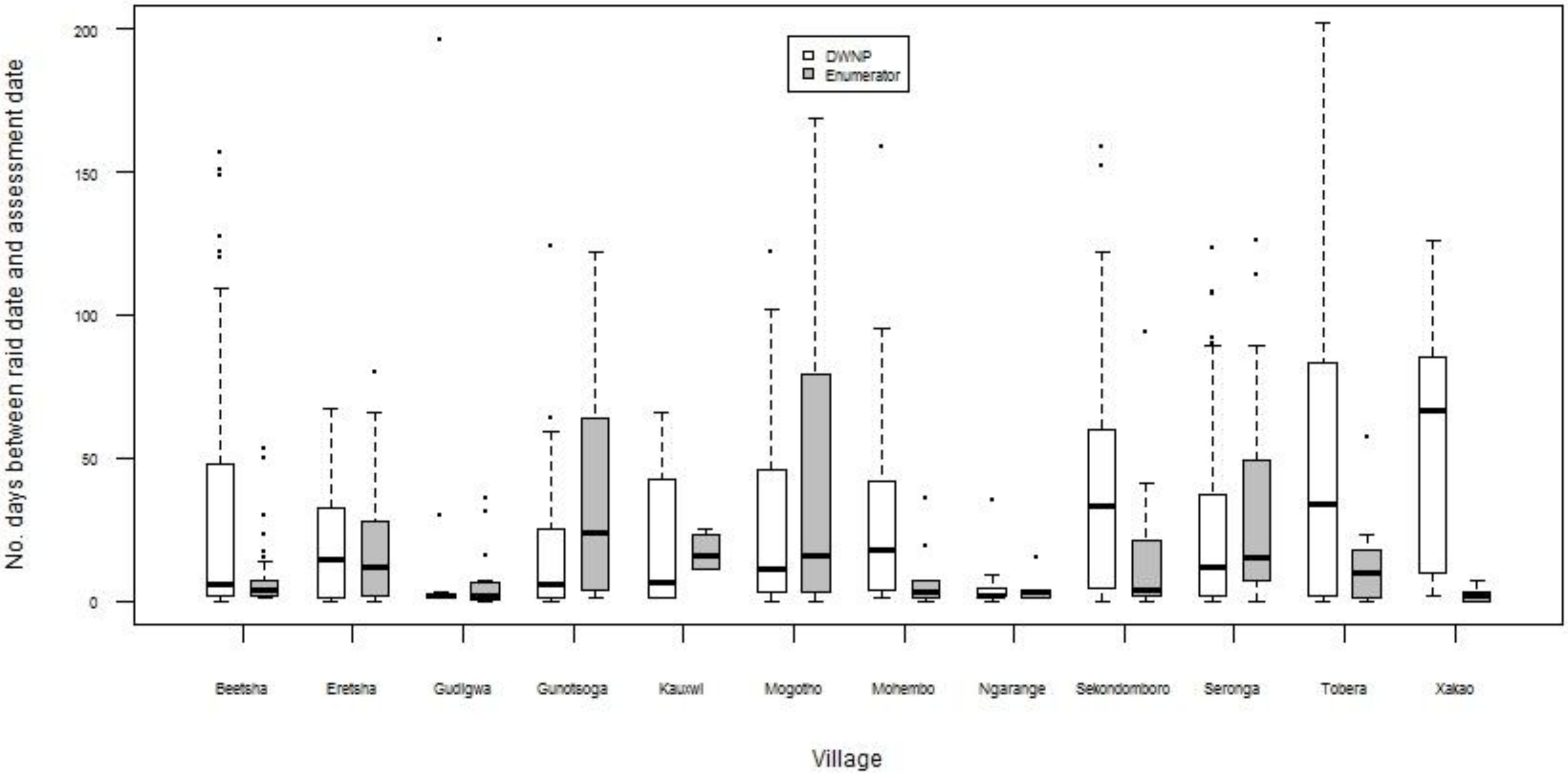


Figure 7. Number of days between incident date and assessment date for DWNP records and Enumerator records within the 12 main villages

A GLM of the number of days between raid and assessment for enumerator data, with quasipoisson error structure, retained the factor village identity (χ^2=-629.59, df = 11, p value < 0.05) as having a significant effect. The villages Gunotsoga, Mogotho and Seronga had significant positive effects (see table 4), indicating that there was a higher chance of the temporal efficiency of data collection being less in these villages.

This model also shows a significant negative interaction between distance to the DWNP office and the number of days it takes DWNP to assess damage in determining the number of days it takes enumerators to assess damage incidents (see table 4).

Table 4. The minimal adequate model for a GLM of the number of days between raid and assessment for enumerator data, with quasipoisson error structure, p<0.05.

Variable	**Estimate**	**SE**	**t value**	**p value**	**χ^2**	**χ^2 p value**
Intercept	1.78	0.49	3.65	0.0004		
DWNP time diff	0.012	0.0045	2.75	0.007		
Dist.DWNP.Office	6.72e-6	1.04e-5	0.64	0.520	0.77	0.86
Eretsha	0.69	0.42	1.44	0.116	723	0.0008
Gudigwa	-0.61	0.84	-0.85	0.476	-	-
Gunotsoga	1.35	0.37	3.53	0.0005	-	-
Kauxwi	0.12	1.57	-0.02	0.941	-	-
Mogotho	1.55	0.42	3.86	0.0003	-	-
Mohembo	0.61	0.81	0.57	0.448	-	-
Sekondomboro	1.01	0.71	1.46	0.159	-	-
Seronga	1.29	0.44	3.31	0.002	-	-
Tobera	0.478	0.61	0.61	0.421	-	-
Xakao	-0.803	1.11	-0.69	0.480	-	-
DWNP time diff:Dist.DWNP. Office	-2.77e-7	1.15e-7	-2.41	0.017	149	0.013

The GLM of number of days between raid and assessment of DWNP data retained the variable distance to DWNP office as having a significant negative effect (less time was taken between report and assessment when villages were closer to the DWNP office), and distance of field to the village as having a significant positive effect (more time was taken between report and raid for fields further from the village), (see table 5). The factor 'village identity' was also retained as having a significant effect (χ^2=-1470, df = 11, p value < 0.05). Gudigwa, Gunotsoga, Mogotho, Seronga and Tobera had significant negative effects (less time was taken between report and assessment in these villages). The model shows a significant interaction between village identity and the distance to the DWNP office (χ^2=-922, df = 11, p value < 0.05) in determining the number of days between raid and assessment in DWNP data.

Table 5. Minimal adequate model for GLM of the number of days between raid and assessment for DWNP data, with quasipoisson error structure, p <0.05.

Variable	**Estimate**	**SE**	**t value**	**p value**	**χ^2**	**df**	**χ^2 p value**
Intercept	8.76	8.311	4.68	4e-6			
Distance to DWNP Office	-1.86e-4	5.4e-5	-3.46	0.0006	68	1	0.196
Distance to Village	0.017	4.8e-3	3.62	0.0003	570	1	0.0001
Eretsha	-3.89	3.71	-1.05	0.30	1470	11	0.0002
Gudigwa	-7.41	2.63	-2.81	0.005	-	-	-
Gunotsoga	-6.68	2.12	-3.14	0.002	-	-	-
Kauxwi	-94	176	-0.54	0.59	-	-	-
Mogotho	-7.23	2.11	-3.42	0.0007	-	-	-
Mohembo East	-5.83	27.2	-0.22	0.83	-	-	-
Ngarange	-9.83	24.6	-0.40	0.69	-	-	-
Sekondomboro	2.44	6.36	0.38	0.70	-	-	-
Seronga	-7.16	1.87	-3.82	0.0002	-	-	-
Tobera	-20.7	7.45	-2.78	0.006	-	-	-
Xakao	-140	98.6	-1.41	0.16	-	-	-
Distance to DWNP Office: Eretsha	7.6e-5	1.3e-4	0.915	0.56	922	11	0.014
Distance to DWNP Office: Gudigwa	1.9e-4	6.2e-5	3.042	0.002	-	-	-
Distance to DWNP Office: Gunotsoga	0.15e-4	8.1e-5	2.925	0.06	-	-	-
Distance to DWNP Office: Kauxwi	0.2e-3	2e-3	0.903	0.56	-	-	-
Distance to DWNP Office: Mogotho	0.9e-4	5.8e-5	3.547	0.001	-	-	-
Distance to DWNP Office: Mohembo East	1.8e-4	3.1e-4	0.267	0.56	-	-	-
Distance to DWNP Office: Ngarange	2e-4	4e-4	0.805	0.63	-	-	-
Distance to DWNP Office: Sekondomboro	6e-5	1e-4	0.105	0.55	-	-	-
Distance to DWNP Office: Seronga	1.6e-4	6e-5	3.225	0.006	-	-	-
Distance to DWNP Office: Tobera	3.8e-4	1.1e-4	3.376	0.0006	-	-	-
Distance to DWNP Office: Xakao	2e-3	1.3e-3	1.436	0.14	-	-	-

3.3.4.3. Comparison of measurements of area damaged

Measurements of fields damaged during elephant crop-raids were taken by the government and IUCN approaches independently. There was a significant difference in the measurement of mean area damaged per incident recorded by the two approaches (t = 21, df = 1070, $p < 0.05$), with DWNP records showing a higher mean area of damage per incident (1.39ha, 95%CI: 1.33-1.46ha) than enumerator records (0.29ha, 95%CI: 0.25-0.32ha). When DWNP measurements of area damaged were plotted against measurements taken by enumerators (see figure 8), it is evident that DWNP measurements are more frequently larger than measurements taken by enumerators in the same field. Linear regression showed that DWNP measurements of area damaged and enumerator measurements were positively correlated ($r^2 = 0.019$, $F_{263} = 5.1$, p value <0.05), however I would have expected a stronger correlation (i.e. $r^2 = 1$) due to measurements being conducted on the same damaged areas.

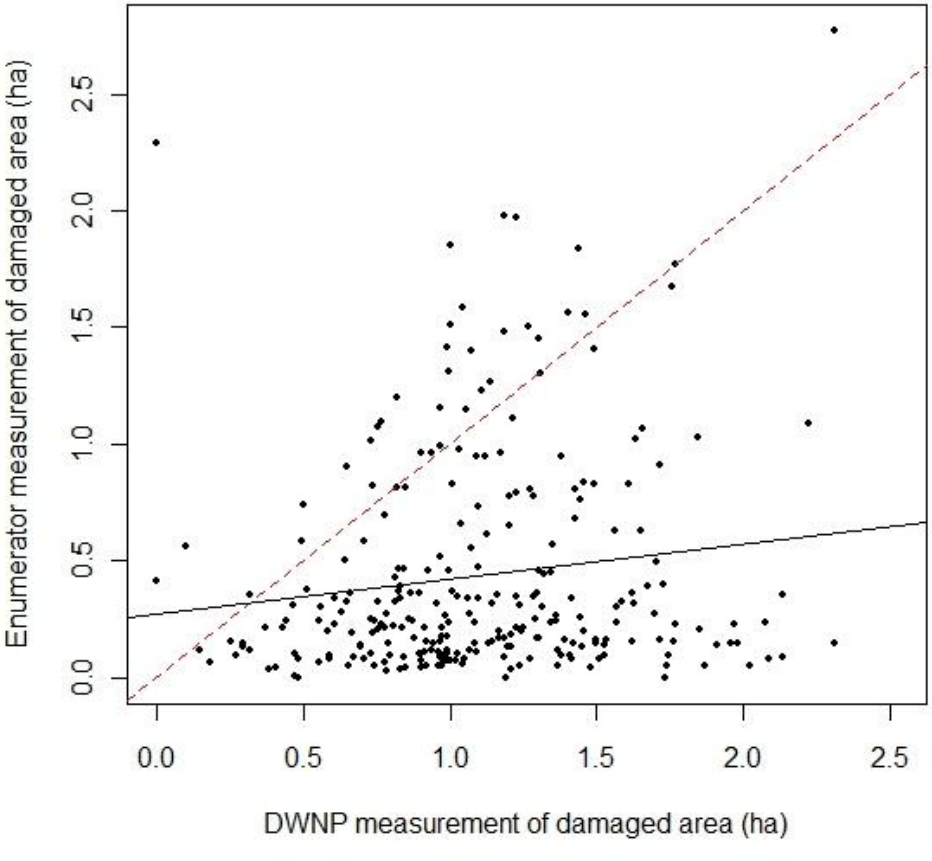


Figure 8. Relationship between area of damage per incident from DWNP records and area of damage per incident from community enumerator records (black line shows actual regression line with $r^2 = 0.019$; red dotted line shows expected regression line if $r^2 = 1$).

The difference between measurements taken by DWNP and enumerators was investigated further to explore variables which may influence the reliability of measurements taken. The minimal adequate model of the GLM of enumerator measurement of area damaged, with normal error structure, retained the variable distance to DWNP office as having a significant positive effect (F = 9.4, $df_M = 1$, $df_R = 568$, p value <0.01) (enumerator measurement of damaged area is likely to be bigger if fields are far from DWNP office).

The minimum adequate model for the GLM of DWNP measurement of area damaged, with normal error structure, retains the variables distance to DWNP office (F= 20.3, $df_M = 1$, $df_R = 366$ p value <0.01) and number of days between raid and assessment for DWNP data (F = 6.4, $df_M = 1$, $df_R = 365$, p value = 0.01) as having significant positive effects (DWNP measurement of damaged area is likely to be bigger if fields are far from DWNP office or if there is a large number of days between raid and assessment).

3.3.5. Assessment of factors affecting area of damage incurred during an elephant crop-raid

Univariate analysis showed that the area of damage inflicted by elephants once they have entered a field and the relative proportion of a field damaged during an elephant raid appear to be influenced by a number of variables, including: group size and sex of perpetrating elephants; raiding history of the field; location, age and elevation of the field; mitigation used; crop types grown; presence of other crop-raiding animals; and surrounding vegetation type. The size of cultivated land appears to affect the area of damage but not the proportion of a field raided.

The minimal adequate model for the GLM of area of damage incurred once elephants have entered a field, with normal error structure, retained the variables: distance of a field to the river, number of elephants in the perpetrating herd, number of years a field had been raided in the past, presence of beans and the use of bush fence for mitigation as having statistically significant positive effects (fields further from the river, or with larger elephant herds raiding, or those more frequently raided in the past, or those growing beans, or those using bush fences were likely to have more damage inflicted). The model shows that an interaction between presence of beans and presence of bush fence had a significant negative effect in determining the area of damage incurred during a raid (fields growing beans with bush fences were less likely to have large areas of damage), (see appendix VII).

The minimal adequate model of the GLM of the proportion of a field damaged, with binomial error structure, retained the variables: number of years previously raided and number of perpetrating elephants, as having significant positive effects (fields with a larger number of elephants raiding and fields which have been raided more often in the past are more likely to have a larger percent of the field damaged by elephant). The variables area of cultivation and surrounding vegetation type were retained as having significant negative effects (larger fields and fields surrounded by mixed vegetation type are more likely to have a smaller proportion of the field damaged by elephants), (see appendix VIII).

The results from the previous two GLMs use all enumerator data from all raided fields in the study area over the past three years and therefore spatial autocorrelation could affect parameter estimates and lead to overstated correlations (Sadoti, Rodhouse & Vierling 2010). When the minimum adequate model was fitted to sub sampled data at smaller spatial scales, it was evident that only the number of elephants in a raiding group has a statistically significant effect on the area damaged and the proportion of damage occurring once elephants have entered a field at the 0.5km^2 scale (see appendices VII and VIII). At coarser sampling scales, no variables are statistically significant for either model.

3.4. Discussion

3.4.1. Comparing insights between IUCN and government approaches

Based on the criteria used to compare IUCN and government approaches in this study, it is evident that both approaches provide sufficient general information on the current extent of HEC incidents in an area that can be used to produce overall status reports, but there are significant differences between the two approaches that lead to discrepancies in results.

The IUCN approach recorded a larger number of crop damage incidents than the DWNP system and could, therefore, be considered more comprehensive. However, recording all incidents of elephant damage does not necessarily mean that a system is more efficient in terms of assessing the current intensity of raiding, because some incidents may have incurred little or no damage. Osborn, (1998a) described elephant crop-raiding incidents in two categories ‘visits’ and ‘raids’, with visits being cases where elephants traverse fields with little damage and raids being cases where crops are eaten by elephants causing greater amounts of damage. Through the IUCN approach, incidents are classified according to intensity (i.e. visit or raid) (Hoare 1999a). My enumerator records showed that 9% (n=74) of cases they assessed in the past 3 years could actually be classified as visits rather than raids in the panhandle, yet all incidents were assessed for damage and included in subsequent analyses. If I exclude incidents classified as visits, however, then there is no longer a significant difference between the number of raids recorded by the two approaches (χ^2= 0.9, df=1, p value = 0.35). Unfortunately, the DWNP records do not identify incidents according to intensity so we cannot substantiate this explanation, but the discrepancy does illustrate the importance of having a clear and standardised data collection and recording system of incidents to facilitate viable comparisons between studies.

The IUCN approach used the same reporting system as that used by the government to locate damaged fields, i.e. farmers report HEC incidents to the village chief, police or DWNP officer, so I would really expect the number of incidents to be the same, regardless of whether they were actually classified as visits or raids. A further explanation for this difference could be that some farmers may report incidents outside of the government reporting system, i.e. directly to an enumerator/DWNP officer at the field. Enumerators were allowed to assess damage immediately, yet DWNP officers can only attend incidents once they have been reported through the correct procedure. Once farmers have reported incidents to a DWNP officer/enumerator they may be reluctant to go and report the incident again through the required procedure and, therefore, such incidents would not appear in the DWNP records. My enumerators encouraged all farmers to report incidents to DWNP if they desired to get compensation, but some farmers do not feel that the compensation is adequate (Mosojane 2004; Jackson *et al.* 2008) and, therefore, may have decided not to re-report incidents resulting in lower numbers of reported incidents recorded by DWNP officers.

It is also evident from our assessment of the two approaches that the time it takes for incidents to be assessed influences the efficiency of damage assessments, because the evidence of the perpetrating wildlife species diminishes and the damage itself is harder to observe after time (Stewart 2004). It is, therefore, important to maximise temporal efficiency for accurate data collection of damage incidents. The number of days between the date of incident and date of assessment of elephant crop-raiding incidents was significantly higher using the DWNP system, indicating that the IUCN approach was temporally more efficient. This can be explained by the increased efficiency with which enumerators took both reports from farmers and damage incident assessments. DWNP officers are based in Seronga village and have to travel to individual villages to collect reports and assess damage incidents, which generally requires more time. My results confirm this by indicating that villages further from Seronga had a greater number of days between raid and assessment among DWNP data. Interestingly, the temporal efficiency of enumerator data also varied between villages. This result, however, is more likely to be attributed to individual enumerator efficiency. The least efficient villages were identified as Gunotsoga and Mogotho, where challenges with human resources were experienced (one enumerator disappeared for a short period and the other enumerator suffered from languor). This highlights one potential challenge of using the IUCN approach for wide scale HEC monitoring; the selection and management of enumerators.

Measurements of damaged areas taken by the two different approaches in the same fields showed differences, with DWNP records being significantly larger. This is an interesting result because I would have expected there to be no significant difference between two measurements from the same field. One explanation for this disparity could be that the measuring techniques used by the two approaches differed. DWNP officers used a surveyor's wheel (which should provide a more accurate measure of area) and enumerators used a pacing method to estimate area of damage. However, estimates of damage should still have been similar. It is more likely that the pattern of elephant damage within a field affected the measurement of area damaged. When an elephant damages a field it often damages patches of crops rather than whole areas. If damaged patches are grouped together then estimates of damage can be highly exaggerated, which is why enumerators are encouraged (under the IUCN approach) to measure damage in patches (Hoare 1999a). If DWNP officers group damaged patches together during their field assessments, however, then larger estimates of areas damaged would be expected.

In addition, discrepancies between the two methods of measuring the area of damage may result from inaccuracies within the DWNP's method owing to the time-lag period of measurement. If there is a large amount of time between the raid report date and assessment date, the evidence of damage is likely to be limited and damage assessments can be difficult. In such cases, it may be necessary to rely on the farmer's perception of damage extent. It has been found, however, that if damage assessments are based on farmers' perception, records may be overinflated (Sekhar 1998; Siex & Struhsaker 1999; Gillingham & Lee 2003; Linkie 2007) and caution is needed. It is not certain whether the DWNP system occasionally relied on farmer estimates for damage assessments, but, if this were the case, it could explain the larger estimates of areas damaged. Regardless, it is clear that damage assessments in any monitoring system should be based on actual evidence rather than farmer perception.

There is also a possibility that damage assessments could be influenced by socio-economic factors or policies such as the government compensation scheme. If this was the case we may have expected enumerator records to show larger measurements of damage, because they were assessing fields of people they know in their own village who they would likely assist in gaining maximum compensation for their loss. To avoid such bias in results, the IUCN approach advises that enumerator measurements be verified by an independent assessor (Hoare 1999a). This was carried out by our research team in every field assessed to verify the amount and intensity of the incident, but no such verification is currently enforced in the DWNP system.

The location of a field also appeared to affect the quantity of damage recorded by the two different approaches. Fields further from the DWNP office had larger amounts of damage recorded by both DWNP officers and enumerators. The DWNP office is located in Seronga village, the largest settlement in the study area. This result could be explained by the effect of human disturbance on the spatial pattern of elephant raiding rather than the monitoring approach used. Sitati *et al.*, (2003) found in Kenya, that crop-raiding by male elephants could be predicted by proximity of a field to major settlements. Similarly, results from another part of this study also showed that fields further from a village are more susceptible to elephant raiding (see chapter four). Interestingly though, there was no significant relationship found between distance to a village and the area of actual or amount of relative damage incurred during a crop-raid in our study. Our findings suggest that intensity of damage may be affected more by size of settlement rather than distance to the settlement, yet Hoare, (1999b) found that problem elephant activity levels were not significantly related to area of human settlement in Zimbabwe. The effect of size and spatial extent of human settlements on elephant crop-raiding incidents warrants further investigation.

Human-elephant conflict situations are complex and often require a multifaceted approach to gain adequate understanding and design appropriate management interventions. As Messmer, (2000) discussed, conflicts can be either real or perceived, economic or aesthetic, social or political. Management and monitoring strategies for HWC that incorporate broader issues are, therefore, likely to have greater impact in successfully quantifying and assessing the impacts of HWC. In this respect, the IUCN protocol has a major advantage over the DWNP approach because data collected under the IUCN protocol includes additional information on environmental and social factors influencing conflict. In addition, the IUCN approach encourages community participation and empowerment. For this reason, it could indirectly assist in reducing the intensity of conflict between humans and elephants. Increasing benefits for people living close to wildlife can increase tolerance towards wildlife conservation and protected areas (Sekhar 1998; Smith & Kasiki 1999; Hill 2004). Therefore, by providing much needed employment opportunities to local communities affected by HEC, the IUCN approach presents a direct benefit to the community which may increase tolerance towards elephants and HEC.

There are, however, logistical and financial restraints with involving local enumerators that need to be considered. Enumerators need to be recruited, trained and employed, which requires personnel and management, as well as sustainable finances. An independent assessor is also required to verify damage assessments to reduce the probability of subjective data being collected. An enumerator based system, therefore, requires financial input and management to keep it sustainable, whereas the government system is implemented by government employees and financed through the Ministry of Environment, Wildlife and Tourism. Support and partnership with the communities by a third party local or international non-governmental organisation (NGO) could be one way to help establish and facilitate a sustainable long-term monitoring system involving community enumerators in Botswana. With improved benefits from the wildlife resource, communities could also invest in such compensation schemes, a system which appears to be effective in neighbouring countries, e.g. the Integrated Rural Development and Nature Conservation (IRDNC) Human Animal Conflict Insurance Scheme (HACIS) in Namibia (Stewart & Diggle 2004).

3.4.2. Patterns of HEC

A major advantage of the standardised IUCN monitoring approach is that results are comparable with studies elsewhere in the elephant range, thus facilitating a broader perspective of the extent of human-elephant conflict in a specific study area. From the data collected under the IUCN protocol in this study it is evident that, contrary to previous reports (e.g. Natural Resources & People 2006b), intensity of crop raiding in the Okavango Delta panhandle appear to be lower than has been recorded in other African countries. The median percentage of damage per raided field was low in this study (2%) compared to Kenya (37.5%), (Sitati, Walpole & Leader-Williams 2005) and Gabon (46%), (Sam *et al.* 2005). A quarter of fields raided in one year in Kenya suffered >66.7% damage (Sitati, Walpole & Leader-Williams 2005), whereas only 7% of fields raided in the Okavango panhandle over three years experienced damage of over 50% of the field. Even though results show proportional crop damage as being low in this study area, as Warren, Buba & Ross, (2007) highlight, this does not mean that people and elephants should be able to co-exist harmoniously. Despite these low levels on a general scale, some fields (n=23) still experienced 100% damage illustrating that elephant raids can have a catastrophic effect on the livelihood of subsistence farmers at the individual field level in the Okavango Delta. Such a result, illustrates the importance of considering HEC on a number of scales when forming conservation policies and mitigation strategies; at the individual farmer level, the community level, as well as regional and international levels.

The number and intensity of HEC incidents varied between the three years of this study, verifying the importance of collecting data over multiple years as identified by Hoare, (1999b). In all three years, elephant raiding exhibited some general temporal patterns found in similar studies (e.g. Sukumar 1990; Hoare 2000a; Osborn 2004; Barnes 2005; Chiyo *et al.* 2005; Fernando *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Warren, Buba & Ross 2007) with raids predominantly occurring during the hours of darkness and raiding being seasonal. Most raiding incidents in 2008 in the Okavango Delta panhandle occurred in the late wet season (April-May) when crops are ripening, which reflects a seasonal pattern in crop-raiding incidents found in comparable studies (Hoare 1995; Kangwana 1995; Kiiru 1995; Tchamba 1995; Hoare 1999b). Yet in 2009 and 2010 most crop damage incidents occurred during February when crops were at the seedling stage. This indicates that the key driver for crop-raiding in the panhandle is unlikely to be the ripening of crops but rather it is determined by other ecological or social factors. For example, from our analysis of the incidents of crop raiding in the Okavango Delta panhandle, the probability of raiding occurring and the amount of damage incurred during a raid were affected by rainfall in the previous months.

The type of data collected under the IUCN approach also allowed me to investigate factors affecting the intensity (area and proportion of damage) of elephant raids. Evidently, the group size of crop-raiding elephants was the most robust key driver of raiding intensity in the Okavango Delta panhandle, with more elephants likely to damage a larger area and a higher proportion of a field during a raid. Similar results were found in Kenya (Sitati, Walpole & Leader-Williams 2005). However, when there were large groups of elephants raiding fields in certain vegetation types (Mixed woodland, Silver terminalia and Umbrella thorn), smaller proportions of the field were damaged than expected in our study area. This could be explained by the nutritional quality of natural forage around the fields providing better natural forage at this time (Sukumar 1990; Osborn 1998b, a, 2004), which i) could have attracted larger groups of elephants to this vegetation type and ii) reduced the amount of crops selected for forage.

Male elephants were more frequently the perpetrators of crop raiding in the Okavango Delta panhandle, a phenomenon that has been recorded in numerous other studies in both Africa and Asia (Bell 1983; Sukumar & Gadgil 1988; Sukumar 1991; Thouless 1994; Osborn 1998a; Hoare 1999b; Sitati *et al.* 2003; Chiyo *et al.* 2005). One explanation for this, based on the optimal foraging theory (Krebs & Davies 1991), is that male elephants may have a greater inclination to take risks to derive better nutrition for enhancing reproductive success (male behaviour hypothesis) (Sukumar 1991), and are therefore more likely to utilise areas close to human activity. Contrary to other studies (Sukumar 1990; Hoare 1999b; Chiyo *et al.* 2005), however, my study showed that repeat raiding within a season did not appear to be common in the panhandle. This difference may be attributed to most raiding incidence being carried out by only adult male elephants, rather than post-pubertal/adolescent males who may be learning to crop raid and hence repeat raiding excursions (Chiyo *et al.* 2005).

3.4.3. What is the best approach for monitoring human-wildlife conflict?

This study highlights advantages and disadvantages of both the IUCN protocol and a government (top-down) approach. However, I cannot conclude that one method is better than the other because the effectiveness of a method depends on the purpose of data collection. For example, if the purpose of the monitoring is to provide a record of incidents for compensation payments, and compensation is administered on account of the area of a field damaged, then it could be argued that damage assessments should be as accurate (as close to the true area damaged) as possible to ensure accurate compensation payments. However, if compensation is paid on reflection of proportional damage, i.e. percentage of total field damaged (IRDNC 2004), then the accuracy of measurements is less important as long as all fields are measured consistently. Similarly, if the purpose of monitoring is to assess trends in conflict incidents then assessments need to be precise but not necessarily highly accurate. However, if the purpose of monitoring is to assess the key drivers of actual and proportional damage inflicted during a raid then true (accurate) measures of area damaged are needed.

The IUCN approach provides a standardised data collection protocol which is advantageous if comparisons are to be made between HWC situations on an international level. However, the government approach in Botswana may be adequate to allow comparisons to be made on a local or national level as long as DWNP officers are consistent in their implementation of monitoring techniques. Although the government system uses a surveyor’s wheel to quantify damage, it is still unclear what measurements are actually the most accurate estimates of damage due to patterns of elephant damage within a field. The IUCN approach does encourage assessments to be verified by an independent assessor and could therefore be considered to be a less biased estimate of damage. Damaged area measurements were significantly smaller using the IUCN methodology, which could be advantageous for the government in reducing compensation pay outs, but disadvantageous to farmers who are claiming compensation.

A system that collects data on environmental and social factors of HEC incidents can facilitate more detailed investigations into temporal and spatial patterns of HEC and ultimately assist the development of successful mitigation strategies. Collection of such data could therefore improve the current government approach, if temporal and spatial patterns of HEC need to be monitored. Evidently, the temporal efficiency of the government system could also be improved if community enumerators (who are based in each village) were employed, which would reduce the time between raid and assessment and may enhance the accuracy of damage assessments. This would be beneficial for the current compensation scheme where area of damage needs to be measured accurately. In addition, community participation could foster trust and acceptance of the compensation scheme that currently suffers from negative perceptions among farmers who feel they have not been compensated sufficiently (see chapter six). However, there are logistical and financial constraints that need to be considered before employing community enumerators.

Involving community members in the monitoring of HWC incidents as suggested by the IUCN approach does, however, have advantages beyond the level of data collection efficiency. Such a system directly increases benefits for people living close to wildlife which can aid in increasing tolerance towards wildlife, thereby, reducing the perceived conflict people are experiencing while increasing acceptance of mitigation strategies. Such a scenario was evident when communities became more involved in monitoring HWC incidents and implementing mitigation strategies in Namibia through IRDNC programmes (Stewart & Diggle 2004) and in the CAMPFIRE project in Zimbabwe (Songorwa 1999). Experience from such studies are generally positive, but the effectiveness of community involvement does need to be monitored (Inamdar *et al.* 1999; Songorwa 1999; Redford & Sanderson 2000; Barrett *et al.* 2001; Berkes 2004; Stewart & Diggle 2004).

Ultimately, it appears that both approaches are effective for general monitoring of human-elephant conflict incidents as long as the purpose of data collection is clearly defined. Aspects such as data quality, temporal efficiency, economic implications and social elements need to be considered in the design of more effective monitoring approaches, to ensure adequate data are collected for consequent management activities.

Chapter 4

Identifying key drivers of elephant crop-raiding and the effect of spatial autocorrelation on such data

4.1. Introduction

One of the biggest challenges in conservation today is managing situations where people and wildlife utilise the same space and compete for similar resources (Balmford *et al.* 2001; Sitati, Walpole & Leader-Williams 2005; Woodroffe, Thirgood & Rabinowitz 2005). As human settlements grow and protected areas become surrounded by human dominated landscapes, human-wildlife conflict (HWC) increases and involves a growing number of wildlife species, especially large mammals (Linnell *et al.* 1999; Naughton-Treves, Grossberg & Treves 2003; Sitati, Walpole & Leader-Williams 2005; Hegel, Gates & Eslinger 2007). Elephants in Africa (*Loxodonta sp.*) and Asia (*Elephas maximus*) (Sukumar & Gadgil 1988; Lahm 1996; Hoare 2000b; Osborn 2004; Chiyo *et al.* 2005; Sitati, Walpole & Leader-Williams 2005), elk (*Cervus elaphus canadensis*) in Canada (Hegel, Gates & Eslinger 2007) and tigers (*Panthera tigris sp.*) in Asia (Saberwal 1997; Sekhar 1998; Tilson & Nyhus 1998; Linkie *et al.* 2003; Nyhus & Tilson 2004a, b; Johnson *et al.* 2006) are prime examples of large mammals that frequently come into conflict with people. Large mammals generally require a large amount of space (Owen-Smith 1988) and due to their physiology and energy requirements, also need to consume large quantities of food and water each day (Laws & Johnstone 1970; Owen-Smith 1988; Sukumar 1990; Ruggiero 1992; Choudhury *et al.* 2008). It is not surprising, therefore, that when these animals live in areas surrounded by a burgeoning human population, they frequently compete with humans for limited resources such as space, water and food (Hoare 2000b; Conover 2002; Hegel, Gates & Eslinger 2007). Despite an increase in the extent of human-wildlife conflict situations (Hoare 1999b; Madden 2004), a major constraint on the development of effective management interventions to reduce such conflict is the inability to effectively predict where conflict is going to occur or in the case of crop-raiding, for example, what makes a field susceptible to attack (Smith & Kasiki 1999; Sitati *et al.* 2003; Hegel, Gates & Eslinger 2007).

Often, wildlife management authorities and community members tackle wildlife crop damage by using short term mitigation measures, such as barrier methods (i.e. bush, wire and electric fences); olfactory repellents (i.e. chilli peppers); and acoustic deterrents (i.e. drums and bangers) (Conover 2002). Although such methods are useful as immediate defensive measures to protect fields from attack (Osborn & Parker 2003) and in reducing negative perceptions of wildlife (Hill 2000), this approach is symptomatic and does not address the underlying causes of the problem (Jackson *et al.* 2008). HWC prevention in both the short and long term requires an understanding and identification of conditions promoting HWC and an ability to focus assistance and mitigation accordingly (Treves *et al.* 2004).

