Exploring the causes of and mitigation options for human-predator conflict on game ranches in Botswana: How is coexistence possible?

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Abstract

Large carnivores in southern Africa are threatened by habitat loss and persecution by humans. Game ranches have the potential to provide habitat for free-ranging predators, but carnivore depredation on game-stock can result in human-predator conflict, and the industry's role in predator conservation has been described as a gap in knowledge.

The density of predators on Botswana commercial farmland was calculated using spoor and camera-trap surveys. Scat-analysis was used to determine the proportion of livestock and game-stock in the cheetah's diet, the species reported to cause the biggest economic losses on Botswana game ranches. Questionnaires to determine the direct costs, drivers and potential mitigation methods of human-predator conflict, were conducted with a representative from 86.2% of registered game ranches in Botswana, plus an additional 27 livestock farmers. The effectiveness of translocating 'problem' predators was analysed using questionnaires with farmers and survival data from 11 GPS-collared 'problem' cheetahs.

Cheetahs were more commonly detected on game ranches than livestock farms; detected cheetah density (0.32/ 100 km^2) was within the previously assumed range for commercial farmland. Leopard density (0.37/ 100 km^2) was 2.6 – 4.1 times lower than previously assumed. Based on scat-analysis cheetahs consumed 1.6 – 5.9 times less game-stock than reported by farmers. Ranchers with only game-stock reported a significantly greater tolerance of predators than ranchers with game-stock and livestock, or farmers with only livestock. The primary drivers of conflict were the presence of cattle and the relative abundance of inexpensive buffer-prey species. Ranchers stated that 'problem' predator hunting, trophy hunting, photographic safaris and the translocation of 'problem' animals were the best solutions to enable farmers and predators to coexist. However, the survival rate of translocated cheetahs was low (18.2%), and translocation was ineffective at reducing stock losses to predators.

Overall, game ranches in Botswana have a potential to conserve predator populations. This potential is limited by a lack of communication and support between government, NGOs and the game ranching industry, resulting in potential uncertainty of the industry's future. Initiatives to support game ranching and to encourage the formation of conservancies are likely to maximise the conservation benefits.

Declaration

I declare that

i) the data used in this thesis are original except where indicated by special reference in the text;

ii) this thesis is my own unaided work, both in concept and execution, and that apart from the normal guidance from my supervisor; I have received no assistance except as stated below:

Cheetah Conservation Botswana (CCB) is responsible for the conception of the predator densities (Chapter 5), prey analysis (Chapter 6) and cheetah *Acinonyx jubatus* translocation studies (Chapter 8). Ann Marie Houser and Dr Vivien Kent on behalf of CCB designed the methodology for the 2007 spoor survey and conducted the initial cheetah translocations.

iii) neither the substance nor any part of the above thesis has been submitted in the past, or is being, or is to be submitted for a degree at this university or at any other university, except as stated below:

In Chapter 8, five of the 12 cheetahs documented in the translocation study had been previously included in a home range analysis (Houser, 2008; Houser *et al.*, 2009a), however, this is the first time data on the cheetahs' survival time, return to capture site and site fidelity to the release site has been presented. It is also the first time the data has been discussed in relation to predator translocation.

The research complied with protocols approved by the ethics committees of the University of Cape Town and adhered to South African and Botswana legal requirements. Opinions expressed, and conclusions arrived at, are those of the author and are not necessarily to be attributed to any of the funding or supporting organisations.

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The photograph on the front cover was taken with a motion-camera, courtesy of Cheetah Conservation Botswana.

List of Acronyms

BMC	Botswana Meat Commission
B-SECR	Bayesian spatially explicit capture-recapture
BWP	Botswana pula
BWPA	Botswana Wildlife Producers Association
CBNRM	Community Based Natural Resource Management
CCB	Cheetah Conservation Botswana
CKGR	Central Kalahari Game Reserve
DA	Data augmentation
DWNP	Department of Wildlife and National Parks
EEC	European Economic Community
GDP	Gross domestic product
GPS	Global positioning system
HPC	Human-predator conflict
HWC	Human-wildlife conflict
IUCN	International Union for Conservation of Nature
KTP	Kgalagadi Transfrontier Park
MMDM	Mean maximum distance moved
NGO	Non-government organisation
PAC	Problem animal control
RRT	Randomised response technique
SADC	Southern African Development Community
SECR	Spatially explicit capture-recapture
US $	United States dollar
USA	United States of America
VHF	Very high frequency
WMA	Wildlife Management Area

Species list

List of animal species used in the main text of this document; their scientific names are stated in accordance with the International Union for Conservation of Nature (IUCN) red list of threatened species (IUCN, 2013).

Common name	Scientific name
Aardvark	*Orycteropus afer*
Aardwolf	*Proteles cristata*
African buffalo	*Syncerus caffer*
African elephant	*Loxodonta africana*
African wild dog	*Lycaon pictus*
African wildcat	*Felis silvestris lybica*
Banded mongoose	*Mungos mungo*
Bat-eared fox	*Otocyon megalotis*
Black bear	*Ursus americanus*
Black-faced impala	*Aepyceros melampus petersi*
Black-footed cat	*Felis nigripes*
Black rhinoceros	*Diceros bicornis*
Black wildebeest	*Connochaetes gnou*
Black-backed jackal	*Canis mesomelas*
Blesbok	*Damaliscus pygargus phillipsi*
Blue wildebeest	*Connochaetes taurinus*
Bontebok	*Damaliscus pygargus pygargus*
Brown bear	*Ursus arctos*
Brown hyaena	*Hyaena brunnea*
Cape fox	*Vulpes chama*
Cape porcupine	*Hystrix africaeaustralis*
Caracal	*Caracal caracal*
Cattle	*Bos primigenius*
Cheetah	*Acinonyx jubatus*
Common duiker	*Sylvicapra grimmia*
Common dwarf mongoose	*Helogale parvula*
Common eland	*Tragelaphus oryx*
Common genet	*Genetta genetta*
Coyote	*Canis latrans*
Domestic dog	*Canis lupus familiaris*
Domestic goat	*Capra hircus*
Domestic sheep	*Ovis aries*
Dorcas gazelle	*Gazella dorcas*
Ethiopian wolf	*Canis simensis*
Eurasian lynx	*Lynx lynx*
Fallow deer	*Dama dama*

Common name	Scientific name
Gemsbok	*Oryx gazella*
Giraffe	*Giraffa camelopardalis*
Golden eagle	*Aquila chrysaetos*
Grant’s gazelle	*Nanger granti*
Greater kudu	*Tragelaphus strepsiceros*
Grey wolf	*Canis lupus*
Helmeted guinea fowl	*Numida meleagris*
Hippopotamus	*Hippopotamus amphibius*
Honey badger	*Mellivora capensis*
Impala	*Aepyceros melampus*
Jaguar	*Panthera onca*
Kea	*Nestor notabilis*
Kob	*Kobus kob*
Leopard	*Panthera pardus*
Lion	*Panthera leo*
Martial eagle	*Polemaetus bellicosus*
Moose	*Alces americanus*
Mountain zebra	*Equus zebra*
Nile crocodile	*Crocodylus niloticus*
Nyala	*Tragelaphus angasii*
Ocelot	*Leopardus pardalis*
Ostrich	*Struthio camelus*
Plains zebra	*Equus quagga*
Puma	*Puma concolor*
Red fox	*Vulpes vulpes*
Red hartebeest	*Alcelaphus buselaphus caama*
Roan antelope	*Hippotragus equinus*
Sable antelope	*Hippotragus niger*
Scrub hare	*Lepus saxatilis*
Serval	*Leptailurus serval*
Side-striped jackal	*Canis adustus*
Sitatunga	*Tragelaphus spekii*
Slender mongoose	*Herpestes sanguineus*
South African ground squirrel	*Xerus inauris*
Southern African hedgehog	*Atelerix frontalis*
Southern lechwe	*Kobus leche*
Spotted hyaena	*Crocuta crocuta*
Springbok	*Antidorcas marsupialis*
Springhare	*Pedetes capensis*
Steenbok	*Raphicerus campestris*
Striped hyaena	*Hyaena hyaena*
Thomson’s gazelle	*Eudorcas thomsonii*

Common name	**Scientific name**
Thylacine (Tasmanian wolf)	*Thylacinus cynocephalus*
Tiger	*Panthera tigris*
Tsessebe (Topi)	*Damaliscus lunatus*
Tsetse fly	*Glossina palpalis*
Warthog	*Phacochoerus africanus*
Waterbuck	*Kobus ellipsiprymnus*
White rhinoceros	*Ceratotherium simum*
White-tailed deer	*Odocoileus virginianus*
Wild boar	*Sus scrofa*
Wolverine	*Gulo gulo*
Zorilla (Polecat)	*Ictonyx striatus*

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Preface

In this manuscript, in line with the policies and regulations of Botswana, the words '**game ranching**' refers to the management of semi-free wildlife species (herein referred to as '**game-stock**') in a semi-open natural system (Republic of Botswana, 2002; Cousins *et al.*, 2008). Game ranching, sometimes referred to as 'wildlife ranching' in the literature, should be distinguished from the intensive management and husbandry of species in captivity, such as Nile crocodile *Crocodylus niloticus* or ostrich *Struthio camelus* (Bulte & Damania, 2005).

A '**game ranch**' refers herein to any farm ranching game-stock, singularly or in conjunction with livestock; all farmers ranching game-stock are referred to as '**game ranchers**'. When specifically referring to ranches or ranchers which farm only game-stock, this distinction will be clearly made (i.e. **'game-stock only ranches**'). Ranches which farm livestock and game-stock will be referred to as described or abbreviated to an '**L/G farm'**, and farmers working on L/G farms are referred to as '**L/G farmers**'. A farm ranching only livestock is referred to as '**livestock farm**', and the farmers as '**livestock farmers**'. Collectively all '**farmers ranching livestock**' are referred to as described. A '**conservancy'** refers to a 'group of adjoining private commercial farms operating under a cooperative management agreement' as defined in ABSA (2003). '**Southern Africa**' is defined as South Africa, Lesotho, Swaziland, Namibia, Botswana, Zambia, Angola, Mozambique and Zimbabwe.

'**Game-stock**' collectively refers herein to wildlife species ranched on game ranches, including giraffe *Giraffa camelopardalis*, plains zebra *Equus quagga*, ostrich and antelope species. '**Free-ranging'** species are defined as wildlife species that naturally occur outside of game ranches or occur on game ranches, but whose movements are not restricted by fences, such as steenbok *Raphicerus campestris* and warthog *Phacochoerus africanus*. Cattle *Bos primigenius* and '**smallstock**' (goats *Capra hircus* and/or sheep *Ovis aries*) are collectively referred to herein as '**livestock**', and '**stock**' refers collectively to livestock and game-stock. '**Exotic species**' are defined as species outside of their historical distribution. The term '**large carnivores**' refers herein to mammalian carnivores with a mass over 20 kg, in southern Africa these are lions *Panthera leo*, leopards *Panthera pardus*, cheetahs *Acinonyx jubatus*, African wild dogs *Lycaon pictus*, spotted hyaenas *Crocuta crocuta* and brown hyaenas *Hyaena brunnea*.

Chapter One: Game ranching and human-wildlife conflict: An introduction

1.1 Chapter overview

The world's human population surpassed 7.2 billion in 2013 and by 2050 is projected to reach 9.6 billion (United Nations, 2012b). By the end of the 20th century, every ecosystem on earth and nearly 75% of the world's habitable land surface had been disturbed by human activities (Hannah *et al.*, 1994; Vitousek *et al.*, 1997). Of the nine 'planetary boundaries', designed as guidelines to avoid major human-induced environmental change, three of the nine (climate change, changes to the global nitrogen cycle and the rate of biodiversity loss) have been surpassed (Rockstrom *et al.*, 2009). Species extinction rates are reported to be 100 – 1000 times faster than pre-human rates (Lawton & May, 1995), and humans have caused the extinction of 5 – 20% of species in most organism groups (Pimm *et al.*, 1995). At the time of writing in 2014, approximately 20% of the world's biodiversity was threatened with extinction (Hoffmann *et al.*, 2010).

In an attempt to reduce the rate of biodiversity loss, by the end of the 20th century, protected areas for wildlife accounted for 11.5% of the Earth's surface (Chape *et al.*, 2003). These areas were invaluable for conservation and globally reflected the number of threatened species within a region (Bruner *et al.*, 2001; Gaston *et al.*, 2008; Loucks *et al.*, 2008). However, protected areas were often selected due to their lack of economic potential, rather than their biodiversity merit (Pressey, 1994; Scott *et al.*, 2001a; Loucks *et al.*, 2008). As a result, the protected area network in 2013 is considered insufficient to conserve the world's biodiversity (Brooks *et al.*, 2004; Rodrigues *et al.*, 2004). The habitat and survival of species whose range exists or extends outside of protected areas, is threatened by conflict with humans for land use and resources (Hutton & Leader-Williams, 2003). As wild habitat is increasingly replaced by cultivated land and domesticated species, in order to support the growing human population, the number of threatened species per region is predicted to increase (McKee *et al.*, 2003).

Conflict between humans and wildlife for the planets limited resources occurs worldwide; however, it is thought to be particularly intense in African savannah ecosystems (Homewood & Brockington, 1999). Africa is a continent famous for its biodiversity, yet the human population is expected to grow at a rate 2.4 times greater than the world average between 2013 and 2050 (United Nations, 2012b), placing enormous strain on the continent's developing infrastructure and ecology. Historically, agriculture and livestock production were the primary income of many southern African countries, and livestock production is still emphasised as a means of poverty reduction in Botswana (Central Statistics Office, 2012a). This has resulted in vast subsidies on cattle *Bos primigenius* farming often to the detriment of wildlife populations (Bond *et al.*, 2004). In an attempt to protect habitat outside of nationally protected areas, governments in Namibia, South Africa, Botswana, Zambia and Zimbabwe passed legislation which enabled landowners to utilise the wildlife on their property. This has resulted in a large increase in wildlife species on private land and has been crucial in the successful recovery of white rhinoceros *Ceratotherium simum* and bontebok *Damaliscus pygargus pygargus* populations (Flack, 2003). However, the game ranching industry has been criticised for its market driven, potentially unsustainable approach to conservation, and its overall role in preserving predator populations has been questioned (Lindsey *et al.*, 2009b).

On private land predators are often in conflict with farmers, due to depredation on livestock or game-stock, or for the potential threat predators can cause to human-life (Woodroffe *et al.*, 2005d). Private game ranches have a potential to increase the available habitat for Africa's large predator species (Lindsey *et al.*, 2012c), however, human-predator conflict (HPC) leading to the removal of predators has been reported (Lindsey *et al.*, 2013c). The direct costs and drivers of conflict between farmers and carnivores on game ranches is largely unknown and has been highlighted as a gap in knowledge (Inskip & Zimmerman, 2009; Balme *et al.*, 2014). The game ranching industry in Botswana is increasing (Rozemeijer, 2002), and the country is fundamental to the conservation of large predator populations in southern Africa (IUCN/SSC, 2007b). Consequently, there is an urgent need to examine the direct costs and drivers of HPC on Botswana's game ranches, and the applicability of methods to mitigate this conflict, which could be applied across the region.

This chapter discusses the limitations of protected areas for biodiversity conservation, and the effect cattle ranching has had upon wildlife biodiversity in southern Africa. The history of game ranching and its role in conservation will be discussed, followed by an introduction to human-wildlife conflict (HWC), focusing on HPC in southern Africa. Lastly, the importance of conserving predators and their role in ecosystems will be introduced. Botswana will be the focal country in all discussions.

1.2 Limitations of protected areas for biodiversity conservation

Although the world's protected area network has an essential role in protecting plant and animal biodiversity (Bruner *et al.*, 2001), it is potentially insufficient to sustain the long-term conservation of many species (Scott *et al.*, 2001b; Andelman & Willig, 2003; Rodrigues *et al.*, 2004). The positioning of protected areas is often biased to particular biomes or areas of low commercial value or productivity (Pressey, 1994; Scott *et al.*, 2001a; Jenkins & Joppa, 2009). For this reason at least 20% of the world's threatened flora and fauna remain largely outside of protected areas, colloquially known as 'gap species' (Rodrigues *et al.*, 2004). Additionally, many species whose range falls within the protected area network remain under threat, because of insufficient protection (for example in areas colloquially termed 'paper parks', where the biodiversity protection exists only on paper) (Langholz *et al.*, 2000; Jenkins & Joppa, 2009) or due to the unsuitability of the habitat (Bruner *et al.*, 2001). For protected areas to successfully conserve biodiversity they must contain all of the essential life-sustaining necessities under the full range of environmental conditions (Mordi, 1989). Yet, many of the world's protected areas contain incomplete ecosystems, are too small for migratory and wide-ranging species, result in ecological isolation and are threatened by continued conflict with people at their boundaries (Cumming, 1990; Wittemyer *et al.*, 2008).

In South Africa, Zimbabwe, Namibia and Botswana 11.2% of land is designated as protected areas for wildlife (Krug, 2001). The reserve network covers the distribution of large mammals, but it is estimated protected areas would need to expand by 30 – 100% (species dependent) to achieve minimal conservation targets for all mammals and amphibians (Fjeldsa *et al.*, 2004; Rondinini *et al.*, 2004). For example, the reserve network is thought to be inadequate to conserve many large carnivore populations (Nowell & Jackson, 1996; Weber & Rabinowitz, 1996; Loveridge *et al.*, 2001).

Large carnivores in particular have been eliminated from much of the world in a fashion disproportionate to species from other trophic levels (Millar *et al.*, 1999). They possess many of the characteristics that make species prone to extinction: they naturally occur at low densities, occupy a high trophic level, require large home ranges, have complex social behaviour and cause conflict with neighbouring human populations (Sillero-Zubiri & Laurenson, 2001). HPC outside of protected areas prevents emigration and immigration of fresh genetics; the removal of predators on park boundaries can act as a ‘sink’, drawing predators from within the protected area to fill the open territories on its borders (Woodroffe & Ginsberg, 1998; Loveridge *et al.*, 2007; Balme *et al.*, 2010). Additionally, the protection of some species such as spotted hyaenas *Crocuta crocuta* and lions *Panthera leo* can restrict the density and distribution of less dominant competitors, such as brown hyaenas *Hyaena brunnea*, cheetahs *Acinonyx jubatus* and African wild dogs *Lycaon pictus* (Mills, 1982; Creel & Creel, 1996; Durant, 2000), therefore, conservation efforts outside of protected areas are especially important for the conservation of these species (Marker & Dickman, 2004; Hayward & Kerley, 2008). Many African protected areas are too small to maintain genetically viable carnivore populations (Bailey, 1993; Loveridge *et al.*, 2001). An estimated 75% of the cheetah species range within southern Africa is outside of protected areas (IUCN/SSC, 2007b), and ca. 90% of leopard *Panthera pardus* habitat and 50% of lion habitat is outside of the protected area network (Martin & de Meulenaar, 1988; Chardonnet, 2002). Consequently, the establishment of protected areas has failed to halt declines in carnivore abundance (Winterbach *et al.*, 2012).

As the human population increases and with declining budgets and financial resources, nationally protected areas are unlikely to increase further (Cousins *et al.*, 2010). In conjunction with poor management and insufficient protection (Krug, 2001; Bauer *et al.*, 2003), it is therefore, widely recognised that conservation within protected areas alone will be insufficient for many African species (Rondinini *et al.*, 2004; IUCN/SSC, 2007b). Any increase in habitat protection for wildlife will need to be derived from private or communal lands (Muir-Leresche & Nelson, 2000), and will require sufficient incentives to justify its occupants tolerating potential conflicts with wildlife over resources (Mbaiwa, 2009). Without the support of people living near protected areas the long-term conservation of wildlife in southern Africa is thought to be unlikely to succeed (Thompson & Homewood, 2002).

1.3 Development of wildlife conservation in southern Africa, focusing on Botswana

Before European colonisation indigenous African communities coexisted with wildlife, utilising them for food, clothing, medicine and socio-cultural uses, in relation to traditional beliefs and practices (Sifuna, 2012). For example, leopard skins were worn as traditional clothing by chiefs, and many wild animals were seen as totems and perceived as sacred (Mbaiwa *et al.*, 2003; Sifuna, 2012). However, following colonisation many communities moved away from traditional lifestyles to commercial agriculture (Darkoh & Mbaiwa, 2002). In these areas land conversion to agriculture, uncontrolled hunting for meat and sport and persecution of wildlife for competing with cattle resulted in the decimation of wildlife populations (Bond *et al.*, 2004).

1.3.1 Establishment of nationally protected areas

In the 1890s, in response to the dramatic decline in wildlife abundance, governments in most southern African countries passed legislation which placed all wildlife under state control, banned all commercial and subsistence use of wildlife and designated the first nationally protected area (i.e. the Kruger National Park). Since then numerous nationally protected areas have been gazetted, most of which were formed between 1950 and 1980, with only small increases since (Krug, 2001). Botswana has designated a larger proportion of its land mass for biodiversity conservation (19.1%) than any other country in Africa (Mordi, 1989). The first national park in Botswana, the Gemsbok National Park, was established in 1932 (Campbell, 2004). It was under the management of South Africa, because Botswana did not have its own wildlife department until 1956 (Campbell, 2004). The Wildlife Department, renamed the Department of Wildlife and National Parks (DWNP) in 1967 (Campbell, 2004), designated a further five game reserves, one national park, five forest reserves and three educational parks as protected areas for wildlife between 1960 and 1985 (IUCN/UNEP, 1987; Child, 2009d). However, the parks were designed based on existing boundaries such as roads or rivers, with the aim of avoiding human settlements, rather than for the needs of wildlife (Campbell, 1973; 2004). As a result, until the late 1980s only the Chobe and Moremi game reserve had permanent water sources, meaning game had to migrate during dry periods through non-protected land, in order to find water and grazing (Mordi, 1989).

1.3.2 Development of the cattle farming industry and its impact on wildlife conservation

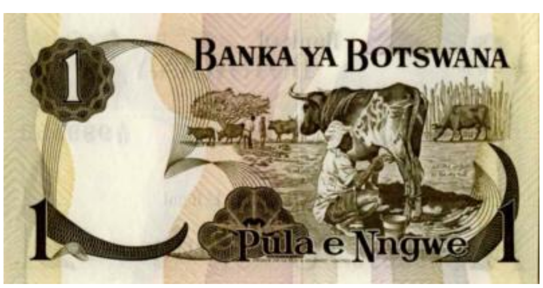
Most of the non-protected land in southern Africa was reserved for agriculture and livestock production. Subsidies to cattle producers and European Union investment in farming resulted in the intensive development of livestock farming across southern Africa, including Botswana.

After Botswana (known at the time as Bechuanaland) became a British protectorate in 1885, cattle production moved from largely subsistence farming to commercial agriculture. Beef was exported to South Africa to meet the increasing demands from the mining industry (Darkoh & Mbaiwa, 2002; Mbaiwa *et al.*, 2003), and during World War II Britain began importing Botswana beef, followed by the European Economic Community (EEC) in the 1950s. Government initiatives were designed to support the industry, including veterinary extension services, slaughter house subsidies, tax write-offs and veterinary cordon fences. As a result, the cost of cattle production was cumulatively subsidised by as much as 55% (Fidzani, 1985). In the 1970s Botswana gained preferential access to the beef market in the EEC, and through the Lome Convention the EEC agreed to buy a specified quantity of Botswana beef with reduced taxation. During the 1970s this enabled Botswana to sell beef at a price 31% greater than would have been gained on the world market (Darkoh & Mbaiwa, 2002). Consequently, the commercialisation of the livestock sector was promoted through the establishment of fenced ranches, and additional international and government subsidies enabled the installation of boreholes to tap ground water in previously dry habitat. This enabled cattle herds to extend into areas that were previously unsuitable for farming (Darkoh & Mbaiwa, 2002).

The number of cattle a family owned was a sign of social status and Botswana citizens viewed cattle as an indication of wealth and prestige; emphasised by pictures of cattle on the local currency (Fig. 1.1). Cattle had and still have a cultural significance, acting as insurance for the future, a dowry before marriage (known as lobola), inheritance and as a resource for family or community celebrations (Central Statistics Office, 2008).

At the time of independence in 1966, agriculture (primarily livestock farming) made up 40% of the country's gross domestic product (GDP), and Botswana was one of the poorest countries in the world (Central Statistics Office, 2008). With the discovery of diamond reserves in 1967 in Orapa (north-east), Lethalkane (north-east) and later Jwaneng (south-central) the mining industry overtook agriculture as the main contributor to GDP. In 2010 mining accounted for 30.3% of the total GDP, compared to 5.9% from tourism and 2.5% from agriculture (70 – 80% of which is livestock, mainly cattle [BEDIA, 2007]) (Central Statistics Office, 2013b). As Botswana diversifies its income, younger generations are relying less on agriculture, and the cultural significance of the cattle-post is diminishing (Child, 2009a; Central Statistics Office, 2012a). However, for rural populations livestock is still the main income, and the national cattle herd from 1983 to 2012 has fluctuated between 1.5 and 3.0 million. Despite evidence that the cattle herd is above carrying capacity and causing severe veldt (open grazing area) degradation (Mordi, 1989; Central Statistics Office, 2013a), the livestock industry is promoted as having a strategic role in increasing food security and reducing rural poverty (Central Statistics Office, 2012a).

The emphasis on cattle farming and the development of the livestock farming industry was similar throughout southern Africa. Through its development and western influences on culture, religion and the commercial value of wildlife products (e.g. ivory) the previous cultural value of wildlife across southern Africa diminished (Mbaiwa, 2002; Sifuna, 2012). Wildlife was neglected, hunted or perceived as competing with cattle for grazing and water, and both large herbivore species and predators were persecuted for the threat they posed to livestock (Bond *et al.*, 2004; Lindsey *et al.*, 2009b). Government campaigns to reduce disease transmission between wildlife and cattle included the culling of ungulate species in order to control tsetse fly *Glossina palpalis* populations (the vector for sleeping sickness, trypanosomiasis) (Matthiessen & Douthwaite, 1985) and the construction of veterinary fences to separate potential foot and mouth disease carrying wildlife



a. Botswana one pula note 1976

b. Botswana 200 pula note 2012

Figure 1.1. Botswana one pula and 200 pula notes from 1976 and 2012. Images from: (World Banknote Pictures, n.d.) and Money All Over the World (2012)

(primarily African buffalo *Syncerus caffer*) from cattle producing areas (Mbaiwa & Mbaiwa, 2006). In Botswana, in the 1980s, these fences prevented wildlife from following their normal migratory routes to water and grazing (Mbaiwa & Mbaiwa, 2006), resulting in mass die-offs of plains zebra *Equus quagga* and blue wildebeest *Connochaetes taurinus* (Owen & Owen, 1980; Ministry of Finance and Development Planning, 1985). The development of the cattle ranching industry and the subsequent neglect and persecution of wildlife populations meant that by the 1980s, the wildlife population had declined in both numbers and diversity, and over 90% of the large herbivore biomass on southern Africa's savannahs was livestock (Mordi, 1989; Cumming & Bond, 1991).