One of the main underlying causes of conflict between people and large mammals is competition for space (Hoare 2000b) and, therefore, one long term mitigation strategy to reduce conflict could be to develop more effective land use planning, e.g. zoning and/or fencing certain areas (Mishra 1971; Taylor 1993; Hoare 1995; Jackson *et al.* 2008; Chase & Griffin 2009). A trial carried out in Kenya, however, found that these options are often costly and require extensive management (Taylor 1993). Allocation of land is usually determined by soil fertility, with the most fertile soils being dedicated to agriculture and livestock production, and the least fertile soils to non-agricultural uses (Martin & Taylor 1983; Happold 1995). Future land use planning, however, should take into consideration spatial determinants that affect the susceptibility of crops and agricultural land to raiding by wildlife, in order to minimise the risk of conflict occurring between people and wildlife (Hoare 2000b; Fernando *et al.* 2005). For example, Fernando *et al.* (2005) found that an area with a designated protected area for elephants combined with traditional slash and burn agriculture had less crop-raiding and facilitated co-existence between elephants and people compared with an area comprising a mosaic of settlements, agriculture and small patches of forest.

Crop-raiding by wild animals occurs all over the world, albeit with different species perpetrators. Studies conducted on a variety of wildlife species causing crop destruction or livestock predation have investigated factors and determined patterns that may help in predicting cases of human wildlife conflict (Tourenq *et al.* 2001; Treves *et al.* 2004; Chiyo *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Woodroffe, Thirgood & Rabinowitz 2005; Cai *et al.* 2008; Graham *et al.* 2010). Despite a variety of research techniques used, common themes arising from these studies indicate that temporal and spatial patterns of wildlife crop-raiding appear to vary across space and time (Goldman 1986; Dudley, Mensah-Ntiamoah & Kpelle 1992; Hillman-Smith *et al.* 1995; Lahm 1996; Naughton-Treves 1998; Weladji & Tchamba 2003; Warren, Buba

& Ross 2007) and may be influenced by animal density (Naughton-Treves 1998; Hegel, Gates & Eslinger 2007), area of cultivated land (Tourenq *et al.* 2001), and/or the location of fields in relation to landscape features such as water availability, protected areas, and wildlife habitat, (Lahm 1996; Hill 1997; Naughton-Treves 1998; Smith & Kasiki 1999; Hill 2000; Saj, Sicotte & Paterson 2001; Linkie 2007; Cai *et al.* 2008). Other factors, such as, season, amount of guarding, human density, hunting, isolation of fields and crop types available, have also been found to effect the degree of damage a field receives (Atteh 1984; Newmark *et al.* 1994; Naughton-Treves 1998; Hill 2000; Warren, Buba & Ross 2007).

HWC is a spatial phenomenon and it is, therefore, important to investigate the effects of spatially explicit factors on its distribution (Smith & Kasiki 1999). For example, a variety of factors have been identified in studies in east and west Africa, which may influence spatial patterns of crop-raiding by elephants (Smith & Kasiki 1999; Sitati *et al.* 2003; Barnes 2005; Sitati, Walpole & Leader-Williams 2005; Graham *et al.* 2010). Smith & Kasiki (1999) describe a GIS-based analysis of HEC data from a region in east Africa and found that HEC incident density in the study blocks was significantly and negatively related to their mean distance to permanent water, elevation and the perimeter that they shared with the national parks. Sitati, Walpole & Leader-Williams (2005) used multivariate statistics to investigate a variety of factors that may make a farm more susceptible to being raided by elephants in the Transmara District, Kenya, and their results showed that larger farms and those that were bordered by hedges or fences were more likely to be raided by elephant. Farms that had been raided in the past were also more likely to be raided and greater guarding effort, with fire and noise, decreased the likelihood of successful crop raiding by elephant. Barnes *et al.* (2005) found that four farming variables increased the risk of a farm being raided by elephant in Kakum Conservation Area in Ghana; distance to the national park boundary, area of cultivation, number of crops planted on the farm and the degree of the farm's isolation. A recent study in Kenya found that the occurrence of crop-raiding was predicted by settlement density, distance from daytime elephant refuges and percentage of cultivation (Graham *et al.* 2010).

It is evident from these studies that the occurrence of elephant crop-raiding appears to be influenced by the field location, farmer mitigation effort, and number and types of crops planted. It is currently unknown, however, what role elephant movement corridors play in determining the distribution and intensity of elephant crop-raiding (Sitati *et al.* 2003). Incidents of human wildlife conflict, particularly elephant crop-raiding are rarely randomly distributed over space. They usually show some degree of spatial clustering. An elephant may raid more than one field on a particular night, and their behaviour towards a field may be influenced by factors from neighbouring fields (i.e. mitigation measures used). When such observations, drawn from different locations, are not independent from one another they can be described as spatially dependent and spatial autocorrelation arises (Cliff & Ord 1981). Positive spatial autocorrelation (data collected at locations closer together are more similar) makes parametric statistical tests too liberal and they can often produce more apparently significant results than the data actually justify (Cressie 1993; Fortin & Dale 2009). Linear estimators, correlation coefficients and variances can also be affected by the presence of spatial autocorrelation (Cliff & Ord 1981; Dutilleul 1993). To understand the spatial patterns of elephant crop raiding incidents one needs to be able to separate the spatial structure of data due to environmental factors from that due to spatial autocorrelation generated by the processes themselves (Smith & Kasiki 1999; Sitati *et al.* 2003).

Given previous findings I chose to collect data on attributes and position of fields, crop types grown, farmer characteristics, and mitigation measures used, in an attempt to elucidate key drivers of crop raiding by elephants in Botswana. In my analyses I also included variables (i.e. distance from main elephant pathways), which haven’t previously been investigated. The objectives of the study were to a) examine whether raided fields were distributed randomly in the study area, b) identify characteristics of elephant crop raiding in the study area, c) investigate factors affecting the susceptibility of a field to elephant raiding, and d) determine the effect of spatial autocorrelation on HEC data.

4.2. Methods

4.2.1. Study area

The study was conducted on the eastern side of the Okavango Panhandle, where the Okavango River reaches the Okavango Delta in Botswana, between January 2008 and July 2010 (see figure 9). The eastern panhandle covers an area of 8,732 km^2 and consists of controlled hunting areas NG11, NG12 and NG13. The Namibian border marks the northern boundary and the northern buffalo fence marks the southern boundary (UTM Zone 34 7910000 – 7990000 South and 580000 – 710000 East). Protected areas occur north of the Namibian border fence, namely Babwata National Park and Mahango Game Reserve. South and east of the northern buffalo fence are wildlife management areas, which are utilized by photographic and hunting tourism operations.

Rainfall averages 360 - 500 mm annually and generally falls between mid October and March (rainy season), (Department of Meteorology, 2010). The mean monthly maximum temperatures in the panhandle range from 26.1 ºC– 35.1ºC, with July being the coldest month and October the hottest (Department of Meteorology, 2010). Deep Kalahari sands dominate throughout the study area, and main vegetation types include mopane woodland (*Colophospermum mopane*), silver terminalia woodland (*Terminalia sericea*), two Acacia vegetation types, namely Candle pod acacia (*Acacia hebaclada*) – Buffalo thorn (*Ziziphus mucronata*) and Umbrella thorn (*Acacia tortillis*) – Combretum (*Combretum imberbe*) valley sandveld, Zambezi Teak (*Baikiaea plurijuga*), False mopane (*Guibourtia coleosperma*), Burkea (*Burkea Africana*), Riverine woodland (e.g. *Diospyros mespiliformis; Croton megalobotrys; Acacia nigrescens; Lonchocarpus capassa; Kigelia africana*), and mixed woodland (*Colophospermum mopane*, *Terminalia sericea*, *Acacia sp.*), (Mendelsohn & Obeid 2004). Subsistence agriculture occurs in lower depressions near the Okavango River and floodplains (Tawana Land Board 2005) and in land <14km from the main road where there are fertile soils.

The 2001 census recorded 15,718 people living in the eastern panhandle (CSO 2001). There are 12 main villages (population > 500 people) in the area, extending from Mohembo-East to Gudigwa, including Kauxwi, Xakao, Tobera, Sekondomboro, Ngarange, Mogotho, Seronga, Gunotsoga, Eretsha, and Beetsha, with additional cattle posts and settlements occurring between villages. The economy is quite diverse, including floodplain and dry land agriculture, livestock farming, wage labour, craft and tourism related enterprises (ODMP 2002; Mendelsohn

& Obeid 2004). Depending on annual rainfall the planting of crops occurs between November and January and harvesting occurs between April and June (Mosojane 2004).

The estimated elephant population in the eastern panhandle is between 9,000 – 15,000 elephants (see chapter two) and elephants range throughout the study area. Telemetry studies by (Jackson *et al.* 2008) in the Okavango Panhandle region indicated that the north-south buffalo fence blocks elephant movements from the Okavango River east to the Kwando River (see figure 9) and it is reported that the Namibian border fence (see figure 9), poses a significant barrier to elephant movements between Namibia and Botswana, (Chase & Griffin 2006). This information suggests that elephant movement is restricted out of the eastern panhandle. It is also evident in the study area that elephants use distinctive pathways to get to the Okavango River, which presents an opportunity to investigate the role of elephant movement as a factor in determining spatial patterns of HEC.

4.2.2. Independent primary HEC data collection

In order to establish a reliable independent conflict reporting system for this study, the standardized data collection protocol of the Human Elephant Conflict Working Group (HECWG) in the IUCN (Hoare 1999a) was followed to collect primary data on HEC. All fields raided by elephant between January 2008 and May 2010 were visited by both the local enumerator and the principal investigator (A. Songhurst) to ensure consistency and reliability of data collection. Details on each damage incident were recorded on standardised data collection forms. The average pace size of each enumerator was measured, and the area of all fields and damaged portions of the fields were estimated in square metres using enumerator paces (Hoare 1999a). Each damage incident was geo-referenced using a Garmin Etrex global positioning unit (GPS) and location details recorded.

4.2.3. Spatial distribution of fields

4.2.3.1. Spatial distribution

Raided and non-raided fields from the whole study period were plotted to examine the spatial distribution of fields.

4.2.3.2. Nearest neighbour distances and Moran's *I*

Spatial point patterns of individuals in a sampling frame can be described by three broad classes: random pattern (distribution of each individual is completely independent of the distribution of every other); regular pattern (individuals are more spaced out than in a random pattern); or aggregated pattern (individuals are more clumped than in a random pattern), (Crawley 2007). The distribution of nearest neighbour distances was calculated as a first step in determining whether there is any evidence to reject the null hypothesis of complete spatial randomness (CSR) of raided fields.

The observed number of raided fields was then distributed randomly across the mapped fields and the mean nearest neighbour distance from this random sample calculated. This was repeated 1,000 times and the frequency distribution of mean nearest neighbour distances plotted. This enabled me to see where in this distribution the observed mean distance was and determine whether fields were raided at random given the distribution of fields in the study area. If fields are not raided at random within the distribution of raided fields then it is evident that the actual locations (coordinates) of observational data matter in explaining the spatial arrangement.

Observations at locations nearer to each other can be either more similar (positive autocorrelation) or less similar (negative autocorrelation) (Fotheringham 2009). A key element in measuring and understanding this spatial autocorrelation is a description of the relationship between the degree of similarity between observations and the distance separating them (Fotheringham 2009). The Moran's *I* spatial autocorrelation statistic was calculated to test whether nearby observations tended to have similar attributes or to be more clustered than expected from randomness alone. The significance of this was tested with a *Z*- test.

4.2.3.3. Spatial-time plots

In ecological studies spatial autocorrelation may contain information that we don’t want to ‘correct for’ in analysis (Dormann 2007). Elephants may raid certain fields because of where they are in relation to landscape features i.e. distance from water etc. This is the spatial pattern we are interested in but this spatial pattern will generate a certain amount of spatial autocorrelation in the data, which we want to detect. The problem arises when elephants raid neighbouring fields on the same night, because I analyse one 'raid' that encompassed many fields as multiple individual raids. Yet, fields raided on the same night, in the same location cannot be considered independent. Because of the lack of independence, at least some of the information of sample *i* is contained in adjacent samples and so instead of having the information of *n* independent samples, we have information appropriate to fewer samples, called ‘effective sample size’ (Cressie 1993; Fortin & Dale 2009). It is therefore necessary to determine what this effective sample size is.

The spatial distance between raided fields was plotted against temporal distance between raided fields, to help identify the spatial scale at which to define a raid location.

4.2.4. Factors affecting the susceptibility of a field to raiding by elephants

A comparative survey of raided and non-raided fields was carried out to determine factors affecting the susceptibility of a field to being raided by elephant.

All fields reporting elephant raids were visited between Jan 2008 – May 2010. At the end of the crop season (June), when all the crops had been harvested and hence no further crop-raiding could occur, a selection of fields were identified by enumerators in each village that had not been raided by elephants. Each enumerator was asked to identify as many non-raided fields near their village as possible, preferably in different localities, giving a representative sample throughout the study area. Although this was not strictly random, it was the best method to use logistically and it is comparable to methods used in other studies both on elephant (Barnes 2005; Chiyo *et al.* 2005; Sitati, Walpole & Leader-Williams 2005) and other wildlife species (Tourenq *et al.* 2001; Ogada *et al.* 2003).

Data collected during the three field seasons were included in analyses. The elephant raid (successful or not) on a particular field was used as the independent unit of analysis in statistical analyses. Fields were coded as raided (1) or non-raided (0), as were mitigation methods (1= used, 0= not used), crop types (1= present, 0= not present), other animals raiding (1 = raiding, 0= not raiding). The data were screened for collinearity and outliers prior to analysis (using box plots and scatter plots). Influential data points were left out one at a time and the model was refitted to check if parameter estimates or standard errors were substantially affected.

Covariates recorded were: distance from field to i) nearest permanent (river) and semi permanent (waterhole) water, ii) next field, iii) main village, iv) main road, and v) main elephant path; mitigation techniques used; number of watch huts and guards; crop types grown; field characteristics (number of years field established, area of field, elevation, number of years previously raided); farmer details (age, ethnicity, gender, family size and livelihood); and predominant vegetation type around the field (see appendix IX for detailed list).

Initially, all data were used in the analyses. This, however, assumes no spatial autocorrelation is present in the data. One method of removing spatial dependency among observations can be done by removing samples until spatial independence has been attained, (Cliff & Ord 1981).The total dataset was, therefore, sub-sampled to select one field raided on a particular night in a particular location (grid cell) at different spatial scales.

Average daily movements of elephants in the Okavango panhandle appear to vary between wet and dry seasons. In the wet season (Oct - March), movements have been recorded to be approximately 3km/day, whereas in the dry season they are larger 6km/day. Crop-raiding incidents occur between January-May and, therefore, the crop-raiding season overlaps with the wet and dry season.

Grids of varying sizes (0.5km^2,1km^2, 1.5km^2, 2km^2, 2.5km^2, 5km^2), which would incorporate daily movement distances of elephants (3 – 6 km/day (Loarie, van Aarde & Pimm 2009a)) were superimposed over the study area, and one field per grid cell chosen for the sub sample (representing one field per location). This sub-sampling process was replicated 500 times at each spatial scale. The minimum adequate model derived from all data was then fitted to the sub-sampled data. Model estimates and p-values from the minimum adequate model using all data were compared with estimates and p-values derived from sub-sampled data at different spatial scales to determine the effect of spatial autocorrelation on model parameters.

4.2.5. Data analysis

All data analysis was carried out using R 2.11.1 (R Development Core Team 2010).

Bootstrapping and simulation techniques were used to investigate whether a field was raided at random within the study area. Moran's *I* was calculated to measure spatial autocorrelation in data and Z-tests were used to test significance.

Prior to developing crop field vulnerability models, all pairwise combinations of continuous predictor variables were assessed for collinearity (Zar 1984). Generalized linear models (GLMs), with binomial errors were used to explore a range of factors affecting the susceptibility of fields to elephant crop-raiding. Initially, univariate analyses were carried out with all explanatory variables. Multivariate GLMs with binomial errors were used to investigate correlates explaining the susceptibility of a field to crop raiding. The maximum model for two-way interactions was fitted and simplified by stepwise deletion of non-significant terms (Crawley 2007).

Model fit was checked using chi-squared goodness-of-fit test and significance determined for all analyses at $p < 0.05$. Using p values of < 0.05, however, could result in only a tiny percentage of deviance being explained. A second set of analysis was conducted to examine factors which explain >1% of the variance in the model, to establish biologically significant variables and interactions, which could be used as practical predictors of conflict in management actions.

4.3. Results

4.3.1 Spatial distribution of fields

4.3.1.1. Nearest neighbour distances

Figure 9 shows the spatial distribution of raided and non-raided fields in the eastern Okavango Delta panhandle, and illustrates a clustered pattern of field distribution in the study area.

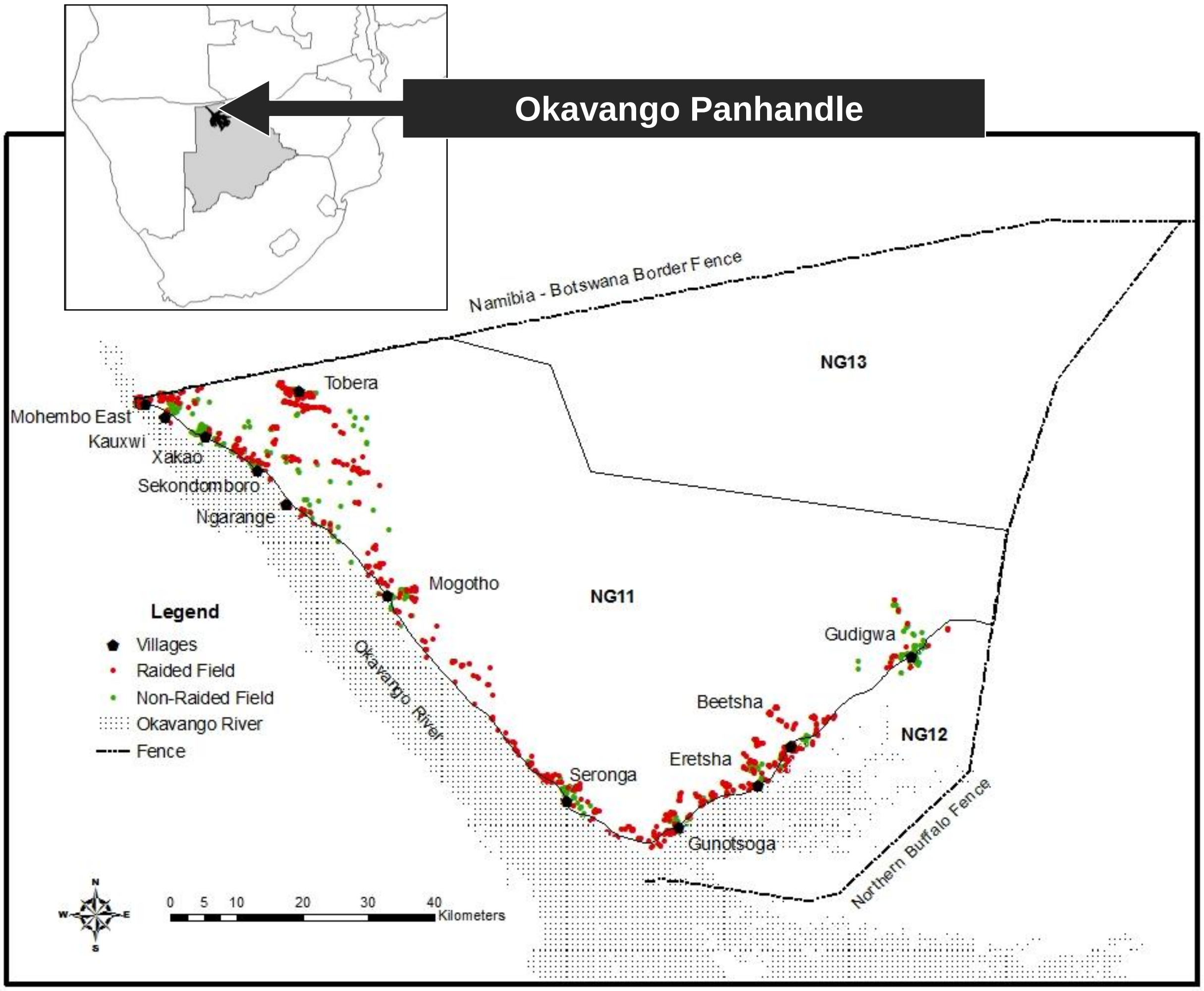


Figure 9. Map of the study area, with raided (red points) and non-raided (green points) fields in three years (2008-2010) illustrated. The two fences which may restrict elephant movement (northern buffalo fence and Namibia border fence) are highlighted with a broken line.

Raiding is spatially non-random. Mean nearest neighbour distances for all fields and just raided fields were respectively, μ = 171.15m and μ = 235.59m (see appendix X for histograms). The distribution of mean nearest neighbour distances of the 1,000 bootstrapped samples of simulated randomly distributed raided fields is illustrated in a histogram (Figure 10). The observed mean nearest neighbour distance of observed raided fields (μ = 235.59m) lies to the left of the distribution of randomly distributed fields outside of the 95% percentiles of the distribution (2.75% = 257.62 and 97.5% = 299.67), suggesting that raiding is spatially non-random and therefore location of fields influence raiding patterns.

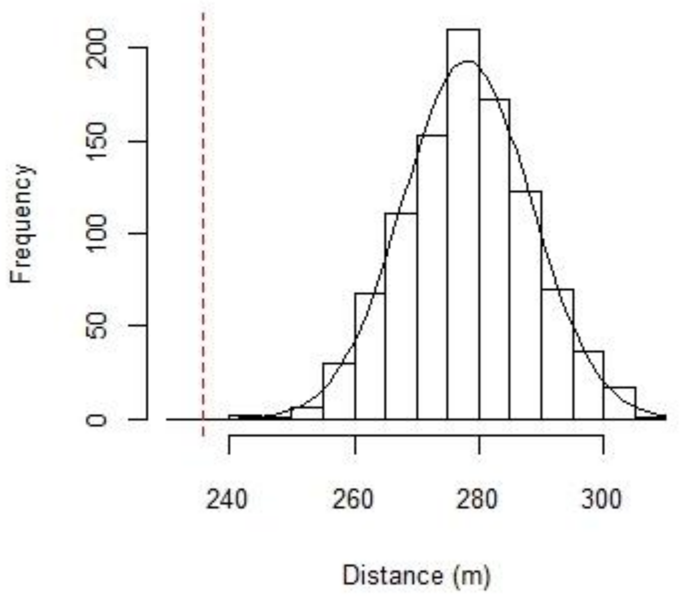


Figure 10. Histogram of 1,000 bootstrapped mean nearest neighbour distances of randomly distributed raided fields, mean nearest neighbour distance of actual raided fields indicated with red dotted line

4.3.1.2. Spatial-time plots

There was a correlation (r_p= 0.067, p = 2.2e^{-16}) between the spatial distance between raided fields and the temporal distance between raided fields. This spatio-temporal correlation generates spatial autocorrelation in the data. The average length of time between raids for fields <0.5km apart was 16 days while for raided fields >2km apart it was 32 days.

The spatial distribution of fields within the study area is clustered and it is, therefore, difficult to investigate the influence of spatial autocorrelation using geostatistical measures. The Moran's *I* statistics showed significant positive spatial autocorrelation in both all fields and raided fields data (I = 0.328, P = 2.2e^{-16} and I = 0.240, P = 2.2e^{-16}, respectively).

However, for raided field data sub-sampled at the 0.5 km^2, 1 km^2, 1.5 km^2, 2km^2 spatial scales Moran's *I* decreased and then levelled out (I = 0.050, P = 0.030; I = 0.042, P = 0.131; I = 0.040, P = 0.283, I = 0.049, P = 0.126; I = 0.050, P = 0.125; I = -0.030, P = 0.617, respectively), illustrating that spatial autocorrelation is reduced from data when sub sampled at larger spatial scales.

4.3.2 Characteristics of elephant crop-raiding

A total of 1421 fields were assessed over three years (Mar 2008- Jul 2010); 788 raided and 633 randomly selected non-raided fields. A total of 162.12 ha of crop were recorded as damaged throughout the three years with a mean amount of damage per incident of 0.289ha (±0.72ha). The median proportion of damage per field was 2.02%, with a quarter of raided farms suffering less than 0.4% and a quarter suffering more than 10.1% damage. Millet was the principal crop destroyed by elephants (82.7% of cases), although all crop varieties were affected. For further details of the characteristics of elephant crop raiding in the Okavango Panhandle see chapter three.

A variety of mitigation techniques were used by farmers in the study area, including bush fence, wire fence, watch huts, drumming, fires, shouting, dogs, tin can fences, plastic on fences, trenches, torches, whips, chilli deterrents and guns. The most commonly used mitigation techniques were drumming (65%), bush fencing (63%), watch huts (58%), and wire fencing (33%). Only a few fields were protected with guns (9%) or chilli (7%).

4.3.3. Key drivers of elephant crop-raiding

The univariate analyses showed a range of factors (see appendix IX for variables tested) with significant effects ($p<0.05$) on whether a field was raided or not by elephants, although some effect sizes were small (see appendix XI for detailed results). These included farmer characteristics, field characteristics, presence of other raiding animals, crop types present, mitigation methods used and surrounding vegetation types.

In summary, results using all data indicate that fields at a higher risk of being raided by elephants are those which are isolated from other fields, further from human settlements, surrounded by certain vegetation types (mixed woodland; Terminalia; or Teak), have pumpkins growing and where porcupines also raid. Fields more likely to be raided by elephant also had watch huts present.

On the other hand, fields less likely to be raided by elephants appear to be relatively newly established, larger, further from elephant paths and waterholes, surrounded by False Mopane vegetation, have groundnuts or other crops (i.e. vegetables) planted, are owned by older farmers and protected by wire fences, shouting farmers, plastic on fences or dogs. Fields were less likely to be raided by elephants if they were raided by livestock, which could be explained by more vigilant guarding effort being implemented in fields raided by livestock.

Only variables that had statistically significant test results in the univariate analysis ($p<0.05$) were entered into the multivariate analysis. The response variable was binomial (raided (1) or not-raided (0)) and explanatory variables were both continuous and categorical, therefore GLM with binomial errors were used to test variables in combination. Two influential data points (398 and 409) were removed after the minimum adequate model was checked using residual plots.

The minimal adequate model containing variables explaining <1% of the overall variance (see table 6), retained variables distance of a field to a main village, number of years a field has been raided in the past, and the presence of pumpkins as having significant positive effects (fields are more likely to be raided if they are far from the village, have been raided lots in the past and have pumpkins growing). It retained variables distance of a field to a waterhole, distance of a field to a main elephant pathway, age of field, and the presence of livestock raiders as having significant negative effects (fields less likely to be raided are older fields, far from waterholes, far from elephant paths, and fields where livestock also raid).

The model shows that the raiding of a field depends on a 2-way interaction between the age of a field and the presence of certain crop types. The number of years a field had been established and the presence of pumpkins interaction was retained (χ^2= -7.47, p = 0.006279) as having a negative effect (older fields growing pumpkin were less likely to be raided by elephant).

Table 6. The minimum adequate model of the GLM for whether a field is raided or not, with binomial error structure and significance at p<0.01.

Explanatory Variable	Estimate	SE	Z Value	P value	χ^2 Deviance	χ^2 P Value
Intercept	0.532	0.391	1.362	0.173		
Distance to Village	0.019	0.003	6.017	$1.77e^{-9}$	-38.98	$4.288e^{-10}$
Distance to waterhole	-0.020	0.007	-2.985	0.003	-8.96	0.003
Distance to Elephant path	-0.039	0.006	-6.997	$2.61e^{-12}$	-51.87	$5.922e^{-13}$
No. years raided in past	1.212	0.085	14.235	$<2e^{-16}$	-315.06	$2.2e^{-16}$
Age of field	-0.490	0.095	-5.160	$2.47e^{-7}$	119.62	$2.2\ e^{-16}$
Pumpkin grown	1.462	0.370	3.951	$7.78e^{-5}$	12.71	0.0004
Livestock raiding	-0.892	0.153	-5.810	$6.23e^{-9}$	-34.38	$4.54e^{-9}$
Age of field : Pumpkin grown	-0.338	0.124	-2.731	0.006	-7.47	0.006

Further chi square analysis showed that biologically significant variables and interactions (explaining >1% of overall variance), which could be used as practical predictors of conflict in management actions were those which had a chi square test residual deviance of >11.2. Variables retained in this analysis included, distance to elephant pathway (χ^2= -51.87, p = $5.92e^{-13}$) explaining 4.6% of variance, distance to village (χ^2= -38.98, p = $4.29e^{-10}$) explaining 3.5% of variance, and presence of pumpkin (χ^2= -15.77, p = $7.15e^{-5}$) explaining 1.1% of variance.

4.3.4. Exploring the effect of spatial autocorrelation

When sub-sampling data at different spatial scales (0.5km^2, 1km^2, 1.5km^2, 2km^2, 2.5km^2, 5km^2), to reduce the spatial autocorrelation in the data, coarser grid sizes ≥2.5km^2 had too few data points and could not be used in subsequent analysis. Results from four grid sizes (0.5km^2, 1km^2, 1.5km^2, 2km^2) only, are shown in table 7.

As expected, as the spatial scale for sub-sampling increases, the sample size of variables decrease, and the standard errors around mean estimates increase for all variables. It is also evident that spatial autocorrelation affects the size of the estimates of different variables, with estimate values either increasing or decreasing as spatial autocorrelation is reduced through sub sampling at different grid sizes. Spatial autocorrelation consequently not only influences standard errors and p-value, but it also influences our biological interpretation. However, patterns of change in the size of estimates across spatial scales were not consistent across variables. Distance of a field to a main elephant pathway or waterhole showed little change in model estimates as spatial autocorrelation was reduced by random sub sampling (see figure 11a and 11c, respectively), indicating that spatial autocorrelation did not have a strong influence in these parameter estimates. Other variables showed an increase e.g. distance of a field to a village, (see figure 11d and 11h) or decrease e.g. number of years raided in the past, (see figure 11b, 11f and 11g) in model estimate as grid size increased, while the pumpkin presence variable showed a fluctuating relationship between mean estimate and grid size, (see figure 11e).

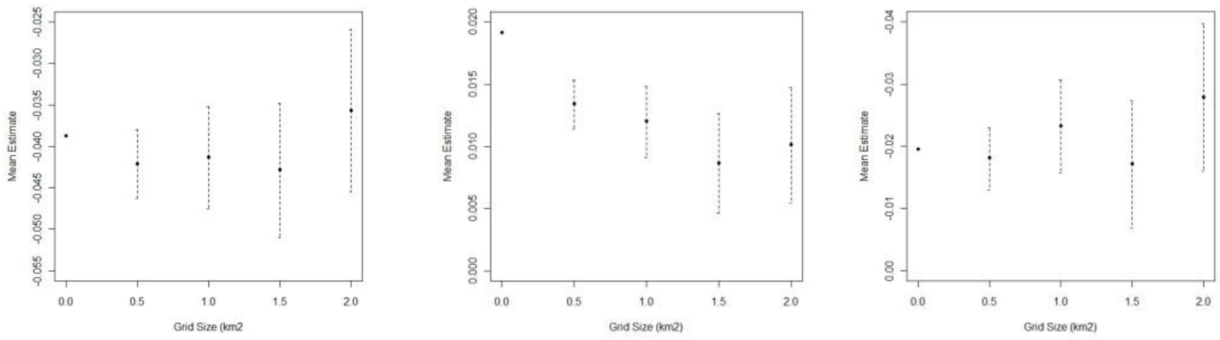
The intercept increased with increasing spatial scale, indicating that the average amount of raiding appears to increase as spatial scale of sub sampling increases, (see figure 11i).

For all variables the p value decreased as sample size increased (spatial scale decreased), as expected (see figure 12). It is evident from this graph that even at a smaller sample size (larger spatial scale) the significance (p-value) of variables such as distance to elephant pathway, number of years previously raided, age of field and presence of livestock do not vary to a large extent and are still statistically significant at $p<0.05$. There does not appear to be a strong association between patterns for the estimates and patterns for the p values. The plots of mean intercept vs grid size show that the intercept increases gradually with increasing grid size.

Table 7. Estimates and p-values for GLM using sub-sampled data at different spatial scales (significant results highlighted in red)

	All Data		**$0.5km^2$ Sample**		**$1km^2$ Sample**		**$1.5km^2$ Sample**		**$2km^2$ Sample**	
Variable	Estimate	p value	Estimate*	p value*	Estimate*	p value*	Estimate*	p value*	Estimate*	p value*
Intercept	0.532	0.173	1.253	0.064	1.876	0.041	1.635	0.157	2.149	0.117
Dist. Village	0.019	$1.77e^{-9}$	0.013	0.007	0.012	0.054	0.009	0.252	0.010	0.232
Dist. Waterhole	-0.020	0.003	-0.018	0.099	-0.023	0.102	-0.017	0.315	-0.028	0.160
Dist. Ele. Path	-0.039	$2.61e^{-12}$	-0.042	$2.95e^{-6}$	-0.041	$5.04e^{-4}$	-0.043	$2.57e^{-3}$	-0.036	$3.48e^{-2}$
Field Age	0.490	$2.47e^{-7}$	0.637	0.0004	0.815	0.001	0.780	0.022	0.975	0.023
No. Years Raided	-1.212	$<2e^{-16}$	-1.285	$3.81e^{-18}$	-1.327	$5.55e^{-12}$	-1.483	$2.69e^{-8}$	-1.528	$3.14e^{-7}$
Pumpkin	1.462	$7.78e^{-5}$	1.143	0.091	0.683	0.405	1.389	0.232	1.120	0.371
Livestock	-0.892	$6.23e^{-9}$	-0.671	0.012	-0.825	0.025	-0.916	0.046	-1.218	0.027
Field Age : Pumpkin	-0.338	0.006	-0.256	0.284	-0.018	0.541	-0.258	0.414	-0.063	0.497

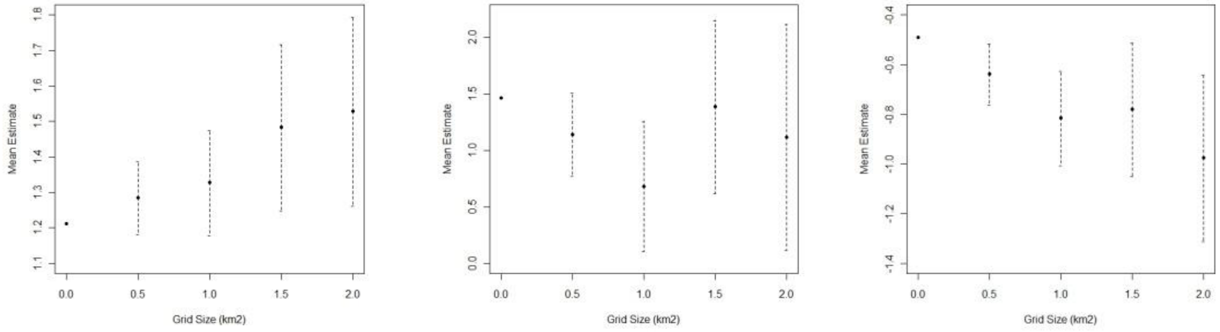
*Mean



a) Distance to elephant pathway

b) Distance to village

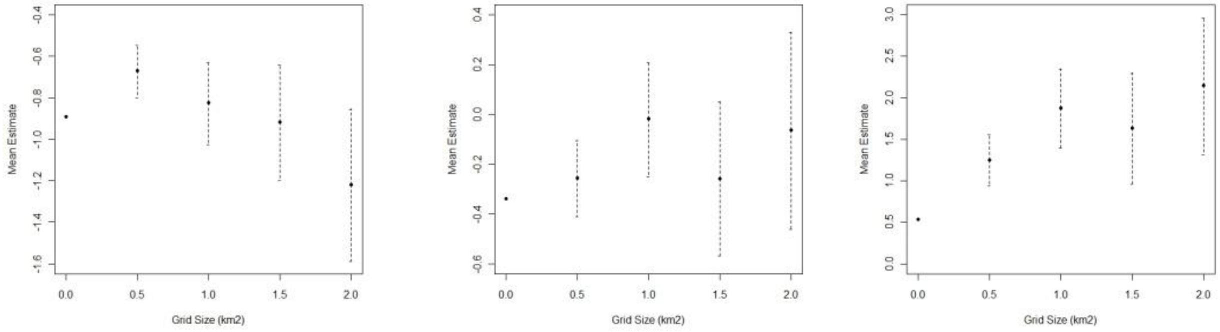
c) Distance to waterhole



d) Number years raided in past

e) Pumpkins present

f) Age of field

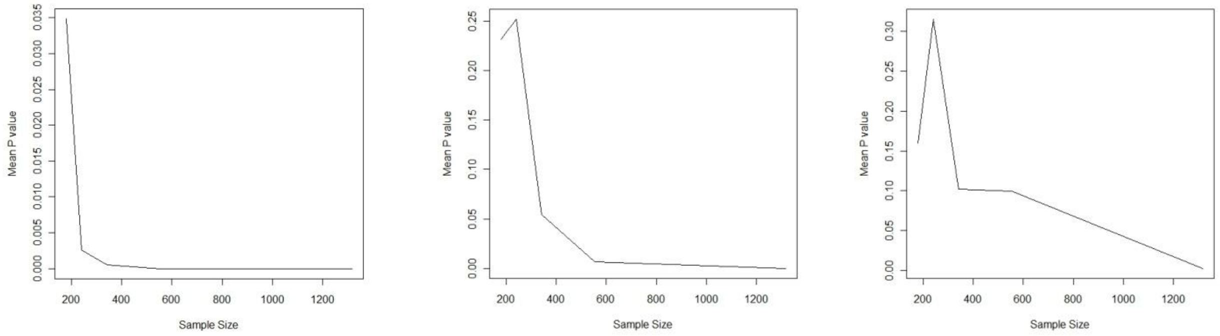


g) Livestock

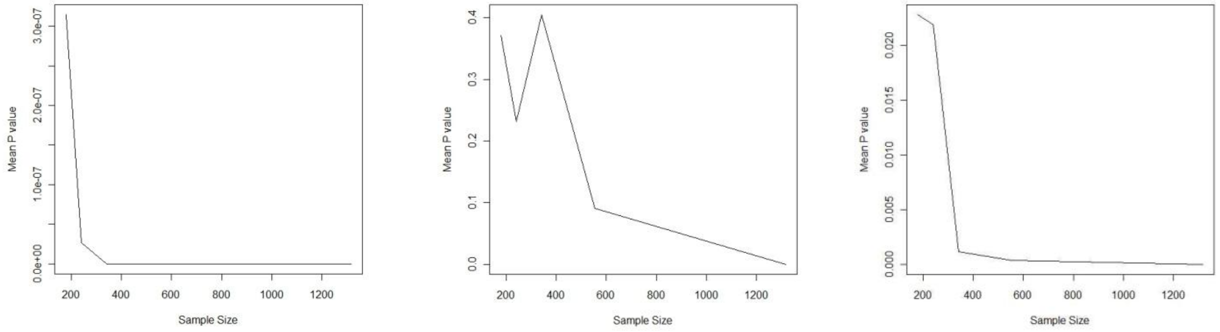
h) Age of field : Pumpkin

i) Intercept

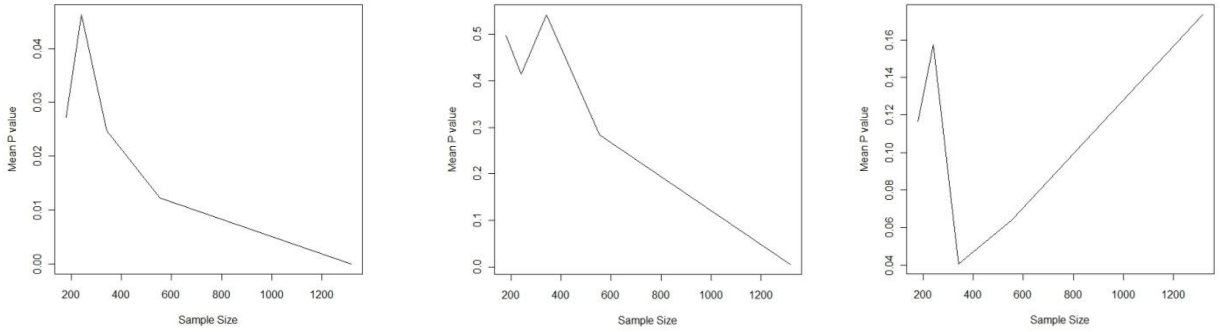
Figure 11. Relationships between mean parameter estimate from GLM and grid size for all variables and intercept



a) Distance to elephant pathway b) Distance to village c) Distance to waterhole



d) Number years raided in past e) Pumpkins present f) Age of field



g) Livestock h) Age of field : Pumpkin i) Intercept

Figure 12. Relationships between mean p value and sample size for all covariates explaining the susceptibility of a field to elephant raiding.

These differences in model estimates (and p values) of variables are driven by the spatial distribution of fields in an area. For example, if fields are close to one village (V1 in figure 13) yet spread out around another village (V2 in figure 13), then when I sub sample one raided/non-raided field per grid cell, the sample size will decrease dramatically for V1 (from n = 5 to n = 1 non-raided and n = 2 to n = 1 raided) but remain relatively stable for V2 (from n = 7 to n = 6 non-raided and n = 3 for raided). The model estimates for the variable “distance of a field to a village” will therefore be dramatically affected by such sub sampling. When all data are used in analyses there will be a lot more variance in the data, but when I throw data points away through sub sampling the intercept and slopes of model variables change. Biologically when I subsample data so they conform to statistical assumptions we potentially lose valuable information. I consequently advocate reporting both regression estimates and p-values across a range of spatial scales as this can help biological interpretation.

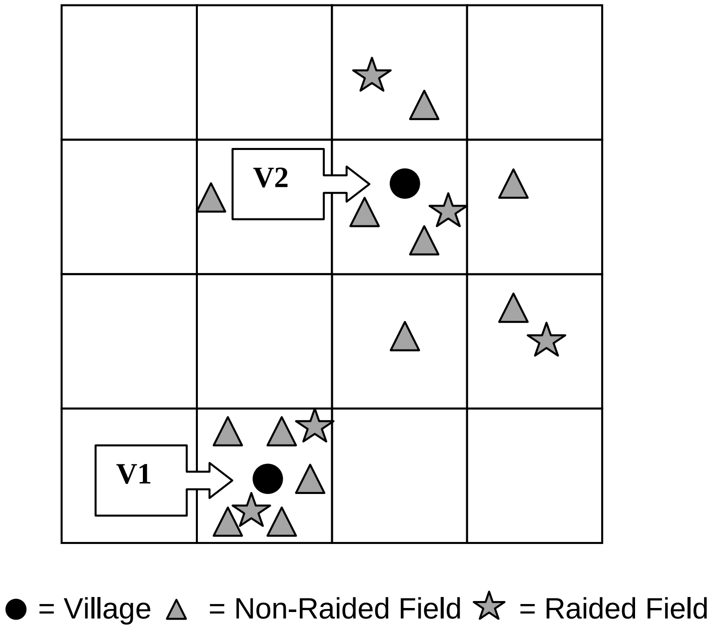


Figure 13. Schematic diagram explaining change in model estimates and p-values

If I use the “distance to village” variable and investigate what happens to model estimates and p-values in a village where fields are dispersed around the village (e.g. Seronga) and compare this to a village where fields are more clumped (e.g. Tobera) it is evident that p-values do indeed decrease more gradually for an area where fields are more dispersed (Seronga) compared with an area where they are clumped (Tobera), (see figure 14) and the model estimates change at a lower grid size for Tobera than Seronga, (see figure 15).

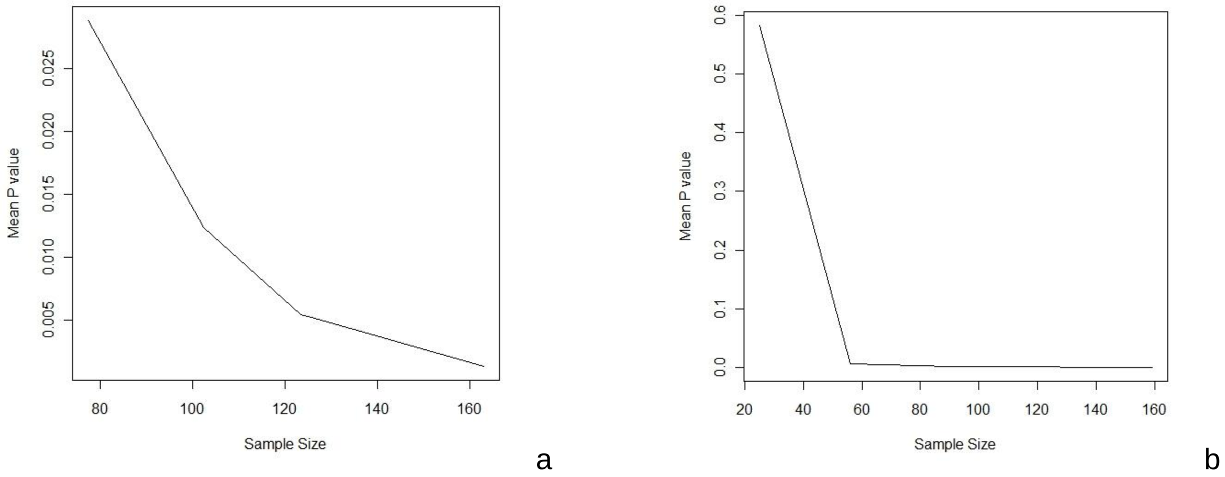


Figure 14. Relationships between mean p value and sample size for distance to village covariate explaining the susceptibility of a field to elephant raiding in two villages a) Seronga and b) Tobera.

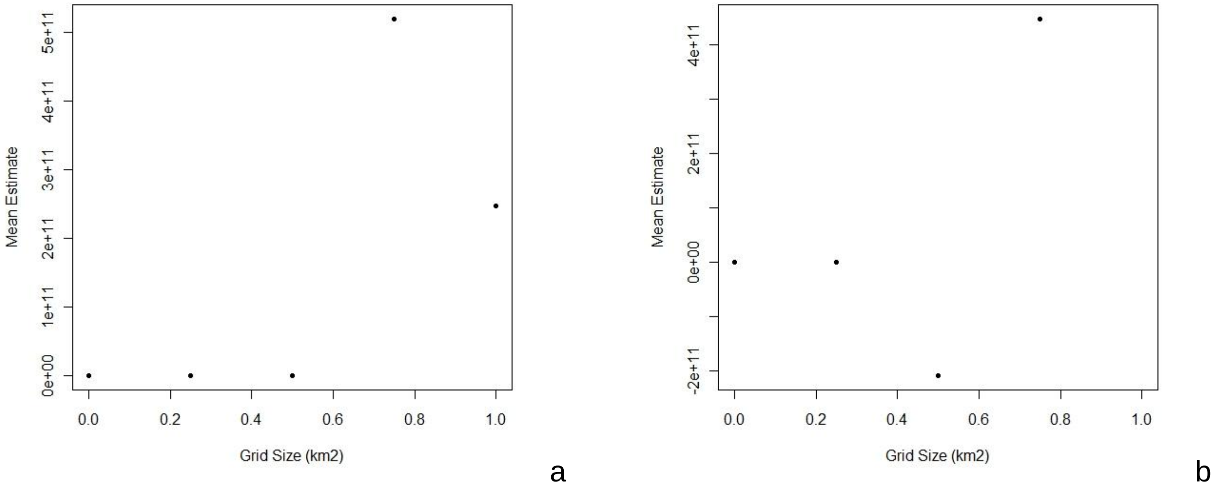


Figure 15. Relationships between mean parameter estimate from GLM and grid size for distance to village variable for two villages a) Seronga and b) Tobera

4.4. Discussion

4.4.1. Spatial patterns and spatial autocorrelation in wildlife crop-raiding data

Spatial patterns in data can act as indicators of the processes that have occurred over a given region, however, several processes can occur simultaneously each operating at a given spatial and temporal scale (Green & Young 1993; Fortin & Dale 2009). The factors and processes that affect data can be coarsely divided into two kinds: those that induce spatial dependence (most spatially distributed environmental factors) and those that generate spatial autocorrelation (Fortin & Dale 2009). When observations drawn from different locations are not independent from one another they can be described as spatially dependent and spatial autocorrelation arises (Cliff & Ord 1969). Spatial patterns of wildlife crop raiding are therefore complex, determined by multiple factors acting at more than one scale (Hegel, Gates & Eslinger 2007), including field locations and the ecology of crop-raiding wildlife species and their heterogeneous use of habitats.

The spatial distribution of agricultural fields often exhibit a clustered pattern, due to field locations being selected based on landscape features such as soil quality (Martin & Taylor 1983; Happold 1995), which induces a degree of spatial dependence between field characteristics (fields closer together may be more similar). When animals raid neighbouring fields on the same night, observations may not therefore be considered independent (Sitati *et al.* 2003), and spatial autocorrelation arises. In this scenario an animal's behaviour towards one field may be influenced by the mitigation used or other factors of a neighbouring field, likewise farmer behaviour and therefore mitigation efforts may be altered in neighbouring fields once they are alerted to the presence of a wildlife crop raider (Sitati, Walpole & Leader-Williams 2005). In the Okavango Delta panhandle, elephant crop-raiding was spatially non-random indicating that the location of fields influenced the pattern of raiding. Localized and persistent crop-raiding occurred, where fields raided on days closer together were spatially closer together as well, a pattern that has been witnessed in other studies (Bell 1983; Damiba & Ables 1993; Naughton-Treves 1998; Sitati *et al.* 2003). As was the case in comparable studies investigating spatial patterns of elephant crop-raiding (Hoare 1999a; Smith & Kasiki 1999; Sitati *et al.* 2003; Graham *et al.* 2010), spatial autocorrelation was present in the raided field data in my study.

4.4.2. What is a "location" of an elephant raid?

One method to remove spatial autocorrelation and avoid pseudoreplication in data is to take a subsample of data (Hurlbert 1990; Hoare 1999b). In the case of crop-raiding data, a suitable subsample would be to select one field raided on a particular night in a particular location (Sitati, Walpole & Leader-Williams 2005). However, it is difficult to determine what a "location" of an independent elephant raid is. From an elephant perspective the location of a raid on a particular night could incorporate fields within the nights foraging expedition or movement route to available water. Elephants can move large distances in search of resources such as food, water and minerals or in response to disturbance even in one night (Blanc *et al.* 2007; Loarie, van Aarde & Pimm 2009a). Loarie, van Aarde & Pimm (2009a) investigated daily movements of elephant in the Okavango panhandle and found that they can move up to 40km/day roundtrip, traversing from areas near rain filled pans to the permanent Okavango river, with average daily movements in the dry season appearing larger than in the wet seasons (6km/day and 3km/day respectively). A location of one elephant foray on a particular night could, therefore, be considered to include all raided fields within a 40km^2 area, or more probable within a 3-6km^2 area. Yet, it is also likely that more than one elephant herd are responsible for crop raiding incidents on any particular night and therefore even fields within 500m from each other could actually be considered independent raids as they were conducted by different groups of elephants. It must also be noted that raiding incident data often relies heavily on information from farmers (who most likely are the only observers of the raid). Therefore, identity of elephants (due to it usually being dark and farmers possibly not having the skill to identify individual elephants) and even exact time of a raid (due to most farmers not having clocks) cannot always be recorded. This makes it difficult to verify whether raids have been conducted by the same or different elephants on the same night in neighbouring fields.

For management purposes a "location" or sampling unit could be considered to be an individual field as used by Graham *et al.* (2010). My study, therefore, included data from all elephant raiding incidents in initial analyses to derive correlates of crop-raiding at the field level. Using all data, however, assumes no spatial autocorrelation is present in the data and therefore it is advised that sub sampling at different spatial scales be used to reduce the effects of spatial autocorrelation without having to define an arbitrary "location" of a raid.

4.4.3. Effect of spatial autocorrelation in wildlife crop-raiding data

The presence of spatial autocorrelation among data can potentially lead to excessive reductions in standard errors of the parameter estimates, exaggerate degrees of freedom and thereby inflate the probability of type I errors, (Dormann 2007; Fortin & Dale 2009). My results found that not only the significance levels (p-values) in regression analysis were affected by spatial autocorrelation, but also the regression estimates changed when data were sub sampled at coarser spatial scales (when spatial autocorrelation was reduced). Legendre *et al.*, (2002) found that if spatial autocorrelation is present in both the response and explanatory variables in regression analysis then the significance of correlation and regression coefficients can be disturbed. I know that spatial autocorrelation was present in the response variable, but results from Legendre *et al.*, (2002) would suggest that spatial autocorrelation was present in the explanatory variables as well. When choosing one raided/non-raided field per cell, the degrees of freedom are reduced and therefore the significance levels increase, but I also found that the regression estimates changed. The regression estimates change if one excludes most of your data with for example a 1 (raided) rather than a 0 (not raided) by sampling only one point from one cell. There is a lot more variance in the response variable (raided/not raided) when one uses all the data, but when data points are discarded during sub sampling the variance decreases, meaning that the intercept changes and also the slope (regression estimate). Most methods used to account for spatial autocorrelation in data attempt to correct for the change in significance levels, yet my study shows that by sub-sampling data at broader spatial scales we can identify changes in regression estimates as well.

The average intercept appeared to increase at broader spatial scales of sub-sampling. The intercept effectively gives an estimate of the amount of raiding occurring, therefore, this result infers that there are larger amounts of raiding occurring at broader spatial scales. When sub-sampling at broader spatial scales it is important to note that the amount of raiding occurring may be inflated. It is also evident that patterns in the changes in significance levels and regression estimates vary for different explanatory variables depending on the spatial distribution of fields and explanatory variables. My results suggest that there is spatial variation in the pattern of spatial autocorrelation in crop-raiding data. For areas where the fields are spatially dispersed there is a less dramatic effect on both p-values and regression estimates when spatial autocorrelation is reduced, than for areas where fields are clumped. Extra caution is therefore needed when sub-sampling data from fields with a clumped distribution. Such findings indicate that models should be developed for areas with fields displaying similar (i.e. all clumped) spatial distributions rather than varied (i.e. dispersed and clumped), for example at a village level rather than on a broader (ward/region/countrywide) scale.

Previous studies investigating spatial correlates of crop-raiding have used districts or known elephant ranges to define the spatial extent to be used in the analysis (Hoare 1999b; Smith & Kasiki 1999; Sitati *et al.* 2003; Graham *et al.* 2010) and it has been found that it is easier to identify predictors of HEC at broader spatial extents (Sitati *et al.* 2003; Graham *et al.* 2010). Graham *et al.* (2010) suggested that if resources are limited, then the use of such parameters to define spatial extent in the analysis of HEC data is adequate for identifying broad priorities for management intervention. My results illustrate, however, that caution should be exercised when defining spatial extents for analysis based on the above criteria, because the effect of spatial autocorrelation in the data varies depending on the spatial distribution of fields, and therefore field distribution in an area should also be considered when choosing the spatial extent for analysis.

Understanding spatial patterns of crop damage is paramount for designing better mitigation and land use planning strategies. If the occurrence of elephant crop raiding can be predicted from spatial and environmental variables, elephant behaviour, field characteristics, farming practices and/or mitigation techniques, then future management of conflict situations could be more effective in reducing the probability of conflict incidents occurring in the future. Although there have been many studies to try and identify predictive factors of elephant crop raiding (Hoare 1999b; Smith & Kasiki 1999; Sitati *et al.* 2003; Osborn 2004; Barnes 2005; Chiyo *et al.* 2005; Sam *et al.* 2005; Sitati, Walpole & Leader-Williams 2005; Graham *et al.* 2010), few universal trends in spatial patterns of crop-raiding incidents have been found, but localised patterns have been identified for specific areas.

In the Okavango Delta panhandle, explanatory variables that remained significant across all spatial scales and therefore are considered to be robust results were distance of a field to a main elephant pathway and the number of years a field has been raided in the past. Fields further from elephant pathways were less likely to be raided by elephants and fields which had been raided frequently in the past were more likely to be raided again. Elephants have long memories (McComb *et al.* 2001) and often utilize traditional movement routes (Low 2000) to get to watering or foraging sites. One explanation for both these results could, therefore, be that elephants are returning to fields they remember having raided successfully in the past (Sitati *et al.* 2003), which were either close to these traditional movement paths, or movement paths have been altered to travel close to these foraging sites. At finer spatial scales the age of a field that had been raided in the past also affected the likelihood of a field being raided by elephant during the study period, which could also indicate that elephants are remembering older fields raided in previous years. Similar patterns, where fields raided in the past were more likely to be raided, have also been identified in Kenya (Sitati, Walpole & Leader-Williams 2005).

Naturally, there is an increased likelihood of elephants using elephant pathways to encounter fields closer to these pathways than fields further away. Therefore, it could also be opportunistic foraging behaviour which explains why fields closer to elephant pathways are more likely to be raided. In the Okavango Panhandle, elephants generally move to the river during the night (Loarie, van Aarde & Pimm 2009a) and hence move into areas with cultivated land at this time. Data collected through another part of this study showed that elephants usually arrive at the river between 9pm-12am and return to foraging sites between 12-3am, as was confirmed by Loarie, van Aarde & Pimm, (2009a) with collared elephant movement data. Most raiding incidents occur after 12am according to farmers in the area indicating that elephants generally crop raid when they return from drinking and bathing at the river. Elephants may then be using the movement pathway to get back to their foraging sites and forage a short distance away from the pathway during this time, opportunistically or accidentally encountering fields close to these pathways. Interestingly, farmers believe elephants are attracted to fields (see chapter six) and have adjusted their pathways to come close to fields. If this hypothesis is true then it could be that elephants are selecting for crops under the theory of optimal foraging, based either on the crops nutritional value as surmised by Sukumar, (1990) or due to the condition of the natural forage available elsewhere as suggested by Osborn, (1998b). More detailed temporal and spatial patterns of elephant foraging strategies are beyond the scope of this study and require further investigation in the panhandle.

4.4.4. Conclusion

This study illustrates the need to exercise caution and consider the effect that spatial autocorrelation may have on regression estimates and significance levels for wildlife crop raiding data when using generalized linear models (GLMs) to identify key drivers of crop raiding and devise a crop vulnerability model. It is advocated that all data be used to find the minimum adequate model containing significant correlates of raiding and encourage the use of sub-sampling data at different spatial scales to reduce the effect of spatial autocorrelation, thereby avoiding the need to define an arbitrary location of a raid. However, because spatial autocorrelation affects both significance levels and model estimates it is advised that both values be reported across a range of spatial scales for all statistically significant explanatory variables to aid in the biological interpretation of the data.

The spatial distribution of fields should be considered when choosing the spatial extent of analyses for wildlife crop-raiding studies. Different explanatory variables are affected in different ways through sub-sampling to reduce spatial autocorrelation in the data, therefore it may be advisable to analyse data from areas where fields either have a clumped distribution or spread out distribution, rather than both together. It is evident from other studies conducted in Africa and Asia both on forest and savannah elephant populations that it is difficult to identify common predictors of crop raiding by elephant. Local variations in elephant ecology and movements, as well as human spatial use have been identified previously as possible causes of this apparent unpredictability; however, I found that field distribution may also be a contributory factor.

The robust key drivers of crop-raiding identified in this study have important management implications. Recommendations can be made to wildlife management and land board authorities as well as subsistence farmers in the Okavango Delta panhandle, that fields situated close to elephant pathways or fields with a long history of elephant crop raiding are at higher risk of future crop damage. Mitigation efforts can therefore be targeted at fields with a higher risk to try and reduce elephant crop-raiding in the future.

Chapter 5

The effect of human habitat modifications on elephant spatial use

5.1. Introduction

Where humans and wildlife live in close proximity competition can arise over limited resources and negative human-wildlife interactions may occur. One of the fundamental underlying factors contributing to human-wildlife conflicts (HWC) is competition for space. Large wide ranging mammals, such as elephants (*Loxodonta sp.* and *Elephas maximus*), require a large amount of space (Laws & Johnstone 1970; Owen-Smith 1988; Sukumar 1990; Ruggiero 1992; Chase & Griffin 2007; Choudhury *et al.* 2008) and therefore often utilise habitats outside of protected areas. Simultaneously, growing human populations increasingly require land for residential and agricultural uses as well as other developments. When natural ecosystems are converted to agricultural land or human settlements, wildlife habitats can become reduced and fragmented, which in turn leads to more frequent interactions between people and wildlife (Linkie *et al.* 2003; Agetsuma 2007). Land use change is one of the most common factors contributing to increased conflicts between humans and wildlife (Hoare 2000b; Conover 2002; Hegel, Gates & Eslinger 2007). Careful land use planning is therefore needed to minimise HWC, which ultimately requires an understanding of how animal spatial use is influenced by people in such heterogeneous landscapes.

In ecological terms, animal habitat selection is determined by availability and accessibility of foraging locations within a home range (foraging scale) as well as by dispersal and the ability to relocate a home range (dispersal scale) (Morris 1992). For example, the spatial distribution of elephants appears to be affected, at a foraging scale, by a combination of environmental and social factors. Such environmental factors include: Water availability (Verlinden & Gavor 1998; Stokke & duToit 2002; Redfern *et al.* 2003; Grainger, van Aarde & Whyte 2005; Chamaille-Jammes, Valeix & Fritz 2007); palatability, availability and nutritional quality of food (Sommerlatte 1976; Jachman 1984; Sukumar 1990; Viljoen & Bothma 1990; Shannon *et al.* 2006); sodium availability (Hanks 1979); vegetation heterogeneity (Murwira & Skidmore 2005; Chamaille-Jammes, Valeix & Fritz 2007; Loarie, van Aarde & Pimm 2009b); land use types, with elephants avoiding cultivated land and unprotected areas, while favouring rivers, bushed grassland, forest and woodland (e.g. Foley 2002; Hien 2005; Wittemyer *et al.* 2007; Mpanduji, East & Hofer 2008); climate, home ranges are larger in drier environments than wet (Lindeque & Lindeque 1991; Chase & Griffin 2007; Young, Ferreira & van Aarde 2009) ranging from as small as 15km^2 in sub-tropical Tanzania (Douglas-Hamilton 1973) to as large as 24,828km^2 in arid Botswana (Chase & Griffin 2007). Social factors include dominance hierarchy among family groups (particularly in the dry season), with dominant groups disproportionately using preferred habitats (Wittemyer *et al.* 2007), sexual segregation between non-mating male only herds and female family groups and intra-sexual avoidance between dominant and sub-ordinate males (Moss & Poole 1983; Stokke 1999; Galanti *et al.* 2005). How elephants utilise different landscapes is also determined by temporal factors such as seasons (Cushman, Chase & Griffin 2005; Douglas-Hamilton, Krink & Vollrath 2005; Leggett 2006c; Chase & Griffin 2007; Wittemyer *et al.* 2007; Chase & Griffin 2008; Young, Ferreira & vanAarde 2009; Loarie, vanAarde & Pimm 2009a), the time of day they move, the speed of their movements and the length of time they spend in any one location (Graham *et al.* 2009).

However, in areas where home ranges overlap with human habitation and infrastructure, animals also have to adapt to cope with changes in the availability and accessibility of resources such as foraging sites, water, resting and breeding sites, as well as avoiding human disturbance (Martin *et al.* 2010). Disturbances to animals may arise due to physical developments impeding movement (i.e. fences, roads, cultivated land), which can affect both foraging and dispersal scale aspects of habitat selection, or as a result of human behaviour such as hunting, farming and wildlife conflict mitigation activities, which are more likely to affect the foraging scale of habitat selection.

Where human–elephant spatial occupation overlaps, changes in elephant distribution and movement patterns are influenced by risk avoidance strategies. For example, elephants tend to move under the cover of darkness (Douglas-Hamilton, Krink & Vollrath 2005; Graham *et al.* 2009) and travel at faster speeds (Douglas-Hamilton, Krink & Vollrath 2005; Galanti *et al.* 2005) in areas close to human habitation. Elephants have been found to avoid unprotected areas (Wittemyer *et al.* 2007), with high human population density (Parker & Graham 1989b; Barnes *et al.* 1991; Eltringham 1991; Happold 1995; Hoare & du Toit 1999; Jackson *et al.* 2008; Graham *et al.* 2009), and high human activity (Graham *et al.* 2009). Male elephants are often more likely to take risks than females (Sukumar 1991) and are, therefore, more likely to utilise areas closer to human habitation (Hoare 1999b), which could account for larger ranging movement patterns among males (Stokke & duToit 2002; Shannon *et al.* 2006; Chase & Griffin 2007). Human presence and infrastructure, therefore, affects elephant behaviour and spatial use, which adds to the complexity of inter-relationships between environmental features and elephant needs and should, therefore, be considered when designing future conservation strategies, management interventions and land use plans (Douglas-Hamilton, Krink & Vollrath 2005).

A large portion (~69%) of known African elephant range exists outside of protected areas (Blanc *et al.* 2007), with many elephant populations now living in close proximity to humans. It is, therefore, imperative, albeit challenging, to understand how habitat modifications by humans (i.e. settlements, cultivated land, fences) affect elephant distribution and movement, so that such movements can be accommodated in conservation plans (Graham *et al.* 2009). This will facilitate appropriate landscape-scale habitat and land use management alongside other mitigation strategies (Davies *et al.* 2011), to aid in reducing negative interactions between humans and elephants.

Previous studies investigating the effect of human disturbance on elephant ecology have used various techniques to collect data, including global positioning system (GPS) tracking data, aerial surveys or ground-transect surveys. For example, GPS collar tracking methods provide detailed data that has been used to determine the effect of human disturbance on elephant activity and habitat use (e.g. Galanti *et al.* 2005; Graham *et al.* 2009), as well as elephant behaviour (e.g. Hoare 1999b; Wittemyer, Douglas-Hamilton & Getz 2005), however, such methods are expensive and therefore not widely available. Aerial surveys have been used to assess interactions between people and elephants at a broader landscape scale (e.g. Parker & Graham 1989b; Hoare & du Toit 1999), but these can only provide a snapshot view of elephant distribution patterns at the specific time of the survey. Ground surveys, however, are a cost effective method that can be used to look at elephant distribution patterns in relation to human infrastructure over broader time periods (Barnes *et al.* 1991; Barnes *et al.* 1997; Blom *et al.* 2005).

This study took place in the Okavango Delta panhandle, Botswana, where it has been observed that elephants use distinctive pathways to move between the Okavango Delta and dry woodland habitat that comprise foraging areas (Jackson *et al.* 2008; Loarie, van Aarde & Pimm 2009a). This presented an opportunity to monitor the usage of such pathways to gain a greater understanding of how human presence and developments affect elephant movements and distribution in the area. The aim of this paper is to investigate a) what human infrastructure influences elephant distribution and b) how human habitat modifications affect the frequency of elephant movement and path use.

5.2. Methodology

5.2.1. Study area

The study was conducted on the eastern side of the Okavango Panhandle, where the Okavango River reaches the Okavango Delta in Botswana, between January 2008 and July 2010 (see figure 16).