1.3.3 Response to wildlife losses in Botswana

In response to the wildlife losses the Botswana government drilled boreholes to provide permanent water in protected areas, restricted citizen hunting and introduced a nationwide annual aerial count of Botswana's wildlife in 1986 (DWNP, 2013). At the time of study, in 2012/2013, wildlife was recognised as crucial to Botswana's economic future, and pictures of wildlife had replaced those of cattle on the local currency (Fig. 1.1). Botswana invests more money into its protected areas per capita income than Namibia or Zimbabwe (Krug, 2001) and generally has a strong policy for the conservation of wildlife. Since 1989, wild herbivore populations have increased (with the exception of springbok *Antidorcas marsupialis* and tsessebe *Damaliscus lunatus*) (Central Statistics Office, 2013b). However, livestock continues to dominate, accounting for 83.2% of the large herbivore biomass of Botswana (Central Statistics Office, 2012a; DWNP, 2013).

1.4 The history of game ranching in southern Africa

In most countries in the world wildlife continues to be owned and regulated by the state in an attempt to ensure its protection (Muir-Leresche & Nelson, 2000). Some countries conduct Community Based Natural Resource Management programmes (CBNRM) to enable local communities to participate in wildlife management and to gain financially from wildlife utilisation (Hackel, 1999). However, overall the 'state owned' approach to wildlife protection has resulted in wildlife being restricted to protected areas, with little incentive for species conservation outside of these areas (Bond *et al.*, 2004). In an attempt to halt biodiversity loss, governments in southern

Africa passed unique legislation that returned the rights to utilise wildlife to landowners. Namibia (at the time known as South-West Africa) was the first country to pass legislation in 1967, which enabled landowners to utilise and profit from the wildlife on their property, if certain regulations regarding fencing and ranch size were met (Joubert, 1974). Wildlife species could be utilised for live sales, meat production, hunting or photographic tourism. Many farmers utilised wildlife in combination with livestock farming; the multiple revenue streams were more resilient to climatic and market changes and, consequently, were often more profitable (Kreuter & Workman, 1997). Between 1975 and 1992 governments in Zimbabwe (1975), Zambia (1985), Botswana (1992 & 2002) and South Africa (1960s; varied by state) passed similar legislation, giving wildlife a value and instigating the game ranching industry (McGranahan, 2008).

Game species are naturally adapted to the often harsh climate in semi-arid zones (ca. 70% of southern Africa [Bond *et al*., 2004]) and require less water, breed faster, mature earlier and cope better with poor grazing and natural diseases than cattle (Dasmann, 1964; Brown, 1969). Therefore, game ranching can be more productive (generating more meat per hectare or per kilogram of live-mass) (Dasmann & Mossman, 1961; Child, 1988) and in some cases more profitable than cattle farming, especially on large farms or in areas with low soil fertility and diverse wildlife communities (Dasmann, 1964; Hopcraft, 1970; Kreuter & Workman, 1997; Tomlinson *et al.*, 2002; Cloete *et al.*, 2007). At the same time cattle farming was becoming less profitable (because of reduced subsidies, land degradation and increased livestock theft), the demand for tourism was increasing, South Africa had re-emerged into the world community and the economic climate in Zimbabwe was uncertain. As a result, many cattle farmers switched to game ranching or incorporated wildlife onto their livestock farm (Kreuter & Workman, 1997; Cousins *et al.*, 2008; Child, 2009c; Lindsey *et al.*, 2013a).

South Africa and Namibia have the most developed game ranching industries; in 2013 game ranches in Namibia encompassed an area of 288,000 km^2 (Lindsey *et al.*, 2013c), and in South Africa in 2008 there were ca. 9000 ranches farming only game-stock, covering an area of 205,000 km^2, plus an additional 15,000 farms ranching game-stock in conjunction with livestock (Cousins *et al.*, 2008). The game ranching industries in Zimbabwe, Zambia and Botswana are smaller; game ranches covered an area of ca. 9000 km^2 in Botswana in 2011 (BWPA, 2011b) and ca. 6000 km^2 in Zambia in 2013 (Lindsey *et al.*, 2013a). The game ranching industry in Zimbabwe has virtually collapsed (Lindsey *et al.*, 2009a), however, in 2000, prior to the land seizures that occurred during the fast-track land reform programme, there were ca. 1000 ranches covering an area of 27,000 km^2 (Bond *et al.*, 2004).

1.5 Development of the game ranching industry in Botswana

The development of game ranching in Botswana was suppressed by the cattle industry (Child, 2009a). As a result, landowners were first granted the rights to utilise wildlife in 1992 (DWNP, 1992), much later than in Namibia, South Africa and Zimbabwe (1967 - 1985) (McGranahan, 2008). The subsequent Game Ranching Policy of 2002 promoted the private sector to develop and drive the industry, whilst the government aimed to facilitate its development (Republic of Botswana, 2002). In 1999 there were 17 registered game ranches (BWPA, 2005) and by 2012 this had increased to 111 ranches (BWPA secretary pers. comm.), with potential for further expansion (Rozemeijer, 2002).

Botswana has a diverse range of wildlife species and a strong tourism sector. Additionally, the climate and vegetation across most of Botswana is more suitable for game ranching than livestock farming or crop production. For example, 77% of the country is characterised by low rainfall (250 – 300 mm per year) and relatively unfertile sandveld soils, and overstocking of the national cattle herd above carrying capacity has contributed to bush encroachment and desertification of the land (Masilo *et al.*, 1999; Ringrose *et al.*, 2002; Moleele & Mainah, 2003). Therefore, the game ranching industry in Botswana has the potential to expand on both private land and in areas with CBNRM programmes (Bond *et al.*, 2004; Mbaiwa *et al.*, 2011; Chaminuka, 2013). The industry is considered by the government as an opportunity to create employment, diversify incomes and conserve biodiversity and habitat (Republic of Botswana, 2002).

1.6 The role of game ranching in biodiversity conservation

The game ranching industry has the potential to protect vast areas of land for habitat and species conservation if planned and managed on a healthy ecological basis (Bothma, 1989). However, the industry has largely developed in an unregulated way, by trial and error and with little ecological monitoring of its impacts (Bond *et al.*, 2004; Cousins *et al.*, 2008). Limited information is available to guide policy (Lindsey *et al.*, 2009b; Cousins *et al.*, 2010), and many ranches lack long-term ecological management plans (Bothma, 1989). Stakeholders stated that the industry's main contribution to conservation was in protecting habitat and in conserving species (Cousins *et al.*, 2008). However, combining the economics of game ranching with conservation objectives often proves difficult (Cousins *et al.*, 2008), and biodiversity conservation has been described as a side effect of the business, rather than the primary aim (Jones *et al.*, 2005).

The lack of regulation has allowed potentially environmentally damaging practices, such as the stocking of exotic species, unethical hunting and the breeding of rare colour variants to occur. As a result, the ecological sustainability of the industry is unclear (Cousins *et al.*, 2008). New game ranching regulations published in South Africa in 2008 (Cousins *et al.*, 2010) and under discussion in Botswana in 2011 have addressed many of these practices (DWNP, 2011b), however, the new regulations have been poorly received by farmers and are likely to be a challenge to implement (Cousins *et al.*, 2010). An overview of the industry's role in conservation is discussed in the following sections.

1.6.1 Habitat protection

The introduction of game ranching has increased the area of land available for wildlife production (compared to the area of nationally protected areas) by 1.1 times in Botswana (extrapolated from BWPA [2011b] and Central Statistics Office [2013a]), 1.9 times in Namibia and Zimbabwe and 4.1 times in South Africa (Lindsey *et al.*, 2009b). This additional wildlife habitat can provide connectivity between protected areas, thereby aiding the dispersal of free-ranging species (Lindberg *et al.*, 2003). However, to register as a game ranch in South Africa, Zambia and Botswana the perimeter of the property must be game fenced (17 strands, ca. 2.3 m high to contain ungulate species that jump), whilst in Namibia the property must be cattle fenced (generally 5 strands ca. 1.5 m high) (Bond *et al.*, 2004; McGranahan, 2008). Fencing establishes ownership and enables the reintroduction of valuable species, and their containment from neighbouring communities (Gusset *et al.*, 2008b). However, the erection of a fence effectively splits the land into small fragmented pockets. Wildlife species are unable to move to new areas to respond to changing environmental conditions, therefore, the carrying capacity and ecological resilience of the land is reduced (Boone & Hobbs, 2004; Cousins *et al.*, 2008; Lindsey *et al.*, 2009c).

This, ultimately, reduces both the number and density of organisms that can be supported in a landscape (Woodroffe *et al.*, 2014), and fencing has been associated with herbivore declines (Owen & Owen, 1980) which can lead to detrimental changes to vegetation diversity (Cassidy *et al.*, 2013). Fences, especially electric fences, prevent access to key resources, disrupt the immigration and emigration of wildlife and potentially result in the injury and death of animals that become caught in the fence (Harrington & Conover, 2006; Cousins *et al.*, 2008; Gadd, 2012; Lindsey *et al.*, 2012d). They are also linked to increased incidence of poaching, supplying materials for snaring (Lindsey, 2010). Additionally, fenced areas, especially smaller farms, require more intensive management to control wildlife populations and prevent inbreeding (Cousins *et al.*, 2008). Open game ranching systems exist in Zimbabwe; however, there were concerns that open systems can encourage ranchers to consumptively utilise game-stock unsustainably, in order to profit from the animals before their neighbours do (Kinyua *et al.*, 2000). Cooperative management agreements between neighbours may reduce this risk.

1.6.2 Veldt management

Game species evolved within their natural environment to exhibit various grazing and browsing strategies, therefore, they utilise a broader range of vegetation than cattle do alone (Walker, 1976; Taylor & Walker, 1978). The ranching of game-stock can protect plant biodiversity (Langholz & Kerley, 2006), makes the farm more adaptable to climate change (Barnes *et al.*, 2012) and can reverse damage on livestock farms caused by poor veldt management and bush encroachment (Bond *et al.*, 2004; Child, 2009b). Namibian farms ranching game-stock and livestock were less bush encroached than ranches farming only cattle (McGranahan, 2008). However, if farms are overstocked, particularly a problem on small game ranches with an available water source or ranches involved in game meat production, this can result in similar damage to the veldt as overstocking with cattle (Bond *et al.*, 2004; Bothma *et al.*, 2009). In a survey of KwaZulu-Natal 50.0% of the game ranches were overstocked (Lindberg *et al.*, 2003). Other ranches engaged in vegetation clearing programmes to create false savannah landscapes or to improve wildlife viewing without consideration of the overall ecosystem function (Sims-Castley *et al.*, 2005).

Figure 1.2. Golden wildebeest. Photo from Ryno Rare Game; http://rynoraregame.co.za/?page_id=127

1.6.3 Biodiversity conservation

Game ranches hold substantial ungulate populations; there were ca. 19 million animals on game ranches in South Africa in 2007 (Carruthers, 2008), ca. 90,000 animals on game ranches in Zambia in 2012, ca. 100,000 animals on game ranches in Botswana in 2011 (extrapolated from DWNP [2013] and BWPA [2011b]) and by 2013 wildlife numbers on Namibian private farms were greater than in the country's protected areas (Lindsey *et al.*, 2013c). Additionally, game ranching has been associated with the reintroduction, conservation and meta-population management of many threatened species including white rhinoceros (of which in 2012 there were more rhinoceros on private land in South Africa than in the rest of Africa together [Emsile, 2012]), mountain zebra *Equus zebra*, black wildebeest *Connochaetes gnou* and bontebok and the natural colonisation of species such as martial eagle *Polemaetus bellicosus* (Flack, 2003; Machange *et al.*, 2005).

The protection of habitat for ungulate populations is, consequently, associated with the overall protection of the region's biological diversity, and it has been claimed that the privatisation of wildlife has been more successful than any other policy, in conserving biodiversity in Africa (Muir-Leresche & Nelson, 2000). However, not everyone agrees with this claim as stocked ungulates are effectively a captive population and their management is often based on maximum profits rather than on long-term genetic fitness and sustainability (Cousins *et al.*, 2008; Frankham, 2009). The hunting and tourism market often drives game ranchers to stock small numbers of multiple species, based on those species that are fashionable (Cousins *et al.*, 2008). This has led to the under-representation of less marketable ecosystems. Some ranches have introduced invasive European species, such as wild boar S*us scrofa* and fallow deer *Dama dama*. Additionally, species endemic to southern Africa such as southern lechwe *Kobus leche*, black wildebeest and blesbok *Damaliscus pygargus phillipsi* have been introduced to ranches outside of their historical range, with little consideration of their habitat needs or species genetics (Hamman *et al.*, 2003; Lindsey *et al.*, 2006; Lindsey *et al.*, 2009b; Maciejewski & Kerley, 2014). Some ranchers have also selectively bred species to express rare colour phenotypes, for example 'white' varieties of springbok or 'golden' varieties of blue wildebeest (Fig. 1.2). Introduced exotic species or rare colour variants often exhibit low breeding rates and decreased survival, which can be mistakenly blamed upon carnivore depredation (Bothma, 2005). Introduced species can displace native species, degrade the habitat and in the case of closely related species stocked together can hybridise, thereby reducing genetic fitness (such as blue- and black wildebeest) (Bothma, 2005; Cousins *et al.*, 2010). If exotic species or colour variants were released or escaped outside of the game ranch they could endanger wild populations, by introducing foreign pathogens and parasites, or through genetic pollution (Bothma, 2005; Cousins *et al.*, 2008). South Africa (in 2008) and Botswana (drafted 2011) have introduced regulations to prevent the stocking of exotic species, however, it is unclear how this policy will be enforced on those farms where they are already stocked (Cousins *et al.*, 2010; DWNP, 2011b).

Overall, the industry is driven by capricious government policies and landowners, and the potentially fickle tourism and hunting trades, therefore, the conservation priorities and direction of the industry can change rapidly (Lindberg *et al.*, 2003). Changes in government policy can reduce the incentives for game ranching making it less economically viable and potentially promoting the conversion of ranch land to other land uses, including a return to livestock farming (Krug, 2001; Jones *et al.*, 2005; Cousins *et al.*, 2008). A decline in registered game ranches in Zambia in 2013 is thought to be related to inappropriate legislation and a lack of government support (Lindsey *et al.*, 2013a). This instability of the industry and its objectives can reduce the long-term conservation potential of game ranches. For example, increased poaching, security costs and a reduction in the perceived incentives for owning white rhinoceros has resulted in an increasing number of owners seeking to sell their animals (Emsile, 2012). This trend threatens to reverse the recent range expansion and meta-population growth of the species (Emsile, 2012).

Conflict between game ranchers and free-ranging herbivore species, particularly African elephants *Loxodonta africana,* has also been reported. Elephants can cause substantial damage to fences which can result in direct costs through the loss of wildlife and indirect costs through revenue spent on patrolling and repairing fence-lines. In Botswana game ranchers have applied for sustainable hunting quotas on 'problem' elephants to offset the costs of damage (BWPA, 2010a), and the hunting of 'problem' elephants in community trusts has been suggested in the nation's draft HWC 'Green Paper' (CAR, 2012). However, with the closure of hunting on state land in Botswana (MEWT, 2013), it is unlikely the hunting of problem elephants would be approved.

1.6.4 Predator conservation

Large carnivores have been introduced into many game ranches with 'predator-proof' fences largely for photographic tourism. By 2009 in South Africa, 22 ranches had reintroduced lions, 70 ranches had reintroduced cheetahs (L. Hunter & K. Marnewick pers. comm., cited in Lindsey *et al*. [2009b]) and 14 ranches had reintroduced African wild dogs (Gusset *et al.*, 2008b). It has been argued that fenced reserves are more economical and have better conservation potential for lions than unfenced protected areas (Packer *et al.*, 2013). However, many game ranches are too small to support viable predator populations (Packer *et al.*, 2013) and the maintenance of predators within 'predator-proof' reserves requires intensive management, to prevent the animals from inbreeding or escaping and to ensure game populations remain sustainable (Davies-Mostert *et al.*, 2009). To maintain genetic diversity, predators are often managed as part of a meta-population plan (Gusset *et al.*, 2008b), however, such management is expensive and consumes vast amounts of conservation resources (Lindsey *et al.*, 2005a). It is, therefore, recommended to avoid keeping predators on 'predator-proof' game ranches in favour of conservation strategies that will promote the self-regulation of populations and natural dispersal (Davies-Mostert *et al.*, 2009).

Game ranches without 'predator-proof' fencing allow predators and other small mammals to cross the fence using holes dug under the fence-line by digging species such as warthog *Phacochoerus africanus* (du Toit, 1996; Van Rooyen *et al.*, 1996) or by passing between the fence wires.

Therefore, game ranches have the potential to protect habitat and provide land connectivity for free-ranging predator populations (Bauer, 2008; Lindsey *et al.*, 2009c; Lindsey *et al.*, 2013c); however, this conservation potential is not always achieved. Unsustainable hunting quotas have been linked to lion and leopard population declines in Tanzania and Zimbabwe (Loveridge *et al.*, 2007; Packer *et al.*, 2011) and 'canned' or 'put and take' hunting of lions (defined by Lindsey *et al* [2009b] as when trophy animals are shot in enclosures with no chance of escape or when captive trophy animals are released onto a ranch shortly before the hunt) have brought the game ranching and hunting industries under scrutiny (Lindsey *et al.*, 2006; Lindsey *et al.*, 2012a; Nelson *et al.*, 2013). Similarly reports of the illegal capture and/or sale of predators in Botswana, often for the canned hunting trade, have brought criticism to the industry (BWPA, 2006a; Lindsey *et al.*, 2009c; Macleod *et al.*, 2013).

However, the greatest threat to predator conservation on game ranches is conflict with landowners due to depredation of game-stock on the ranch (Lindsey *et al.*, 2013c). Research in Namibia, Zimbabwe and South Africa suggests that game ranchers are more tolerant towards predators than livestock farmers, especially when ranchers utilise high income, low off-take activities such as trophy hunting or photographic tourism (Lindsey *et al.*, 2005c; Swanepoel, 2008; Thorn *et al.*, 2012; Lindsey *et al.*, 2013b). However, for maximum profitable off-take of game-stock for live sales or hunting for meat, game-stock populations need to be larger than carnivore depredation would naturally allow. In these cases predators are seen as directly competing with ranch owners for game-stock animals (Lindsey *et al.*, 2005c). As the game species are non-domesticated, free-ranging within the ranch and the predators' natural prey, these losses can be substantial and difficult to prevent (Marker *et al.*, 2003a; Winterbach *et al.*, 2012; Lindsey *et al.*, 2013b).

1.7 Introduction to human-wildlife conflict

Human-wildlife conflict is defined as 'when the needs and behaviour of wildlife impacts negatively on the goals of humans, or when the goals of humans impact negatively on the needs of wildlife' (Recommendation 5.20, [World Parks Congress, 2003]). Due to the increasing human population and the resulting conversion of land to anthropic activities, HWC is expected to increase worldwide, impacting on the long-term survival of many threatened species (Hutton & Leader-

Williams, 2003; Madden, 2004). Conflict occurs due to competition over resources such as grazing (Amano *et al.*, 2007), loss of human-life (Löe & Röskaft, 2004), potential disease transmission between wildlife and domesticated livestock or humans (Osofsky, 2005) or due to damage to crops (Hill & Wallace, 2012; Hoare, 2012), property (Enari & Suzuki, 2010), fish stocks (Schakner & Blumstein, 2013), livestock or game-stock (Woodroffe *et al.*, 2005d). Globally, the most commonly reported cause for HPC is carnivore depredation on livestock (Sillero-Zubiri & Laurenson, 2001). HPC occurs with species ranging from kea *Nestor notabilis*, a New Zealand alpine parrot that mostly feeds on berries and shoots, but occasionally feeds on the body fat of sheep *Ovis aries* (Birdlife International, 2012), to lions in Africa which can pose a threat to both stocked animals and human-life (Macdonald & Sillero-Zubiri, 2002). An estimated 75% of felid species and the majority of canid species worldwide are involved in HPC (Sillero-Zubiri & Stwizer, 2004; Inskip & Zimmerman, 2009), and in conjunction with habitat loss it is the most important identified threat facing African carnivores (Ray *et al.*, 2005).

The intensity of HPC and an individual's ability to cope with this conflict can vary with geography, society, culture, history, economics and politics (Osborn & Parker, 2003) and in some cases can impose substantial costs to both humans and wildlife. Generally, the direct and indirect costs of conserving large carnivores are felt by the local communities and landowners, whilst the benefits of large carnivore conservation, through tourism or existence values, materialise at the national or international level (Norton-Griffiths & Southey, 1995; Rasker & Hackman, 1996; Messmer, 2000; Dickman *et al.*, 2010).

1.7.1 Direct costs of human-predator conflict to humans

The direct costs of HPC are the number of livestock and game-stock lost to predators and the financial implications of this loss. These costs impact not only on individual farmers, but also on the cattle and game ranching industries and on local and national government through expenditure on conflict mitigation programmes (Swenson & Andren, 2005; CAR, 2011). The number of animals predated upon by carnivores can vary substantially with farm location, livestock husbandry, species stocked and the predators present. At a national level livestock and game-stock losses are generally low, however, 'hot-spots' of HPC can exist and financial losses from carnivore depredation on stocked animals for individual households within these 'hot-spots' can be substantial (Oli *et al.*, 1994; Mishra, 1997).

Habitat for many large African carnivore species overlaps with some of the world's poorest human communities, who are often reliant on agriculture for subsistence living (Dickman *et al.*, 2010; Central Statistics Office, 2012a). For these subsistence farmers the loss of even one livestock animal can have a major impact upon the household's income (Oli *et al.*, 1994; Mishra, 1997). Commercial farms are likely to have more resilience to carnivore depredation than subsistence farms, but even low predation rates can result in large economic losses if valuable stock such as stud cattle or expensive game-stock species are killed (Thorn *et al.*, 2012). Previous studies have estimated livestock losses to predators on farms in southern Africa to be between 0.8 and 5.0% of a farm's cattle herd per year (Butler, 2000; Marker *et al.*, 2003d; Schiess-meier *et al.*, 2007; Thorn *et al.*, 2012). The number of game-stock animals killed and the financial costs have the potential to be greater, and the Botswana Wildlife Producers Association (BWPA) estimate 50% of the game-stock calf crop will be lost to predation in the first year if predators are present (BWPA, 2005). This could be financially unsustainable, especially on new farms that are trying to establish their breeding herds and increase game numbers to carrying capacity (Bothma, 1989). The presence of game-fencing can also alter predators' natural feeding patterns; African wild dogs, in particular, are reported to deliberately chase animals into fences. This enables them to kill healthier and larger animals and species, and may also result in damage to fences or the escape of game-stock animals through the fence, increasing the associated economic losses (Van Dyk & Slotow, 2003; Rhodes & Rhodes, 2004; Davies-Mostert *et al.*, 2013).

1.7.2 Indirect costs of human-predator conflict to humans

The indirect costs of carnivore depredation relate to financial costs, social and opportunity costs, anxiety and stress (Barua *et al.*, 2013). Financial losses include revenue lost from by-products from the killed animal, including wool, milk or future offspring (Mertens & Promberger, 2001) or expenses related to the costs of materials, staffing and maintenance to introduce management practices such as kraaling (enclosing livestock within a fenced area) and 'predator-proof' fencing (Dickman, 2008). Kraaling has been linked to increased risks of disease or reduced growth rates of calves, thereby potentially causing financial losses as a result of decreases in livestock production or adding additional costs to prevent these losses (Ogada *et al.*, 2003). Overall, the indirect financial costs can be substantial and may, in some cases, be equal to or more than the direct costs (Taylor *et al.*, 1979).

Management strategies to reduce predation also take time to implement, time which would otherwise be spent on other aspects of farm management, work or leisure (Dickman, 2008). Predators and conflicting attitudes to predators can also impose social costs, often causing conflict between farmers, including peer pressure from neighbours to lethally control predator populations (Bezuidenhout, 2012). The anxiety and stress of living with predators either due to their financial costs or their impact upon freedom can affect health and quality of life (Bowie, 2009). Additionally, the conservation of predators can impose opportunity costs, preventing people from utilising resources inside protected areas or utilising the predator itself for cultural beliefs or financial gain (Norton-Griffiths & Southey, 1995; Infield, 2001; Goldman *et al.*, 2010).

1.7.3 Impact of human-predator conflict on wildlife

Colonial administrations have traditionally viewed predators as vermin and until the 1970s, many government bounty programmes encouraged the reduction of predator numbers in order to protect livestock or wild ungulates (Linnell *et al.*, 2001). A government bounty in New Zealand resulted in the death of at least 150,000 kea, until its protection in 1971 (Birdlife International, 2012). Similarly grey wolves *Canis lupus*, Eurasian lynx *Lynx lynx* and brown bears *Ursus arctos* were eliminated across Europe in the 18th and 19th centuries to protect livestock and farmed deer. Until the 1970s large carnivores were killed in order to protect moose *Alces americanus* in protected areas in the United States of America (USA) and African game species in numerous East African and southern African national parks (Woodroffe *et al.*, 1997; Breitenmoser, 1998; Linnell *et al.*, 2001; Schwartz *et al.*, 2003). For many people this view of predators as vermin or pests has remained unchanged (Sillero-Zubiri & Laurenson, 2001), and the conservation of predators outside of national parks and reserves is rarely supported by farming communities (Selebatso *et al.*, 2008). On non-protected land and in some protected areas, human-mediated death is the greatest cause of mortality and is one of the primary threats to the survival of nearly all large carnivore species (Ogada *et al.*, 2003; Woodroffe & Frank, 2005; Woodroffe *et al.*, 2005c).

Advanced weaponry and its increased availability have made lethal control easier and more efficient than it was in the past (Frank *et al.*, 2005). In recent history HPC has been associated with the extinction of three tiger subspecies; *Panthera tigris balica* went extinct in the 1940s, and *P.t virgata* and *P.t. sondaica* in the 1970s (Seidensticker, 1987; Nowell & Jackson, 1996). Additionally, the Zanzibar leopard (*P. pardus adersi*) was exterminated in the 1990s due to concerns that leopards were being used in witchcraft (Walsh & Goldman, 2007), and the Thylacine (Tasmanian wolf) *Thylacinus cynocephalus* became extinct in 1936, largely due to a government bounty programme (McKnight, 2008). Dramatic declines in population and range have also occurred in cheetahs, African wild dogs, pumas *Puma concolor*, tigers and lions (Nowell & Jackson, 1996; Woodroffe *et al.*, 1997; Marker & Dickman, 2004).

Predators can be legally killed in order to protect livestock across southern Africa and can also be legally killed in order to protect game-stock in South Africa and Namibia (MET, 2009; Cousins *et al.*, 2010). Lethal control is immediately satisfying to the farmer, convenient and often effective, at least temporarily, at reducing stock losses (Swarner, 2004). Usually it is a minority of predators that repeatedly prey on livestock, termed ‘problem’ animals (Stander, 1990; Mizutani, 1999). Cheetahs that habitually kill livestock often show physical or behavioural abnormalities and there is a general assumption that these animals are weaker or less dominant than other predators (Marker *et al.*, 2003c). The removal of a specific ‘problem’ animal can reduce stock losses, allowing farmers to coexist with the remaining predators (Linnell, 2011).