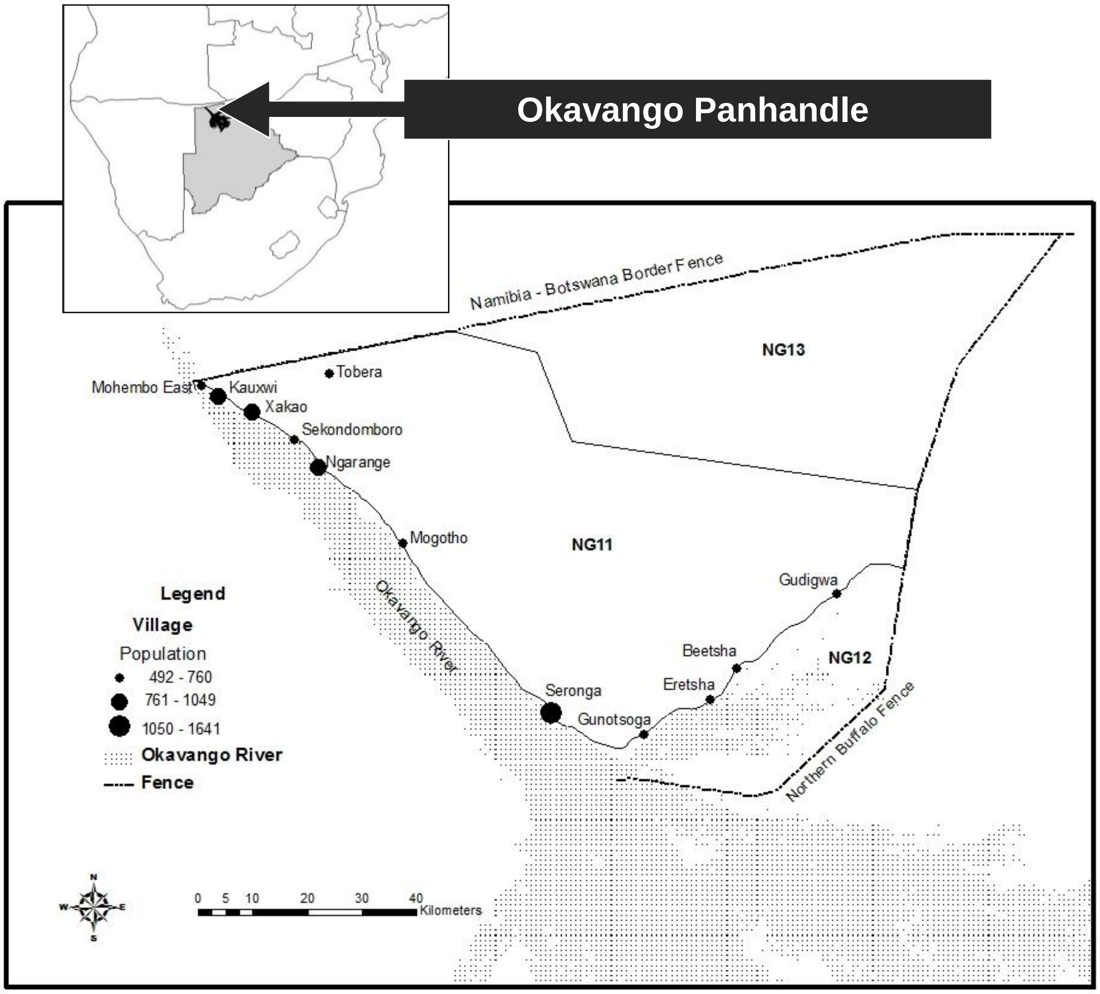


Figure 16. Study area with 12 villages illustrated according to population size

The eastern panhandle covers an area of 8,732 km^2 and consists of controlled hunting areas NG11, NG12 and NG13. The Namibian border marks the northern boundary and the northern buffalo fence marks the southern boundary (UTM Zone 34 7910000 – 7990000 South and 580000 – 710000 East). Protected areas occur north of the Namibian border fence, namely Babwata National Park and Mahango Game Reserve. South and east of the northern buffalo fence are wildlife management areas, which are utilized by photographic and hunting tourism operations.

Rainfall averages 360 - 500 mm annually and generally falls between mid October and March (rainy season), (Department of Meteorology, 2010). The mean monthly maximum temperatures in the panhandle range from 26.1 ºC– 35.1ºC, with July being the coldest month and October the hottest (Department of Meteorology, 2010). Deep Kalahari sands dominate throughout the study area, and main vegetation types include mopane woodland (*Colophospermum mopane*), silver terminalia woodland (*Terminalia sericea*), two Acacia vegetation types, namely Candle pod acacia (*Acacia hebaclada*) – Buffalo thorn (*Ziziphus mucronata*) and Umbrella thorn (*Acacia tortillis*) – Combretum (*Combretum imberbe*) valley sandveld, Zambezi Teak (*Baikiaea plurijuga*), False mopane (*Guibourtia coleosperma*), Burkea (*Burkea Africana*), Riverine woodland (e.g. *Diospyros mespiliformis; Croton megalobotrys; Acacia nigrescens; Lonchocarpus capassa; Kigelia africana*), and mixed woodland (*Colophospermum mopane*, *Terminalia sericea*, *Acacia sp.*), (Albertson 1998; Mendelsohn & Obeid 2004). Subsistence agriculture occurs in lower depressions near the Okavango River and floodplains (Tawana Land Board 2005) and in land <14km from the main road where there are fertile soils.

The 2001 census recorded 15,718 people living in the eastern panhandle, with Seronga village being the largest settlement with 1,641 inhabitants (CSO 2001). There are 12 main villages (population > 500 people) in the area, extending from Mohembo-East to Gudigwa, including Kauxwi, Xakao, Tobera, Sekondomboro, Ngarange, Mogotho, Seronga, Gunotsoga, Eretsha, and Beetsha, with additional cattle posts and settlements occurring between villages. Three main ethnic groups, namely Bahambukushu, Bayei, and Basarwa (Anikhwe and Bukakhwe river bushmen), reside in the area, each with its own ethnic identity and language. Livelihoods are closely interwoven with the diversity of natural resources, with people living off the goods and services the delta provides. The economy is, therefore, quite diverse and includes floodplain and dry land agriculture, livestock farming, wage labour, craft and tourism related enterprises (ODMP 2002; Mendelsohn & Obeid 2004). Depending on annual rainfall the planting of crops occurs between November and January and harvesting occurs between April and June.

The estimated elephant population in the eastern panhandle is between 9,000 – 15,000 elephants (see chapter 2) and elephants range throughout the study area. Telemetry studies by Jackson *et al.*, (2008) in the Okavango Panhandle region indicated that the north-south buffalo fence blocks elephant movements from the Okavango River east to the Kwando River (see figure 16) and it is reported that the Namibian border fence (see figure 16), poses a significant barrier to elephant movements between Namibia and Botswana, (Chase & Griffin 2006), suggesting that elephant movement is restricted out of the eastern panhandle.

5.2.2. Data collection

Elephants in the Okavango panhandle are very difficult to observe directly. At night when elephants cross the main (dirt) road to the river it is too dark to see individuals clearly and during the day they travel into thick inaccessible bush ~20-30km (Jackson *et al.* 2008; Loarie, van Aarde & Pimm 2009a) away from human habitation where it is dangerous to follow and thus visual observations are inhibited. It is consequently very difficult to estimate elephant age and sex from photogrammetric techniques as has been done in other studies (e.g. Laws 1966; Whitehouse & Hall-Martin 2000; Moss 2001). Therefore, an indirect method to estimate group size and group composition of elephant was used, which involves identifying and measuring hind footprints (Western, Moss & Georgiadis 1983), similar to studies such as Chiyo & Cochran, (2005).

Elephants are generally found in three types of groups: a) all male groups, b) cow/calf groups, and c) mixed groups (cow/calf groups with adult males present), (Moss 1996). However, because indirect methods were used to identify elephant age and sex I used two classifications for group composition, namely i) female led family groups and ii) male only groups. Groups containing babies and juveniles under the age of ~8 years (footprint length ≤30cm) were classified as female led family groups, as male elephants ≥8 years begin to spend more time outside of family groups with male only herds (Moss 2001). Groups containing elephants with footprints >30cm were classified as male only groups, (Lee & Moss 1995; Whitehouse & Hall-Martin 2000; Moss 2001; Chiyo & Cochrane 2005). Elephant group size was estimated by counting the number of different individual elephant's footprints present. This was achieved, in a similar way to Chiyo & Cochran, (2005) by following elephant tracks at crossings back along pathways to see how many elephants utilized those paths. Although, the substrata in the area comprises mostly fine Kalahari sands and alluvial clays, which contribute to a clear imprint of a footprint (see appendix XII), the identification of individual elephant tracks is difficult and therefore calculations of numbers of elephants per group are considered to be estimates of group size. Subsequent analyses, therefore, used two response variables in generalised linear mixed effect models (see section 5.2.3), namely i) number of herds using path and ii) number of elephants using path.

Ground surveys of the main road (180km) were conducted bi-monthly over three years to assess recent elephant activity. All elephant footprints along this transect were recorded, number of elephant groups at each crossing counted, and number of individual elephants per group estimated. Hind footprints were measured to estimate elephant age using standard methodology (Western, Moss & Georgiadis 1983; Lee & Moss 1995). GPS locations of the start and end of a crossing section, vegetation type, and elevation were recorded at all footprint sightings. The direction of elephant footprints were recorded (towards river or away) for each crossing. Only fresh footprints were recorded. All footprints were cleared (by dragging branches) after each assessment to avoid double counting footprints in subsequent assessments.

Aerial (1:50,000) orthophotographs were used to digitise well-worn elephant paths coming from the hinterland to the Okavango Delta/River. These maps were then verified using a combination of methods: ground surveys; comparisons made with existing maps from collared elephant data (Mosojane 2004; Jackson *et al.* 2008; Loarie, van Aarde & Pimm 2009a), management plans (ODMP 2002) and reports (Albertson 1998); and through consultations with local community members.

Two aerial surveys were conducted in 2008 and 2010 (see chapter 2 for detailed methodology). All elephant sightings were recorded (GPS location, number, sex, group size) during these aerial surveys.

Human settlements, fences, and cultivated land were mapped from aerial orthophotographs and Google earth using ArcMap 9.3 (ESRI). Data on human population numbers in the study area were collated from the Central Statistics Office (CSO 2001). The area of cultivated land within a 1km buffer of each elephant pathway was calculated and the distance of each pathway to human settlements and villages were measured using tools in ArcMap 9.3.

5.2.3. Statistical analysis

All analysis was carried out using the statistical package R 2.11.1 (R Development Core Team 2010).

Generalised linear models, with Poisson errors, were used to explore factors affecting the number of elephant observations recorded in aerial surveys at varying distances to human landscape modifications. Continuous explanatory variables included, closest distance to a fence, closest distance to the centre of a village and closest distance to the edge of cultivated land. The maximum model for two-way interactions was fitted and simplified by stepwise deletion of non-significant terms. Model fit was checked using chi-squared goodness-of-fit test and significance determined at $p < 0.01$.

Each elephant pathway road crossing was monitored bi-monthly for elephant activity, which may lead to temporal pseudoreplication in the data. This can cause non-independence in errors, causing a higher risk of spuriously significant results (Hurlbert 1990; Crawley 2007). Generalised linear mixed effect models (GLMM) provide a powerful tool for the analysis of such data (Pinheiro & Bates 2000; Crawley 2007) and have recently been used in several conservation studies (Rhodes *et al.* 2006; Bunnefeld *et al.* 2009; Stokes *et al.* 2010; Davies *et al.* 2011; Woodroffe 2011).

Generalised linear mixed effect models were fitted to the response variables ‘elephants present on pathway or not’ (coded 1 or 0), ‘if elephants present, estimated number of herds per crossing’, and ‘if elephants present, estimated number of elephants per crossing’. The models were fitted with two crossed random effects, elephant pathway and month. Continuous explanatory variables included, the area of cultivated land within a 1km buffer of an elephant path, the distance of a path edge to the nearest human settlement, the number of people in the nearest settlement, and elevation where elephant path intersected main road. Continuous response and explanatory variables were square root transformed to conform better to a normal distribution where appropriate. Significance tests of the fixed effects were limited to two-way interactions to avoid over fitted models and model selection was carried out using Akaike information criterion (AIC) as suggested by Whittingham *et al.*, (2006). All possible combinations of fixed effects and two-way interactions were explored and models compared using Akaike weights (see Burnham & Anderson 2002). The model with the highest Akaike weight (weight of evidence in it its favour) was considered to be the best model within the total set of models (Burnham & Anderson 2002). Interactions were plotted to aid in interpreting model outcomes. A variance components analysis of the final mixed effects models was carried out (Pinheiro & Bates 2000; Crawley 2007) in order to estimate the variation explained by the crossed random effects - elephant pathway and month.

5.3. Results

5.3.1. Elephant distribution

From the distribution of elephant herds sighted during the 2008 and 2010 aerial surveys, elephant distribution appeared to be affected by the availability of water and human landscape modifications (see figure 17). Herds were most abundant near rain filled pans or in floodplain vegetation in both years and most herds were observed in NG11. Few herds were observed along the Okavango River, where numerous human settlements occur. Where herds were observed in close proximity to settlements they were generally large, with one observation recording over 500 elephants.

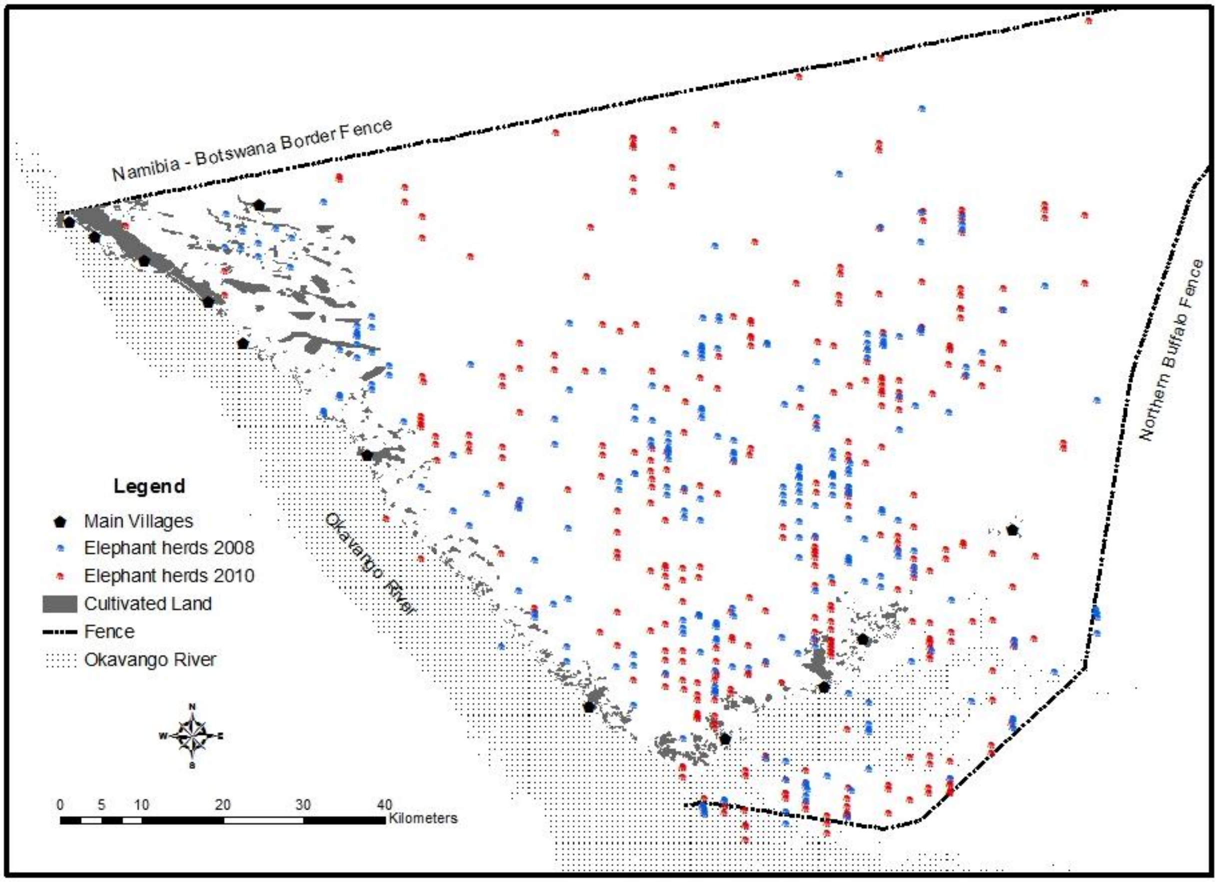


Figure 17. Distribution of elephant herds observed in the 2008 and 2010 surveys

Generalised linear models with Poisson errors for the number of elephant observations recorded, retained the variables distance to cultivated land (χ^2 = 199, df = 1, p<0.001), distance to village centres (χ^2 = 97.9, df = 1, p<0.001) and distance to fences (χ^2 = 9.7, df = 1, p<0.001) as having significant negative effects, (indicating that less elephant herds were seen near agricultural land, villages and fences). The highest percentage of elephant observations occurred 1-10km from cultivated land, 1-10km from a village and 21-30km from a fence (see table 8).

Table 8. Percentage of elephant observations from 2008 and 2010 at different distances from a) cultivated land, b) a fence and c) a village

Distance (km)	Elephant observations near cultivated land (%)	Elephant observations near a fence (%)	Elephant observations near a village (%)
<1	10.4	5.6	0.3
1-10	33.9	16	33.7
11-20	27.1	16.7	31.5
21-30	17.7	28.4	19.8
31-40	8.9	25.4	9.4
41-50	1.3	7.8	3.7
>50	0.7	0	1.5

5.3.2. Elephant pathways

Over three years, 106 main elephant pathways were identified along the main road transect (see figure 18 for map). A total of 12,868 elephant herd crossings were recorded between March 2008 and June 2010, with a mean number of 119 (± 255) herds crossing per pathway. Significantly more family groups were recorded than male only herds (χ^2 = 171.7, df = 1, p<0.001).

It is evident from figure 18 that there are areas along the panhandle which elephants seem to avoid, these less utilised areas mainly occur near villages. From examining orthophotos and satellite images it is apparent that pathways lead towards easily accessible permanent water (i.e. lagoon or accessible sections of the main Okavango River channel).

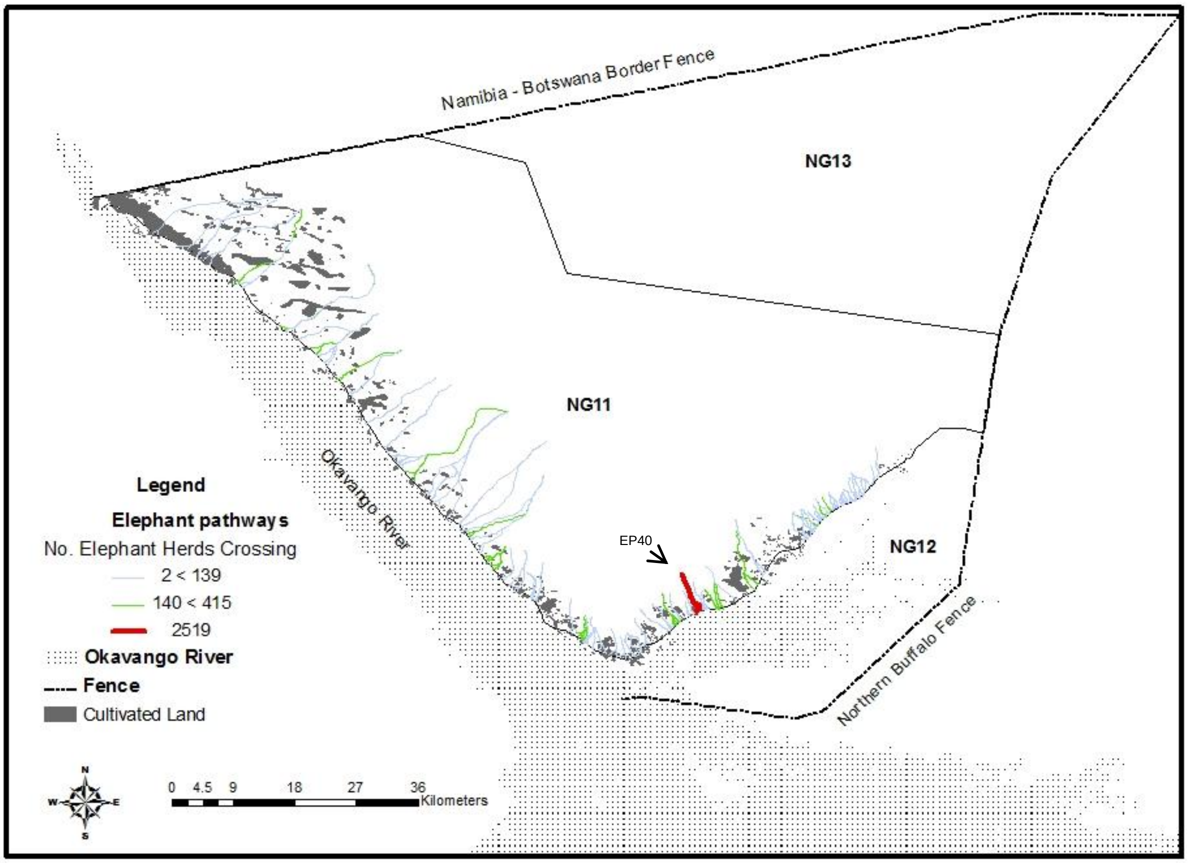


Figure 18. Digitised main elephant pathways in the Okavango Delta Panhandle. Paths are colour coded to show low (light blue), medium (green), and high (red) utilisation gradients based on number of herds crossing the road over 3 years. Pathways were numbered from east-west (i.e. furthest to the east is EP1)

Male herds and female led family groups appeared to prefer different pathways (see figure 19), with some paths only used by male herds, while others were more frequently used by family groups. Certain pathways were used more frequently than others by both male herds and female led family groups. One elephant pathway (EP40) was utilised significantly more frequently throughout the year than all other pathways combined (t = 8.8, df = 26, p <0.001) (see figures 18 and 19). Consequent analysis was therefore conducted with and without this pathway, to assess the effect of this heavily utilised pathway on results.

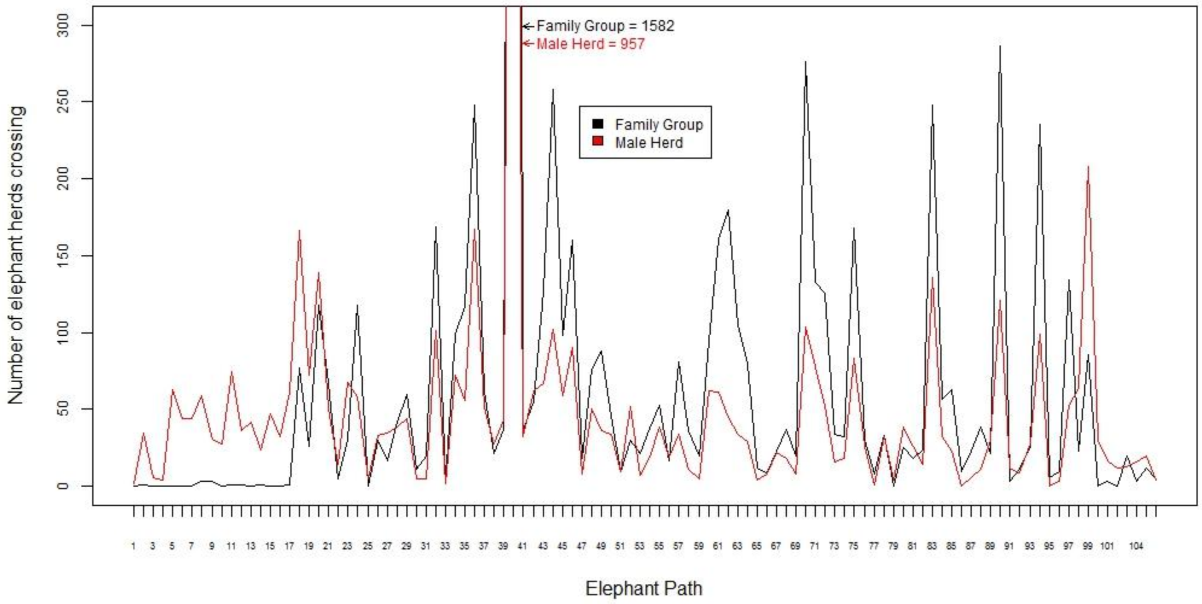


Figure 19. Number of male and female elephant herds utilising each pathway over three years (2008-2010)

Significantly more elephant crossings were recorded travelling away from the river (χ^2 = 84.7, df = 1, p<0.001). To avoid double counting elephant crossings by the same herd, i.e. counting their going and coming from the river as two separate groups, only data from groups travelling away from the river (n = 6757) were used in further analyses.

5.3.3. Factors affecting elephant path usage

The top five generalized linear mixed effect models (GLMM) of the three response variables i) elephants using a pathway or not; ii) number of herds using a pathway; iii) estimated number of elephants using a pathway, for main effects and two-way interactions ranked according to AIC weights are shown in tables 9,10 and 11 respectively.

The generalized linear mixed effect model, of whether elephants used a pathway or not, with the smallest AIC value (AIC = 3010) and highest model weight (w_i = 0.34) retained main fixed effects elevation (ML estimate = -5.8, SE = 2.1) and human population (ML estimate = -4.7, SE = 1.2) and the two-way interaction between these two main effects (ML estimate = 0.18, SE = 0.07). This model shows that there is likely to be elephants present when the human population is less, but the probability of elephants occurring when human populations are larger increases when the elevation is higher (see appendix XIII for interaction plot). The variance components of the random effects show that 38.8% of variation is between elephant paths and 61.2% is between months at each pathway. When the GLMM was fitted to a subset of data where crossings on the heavily utilised elephant path "EP40" were removed, the model estimates of the fixed effects were not significantly affected.

Table 9. Summary of Akaike Information Criteria (AIC) and Akaike model weights (w_i) (estimated probabilities of model truth) relating probability of elephants using a path or not (coded 1 or 0) to combinations of environmental and human habitat modification variables. The top five models ranked according to AIC model weights are presented here.

	Model	AIC	w_i
Main effects	H+E	3014.9	0.04
	A+H+E	3016.5	0.02
	D+H+E	3016.8	0.01
	A+D+H+E	3018.5	0.01
	A+H	3019.7	0.00
2-way interactions	H:E	3010.4	0.34
	H:E+A:E	3012.3	0.14
	H:E+A:H	3012.4	0.13
	H:E+A:D	3012.4	0.13
	H:E+A:E+A:D	3014.2	0.05

Variable codes: A = sqrt(Area agriculture within 1km buffer from path); E = sqrt(Elevation); D = sqrt(Distance to nearest settlement); H = sqrt(Human population in nearest settlement)

The generalized linear mixed effect model, of the square root of the number of elephant herds (if present) using a pathway, with the smallest AIC value (AIC = 631.1) and highest model weight (w_i = 0.21) retained main fixed effects human population in nearest settlement (ML estimate = -0.01, SE = 0.005), area of agriculture within a 1km buffer of path (ML estimate = -0.03, SE = 0.01) and distance to nearest human settlement (ML estimate = -0.003, SE = 0.003) and the two-way interaction between the latter two main effects (ML estimate = 0.0008, SE = 0.0003). This model shows that more herds are likely to cross on pathways near less populated settlements. There is also likely to be less elephant herds crossing where there is a larger area of agriculture near a pathway and where pathways are close to human settlements, but the likelihood of more herds crossing when there are larger areas of agriculture increases when the distance to human settlements increases (see appendix XIII for interaction plot). Variance components show that 67.8% of variation is between elephant pathways and 32.2% is between months at each pathway. When the GLMM was fitted to a subset of data where crossings on elephant path "EP40" were removed, the model estimate for the fixed effect was not significantly affected.

Table 10. Summary of Akaike Information Criteria (AIC) and Akaike model weights (w_i) (estimated probabilities of model truth) relating the likelihood of a larger number of herds (>0) present at a pathway crossing to combinations of environmental and human habitat modification variables. The top five models ranked according to AIC model weights are presented here.

	Model	AIC	w_i
Main effects	H	633.8	0.05
	A+H	635.3	0.03
	D+H	635.6	0.02
	A+D+H	636.5	0.01
	A+H+E	637	0.01
2-way interactions	A:D	631.1	0.21
	A:D+A:E	632.1	0.13
	A:D+A:H	632.1	0.13
	A:D+H:E	632.4	0.11
	A:D+A:E +H:E	633.3	0.07

Variable codes: A = sqrt(Area agriculture within 1km buffer from path); E = sqrt(Elevation); D = sqrt(Distance to nearest settlement); H = sqrt(Human population in nearest settlement)

The generalized linear mixed effect model, of the square root of the estimated number of elephants (if present) using a pathway, with the smallest AIC value (AIC = 1662.3) and highest model weight (w_i = 0.33) retained main fixed effects human population in nearest settlement (ML estimate = -0.02, SE = 0.006), elevation (ML estimate = 1.33, SE = 0.48), area of agriculture within a 1km buffer of path (ML estimate = 3.49, SE = 0.99), and distance to nearest human settlement (ML estimate = -0.007, SE = 0.003) and the two-way interaction between elevation and area of agriculture (ML estimate = -0.11, SE = 0.03) and distance to nearest settlement and area of agriculture (ML estimate = 0.001, SE = 0.0004). This model shows that larger groups of elephants are more likely to cross on pathways near less populated settlements. There are also likely to be more elephants using a path if it is near larger amounts of agriculture and further from settlements. Although it is likely that more elephants will use a path if it is near large areas of agriculture, this likelihood will decrease when elevation increases. However, graphs of these interactions show a small amount of curvature indicating that such relationships are unlikely to be biologically important (see appendix XIII for interaction plots). Variance components show that 53% of variation is between elephant pathways and 47% is between months at each pathway. When the GLMM was fitted to a subset of data where crossings on elephant path "EP40" were removed, the model estimates for the fixed effects were not significantly affected.

Table 11. Summary of Akaike Information Criteria (AIC) and Akaike model weights (w*i*) (estimated probabilities of model truth) relating the likelihood of a larger number of elephants (>0) present at a pathway crossing to combinations of environmental and human habitation modification variables. The top five models ranked according to AIC model weights are presented here.

	Model	AIC	w_i
Main effects	A+H	1688.1	8.6e-07
	H+D	1688.4	7.3e-07
	H	1688.9	5.6e-07
	A+H+D	1689.1	5.1e-07
	H+D+E	1689.9	3.5e-07
2-way interactions	A:D+A:E	1662.3	0.33
	A:D+A:E+H:E	1662.9	0.25
	A:D+A:E+A:H	1664.1	0.14
	A:D+A:E+ D:E	1664.3	0.12
	A:D+A:E+D:E+H:E	1664.6	0.12

Variable codes: A = sqrt(Area agriculture within 1km buffer from path); E = sqrt(Elevation); D = sqrt(Distance to nearest settlement); H = sqrt(Human population in nearest settlement)

5.4. Discussion

Where human land occupation and elephant habitat overlap, changes in elephant distribution and movement patterns occur. In previous studies, elephants have been found to avoid unprotected areas (Wittemyer *et al.* 2007) with high human population density (Parker & Graham 1989b; Barnes *et al.* 1991; Eltringham 1991; Happold 1995; Hoare & du Toit 1999; Harris *et al.* 2008; Jackson *et al.* 2008; Graham *et al.* 2009), human settlements (Harris *et al.* 2008) and high human activity (Graham *et al.* 2009). Similarly, my study indicates that the movements and spatial use of elephants utilising the eastern Okavango Delta panhandle appear to be affected by human land occupation and associated activities. Less elephant groups were observed close to human habitat modifications such as cultivated land, settlements and fences. In general, elephants were more likely to utilise areas with a lower human population. Specifically, my results show that elephant paths near sparsely populated human settlements were more likely to be utilised by elephants and the number of elephants utilising paths decreased as the number of people in the nearest settlement increased and the distance of a movement path to a settlement increased. When the area of agriculture near a path increased the number of elephant herds utilising a path decreased but the number of elephants in a herd increased.

Such findings have important implications for the management of elephants in the area and for addressing human-elephant conflict through mitigation strategies such as land use planning. As Morris, (1992) discussed, animal movements and habitat utilisation can be influenced by factors at different scales. Some factors may affect the movement of animals between home ranges and others on a more local scale, between foraging sites. Human habitat modifications such as fences can have detrimental effects on large scale animal migration routes by impeding movement across fences (e.g. zebra (*Equus burchelli*) and wildebeest (*Connochaetes taurinus*) (Albertson 1998)). This is thought to be the case in the Okavango panhandle, where elephants may potentially be trapped between the Namibian border fence, the northern buffalo fence and the Okavango River (Jackson *et al.* 2008; Chase & Griffin 2009; Loarie, van Aarde & Pimm 2009a). Small scale, localised movements may also be affected by human modifications such as settlements or cultivated land, which reduces the natural habitat and available foraging sites and also leads to an increase in human activity in the area. Where people and wildlife share space, wildlife movements may, therefore, not only be affected directly by human habitat modifications on both the dispersal and foraging scale, but also due to risk avoidance strategies in response to human behaviour (Graham *et al.* 2009).

Avoiding potential sources of disturbance may be a key factor in determining movement patterns of elephant in human dominated landscapes (Graham *et al.* 2009), a phenomenon which has been recorded with other large herbivore species too, such as roe deer *Capreolus capreolus* (Coulon *et al.* 2008) and Mongolian gazelles (*Procapra gutturosa*) (Olson *et al.* 2011). My results show that this is likely true in the eastern Okavango Delta panhandle. Elephants in the panhandle move towards the river or closer to human activity during the cover of darkness (Loarie, van Aarde & Pimm 2009a), a behaviour which has been recorded in other study sites (e.g. Hoare 1999b; Hoare & du Toit 1999; Graham *et al.* 2009) and illustrates an example of a risk avoidance strategy. During my aerial surveys, elephants were observed aggregating in larger groups near human habitations, which could also be a strategy to minimise risk (safety in numbers). This aggregation strategy to minimise risk could explain why the highest percentage of elephants observed in surveys was within the 1-10km distance category from cultivated land and settlements. In addition, elephant movement behaviour likely adopted risk avoidance strategies; elephants avoided movement pathways near anthropogenic disturbances such as large human populations and large areas of cultivated land.

Larger numbers of elephants, however, were more likely to use paths close to larger areas of cultivated land. This could indicate that the cultivated land is attracting a larger number of elephants as a foraging resource (due to the high nutritional content in crops compared to natural forage) as surmised in previous studies (Sukumar 1989; Osborn 1998b; Chiyo *et al.* 2005). This may certainly be true during the crop season as cultivated land would be more attractive to elephants when there are crops in the fields. However, a brief examination of the temporal patterns of elephant path use shows that significantly more elephant herds used paths closer to crop-land between May and November (see appendix XIV for graph), yet the crop season in the study area occurs between January and May, which indicates that elephants may actually be avoiding cultivated areas during this time, again to avoid increased human activity. Although crop raiding by elephants does occur in the Okavango Delta panhandle (see chapters three and four), there is no clear evidence to suggest that elephants are actually attracted to cultivated land. However, currently many people in the Okavango Delta panhandle believe that elephants are attracted to their fields to eat crops. Such a result could, therefore, be important in improving farmer perception of elephants in this area, which can help reduce perceived levels of conflict (see chapter 6).

Alternatively, it may be that elephants are moving in larger groups in areas where there is more cultivated land as a risk avoidance strategy, i.e. owing to increased risk associated with cultivated land elephants prefer to move in larger groups (safety in numbers). However, results from the GLMM of the number of elephant herds utilising a path showed that the likelihood of more herds crossing in highly cultivated areas increases when the distance to human settlements increases. This indicates that elephants may be avoiding humans themselves, i.e. around settlements, rather than cultivated land. Areas of crop-land present a lower risk of disturbance than areas near settlements, thereby providing a safer route of passage and hence attracting larger numbers of elephants.

Elephant movements and habitat use have been found to be influenced by environmental factors such as palatability, availability and nutritional quality of food (Sommerlatte 1976; Jachman 1984; Sukumar 1990; Viljoen & Bothma 1990; Shannon *et al.* 2006); and vegetation heterogeneity (Murwira & Skidmore 2005; Chamaille-Jammes, Valeix & Fritz 2007; Loarie, van Aarde & Pimm 2009b). In dry savanna ecosystems, like Botswana, abundance and distribution of woody vegetation is affected by landscape and rainfall gradients, with elevation and substrata gradients providing strong controls on species assemblages (Coughenour & Ellis 1993). My study found that elephant movements were influenced by changes in elevation, which could be explained by changes in resource availability or quality at different elevations. Elephants were more likely to use movement paths where human populations are large if the elevation was higher, yet at higher elevations the likelihood of more elephants using a path near large areas of agriculture decreases. Elephant movement appears to be affected by a two-tiered set of factors, firstly by anthropogenic disturbances such as human population numbers or cultivated land, and second by changes in elevation and hence available food resources.

My study investigated a limited number of possible drivers of elephant movement, which could account for the relatively small proportion of variance that my models explained. For example, I did not account for the affect of water availability (Verlinden & Gavor 1998; Stokke & duToit 2002; Redfern *et al.* 2003; Grainger, van Aarde & Whyte 2005; Chamaille-Jammes, Valeix & Fritz 2007) or social factors such as dominance hierarchy among family groups (Wittemyer *et al.*

2007), or sexual segregation (Moss & Poole 1983; Stokke 1999; Galanti *et al.* 2005), or temporal factors (e.g. Graham *et al.* 2009). Further investigations are required to fully understand how resource availability affects elephant movements in the Okavango Delta panhandle, but my study indicates that human activity has a major role in determining elephant movements and could, indeed, be more influential than resource availability in areas where elephant occur in human-inhabited landscapes.

Elephants appear to avoid potential sources of disturbance in human dominated landscapes in Botswana, as well as in other parts of their range i.e. Kenya (Graham *et al.* 2009). This is encouraging for managers of elephant populations where this appears to be the case, as it indicates there is potential for minimising overlap between human and elephant resource use through land use planning interventions. It is believed realistic to consider creating zones within a multi-use landscape as a method of minimising negative interactions between humans and wildlife (Fernando *et al.* 2005; Linnell *et al.* 2005). In the Okavango panhandle, it is evident that elephants use certain pathways more frequently than others. Elephants appear to avoid areas where human settlements are larger or in areas that are more developed by humans. For example, the north-west corner of the study area is heavily utilised by people with five main villages occurring within a 40km strip of the road and where lots of land has already been converted to arable-land. This area has fewer elephant paths, which are less utilised compared with those pathways further south and east. This area could theoretically be zoned for future agricultural development and could even be protected from elephant crop-raiding by large scale or co-operative fencing, because it is already infrequently utilised by elephant (see appendix XV). Such land use zoning could therefore reduce the amount of crop damage by elephants, whilst having a minimal effect on elephant movements in the area. Conversely, those pathways that are frequently used, e.g. EP40 and evidently important for elephant movement and resource use should be afforded a free movement buffer zone along their route, where arable land allocation should be prohibited, thereby reducing the likelihood of creating a barrier to elephant movement, reducing the human induced disturbance factor and, ultimately, reducing human-elephant overlap and its resulting conflict. My study provides evidence to suggest that such land use management interventions should be used as a form of human-elephant conflict mitigation in the eastern Okavango Delta panhandle, and indeed, other nearby areas around the Okavango and other major waterways in the region, where elephants regularly use pathways to and from resources in human inhabited landscapes.

Chapter 6

Exploring the complexities of perceived conflict between people and wildlife

6.1. Introduction

People's perception of the cost of living with or close to certain wildlife species is greatly influenced by social determinants that are borne from complex interactions between cultural, social and personal factors (Dickman 2010). The perception of the level of negative interactions they are experiencing with wildlife are as important as the actual losses from wildlife damage (Naughton-Treves & Treves 2005), as they affect the level of antagonism felt towards the perpetrating wildlife species and consequently influence what actions people are likely to take in such situations (Dickman 2010). Gaining a social perspective of negative interactions with wildlife can enable assessment of the perceived severity of these interactions and identify the varying capacity of individuals to cope with such losses (Naughton-Treves & Treves 2005). This can then illuminate the social factors that intensify negative interactions or favour coexistence (Knight 2000) and help guide effective management interventions that will be more acceptable or implementable (Jonker *et al.* 2006). Indeed, understanding the social determinants of negative interactions between people and wildlife is critical in order to develop and implement successful, long term mitigation strategies.

Perception of dangerous wildlife or pest species and the extent of negative people-wildlife interactions can often appear to be over exaggerated or fabricated. Larger animals, protected animals and those that cause more conspicuous damage tend to receive a greater number of reports in interview studies (Bell & McShane-Caluzi 1984; Dudley, Mensah-Ntiamoah & Kpelle 1992; Lahm 1996; Hill 1997; Naughton-Treves 1998; Siex & Struhsaker 1999; Weladji & Tchamba 2003; Linkie 2007; Warren, Buba & Ross 2007). For example, Arlet & Molleman (2007) found that the Agile mangabey (*Cercocebus agilis*) was regarded as a significant pest species from community interviews in Cameroon, despite the fact that little damage by Agile mangabey was observed during field studies. Likewise, Linkie *et al* (2007) found that Wild boar (*Sus scrofa*) were considered the worst crop-raiders in Sumatra, yet field monitoring showed the pig-tailed macaque (*Macaca nemestrina*) actually caused most damage. Gillingham & Lee (2003) identified reported crop damage to be greater than actual damage near the Selous game reserve, Tanzania. In an extreme case of disparity between actual and perceived conflict Siex & Struhsaker (1999) found that, contrary to farmers' perceptions, the Zanzibar red colobus (*Procolobus kirkii*) monkey actually improved coconut yields in Zanzibar rather than impede them. Such discrepancies indicate that factors other than actual quantifiable damage contribute to perceived conflict levels (Gillingham & Lee 2003; Linkie 2007). Although, previous studies have documented such discrepancies and consequently highlighted the need to assess both perceived as well as actual dimensions of negative wildlife interactions (e.g. Siex & Struhsaker 1999; Gillingham & Lee 2003; Hill 2004; Naughton-Treves & Treves 2005; Linkie 2007), a fundamental gap still exists in current knowledge surrounding what factors actually contribute to determine perceived conflict.

The extent to which people accept the presence of wildlife and tolerate interactions with wild animals, known as people's 'wildlife acceptance capacity' (WAC) (Decker & Purdy 1988), is influenced by people's experiences, attitudes and values (Jonker *et al.* 2006). Human attitudes towards wildlife species and consequent tolerance of living close to animals causing negative interactions appears to be affected by species of animal, people's religious and cultural beliefs, education levels and other social factors (Bjerke, Reitan & Kellert 1998; Sekhar 1998; Holmern, Nyahongo & Roskaft 2007). Negative feelings towards some wild animals, can also be exacerbated by past experiences e.g. the occurrence of crop raiding, property destruction or livestock depredation (Getz *et al.* 1999; O'Connell-Rodwell *et al.* 2000; Hill 2004; Madden 2004; Naughton-Treves & Treves 2005; Dickman 2010). Kellert, *et al.* (1996), surmised attitudes towards wildlife as being a consequence of four interacting variables: i) basic values toward animals and nature; ii) physical and behavioural characteristics of a species, including the animal's size, perceived intelligence, morphology, mode of locomotion, cultural and historical associations, and other factors; iii) knowledge and understanding of a species, including factual, conceptual, and conservation awareness; and iv) past and present interactions with particular species, including conflict, recreational use, property relationships, and management status. Attitudes toward dangerous animals such as wolves or elephants may also be influenced by the perceived risks of living near such wildlife species (Knight 2000; Naughton-Treves, Grossberg & Treves 2003; Kaltenborn, Bjerke & Nyahongo 2006). If animals are perceived differently they are likely to be treated differently (e.g. Warren, Buba & Ross 2007) and hence their management may need to vary accordingly, which has implications for conservation policy. People's view or acceptance of wildlife management interventions are often influenced by WAC and it is, therefore, essential to gain an understanding of people's attitudes in order to develop socially acceptable wildlife management strategies (Jonker *et al.* 2006).

A combination of social and cultural factors as well as prior experience also contribute to people's perception of risk to wildlife damage (Hill 2004). Factors such as, the species and size of animals, how dangerous wildlife are considered to be, rare or extreme events, and the degree of control an individual feels they have over wildlife activities are all thought to influence perceptions of risk (Hill 2004; Naughton-Treves & Treves 2005). If people believe they have little control over a conflict situation or have limited coping strategies then they are likely to further inflate perceptions of risk (Hill 2004), which in turn could increase negative attitudes towards wildlife and intensify the perceived level of conflict. An understanding of the consequences of a conflict situation and the available coping strategies to losses incurred by people is, therefore, important when addressing such issues. If the aim of managing a human-wildlife conflict situation is to reduce the conflict, then reducing perceived risk, raising people's tolerance of damage and improving their attitude toward the perpetrating species appear to be as important as reducing the damage itself (Osborn & Hill 2005).

Naughton-Treves & Treves, (2005) also highlighted that local intolerance for wildlife may be enlarged by institutional constraints on coping strategies, with people being less tolerant of imposed risk than voluntary risk. Therefore, when wildlife species are perceived to belong to the government, people's perception of risk may be heightened and tolerance reduced. For example, African elephants (*Loxodonta africana*) who are protected under international and national laws, are large and potentially dangerous animals who inspire animosity and fear among people living within their range (Naughton-Treves 1999) and often compete with humans for space and limited resources (Siex & Struhsaker 1999; Osborn & Hill 2005). People living close to elephants, therefore, often view them as a threat to their livelihoods and indeed their lives (Smith & Kasiki 1999). Exploring peoples' perceptions of negative interactions with elephants and attitudes towards elephants may therefore provide insights into underlying issues of human-wildlife conflict scenarios and aid in our understanding of how to focus management strategies in cases where large and dangerous protected wildlife species are the perpetrators of human-wildlife conflict.

From the literature, it is clear that when designing mitigation and management strategies to reduce HWC it is important to consider perceived conflict levels alongside actual conflict measurements. To understand perceptions and attitudes, however, it is necessary to identify factors which influence them. This paper, therefore, aims to determine factors affecting attitudes towards elephants and perceptions of conflict of subsistence farmers in the Okavango Delta panhandle, Botswana. Four main questions are addressed: a) What social factors affect attitudes towards wildlife and elephants? b) How severe is the perceived level of conflict with elephants? c) What are the varying coping strategies of people experiencing negative interactions with elephants? and d) do the views of rural farmers indicate possibilities for coexistence?

6.2. Methodology

6.2.1 Study area

This study took place on the eastern side of the Okavango Delta panhandle in Botswana (see figure 20), between 2008 and 2010. The Okavango panhandle describes the area where the Okavango River enters Botswana before dispersing across the alluvial fan of the Okavango Delta (Kgathi *et al.* 2006). My study area covered an area of 8,559 km^2 and consists of controlled hunting areas NG11, NG12 and NG13, with the Namibian border marking the northern boundary and the northern buffalo fence marking the southern boundary (UTM Zone 34 7910000 – 7990000 South and 580000 – 710000 East). Most agricultural activity takes place on the periphery of the Delta in NG11 and NG13 of the Okavango panhandle, while tourism activities mainly take place in NG12.

Deep Kalahari sands dominate throughout the study area, with fertile soils that support subsistence agriculture occurring in lower depressions on land near the Okavango River and floodplains (Tawana Land Board 2005) and in dry land <14km from the main road. Rainfall averages 360 - 500 mm annually and generally falls between mid October and March (wet season). The mean monthly maximum temperatures in the panhandle range from 26.1 ºC– 35.1ºC, with July being the coldest month and October the hottest. The estimated elephant population in the eastern panhandle is 15,429 elephants and elephants range throughout the study area (see chapter 2).

The 2001 census recorded 15,718 people living in the eastern panhandle, with Seronga village being the largest settlement with 1,641 inhabitants (CSO 2001). There are 12 main villages in the area, extending from Mohembo-East to Gudigwa, including Kauxwi, Xakao, Tobera, Sekondomboro, Ngarange, Mogotho, Seronga, Gunotsoga, Eretsha, and Beetsha, with additional cattle posts and settlements occurring between villages. Three main ethnic groups, namely Bahambukushu, Bayei, and Basarwa (River Bushmen), reside in the area, each with its own ethnic identity and language. Livelihoods are closely interwoven with the diversity of natural resources, living off the goods and services the delta provides. The economy is, therefore, quite diverse and includes floodplain and dry land agriculture, livestock farming, wage labour and craft and tourism related enterprises (ODMP 2002; Mendelsohn & Obeid 2004). Depending on annual rainfall the planting of crops occurs between November and January and harvesting occurs between April and June (Mosojane 2004).

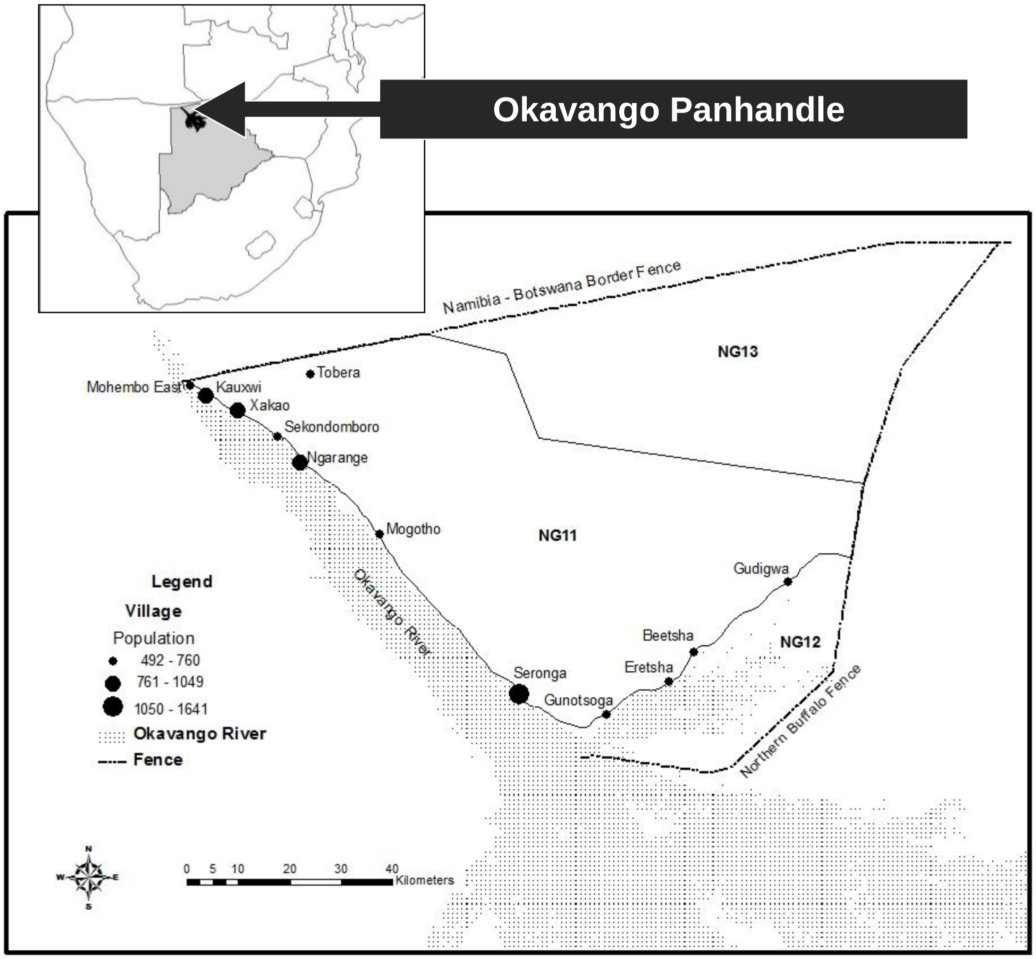


Figure 20. Study area with 12 villages illustrated according to population size

6.2.2. Sampling

My target population for the survey were subsistence farmers in the eastern Okavango Delta panhandle. To gather representative data and record the diverse opinions and views of the farming community, participants were selected through a stratified and randomized approach. Only one farmer per household was interviewed. Participants represented both genders, a range of ages and included individuals from different economic, social and cultural backgrounds. A random sample of participants were selected from 2008-2010 elephant raided and non-raided fields and the number of participants selected from each village aimed to represent 5-10% of the total village population. Potential respondents were approached by the interviewer and asked if they would consent to an interview about their farming practices, and knowledge and feelings about elephants. If the farmer agreed, the interview proceeded; only three farmers refused to be interviewed.

People living in and around the Delta have been involved in numerous socio-economic surveys, therefore questionnaires were designed to ask direct and relevant questions, and time being interviewed minimized (35 - 40min) to prevent respondent fatigue. To reduce bias the time-recall period was minimised, the principal investigator (A. Songhurst) spent two years in the community so that interviewees' and researcher understanding of each other's motivations were more closely aligned, and all research assistants and translators were thoroughly trained. During the surveys, interviewers and translators practiced awareness of national policies on gender, HIV and other sociological and economic issues. Ethical approval for this study was granted by the Imperial College Research Ethics Committee.

6.2.3. Administration of questionnaire

Structured and open-ended questions were used in interviews to collect quantitative and qualitative information (see appendix XVI for questions used). A simple question-and-answer format was adopted for the questionnaires, to maximize the accuracy of data by minimizing possible biases caused by misinterpretation by respondents or researchers (White *et al.* 2005). Open-ended questions were avoided when gathering factual information but used when the questions were seeking to understand views or motivations of respondents. Respondents had the option to decline to answer individual questions if they so wished.

Respondents were interviewed either at their fields or in their homestead at the village. The questionnaire was written in English then translated into written Setswana and checked by a university lecturer (Dr. C. Bonyongo) at the University of Botswana for appropriateness and correctness of interpretation. Pilot studies were conducted on a random selection of 20 farmers from different cultural groups until questions were being answered consistently, to ensure question clarity and effectiveness of questions in gaining the required information. Interviewers were fluent in English and were familiar with the study area and the locations of fields/compounds where interviews were conducted. Each interviewer was assisted by a translator who was both fluent in English, Setswana and the local tribal language (either: Hambukushu; Seyei; or Sisarwa) and familiar with farmers in the village. The researcher who designed the questions (A. Songhurst) discussed each question with the translators to ensure that they understood the questions prior to interviews. To ensure accuracy, the same interviewers were used and the same interview questions were administered during the study.

A wealth ranking criterion was established which included ownership status of field; number of cattle owned; house type and other household possessions owned (see appendix XVI). A conflict level ranking criterion was established based on the number of years a respondent's field had been raided in the past and the number of times a respondent's field had been raided in year of interview (see appendix XVII).

Qualitative data arising from open-ended questions in the survey were recorded, systematically evaluated, and coded, and any emerging themes identified (Maxwell 2005). These themes were matched with quantitative findings. This mixed methods approach was used to both triangulate responses and to further interpret the results, in a comparative way to other studies (e.g. Mauro, McLachlan & van Acker 2009).

Each interview covered several topics including: (1) interviewee socioeconomic details and wealth ranking; (2) agricultural practices; (3) opinions towards wildlife and elephants; (4) local knowledge of elephants; (5) problems with wildlife and elephants; (6) possible options for mitigating human–elephant conflicts; (7) feelings towards living with elephants.

6.2.4. Hypotheses testing

6.2.4.1. Social factors

The first question I aimed to address was: What social factors affect attitudes towards wildlife and elephants? Specifically, I aimed to determine what the key drivers of people's feelings towards elephants and towards living near wildlife were. Respondents were asked to identify how they felt towards elephants (1 = love, 2 = like, 3 = neutral, 4 = dislike, 5 = hate) and whether they liked living near wildlife (1 = yes, 0 = no). The level of feeling towards elephant (coded 1-5) and attitude towards wildlife (coded 0 or 1) were used as the independent units of analysis in statistical analyses.

Ordered probit models (see section 6.2.5. for details of ordered probit model analysis) were used to explore factors affecting feelings towards elephants (coded 1-5), with continuous and categorical explanatory variables tested in combination. Explanatory variables included socioeconomic factors (gender, age, highest education level reached, wealth ranking, livelihood strategy other than farming), indicators of actual conflict level (number of times field raided this year, number of years field raided, area damaged, number of known people killed by elephants), perceived conflict level (coded 0-4), attitude towards living near wildlife (1 = like, 0 = dislike) and whether or not respondent’s village received a direct source of income from tourism (coded 0 or 1), (see appendix XVIII for details of variables).

A Generalised Linear Model (GLM) for whether or not people liked living near wildlife (coded 0 or 1), with binomial error structure, was used to test continuous and categorical explanatory variables independently and in combination. Explanatory variables included socioeconomic factors (gender, age, highest education level reached, wealth ranking, livelihood strategy other than farming), indicators of actual conflict level (number of times field raided this year, number of years field raided, area damaged, number of known people killed by elephants), perceived conflict level (coded 0-4), attitude towards elephants (coded 1-5) and whether or not respondent’s village received a direct source of income from tourism (coded 0 or 1), (see appendix XVIII for details of variables).

6.2.4.2. Perceived conflict

The second question I aimed to address was: How severe is the perceived level of conflict with elephants? Respondents were asked if they felt like they were in conflict with elephants and if they answered yes they were asked to grade the level of conflict (0 = none to 5 = high) they were experiencing. Data were collected on actual conflict levels per farmer, by asking questions such as: How many times has your field been raided by elephants this year?; In which previous years has your field been raided by elephants?; How many people in or near your village have been killed by elephants in the past 5 years?, and combined with data on actual measurements of crop damage collected in another part of this study (see chapter three).

People have been found to often over-report the scale of a human-wildlife conflict problem, leading to a mismatch between local perceptions of wildlife damage and its actual occurrence (Gillingham & Lee 2003; Linkie 2007). I compared perceived level of conflict with indicators of actual conflict using ANOVA. Continuous actual conflict indicators (number of times raided this year, number of years previously raided, known number of people killed by elephants, and the area damaged in most recent elephant raid) were used as response variables.

To explore the effectiveness of using crop-raiding (the most prevalent form of HEC in the study area) as an indicator of actual conflict, I developed a conflict ranking for respondents based on elephant crop-raiding history of fields (see appendix XVII). This categorical indicator of actual conflict (0 = none to 5 = high) was compared to the perceived conflict ranking levels using a chi-squared goodness-of-fit test. The spatial distribution of calculated and perceived crop-raiding levels were plotted to give a visual representation of differences.

To determine what factors influence perception of conflict, I used ordered probit models to test possible continuous and categorical explanatory variables in combination. Explanatory variables represented socioeconomic factors, attitudes towards wildlife and elephants, field location, raiding history of field, and preferred mitigation options of respondents (see appendix XVIII for details of variables used).

6.2.4.3. Coping strategies

The third question this study aimed to address was: What are the varying coping strategies of people experiencing negative interactions with elephants? First, I summarised productivity of fields in good and bad harvests relative to productivity levels needed to feed dependant people. Second, I investigated what coping strategies were available to respondents during years of bad harvests and which of these were the preferred strategies. A pie chart was drawn to illustrate the percentage of respondents favouring different strategies.

The fourth question investigated was: Do the views of the farming community indicate possibilities for coexistence? First, a generalised linear model for whether or not people believed there is a solution to reduce the HEC they are experiencing (coded 0 or 1), with binomial error structure, was used to test continuous and categorical explanatory variables independently and in combination. Explanatory variables included socioeconomic factors (gender, age, highest education level reached, wealth ranking, livelihood strategy other than farming), indicators of actual conflict level (number of times field raided this year, number of years field raided, area damaged, number of known people killed by elephants), perceived conflict level (coded 0-4), productivity of field (amount of crop usually harvested, amount of crop harvested in a bad year, amount of crop needed to support dependants), acceptable management strategies (favoured strategies, whether respondent willing to move field (coded 1 = yes, 0 = no), whether elephant paths should be considered during land use planning (coded 1 = yes, 0 = no), whether respondent agrees with killing elephants (coded 1 = yes, 0 = no)), whether respondent believes compensation rates are fair (coded 1 = yes, 0 = no) and who respondent feels is responsible for reducing crop raiding by elephants and for protecting their field, (see appendix XVIII for details of variables).

Second, favoured management strategies were summarized and chi squared tests used to investigate differences between attitudes towards who is responsible for reducing crop raiding and who is responsible for protecting individual fields. Four main development issues were identified in the study area prior to questionnaire surveys through informal interviews with local residents, namely no electricity, no bridge across river, no tarmac road and problems with elephants. To investigate how serious people viewed elephant crop-raiding in the area, respondents were asked “which of these development issues should be the priority to address in the study area?” Chi squared tests were used to compare responses to this question.

6.2.5. Data analysis

All data analysis was carried out using R 2.11.1 (R Development Core Team 2010).

The data were screened for collinearity and outliers prior to analysis (using box plots and scatter plots). Descriptive statistics were derived for all factual and attitudinal questions. Continuous data were square root transformed to approximate better to a normal distribution where appropriate.

Questionnaires frequently produce a considerable amount of data, which may be interrelated, therefore multivariate analysis is important (White *et al.* 2005). I used multivariate ordered probit models and generalized linear models to test variables in combination.

Ordered Probit Model Analysis

Ordered probit model analysis has been shown to be useful for ecological studies where ordered categorical response variables are present (Guisan & Harrell 2000), for example for determining factors affecting attitude levels towards animals (e.g. Karlsson & Sjostrom 2007). For an introduction into the methods of ordered probit model analysis see Cameron & Trivedi, (1998) and McCullagh, (1980).

Following methods outlined in Karlsson & Sjostrom, (2007) and Cameron & Trivedi, (1998), the dependent variable y_i (perceived conflict level (0-4) or attitude towards elephant (1-5)) in my ordered probit models is discrete and ordered. It is assumed to be determined by the underlying continuous variable, y^*_i, and the cut off points, μ_i, given by the possible response alternatives.

$$y_i = \begin{cases} 0 \text{ if } y^*_i \leq \mu_1 \\ 1 \text{ if } \mu_1 \leq y^*_i \leq \mu_2 \\ 2 \text{ if } \mu_2 \leq y^*_i \leq \mu_3 \\ 3 \text{ if } \mu_3 \leq y^*_i \leq \mu_4 \\ 4 \text{ if } \mu_4 \leq y^*_i \end{cases}$$

Assuming that u_i is independent with a normal standard distribution, the probability of the individual i stating conflict level/attitude j (j=0,....,j=4) will be:

$$\text{Prob}(y = 0) = \Phi(-\beta' x)$$
$$\text{Prob}(y = 1) = \Phi(\mu_1-\beta' x) - \Phi(-\beta' x)$$
$$\text{Prob}(y = 2) = \Phi(\mu_2-\beta' x) - \Phi(\mu_1-\beta' x)$$
$$\text{Prob}(y = 3) = \Phi(\mu_3-\beta' x) - \Phi(\mu_2-\beta' x)$$
$$\text{Prob}(y = 4) = 1 - \Phi(\mu_4-\beta' x)$$

where Φ is the standard normal cumulative distribution. The parameters are estimated by Maximum Likelihood and should be interpreted with caution, as they do not express marginal effects of the independent variable. However, inference of the marginal effects of the regressor on the underlying variable is possible (Cameron & Trivedi 1998; Karlsson & Sjostrom 2007).

Predictor variables were removed successively from the model and the likelihood ratio (χ^2 statistic) calculated to determine whether the deviance increase by removing each predictor was significant (Guisan & Harrell 2000). The models were used to predict values of either perceived conflict level or feeling towards elephant and the difference between predicted and actual values calculated. Frequency distributions of this difference were drawn to illustrate the predictive capability of the model.

Generalized Linear Models (GLM)

Generalised Linear Models (GLMs), with binomial errors were used to explore a range of factors affecting respondent's attitude towards wildlife. Initially, univariate analyses were carried out with all explanatory variables. Only significant variables ($p<0.05$) were included in multivariate analysis. Multivariate GLMs with binomial errors were used to investigate correlates in combination. The maximum model for two-way interactions was fitted and simplified by stepwise deletion of non-significant terms. Model fit was checked using chi-squared goodness-of-fit test and significance determined for multivariate analyses at $p < 0.01$. Influential data points were left out one at a time and the model was refitted to check if parameter estimates or standard errors were substantially affected.

6.3. Results

A total of 909 structured interviews were conducted in 2010, over an 8 month period (January-August). Non-respondents were minimal, with only 3 farmers declining to be interviewed (because they were “too angry with the elephants to answer questions about them”), giving a 99.7% response rate. The mean interview time was 36.9 (± 12.5) minutes, with 2 main interviewers consistently carrying out the questionnaires and 14 different translators used to translate questions into Setswana or a local tribal language where necessary.

6.3.1. Demographics and background information of respondents

The majority of respondents were Bahambukushu (64%), which reflects the predominance of the Bahambukushu tribe in the ethnic make-up of the study area. Most respondents (71.2%) were placed in the poor or very poor wealth ranking, while 15% were classified as wealthy. Over half of the farmers interviewed relied solely on subsistence farming as a livelihood (56%), whereas only 7% were in fulltime employment. The median age of respondents was 48 years (range 18 – 92) and there was a higher number of female respondents (n = 565). This higher number of female respondents could be because most interviews were conducted during the day at fields. Traditionally the women tend to the fields during the day (weeding and scaring birds), thus making more women available to answer questionnaires.

6.3.2. Feeling towards elephants

Most respondents (56%) did not like living with wildlife around them, and even more had negative views towards elephants (82%). When asked to select a level of feeling towards elephants (1 = love, 2 = like, 3 = neutral, 4 = dislike, 5 = hate), the mean feeling was 4.2 (± 1.1) indicating a general dislike for elephants. Main reasons given for disliking elephants were because they crop-raid and are dangerous animals that can kill people. Those people that liked elephants or saw an advantage of living near elephants gave reasons such as they bring income, employment and tourism as well as that they “beautify the environment” and are “important for future generations”.

The ordered probit model for level of feeling towards elephants retained the variables gender and ethnicity of farmer, and attitude towards wildlife, as having statistically significant effects (see table 12). Interactions between attitude towards wildlife and gender, as well as attitude towards wildlife and ethnicity were also retained. Indicating that farmers who don’t like wildlife have a higher probability of disliking elephants and male farmers were more likely to like elephants. The ethnicity of a farmer affected the feeling towards elephants, with Basarwa and Baxereku farmers having a higher probability of disliking elephants, while Bakalaghadi and Bayei farmers had a higher probability of liking elephants. Farmers who were male and liked wildlife were even more likely to like elephants, and interestingly if Basarwa farmers liked wildlife they were then more likely to like elephants than farmers with other ethnicities. A respondent’s feeling towards elephants did not appear to be affected by the raiding history or location of their field, or whether they had an elephant totem or not.

Table 12. Predictors retained in ordered probit model for level of feeling towards elephant (FTE), their coefficient, standard error, t value and significance. P(χ): p value of the Likelihood Ratio - χ^2 statistic for testing whether the deviance change obtained by successively removing each predictor in the model is significant. The α_i is the intercept estimated to predict the different ordinal classes.

	Variable	Standard Coefficient	Standard Error	t value	χ^2	df	p value
FTE α_2: -2.51 α_3: -1.67 α_4: -1.33 α_5: -0.38							
	Ethnicity – Basarwa	0.71	0.20	3.50	13.0	4	0.01
	Ethnicity – Baxereku	0.29	0.21	1.36	-	-	-
	Ethnicity – Bayei	0.16	0.17	0.97	-	-	-
	Ethnicity – Bakalaghadi	0.05	0.50	0.10	-	-	-
	Gender – male	-0.13	0.12	-1.1	26.1	1	$3.3e^{-7}$
	Attitude towards wildlife – like	-0.32	0.12	-2.8	88.7	1	0
	Ethnicity–Bakalaghadi: like wildlife	-0.34	0.12	-2.8	17.7	4	0.001
	Ethnicity–Baxereku: like wildlife	-0.47	0.31	-1.5	-	-	-
	Ethnicity–Bayei: like wildlife	-0.74	0.22	-3.3	-	-	-
	Ethnicity–Basarwa: like wildlife	-0.78	0.27	-2.9	-	-	-
	Gender-male : like wildlife	-0.54	0.16	-3.3	10.9	1	0.001
NOBS	892						
-ln L	957						

The ordered probit model appeared to predominantly predict correct attitudes towards elephants (0 on histogram) or more negative feelings towards elephants than actually occurred (see figure 21).