However, farmers’ definitions of a ‘problem’ animal often vary, and some farmers may consider all predators as a problem. In a review of HPC, Linnell *et al.* (1999) found no evidence that dispersing juveniles and infirm adults kill more livestock than adults in prime condition. Predators are opportunistic, and Linnell *et al.* (1999) argues that all predators have the potential to be a ‘problem’ animal. If specific ‘problem’ individuals do not exist (Linnell, 2011), preventative indiscriminate lethal control could be considered the best way to control carnivore depredation. On some farms all predator tracks are followed with the aim of killing the predator regardless of any recent losses (pers. obs.). The persecution of African wild dogs in particular is rarely in direct response to livestock losses (Gusset *et al.*, 2009), and cheetah marking-trees are commonly used to capture and remove cheetahs indiscriminately in Namibia (Marker *et al.*, 2003d) and Botswana (pers. obs.).

The definition of ‘problem’ animals on game ranches is more difficult as predators can be labelled as a ‘problem’ for consuming their natural prey, and the removal of multiple predators is likely to be necessary before a reduction of game-stock losses is visible (Massei *et al.*, 2010). However, lethal control is often only a short-term solution; if the wrong individual or species is targeted losses will continue and could increase as predators from surrounding territories move in to fill open territorial vacancies. This can influence a species’ social dynamics over a large area, and the killing of predators on boundaries of national park and reserves is associated with reductions in predator populations within the protected area (Woodroffe & Ginsberg, 1998; Castley *et al.*, 2002; Loveridge *et al.*, 2007; Balme *et al.*, 2010). The removal of individuals from social species, such as African wild dogs, lions or male cheetah coalitions, can potentially reduce hunting success and survival of the remaining individuals (Marker-Kraus *et al.*, 1996). Additionally, the lethal control of territorial males in species which practise infanticide, such as leopards (Balme & Hunter, 2013) and lions (Estes, 1992), will result in new individuals moving into the area, potentially killing the last male’s offspring, thereby affecting the predator population on a much wider scale (Loveridge *et al.*, 2007).

The lethal control of predators can also affect many non-target individuals or species; the use of poison bait or snares can result in indiscriminate deaths, whilst innocent species such as aardwolf *Proteles cristata* or bat-eared foxes *Otocyon megalotis* are sometimes killed when wrongly associated with livestock losses (Mills & Hofer, 1998; Treves & Naughton-Treves, 2005; Mateo-Tomás *et al.*, 2012). Additionally, predators play a keystone role in ecosystems, and their removal has been linked to herbivore and meso-predator release, which disrupts food webs, herbivore communities, and vegetation and consequently can have wide-ranging effects on ecosystems (Crooks & Soulé, 1999; Schmitz *et al.*, 2000; Ripple & Beschta, 2004; Ritchie & Johnson, 2009; Estes *et al.*, 2011).

1.8 Human-wildlife conflict in Botswana

Historically, Botswana has relied heavily on agriculture for the nation's income and for poverty reduction (Darkoh & Mbaiwa, 2002). Livestock have both economic and cultural significance to the citizens of Botswana (Central Statistics Office, 2008). Coupled with a relatively large area of unfenced land for wildlife conservation (19.1% of land mass is protected areas), there are vast opportunities for HWC.

National data on HWC is derived from applications to the government for compensation for livestock and property damage cause by wildlife species. Compensation is provided for damage caused only by lions, leopards, cheetahs, crocodiles *Crocodylus niloticus*, elephants, rhinoceros, hippopotamus *Hippopotamus amphibius* and African buffalo (DWNP, 2012). Damage caused by species for which compensation is not received, such as jackals and hyaenas, are rarely reported (Kent, 2011) or may be wrongly blamed on other species in order to receive compensation. Losses should be verified by a DWNP officer, however, this does not always occur and in some cases farmers may over-report losses. Game-stock species killed on game ranches are not eligible for compensation and, therefore, are rarely reported (DWNP, 1992); nor do all farmers claim compensation (Schiess-meier *et al.*, 2007; Kent, 2011). As a result, the estimated number of HWC incidents in national records and their potential impact on Botswana's farmers is likely to be inaccurate.

The intensity of conflict differs both spatially and seasonally. 'Hot-spots' of HWC occur in the Chobe, Ngamiland, Ghanzi and Kgalagadi districts (based on the number of 'problem' animal control (PAC) reports per resident), and the number of conflict incidents peaks during the wet season (CARACAL, 2009; CAR, 2011). Following worldwide trends, carnivore depredation on livestock is the most commonly reported form of HWC in Botswana (CAR, 2011). Nationwide it is estimated to cost farmers at ca. US $ 1,176,000 (United States dollars) per year and accounts for 75.4% of the total number of PAC reports recorded in a one year period (2010/2011, n = 5553) (CAR, 2011). During this period, damage to crops accounted for the second highest proportion of PAC reports (number of reports not specified) followed by damage to property (9.1% of reports) (CAR, 2011). Four human deaths and 22 injuries caused by wildlife occurred in this one year period (CAR, 2011; Central Statistics Office, 2013b). Elephants were the species which caused the most

PAC reports in 2010 – 2011 (26.3% of the total PAC reports) and during this period leopards (23.6% of predation reports), lions (24.1%) and African wild dogs (16.0%) caused the majority of reports of carnivore depredation on livestock (Fig. 1.3) (CAR, 2011). Carnivore depredation incidents on game-stock are rarely reported, hence data was not available on their frequency. However, the BWPA raised concerns in 2006 that depredation by cheetahs and African wild dogs was negatively affecting game ranches (BWPA, 2006b) and asserted that cheetahs 'probably cause the most economic losses on Botswana game ranches' (BWPA, 2005).

Although many farmers support predator conservation in general, this support does not always extend to farming areas (Selebatso *et al.*, 2008; Kent, 2011). Predators have no value to the majority of Botswana farmers and cattle are generally considered more important than wildlife (Mordi, 1989; Okavango Delta Management Plan, 2006). Retaliatory or preventative killing of predators occurs frequently, and the lethal control of 'problem' animals accounts for 59.2% of reported wildlife mortality, the remainder being due to road accidents (29.4%), poaching, fences

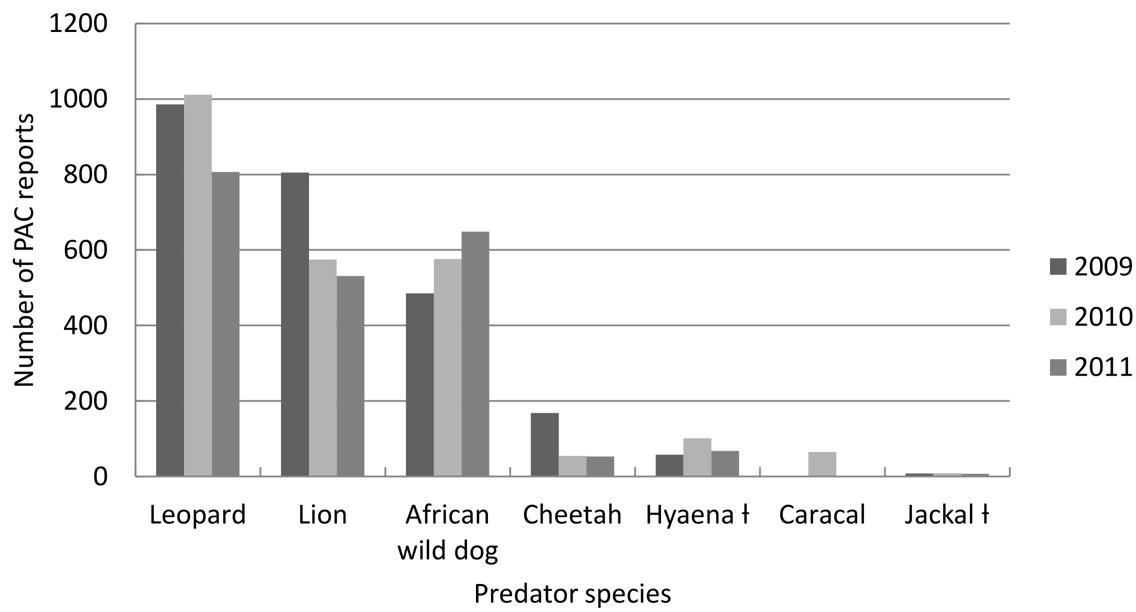


Figure 1.3. Carnivore species involved in 'problem' predator reports between 2009 and 2011, recorded by the Botswana Department of Wildlife and National Parks. Source: (Central Statistics Office, 2013b)

ł Records do not generally distinguish between brown hyaenas and spotted hyaenas or black-backed jackals and side-striped jackals or unknown factors (11.4%) (CAR, 2011; Central Statistics Office, 2013b). By law predators can be legally killed in Botswana to protect livestock, but not to protect game-stock (DWNP, 1992). Animals killed due to HWC or accidental death must be reported to the DWNP within seven days (DWNP, 1992). Approximately 160 leopards, 29 cheetahs, 30 lions and 31 African wild dogs are reported to have been killed per year, between 2004 and 2008 (Table 1.1). However, many deaths go unreported and official estimates are likely to be an underestimate of the true number of removals occurring.

The number of HWC cases reported to the Botswana DWNP doubled between 2000 and 2008 (CAR, 2011). Although changes in compensation policies may have altered reporting frequencies, the increase in HWC is primarily thought to be due to poor land use planning, an increase in human populations and activities, depletion of natural prey and abandonment of traditional livestock husbandry practices (Tjibae, 2001; CAR, 2011). A HWC 'Green Paper' (the first draft of a policy document) was submitted for discussion with stakeholders in 2012; it outlines the aim that people living with wildlife should be no worse off than other Botswana citizens (CAR, 2012). HPC in the game ranching industry was not originally mentioned in the phase one draft of the paper (CAR, 2011), but was subsequently added to the 2012 draft (CAR, 2012). The 'Green Paper' calls for changes in land use planning, improved CBNRM programmes, increased research and baseline data collection, changes to the compensation programme and the promotion of livestock husbandry techniques which reduce predation (CAR, 2012).

Table 1.1 Number of large predators of each species reported to have been killed during a five year period between 2004 and 2008. Source: (CAR, 2011)

Predator species	Number of predators killed 2004 - 2008	Mean per year
Leopard	806	161
Hyaena ł	180	36
African wild dog	156	31
Lion	151	30
Cheetah	146	29

ł Records do not generally distinguish between brown hyaenas and spotted hyaenas

1.9 The value of predators

Carnivores are valued for their ecological, cultural, financial, bequest and existence values (Dickman, 2008). Bequest values refer to passing an asset on to future generations and existence values refer to the importance that a species should survive (Attfield, 1998; Edwards & Abivardi, 1998). Financially, carnivores are valued for their extractive use as a food source, medicine or hunting trophy or for their non-consumptive use in tourism. Cultural values relate to carnivores' involvement in local customs and rituals (Dickman, 2008), such as the killing of a lion as a rite of passage for the Maasai in Tanzania (Kruuk, 2002). Predators can also be valued for their role in deterring poaching, with less poaching occurring on farms with resident lion populations (farming community member, pers. comm.).

However, the ecological value of predators is potentially the most important, but often the least recognised of the values. Carnivores can have a cascading effect on lower trophic levels and have an essential role in maintaining ecosystems (Terborgh *et al.*, 1999). Generally, large carnivores are among the first taxa to disappear from a site (Ray *et al.*, 2005), and by protecting habitat for top predators the habitat will in turn be conserved for other species associated with the ecosystem (Ripple & Beschta, 2004). As a result, they are often described as umbrella or flagship species. For example, in Namibia, lions only occurred in areas with populations of cheetahs, spotted hyaenas, brown hyaenas and leopards, therefore, the presence of lions is likely to indicate suitable habitat is available and has been conserved for all large predator species (Lindsey *et al.*, 2013b).

Predators limit the density of some non-migratory prey species; this can reduce foraging pressure on plant species, maintain vegetation biodiversity and buffer the impact of drought and climate change on herbivorous prey populations (Ripple & Beschta, 2004; Sala, 2006). Declines in large predator populations can result in herbivore release; the resulting increase in herbivore populations can disrupt food webs and herbivore and vegetation communities (Schmitz *et al.*, 2000; Ripple & Beschta, 2004). Predator declines have also been linked to increased intra-guild competition and aggression at lower trophic levels, potentially resulting in the domination of one particular species or group of species and the displacement of others (Terborgh *et al.*, 1999). Additionally, species that hunt prey by chasing, such as spotted hyaenas and African wild dogs, preferentially select sick and injured prey (Pole *et al.*, 2004; Hayward, 2006). This removal of weaker individuals may improve species' genetics and limits disease transmission in prey populations (Packer *et al.*, 2003).

The removal of large carnivores has also been associated with meso-predator release (Crooks & Soulé, 1999; Ritchie & Johnson, 2009). For example, the removal of lions and spotted hyaenas has been associated with high densities of cheetahs (Anderson, 1984; Kelly *et al.*, 1998). In some areas of South Africa, where cheetahs and leopards have been removed, black-backed jackals *Canis mesomelas* and caracals are the top predators (Klare *et al.*, 2010). Subsequently, on farms in the Northern Cape of South Africa where black-backed jackals and caracals are persecuted for killing smallstock, the numbers of bat-eared foxes, common genets *Genetta genetta* and cape foxes *Vulpes chama* have increased (Blaum *et al.*, 2009).

1.10 Importance of Botswana for predator conservation

With a low human population density and ca. 19.1% of land designated as protected areas Botswana provides a suitable habitat for mammalian carnivore populations (Central Statistics Office, 2012b). Within southern Africa, Botswana has the largest percentage of land where viable lion populations are thought to exist (Bauer, 2008), and Botswana was highlighted as the core region for the southern African cheetah population (IUCN/SSC, 2007b). The country is estimated to host 25.1% of the African wild dog, 23.6% of cheetah, 6.0% of lion, up to 52.7% of brown hyaena and up to 10.5% of spotted hyaena global populations (Hofer & Mills, 1998a; Durant *et al.*, 2008; Höner *et al.*, 2008; Winterbach, 2008; Woodroffe & Sillero-Zubiri, 2012; Riggio *et al.*, 2013). Therefore, Botswana is a vitally important area for predator conservation across the region and efforts to reduce HPC both within the cattle industry and the growing game ranching industry are essential for the long-term conservation of African large carnivore species.

1.11 Research questions

The distribution and abundance of African ungulate species and carnivore populations in southern Africa have declined substantially since European colonisation in the 17th century, due to land conversion to agriculture, uncontrolled hunting for meat and sport and persecution of wildlife for competing with livestock (Bond *et al.*, 2004; IUCN, 2013). The designation of protected areas have failed to halt these declines (Winterbach *et al.*, 2012), and the protected area network is considered inadequate to conserve many African mammal species (Fjeldsa *et al.*, 2004; Rondinini *et al.*, 2004). The contribution of private land to biodiversity conservation in southern Africa has increased since the introduction of game ranching in 1967 (Muir-Leresche & Nelson, 2000). The industry has played a role in the conservation of many threatened species and has been associated with increases in wildlife abundance (Flack, 2003). Game ranching has the potential to conserve vast tracts of land for mammalian predators (Lindsey *et al.*, 2009c; Lindsey *et al.*, 2013c). However, persecution by game ranchers due to carnivore depredation on game-stock has reduced this potential benefit (Lindsey *et al.*, 2013c), and HPC is one of the biggest threats to all large carnivore populations in Africa (Woodroffe *et al.*, 2005c).

Game ranching was introduced into Botswana in 1992 (DWNP, 1992). Despite the industry being smaller than Namibia or South Africa (Lindsey *et al.*, 2013a), it has increased rapidly. Game ranches in 2011 covered an area of ca. 9000 km^2 (BWPA, 2011b), and the industry has the potential to expand on Botswana's private and communal land. Botswana is at the core of the southern African cheetah population (IUCN/SSC, 2007b), the species believed to cause the largest economic losses to Botswana game ranchers (BWPA, 2005). The country also hosts a substantial proportion of the remaining global populations of the other African large carnivores, including lions, leopards and brown hyaenas. Consequently, the reduction of HPC within Botswana's growing game ranching industry and livestock farms would benefit the conservation of predator populations across the southern Africa region. However, few studies have examined HPC and potential mitigation methods to alleviate conflict on game ranches.

This study aims to determine the direct costs and drivers of conflict between game ranchers and predators in Botswana and to examine the use and perceived effectiveness of the available conflict mitigation methods. The impact of predator distribution and density and predators' prey preferences on HPC will be discussed. In order to achieve these aims the following questions and sub-questions were asked:

1. What are the direct costs and drivers of HPC on game ranches in Botswana, and how do they differ from commercial livestock farms?
    a. What factors do farmers consider as problems to the farm? How much of a problem is carnivore depredation in relation to these other problems?
    b. What are the number, proportion and financial costs of stock losses due to carnivore depredation? How does this vary with farm type? How do these losses compare with reported losses to other factors, such as disease?
    c. What factors (e.g., predators species present, farming techniques used, farm and farmer characteristics) are associated with stock losses to carnivore depredation?
    d. Is there a difference in the tolerance of predators between game ranchers and livestock farmers? What are the potential reasons for any differences?
    e. Does farmers' tolerance of predators differ between predator species? What are the potential reasons for any differences?
    f. What are the drivers of tolerance on game ranches and livestock farms? Are the direct costs of stock losses an important factor?

2. What is the distribution of large predators on commercial game ranches in Botswana, and how does the density of cheetahs, leopards and brown hyaenas vary between commercial livestock farms and game ranches in the Ghanzi farming block, in north-west Botswana?
    a. What is the distribution of large predators on commercial game ranches and nearby livestock farms in Botswana?
    b. What factors are significantly associated with the presence or absence of a predator species on a farm?
    c. What is the density of large predators on Ghanzi farmland?
    d. How does the calculated density compare with previous estimates?
    e. Is there a difference in predator densities between game ranches and livestock farms?
    f. Does the potential greater density of predators on game ranches cause social conflict between game ranchers and neighbouring livestock farmers?

3. Are farmers' perceptions of cheetahs' preferred prey on commercial farmland comparable to prey preferences observed by scat-analysis, and what is the proportion of livestock and game-stock in the cheetahs' diet, based on scat-analysis?
    a. What prey species do cheetahs consume on commercial farmland and what is the relative contribution of each species to the cheetahs' diet?
    b. What proportion of a cheetahs' diet on commercial farms is livestock and game-stock?
    c. Are remains of large prey species more commonly found in cheetah scat during a prey species peak birthing period?
    d. Do cheetahs consume prey species in relation to their relative abundance or do they show a preference or avoidance of certain species?
    e. How do cheetahs' prey preferences compare with farmers' perceptions of the cheetahs' diet?
    f. How does the estimated number of livestock or game-stock consumed by cheetahs from scat analysis compare with farmers' estimates of stock losses to cheetahs?
    g. Do farmers' perceptions of predators' primary prey items influence their tolerance of predators?

4. What mitigation methods are used by farmers to reduce conflict with predators, and how effective do farmers think these methods are at enabling humans and predators to coexist on commercial farmland?
    a. What methods are commercial farmers utilising to mitigate conflict with predators?
    b. If applicable, for what reasons do farmers state they have chosen not to use a mitigation method?
    c. Are certain factors, such as the predator presence, the number of stock killed and the location and characteristics of the farm associated with the use of livestock or game-stock management techniques?
    d. How effective do farmers think mitigation methods are at enabling farmers and predators to co-exist on farmland?

e. Does the utilisation of a mitigation method or a farmers' rating of its effectiveness correlate with tolerance score to predators?
f. How many farmers have lethally removed a predator species from the farm within the last 12 months, and what proportion of farms have used poison to kill predators?

5. How effective is predator translocation as a conflict mitigation tool in relation to the survival and post-release movements of translocated 'problem' cheetahs, and in relation to farmers' perceptions of the effectiveness of translocation at reducing stock losses to carnivores?
   a. How many predator translocations are conducted each year in Botswana?
   b. What is the survival of translocated 'problem' cheetahs?
   c. How effective do farmers think translocation is at reducing stock losses on the farm?
   d. Is predator translocation a cost effective solution to human-predator conflict?

1.12 Structure of the thesis

The thesis has been structured as nine chapters:

Chapter 1 provides an introduction to the history of wildlife conservation in southern Africa focusing on Botswana, including the limitations of protected areas, the impact of cattle ranching on biodiversity and the development of the game ranching industry. The potential role of game ranches in biodiversity conservation is discussed, including its impact upon predator populations. The topic of HWC is introduced, focussing on carnivore depredation of livestock and game-stock in Botswana. The importance of conserving large predators and Botswana's importance in predator conservation are introduced. The chapter ends with the research questions being posed.

Chapter 2 provides background to the study area. The location of Botswana, its history of establishment, climate, geography, vegetation, wildlife biodiversity, human population and land use are briefly described. The predators involved in HPC are introduced, and an overview of legislation in Botswana relevant to wildlife conservation and HWC is described. The status of the cattle, game ranching and the national wildlife industry, at the time of study (2012/2013), is explained. The chapter ends with a description of the three main game ranching regions: the Tuli-block farms, the Ghanzi farming block and the Hainaveld and Makalamabedi farms.

Chapter 3 provides an overview and background to the methodologies used is this study, namely spoor surveys, camera-trap surveys, scat-analysis, Global Positioning System (GPS) monitoring of translocated cheetahs and questionnaires.

Chapter 4 investigates the direct costs and the potential drivers of conflict between free-ranging predators and commercial farmers on livestock farms and game ranches, using questionnaires**.** It examines the relative importance of farmers' perceptions of predators as conflict drivers compared to the financial costs caused by carnivore depredation on livestock and game-stock.

Chapter 5 defines the distribution of large predators across Botswana's game ranches and calculates the density of large predators on the Ghanzi farming block using spoor surveys and camera-trap surveys.

Chapter 6 examines farmers' perceptions of the prey species most commonly killed by cheetahs on the Ghanzi farming block, compared to data from prey analysis of cheetah scat samples.

Chapter 7 describes the methods available to mitigate conflict between farmers and predators and their use in Botswana, based on questionnaire data. The effectiveness of these methods as perceived by the farming community and farmers' comments regarding the use of mitigation methods are examined.

Chapter 8 examines the effectiveness of translocating 'problem' predators as a conflict mitigation tool. Questionnaires are used to investigate how effective farmers thought translocation was at reducing stock losses, and GPS tracking collars are used to examine the survival and post-release movements of translocated 'problem' cheetahs.

Chapter 9 summarises the main results from the research, their limitations and their potential relevance to the wider field of HWC and predator ecology and conservation. Management recommendations and suggestions for further research will be made.

Chapter Two: Study area - Botswana commercial farmland

2.1 Chapter overview

The previous chapter provided a background to the game ranching industry, HPC and the importance of predator conservation, focusing on Botswana. This chapter will provide further information about the study area for this research, namely Botswana commercial farmland. It will give an overview of the country's location, climate, biodiversity (focusing on medium- to large-sized carnivores), wildlife conservation legislation, land use, human population and the status of the national wildlife, cattle farming and game ranching industries, at the time of study (2012/2013). Questionnaires were conducted as face-to-face interviews with game ranchers throughout Botswana. The primary game ranching regions, the Ghanzi farming block (also the site of data collection on cheetahs' prey preferences and predator densities), the Tuli-block and Hainaveld/Makalamabedi farms will be briefly described.

2.2 Location and brief history of establishment

Botswana is located at the centre of southern Africa; it is land-locked, sharing borders with Namibia, South Africa, Zambia and Zimbabwe (Fig. 2.1). It is situated between latitudes 17^{o} 45' and 27^{o} 45' south and longitudes 20^{o} 00' and 29^{o} 25' east (United Nations, 2012a). Never formally

Figure 2.1. Location of Botswana within southern Africa

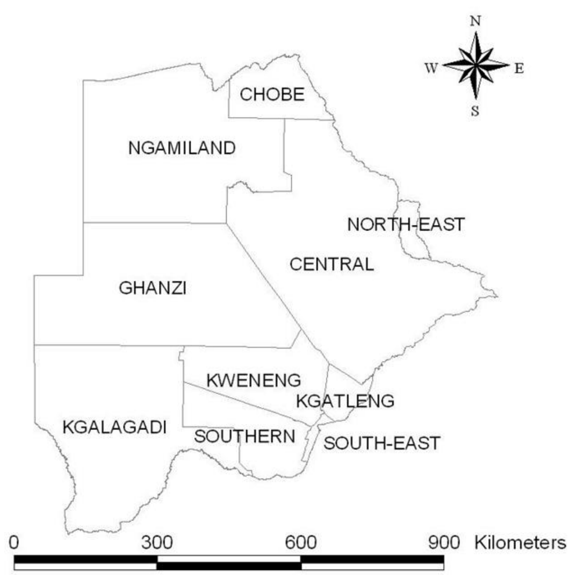


Figure 2.2. Regional districts of Botswana

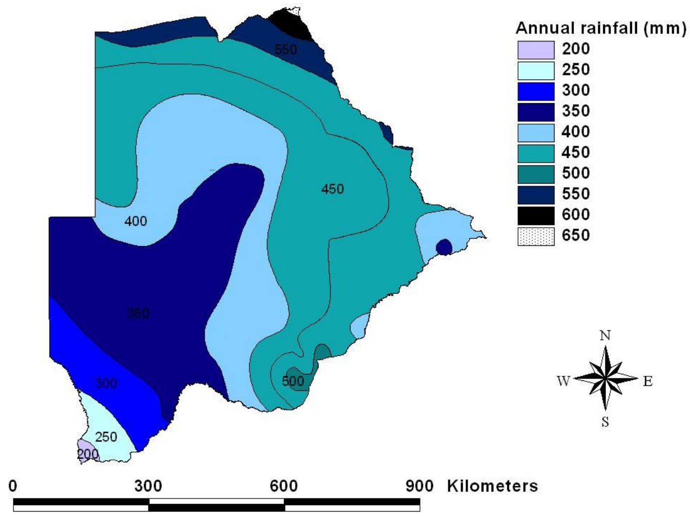


Figure 2.3. Annual rainfall distribution in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

colonised, Botswana became a British protectorate known as Bechuanaland in 1885, in response to appeals from Botswana's leaders for assistance and protection against a potential invasion by the South Africa republic (an independent country ruled by the Boer's republic, known as the Transvaal province within modern day South Africa) (Ramsay, 1996). Botswana became independent in September 1966 and Sir Seretse Khama became the first president; his son Lieutenant General Ian Khama was president at the time of study in 2012/2013 (BBC, 2014). Botswana has a democratically elected parliament and is known as one of the least corrupt and one of the most financially and politically stable countries in Africa (Robinson, 2009). It is divided into 10 regional districts: Central, Chobe, Ghanzi, Kgalagadi, Kgatleng, Kweneng, Ngamiland, North-east, Southern and South-east (Fig. 2.2) (Central Statistics Office, 2013b).

2.3 Climate and geography

Botswana has an area of 581,730 km^2 (similar in size to Texas or France), it is 900 m above sea level and predominantly flat (United Nations, 2011; 2012a). Unreliable rainfall occurs between October and March (the wet season), and the country is classified as semi-arid (Parida & Moalafhi, 2008). The north-east is the wettest region receiving more than 600 mm of rain per year, whilst areas in the south-west receive less than 250 mm of rain per year (Fig. 2.3) (Barnes, 2001).

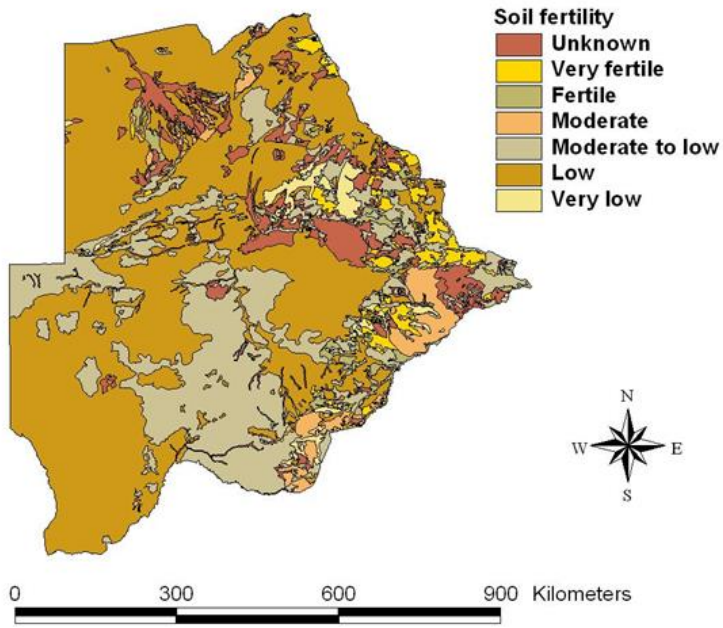


Figure 2.4. Soil fertility in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

Soil fertility in Botswana is generally low (Fig. 2.4); the soils can be broadly classified as hardveld, covering 23% of the land mass and sandveld, covering the remaining 77% of the country (Central Statistics Office, 2008). Hardveld soils are clay-based, moderately to very fertile, have a shallower water-table and generally experience greater rainfall (350 – 650 mm a year) than the sandy sub-soils of the sandveld. The sandveld has very low to moderately fertile soils with poor water retention (Fig. 2.4); in conjunction with the area's high evaporation rates and low rainfall (250 − 300 mm per year) it is classified as a desert, known as the Kalahari Desert (Central Statistics Office, 2008).

Temperatures range from 40°C in summer to below zero overnight in winter (Central Statistics Office, 2008). The country experiences regular periods of drought and flooding and was declared drought stricken for the 2012/2013 wet season (Central Statistics Office, 2013b). Natural pans and dry river beds fill with water during wet periods; however, the only natural permanent surface water is found in the Chobe and Okavango River systems (Fig. 2.5). The largest inland delta in the world, the Okavango delta, is located in the north-west of the country and is home to the majority

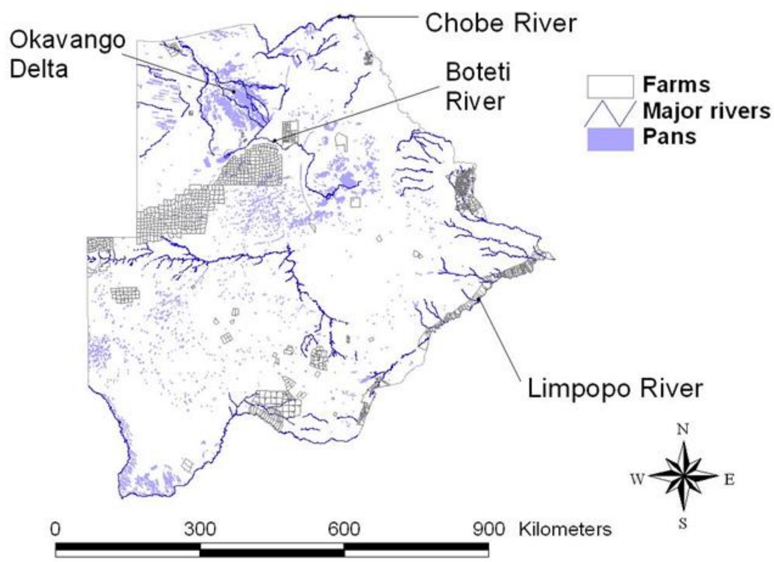


Figure 2.5. Major rivers and natural pans in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

of the country's biodiversity (Central Statistics Office, 2013b). The Botswana government applied for the Okavango delta to be classified as a UNESCO world heritage site in 2010 (UNESCO, 2010), but this status had not been granted at the time of writing, in 2014.

2.4 Vegetation

There are more than 2600 identified plants species in Botswana (UNDP, n.d.). The vegetation changes from scrub savannah and small trees in the drier south-west of the country to tree savannah and woodland as precipitation increases in the north-east (Burgess, 2003). The drier southern region is classified as sandveld vegetation (Fig. 2.6) and trees and shrubs are primarily *Acacia spp.*, predominantly camel thorn *Acacia erioloba,* Silver terminalia *Terminalia sericea* and Kalahari apple-leaf *Lonchocarpus nelsii* (Burgess, 2003); common grasses include *Aristida meridionalis*, *Digitaria eriantha, Cenchrus ciliaris* and *Anthephora pubescens* (Burgess, 2003). The vegetation in the wetter northern region is largely classified as mopani *Colophospermum mopane* dominated or miombo *Brachystegia spp*. (Fig. 2.6); additional species include broad-leaved tree species such as rain tree *Lonchocarpus capassa*, interspersed with *Acacia* spp. (Burgess, 2003). The Okavango delta wetland vegetation is dominated by paper reeds *Cyperus paprus* and common reeds *Phragmites australis* (Burgess, 2003). Trees and shrubs in the hardveld (Fig. 2.6) include

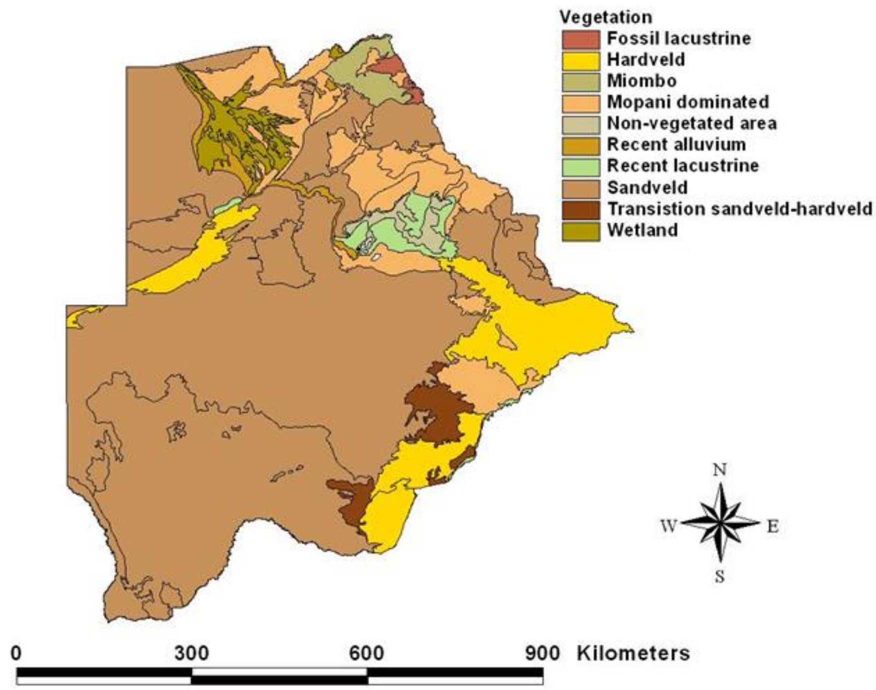


Figure 2.6. Vegetation types in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

camel thorn, trumpet thorn *Catophractes alexandri* and lowveld cluster leaf *Terminalia prunioides* (Burgess, 2003) (Fig. 2.6); common grasses are *Panicum maximum*, *Urochloa trichopus* and *Bothriochloa insculpta* (Burgess, 2003).

A large proportion of the livestock-producing areas suffer from bush encroachment (dominated by species such as black thorn *Acacia mellifera*) because of overgrazing in conjunction with climate change and a reduction in fires (Moleele *et al.*, 2002; Ringrose *et al.*, 2002; Moleele & Mainah, 2003). This has resulted in reductions in habitat and grazing quality for wildlife and livestock (Ringrose *et al.*, 2002). Several plants, mainly in the deep sandy soils, are poisonous to livestock, including *Urginea sanguinea* and *Pavetta harborii*. These plants produce green leafy material at the beginning of the wet season ahead of most palatable plants, potentially resulting in livestock deaths (Burgess, 2003).

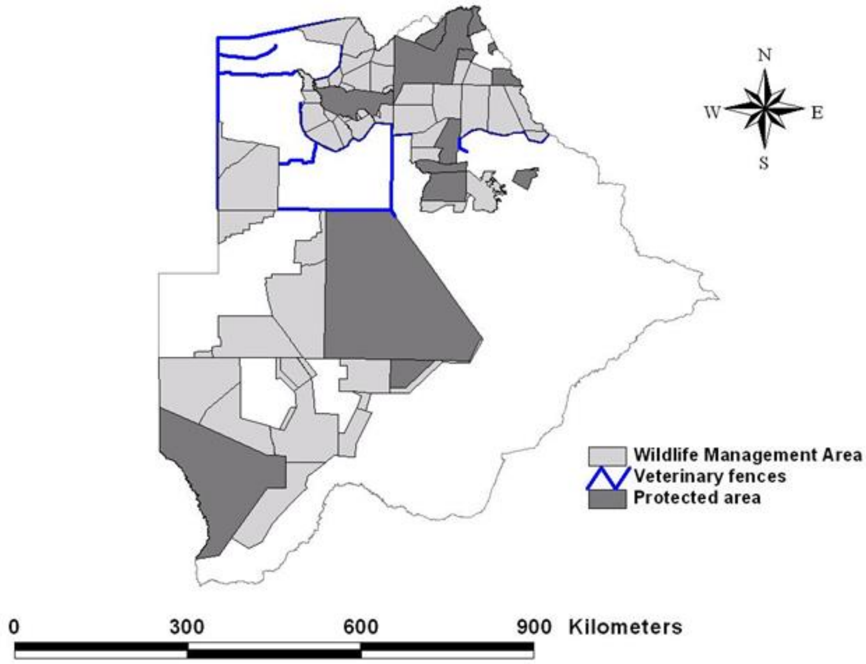


Figure.2.7. Location of veterinary fences in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

2.5 Wildlife biodiversity

The wetlands and woodlands of the north, including the Okavango delta and the Chobe River are home to a vast array of birds, mammals, fish, reptiles and amphibians; large mammals include elephants, African buffalo, southern lechwe and sitatunga *Tragelaphus spekii*. Wildlife biodiversity further south in the drier Kalahari Desert occurs at a lower density and is characterised by species adapted to desert conditions, including springbok, gemsbok *Oryx gazella* and red hartebeest *Alcelaphus buselaphus caama* (Hachileka, 2003). According to records from the UNDP (n.d.) there are 164 species of mammals, 157 species of reptiles, 38 species of amphibians, 80 species of fish and ca. 500 species of birds native to Botswana. An estimated 154 species are threatened with extinction, and 32 mammal species and numerous bird species (including nearly all birds of prey, excluding owls) are listed on the country's protected or partially protected species list (DWNP, 1992; UNDP, n.d.).

Veterinary cordon fences erected to control the spread of livestock diseases separate wildlife populations into a north-eastern (Ngamiland, Chobe and Central districts) and a south-western

Figure 2.8 Snares collected by the Department of Wildlife and National Parks in the Tuli-block region

system (Ghanzi, Kgalagadi, Southern and Kweneng districts; Fig. 2.7) (Crowe, 1995). The fences erected from 1954 onwards are considered one of the primary threats to wildlife (Mbaiwa & Mbaiwa, 2006). Other threats to the conservation of Botswana's biodiversity include poaching and the associated bush meat trade (Fig. 2.8), HWC, land degradation due to overgrazing, drought, human population growth and the resulting increase in human economic, social and infrastructure needs (Hachileka, 2003; Lindsey *et al.*, 2012b).

2.6 Botswana legislation relevant to wildlife conservation and human-predator conflict

The legislation relevant to wildlife conservation and HPC in Botswana are listed below:

- **Wildlife Conservation Policy** (1986; and the 2012 updated draft) aims to direct Botswana to sustainably utilise the full potential of its wildlife resources, with the establishment of Wildlife Management Areas (WMAs) and a focus on wildlife management by communities and private landowners (National Assembly, 1986; CAR, 2007; Botswana Press Agency, 2013).
- **National Conservation Strategy** (1990) and the **National Biodiversity Strategy and Action Plan** (2007) aim to maintain Botswana's biological heritage through the sustainable use and conservation of the country's biodiversity (National Assembly, 1990a; MEWT, 2007).
- **Tourism Policy** (1990) aims to limit tourism-related disturbances to the natural environment by promoting high-cost/low-volume tourism (National Assembly, 1990b).

- **Wildlife Conservation and National Parks Act** (1992) and its subsidiaries set the rules and regulations for all wildlife-related activities, including game ranching and the lethal control of predators due to HPC (DWNP, 1992).
- **Game Ranching Policy** (2002) in association with the 1992 Wildlife Act and the forthcoming game ranching regulations (drafted in 2011) promotes and governs the utilisation of game-stock species on private land, promoting biodiversity conservation and commercial viability (MTIWI, 2002; DWNP, 2011b).
- **Captive carnivore guidelines** (2011) direct the ownership of predators in captivity. Permission to own a captive predator is restricted to individuals with permits issued by the director of the Department or Wildlife and National Parks. 'Canned hunting' is illegal; however, 'put and take' hunting is permitted of partially protected or unprotected species (DWNP, 2011a).
- **National predator strategy** (2002 and draft version revised in 2008) focuses on ensuring the long-term conservation of large predators in Botswana in 'ways that gain enduring public support' (Winterbach, 2002; 2008).

The level of protection ascribed to specific wildlife species by these regulations is divided into three categories, defined in the Wildlife Conservation and National Parks Act (1992):
- **Protected species** cannot be hunted or captured by any person without permission from the director of the DWNP (predators are brown hyaenas, cheetahs, African wild dogs, servals *Leptailurus serval* and black-footed cats *Felis nigripes*).
- **Partially protected species** can be hunted or captured with a licence or permit issued by the DWNP (predators are lions and leopards; however, a moratorium on hunting permits was issued in 2007 for lions [Packer *et al.*, 2006] and in 2013 for leopards [BWPA, 2012a]).
- **Unprotected species** can be hunted under licence (predators are black-backed jackals, spotted hyaenas, caracals, [African] wildcats *Felis silvestris lybica* and side-striped jackals *Canis adustus*).

The allocation of species to these categories has not been updated since the inception of the policy in 1992, and more information is required to understand the status and trends of protected animals in Botswana (UNDP, 2007). Some species, for example lions and leopards, which are listed as globally threatened on the International Union for Conservation of Nature (IUCN) threatened species red list (IUCN, 2013), do not receive full protection in Botswana (UNDP, 2007).

However, most predator species (including protected species) can be killed if it has ‘caused, is causing or threatened to cause damage to any livestock, crops, water installation or fence’ (game-stock is not included) (DWNP, 1992). Restrictions were implemented in 2000 and formally added as a statutory instrument in 2005, which prohibited the killing of cheetahs for causing damage to livestock under any circumstances (DWNP, 2005a; Klein, 2007) and restricted the killing of lions to one member of the pride, if it can be proven the pride has killed livestock (DWNP, 2005b). However, these restrictions are difficult to police and are rarely enforced (Klein, 2007). Additionally, there are no penalties for accidently killing wildlife whilst driving a vehicle or in the protection of human-life (DWNP, 1992), and predators are sometimes killed by abusing these conditions (pers. obs.). Therefore, a predator’s protected status is largely limited to areas without livestock.

2.7 Botswana’s involvement in international environmental policies and conventions

Botswana as a member of the Southern African Development Community (SADC) upholds various regional environmental policies including SADC protocols on Wildlife Conservation and Law Enforcement, Fisheries, Forestry and Shared Water Courses (CAR, 2011). The former promotes the development of regional approaches to wildlife conservation including the establishment of protected areas which span across international borders (CAR, 2011). Additionally, Botswana is signatory to and has ratified all major international wildlife conventions and treaties (CAR, 2011), including the Convention on Biological Diversity (since 1992 [United Nation, 2014]), Convention on International Trade in Endangered Wild Fauna and Flora (since 1977 [CITES, 2014]) and the Convention on the Conservation of Migratory Species of Wild Animals (CMS, 2014). Botswana’s contributions and progress to adhering to these conventions is regularly reviewed by a committee and progress reports are submitted to the secretariat of the various conventions (officer at the Department of Environmental Affairs pers. comm.).

2.8 Predators involved in human-predator conflict in Botswana

There are 28 carnivore species present in Botswana, ranging in size from a common dwarf mongoose *Helogale parvula* at 0.21 - 0.35 kg to a lion weighing 150 - 260 kg (Kingdon, 1997; Mills & Hes, 1997). The mammalian predators causing the majority of HPC in Botswana are African wild dogs, brown hyaenas, caracals, cheetahs, black-backed jackals, leopards, lions and spotted hyaenas (Selebatso *et al.*, 2008; Gusset *et al.*, 2009; Central Statistics Office, 2013b). Crocodiles, baboons, snakes and large birds of prey are also reported as 'problem' predators to livestock (Waite & Philips, 1994; Butler, 2000; CAR, 2011; Wallace *et al.*, 2011), but are outside the scope of this study. A brief introduction to these mammalian predators and their conservation status follows; their distribution and abundance within Botswana will be discussed in Chapter 5.

2.8.1 African wild dog, Lycaon pictus

Ecology: African wild dogs sometimes referred to as the African painted wolf, occupy a range of habitats including short grass plains, semi-desert, bushy savannahs and upland forest (Woodroffe & Sillero-Zubiri, 2012). They are a pack animal; the pack consists of non-breeding adults and a dominant breeding pair of adults and their off-spring (on average 10 pups per litter) (Estes, 1992). They are largely diurnal, but will also hunt on moon-lit nights (Estes, 1992; Rasmussen & Macdonald, 2012). They are generalist feeders and mostly hunt medium-sized ungulate species by chasing and exhausting their prey; the principal prey species are impala *Aepyceros melampus*, greater kudu *Tragelaphus strepsiceros* (herein referred to as kudu), Thomson's gazelles *Eudorcas thomsonii* and blue wildebeest (Woodroffe & Sillero-Zubiri, 2012).

Conservation status: Botswana status: Protected

IUCN Red list status: Endangered Population trend: Decreasing

At the beginning of the 19th century African wild dogs occurred across sub-Saharan Africa, however, in 2005 the species was restricted to 10% of this historic range (Ray *et al.*, 2005). It has been largely extirpated in North and West Africa and populations are greatly reduced in north-east and Central Africa; the largest populations remain in southern Africa and the southern part of East Africa (Woodroffe & Sillero-Zubiri, 2012). The most recent population estimate is 6600, of which 1658 (25.1%) are estimated to occur in Botswana (Winterbach, 2008; Woodroffe & Sillero-Zubiri, 2012). Populations continue to decline due to habitat fragmentation, HPC, accidental death by humans due to road accidents or in snares and through infectious disease (Woodroffe & Sillero-Zubiri, 2012). African wild dogs are susceptible to rabies and canine distemper transmitted from domestic dogs *Canis lupus familiaris*, and disease management programmes form part of the conservation strategy in southern Africa (IUCN/SSC, 2007b). The species is exposed to human impacts more than the other large predator species (with the possible exception of cheetahs) due to its naturally low densities and wide-ranging habits (IUCN/SSC, 2007b). A region-wide conservation strategy for African wild dogs was implemented in southern and East Africa in 2007 (IUCN/SSC, 2007b; a), and action plans were in formation for North and West Africa at the time of writing, 2014 (Durant, n.d.). A Botswana national action plan was formalised in 2009 (DWNP, 2009).

2.8.2 Black-backed jackal, Canis mesomelas

Ecology: Black-backed jackals are small, relatively unspecialised canids, which occur in a wide range of habitats, including savannah, coastal desert, grassland, scrubland and woodland savannah (Loveridge & Nel, 2008). They are omnivorous, eating invertebrates, fruits and berries; they will actively scavenge and hunt small lambs of ungulate species, smallstock and other small vertebrates (Kaunda & Skinner, 2003; Klare *et al.*, 2010). They are mainly nocturnal but can also be active during the day. They generally occur in family groups of a monogamous pair and their offspring; two to six pups are born in one litter, and offspring from the previous litter often help with raising the young (Estes, 1992).

Conservation status:	Botswana status: Can be hunted under licence
IUCN Red list status: Least concern	Population trend: Stable

Black-backed jackals are endemic to Africa and occur widely throughout their historic range (Ray *et al.*, 2005); they are found in two sub-populations, one in East Africa and one in southern Africa (Loveridge & Nel, 2008). Despite persecution by farmers the species is not thought to be threatened and has no legal protection outside of protected areas (Loveridge & Nel, 2008). Attempts at population control are largely ineffective causing only temporary reductions in local numbers (Loveridge & Nel, 2008).

2.8.3 Brown hyaena, Hyaena brunnea

Ecology: Brown hyaenas are principally scavengers of vertebrate remains, supplemented by fruits, birds' eggs and insects (Wiesel *et al.*, 2008). Hunting, primarily of small prey, accounts for ca. 4.2% of their diet (Mills & Mills, 1978) and they are reported to rarely kill livestock (Maude & Mills, 2005). They are nocturnal and form clans of four to 14 hyaenas; they forage separately, but pups are raised in a communal den, with each female having one to four pups (Estes, 1992). They are found in desert areas, particularly along the Namibian coast, semi-desert, open scrub and open woodland savannah (Wiesel *et al.*, 2008).

Conservation status:	Botswana status: Protected
IUCN Red list status: Near threatened	Population trend: Decreasing

Brown hyaenas are endemic to southern Africa, occurring in Botswana, Namibia, Lesotho, Mozambique, South Africa and Zimbabwe (Wiesel *et al.*, 2008).They remain widespread and occur in ca. 72% of their historic range, as defined at the beginning of the 19th century (Ray *et al.*, 2005; Wiesel *et al.*, 2008). The total population was last estimated in 1998 as 5000 - 8000 individuals, of which Botswana has the largest population ca. 2636 individuals (33.0 – 52.7% of the global population) (Hofer & Mills, 1998a; Winterbach, 2008). The main threat to brown hyaena survival is HPC (Wiesel *et al.*, 2008); their conservation is guided by the Hyaena Conservation Status and Action Plan (Mills & Hofer, 1998).

2.8.4 Caracal, Caracal caracal

Ecology: Caracals are a medium-sized cat; found in a wide variety of habitats, including semi-desert, open savannah, scrubland and woodland (Stuart & Stuart, 2013). They prey on birds and small- to medium-sized mammals, including ungulate species up to ca. 50 kg (Stuart & Stuart, 2013), and they are also known to scavenge (Nowell & Jackson, 1996). Caracals are solitary, territorial and mainly nocturnal; they have one to four kittens per litter (Estes, 1992).

Conservation status:	Botswana status: Can be hunted under licence
IUCN Red list status: Least concern	Population trend: Unknown

Caracals are widely distributed; they are relatively common in southern and eastern Africa, threatened in northern Africa, and they are rare and declining in Central and West Africa and through most of the their range in south-west Asia (Nowell & Jackson, 1996; Stuart & Stuart, 2013). Their range is believed to have declined by 38% since the mid-nineteenth century; a population estimate is not available (Ray *et al.*, 2005). Habitat destruction is an important threat to populations in Asia and Central, West and North Africa (Sunquist & Sunquist, 2002; Ray *et al.*, 2005); in southern and East Africa HPC is the primary threat (Breitenmoser-Wursten *et al.*, 2008).

2.8.5 Cheetah, Acinonyx jubatus

Ecology: Cheetahs are the fastest land mammal (Sharp, 1997), reaching hunting speeds of 18.9 m/s^{-1} (Wilson *et al.*, 2013). They occur in open grasslands, dry forest, savannah woodland and scrubland (Durant *et al.*, 2008). They are largely crepuscular but can also be active at night (Grünewälder *et al.*, 2012). They prey on small- to medium-sized ungulate species favouring the most abundant prey species within a body mass of 23 - 56 kg (Hayward *et al.*, 2006). Females are solitary; males can be solitary or occur in coalitions of generally two to four individuals (normally brothers) (Caro, 1994). Cheetah densities in protected areas are generally limited due to interspecific competition and by the killing of cheetah cubs by lions (Caro, 1994; Laurenson, 1994). Consequently, in southern Africa the majority of the cheetah population occurs on farmland where lions and spotted hyaenas have largely been eradicated (IUCN/SSC, 2007b). However, cheetah cub survival was not limited by lions in the Kgalagadi Transfrontier Park (KTP) in Botswana (Mills & Mills, 2014). Cheetahs begin feeding on the hind-quarters of large prey animals and generally will not return to kills in order to avoid stronger competitors and scavengers which could be attracted to the carcass (Phillips, 1993; Caro, 1994).

Conservation status: Botswana status: Protected

IUCN Red list status: Vulnerable Population trend: Decreasing

At the beginning of the 19^{th} century cheetahs were present throughout Africa and Asia (IUCN/SSC, 2007b). As of 2005 they are estimated to remain in 23% of this historic range with strongholds in East and southern Africa (Ray *et al.*, 2005). The known cheetah population is 7500 adult individuals, of which 1768 are estimated to occur in Botswana (23.6% of population) (Durant *et al.*, 2008; Winterbach, 2008). The Asiatic subspecies *A.j. venaticus* and the North and West African subspecies *A.j. heckii* are critically endangered. The Asiatic population has declined due to loss of prey, development of land and the persecution and capture of wild cheetahs for captivity and hunting (Divyabhanusinh, 1995; Mallon, 2007); ca. 60 - 100 individuals remain in an isolated population in Iran (Hunter *et al.*, 2007b). The main threats to the survival of cheetah populations are habitat loss and fragmentation, HPC (the primary reason in southern Africa (IUCN/SSC, 2007b; a; Durant *et al.*, 2008), and a reduction of available prey (primarily a problem in North and East Africa). A region-wide conservation strategy for cheetahs was implemented for southern and East Africa in 2007 (IUCN/SSC, 2007b; a), and action plans were in formation for North and West Africa at the time of writing, in 2014 (Durant, n.d.). A Botswana national action plan was formalised in 2009 (DWNP, 2009).

2.8.6 Leopard, Panthera pardus

Ecology: After coyotes *Canis latrans*, leopards are considered the most adaptable predator in the world (Eaton, 1978). They have a wide habitat tolerance, occurring in habitats ranging from rainforest to desert (Henschel *et al.*, 2008) and consume a diverse range of prey, ranging from arthropods to large ungulates (Hunter *et al.*, 2013). Leopards will hunt or scavenge, are nocturnal, solitary and typically have one to three cubs in a litter (Estes, 1992).

Conservation status:	Botswana status: Partially protected
IUCN Red list status: Near threatened	Population trend: Decreasing

Leopards have a wide distribution across sub-Saharan Africa and Asia and occur in ca. 63% of their historic range in Africa, as defined at the beginning of the 19th century (Ray *et al.*, 2005). They are locally common in some areas, but patchily distributed throughout the rest of their range, with some sub-populations, for example in North Africa, threatened with extinction (Henschel *et al.*, 2008). There are no reliable population estimates for the whole species (Henschel *et al.*, 2008), but the Botswana population estimate is 5617 individuals (Winterbach, 2008). Leopards are highly adaptable and have been known to occur in areas with large human population densities (Hunter *et al.*, 2013). However, populations are under threat due to habitat conversion and HPC (Ray *et al.*, 2005).