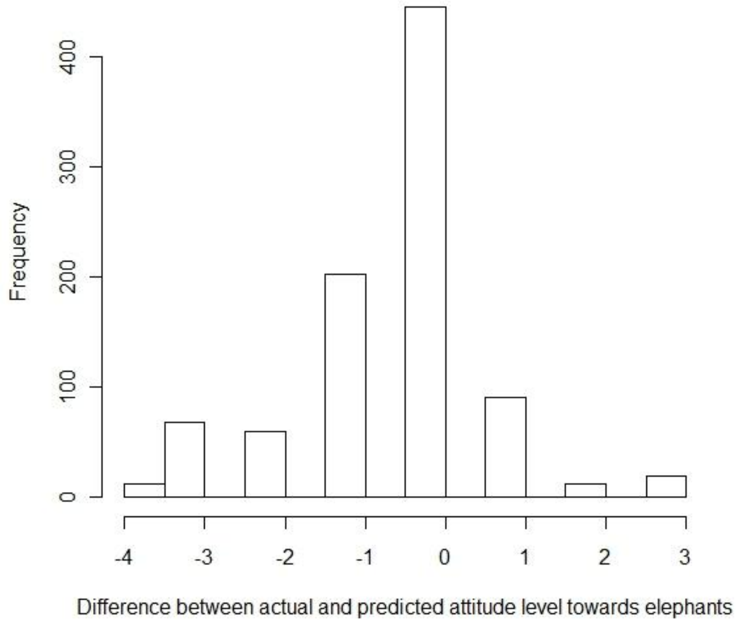


Figure 21. Frequency distribution of difference between actual values and model predictions of feelings towards elephants

Interestingly, many respondents did not eat elephant meat (38%), either because they are Christian and believe it is forbidden in the bible, or they were allergic, disliked the taste, forbidden under traditional beliefs (e.g. A traditional Bahambukushu belief states that elephant meat is made up of lots of other animal meat (including human) and therefore people should not eat elephants), or felt that elephant bodies are too similar to humans. Some farmers also had elephant as their totem (n=124), however less than half of these respondents believed they should not eat elephant meat (n = 50).

6.3.3. Attitude towards wildlife

The GLM, with binomial error structure, for attitude towards all wildlife retained respondent's sex ($\chi^2 = 9.1$, df = 1, $p < 0.01$) and whether the respondent's village benefits directly from tourism ($\chi^2 = 23.3$, df = 1, $p < 0.001$) as significant positive effects (male farmers and farmers living in villages directly benefitting from tourism are more likely to like all wildlife) (see appendix XIX for

GLM results). A respondent’s feeling towards elephant ($\chi^2 = 129.9$, df = 4, $p < 0.001$) was retained as a significant negative effect, with famers who expressed a neutral, dislike or hate attitude towards elephants more likely to dislike all wildlife. Education level ($\chi^2 = 15.7$, df = 4, $p < 0.01$) was retained as a significant effect, with farmers who had reached a non-formal education more likely to dislike wildlife and those who had reached junior secondary level more likely to like wildlife. Attitude towards wildlife also appeared to be affected in a complicated way by the interaction between age and education level ($\chi^2 = 13.3$, df = 4, $p < 0.01$), with older farmers who had had a non-formal education more likely to like wildlife.

6.3.4. Perception of conflict

A number of difficulties were identified by farmers with growing crops in the Okavango Delta panhandle, ranging from unpredictable rains, to domestic and wildlife crop raiders. Most respondents (63%) identified elephant raiding as being the biggest difficulty with farming in the area. When asked to identify the wildlife species causing them most problems with everyday life, 77% of respondents ranked elephants as the worst. Very few respondents felt they did not have a problem with elephants (n = 5). Most respondents (77%) who did have a problem with elephants ranked crop-raiding as their biggest problem, while 16% felt that elephants killing and injuring people was the biggest problem. Crop-raiding by elephants was found to be the most prevalent form of HEC in the study area during another part of this study independently monitoring HEC incidents (see chapter three).

Most respondents (n = 872) felt they were in conflict with elephants and when asked to grade the level of conflict they were experiencing (scale 0 = none to 4 = high), 70% felt it was high, with a mean conflict rate of 3.5 (± 0.99). Out of the 909 participants, 147 had never been raided by elephants (16%), while the median number of years a respondent had been raided in the past was 1 (range 0 – 70 years) and the median number of times raided this year was 0 (range 0-10 times).

Relationships between perceived conflict levels, and number of times raided this year ($F = 6.5$, df = 4, $p<0.001$), number of years previously raided ($F = 3.9$, df =4, $p<0.001$), and known number of people killed by elephants ($F = 3.9$, df = 4, $p<0.001$), were significant, but the area damaged in the most recent elephant raid was not. Levels of perceived conflict appear to be higher when certain indicators of actual levels of conflict are high.

Perceived conflict levels were significantly different to actual crop-raiding levels ($\chi^2 = 44.8$, df = 16, $p<0.001$), with perceived conflict levels generally higher than actual crop raiding levels (see figure 22)

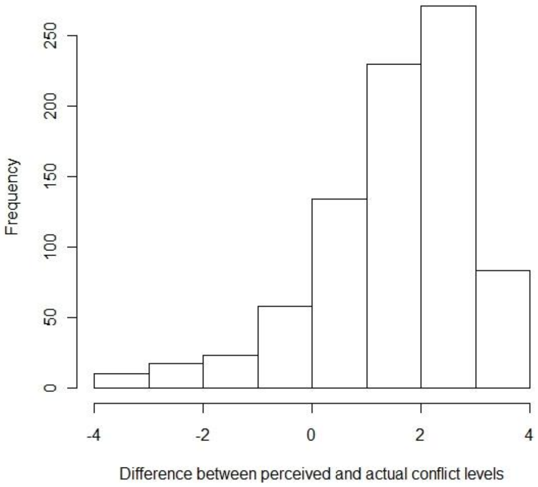


Figure 22. Frequency distribution of differences between perceived and actual conflict levels

Figure 23 illustrates the spatial distribution of differences between perceived conflict levels and actual crop-raiding levels. These results indicate that either, i) farmers are over inflating conflict levels or ii) that crop-raiding level ranks alone are not a good indicator of actual conflict levels.

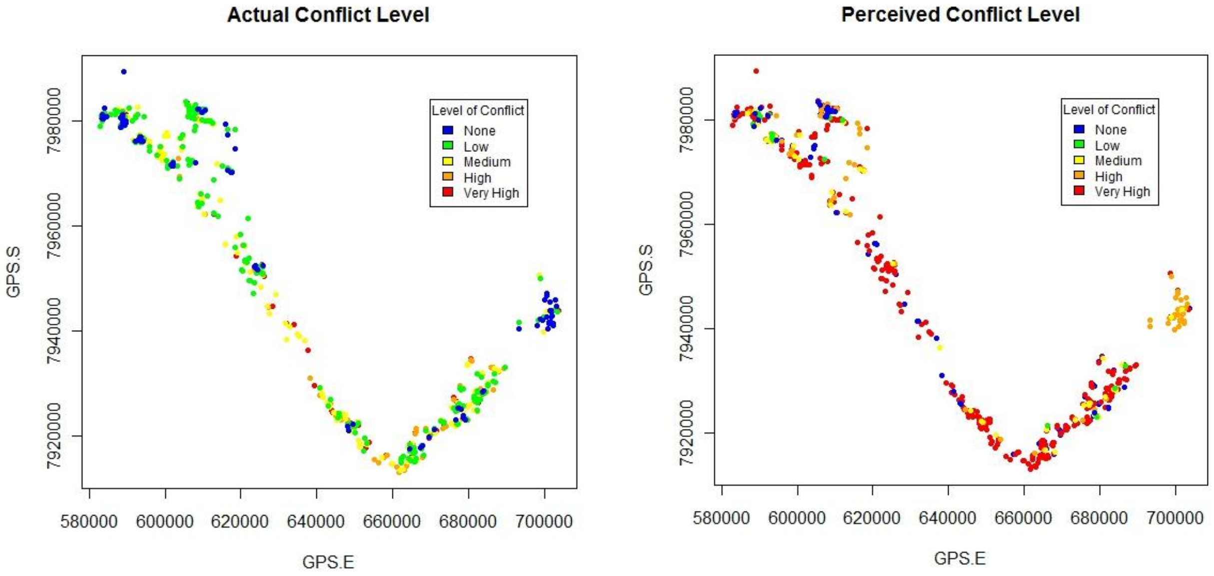


Figure 23. Variation between actual crop-raiding levels and perceived conflict levels per field in the Okavango Delta panhandle

The ordered probit model for perceived conflict level retained ethnicity of farmer, feeling towards elephants, number of years previously raided and distance of a field to an elephant pathway as having statistically significant effects (see table 13). Indicating that farmers who have a greater dislike for elephants have a higher probability of perceiving their level of conflict with elephants is high. The ethnicity of a farmer affected the perceived level of conflict, with Basarwa farmers having a higher probability of lower levels of perceived conflict. Farmers who had been raided more frequently in the past were more likely to have higher levels of perceived conflict and the location of a farmer's field appeared to influence their perceived conflict level, with farmers closer to elephant pathways likely to have higher perceived levels of conflict.

Table 13. Predictors retained in ordered probit model for perceived conflict level (PCL), their coefficient, standard error, t value and significance. P(χ): p value of the Likelihood Ratio - χ^2 statistic for testing whether the deviance change obtained by successively removing each predictor in the model is significant. The α_i is the intercept estimated to predict the different ordinal classes.

	Variable	Standard Coefficient	Standard Error	t value	χ^2	df	p value
PCL α_1: -0.90 α_2: -0.69 α_3: -0.26 α_4: 0.39							
	Feeling towards elephants – Hate	1.40	0.26	5.43	40.1	4	$4.1e^{-8}$
	Feeling towards elephants - Dislike	1.24	0.26	4.72	-	-	-
	Feeling towards elephants – Neutral	0.54	0.30	1.83	-	-	-
	Feeling towards elephants – Like	0.67	0.29	2.30	-	-	-
	Ethnicity – Baxereku	0.56	0.22	2.53	21.2	4	0.0003
	Ethnicity – Bayei	0.39	0.15	2.60	-	-	-
	Ethnicity – Bakalaghadi	0.27	0.50	0.54	-	-	-
	Ethnicity – Basarwa	-0.69	0.13	-5.24	-	-	-
	No. Years Raided	-0.02	0.01	-1.57	3.8	1	0.05
	sqrt(Distance to elephant pathway)	-0.007	0.003	-2.54	4.7	1	0.03
NOBS	735						
-ln L	651						

The ordered probit model appeared to predominantly predict correct perceived conflict levels (0 on histogram) or higher levels of perceived conflict levels than actually occurred (see figure 24).

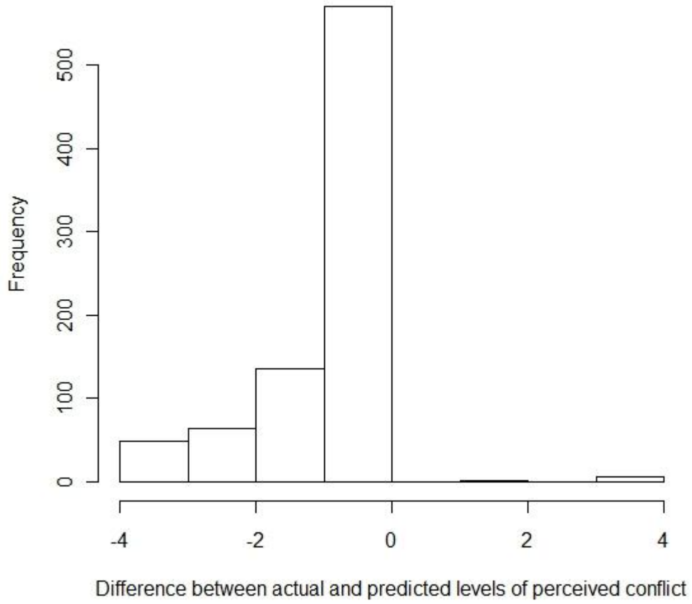


Figure 24. Frequency distribution of difference between actual values and model predictions of perceived conflict levels

In summary, it is evident that a respondent's perceived conflict level is influenced in a complicated way by a combination of factors both directly and indirectly. To illustrate the relationship between factors affecting perceived conflict levels in the Okavango Delta panhandle, a path diagram was constructed (see figure 25).

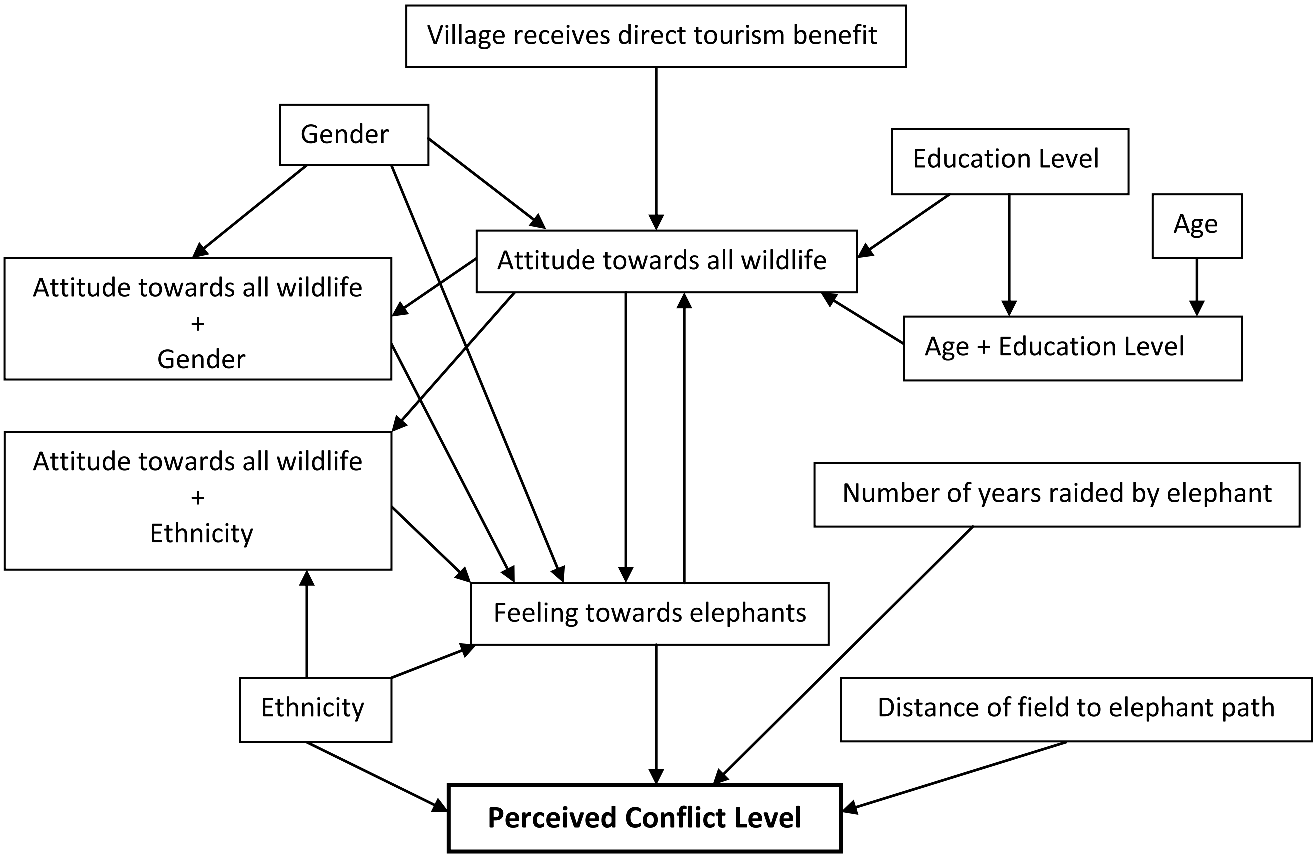


Figure 25. Path diagram explaining relationship between factors affecting farmer perception of human-elephant conflict in the Okavango Delta panhandle

6.3.5. Coping strategies

The main purpose of growing crops in the study area was for subsistence (n = 813), with only 10% of farmers selling some crops for cash alongside crops for personal consumption and no farmers interviewed growing crops commercially. The median number of people a field supports (dependants) was 7, ranging from 1-34, and the number of dependants was significantly higher than the number of people in the respondents immediate family (t = 3.09, df = 1615, p<0.01). The mean estimated harvest was 499 (±486) kg per field, with the mean quantity of cereal crop required for a year per respondent being 444 (±382) kg. In bad harvests the estimated mean amount of crop produced per respondent was 213 (±199.5) kg, which is less than half the estimated amount needed to support dependants for a year. This indicates that on average farmers may not be able to produce enough food to support the number of people dependant on a field during bad harvests and are therefore vulnerable to the effects of elephant crop-raiding which was most frequently attributed as the main cause of bad harvests (63% of respondents).

Various coping strategies were identified by farmers for dealing with bad harvest years (see figure 26). Many respondents (26.4%) indicated that the only coping strategy available to them was to eat less (budget) to ensure there is food for the year. Other popular coping strategies included establishing small businesses (e.g. selling baskets) (23.2%) or looking for piece jobs (18.3%) to be able to buy extra food.

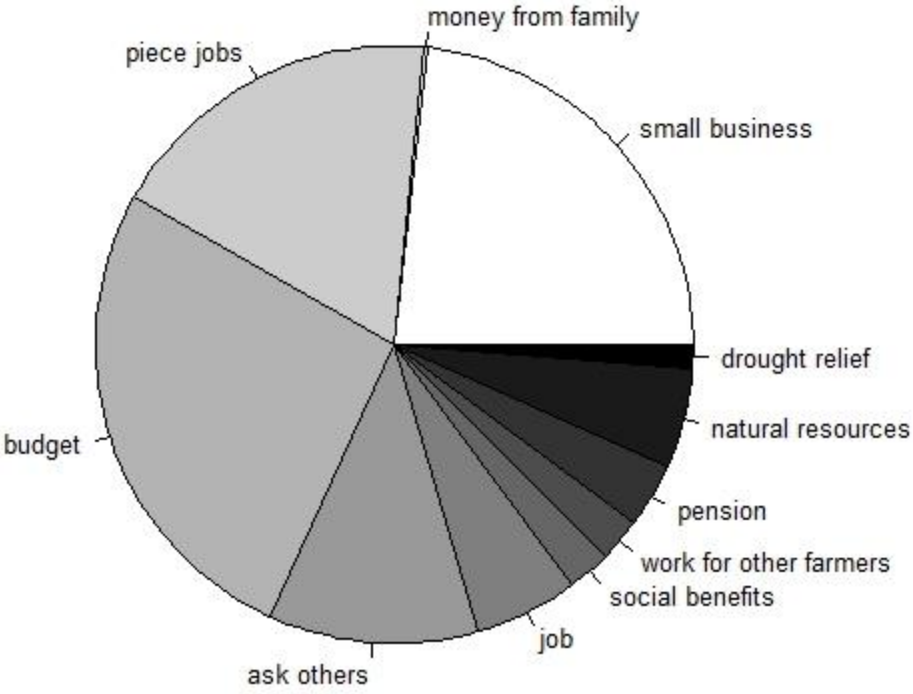


Figure 26. Coping strategies of farmers during bad harvests

When respondents were asked “Do you think it is possible to reduce the problem with elephants here?” under half (n = 415) replied yes, while 30% replied no and the remaining 24% were unsure. The GLM, with binomial error structure, for whether people thought there is a possible solution to reduce the problem with elephants or not (coded 1 or 0) retained the variables respondent’s education level (χ^2 = 30.4, df = 4, p<0.001) and whether the respondent thought elephant movement paths should be considered in land use planning (χ^2 = 6.3, df = 1, p=0.01) as having significant positive effects (respondents were more likely to think there is a solution if they had been educated either non-formally, to primary level, or to junior secondary level, or if they thought elephant paths should be considered in land use plans). The amount of damage a farmer had incurred from a recent (within the last 3 years) elephant raid (χ^2 = 7.1, df = 1, $p<0.01$) was retained as having a significant negative effect (farmers who had incurred more damage were less likely to think there was a possible solution). The opinion of respondents towards whether there was a possible solution to reduce the current problem with elephants was also affected by their preferred elephant management strategy (χ^2 = 34.8, df = 9, $p<0.001$), with farmers favouring improving mitigation for management more likely to believe there was a solution and farmers favouring leaving elephants alone less likely to think there is a solution (see appendix XX for detailed results table).

The top four management strategies to reduce the problems people are experiencing with elephants were to kill some elephants (47%), move elephants away (20%), kill all elephants (16%) and mitigate crop damage (10%). Significantly more respondents believed that the government (rather than farmers) were responsible for reducing elephant crop-raiding (χ^2 = 781.3, df = 1, $p<0.001$), however, most people felt that farmers should be responsible for protecting their own fields from elephant raids (χ^2 = 267.7, df = 1, $p<0.001$). When asked what the government's priority should be to reduce human-elephant conflict, there was no significant difference between the following three responses: i) increase compensation; ii) improve mitigation; and iii) reduce the elephant population. When asked to choose which development issue should be the priority to address in the area, reducing the elephant problem was selected most frequently (χ^2 = 28.2, df = 3, $p<0.001$), indicating that human-elephant conflict is a priority development issue for subsistence farmers in the Okavango Delta panhandle.

6.4. Discussion

Most subsistence farmers in the Okavango Delta panhandle believe they are experiencing high levels of conflict with elephants and consider such negative interactions to be a major factor in prohibiting development in the area. Crop-raiding by elephants was reported as the predominant problem and could therefore, potentially, be a good indicator of actual human-elephant conflict intensity. Yet, when I established a ranked conflict level based on actual crop raiding history of a respondent, the perceived conflict levels of respondents were higher than our predicted actual conflict levels. Such discrepancies between actual and perceived conflict levels have been highlighted in previous studies (e.g. Gillingham & Lee 2003; Linkie 2007; Hazzah, Mulder & Frank 2009), and have sometimes been interpreted as “people over-reporting the scale of the problem” (Gillingham & Lee 2003). However, another explanation for such a disparity, as indicated by my results, is that the crop-raiding history of a farmer alone is not adequate to assess the overall level of conflict being experienced, because other social and environmental factors are also involved in determining farmer perception of conflict (Dickman 2010). By using a combination of multivariate ordered probit models and generalized linear models I have been able to evaluate the complexities of farmer perception and attitudes, revealing underlying influential factors which shape perceived levels of conflict.

6.4.1. Complexities of perceived human-elephant conflict

Farmers’ perceptions of negative interactions with elephants appear to be shaped by more than their past negative experiences with elephants. Although perceived conflict levels were generally higher when indicators of actual conflict increased, only the number of years a farmer’s field had been raided in the past remained significant in the ordered probit model explaining perceived conflict levels. Evidently, people’s perception of the intensity of negative interactions is influenced by an array of factors, both directly and indirectly, many of which are socio-demographic characteristics of the respondent, rather than quantifiable indicators of actual conflict. In our study, the direct key drivers of subsistence farmers perceived conflict levels with elephants included ethnicity of respondent, attitudes towards elephants, number of years raided by elephant in the past, and the location of a field in relation to main elephant pathways. Indirectly, perception of conflict was influenced by factors affecting feelings towards elephants, such as ethnicity and gender of respondents, as well as attitudes towards wildlife.

This in turn was affected by factors shaping attitudes towards wildlife, including feelings towards elephants, gender and education level of respondents and whether or not a respondent's village benefitted directly from tourism.

Clearly peoples' perception of the conflict they are experiencing with elephants is complex. As Dickman (2010) explains, perceptions and attitudes are formed from personal as well as wider societal experiences, facts, cultural norms, expectations and beliefs. If people with low levels of actual quantifiable damage from elephants still feel that they are in high conflict with elephants, then it seems that indirect experiences with elephants and hyper-awareness of risk could also be influencing perceptions. For example, in a community where one person sustains damage from elephants, other people may become fearful of such damage even if they have never had a personal experience of it (Dickman 2010). Indeed, it has been found that attitudes towards potentially dangerous wildlife species (e.g. wolves) are shaped by such indirect experiences including reports in the media (e.g. Karlsson & Sjostrom 2007), and therefore perceptions of conflict with wildlife species such as elephant are likely to be affected. People with different ethnicity, gender, or education levels are likely to have had different types of direct and indirect experiences with elephants, which could explain the importance of such socio-demographic factors in shaping perception of conflict.

6.4.2. Factors driving attitudes and perceptions

Indirect experiences with elephants may also explain the complicated interaction between gender and attitude towards wildlife in shaping feelings towards elephants. Male respondents in the Okavango panhandle appeared to be more likely to like wildlife and elephants than women, a trend that was also observed in Uganda (Hill 1998), and men who liked wildlife were more likely to like elephants than men who didn't. Attitudinal differences between and within genders may be attributed to differences in daily activities of respondents in these study areas. For example, women in the Okavango Delta are often exposed to and disturbed by wildlife (including elephants) during daily activities such as crop protection, gathering of firewood, fetching water and collecting thatching grass, while some men who are responsible for activities such as cattle herding and fishing may also encounter elephants regularly. Frequent exposure to elephants may incite more negative feelings towards them as Williams, Ericsson & Heberlein (2002) surmised with wolves. Fear could also influence attitude towards elephants (Kaltenborn, Bjerke & Nyahongo 2006), indicating that women may be more afraid of elephants than men in the panhandle.

The ethnicity of respondents appeared to affect both feelings towards elephants and perceived conflict levels, with Bayei and Basarwa farmers more likely to have lower perceptions of conflict and Bayei more likely to like elephants, than the other tribes. Such differences between ethnic groups could be due to cultural traditions and beliefs towards elephants (Knight 2000; Dickman 2010). For example, a traditional Bayei belief states that elephants descended from people (Botumile 2006) and some Bayei view elephants as “our parents”. Elephants are therefore respected by many Bayei, which may enhance positive feelings towards them and hence reduce severity of perceived conflict. Basarwa people are traditionally hunter gatherers (Vanderpost 2003), who began cultivating crops a lot more recently than other tribes in the area (Tlou & Campbell 1997). Basarwa people therefore relied heavily on natural resources for their livelihoods, thereby increasing the importance of wildlife, whilst reducing the reliance on crops, for food. Such traditional values could enhance positive feelings towards wildlife and reduce the impact of losses incurred from wildlife/elephant crop-raiding due to available alternative food resources.

The level of education of a respondent appeared to affect attitude towards wildlife, with more educated people more likely to like wildlife in the study area. Although level of education did not directly affect peoples’ feelings towards elephants or perceptions of conflict levels in my study, it has been shown to be an important factor in increasing tolerance to wildlife damage in other studies (e.g. Marker, Mills & Macdonald 2003; Holmern, Nyahongo & Roskaft 2007). More positive attitudes towards wildlife appear to increase positive feelings towards elephants which in turn reduce the level of perceived conflict a respondent is experiencing. Therefore, by raising education levels and improving attitudes towards wildlife, managers may ultimately reduce negative feelings towards elephants and reduce conflict in this area.

It has been shown in some studies that distance of a respondent’s residential location to an animal’s habitat can affect attitudes towards that animal (e.g. Karlsson & Sjostrom 2007), however, this did not appear to be the case with the location of a respondent’s field. However, I did find that the distance of a respondent’s field to an elephant pathway affected their perceived level of conflict, with higher perceived conflict levels more likely when fields were closer to elephant pathways. In fact, in chapter four, I showed that the closer a field is to an elephant pathway the more likely it is to be raided by elephant. My survey results therefore suggest that farmers in the panhandle are fully aware of the increased risk of raiding when crops are planted close to elephant movement paths. Interestingly, however, most farmers did not agree that such paths should be considered in future land use plans ($\chi^2 = 12.9$, df =1, $p<0.001$), most would not be willing to move the location of their current field ($\chi^2 = 33.6$, df =1, $p<0.001$) and some people are still choosing to plant crops close to well known pathways in this area (*personal observation*). This has implications for future management strategies, which involve changes in land use planning to reduce human-elephant conflict in this area. If, farmers are aware of the risks of planting crops near elephant paths but reluctant to change field locations or consider such paths when choosing new fields, then changes in land use planning policy which restricts fields from occurring near such movement paths may not be accepted by the majority of the farming community. If mitigation strategies are not accepted by affected communities, then perceived levels of conflict may be heightened rather than reduced and interventions may be regarded as ineffective or inappropriate (Hill 2004).

A common strategy to decrease the costs involved with living close to wildlife is to increase the benefits local people receive from living close to protected areas or wildlife (Thouless 1994; Newmark & Hough 2000; Johannesen & Skonhoft 2005; Walpole & Thouless 2005). In the eastern Okavango panhandle, the Okavango Community Trust has been established to administer funds received from two wildlife management concessions leased to tourism operators. Five villages benefit from this income through different development projects, while the seven remaining villages in the panhandle receive no direct benefits from tourism or wildlife related enterprises. All the communities, however, still experience problems with living close to wildlife (especially elephant). In addition, the Government of Botswana administers a state funded compensation scheme to minimise the economic losses of elephant crop damage in the area, which may help to increase tolerance of the problem (Macdonald & Sillero-Zubiri 2004). This situation provided an opportunity to investigate too important issues: i) the role of tourism-based benefits and; ii) the role of government compensation schemes in affecting attitudes towards wildlife and elephants, and perceptions of human-elephant conflict.

Evidently, attitudes towards wildlife improved when a respondent’s village received direct benefits from tourism, which reflects a trend observed in other studies investigating attitudes towards wildlife and conservation (e.g. Infield 1988; Archabald & Naughton-Treves 2001), yet feelings towards elephants or perceptions of conflict were not significantly affected. An explanation for this later result could be that the costs incurred from living with elephants accrue to individuals, whereas the benefits received through tourism is received on a community basis, therefore benefits may not offset individual costs of living close to elephants (Walpole & Thouless 2005) and consequently has little effect on improving perceptions of conflict. Although there was no apparent direct effect of tourism benefits on perceived conflict levels, they could be affected indirectly through influencing attitudes towards wildlife, which affects feelings towards elephants, which influences perceived conflict levels. This indicates that tourism based community incentive programmes, can indeed improve attitudes towards wildlife and may consequently in the long term reduce people's perceived level of conflict with certain problem wildlife species such as elephants. Whether or not people received compensation, however, did not appear to influence attitudes or perceptions of farmers in the panhandle.

6.4.3. Coping strategies

Where human-wildlife conflict (HWC) occurs in areas with low income or where people are dependent upon a single livelihood strategy, potential consequences of HWC incidents are intensified by a lack of alternative assets or income strategies, which can exacerbate the antagonism of people towards conflict causing animals (Naughton-Treves & Treves 2005; Dickman 2010). If a person is wealthy, has alternative sources of income and/or engages in social reciprocity with their family and community, then it is surmised that they could be less vulnerable than other people (Naughton-Treves & Treves 2005).

Aspects of a farmer's cultural and socio-economic status can, therefore, affect if, and how, crop-raiding impinges on their livelihood (Goldman 1986; Naughton-Treves 1998; Sekhar 1998; Hill 2000; Linkie 2007), and therefore affect tolerance towards such negative interactions with wildlife. In the Okavango panhandle, it is apparent that most subsistence farmers are highly vulnerable to the effects of elephant crop-raiding due to low crop yields and limited coping strategies available to them to survive further crop loss. The level of conflict people feel they are experiencing with elephants is likely to be intensified in such circumstances, which could help account for the perceived high conflict levels and negative feelings towards elephants that we recorded. It is apparent, therefore that such socio-economic issues are indeed an important factor to address in conflict resolution programmes, as has been suggested in earlier research (Hill 2004; Madden 2004; Naughton-Treves & Treves 2005; Osborn & Hill 2005; Dickman 2010).

It is encouraging that many people in the panhandle felt there was a possibility to reduce the problems they are experiencing with elephant, but, unfortunately the preferred management strategies (to kill some elephants or move them away) of the community are not currently favoured by the Government of Botswana in their Elephant Management Plan (MEWT 2000). Such differences in opinions could lead to feelings of antagonism in the community towards both government and wildlife, especially if people feel their needs are not being addressed or judgments not duly considered (Madden 2004). Evidently, an environmental education awareness campaign could play an important role in improving farmer perception, as respondents with higher education levels tended to have more positive opinions about finding a solution. To avoid unrealistic expectations of farmers, part of such a campaign could focus on ensuring people have a realistic vision of what can be achieved through certain conflict mitigation interventions (Osborn & Hill 2005), as well as explaining which strategies are favoured by the government and why.

Understandably, respondents who had experienced large amounts of damage from elephant crop-raiding recently were more inclined to feel there was no solution to the elephant problem, highlighting the need to try and reduce amounts of actual damage in order to improve farmer perception of the problem (Hill 2000). Farmers who indeed favoured mitigating damage as an elephant management strategy were more likely to believe there is a solution to the elephant problem, whereas those who favoured leaving elephants alone were more likely to have a negative opinion. Although most respondents felt that the Government of Botswana (including the President, Department of Wildlife and National Parks, and village Chiefs) is responsible for reducing the level of crop-raiding by elephants, most farmers did feel that they had the responsibility of protecting their own fields. This shows that management strategies which empower farmers to protect their own fields (e.g. Osborn & Parker 2003; Osborn & Hill 2005; Parker & Osborn 2006) should theoretically be accepted by the community in this area. However, there may still be a danger of resistance if such strategies are enforced by the government, because many people believe the government is ultimately responsible for reducing damage. It is, therefore, advised that such mitigation strategies be introduced and encouraged by a third party to avoid misconceptions over responsibilities.