2.8.7 Lion, Panthera leo

Ecology: Lions are found in a diverse range of habitats in sub-Saharan Africa, absent only from tropical rainforest and the Sahara desert (Nowell & Jackson, 1996). They form prides of four to six adults, consisting of a coalition of related or unrelated males, adult females and juveniles; typically three cubs are born in one litter (Estes, 1992; Bauer *et al.*, 2012). Lions typically prey upon medium- to large-sized ungulates, however, they will also eat carrion or hunt almost any animal from rodents to rhinoceros (Bauer *et al.*, 2012). They are nocturnal, territorial and often perform infanticide when a new male takes over the pride (Estes, 1992).

Conservation status: Botswana status: Partially protected

IUCN Red list status: Vulnerable Population trend: Decreasing

Lions formally ranged throughout Africa, south-west Asia and India (Nowell & Jackson, 1996; Sunquist & Sunquist, 2002). Lions occur in approximately 27% of their historic range in sub-Saharan Africa, as defined at the beginning of the 19th century (Ray *et al.*, 2005). The Asiatic, North Africa population *P.leo persica* exists as one isolated population of ca. 200 individuals in the Gir forest national park, in India (Nowell & Jackson, 1996; Bauer *et al.*, 2012). The West African lion population is considered critically endangered (Henschel *et al.*, 2014) and the southern and East African populations, where 77% of lion habitat remains, are patchily distributed. Seventeen areas within the species range were described as strongholds for lion conservation, two of which were within Botswana (Bauer, 2008).

The global lion population is reported to have declined by 30% since 1988 − 2008 (Bauer, 2008); at this rate of decline and with the increased pressure from the growing human population, lions are expected to be listed as endangered by 2030 (Cardillo *et al.*, 2004). The latest population estimate is 32,000 lions (Riggio *et al.*, 2013), of which 2918 are thought to occur in Botswana (6% of the worldwide population). The primary threats are habitat loss and fragmentation, the indiscriminate killing of lions in relation to HPC and a reduction in available prey (Bauer, 2008). Regional conservation strategies for lions in West and Central Africa and East and southern African have been developed (IUCN, 2006b; a).

2.8.8 Spotted hyaena, Crocuta crocuta

Ecology: Spotted hyaenas are nocturnal and occur in a wide range of habitats including semi-desert, savannah and open woodland (Hofer & East, 2013). In areas with numerous prey spotted hyaenas form clans numbering 35 to 80 adults, whilst in other areas the species is known primarily as a solitary forager (Estes, 1992). Individuals within spotted hyaena clans will hunt and scavenge together or singly; they hunt a variety of vertebrate prey and hunted prey can account for up to 95% of their diet (Estes, 1992; Cooper *et al.*, 1999). They have one to four cubs in a litter; the cubs stay at a communal den with other juveniles from the clan (Estes, 1992).

Conservation status: Botswana status: Can be hunted under licence

IUCN Red list status: Least Concern Population trend: Decreasing

Spotted hyaenas are widely but patchily distributed in sub-Saharan Africa, often restricted to protected areas, especially in West Africa (Höner *et al.*, 2008). In East and West Africa, population declines are occurring inside protected areas because of snaring and poisoning; southern African populations are thought to be stable (Hofer & Mills, 1998b). The global population was estimated in 2008 as between 27,000 and 47,000 individuals, with 2829 occurring in Botswana (6.0 – 10.5% of the total population). Spotted hyaenas are thought to occur in 73% of their historic range, as defined at the beginning of the 19th century (Ray *et al.*, 2005; Höner *et al.*, 2008; Winterbach, 2008). Major threats throughout their range are human persecution, snaring and a reduction in the available prey due to increased human settlement and overgrazing (Höner *et al.*, 2008).

2.9 Human population

Botswana has one of the lowest human population densities in the world; in 2011 the population was estimated at 2,024,904, with a population growth rate of 1.2% (Central Statistics Office, 2012b). The population density in towns is increasing; the country's two largest towns are Gaborone the capital city (population: 231,592) and Francistown (population 98,961) (Central

Statistics Office, 2012b). The majority of the population live in the more fertile eastern region (Fig. 2.9) (Central Statistics Office, 2012a).

Citizens of Botswana, referred to as Motswana, mainly descend from the Khoesan groups (the earliest inhabitants [Mbaiwa *et al*., 2003]), the Kalanga tribes and the Setswana speaking tribes (e.g., Bangwato and Batswana) (Mooko, 2006). There is also a large white Motswana population of Afrikaans or English descent, many of whom own farms in the commercial sector. The rural population of Botswana largely depend on agriculture for their livelihood and subsistence. There are numerous indigenous languages, however, the official language is English and the national language is Setswana (Mooko, 2006). Botswana has the second highest HIV prevalence in the world, with 23.4% of adults (15 – 49 years) testing HIV positive (UNAIDS, 2012). Despite the introduction of free anti-retroviral drugs, the high infection rate has had a major impact on Botswana's workforce and economy (Econsult, 2006).

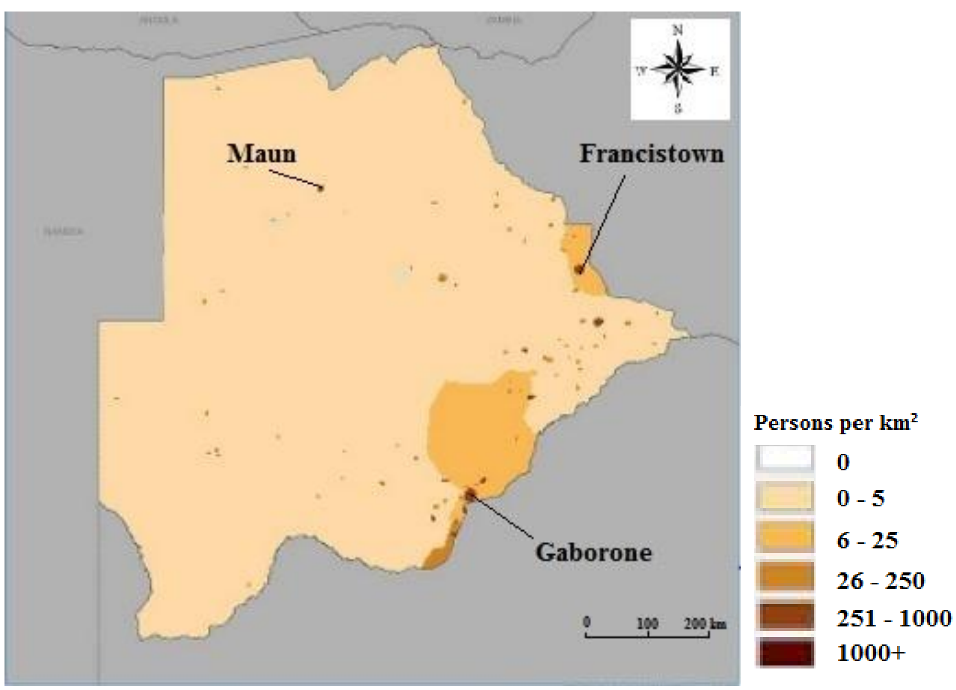


Figure 2.9. Human population density in Botswana. Source: CIESIN (2012)

2.10 Land use

The land in Botswana is divided into communal (54.8%), state (41.8%) and freehold land (3.4%; Table 2.1). Communal land consists of ranches and arable, residential and pastoral land and can be divided into three categories: reserved land is held by the state for future use; commercial land is leased to individuals or groups for farming and communal land is land in which every citizen, regardless of gender, is entitled to apply for a plot of land for agriculture or residential use (Central Statistics Office, 2008). The latter is allocated free-of-charge, must be used for the allocated purpose and cannot be sold, however, any improvements to the land (e.g. buildings) can be leased or sold (Adams *et al.*, 2003). State land is composed of WMAs, city land and protected areas (e.g., national parks, game reserves and forest reserves, Table 2.1; Fig. 2.10 and 2.11). Freehold land covers a much smaller proportion of land (3.4%) than found in neighbouring Zimbabwe (35%), Namibia (44%) and South Africa (73%) and is largely developed as commercial livestock farms, game ranches and arable blocks (Krug, 2001; Central Statistics Office, 2013b).

Protected areas constitute 19.1% of Botswana's land mass (Table 2.1; Fig. 2.11). The KTP was the first protected area in southern Africa to span across international borders (an amalgamation of the Gemsbok National Park and the Kalahari Gemsbok National Park in South Africa) (Peaceparks, 2011). All national parks and game reserves are unfenced enabling the free movement of wildlife populations, with the exception of the south-western boundary of the Makgadikgadi National Park (which is electrified but poorly maintained), the Kuke veterinary fence along the northern boundary of the Central Kalahari Game Reserve (CKGR) and the southern boundary of Khutse Game Reserve (Fig. 2.11).

In addition to nationally protected areas, WMAs extend the protected area network by 22.1% of Botswana's land mass. They are primarily designed for wildlife conservation, utilisation, and management and act as wildlife corridors and buffer zones between protected areas and farmland (Hachileka, 2003). However, unlike nationally protected areas (with the exception of the CKGR[1]), people are permitted to reside within WMAs and to own livestock, if they abide by regulations regarding numbers, husbandry and the erection of buildings (DWNP, 1992). At least five .

Table 2.1. Land use in Botswana. Source: Botswana Land use map, Cartographic Section, Ministry of Agriculture, cited in Central Statistics Office (2013b)

Land use	Land area (km^2)	Percentage of total land mass
Communal Land		**54.8%**
Pasture, Arable and Residential	253,223	43.5%
NADP ƚ Ranches	28,392	4.8%
Tribal Grazing Land Policy Ranches	24,292	4.2%
Lease Ranches	13,090	2.3%
State Land		**41.8%**
Wildlife Management Areas	128,574	22.1%
Game Reserves	60,558	10.4%
National Parks	45,900	7.9%
Forest Reserves	4555	0.8%
Quarantine and BLDC ǂ Ranches	3717	0.6%
City land	Not stated	Not stated
Freehold Land		**3.4%**
Freehold Farms	19,109	3.3%
Arable Blocks	320	0.1%
TOTAL	581,730	100.0%

ƚ NADP = National Agriculture Development Policy

ǂ BLDC = Botswana Livestock Development Corporation

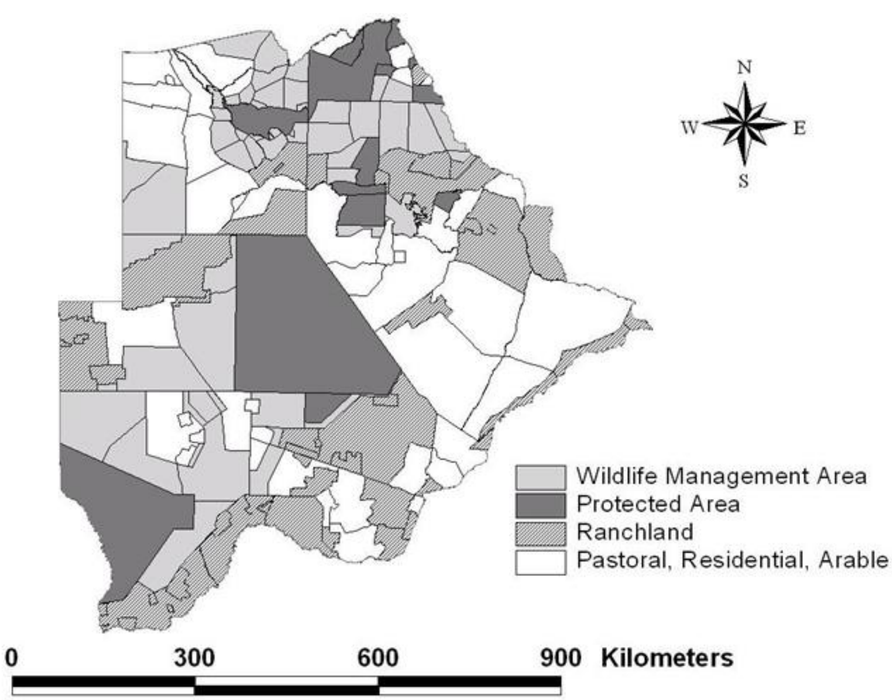


Figure 2.10. Major land uses in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

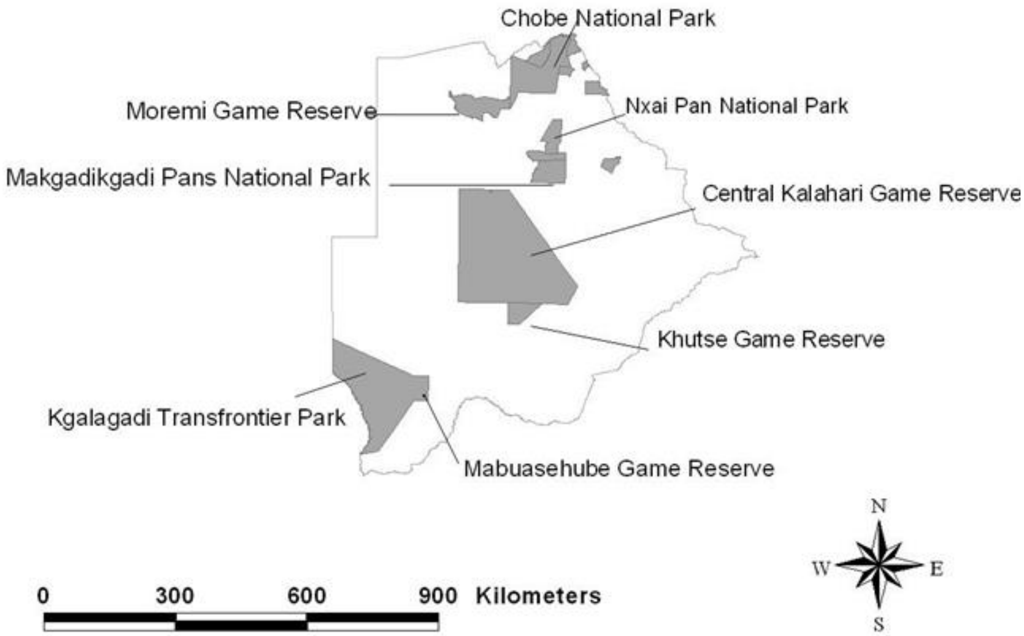


Figure 2.11. Protected areas in Botswana. Data source: Department of Wildlife and National Parks (n.d.)

community trusts have been formed within WMAs, which aim to cooperatively manage the natural resources, providing income for the community through commercial hunting and tourism (Pienaar *et al.*, 2013). However, the number and distribution of cattle have expanded in WMAs, resulting in HWC between residents and wildlife, due to carnivore depredation of livestock and with herbivores for competing with cattle for water and grazing (CAR, 2011).

[1] 260 San bushmen have been given permission by the Botswana government to return to their native lands in the CKGR (Majelatle, 2012).

2.11 Current status of the cattle farming industry in Botswana

In 2012 the livestock population in Botswana was estimated as 2.6 million cattle, 1.8 million goats *Capra hircus* and ca. 300,000 sheep, most of which were located on the more fertile eastern side of the country (Fig. 2.12). The majority of cattle in 2012 were kept by traditional farmers (2.26 million cattle owned by ca. 74,600 farmers) with ca. 294,000 cattle in the commercial sector (659 farmers). All exported beef must be processed by one of the abattoirs owned by the Botswana Meat Commission (BMC), based in Lobatse, Francistown and Maun. The large distances between farms and the BMC abattoirs, restricts traditional farmers' access to the market, contributing to a lower average price for traditionally farmed cattle (ca. US $405 per animal) compared to cattle produced in the commercial sector (ca. US $461 per animal) (Darkoh & Mbaiwa, 2002; Central Statistics Office, 2012a).

To export to the European Union all beef must be identifiable and traceable from 'the farm to the fork' (Marumo & Monkhei, 2009). However, failures in Botswana's traceability system, in conjunction with disease outbreaks, have resulted in export restrictions and market closures, most recently in 2010 to 2012 (Bahta & Malope, 2013).

2.12 Current status of the national wildlife industry in Botswana

Until the late 1990s, Botswana gained most of its direct revenue from wildlife through hunting; the tourism industry being underdeveloped and lacking facilities and infrastructure (Ministry of Finance and Development Planning, 1985). The hunting industry was one of the largest hunting operations in Africa, generating ca. US $20 million a year in revenue (Chardonnet, 2002; Lindsey *et al.*, 2007b) and earning more per client than any other country in the Southern African Development Community (Lindsey, 2011). Plains game, elephants, lions and leopards could be hunted under a permitted quota system allocated by the DWNP on both private game ranches and

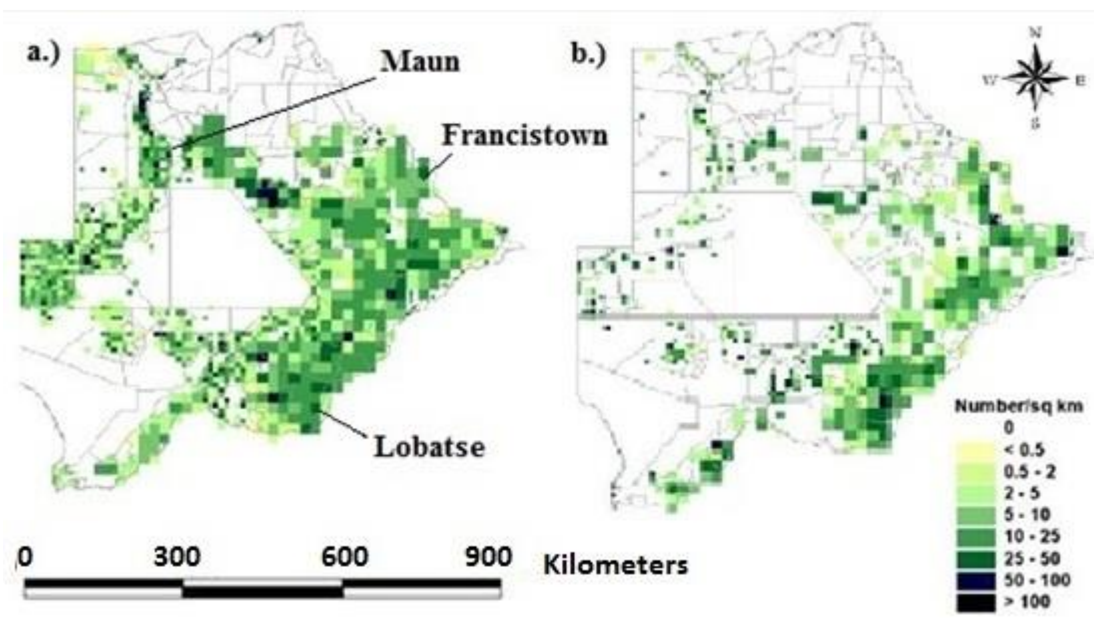


Figure 2.12. Distribution and density of a.) cattle and b.) smallstock in Botswana. Source: Department of Wildlife and National Parks aerial survey (DWNP, 2013)

in wildlife concession areas. Private hunting operators were able to buy permits directly from the government or were able to purchase citizen hunting permits from community trusts as part of CBNRM. However, in January 2014 the hunting of wildlife outside of private land ceased (MEWT, 2013); a moratorium was placed on leopard trophy hunting from January 2013 (BWPA, 2012c), and a moratorium on lion trophy hunting has been in place since 2007 (and previously between 2003 and 2005) (Packer *et al.*, 2006).

Since the 1990s the Botswana tourism industry has grown substantially (Mbaiwa, 2003; 2011) and is expected to grow further with the conversion of the previously used hunting concessions to tourism. Botswana has a high-cost/low-volume tourism policy (National Assembly, 1990b). In the Okavango delta this is largely the case and concessions are leased for the establishment of high-end lodges to which clients fly in and out. However, in more accessible areas especially along the Chobe river, littering, motorised boats and the increased movement of tourists is thought to pose a threat to wildlife conservation (Hachileka, 2003).

2.13 Current status of the game ranching industry in Botswana

The majority of game ranches in Botswana are in the Tuli-block, Ghanzi, Hainaveld and Makalamabedi farms (Fig. 2.13). All game ranches must be game-fenced and registered with the DWNP (BWPA, 2005). In 2012 there were 111 registered ranches (BWPA secretary pers. comm.). The minimum size is 4000 ha, however, due to low annual rainfall and poor soil fertility in the west of the country, it is recommended that game ranches in the west are larger (BWPA, 2005). Game-stock species are often reintroduced into the farm or in some cases were naturally present when the game-fence was constructed.

The industry is largely based upon hunting (for meat or trophy hunting) and tourism; many ranchers (specific number not reported) farm game-stock in addition to cattle production (BWPA secretary pers. comm.). Live game sales have been limited due to the imposition of periodic export restrictions imposed in 2009, 2013 and 2014 (BWPA, 2009b; 2013; MEWT, 2014) and the saturation of the game-stock sales market in South Africa (Lindsey *et al.*, 2009c). Similarly, the export of game meat has been restricted due to export regulations and a lack of access to the

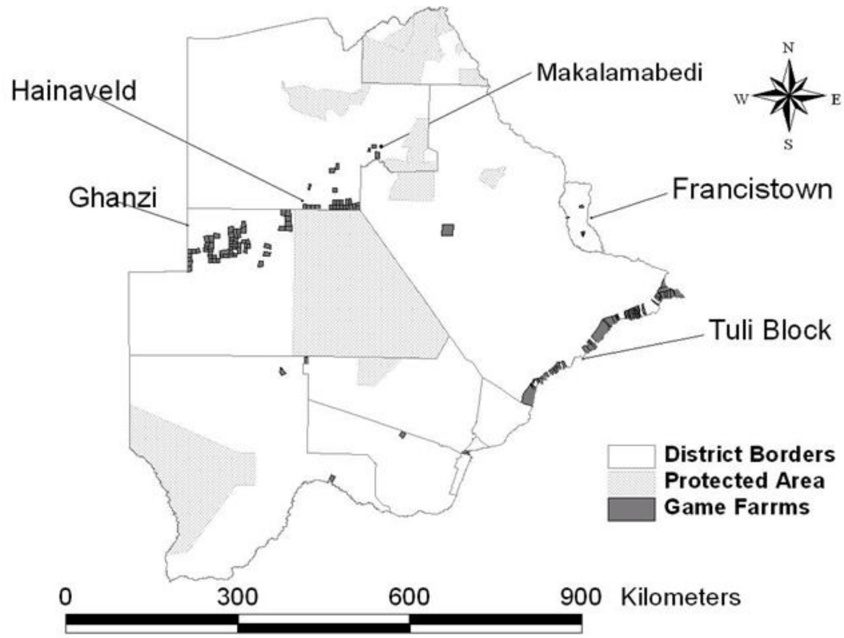


Figure. 2.13. Location of game ranches in Botswana. Data Source: Department of Wildlife and National Parks (n.d.) and BWPA (2012a)

European market (BWPA, 2007; 2009a). Access to the local game meat market has also been hindered by the introduction of abattoir standards (introduced in the Botswana Livestock and Meat Act of 2007 (Government of Botswana, 2007) which has increased the costs of harvesting and processing and has imposed stringent hygiene and veterinary control regulations (BWPA, 2009b). The ownership of captive predators requires permission from the director of the DWNP and is generally discouraged (DWNP, 2011a); at the time of study four farms owned captive predators, primarily lions and one farm was involved in the breeding and selling of lions to South Africa. However, this was heavily criticised in the local media (Macleod *et al.*, 2013).

The Botswana Game Ranching Policy formed in 2002, acknowledges that the industry has been limited by difficulties in securing land (due to the small proportion of freehold land in Botswana compared to neighbouring countries), insufficient capital or collateral to develop a game ranch and a general lack of experience, support and information dissemination in game-stock management (Republic of Botswana, 2002). The policy aims to address these issues and facilitate the growth of the industry, with the assistance of the BWPA, set up in 2002, to aid communication between government, non-government organisations (NGOs) and ranchers (BWPA, 2003).

2.14 Main game ranching regions

2.14.1 Tuli-block farms

The Tuli-block farms are located in south-east Botswana along the Limpopo River, which acts as the border between Botswana and South Africa (Fig. 2.14); thirty-six of the registered game ranches are within this farming block (BWPA secretary pers. comm.). The river periodically has water and farmers experience conflict with Nile crocodiles, African elephants and hippopotamus in addition to the large mammalian carnivores. The farms are freehold, enabling private ownership of the land and include livestock farms, game ranches and farms ranching livestock and game-stock. The vegetation is predominantly hardveld, the soils are moderately fertile and the area receives ca. 450 mm of rain per year. Free-ranging wildlife in the Tuli-block includes impala, bushbuck, red-hartebeest, tsessebe, plains zebra and elephants; 'golden' wildebeest also naturally occur in the area (farming community member pers. comm.). All of the large carnivore species are reported to be present with the exception of African wild dogs; lions are generally restricted to the eastern farms (CCB, n.d.).

The farms in the east have formed a private game reserve, known as the Northern Tuli Game Reserve, for the protection and conservation of wildlife. The reserve conducts photographic

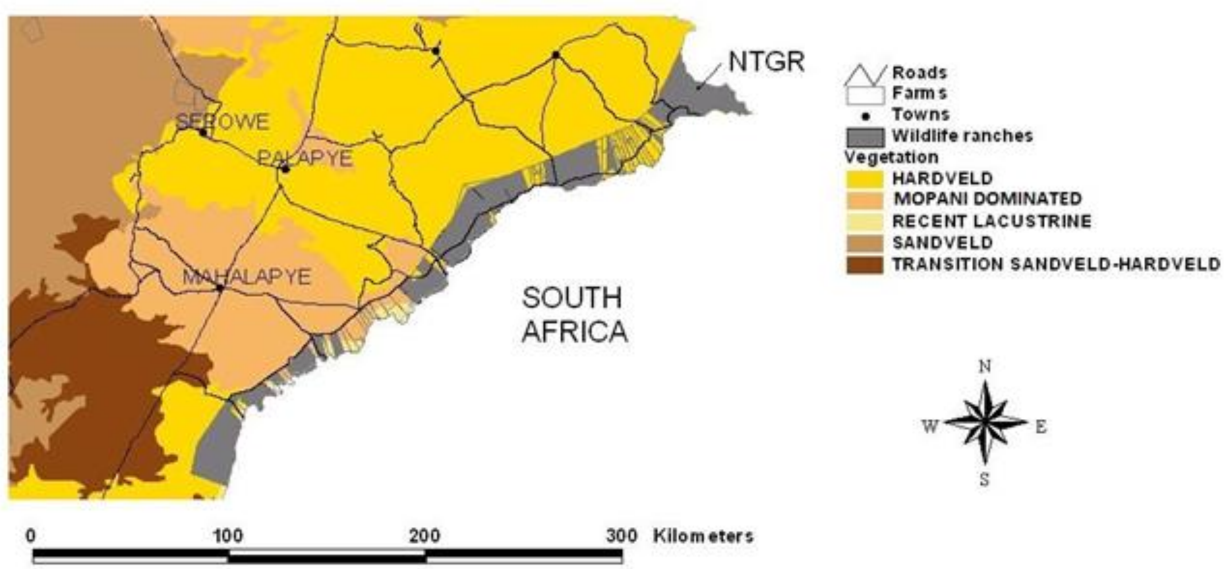


Figure 2.14. Location of Tuli-block farms. (NTGR = Northern Tuli Game Reserve). Data Source: Department of Wildlife and National Parks (n.d.) and BWPA (2012a)

tourism and is to be incorporated into the Shashe-Limpopo trans-border park with areas in Zimbabwe and South Africa, in the future (SANParks, n.d.).

2.14.2 Ghanzi farming block (site of prey analysis and predator densities)

The Ghanzi farming block located in the Ghanzi district, north-west Botswana is one of the largest commercial farming areas in Botswana; 17 ranches are registered as having game-stock (BWPA secretary pers. comm.). With an approximate area of 13,152 km^2, the farming block is bordered by the CKGR to the east and the Namibian border to the west (Fig. 2.15). The mean annual rainfall is 400 mm and vegetation is hardveld and sandveld (Fig. 2.15). The first farms to establish in the area in the late 19th century were located along a limestone ridge. This ridge is characterised by a shallow water-table, natural pans, that fill with water during the wet season, and grasses with a higher crude protein content than located on adjacent sandveld soils (Cole & Brown, 1976; Burgess, 2003; Kent, 2011). With improvements in bore-hole drilling technology, the farming block has subsequently expanded onto the sandveld soils (Kent, 2011), which are characterised by a much deeper water-table and less fertile soils than the hardveld (Cole & Brown, 1976).

Farms are freehold and are primarily owned by farmers of European descent, predominantly Afrikaners (Kent, 2011). The farming block is a mixture of commercial livestock farms, game-stock only ranches and dual-use livestock/game-stock farms. Many farms keep smallstock, but these are predominantly for private use or to sell locally (pers. obs.). Individual farm plots are ca. 5000 ha, but often farms are made up of multiple plots and the majority of land is owned or managed by ca. 50 families (Kent, 2011). Free-ranging ungulate species include common duiker *Sylvicapra grimmia* (herein referred to as duiker), steenbok *Raphicerus campestris,* warthog and kudu. Game ranches stock giraffe *Giraffa camelopardalis*, plains zebra and a mixture of large and small antelope species, including springbok, impala, gemsbok, common eland *Tragelaphus oryx*, blue- and black wildebeest, red hartebeest and kudu (Kent, 2011).

The large predator species present in the area are cheetahs, African wild dogs, leopards and brown hyaenas; spotted hyaenas are occasionally reported as present on the northern farms and lions are present on the farms bordering the CKGR (Kent, 2011; farming community member, pers.

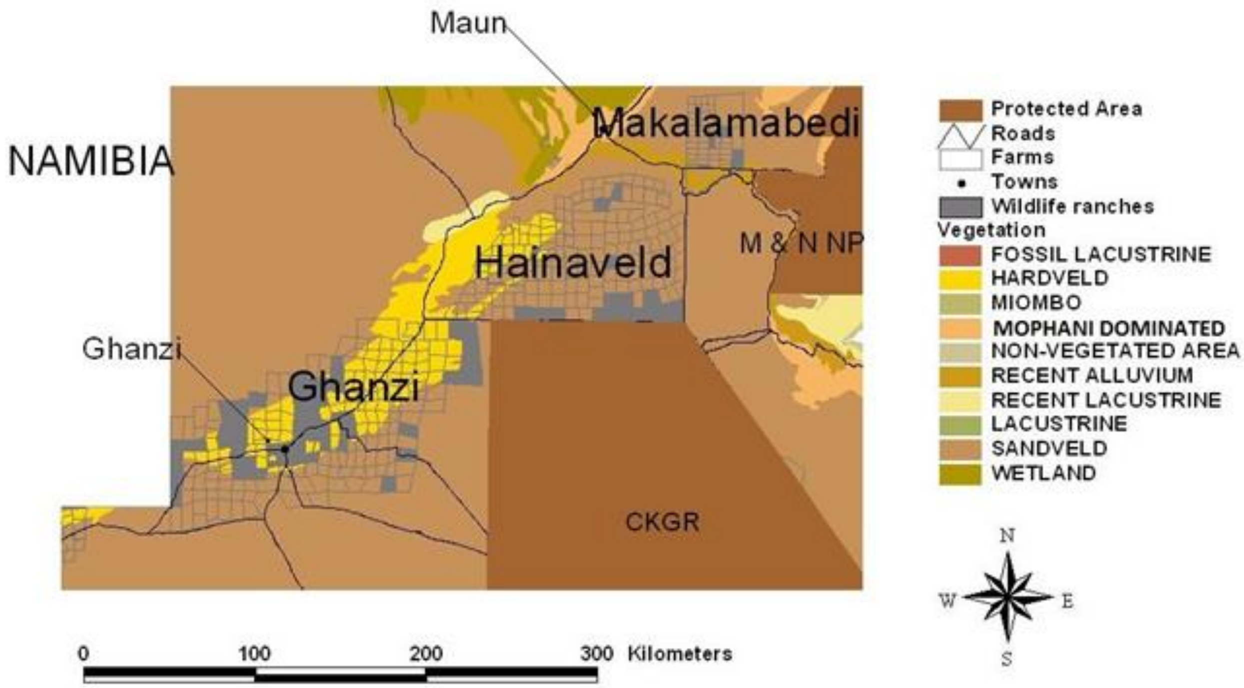


Figure 2.15. Location of Hainaveld, Makalamabedi and Ghanzi farms (CKGR = Central Kalahari Game Reserve; M & N NP = Makgadikgadi and Nxai Pan National Park). Data Source: Department of Wildlife and National Parks (n.d.) and BWPA (2012a)

comm.). The Ghanzi district has the third largest number of 'problem' animal reports (per citizen), after the Chobe and Ngamiland districts (CAR, 2011). Farmers thought that leopards, jackals, African wild dogs and cheetahs are the species causing the majority of predator problems in the area (Selebatso *et al.*, 2008).

2.14.3 Hainaveld and Makalamabedi farms

The Hainaveld and Makalamabedi farms are in the Ngamiland district in north-west Botswana (Fig. 2.15); there are 16 registered game ranches in the area (BWPA secretary pers. comm.). The Hainaveld farms are to the north of the CKGR. The farms bordering the CKGR are predominantly game ranches but are separated from the reserve by the farm's private game-fence and the Kuke veterinary fence (itself a double fence in sections). This corridor fencing can endanger wildlife populations (Mbaiwa & Mbaiwa, 2006), with ungulate species becoming stuck and dying in the corridor (Fig. 2.16). The majority of farms further north are livestock farms.

Figure 2.16.a. Red hartebeest within the corridor formed by the Kuke veterinary fence

b. Red hartebeest carcass by the Kuke veterinary fence

North-east of the Hainaveld farms are the Makalamabedi farms. Due to a lack of available surface water and salty underground water the majority of livestock farms rely on the Boteti River for water. The river had been dry since the mid-1990s, but began flowing again in 2009 and livestock farming was starting in the area again at the time of study (2012/2013) (farming community member pers. comm.). The Makgadikgadi and Nxai pans national park is to the east of the farms. The soil fertility is low and rainfall is generally between 350 and 400 mm per year (Fig. 2.3 and 2.4). The vegetation is predominantly sandveld. Free-ranging game species are adapted to dry environments and include kudu, red hartebeest and springbok (pers. obs.); elephants are present in the area during the wet season (farming community member pers. comm.). All of the large carnivores are considered present, with the exception of brown hyaenas (CCB, n.d.).

Chapter Three: Methodology

3.1 Chapter overview

The previous two chapters provided an introduction to the research topic of HPC on game ranches and a background to the study area (i.e. commercial farms in Botswana). Chapter 3 will provide a brief description of the research methods used in this study, namely questionnaires, spoor surveys, camera-trap surveys, scat-analysis and GPS collaring and translocation of 'problem' cheetahs. As cheetahs are thought to cause the largest economic losses on game ranches in Botswana (BWPA, 2005) and are the third most threatened carnivore in Africa, after African wild dogs and Ethiopian wolves *Canis simensis* (Ray *et al.*, 2005; IUCN, 2013), the majority of studies with the exception of the questionnaire data were primarily designed for cheetahs. This chapter provides an overview of the research methodology, the reasons for its use and a brief background to the technique; specific details will be included in the relevant chapters.

3.2 Questionnaires

3.2.1 General questionnaire design

This study used questionnaires to determine the direct costs and drivers of HPC (Chapter 4) and the distribution of predators on commercial farmland (Chapter 5). The questionnaire also discussed farmers' perceptions of the primary prey items consumed by predators (Chapter 6) and the use and effectiveness of conflict mitigation techniques (Chapters 7 and 8). Interviews and questionnaires are a useful and commonly used tool to examine human attitudes and behaviours towards wildlife species (White *et al.*, 2005). A participatory focus group approach was considered; however, it was felt that the transaction costs of participating would potentially bias the participants to an unrepresentative sample of the game ranching industry (Treves *et al.*, 2009). For an initial survey of the industry it was thought to be important to consult as many of the game ranchers as possible, which could be best achieved through questionnaires.

The questionnaire was designed using previous HPC surveys as a guide (Marker, 2002; Dickman, 2008; Selebatso *et al.*, 2008; Stein *et al.*, 2010; Kent, 2011; CCB, unpubl. data.), in consultation with questionnaire design guidelines from White *et al.* (2005) and Websurveyor (2003). The majority of questions were closed-ended; associated open-ended questions, such as 'please explain why' provided opportunities for further explanation and comments. These open-ended questions enabled a deeper understanding of farmers' attitudes and reasoning (O'Cathain & Thomas, 2004). A semi-structured questionnaire was chosen over an unstructured approach because it would enable the most effective and standardised data collection in a limited time period.

Questionnaires were administered during a face-to-face interview with farmers. Conducting the questionnaires as telephone interviews was considered, but telephone reception was largely restricted to the patchy and unreliable cellular telephone network and, therefore, was deemed to be impractical for interviews. Similarly, postal or email questionnaires were rejected because the response rate would potentially be too low to collect meaningful data (White *et al.*, 2005). Previous attempts to collect data from Botswana's game ranchers via email had poor response rates (BWPA, 2011a) and generally postal/email questionnaires have a response rate of less than 50% (Weisberg *et al.*, 1996).

Attempts were made to contact all game ranchers in Botswana to ensure that the sample for the research was representative of the game ranching industry as a whole. Individuals who chose to or chose not to participate in surveys may be fundamentally different; this non-response bias reduces the external validity of the survey (Oppenheim, 1992). Face-to-face interviews were conducted with the aim of increasing the response rate to above 85%, for which non-response bias is no longer thought to be a problem (Lindner, 2002). Livestock farmers were selected from the main game ranching regions for comparison purposes, using 'snowball' sampling, in which the individuals questioned were asked to provide contact details for their neighbours as suggestions for additional people to contact. 'Snowball' sampling can reduce external validity and has the potential to introduce bias (Sadler *et al.*, 2010). However, because contact lists for livestock farmers were not readily available or were outdated, prohibiting random or stratified sampling, 'snowball' sampling was considered an appropriate method to identify and question the livestock farmers living adjacent to game ranches. 'Snowball' sampling has the added advantage of higher response rates as the interviewer has an opportunity to be 'introduced' to new participants (Sadler *et al.*, 2010).

Gaining the trust of participants is essential for accurate data collection (Hazzah, 2006). The questionnaires were introduced as university research which is often seen as neutral and non-threatening (Mitchell & Carson, 1989). Some of the participants knew the interviewers which is likely to have helped to establish trust and openness. However, previous knowledge of the interviewers or preconceptions of the researcher's beliefs may have also biased the responses. Participants may report what they believe the interviewer wants to hear, or may manipulate answers in an attempt to influence findings and subsequent policy decisions. This bias could reduce the validity of the results (Hazzah, 2006).

3.2.2 Randomised response technique

The final section of the questionnaire consisted of yes/no questions regarding the lethal removal of predators and was conducted using the 'forced response' randomised response technique (RRT) developed by Warner (1965). The RRT has been commonly used in social science to collect information on sensitive topics that participants are unlikely to wish to discuss, because of concerns of confidentiality and/or social embarrassment (Boruch, 1971). This unwillingness to answer can lead to non-response and social-desirability bias, in which individuals refuse to participate or answer dishonestly in order to promote a favourable image of themselves, that conforms to social norms (Fisher, 1993). Individuals can often be uncomfortable talking to an outsider about illegal activities (Hussain, 2000) and previous HWC studies have reported high rates of non-response to lethal control questions, for example only 24% of participants answered questions regarding killing cheetahs on farmland in Namibia (Marker *et al.*, 2003d). It was, therefore, hoped the use of the RRT method would increase the response rate and validity of the data.

There are multiple versions of the RRT, all of which use a randomising device, commonly dice, to increase the participants' privacy by adding an element of chance to the answering process (Lensvelt-Mulders *et al.*, 2005b). The 'forced response' RRT method is one of the most statistically efficient methods (Boruch, 1971; Lensvelt-Mulders *et al.*, 2005a). Participants are instructed (figuratively 'forced') to answer sensitive questions with a 'Yes' or a 'No' irrespective of the truth or truthfully 'Yes' or 'No' based on the sum of the numbers rolled on two dice (Boruch, 1971; St John *et al.*, 2010). The dice roll is not seen by the interviewer; therefore, the interviewer is unaware if the question is being answered truthfully or if a 'forced' answer is being given. This introduced level of anonymity has been shown to result in a greater incidence of honest reporting on sensitive issues, for which the true answer was already independently known, than conventional methods (Lensvelt-Mulders *et al.*, 2005a; Lensvelt-Mulders *et al.*, 2005b). St John *et al.* (2010) applied the technique to ecological studies and estimated a greater prevalence of rule-breaking in fly fishing using the RRT compared to self-completed questionnaires. This was seen as an indication of greater honesty; the technique was subsequently applied to the lethal control of predators (St John *et al.*, 2012).

The incidence of the behaviour in the population as a whole can then be calculated based on the probability of rolling a number that results in a forced 'Yes', 'No' or an honest answer, using the Hox and Lensvelt-Mulders (2004) model (described in Chapter 7). The current study asked questions regarding the lethal removal of large carnivore species, the use of poisoned bait and the illegal removal of protected predator species.

3.3 Predator densities

Species density can be determined using direct methods such as mark and recapture or physical counts, or by using indirect methods that rely on the detection of field signs as a relative measure of species density (such as scat, den-sites or spoor [footprints]) (Mills, 1997). Although direct methods are often more accurate and precise, for low-density, wide-ranging species such as large carnivores, direct methods are often expensive, difficult to conduct and time-consuming (Mills, 1997; Stander, 1998; Wilson & Delahay, 2001; Gusset & Burgener, 2005). Consequently, the density of large predators on commercial farmland in Botswana was estimated using the indirect methods of spoor surveys and camera-trap surveys. These methods are considered more cost-effective and less invasive than direct methods (Jewell *et al.*, 2001), yet remain repeatable, objective and accurate (Stander *et al.*, 1997a; Gusset & Burgener, 2005; Balme *et al.*, 2009a). Both techniques have limitations; therefore, it was chosen to use both methods to calculate a more informative estimate of species density.

3.3.1 Spoor surveys

Spoor surveys have been used to monitor population size and trends in several African carnivore species, including leopards, lions, brown hyaenas, caracals and cheetahs (Stander, 1998; Funston *et al.*, 2001; Melville & Bothma, 2006; Houser *et al.*, 2009b; Funston *et al.*, 2010). The current study used three independent spoor surveys to achieve an overall estimate of large predator densities in the Ghanzi farming block. The surveys were conducted in an area of livestock farms, in an area of ranches farming only game-stock and in a mixed area of livestock farms and game-stock only ranches. The spoor surveys were conducted upon fixed routes, along sand roads, and a skilled tracker identified the spoor from a vehicle driven at slow speeds. Any fresh spoor (< 24 hours old) found to belong to a large predator was identified and recorded.

Spoor surveys calculate a relative estimate of species density based on the number of spoor detected per 100 km of road surveyed (Houser *et al.*, 2009b). To improve the accuracy of the estimate a quantifying technique can be applied. One option that has been used with pumas (Lewinson *et al.*, 2001), leopards (Gusset & Burgener, 2005) and tigers (Sharma *et al.*, 2005), is to use unique identifying measurements of the spoor track to identify and count individuals within a population. The technique is being trialled with cheetahs (F. Weise pers. comm.), however, it requires refinement before it can be used in the field, where substrates and, therefore, spoor measurements will vary (Lewinson *et al.*, 2001).

The more commonly used alternative, is to double sample the population by conducting a spoor survey and a direct technique, such as capture-mark and recapture. The relationship between the two estimates can be explained with a regression equation, which can be used to calibrate future spoor survey data (Eberhardt & Simmons, 1987; Wilson & Delahay, 2001). Calibration equations have been calculated for lions, leopards, African wild dogs (Stander, 1998; Funston *et al.*, 2001) and cheetahs (Houser *et al.*, 2009b). The Funston *et al.* (2001) calibration equation was calculated in lions and extrapolated to cheetahs, brown hyaenas, spotted hyaenas, and leopards (Funston *et* *al.*, 2001). This equation was refined with real data sets in 2010 and is considered suitable for all large predators on sandy roads above a spoor density of 0.32/ 100 km (Funston *et al.*, 2010). Below this threshold low detected spoor densities result in negative estimates of population density. Therefore, the current study used the Funston *et al.* (2010) calibration equation when possible, or the Stander (1998) calibration equation for applicable species with a low spoor density.

3.3.2 Camera-trap surveys

For species with individual markings, camera-trap surveys can be used to estimate species' abundance using capture-recapture based methodology (Karanth, 1995). The technique has been widely used for many carnivore species worldwide (O'Connell *et al.*, 2011). African carnivores for which camera trap surveys have been utilised include cheetahs (Marker *et al.*, 2008; Marnewick *et al.*, 2008), brown hyaenas (Kent & Hill, 2013), striped hyaenas *Hyaena hyaena* (Gupta *et al.*, 2009) and leopards (Stein *et al.*, 2011). However, to be biologically relevant, abundance estimates must be converted to densities, either by estimating the size of the surveyed area utilising buffer-strip methods or by using spatially explicit capture–recapture (SECR) models (Foster & Harmsen, 2012).

Buffer-strip methods are based on the assumption that at least some of the study animals' home ranges are likely to extend beyond the polygon formed by the outer camera-trap locations, known as the sampling area. If movements outside of the sampling area were unaccounted for, the assumption of geographic closure would be violated. As a result, the size of the total area surveyed would be underestimated and the density estimate would be overestimated. To account for these movements a buffer-strip is added around the outer camera-traps (Dice, 1938; Silver *et al.*, 2004). The width of this buffer-strip is based on assumptions as to the distances that individuals move, most commonly it is equivalent to the radius of the home range of the target species, specific to that geographical area or habitat (Karanth & Nichols, 2000; Soisalo & Cavalcanti, 2006). If home range data are unavailable, the mean maximum distance moved (MMDM) of all individuals captured more than once during the camera-trap survey (Soisalo & Cavalcanti, 2006; Dillon & Kelly, 2008; Maffei & Noss, 2008) or half of the MMDM can be used as the estimated buffer width (Wilson & Anderson, 1985; Karanth, 1995; Silver *et al.*, 2004; Jackson *et al.*, 2006; Balme *et al.*, 2009a). Repeat captures at the same camera station may (Kelly, 2003; Dillon & Kelly, 2007) or may not be included (Trolle & Kery, 2003; Maffei *et al.*, 2005). Although buffer-strip methods have been used for many large felid populations, they are used in an *ad hoc* manner, are difficult to characterise theoretically, there is little consensus in their use and often the reporting of the specific techniques used is poor (Royle *et al.*, 2009b; Foster & Harmsen, 2012).

An alternative to buffer-strip methods are spatially explicit capture-recapture models, which use the capture history of each individual in combination with their spatial locations to calculate density, thereby removing the assumption of geographic closure and the requirement to arbitrarily define the size of the area surveyed (Efford, 2004; Borchers & Efford, 2008; Royle *et al.*, 2009a). In addition, SECR models are able to account for more of the variation observed in camera-trap data than is possible using non-spatial buffer-strip methods. SECR models can account for lost camera-trap nights (for example due to camera failure) and can exclude non-suitable habitat such as towns from their spatial and, therefore, density estimates (Efford & Fewster, 2012). SECR models, unlike buffer-strip methods, do not require multiple days of data to be condensed into sampling occasions (Royle *et al.*, 2009a). This condensing of data often results in the loss of individual captures and can introduce individual and temporal heterogeneity within and across the sampling occasions (Foster & Harmsen, 2012). SECR models can also account for variation in the probability of capture for each individual caused by the distribution of individuals relative to the camera locations (Efford *et al.*, 2009; Royle *et al.*, 2009b; Singh *et al.*, 2010) and, if sample size allows, due to age, social grouping, status and sex (Borchers & Efford, 2008; Royle *et al.*, 2009b; Sollmann *et al.*, 2011; Blanc *et al.*, 2013). All of these factors are known to reduce the reliability of the estimate of abundance and, therefore, density (Foster, 2008; Gardner *et al.*, 2010b; Harmsen *et al.*, 2011; Sollmann *et al.*, 2011; Tobler *et al.*, 2013). There are two varieties of SECR, the maximum likelihood method (Efford, 2004; Efford & Fewster, 2012) and the Bayesian model (B-SECR) (Royle & Young, 2008; Royle *et al.*, 2009a). The B-SECR model uses non-asymptotic inferences which are more appropriate for small sample sizes and is considered more suitable for elusive, low-density predators (Kery *et al.*, 2011; Gopalaswamy *et al.*, 2012b). The B-SECR model has been used to estimate density in many individually recognisable carnivores (Royle *et al.*,

Table 3.1. Assumptions of capture-recapture analysis for use with buffer-strip methods and the SECR model, extracted from Foster and Harmsen (2012) and Tobler and Powell (2013). Applicable assumptions are marked with an 'X'

Assumption	**Buffer-strip methods**	**SECR model**
All individuals are individually recognisable	X	X
All individuals have an opportunity for capture	X	X
Demographically closed population	X	X
Capture probability not dependent on trap location	X	X
Each capture is an independent event	X	X
Geographically closed population	X	-
Assume individuals have circular home ranges	X ł	X
Distribution of individuals follows a Poisson distribution	-	X
Individuals have independent activity centres	-	X
Location of activity centres/ home ranges is fixed during survey period	-	X
Probability of detecting an individual at a camera station declines with distance of the activity centre from the trap	-	X
Area around cameras is homogenous in terms of habitat suitability	-	X

ł Only applicable if buffer-strip methods are determined using home range data

2009b; Gardner *et al.*, 2010a; Sollmann *et al.*, 2011; Noss *et al.*, 2012; Chase Grey *et al.*, 2013; Kent & Hill, 2013), but to the author's knowledge is yet to be applied to cheetahs.

The camera-trap survey for the current study was conducted over a period of 84 days to maintain the assumption of demographic closure (Karanth & Nichols, 1998). Twenty-six pairs of cameras were mounted on roadways. It is generally agreed that random camera placement should be avoided and cameras should be positioned to maximise the probability of capture of the target species (Carbone *et al.*, 2001), in this case cheetahs. However, camera surveys generate a large quantity of data which, if species ecology is similar, can be used to estimate population density in other non-target species (Kelly *et al.*, 2008). All large predators commonly use roadways; therefore, the survey was used to estimate the density of all large carnivore species with individually recognisable pelage.

The design of camera-trap surveys is often constrained by practical considerations, such as the number of cameras available. It is a balance between maintaining a suitable camera density to avoid missing individuals and exposing as many individuals as possible to being photographed (Foster & Harmsen, 2012). This study used a 'blocked' survey design in order to maximise camera density and the size of the sampled area (Karanth, 1995). Cameras were placed at 13 sites for 42 days before removal to an additional 13 sites in an adjacent area for a further 42 days.

The number of times a species was independently photographed (captured) was calculated per 100 camera-trap nights. Captures were considered independent if recorded more than one hour after the previous capture of that species. Individuals of carnivore species which are individually recognisable, namely cheetahs, leopards, African wild dogs, spotted hyaenas and brown hyaenas, were identified manually, by looking for identical spot or stripe patterns, primarily on the fore- and hind legs. Only individuals deemed as adults with independent captures were included in the data analysis, in line with the assumptions of capture-recapture analysis (Table 3.1). The density of individually recognisable species was estimated using the B-SECR model and the non-spatial, buffer-strip methods (the MMDM and the home range radius) to enable comparison to older studies.

3.4 Prey analysis

The prey a carnivore consumes can be broadly categorised as browsers or grazers by analysing the stable carbon isotope ratio of the carnivores breath (Voigt *et al.*, 2013). To identify the species consumed, commonly used methods are morphological or genetic analysis of prey remains in scat (Casper *et al.*, 2007; Klare *et al.*, 2011), stomach contents analysis (Brassine, 2011) or through predator kill identification either opportunistically or by using data from GPS-collared individuals or by physically following an individual or its spoor (Bissett & Bernard, 2007; Martins *et al.*, 2011b; Tambling *et al.*, 2012). Stomach content analysis would be unethical and impractical in a threatened species like the cheetah. Opportunistically locating kills or following individuals and spoor in dense bush would be time-consuming and difficult (Marker *et al.*, 2003e). Using clusters of GPS points from GPS collared individuals to locate kills would also be impractical; cheetah's handling time of kills is shorter than predators, such as leopards, for which GPS clustering has been used (Caro, 1994; Martins *et al.*, 2011a) and kills with short handling times can often be missed when being located by GPS clusters (Tambling *et al.*, 2012; Pitman *et al.*, 2014).

Additionally, kill identification biases sampling to larger prey items i.e. to those kills with identifiable remains (Mills, 1992; Marker *et al.*, 2003e). The detection of prey remains in scat was, therefore, the preferred option. As the aim was to identify prey to the species level in a large number of scats, it was chosen to use morphological rather than genetic analysis due to the lower costs.