6.4.4. Management implications

Such findings have important implications in terms of managing a situation where negative interactions arise between people and wildlife. Ultimately the level of human-elephant conflict appears to be a combination of both actual elephant damage and perceived conflict (Dickman 2010), therefore mitigation efforts need to address both aspects. If loss (actual and perceived) is matched by benefit (actual and perceived) then overall conflict can be reduced. Actual wildlife damage can be reduced through mitigation initiatives; however, improving perceived conflict levels is more difficult because they are so complex. Perceived conflict is affected by a myriad of factors including socio-demographic characteristics, cultural beliefs, socio-economic circumstances of farmers, location of a field, crop-raiding history of a field, indirect and direct experiences with elephants, and acceptability of management options, therefore investigations into perceptions of conflict are likely to be needed on a case by case basis in order to effectively manage human-wildlife conflict issues.

By gaining a greater understanding of the factors that shape attitudes towards wildlife and perceptions of conflict in the Okavango Delta panhandle, I can start to identify groups of farmers to concentrate different management strategies on. For example, I found that women were more likely to dislike wildlife and elephants, and people from Bahambukushu, Bakalaghadi and Baxereku tribes were more likely to have higher perceived conflict levels and dislike elephants than other tribes. Therefore, education programmes to improve attitudes could initially target these social groups in order to reduce levels of perceived conflict.

My study suggests that such farmers who have a high perceived level of conflict are also likely to have negative feelings towards elephant which may influence their adoption of mitigation techniques. People who hate elephants may be less willing to adopt passive mitigation techniques to reduce conflict. If one endeavours to introduce passive mitigation strategies such as chilli methods to deter elephants in this area a different management approach may, therefore, be needed, which starts by improving attitudes towards elephants through an education campaign before introducing the idea of passive mitigation methods to reduce conflict levels. As farmers feel they are responsible to protect their own fields in this area, mitigation strategies which empower farmers to do so should also be a priority.

Evidently, one needs to assess what management strategies will appease community perceptions and improve negative attitudes, yet remain a viable and acceptable strategy to the government. Many farmers advocate killing some elephants to reduce the problems they are facing, however, hunting of elephant is currently prohibited in the study area (wildlife management areas NG11, NG12 and NG13), apart from the use of lethal control to remove problem animals. Although killing elephants may appease some farmers and reduce levels of perceived conflict on an individual level. It is clear that many people in the community do not actually eat elephant meat. Therefore, direct benefits from such management strategies i.e. protein from meat, will not be shared equally within the community. When benefits are received on a community level (i.e. from tourism), however, attitudes towards wildlife appeared to improve and perceived levels of conflict can then potentially be reduced on a community level. Increasing tourism or wildlife related enterprises in the Okavango panhandle or finding management strategies which provide benefits to the community are therefore likely to be most effective in reducing perceived conflict levels here.

Chapter 7

Reducing competition between people and elephants

7.1. Complexities of human-wildlife competition

Competitive interactions between humans and wildlife are complex, arising from an array of interlinked contributing factors. Such complexities make assessments of trends, identification and quantification of key drivers of competition and development of appropriate management strategies challenging. Negative interactions can often arise as a result of such competition and are commonly defined as human-wildlife conflict (HWC). My study has identified contributing factors of human-elephant conflict (HEC) in the eastern Okavango Delta panhandle, Botswana through a multifaceted approach of data collection and analysis, giving greater insight into the dynamics of their interconnected driving forces and highlighting the complex nature of competition. This has facilitated a better understanding of how contributors of competition can be addressed to develop effective management strategies for reducing negative interactions between people and wildlife.

This study shows that combinations of social and ecological factors are involved in shaping competition (see figure 27). Social factors include: actual and perceived conflict levels; farmer vulnerability to risk and available coping strategies; methods used to measure damage; and human feelings and perception towards elephants and the situation, which are influenced by an array of socio-economic factors. While ecological factors include: natural and modified behaviour of people and elephants affecting resource and spatial use as well as how each species reacts to living in close proximity to each other; and the vulnerability of crops to elephant foraging, which affects both actual and perceived conflict levels. Boundary solutions to stop competition and conflict between humans and wildlife would inevitably involve killing all wildlife or prohibiting people from growing crops or encroaching on wildlife habitats. However, it is assumed that such extreme measures are likely to be unacceptable. Clear cut win-win solutions for reducing such competition and conflict are likely to be difficult to identify because a) there are complex interactions between contributing factors and b) both people and wildlife will have to experience an element of discomfort when adapting to living in close proximity to each other. To be successful, effective conflict resolution and management strategies will require consideration of short and long term dynamics, as well as a combination of mitigation approaches that consider all elements affecting conflict extent.

The composite structure of underlying processes driving human-elephant conflict also makes the identification of simple trends difficult, e.g. whether or not competition is increasing. In the eastern Okavango Delta panhandle, numerous types of conflict occur between people and elephants. Elephants damage crops, break fences, damage property and chase, injure and sometimes kill livestock and people. At the same time, people are modifying elephant habitat through developments such as cultivated land and settlements, and chase, injure and sometimes kill elephants. Monitoring elephant crop-raiding incidents over the past three years revealed that crop damage frequency and intensity fluctuated annually, with the highest levels recorded in 2008, a decrease in 2009 and then an increase in 2010. Crop-raiding patterns differed spatially as well, with some villages experiencing higher raiding levels and certain fields were more vulnerable to raiding than others. Nevertheless, there was no quantifiable evidence to suggest that crop damage is increasing. Indeed, my results showed that the intensity of crop-raiding in the panhandle appears to be lower than intensities recorded in other countries in the region. These findings suggest that people and elephants may be coexisting in relative harmony despite both human and elephant population increases.

Contrary to these findings, however, my results from an examination of rural farmer perceptions, suggest that problems with elephants are increasing and HEC is getting worse in the Okavango Delta panhandle. According to my analysis, most people believe they are experiencing high levels of conflict with elephants and consider such negative interactions to be a major factor in prohibiting development in the area. People may have reported an inflated concern for HEC issues, because they were aware that my research was focusing on such issues. However, I spent a year working in the communities before interviewing people, to improve mine and my interviewee’s understanding of each other’s motivations (Milner-Gulland & Rowcliffe 2007), which hopefully reduced bias in people’s opinions. Evidently, quantifying wildlife crop damage provides information on the physical symptoms of HEC, but not the full extent of the problem, which includes the additional indirect impacts on wildlife perception among the communities. Consequently, damage incidents are not a true representation of the full extent of competition and cannot be treated in isolation. While measurements of wildlife damage are necessary to provide information for developing crop/field vulnerability models and devising short-term mitigation programmes to minimise damage intensity, it is also crucial to consider other contributory factors that exacerbate conflict, such as human perceptions and elephant behaviour, if long-term management strategies are to be successful.

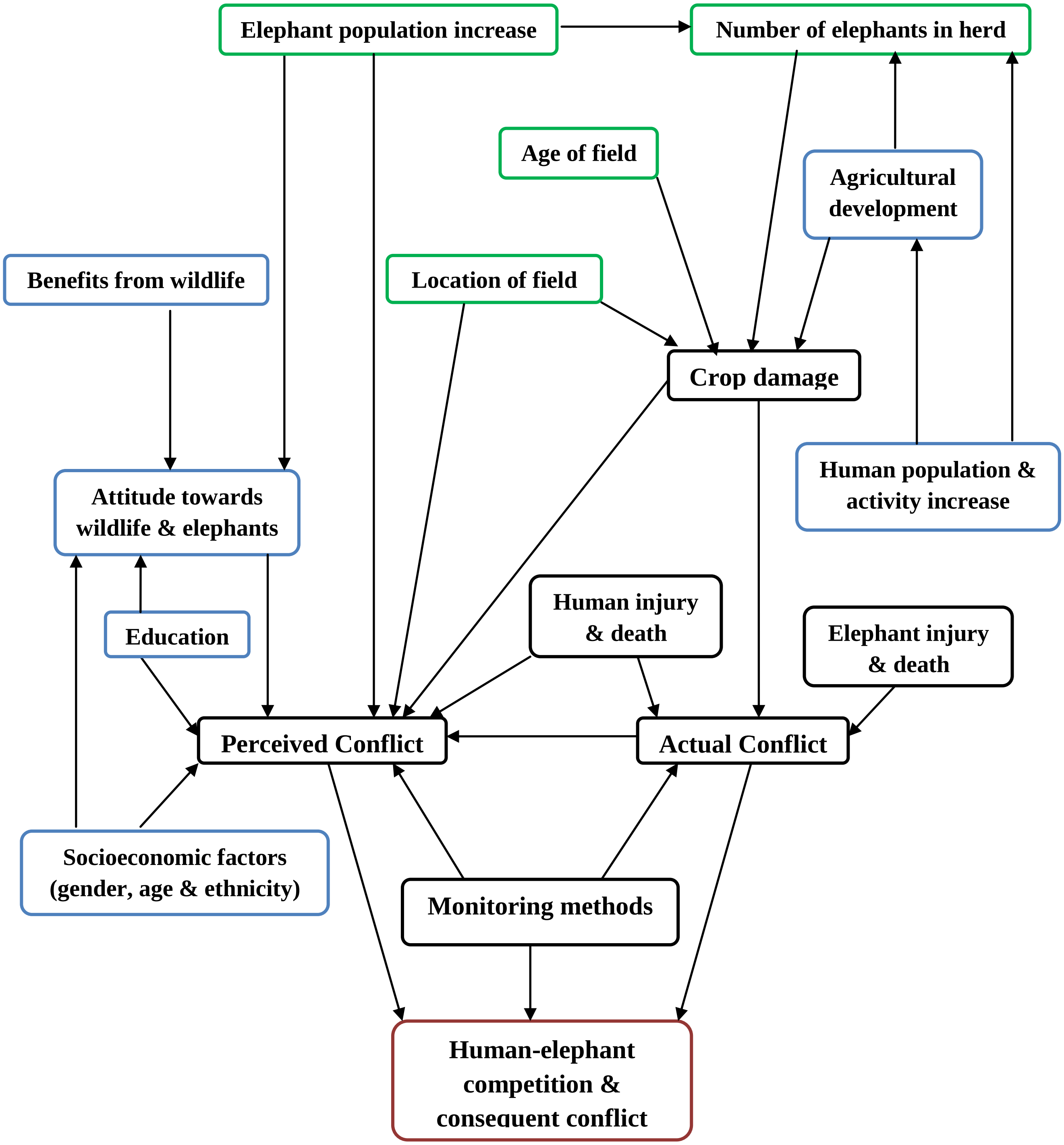


Figure 27. Path diagram illustrating interacting environmental (green) and social (blue) factors shaping and contributing to the complexities of competition and conflict between elephants and people.

7.2. Patterns, underlying processes and management strategies

The findings of this study identified a number of ecological and social patterns and underlying processes, as well as aspects of recording and quantification methodology that, with due consideration to their impact on HEC could aid in the identification of appropriate management strategies that will ultimately reduce competition between humans and elephants.

When choosing a monitoring programme for human-wildlife conflict situations, it is imperative to clearly define the purpose of data collection and, therefore, what information to collect. Aspects such as data quality, temporal efficiency, economic implications and social elements also need to be considered to ensure adequate data is collected for consequent management activities. HWC monitoring strategies that address broader issues beyond providing a record of damage incidents are likely to have a greater impact in reducing HWC in the long-term (see chapter three). In this respect the IUCN protocol has some major advantages over a government (top-down) approach that is driven by the need to record damage incidents for compensation purposes. In addition, the IUCN approach involves community enumerators, which provides employment and empowers community members in monitoring HWC incidents, thereby encouraging participation and ownership in such schemes whilst providing direct benefits to the community. An increase in benefits from wildlife can aid in increasing tolerance towards wildlife and reduce perceived conflict levels. Many people interviewed in my study expressed enthusiasm for the involvement of community enumerators in monitoring HEC incidents and appreciated their support with training and implementing mitigation techniques. In addition, I found that community enumerators facilitated information exchange between farmers and researchers or Department of Wildlife and National Parks (DWNP) officers, which can facilitate conflict resolution by improving communication between stakeholders. Adopting these aspects of the IUCN approach to HEC data collection could, therefore, indirectly assist in reducing the intensity of conflict between humans and wildlife whilst simultaneously increasing acceptance of mitigation strategies.

Peoples' perception of the conflict they are experiencing with elephants/wildlife is a major factor in determining the extent of competition occurring and therefore needs to be considered when devising wildlife conflict management strategies. However, it is also complicated, influenced by an array of factors, both directly and indirectly, many of which are socio-demographic characteristics of the farmer, rather than quantifiable indicators of actual conflict. For example, my study (see chapter six) found that perceived human-elephant conflict in the Okavango Delta panhandle is affected by the ethnicity of a farmer, raiding history of a field, distance of a field to an elephant pathway, and a farmers feeling towards elephants (which is affected by attitudes towards wildlife, as well as gender and ethnicity of the farmer). With a greater understanding of factors shaping attitudes towards wildlife and perceptions of conflict it is possible to identify target groups of farmers to concentrate different management strategies on. In the Okavango Delta panhandle focused education and awareness campaigns with women and farmers from certain tribes could improve attitudes towards elephants and, thereby, reduce perceived levels of conflict. Improving attitudes towards elephants is essential if passive mitigation strategies such as chilli deterrent methods are to be successful at reducing HEC in this area, because people who dislike elephants may be less willing to adopt such passive mitigation techniques.

My analysis in chapter six, also found that feelings towards elephants are affected by attitudes towards wildlife which are improved by benefits accrued to the community through tourism and other wildlife related enterprises. In the eastern Okavango Delta panhandle there is currently little direct benefit to many communities from elephants, apart from protein for some people when animals are killed through problem animal control activities. An appropriate management intervention, therefore, would be to try and improve this situation, by increasing benefits through tourism in those villages that currently receive no direct benefits. Responsibility for wildlife and land is another big factor to consider when devising HWC management strategies. My study found that many people in the Okavango Delta panhandle felt that the Government of Botswana is responsible for reducing the problems they are experiencing with elephants (i.e. level of elephant crop damage), because the government own elephants. Increasing community ownership of wildlife resources could also, therefore, facilitate improving perceived conflict. Strategies such as increasing community hunting quotas to allow sustainable use of elephants could increase direct benefits whilst raising community responsibility for their wildlife resources, which may aid in reducing HEC. On the other hand, most farmers feel they are responsible for protecting their own fields in my study area; therefore, community based conflict management strategies which empower farmers to protect their own fields are likely to be accepted by the community and should be a priority. Community based insurance schemes, like the Human Animal Conflict Insurance Scheme (HACIS) in Namibia (Stewart & Diggle 2004), that encourage community members to take part/full ownership of a compensation scheme could also be an effective management strategy to reduce perceived conflict levels in the Okavango Delta panhandle. Such a strategy would increase community responsibility (thereby decreasing perceived government responsibility) for monitoring and managing HEC incidents, which could improve community perception of both HEC itself and management strategies involving compensation. However, it would require increased wildlife generated revenue to supplement government money for insurance/compensation pay outs. Increased wildlife tourism enterprises may, therefore, also be required for such strategies to be sustainable.

As well as targeting certain social groups of farmers in management strategies, it is also possible to design crop damage mitigation approaches based on field location or vulnerability to elephant/wildlife raiding. My results in chapter four indicate that fields that have a long history of elephant crop raiding or those situated close to elephant pathways in the Okavango Delta panhandle are more vulnerable to elephant damage in the future. Mitigation efforts should, therefore, focus on such high risk fields. Similar local spatial patterns have been found in previous studies where crop vulnerability models have been developed (e.g. Sitati, Walpole & Leader-Williams 2005). It still appears, however, to be difficult to identify common predictors of crop-raiding across larger spatial scales. Such apparent unpredictability has been attributed to local variations in elephant ecology and movements, and human spatial use. My study suggests that the spatial distribution of fields may also be a major contributing factor. Spatial autocorrelation in wildlife crop-raiding data can be problematic, sometimes leading to spuriously significant results. However, when data are sub sampled to remove the effect of spatial pseudoreplication important biological information may be lost, because both significance levels (p values) and model estimates are affected by such autocorrelation. Both estimates and p values should, therefore, be reported across a range of spatial scales. From such analysis, it became evident that the spatial distribution of fields should also be considered when choosing the spatial extent of analyses for wildlife crop-raiding studies because different spatial distributions of fields and explanatory variables are affected in different ways through sub-sampling to reduce spatial autocorrelation in the data. Consequently, it would be advantageous to assess HEC incidents at a village level where clumped field distributions could be separated from dispersed distributions. Such localised research would facilitate the development of relevant crop vulnerability models and enable effective management strategies to be designed at a local level.

Elephant movements are strongly influenced by the presence of human habitat modifications and activities in the Okavango Delta panhandle, which indicates there is potential for minimising overlap between human and elephant resource use through land use planning interventions. Analysis of localised elephant movements in the study area (see chapter five) indicate that elephants use certain pathways more frequently than others and appear to avoid areas where human settlements are larger or where there is more development (i.e. cultivated land) and associated activity by humans. This highlights the possibility for micro level land use planning to be developed (Taylor 1993; Fernando *et al.* 2005; Linnell *et al.* 2005). To facilitate accurate land use planning; appropriate field allocation zoning, these elephant movement behaviour characteristics should be combined with information from crop/field vulnerability models. As a result, agricultural use zones could be designated in such a manner as to avoid frequently used elephant paths as well as high-risk fields. Alternatively, elephant pathways that are frequently used should be allocated a free movement buffer zone along their route, where arable land allocation should be prohibited, thereby reducing the likelihood of fields being closer to main elephant pathways, which a) can create a barrier to elephant movement and b) make fields more vulnerable to raiding. Such land use zoning would, therefore, reduce the amount of crop damage by elephants, whilst having a minimal effect on elephant movements in the area.

It appears that farmers in the Okavango Delta panhandle are fully aware of the increased risk of raiding when crops are planted close to elephant movement paths. However, most farmers are reluctant to change field locations or consider such paths when choosing new field sites. Presumably, this is because such actions are costly and time consuming and, as figure 18 in chapter five illustrates, elephant pathways are almost ubiquitous throughout the study area. Changes in land use planning policy, which restricts fields from being allocated near such movement paths (such as those described above) may not, therefore, be accepted by the majority of the farming community. The Government of Botswana currently recommends avoiding wildlife movement paths when allocating fields in the eastern Okavango panhandle, (Tawana Land Board 2005). However, my results suggest that the underlying thinking and concept behind such policies has not been adequately conveyed to the community and therefore there is a lack of understanding to the justification for such decision making. If mitigation strategies are not accepted by affected communities, then perceived levels of conflict may be heightened rather than reduced, and interventions may be regarded as ineffective or inappropriate (Hill 2004). Education campaigns, which explain such high risk zones and reasons behind land use plans, could improve the community perception of such management initiatives.

When human or wildlife populations increase the likelihood of spatial and resource use overlap increases, which ultimately intensifies competition. In the Okavango Delta panhandle, the elephant population appears to be increasing at a rapid rate (see chapter two). Yet, there was no statistically significant relationship between elephant numbers or elephant density and crop raiding incidents per year, suggesting that a growing elephant population does not necessarily result in more crop-raiding incidents. This result, however, does not confirm that a growing elephant population does not exacerbate competition between people and elephants. Evidently, many farmers in the Okavango panhandle believe reducing the number of elephants in the area will reduce the problems they are facing with living close to elephants, which indicates that perceived conflict could be inflated by a larger elephant population, perhaps as a result of an increased likelihood of encounters with elephant.

Consequently, it is also important to include elephant population management policies into a human-elephant conflict management strategy for this area. In order to devise effective conservation and management strategies for an elephant population, however, we need to be able to reliably estimate population numbers, trends and densities as well as gain an understanding of why such patterns are occurring. Aerial surveys are an effective method for surveying elephants in the Okavango panhandle, due to the inaccessibility and size of the area, but such sampling methods are subject to a number of potential sources of bias (Caughley 1974; Marsh & Sinclair 1989; Jachman 2002; Elphick 2008; Laake, Dawson & Hone 2008). In my study of the elephant population in the area, simulations were used to verify the precision of aerial survey abundance estimates and trends in population numbers. These simulations substantiated the high population growth rates calculated for this area. It is difficult, however, to be certain why the population is increasing at such a high growth rate. Population dynamics can be driven by both intrinsic demographic factors (such as low mortality rates and low inter calving rates) as well as extrinsic environmental factors (such as high immigration rates and resource availability). Different driving factors will require different management strategies. For example, if the population is increasing due to immigration across fences, then a suitable management strategy would be to improve fence maintenance and promote improved transboundary monitoring. If, however, the population is increasing due to low mortality rates then management strategies such as increasing the off-take of elephants under community hunting quotas may need to be considered. More detailed research on the population dynamics and movements of elephants in this area, as well as more accurate estimates of population numbers (through high intensity aerial surveys with small strip widths) are needed to facilitate more informed elephant population management decisions and policies. Very little research has been conducted in northern Botswana on, for example, the detailed population structure of elephant herds and demographic characteristics to establish confident detailed information about their population growth rates. Such a study would certainly provide very useful information for more informed decision making in this regard.

7.3. Conclusion

The level of competition between humans and elephants is driven by a combination of actual conflict (i.e. elephant damage and human disturbance) and perceived conflict. Mitigation efforts therefore need to address both aspects of HEC in their design and implementation. If loss (actual and perceived) is matched by benefit (actual and perceived) then overall conflict can be reduced. Actual wildlife damage can be reduced through management initiatives such as short-term mitigation methods like chilli deterrents or longer-term land use and elephant management planning, with the right combination of information on localised patterns in both the ecological and social contributing factors. Perceptions of conflict are more difficult to address because they are so complex. Attitudes towards wildlife and subsequently perceived conflict levels can, however, be improved through environmental education awareness campaigns and by increasing benefits to the community from wildlife through tourism or other wildlife related enterprises.

In the Okavango Delta panhandle, perceived conflict appears to be greater than actual measurable conflict. Therefore, strategies to improve perceptions of conflict and attitudes towards wildlife are needed. It is also important to address actual conflicts by implementing appropriate mitigation methods because this will reduce actual damage and help to improve perceived conflict levels. This study has provided information that can aid in designing appropriate management approaches that will likely be accepted by both government and community stakeholders. Such strategies include involving and empowering community members in both monitoring and mitigation efforts; establishing micro level land use plans for agricultural development, in order to avoid high risk crop damage areas and providing agricultural free zones to facilitate undisturbed elephant movements; management suggestions for a burgeoning elephant population, i.e. improve fence maintenance to restrict elephant immigration; increase benefits to communities from wildlife (particularly elephants) through tourism enterprises; and introduce environmental awareness campaigns to improve perceived conflict levels and attitudes towards elephants. It is imperative that people have realistic expectations of what can be achieved through conflict management and so adequate information exchange between stakeholders (e.g. government departments and communities) is an essential element of the successful implementation of such management strategies.

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Appendix I. Estimates of elephant numbers in the eastern Okavango Panhandle by block/strata, 1996-2010

Source	Block/ Strata	Year	Area	No. Animals Observed	Population Estimate	SE	95% Range		t	95% CI	95% CI as % Estimate	Density	Strip width	Transect space	Sampling Intensity (%)
DWNP (1996)	Total	1996	9835	114	3782	2021	*114	8131	2.15	4349	115	0.384	400	12	3.01
DWNP (1999)	Total	1999 (Wet)	9835	222	7353	3243	368	14338	2.15	6985	95	0.748	400	12	3.02
DWNP (1999)	Total	1999 (Dry)	9835	126	3886	2647	*126	9598	2.16	5712	147	0.395	400	12	3.24
DWNP (2001)	Total	2001	9919	458	13173	6355	*458	26873	1.04	13700	104	1.328	400	12	3.48
DWNP (2002)	Total	2002	9919	218	6660	2681	866	12454	0.87	5794		0.671	400	12	3.27
DWNP (2003)	Total	2003	9919	211	5261	2393	*211	10417	0.98	5156	98	0.53	400	12	4.01
DWNP (2004)	Total	2004	9841	447	11870	4151	2849	20891	0.76	9021	76	1.206	400	12	3.77
DWNP (2005)	Total	2005	9142	177	5088	2440	*177	10380	1.04	5292	104	0.557	400	12	3.48
DWNP (2006)	Total	2006	9919	280	9212	3556	1566	16858	0.83	7646	83	0.929	400	12	3.04
This study	Total	2008	9732	1927	9963	1170	7642	12284	1.984	2321	23.3	1.14	500	3.2	19.7
This	Total	2010	9732	2771	15027	2008	11043	19011	1.984	3984	25.7	1.72	500	3.2	19.2

study															
Jackson et al, (2008)	NG11	2003 (Dry)	5952	1806	3579	604	2373	4785	1.997	1206	33.7	0.71	800	2	40
Jackson et al, (2008)	NG11	2004 (Wet)	5280	456	1060	250	561	1559	1.997	499	47	0.21	800	2	40
This study	NG11	2008	5140	1555	6383	846	4692	8074	1.999	1691	26.5	1.24	500	2	23.7
This study	NG11	2010	5140	1800	7580	1120	5346	9824	1.999	2239	29.5	1.47	500	2	23.1
This study	NG12	2008	1092	270	1303	292	700	1906	2.064	603	46.2	1.19	500	2.5	18.6
This study	NG12	2010	1092	593	2937	1054	762	5112	2.064	2175	74	2.69	500	2.5	18.1
This study	NG13	2008	2500	102	1219	425	293	2145	2.179	926	75.9	0.49	500	5	10.3
This study	NG13	2010	2500	378	4591	1253	1861	7321	2.179	2730	59.4	1.84	500	5	10
This study	Cumulative Total	2008	8732	1927	8905	-	-	-	-	-	-	1.02	500	3.2	19.7
This study	Cumulative Total	2010	8732	2771	15429	-	-	-	-	-	-	1.73	500	3.2	19.2

*Where the lowest 95% CI was below the number of animals observed, the number of animals observed replaced this value

Appendix II. Jolly's Method II

N = the number of sample units in the population
n = the number of sample units in the sample
Z = the area of the Census Zone
z = the area of any one sample unit
y = the number of animals counted in that unit
$\hat{R}$ = the ratio of animals counted to area searched = $\Sigma y / \Sigma z$

s_y^2 = the variance between animals counted in all the units
$= 1 / n-1 * [\Sigma y^2 – [(\Sigma y)^2 /n]]$

s_z^2 = the variance between the area of all sample units
$= 1 / n-1 * [\Sigma z^2 – [(\Sigma z)^2 /n]]$

s_{zy} = the covariance between the animals counted and the area of each sample unit
$= 1 / n-1 * [\Sigma z * y – [((\Sigma z) * (\Sigma y)) /n]]$

Population Total: $\hat{Y}$ **=** Z.R

Population Variance: Var $(\hat{Y})$ **= [[**N(N-n)] /n] *** (**$s_y^2 – 2* \hat{R}* s_{zy} + \hat{R}^2 * s_z^2$)

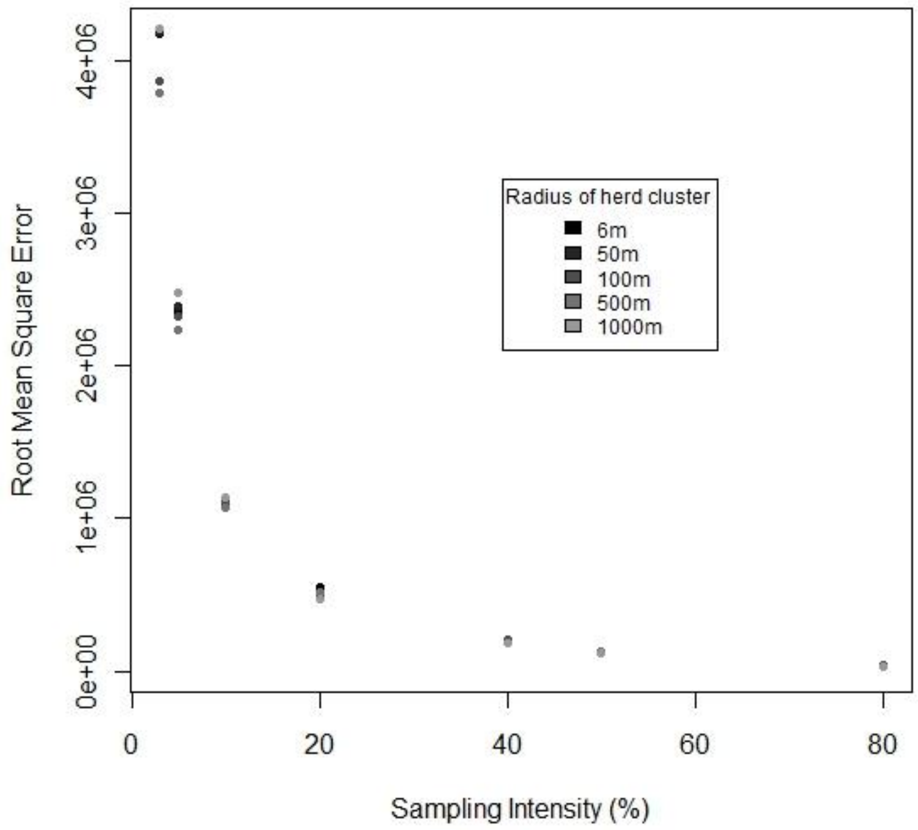
Population Standard Error: SE$(\hat{Y}) = \sqrt{[\text{Var}(\hat{Y})]}$

95% confidence limits of $\hat{Y}$ +/- t*SE$(\hat{Y})$
(where t is for n-1 degrees of freedom)

Appendix III. Herd number, type and mean herd size by observer on strip transects

Observer	Year	No. Herds Observed	No. Un-class. Herds	No. Bull Herds	No. Family Groups	X bull herd size (SE)	X family herd size (SE)
L	2008	143	3	62	52	3.03 (0.4)	11.59 (0.8)
L	2010	173	0	82	91	2.17 (0.26)	14.74 (1.36)
R	2008	121	4	65	78	2.54 (0.3)	11.98 (1.2)
R	2010	158	0	70	88	2.3 (0.3)	12.4 (1.35)
L/R	2008	1	0	0	1	-	-
Both	2008	265	7	127	131	2.8 (0.3)	11.8 (0.7)
Both	2010	331	0	152	179	2.23 (0.2)	13.59 (0.96)

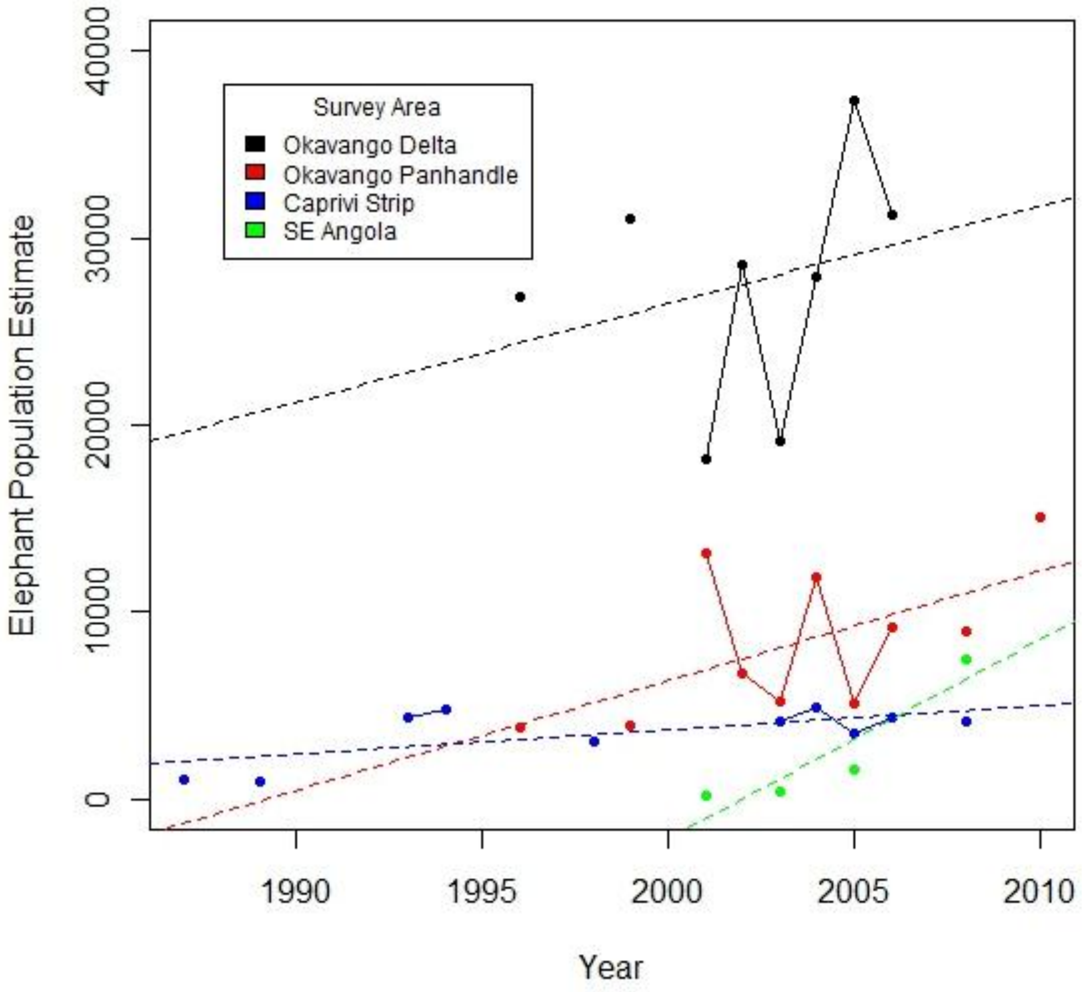
Appendix IV. Plot showing root mean square errors at different sampling intensities, different shades represent varying degrees of spatial clustering in elephant distribution (radius of herd)



Appendix V. Population estimates within the potential elephant range of the Okavango panhandle population, including the Okavango Delta, the West Caprivi and South East Angola

Year	Population Okavango Panhandle	Population Okavango Delta	Source	Population West Caprivi	Source	Population SE Angola	Source
1987	-	-		1037	Rodwell, Tagg & Grobler (1994)	-	
1988	-	-		-		-	
1989	-	-		902	Rodwell, Tagg & Grobler (1994)	-	
1990	-	-		-		-	
1991	-	-		-		-	
1992	-	-		-		-	
1993	-	-		4332	Rodwell, Tagg & Grobler (1994)	-	
1994	-	-		4733	Rodwell, Tagg & Grobler (1994)	-	
1995	-	-		-		-	
1996	3782	26795	DWNP (1996)	-		-	
1997	-	-		-		-	
1998	-	-		3068	Craig (1998)	-	
1999	7353 (wet) 3886 (dry)	30971	DWNP (1999)	-		-	
2000	-	-		-		-	
2001	13173	18175	DWNP (2001)	-		126	Chase & Griffin (2006)
2002	6660	28550	DWNP (2002)	-		-	
2003	5261	19079	DWNP (2003)	4136	Chase & Griffin (2004)	350	Chase & Griffin (2006)
2004	11870	27917	DWNP (2004)	4868	MET (2004)	-	
2005	5088	37351	DWNP (2005)	3456	Chase & Griffin (2006)	1550 (dry) 2030 (wet)	Chase & Griffin (2006)
2006	9212	31191	DWNP (2006)	4332	Chase & Griffin (2009)	-	
2007	-	-	-	-		-	
2008	8905	-	This study	4136	MET (2008)	7500	Chase & Griffin (2009)
2009	-	-	-	-		-	
2010	15113	-	This study	-		-	

Appendix VI. Population estimates from the Okavango Delta, Okavango Panhandle, West Caprivi and South East Angola, (see appendix IV for data used)



Appendix VII. The minimum adequate model of the GLM of the actual area damaged during an elephant crop-raid, with normal error structure, $p<0.01$ and results from model using sub-sampled data at the $0.5km^2$ scale

	All Data							**$0.5km^2$ Sample**	
Variable	**Estimate**	**Standard Error**	**t value**	**p value**	**F**	**df**	**F test p value**	**Estimate**	**p value**
Intercept	-21.17	5.72	-3.7	0.0002				-23	0.067
Distance to River	0.12	0.04	2.8	0.005	8.1	1	0.005	0.15	0.1
No. elephants in raiding herd	11.12	1.85	6.0	$3.64e^{-09}$	36.1	1	3.6e-9	13.31	0.0004
No. years raided in past	4.98	1.14	4.3	$1.75e^{-05}$	18.8	1	1.7e-5	5.17	0.06
Beans growing	13.93	4.46	3.1	0.002	2.3	1	0.1	16.96	0.07
Bush Fence	8.02	4.43	1.8	0.07	0.2	1	0.6	8.57	0.35
Beans : Bush fence	-16.43	5.74	-2.9	0.004	8.2	1	0.004	-16	0.2

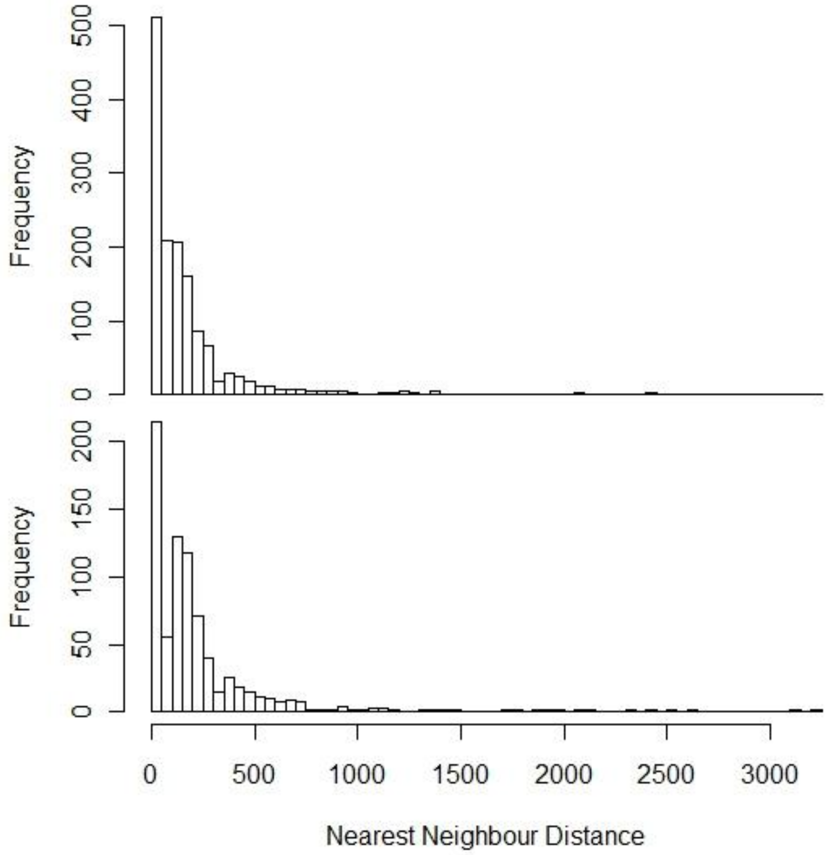
Appendix VIII. The Minimal adequate model using all data for the proportion of a field damaged during an elephant raid, with binomial error structure at $p<0.01$, and results from model using sub-sampled data at the $0.5km^2$ scale

	All Data							$0.5km^2$ Sample	
Variable	**Estimate**	**Standard Error**	**t value**	**p value**	**χ value**	**df**	**χ p value**	**Estimate**	**p value**
Intercept	-2.9	0.49	-5.83	1.04e-8				-3.1	0.0003
NYR	0.18	0.07	2.77	0.006	287.8	1	0.006	0.16	0.29
NE	0.89	0.12	7.21	2.2e-12	2068	1	2.6e-13	0.98	4.5e-6
Area	-0.008	0.002	-4.52	7.9e-6	916	1	1.1e-6	-0.006	0.07
Acacia	-0.18	0.35	-0.52	0.6	702.7	11	0.006	-0.50	0.5
FM	-1.01	1.30	-0.78	0.4	-	-	-	-1.1	0.5
Mix	-0.91	0.30	-3.04	0.002	-	-	-	-0.9	0.08
Riv	-0.76	0.42	-1.8	0.07	-	-	-	-0.7	0.32
Ser	0.75	0.51	1.49	0.13	-	-	-	0.7	0.39
Termteak	-0.44	0.34	-1.23	0.2	-	-	-	-06	0.3

Appendix IX: Explanatory variables measured for all fields, both raided and non-raided in 2008

Environmental Variable	**Mean (± se) of continuous variable**
Distance variables	
Distance to temporary water (waterholes) (m)	985.1 (± 20.64)
Distance to permanent water (river) (m)	3,771 (± 115.77)
Distance to next field (m)	90.25 (± 8.68)
Distance to village (m)	3,979 (± 101.14)
Distance to settlement (m)	2,363 (±67.71)
Distance to main road (m)	3,363 (± 115)
Distance to elephant path (m)	918.4 (±22.49)
Distance damage to watch hut (m)	221.8 (± 6.24)
Field Variables	
Area of field (m^2)	30,350 (± 1,206.02)
Elevation (m)	992.9 (± 0.52)
No. Adjacent fields	1.67 (± 0.03)
No. watchuts < 50m	1.34 (± 0.05)
No. Years field established	11.48 (± 0.31)
Harvest Month	Categorical (March; April; May; June)
Vegetation type	Categorical (Mopane; Acacia 1; Acacia 2; Teak; False Mopane; Wild Syringa; Riverine; Terminalia)
No. Years previously raided	4.69 (± 0.22)
Crop Variables	
Crop types present: (Pumpkin; Millet; Maize; Sorghum; Beans; Groundnuts; Watermelon; Sweet reed; Other (cow peas, rape, tomatoes, etc.))	Categorical (present =1, absent = 0)
Total no. crop types grown	5.1 (±2.15)
Mitigation Variables	
Mitigation used: (Bush fence; Wire fence; Watch hut; Tin cans; Plastic; Shouting/whistling; Gun; Dogs; Chilli; Drumming; Trench; Whip; Torch)	Categorical (present =1, absent = 0)
Number of Guards	1.82 (± 0.06)
Farmer Characteristics	
Age of farmer	49.5 (± 0.44)
Sex of farmer	Categorical (male; female)
Ethnicity of farmer	Categorical (Bahambukushu; Bayei; Basarwa; Other)
Family Size	7.3 (± 0.11)
Livelihood	Categorical (Farmer; Kgosi; Builder; Headman; VTC chair; VTC member; Nurse assistant; Elephant research; Farmer union chair)
Other crop raiding species	
Animal species: (Hippo; Monkey; Baboon; Porcupine; Warthog; Duiker; Livestock)	Categorical (present =1, absent=0)
Elephant Characteristics	
Elephant Sex	Categorical (male; female)
Elephant Age	Categorical (adults only (footprint > 35cm) = 1, adolescents only (30cm < footprint < 35cm) = 2, adults and adolescents (footprint > 30cm) = 3, all ages including juveniles (footprint > 15cm) = 4)
Elephant herd size	4.75 (±3.46)

Appendix X: Histograms of nearest neighbour distances for a) all fields (raided and non-raided in 2008-2010), b) raided fields



a

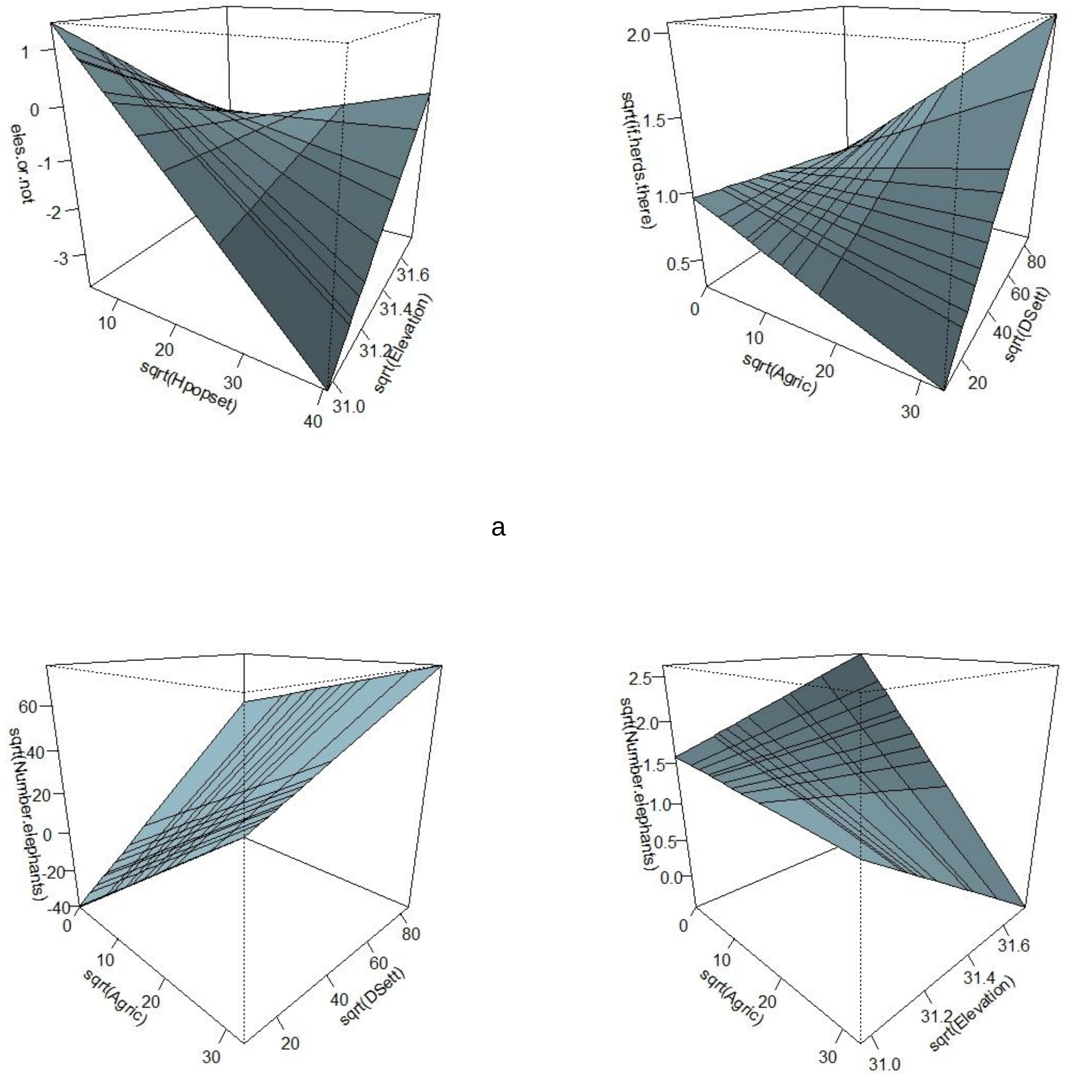
b

Appendix XI: Statistically significant univariate results for variables explaining the susceptibility of a field to raiding by elephants (using all data)

Explanatory Variable	Estimate	P value	Influence on susceptibility of a field to raiding by elephant
Farmer characteristics			
Farmers Age	-0.010	0.006	Older farmers less likely to be raided
Basarwa Ethnicity	-0.878	$3.27e^{-7}$	Basarwa farmers less likely to be raided
Baxereku Ethnicity	-1.026	0.001	Baxereku farmers less likely to be raided
Field characteristics			
No. Years Raided in past	0.158	$<2e^{-16}$	Fields raided frequently in past more likely to be raided
Area of field	-0.0004	0.003	Larger fields less likely to be raided
Age of field	-0.018	0.0002	Older fields less likely to be raided
No. Adjacent fields	-0.418	$<2e^{-16}$	Fields with more adjacent fields less likely to be raided
Dist to village	$1.965e^{-4}$	$<2e^{-16}$	Fields further from the village more likely to be raided
Dist to main road	$5.528e^{-5}$	$1.78e^{-5}$	Fields further from the main road more likely to be raided
Dist to Elephant Path	$-9.796e^{-4}$	$<2e^{-16}$	Fields further from Elephant paths less likely to be raided
Dist to River	$5.100e^{-5}$	$6.12e^{-5}$	Fields further from the river more likely to be raided
Dist to waterhole	$-3.428e^{-4}$	$9.46e^{-7}$	Fields further from waterholes less likely to be raided
Dist to next field	0.0005	0.019	Isolated fields more likely to be raided
Other Raiding Animals			
Livestock raiders present	-1.223	$<2e^{-16}$	Fields with livestock raiding less likely to be raided
Porcupine raiders present	0.382	0.0005	Fields with porcupine raiding more likely to be raided
Warthog raiders present	-1.287	0.007	Fields with warthog raiding less likely to be raided
Crop Types			
Other crops (i.e. cow peas; spinach; tomatoes)	-0.431	0.011	Fields growing other crops less likely to be raided
Groundnuts	-0.241	0.028	Fields growing groundnuts less likely to be raided
Pumpkins	0.310	0.004	Fields growing pumpkin more likely to be raided
Mitigation Used			
Wire fence	-0.470	$3.67e^{-5}$	Fields with wire fence less likely to be raided
Shouting	-1.480	$<2e^{-16}$	Fields where farmers shout less likely to be raided
Plastic on fence	-0.406	0.017	Fields with plastic on fence less likely to be raided
Dogs	-0.525	0.0002	Fields with dogs less likely to be raided
Watch hut	0.241	0.026	Fields more likely to be raided had watch huts
Surrounding Vegetation			
False Mopane	-0.733	0.047	Fields surrounded by False Mopane less likely to be raided
Mixed woodland	0.589	0.0008	Fields surrounded by mixed vegetation more likely to be raided
Terminalia	0.851	0.002	Fields surrounded by Terminalia more likely to be raided
Teak	0.761	0.0005	Fields surrounded by Teak more likely to be raided

Appendix XII. An example of an elephant path (a) and the identification of different individual elephant tracks (b), see arrows

Appendix XIII. Perspective plots of the effect of interactions between a) human population in nearest settlement and elevation on the probability of a pathway being used or not by elephant; b) distance to the nearest settlement and area of cultivated land on the likelihood of number of elephant herds crossing a pathway; c) area of cultivated land and distance to the nearest settlement on the likelihood of estimated number of elephants crossing a pathway; and d) area of cultivated land and elevation, on the likelihood of estimated number of elephants crossing a pathway



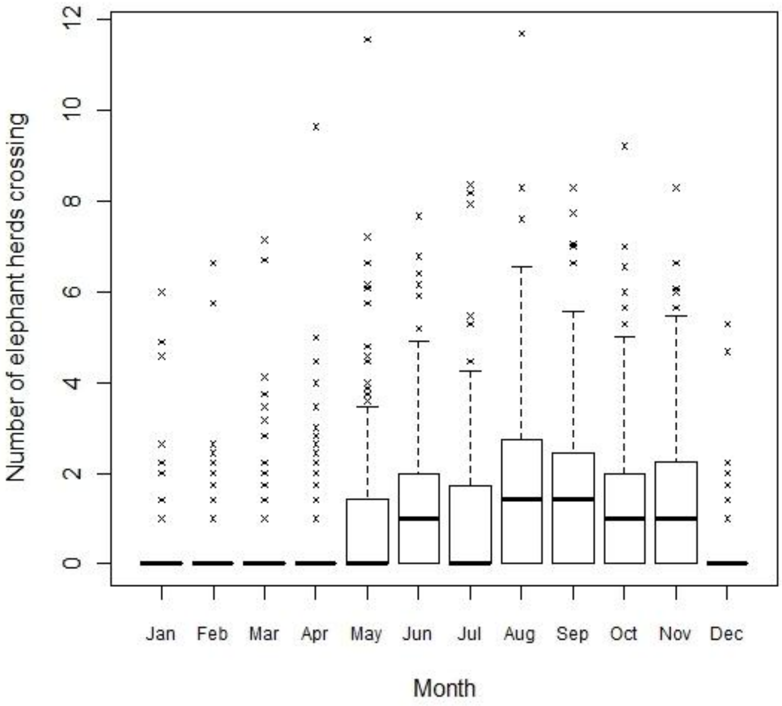
a

b

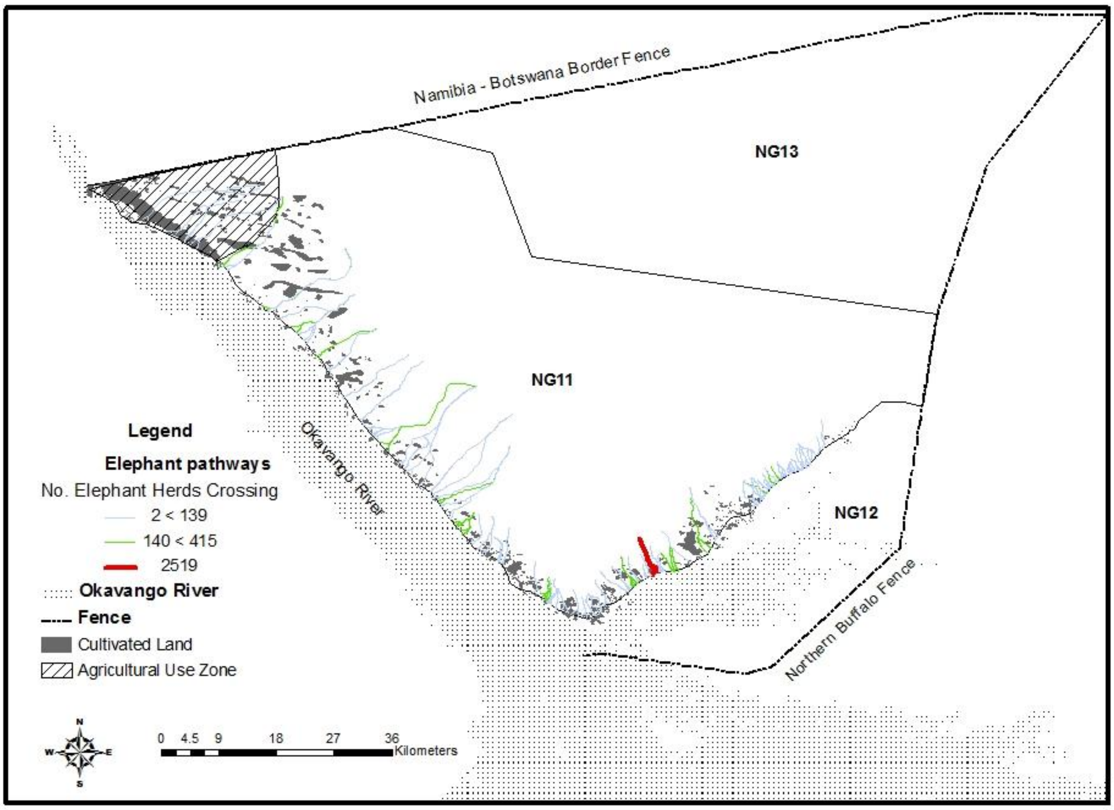
c

d

Appendix XIV. Number of elephant herds crossing the main road each month over three years (2008-2010).



Appendix XV. Map showing elephant path use, human activity (cultivated land and settlements) and proposed agricultural use zone (shaded)



Appendix XVI: Questions used in questionnaire survey

Category	**Questions**
Socioeconomic Factors	*Farmer details (Name/Age/ Gender /Village)*
	Ethnicity (Bahambukushu/Bayei/Basarwa/Other)
	Religion (Christian/African tradition/ Other)
	What is your totem?
	Besides farming, what else do you do for a living? What are your other sources of income? (Full time Employment/ Piece Jobs/Small Businesses/Other/None)
	Highest Education Completed?
	Wealth ranking: A (high) B (upper middle) C (lower middle) D (low) E (very low)
	How many cattle do you or your spouse own? (none (E)/1-10 (D)/11-20 (C)/21-50 (B)/50+ (A))
	Do you own your field? (yes/no)
	What type of roof and walls do you have in your house?(Cement-iron (A)/Cement-grass (B)/Reed-iron (C)/Mud-iron (D)/Reed-grass (D)/Mud-grass (E)/Other)
	Do you or any members of your immediate family own any of the following? (Radio (D)/Mokoro(C-B)/Car(A))
Agricultural Practices	*How many people does your field support?*
	Why do you grow crops?(Personal Consumption/Cash Revenue/Personal Consumption and Sell some/Other)
	How productive is your field?(Very Good/Good/Medium/Poor)
	How many bags do you harvest on average?
	How many seeds did the government give you in this crop season?
	How much money did the government give you in this crop season?
	How much does it cost you to farm each crop season?
	How many bags feed your family for a year?
	What difficulties do you encounter with growing crops? What is the most serious difficulty?
	How many bags do you get in a bad harvest?
	How do you provide for your family if you have a bad harvest?
Attitudes towards wildlife & elephants	*Do you like living with wildlife around you?(yes/no/unsure)*
	What are the benefits of having wildlife around you?
	How do you feel about Elephants? (Love/Like/Neutral/Dislike/Hate)
	What are the advantages of living close to elephants?
History of HEC	*Do you have a problem with elephants? (yes/no)*
	Which (if any) of the following problem(s) do you have with elephants? (Crop Raiding/Fence Damage/Fear of walking/Property Destruction/Competition for water/Injuring-killing people/Other)
	Which is the biggest problem?
	Was your field raided by elephants this year?

	How many times was your field raided this year? *Which previous years has your field been raided by elephants?* *Do you know anybody in your village that has been killed/injured by an elephant in the last 5 years? If yes, how many people?* *How many elephants have been killed in or near your village (in the last 5 years)?* *Do you agree with killing elephants? If no, why not?* *Do you eat elephant meat? If no, why not?*
Potential options for mitigating HEC	*What do you want to do about the elephants in this area? (Kill All/Kill Some/Kill One/Move them/Mitigate damage/Leave them/Other)* *Who do you feel is responsible for reducing the crop-raiding?(Farmers/Kgosi/DWNP/Land board/Department of Agriculture/President/Other)* *Who do you think is responsible for protecting your field?(Farmer/DWNP/Land board/Other)* *Have you ever received compensation for elephant damage? (yes/no)* *If yes, how much did you receive for the most recent payment?* *Do you think the compensation for crop damage is fair? (yes/no)* *If no, is it because it doesn't cover: (the cost of growing the crop/the resale value of the crop/the cash required to purchase additional food/other reason)* *What do you think the government's priority should be? (Increase compensation/Improve mitigation/Reduce the elephant population/Other)* *Do you think elephant movement paths should be considered when land is chosen and allocated for fields? (yes/no)* *In principal, would you ever be willing to move your field to reduce the crop-raiding by elephants? (yes/no)*
Conclusion	*Do you feel like you are in conflict with elephants?* *How would you describe the conflict you are experiencing? (High/Quite high/Medium/Low/None)* *Do you think it is possible to find a solution to reduce the problem with elephants here in the future? (yes/no/unsure)* *Which one of these development issues do you think should be the priority to address in this area? (Electricity supply/Building bridge at Mohembo/Tarring the road/Elephant problem in area/Other)*

Appendix XVII. Conflict Level Ranking Criteria

CLS1 = Number of times raided this year rank
CLS2 = Number of years raided rank

Conflict Level Score (CLS)	No. times raided this year	No. years raided
0	0	0
1	1	1
2	2	2
3	3	3
4	>3	>3

Conflict Level = CLS1+CLS2 range

Conflict Level	CLS1+CLS2 range
0	0
1	1-2
2	3-4
3	5-6
4	7-8

Appendix XVIII. Questionnaire Variables

Category	Variable	Description	Variable type	Analysis*
Socioeconomic Data	Gender	M or F	Categorical	AW FE PCL SOL
	Age	Range 18 – 92	Continuous	AW FE PCL SOL
	Highest Education Completed	0 = none, 1 = non-formal, 2 = S1, 3 = S2, 4 = S3, 5 = S4, 6 = S5, 7 = S6, 8 = S7, 9 = F1, 10 = F2, 11 = F3, 12 = F4, 13 = F5	Ordered factor	AW FE PCL SOL
	Additional income source	0 = none, 1 = full-time employment, 2 = piece jobs, 3 = small business, 4 = other	Categorical	AW FE PCL SOL
	Wealth Ranking	1 = very low, 2 = low, 3 = lower middle, 4 = upper middle, 5 = high	Ordered factor	AW FE PCL SOL
Agricultural Practices	No. people field supports		Continuous	SOL
	Reason for growing crops	1 = personal consumption, 2 = cash revenue, 3 = both	Categorical	
	Perceived field productivity level	1 = poor, 2 = medium, 3 = good, 4 = very good	Ordered factor	
	Average harvest		Continuous	SOL
	Average bad harvest		Continuous	SOL
	Crop needed to support family		Continuous	SOL
	Most serious difficulty with farming		Category	
Attitudes towards wildlife & elephants	Like wildlife	1 = yes, 0 = no	Binomial	FE PCL
	Feeling towards elephants	1 = love, 2 = like, 3 = neutral, 4 = dislike, 5 =hate	Categorical	AW PCL
	Advantages of elephants	0 = none, 1 = tourism, 2 = employment, 3 = income, 4 = meat, 5 = beautify environment, 6 = future generations	Categorical	
	Elephant numbers increased	1 = yes, 0 = no	Binomial	
	Since when increasing	1 = one year ago, 2 = 2-5yrs, 3 = 6-10yrs, 4 = >10yrs	Categorical	
	Elephants cross fences	1 = yes, 0 = no	Binomial	
Problems with wildlife	Which animals cause problems (list animal)	1 = yes, 0 = no	Binomial	

	Animal causing biggest problem		Categorical	
	Problem with elephants	1 = yes, 0 = no	Binomial	
	What problems experienced with elephants (list problems)	1 = yes, 0 = no	Binomial	
	Biggest problem with elephants	1 = crop-raiding, 2 = fence damage, 3 = fear of walking, 4 = property destruction, 5 = competition for water, 6 = injury/killing people	Categorical	
	Is elephant crop-raiding getting worse	1 = yes, 0 = no	Binomial	
	Year noticed getting worse		Continuous	
	Reason for problem increasing	1 = raid/yr increase, 2 = damage/raid increase, 3 = elephants not scared	Categorical	
	Severity of raiding last season	0 = none, 1 =minimal, 2 = minor, 3 = quite serious, 4 = very serious	Ordered factor	
	Parents experience elephant problems	1 = yes, 0 = no	Binomial	
	No. people killed by elephant in last 5 yrs		Continuous	FE PCL SOL
	No. elephants killed by people in last 5 yrs		Continuous	
	Agree to kill elephant	1 = yes, 0 = no	Binomial	SOL
	Eat elephant meat	1 = yes, 0 = no	Binomial	
	Reason for not eating elephant	1 = totem, 2 = religion, 3 = allergy, 4 = dislike taste, 5 = human likeness, 6 = other	Categorical	
	Utilise other parts	1 = yes, 0 = no	Binomial	
Elephant Raiding History				
	No. years field raided by elephants		Continuous	PCL SOL
	Raided this year	1 = yes, 0 = no	Binomial	
	No. times raided in year of questionnaire		Continuous	PCL SOL
	Area damaged in recent raid (within last 3 years)		Continuous	PCL SOL
Options for Mitigation	Preferred management strategy	1 = kill all, 2 = kill some, 3 = kill one, 4 = move, 5 = mitigate, 6 = other	Categorical	PCL
	Body responsible for reducing crop-damage	1 = farmer, 2 = chief, 3 = DWNP, 5 = Land board, 4 = Dept. Agriculture, 6 = president, 7 = all, 8 = other	Categorical	PCL
	Body responsible	1 = farmer, 2 = DWNP, 3 =	Categorical	

	for protecting individual fields	Land board, 4 other		
	Are farmers responsible?	1 = yes, 0 = no		PCL
	Compensation received	1 = yes, 0 = no	Binomial	
	Recent amount received		Continuous	
	Compensation fair	1 = yes, 0 = no	Binomial	PCL SOL
	Reason for unfair	1 = cost of growing, 2 = resale value, 3 = replacement value	Categorical	
	Government priority	1 = increase compensation, 2 = improve mitigation, 3 = reduce elephant numbers	Categorical	
	Consider elephant paths	1 = yes, 0 = no	Binomial	SOL
	Consider moving field	1 = yes, 0 = no	Binomial	SOL
Conclusion	Community attitude toward elephants	1 = like, 2 = tolerate, 3 = hate	Categorical	
	Personal happiness with elephants	1 = yes, 0 = no	Binomial	
	Personal conflict with elephants	1 = yes, 0 = no	Binomial	
	Perceived conflict level	0 = none, 1 = low, 2 = medium, 3 = quite high, 4 = high	Ordered factor	PCL SOL
	Possible to reduce conflict	1 = yes, 0 = no	Binomial	PCL SOL
	Priority socioeconomic issue in area	1 = electricity, 2 = bridge, 3 = road, 4 = elephant problem	Categorical	
Field Location	Dist. Field to elephant path		Continuous	PCL
	Dist. Field to road		Continuous	PCL
	Dist. Field to river		Continuous	PCL
	Dist. Field to village		Continuous	PCL
	Dist. Field to waterhole		Continuous	PCL
Tourism Benefit	Does village benefit from tourism	1 = yes, 0 = no	Binomial	AW FE PCL

*Analysis categories:

AW = attitude towards wildlife generalised linear model

FE = feeling towards elephants ordered probit model

PCL = perceived conflict level ordered probit model

SOL = possible solution generalised linear model

Appendix XIX. Results of GLM, with binomial error structure, for whether a respondent likes wildlife or not, with influential points 823, 83 and 549 removed.

Coefficients	Estimate	Std. Error	z value	p value	χ^2	df	p
(Intercept)	1.38	0.87	1.60	0.11			
Male	0.54	0.18	3.02	0.002	9.1	1	0.003
Age	-0.001	0.01	-0.13	0.90	0.20	1	0.7
Ed.level - non-formal	-5.10	2.32	-2.20	0.03	15.7	4	0.004
Ed.level - primary	1.30	0.70	1.87	0.06	-	-	-
Ed.level - junior sec	2.70	0.98	2.75	0.006	-	-	-
Ed.level - senior sec	-0.41	3.66	-0.11	0.91	-	-	-
feel.elephants - like	0.13	0.86	0.15	0.88	129.9	4	<0.001
feel.elephants - neutral	-0.60	0.84	-0.72	0.47	-	-	-
feel.elephants - dislike	-2.86	0.76	-3.76	0.0002	-	-	-
feel.elephants - hate	-2.87	0.76	-3.80	0.0001	-	-	-
Village tourism benefit	0.81	0.17	4.79	$1.7e^{-06}$	23.3	1	<0.001
Age:Ed.level1	0.09	0.04	2.42	0.02	13.3	4	0.01
Age:Ed.level2	-0.02	0.01	-1.11	0.27	-	-	-
Age:Ed.level3	-0.06	0.03	-1.86	0.06	-	-	-
Age:Ed.level4	0.002	0.12	0.02	0.99	-	-	-

Appendix XX. Results of GLM, with binomial error structure, for whether a respondent felt there was a solution to reduce the problem with elephants or not

Variables	Estimate	Std. Error	z value	p value	χ^2	df	p
(Intercept)	-0.13	0.25	-0.22	0.80			
Consider elephant paths	0.54	0.25	2.19	0.03	6.3	1	0.01
Ed.level - non-formal	2.91	1.10	2.64	0.008	30.4	4	$4.1e^{-6}$
Ed.level - primary	0.71	0.25	2.87	0.004	-	-	-
Ed.level - junior sec	1.02	0.33	3.12	0.002	-	-	-
Ed.level - senior sec	-0.92	0.97	-0.95	0.34	-	-	-
sqrt(Area.damaged)	-0.007	0.003	-2.45	0.01	7.1	1	0.008
management - kill some	0.16	0.28	0.59	0.56	34.8	9	$6.4e^{-5}$
management - kill one	-1.68	1.24	-1.36	0.17	-	-	-
management - move	-0.40	0.31	-1.26	0.21	-	-	-
management - mitigate	1.58	0.51	3.11	0.002	-	-	-
management - leave	-1.85	0.94	-1.98	0.05	-	-	-
management -other	15.6	685	0.02	0.98	-	-	-
management – kill&move	-0.68	1.49	-0.46	0.65	-	-	-
management – kill&mitigate	-0.43	0.69	-0.62	0.54	-	-	-
management - All	15.63	1029	0.02	0.99	-	-	-