The morphological analysis of prey remains in scat has been used to examine the diet of numerous African carnivore species, including cheetahs (Marker *et al.*, 2003e; Wachter *et al.*, 2006; Wachter *et al.*, 2012), brown hyaenas (Stein *et al.*, 2013), black-backed jackals (Kaunda & Skinner, 2003) and African wild dogs (Davies-Mostert *et al.*, 2010). When prey is consumed indigestible material, such as feathers or hair, passes through the digestive system to be excreted in the scat. The scats

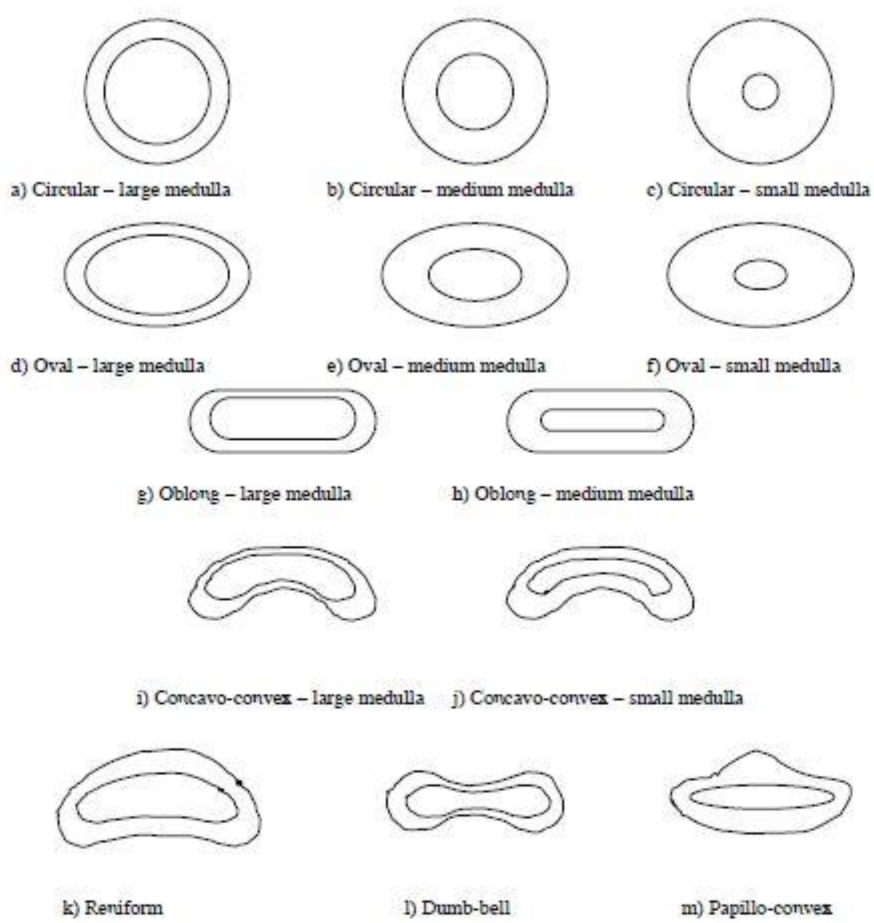


Figure. 3.1. The most common cross-section shapes of hair shafts. Source:Kent (2004) based on Keogh (1983) and Dreyer (1966)

can be collected, dried and the prey remains removed for identification. Hair is composed of keratin and the shaft has three components: the medulla, the cortex and the cuticle. The medulla is the cellular column at the centre of the hair and varies in size and shape (Fig. 3.1), the cortex surrounding the medulla contains pigment granules determining hair colour and varies in size, shape and colour. The cuticle, which surrounds the cortex, is the outer layer of the hair shaft and is composed of overlapping scales that also vary in size and shape (Kent, 2004). As a result of this variation, the cuticle scale pattern and the cross-sectional appearance of hairs vary between species. Therefore, hairs from the scat can be identified to a prey species by comparison with hair samples collected from known species (Keogh, 1983). The cross-sectional pattern is generally considered to be more definitive and reliable than scale patterns to identify between species (Keogh, 1983; Henschel & Skinner, 1990), therefore, cross-sectional analysis was used in the current study.

The varying proportions of indigestible material in different prey items cause food to pass through the digestive system at different rates (Floyd *et al.*, 1978). For example, smaller prey items have a larger ‘surface to volume ratio’ and contain lots of indigestible material (Floyd *et al.*, 1978; Wachter *et al.*, 2012). This results in a larger number of field collectable scats than feeding on meat alone. Without compensating for these differences, the biomass consumed of smaller prey items would be overestimated (Floyd *et al.*, 1978; Marker *et al.*, 2003e; Wachter *et al.*, 2012). To quantify the relative passage rates of different prey items and calculate correction factors to compensate for this variation, feeding trials have been conducted in numerous carnivore species, including cheetahs (Marker *et al.*, 2003e). Carnivores have different feeding ecology, for example, cheetahs eat more muscle compared to skin and bone than other carnivores (Wrogemann, 1975; Van Valkenburgh, 1996); therefore, the correction factor used must be specific to the target species (Floyd *et al.*, 1978; Hiscocks & Bowland, 1989; Klare *et al.*, 2011). This study used the Marker *et al.* (2003e) correction factor for cheetahs to determine the prey items consumed by cheetahs on commercial farmland in the Ghanzi farming block.

3.5 GPS collaring of translocated cheetahs

The translocation of 'problem' predators from farmland to protected areas is conducted globally as a more humane method to deal with 'problem' animals than lethal control (Massei *et al.*, 2010). Translocated individuals are fitted with Very High Frequency (VHF) radio or GPS tracking collars to monitor their movements and survival post-release. VHF radio collars transmit a VHF signal which can be detected with a receiver and used to triangulate the position of the monitored animal. Visual sightings of the individual are often obtained to monitor their condition and behaviour post release. GPS tracking collars have a VHF function, but additionally identify the position of the animal using GPS technology and transmit the locations by either cellular telephone or satellite signals. GPS or radio collars have been used to monitor the translocation of many large carnivore species, including leopards, lions and cheetahs (Trinkel *et al.*, 2008; Marnewick *et al.*, 2009; Weilenmann *et al.*, 2010).

During this study cheetahs were live-captured in double ended box traps using limited access or bait trap-sets as described in Houser *et al.* (2009a). Cheetahs were captured by Cheetah Conservation Botswana (CCB), the DWNP or independently by farmers. Captured cheetahs were reported to the DWNP and collected for translocation by CCB staff. Upon collection, cheetahs were either driven directly to their release site or were temporarily moved to the CCB research camp. Adult cheetahs were anaesthetised and underwent a physical health check, including the fitting of a VHF radio, GPS-cell or GPS-satellite collar using methods adapted from Marker (2002). Cheetahs were translocated and hard released (i.e. directly into the release site without any temporary holding at the site) in unfenced release sites, primarily within protected areas. Visual follow-up was not conducted as it was considered to be impracticable because of the unfenced release sites, which would enable cheetahs to move widely post-release. Post-release movements, including homing behaviour, site fidelity to the release site and survival time were analysed using the GPS data. Attempts were made to recapture and remove the collars from all cheetahs. Drop-off collars were not used, because the technology was deemed too unreliable and too heavy to place on cheetahs at the time of study (2003 − 2011), however, with advancements in technology drop off collars are recommended for future studies.

3.6 Statistical analyses

All data were numerically coded for use in the statistical computer package SPSS (version 11.0.1; SPSS Inc.) and Genstat (version 15; VSN International). Open-ended questions within the questionnaire data were coded based on content analysis to identify consistent themes within the answers. Data were cross-tabulated and frequencies were counted to check for discrepancies before commencing analysis. All financial data are quoted in the local currency, Botswana pula (BWP) and/or US dollars, based on the currency exchange rate at the commencement of the questionnaires of one US dollar to BWP 7.649.

The Kolmogorov-Smirnov or Shapiro Wilk test (for sample sizes under 50) were used to determine if continuous variables were normally distributed, and homogeneity of variances was established using the Levene's test of equality. Outliers were detected using box-plots and were identified as values greater than 1.5 box lengths from the edge of the box. Outlying values were transformed or were removed before running tests on a case by case basis as stated in the relevant tables. All necessary assumptions of the statistical test selected were upheld. T-tests or Mann-Whitney U-tests were used when comparing continuous variables with a categorical variable with two states. For categorical variables with multiple states an ANOVA or Kruskal-Wallis ANOVA with *post hoc* testing (Turkey HSD or Mann-Whitney U-tests with the Bonferroni correction) was utilised. When variables had non-homogenous variances the Welch correction was used for t-tests and ANOVA with the Games-Howell *post hoc* test. A paired samples t-test or the Wilcoxon signed rank test was used to compare variables with paired results. The Mann-Whitney U-test, Kruskal-Wallis ANOVA and the Wilcoxon signed rank test was used to compare ordinal variables. The correlation between two continuous variables was compared with Spearman's correlation. Chi-squared tests using Yate's correction factor for tests with one degree of freedom were used to compare proportions. Variables with a low incidence were collapsed into fewer categories to avoid sample size violation, and significant categories were defined as those with a standardised residual greater than $\pm$ 1.96, equivalent to a probability of 0.05. Multiple regression using the general linear model or multi-nominal logistic regression was used to assess the importance of various factors and covariates on continuous and categorical variables, respectively. The assumptions of homoscedasticity, linearity, unusual points, independence of errors and the normality of residuals were upheld. Means are presented with their standard deviation (SD). All tests were two-tailed and statistical significance was determined at $p < 0.05$.

Chapter Four: The direct costs and drivers of human-predator conflict on commercial game ranches and livestock farms in Botswana

4.1 Chapter overview

This chapter aims to determine the direct costs of conflict between predators and farmers and the potential drivers of conflict on game ranches compared with livestock farms. Questionnaire surveys were conducted with representatives from 86.2% of the registered game ranches in Botswana. Questionnaires gathered information on farm characteristics, the problems farmers faced and the number and financial cost of livestock and game-stock losses to predators. Farmers' tolerance of predators was measured in relation to how many livestock or game-stock animals they would tolerate losing before removing a predator from the farm. Overall, game ranchers were more tolerant of predators than farmers farming only livestock. Of the predator species African wild dogs and lions were the least tolerated. Spotted hyaenas and leopards (which were available for trophy hunting at the time of study [2012/2013]), were more tolerated for causing game-stock losses than cheetahs (which cannot be legally trophy-hunted), possibly because of their potential economic benefits in association with farmers' preconceptions of each predator species. The number of game-stock lost to carnivore depredation was more difficult to establish than the number of livestock losses. The reported number of game-stock lost to predators was greater if lions were present and was positively correlated with the number of years of game ranching experience the farmers had. Farmers based on game ranches with a large proportion of inexpensive game-stock species (buffer species) relative to the total game-stock herd, or farmers based on livestock farms where free-ranging small antelope species were perceived to be increasing, reported a high tolerance of predators. It is hypothesised that the increased availability of diversionary prey species may cause farmers to perceive predators as less of a threat to their expensive game-stock or livestock, resulting in a higher tolerance.

4.2 Introduction

Conflict between humans and predators most commonly occurs due to carnivore depredation on livestock and game-stock (Sillero-Zubiri & Laurenson, 2001; Woodroffe *et al.*, 2005d). As a result of HPC, predators are frequently persecuted in direct response to livestock and game-stock being killed or to prevent future losses. In most large carnivore species outside of protected areas human-mediated death is the primary cause of mortality (Ogada *et al.*, 2003; Woodroffe & Frank, 2005). Understanding the direct costs and drivers of HPC is essential for the implementation of appropriate measures to reduce conflict and to encourage predator-human coexistence (Messmer, 2000; Mattson *et al.*, 2006); therefore, promoting the long-term survival of predator populations.

Research into HPC has primarily focused on livestock farms, both subsistence and commercial. Generally, the number of livestock lost to carnivore depredation is small relative to other morbidity factors such as disease and climate (Breitenmoser, 1998); however, 'hot-spots' of conflict can exist, where a locally large number of losses are reported (Oli *et al.*, 1994; Mishra, 1997). Studies have found a direct relationship between the financial costs incurred due to carnivore depredation on livestock, farmers' reported tolerance of predators and the incidence of lethal predator removal (Oli *et al.*, 1994; Mishra, 1997; Ogada *et al.*, 2003; Hazzah, 2006; Romañach *et al.*, 2011; Lindsey *et al.*, 2013b). However, these findings have not been universal; wildlife interactions are often emotionally charged events (Hudenko, 2012), and other studies have shown that farmers' general perceptions and attitudes to predators are often equal to, or greater drivers of conflict than the direct costs (Conforti & de Azevedo, 2003; Marker *et al.*, 2003d; Zimmerman *et al.*, 2005; Dickman, 2008; Selebatso *et al.*, 2008; Swanepoel, 2008; Thorn *et al.*, 2012). Both intrinsic (individual experience) and extrinsic (economic, social and cultural) factors shape perceptions and attitudes to predators (Treves, 2009a). Generally, younger, wealthier, more educated-livestock farmers with alternative sources of income, have more positive attitudes towards predators and are less likely to remove them from their farms (Lindsey *et al.*, 2005c; Zimmerman *et al.*, 2005; Hazzah, 2006; Selebatso *et al.*, 2008; Swanepoel, 2008; Romañach *et al.*, 2011; Thorn *et al.*, 2012; Lindsey *et al.*, 2013b). Conversely, individuals who have experienced conflict with other wildlife species, live close to protected areas or have had previous adverse experiences with predators are more likely to exhibit negative attitudes to predator conservation (Hazzah, 2006; Dickman, 2008; Siemer *et al.*, 2009). Additionally, other factors such as ethnicity, religion, land use, residency in the area, and the size and value of the livestock herd owned have also been linked to farmers' tolerance of predators (Conforti & de Azevedo, 2003; Marker *et al.*, 2003d; Lindsey *et al.*, 2005c; Bagchi & Mishra, 2006; Hazzah, 2006; Dickman, 2008; Swanepoel, 2008; Romañach *et al.*, 2011; Thorn *et al.*, 2012).

In southern Africa the private ownership of game through the introduction of game ranches has led to an increase in wildlife habitat, diversity and abundance and game ranches have the potential to increase land connectivity for free-ranging predators (Flack, 2003; Lindsey *et al.*, 2009b). Game ranchers are able to benefit financially from predators through trophy hunting and photographic safaris (Lindsey *et al.*, 2009b). However, due to carnivore depredation on game-stock, which can have a greater financial value than livestock (Thorn *et al.*, 2012), game ranchers do not always tolerate the presence of free-ranging predators on their ranch (Treves, 2009b; Funston *et al.*, 2013). Game-stock animals, being non-domesticated, are ranched in extensive systems. Consequently conventional management techniques used to protect livestock from predators, such as kraaling and herding are not possible. As a result, game-stock can be more difficult than livestock to protect from carnivore depredation (Winterbach *et al.*, 2012; Lindsey *et al.*, 2013b).

Research in Namibia, Zimbabwe and South Africa suggests that game ranchers are more tolerant towards predators than livestock farmers (Lindsey *et al.*, 2005c; Swanepoel, 2008; Thorn *et al.*, 2012; Lindsey *et al.*, 2013b). However, when conflict does occur it can be more intense than other forms of HPC (Marker *et al.*, 2003d; Cousins *et al.*, 2008), and in Namibia cheetah removals were greater on game ranches than livestock farms (Marker *et al.*, 2003d). Due to the large economic set-up costs involved in starting a game ranch (e.g. in Botswana all game ranches must have a game-fence), the difficulties in protecting game-stock from predators and conversely, the potential financial benefits from predators, the direct costs and drivers of HPC on game ranches are likely to differ from livestock farming. However, few studies have specifically examined conflict between game ranchers and wildlife, and the extent and the impact of HPC on game ranches in southern Africa is considered a ‘gap in knowledge’ (Inskip & Zimmerman, 2009; Balme *et al.*, 2014).

Botswana plays a fundamental role in the conservation of many large predator species (IUCN/SSC, 2007b; Bauer, 2008). The number of registered game ranches in the country has increased six-fold between 1999 (BWPA, 2005) and 2012 (BWPA secretary pers. comm.) and HPC on game ranches has been reported (Swarner, 2004; BWPA, 2006b; Selebatso *et al.*, 2008). Therefore, conflict mitigation to encourage coexistence between game ranchers and large predators is required. This study aims to establish the direct costs and drivers of HPC on game ranches in Botswana, in order that appropriate advice on conflict mitigation methods can be offered (discussed in Chapter 7).

4.3 Study Area and methodology

4.3.1 Study area and sampling

Questionnaire surveys were conducted with game ranchers from across Botswana. A list of 111 registered game ranches was obtained from the BWPA, including email and telephone contact details and attempts were made to contact a representative from all of the ranches on the list. For comparative purposes, questionnaires were completed with livestock farmers from the largest game ranching areas, primarily Tuli-block farms, Ghanzi farming block, and Hainaveld and Makalamabedi farms (Fig. 2.13). Livestock farmers were selected using ‘snowball’ sampling, in which previously questioned farmers provided contact details for neighbouring farmers.

The perimeter of livestock farms was generally fenced with standard cattle fencing (five wires, ca. 1.5 m high). Game ranches were perimeter-fenced with standard game-fencing (12 − 17 wires, ca. 1.4 - 2.3 m high), and were rarely electrified (10%, n = 80, Chapter 7), enabling free-ranging predators to access the farms through the wires or through holes under the fence-line. Conservancies were rare, accounting for 5.0% of the interviewed game ranches and none of the livestock farms (Chapter 7). Further details of the study area are in Chapter 2.

4.3.2 Questionnaire methods

Questionnaires were designed using previous HPC surveys as a guide (Marker, 2002; Dickman, 2008; Selebatso *et al.*, 2008; Stein *et al.*, 2010; Kent, 2011; CCB, unpubl. data.). Due to the emphasis on game ranches, additional sections specifically designed for farmers ranching game-stock, concerning such issues as leopard trophy hunting and game-stock management, were created and piloted.

Questionnaires were divided into 11 sections (A-K), covering general farm and participant characteristics, problems and threats to the farm, the predators present, causes of livestock and game-stock losses, trophy hunting, livestock and game-stock management, photographic safaris, translocation and lethal control of ‘problem’ predators and the effectiveness of methods to enable predators and ranchers to coexist (Appendix 3). Three versions of the questionnaire were used depending on the species farmed: livestock, livestock and game-stock (referred to as L/G) and only game-stock. The L/G questionnaire was a combination of the livestock specific and game-stock specific questions. Questionnaires were completed as face-to-face interviews or were self-completed by the farm owner or manager. Sections A-J were mainly comprised of closed-ended questions; some open-ended questions were included for further comments or explanations. Section K consisted of yes/no questions regarding the lethal removal of predators and was conducted using the RRT (as described in Chapter 7). The original questionnaire was piloted with 10 farmers and one change was made; farmers’ assessment of the problems facing the farm was altered from a ‘ranking’ to a ‘score out of 10’. No other major changes were made to the questionnaire; therefore, data from the pilot study were included in the statistical analyses.

Before commencement of the interview, the interviewer explained the project, stated that all answers were confidential and gained written consent to conduct the questionnaire (Appendix 1 and 2). Farmers were then shown a predator ID sheet (Appendix 5) and were asked to name the predators pictured. If farmers were unsure of the English name for a predator species, the interviewer additionally referred to the predator in Afrikaans or Setswana, for the duration of the questionnaire. For self-completed questionnaires the interviewer established understanding and discussed any ambiguous answers or confusions before and after completion of the questionnaire, face-to-face or by telephone. Farmers were given a copy of the questions in their preferred language (English or Afrikaans) and the interviewer read the questions aloud in English and recorded the answers and any additional comments on a separate answer sheet. A translated copy of the questionnaire was not available in Setswana (the national language). However, the official language of Botswana is English, and because the questionnaires were targeted at commercial farmers, who generally had an excellent understanding of English, this was not problematic.

GPS locations of the interviewed farms were recorded and farm boundaries were mapped using national GIS data (Department of Wildlife and National Parks, n.d.) in ArcView 3.2 (Environmental Systems Research Institute Inc. 1992 – 2000). The shortest straight line distance from the boundary of each farm to the nearest protected area was calculated in kilometres; the annual rainfall, human population density and vegetation type were assigned by reference to national GIS data, from the Botswana DWNP (Department of Wildlife and National Parks, n.d.) and from the Centre for International Earth Science Information Network (CIESIN, 2012). When questionnaires were completed with multiple representatives from the same farm (e.g., owner and manager or livestock manager and game-stock manager), farm characteristics such as the number of stock owned or the farming techniques used were only recorded once. When farmers owned multiple farms, details were collected for the farm which they were most familiar, normally the one they were living or working on. As small antelope species (i.e. steenbok, duiker and/or bushbuck) were present on livestock farms and game ranches nationwide, they were used as a relative indicator of free-ranging wildlife biomass. If farmers reported that they saw a predator species or found it's spoor at least once every three months, the species was recorded as 'present' on the farm. If the predator was reported as being seen less frequently than every three months it was recorded as 'absent' (Chapter 5).

Participants' tolerance of predators was measured as a 'tolerance score' (TS), based on methods adapted from Romañach *et al.* (2007). Farmers were asked to state the hypothetical number of cattle and/or game-stock they would tolerate losing to a predator (in a single calving season, assumed to be a three month period), before they would remove the predator from their farm. Participants were asked to state whether they would remove the predator upon first sight (i.e. before any stock had been killed; score: zero), after the first loss (score: one), after multiple losses (score: two to four) or never (score: five). If farmers stated multiple losses, they were asked to specify how many, which was later coded into three categories (two to five [score: two], six to 10 [score: three] and equal to or greater than 11 [score: four]). If participants failed to provide a specific number for multiple losses (occurred in 8.3 - 14.0% of cases), they were allocated the mid value; the six to 10 category. A tolerance score was calculated for each large predator species for both cattle and game-stock, hypothetically in the case of species that were not present on the farm and a mean tolerance score was calculated. The questionnaire posed two independent measures of HPC: in section B farmers were asked to score how large a problem they thought that predators were to their farm (from one [small problem] to ten [large problem]) and in section C they were asked to state how 'happy' they were sharing their land with predators on a five point scale; 'very happy', 'happy', 'neutral', 'unhappy' to 'very unhappy'. The correlation between 'tolerance scores' and these independent measures of conflict was tested to ensure the former was a reliable measure. Participant and farm characteristics, predators present (discussed in Chapter 5) and stock management techniques (discussed in Chapter 7) were compared with the reported number of livestock and game-stock killed by predators. To establish the potential drivers of tolerance of predators these factors in addition to the reported number, proportion and financial costs of stock losses were compared with tolerance scores.

4.3.3 Statistical analyses

All data were coded for use in the statistical computer package SPSS (version 11.0.1; SPSS Inc.) or Genstat (version 15; VSN International). The Kolmogorov-Smirnov or Shapiro Wilk tests were used to determine if continuous variables were normally distributed. T-tests or Mann-Whitney U-tests were used when comparing continuous variables with a categorical variable with two states. For categorical variables with multiple states an ANOVA or Kruskal-Wallis ANOVA with *post hoc* testing (Turkey HSD or Mann-Whitney U-tests with the Bonferroni correction) was utilised. The Welch correction was used for t-tests and ANOVA (with Games-Howell *post hoc* testing) when variables had non-homogenous variances. A paired samples t-test or the Wilcoxon signed rank test was used to compare variables with paired results. The correlation between two continuous variables was compared with Spearman's correlation. Chi-squared tests using Yate's correction factor for tests with one degree of freedom were used to compare proportions. Significant categories in Chi-squared tests were defined as those with a standardised residual greater than ± 1.96, equivalent to a probability of 0.05.

Multiple regression using a general linear model was used to assess the importance of various factors and covariates on tolerance scores to predators and the reported number of game-stock and cattle killed by predators. The model which accounted for the most variation in tolerance scores, whilst all individual variables remained significant, was considered the most appropriate model. Means are presented with their standard deviations (SD). All tests were two-tailed.

4.4 Results

4.4.1 Farm and participant characteristics

Questionnaire surveys were conducted with 44 ranchers farming only game-stock on 43 ranches and 44 L/G farmers based on 37 game ranches. An additional 27 livestock farmers were questioned on 27 livestock farms (Table 4.1). From the original list of 111 registered game ranches in Botswana, 109 were in existence at the time of survey. As some participants represented multiple registered game ranches, at least one representative from 86.2% of the game ranches in existence was questioned. Of the 15 non-responses, contact details were not available for two of the farms and 13 farms either explicitly refused to participate (two farms) or did not take part (i.e. a suitable time to interview them could not be arranged and/or paper-questionnaires were not completed). Eighty-nine percent (88.7%, $n = 115$) of the questionnaires were conducted as face-to-face interviews, and 11.3% of surveys were self-completed by farmers; reported tolerance scores (TS) to predators did not differ between the questionnaire methodologies ($U(112)$, $z = -1.43$, $p = 0.152$; TS were not calculated for three participants). Interviews were primarily conducted with the owner (65.2%) of the farm. The majority of questionnaire surveys were conducted in the Central (30.8%) and Ghanzi (29.0%) districts (i.e. those districts with the largest number of game ranches; Fig. 2.13, Table 4.1).

The majority of participants were male (89.6%); most spoke Afrikaans (46.1%) or English (25.2%) as their first language and the mean age was 47.2 years old (SD = 12.4, range 22 - 74). The farmers' primary religion was Christianity (78.0%) and 61.9% of farmers had tertiary education. Livestock accounted for 63.0% of the income earned on L/G farms; the remainder was derived from game-stock ranching activities (34.4%; primarily hunting for meat and game meat sales) and the

Table 4.1. Differences in farm characteristics between the surveyed livestock farms, game-stock only ranches and ranches farming livestock and game-stock (L/G). Significant results were determined by the Chi-squared test and the Mann-Whitney U-test, and are marked with an '*'. Significant variables in *post hoc* testing are marked with α or β

		All farms	Livestock	Game-stock only	L/G	Statistic	df/n	Prob
Sample size	Farmers	115	27	44	44			
	Farms	107	27 (‡26)	43 (‡38; §41; λ39)	37 (‡35; λ36)			
		Proportion	**Proportion**	**Proportion**	**Proportion**			
Farms in district ƚ	Southern	7.5%	0.0%	14.0%	5.4%	χ^2 = 8.15	4	0.086
	Ghanzi	29.0%	44.4%	11.6%	37.8%			
	Kgalagadi	6.5%	0.0%	7.0%	10.8%			
	North East	10.3%	11.1%	14.0%	5.4%			
	Ngamiland	15.9%	14.8%	20.9%	10.8%			
	Central	30.8%	29.6%	32.6%	29.7%			
Farms conducting commercial activity ¶ (average proportion of farm's income generated from activity)	No commercial use	12.5%	0.0%	23.3%	2.7%	ƚƚ		
	Livestock farming	98.4%	100.0% (100%)	-	97.1% (63.0%)	χ^2 = 0.00	1	1.000
	Trophy hunting	42.5%	-	41.9% (22.8%)	43.2% (4.9%)	χ^2 = 0.00	1	1.000
	Hunting for meat	46.3%	-	34.9% (11.4%)	59.5% (11.2%)	χ^2 = 3.89	1	0.048*
	Bow hunting	21.3%	-	20.9% (5.1%)	21.6% (0.8%)	χ^2 = 0.00	1	1.000
	Photo-tourism	50.0%	-	62.8% (36.0%)	35.1% (8.0%)	χ^2 = 5.03	1	0.025*
	Live game sales	23.8%	-	23.3% (6.6%)	24.3% (2.8%)	χ^2 = 0.00	1	1.000
	Game meat sales	41.3%	-	32.6% (5.6%)	51.4% (6.7%)	χ^2 = 2.18	1	0.140
	Agricultural crops	4.7%	11.1%	2.3% (1.9%)	2.7% (2.6%)	ƚƚ		
	Other	5.6%	0.0%	11.6% (10.6%)	2.7% (0.0%)	ƚƚ		
Household income farm generated ‡	0 ≤ 25%	57.6%	42.3%	84.2%$^{\alpha}$	40.0%	χ^2 = 17.94	2	< 0.001*
	> 25%	42.4%	57.7%	15.8%$^{\alpha}$	60.0%			
		Median (range)	**Median (range)**	**Median (range)**	**Median (range)**			
Farm size (km^2) §		71.5 (3 – 2000)§§	72.0 (16 – 870)	50.0 (3 - 600)	85.0 (5 - 2000)	χ^2 = 5.10	2	0.078
Number of cattle ‡		400 (0 – 9000)	486 (28 - 9000)	-	400 (0 - 6500)	z = −0.60	2	0.546
Number of game-stock λ		1216 (97 – 26,447)	-	1313 (97 - 11,729)	1217 (109 - 26,447)	z = −0.51	72	0.607
Number of years working on farm		8.0 (0 – 43)	16.0 (1 – 42)$^{\beta}$	5.5 (0 – 27)$^{\alpha\beta}$	11.5 (0 – 43)$^{\alpha}$	χ^2 = 11.37	2	0.003*

ƚ Statistical comparison refers to Central, Ngamiland and Ghanzi districts; Southern refers to the Southern, South-east, Kweneng and Kgatleng regional districts

‡, §, λ Sample sizes varied as stated due to non-response

¶ Most farms were conducting multiple activities, therefore, totals do not add up to 100%

ƚƚ Violates the assumptions of Chi-squared test production of agricultural crops (2.6%; Table 4.1). Photographic tourism (36.0% of income) and trophy hunting (22.8%) were the main sources of income on ranches farming only game-stock; however, 23.3% of game-stock only ranches were not conducting any commercial activities (Table 4.1). Income from the farm contributed to more than 25% of the household income on 15.8% of game-stock only ranches, compared to 57.7% of livestock farms and 60.0% of L/G farms (χ^2 = 17.94, df = 2, $p < 0.001$; Table 4.1). The game ranches surveyed covered a total area of 11,771 km^2 and stocked ca. 142,350 wild ungulates. Exotic game species were stocked on 20.0% of farms and 2.5% of farms were breeding rare colour variants of ungulate species, specifically 'golden' wildebeest; 7.5% of farms stocked both blue- and black wildebeest (n = 80).

4.4.2 Problems facing farmers

Livestock farmers regarded an unreliable market (score: 4.4 out of 10) and the lack of government support (3.4/10) as the first and second biggest problem to their farm, followed by carnivore depredation (3.0/10; Fig. 4.1). Farmers on game-stock only ranches and L/G farms considered a

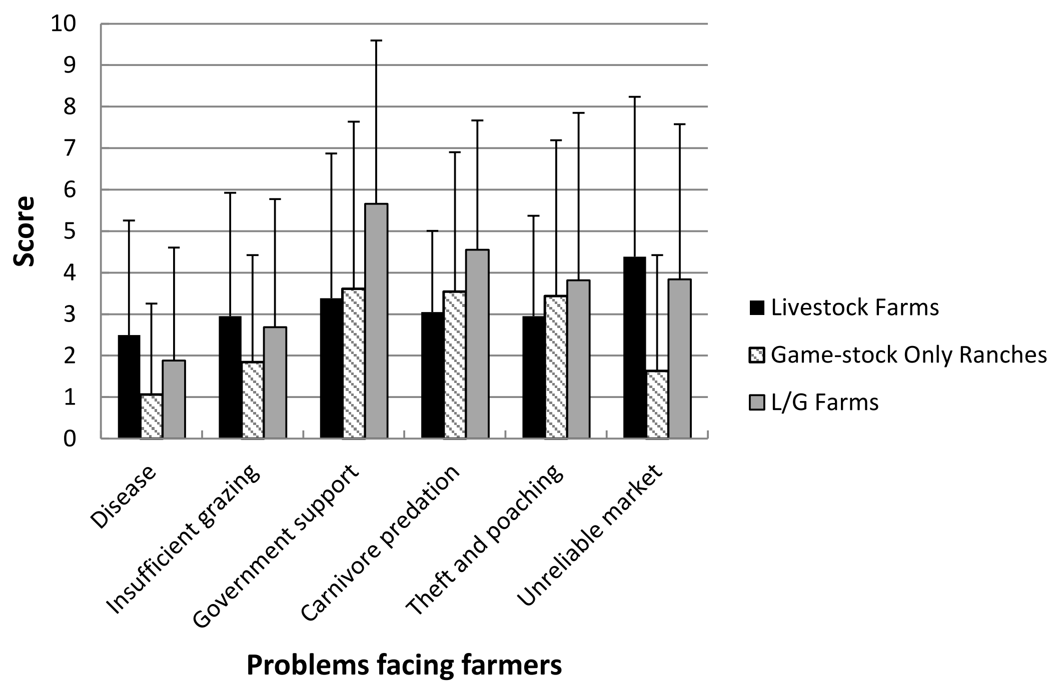


Figure 4.1 Problems facing livestock farmers (n = 21), game-stock only ranchers (n = 33) and farmers ranching both game-stock and livestock (L/G farmers, n = 30). Problems were scored from 0 to 10 (small - large problem); mean scores are shown and error bars represent one standard deviation

lack of government support (3.6/10 and 5.7/10, respectively) as the primary problem to the farm, followed by carnivore depredation (3.5/10 and 4.6/10, respectively; Fig. 4.1). Disease ($z = -2.94$, $df = 69$, $p = 0.003$) and unreliable markets ($z = -2.65$, $df = 69$, $p = 0.008$) were considered a significantly greater problem to the farm by livestock farmers than game ranchers. Game ranchers commented that changes to legislation proposed in the draft game ranching regulations in 2011 (DWNP, 2011b) could impose large financial costs to game ranches and could make the game ranching industry no longer viable. Additionally, from January 2014 hunting on state land in Botswana was banned (MEWT, 2013), rather than increasing the demand for trophy hunting on private land, ranchers predicted their business would decline as many of their customers primarily came to Botswana to hunt large-game species (e.g. elephants) on state land. Some farmers stated that they were considering selling their farm or returning to livestock farming. Other problems reported were conflict with elephants (14.2% of farms), fire (11.5%), water supply (0.8%), drought (0.8%), lack of funding (0.8%) and the HIV prevalence rate amongst staff (0.8%).

4.4.3 Direct costs of carnivore depredation

From the total number of livestock and game-stock losses reported to have occurred in the 12 months prior to the survey, predators were described as causing 34.3% of cattle, 37.2% of smallstock and 63.9% of game-stock losses (Table. 4.2). Predation was considered the primary cause of livestock and game-stock losses, followed by disease in livestock and theft/poaching in game-stock (Table. 4.2). The majority of farms reported losing cattle (83.1%), smallstock (68.8%) or game-stock (93.2%) to predators in the 12 months prior to the survey (Table 4.3). Predators

Table 4.2. Reported cause of cattle (n = 54), smallstock (n = 26) and game-stock (n = 37) losses as a proportion of total losses, on commercial farmland in Botswana

Cause of loss	Smallstock	Cattle	Game-stock
Predators	37.2%	34.3%	63.9%
Calving problems	7.3%	15.2%	0.6%
Disease	31.1%	22.9%	0.4%
Poisonous plants	2.0%	3.7%	1.2%
Starvation	6.4%	6.7%	10.7%
Theft/poaching	14.2%	11.3%	21.2%
Fire	0.0%	0.6%	0.0%
Other	1.8%	5.2%	2.0%

were reported to have killed a median of seven cattle (1.0% of the farm's cattle herd), four smallstock (3.6% of a farm's smallstock herd) and 43 game-stock (3.0% of the farm's total game-stock) per farm in the 12 months prior to study (Table 4.3). Farmers were 27 times more likely to answer 'do not know' to the number of game-stock lost to predators, than to the number of cattle lost to predators ($\chi^2 = 22.70$, df = 1, $p < 0.001$). Farmers reported they found 50.0% of game-stock carcasses and 92.5% of livestock carcasses (U(56), $z = -4.01$, $p < 0.001$; Table 4.3).

The reported number of livestock and game-stock lost to predators was positively correlated with the financial losses farmers' estimated that had been caused by carnivore depredation (livestock: $R_s = 0.73$, n = 55, $p < 0.001$; game-stock: $R_s = 0.59$, n = 40, $p < 0.001$). Ranchers farming only game-stock reported economic losses of US \$2.1 per hectare; seven times greater than livestock farmers (US \$0.3/ha; U(51), $z = -3.07$, $p = 0.002$) and 3.5 times greater than L/G farmers (US \$0.6/ha; U(57), $z = -2.58$, $p = 0.010$). The median reported financial loss due to livestock losses to carnivore depredation, per farm in the 12 months prior to study was US \$3399; 1.9 times smaller than reported due to game-stock losses (US \$6536; Table 4.3). Total financial losses to carnivore depredation of over BWP 250,000 (US \$32,684) were reported on 20.0% of game-stock only ranches (n = 40) and on none of the L/G farms (n = 36) or livestock farms (n = 25). These large

Table 4.3. The direct costs of cattle, smallstock and game-stock losses to carnivore depredation in the 12 months prior to the survey (2011 – 2012), as reported by commercial livestock farmers and game ranchers in Botswana

	Cattle	**Smallstock**	**Game-stock**
Proportion of farms that reported losing animals	83.1% n = 59	68.8% n = 32	93.2% n = 44
Median number of animals lost (range)	7 (0 – 100) n = 59	4 (0 – 30) n = 32	43 (0 – 1275) n = 44
Median number of animals lost as a proportion of total animals owned (range)	1.0% (0.0 – 27.8%) n = 57	3.6% (0.0 – 20.2%) n = 30	3.0% (0.0 – 85.0%) n = 42
Median financial cost of losses (range)	US \$3399 (0 – 32,684), n = 55 BWP26,000; (0 – 250,000)		US \$6536 (0 – 235,325) n = 53 BWP50,000; (0 – 1,800,000)
Median proportion of carcasses found (range)	Livestock: 92.5% (n = 25)		50.0%; (n = 31)

losses were calculated by farmers based on the predation of expensive game-stock species such as sable antelope *Hippotragus niger* (one incidence) or based on estimates as to the number of predators on the farm, the frequency with which they kill and their preferred prey species, in conjunction with observed declines or failures to increase in game-stock numbers.

The number of smallstock reported to have been killed by predators was 4.8 times lower on farms where small antelope species (e.g., steenbok, duiker and/or bushbuck) were seen daily (median 3.0, range 0 - 26, n = 25) than on farms where they were seen less than daily (median 14.5, range 6 - 30, n = 6; U(31), z = −2.60, p = 0.009). The number of cattle or game-stock killed was associated with the number of cattle or game-stock owned (which correlated with the size of the farm), the predators present, the livestock or game-stock management techniques used and the farmers' age and farming experience (Table 4.4 and 4.5). Comparisons of multiple regression models identified the number of cattle owned and the presence/absence of African wild dogs, lions and/or spotted hyaenas (Awd_li_sh) as the most significant factors to predict the number of cattle killed by predators (f = 9.36, df = 2, p < 0.001, adj. R^2 = 0.23). The regression equation was:

y = (0 [**Awd_li_sh absent**] + 0.40 [SE = 0.13; **Awd_li_sh present**]) + 0.0002 (SE = 0.00) * **number of cattle owned** + 0.48 (SE = 0.11)

Table 4.4. Factors associated with the number of cattle killed by predators as reported by commercial farmers in Botswana. Variables in blue are those factors which were included in the final regression model ɫ

Factor	Median number of cattle killed (n; range)		Statistic	n	Prob
Distance to protected area	≤ 100 km: 10.0 (25; 0 − 100)	> 100 km: 4.8 (26; 0 − 60)	z = −2.03	51	0.042
African wild dogs, lions and/or spotted hyaenas present	Present: 10.0 (33; 0 − 100)	Absent: 4.3 (26; 0 − 45)	z = −2.83	59	0.005
Kraal calves ≥ 3 months ǂ	Use: 4.0 (7; 0 − 18)	Do not use: 15.0 (17; 0 − 100)	z = −2.23	17	0.026
No of cattle owned			R_s = 0.48	59	< 0.001
Farm size (km^2)			R_s = 0.39	59	0.003
Farmers' age			R_s = −0.30	59	0.021

ɫ The model which accounted for the most variation in tolerance scores, whilst all individual variables remained significant Tolerance scores were transformed using: log10 (χ + 1); one participant was removed as an outlier.

ǂ Sample restricted to farms within 100 km of a protected area (refer to Chapter 7 for further information on livestock husbandry)

Table 4.5. Factors associated with the number of game-stock killed by predators as reported by game ranchers in Botswana. Variables in blue are those factors which were included in the final regression model ł

Factor	Median number of game-stock killed (n; range)		Statistic	n	Prob
Lions present	Present: 175.0 (10; 5 – 1275)	Absent: 27.5 (34; 0 – 450)	z = −2.44	44	0.015
Perimeter-fencing electric or bonnox ǂ	Use: 12.5 (4; 8 – -25)	Do not use: 85.5 (20; 4 – 1275)	z = −2.05	24	0.040
No of game-stock owned			R_s = 0.47	42	0.002
Farm size (km^2)			R_s = 0.53	43	< 0.001
Farmers' game ranching experience (years)			R_s = 0.50	44	0.001

ł The model which accounted for the most variation in tolerance scores, whilst all individual variables remained significant. Tolerance scores were transformed using: log10 (χ + 1)

ǂ Sample restricted to farms within 100 km of a protected area (refer to Chapter 7 for further information on livestock husbandry)

Similarly, the number of game-stock killed by predators was best predicted by the presence/absence of lions and the number of years of game ranching experience (f = 9.87, df = 2, $p < 0.001$, adj. $R^2 = 0.29$). The regression equation was:

y = (0 [**Lion absent**] + 0.59 [SE = 0.24; **Lion present**]) + 0.03 (SE = 0.01) * **number of years of game ranching experience** + 0.98 (SE = 0.18)

4.4.4 Predator species causing stock losses

Nationwide the majority of cattle losses were reported to have been caused by leopards (50.6%), followed by cheetahs (15.8%; Table. 4.6). Black-backed jackals were reported to have caused the majority of smallstock losses (74.4%), and leopards (32.7%) followed by cheetahs (23.6%) were reported to have caused the majority of game-stock losses (Table. 4.6).

4.4.5 Total tolerance scores to predators

Farmers' tolerance scores (TS) to predators correlated with how 'happy' or 'unhappy' farmers reported they were to share their land with predators ($R_s = -0.55$, n = 112, $p < 0.001$) and with the size of the problem farmers stated that predators were causing to their farm ($R_s = -0.32$, n = 87

Table 4.6. Proportion of cattle (n = 49), smallstock (n = 19) and game-stock (n = 54) losses to carnivore depredation reported to have been caused by each predator species

Predator species	Smallstock	Cattle	Game-stock
African wild dog	7.1%	11.0%	7.4%
Baboon	6.1%	0.2%	0.8%
Black-backed jackal	74.4%	3.9%	12.8%
Brown hyaena	0.0%	2.6%	6.5%
Caracal	4.9%	0.6%	2.3%
Cheetah	4.3%	15.8%	23.6%
Domestic dog	0.6%	1.0%	2.3%
Leopard	0.3%	50.6%	32.7%
Lion	0.0%	9.1%	7.4%
Spotted hyaena	2.4%	5.1%	4.2%

$p = 0.003$). L/G farmers (TS = 3.5, range 0.6 – 5, n = 44) had significantly greater tolerance scores to predators than livestock farmers (TS = 2.6, range 0 - 4.5, n = 27; U(71), $z = -2.83$, $p = 0.005$). Ranchers farming only game-stock had significantly greater tolerance scores to predators (TS = 4.4, range 0 – 5, n = 41) than livestock farmers (U(68), $z = -4.18$, $p < 0.001$) and L/G farmers (U(85), $z = -3.05$, $p = 0.002$). Farmers' reported tolerance of predators varied between individuals, from those who considered that carnivore depredation was part of owning game-stock, to those who were concerned that predators were making their ranch financially unviable. Tolerance scores of less than one (i.e. would remove all predators species after or before the first livestock or game-stock loss) were reported by 25.9% of livestock farmers and 5.7% of game ranchers. Farmers had greater tolerance scores to all predator species in relation to game-stock losses as opposed to cattle losses (Fig. 4.2).

4.4.6 Tolerance scores to specific predator species

Overall, African wild dogs and lions were the least tolerated of the predator species and had the lowest tolerance scores; conversely brown hyaenas and caracals were the most tolerated of the predator species (Table 4.7). For cattle losses, spotted hyaenas were the third least tolerated species, followed by leopards, black-backed jackals then cheetahs; for game-stock losses black-backed jackals were the third least tolerated predator species followed by cheetahs, spotted hyaenas then leopards (Table 4.7). L/G farmers had significantly greater tolerance scores for cattle losses caused by leopards (median TS: 2.0, n = 41) and brown hyaenas (TS: 5.0, n = 37) than

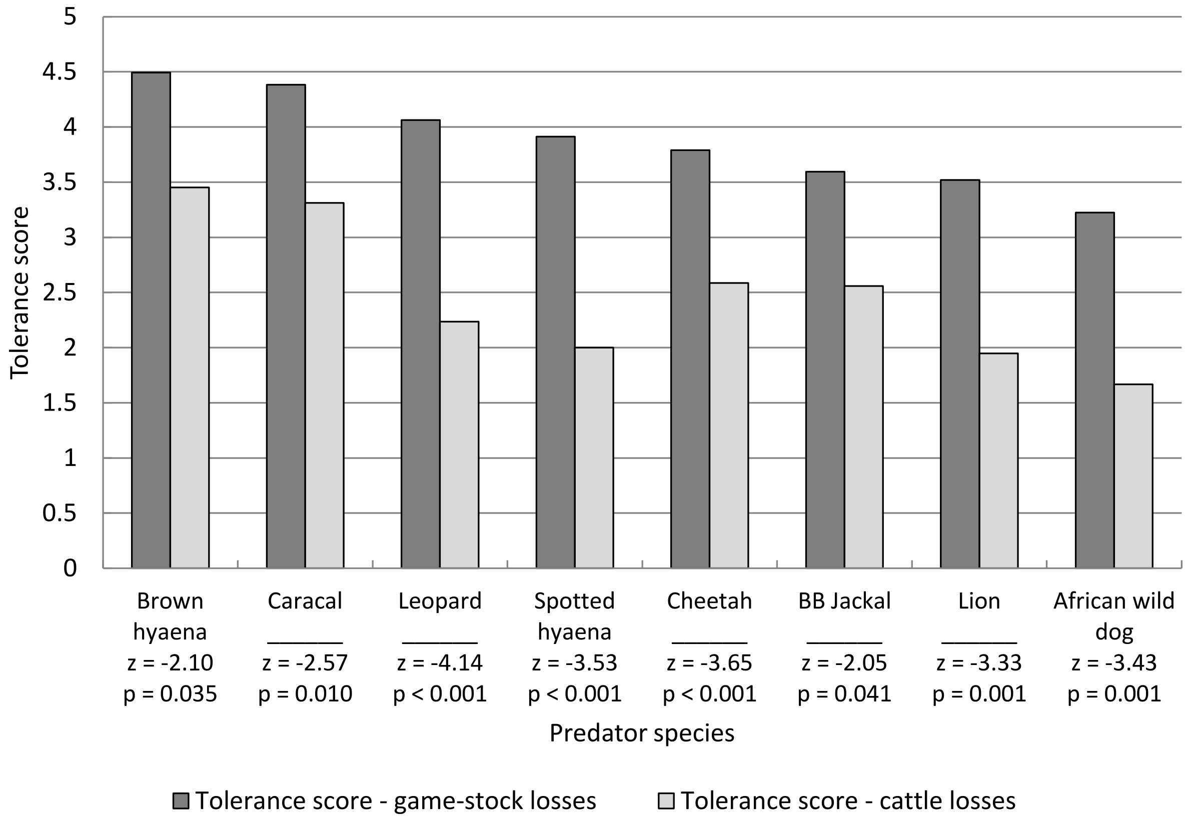


Figure 4.2. Differences in farmers' reported tolerance of predator species killing cattle compared to game-stock. Tolerance scores were scored from 0 – 5, (low to high tolerance); BB Jackal refers to black-backed jackal

livestock farmers (leopard: TS = 1.0, n = 27; U(68), z = −2.00, p = 0.046; brown hyaena: TS = 3.0, n = 25; U(62), z = −2.24, p = 0.025).

4.4.7 Drivers of tolerance of predators on livestock farms

Farmers' reported tolerance scores to predators were positively correlated with the farmer's age (R_s = 0.42, n = 27, p = 0.027). Farmers with tertiary education (TS = 3.0; z = −2.30, df = 27, p = 0.021), or farmers based on farms where small antelope numbers were perceived to have increased in the 10 years prior to the study (TS = 3.2; U(14), z = −2.61, p = 0.009) had greater tolerance scores to predators than individuals without tertiary education (TS = 1.6), or those based on farms where small antelope numbers were perceived to have decreased (TS = 0.6). Farmers whose first language was an indigenous African language had lower tolerance scores (TS = 1.1) than individuals whose first language was Afrikaans (TS = 2.7) or a European language (including

Table 4.7. Farmers reported tolerance score (TS) to each predator species causing cattle or game-stock losses on commercial farmland in Botswana Tolerance scores were scored from 0 – 5, (low to high tolerance). Significance testing was conducted with the Wilcoxon Signed Ranks Test.

Cattle losses Tolerance Score

Predator	Median (n); mean (SD)	Less tolerated than	z	n	Prob
Brown hyaena	4 (62); 3.45 (1.78)	None			
Caracal	4 (61); 3.31 (1.79)	None			
Cheetah	3 (63)	Caracal	−3.03	59	0.002
	2.59 (1.74)	Brown hyaena	−2.35	59	0.019
BB Jackal ł	3 (59)	Brown hyaena	−2.28	57	0.022
	2.56 (2.01)	Caracal	−2.35	57	0.019
Leopard	2 (68)	Brown hyaena	−4.08	62	< 0.001
	2.24 (1.55)	Caracal	−4.08	61	< 0.001
Spotted hyaena	2 (55)	Brown hyaena	−4.35	55	< 0.001
	2.00 (1.66)	Caracal	−3.55	54	< 0.001
		Cheetah	−2.67	52	0.008
Lion	2 (57)	Brown hyaena	−4.21	54	< 0.001
	1.95 (1.76)	Caracal	−3.54	53	< 0.001
		Cheetah	−2.91	56	0.005
		BB Jackal	−2.32	59	0.020
African wild dog	1 (60)	Brown hyaena	−4.45	58	< 0.001
	1.67 (1.81)	Caracal	−4.26	57	< 0.001
		Cheetah	−3.92	58	< 0.001
		BB Jackal	−2.82	55	0.005
		Leopard	−2.26	60	0.024

Game-stock losses Tolerance Score

Predator	Median (n); mean (SD)	Less tolerated than	z	n	Prob
Brown hyaena	5 (77); 4.49 (1.21)	None			
Caracal	5 (73); 4.38 (1.33)	None			
Leopard	5 (78)	Brown hyaena	−2.05	77	0.041
	4.06 (1.50)				
Spotted hyaena	5 (70)	Brown hyaena	−2.76	70	0.006
	3.91 (1.72)	Caracal	−2.46	70	0.014
Cheetah	5 (76)	Brown hyaena	−2.60	76	0.009
	3.79 (1.81)	Caracal	−2.08	73	0.037
BB Jackal	5 (74)	Brown hyaena	−3.30	74	0.001
	3.60 (1.96)	Caracal	−2.77	73	0.006
		Leopard	−2.34	74	0.019
Lion	5 (71)	Brown hyaena	−3.32	71	0.001
	3.52 (1.99)	Caracal	−2.76	71	0.006
		Leopard	−2.64	71	0.008
		Spotted hyaena	−2.04	70	0.042
African wild dog	5 (76)	Brown hyaena	−4.14	76	< 0.001
	3.22 (2.19)	Caracal	−3.98	73	< 0.001
		Leopard	−3.15	76	0.002
		Cheetah	−2.56	76	0.010
		Spotted hyaena	−2.01	70	0.045

ł BB Jackal = Black-backed jackal

English) (TS = 3.3; χ^2 = 7.6, df = 2(27), p = 0.023). Tolerance scores to predators was negatively correlated with the reported number of livestock killed by predators in the past 12 months (R_s = −0.44, n = 27, p = 0.022) and with the financial costs of carnivore depredation on livestock (appr. sig. Table 4.8). Changes in small antelope numbers, the farmers' age and education level were the variables which accounted for the most variation in tolerance scores (f = 6.46, df = 4, p = 0.002, adj. R^2 = 0.49). The regression equation for this model was:

y = (0 [**small antelope perceived to be increasing**] −1.76 [SE = 0.59; **decreasing**] −0.58 [SE = 0.50; **stable**]) + (0 [**educated < tertiary**] + 1.19 [SE = 0.45; **educated ≥ tertiary**]) + 0.05 (SE = 0.02) * **farmers' age** - 0.18 (SE = 1.15)

4.4.8 Drivers of tolerance of predators on game ranches

Farmers who did not practise a formal religion (TS = 4.4), who were working on smaller ranches or worked on ranches located in the Central district of Botswana (TS = 4.4) had greater tolerance scores to predators than farmers who were Muslim (TS = 2.0; U(16), z = −2.69, p = 0.007), whose farms were larger (R_s = −0.25, n = 81, p = 0.022) or were located in the Ghanzi district (TS = 3.1; U(51), z = −3.46, p = 0.001). Farmers had lower tolerance scores to predators if they were based on farms which ranched large numbers of springbok (R_s = −0.40, n = 83, p < 0.001), on farms with sandveld vegetation (TS = 3.4, U(84), z = −1.98, p = 0.048), on farms where cattle (TS = 3.3, U(85),

Table 4.8. Variables shown to be potential drivers of human-predator conflict on commercial livestock farms in Botswana. Factors marked with an 'α' or 'β' were significant in post-hoc testing. Variables in blue are those factors which were included in the final regression model (i.e. the model which accounted for the most variation in tolerance scores, whilst all individual variables remained significant)

Variable	TS Score (n)	TS Score (n)	TS Score (n)	Statistic	df /n	Prob
Small antelope numbers	Increased:	Stable:	Decreased:	χ^2 = 8.63	2	0.013
changed	3.2 (9)$^{\alpha}$	1.6 (10)	0.6 (5)$^{\alpha}$	$^{\alpha}$ z = −2.61	14	0.009
Tertiary educated	Yes: 3.0 (15)	No: 1.6 (12)		z = −2.30	27	0.021
First language spoken	Afrikaans:	English/other European:	Indigenous African:	χ^2 = 7.56	2	0.023
				$^{\alpha}$ z = −1.97	23	0.048 ł
	2.7 (15)$^{\beta}$	3.3 (4)$^{\alpha}$	1.1 (8)$^{\alpha\beta}$	$^{\beta}$ z = −2.21	12	0.027 ł
Farmers' age (yrs.)				R_s = 0.42	27	0.027
Number of livestock lost to predators				R_s = −0.44	27	0.022
Financial cost of livestock losses to predators				R_s = −0.39	25	0.057

ł non-significant with Bonferroni correction

$z = -3.39$, $p = 0.001$) or cheetahs were present (TS = 3.5; appr. sig) or on farms that had a smaller proportion of buffer species (kudu and impala; as a proportion of total game-stock; $R_s = 0.37$, $n = 73$, $p = 0.001$; Table 4.9). Farmers who were earning less than 50% of the farm's income from photographic safaris (TS = 3.6), as opposed to more than 50% (TS = 4.5, U(85), $z = -2.27$, $p = 0.023$), or farmers who relied on the farm for more than 25% of their household income (TS = 3.5), as opposed to less than 25% (TS = 4.4, U(76), $z = -2.37$, $p = 0.018$), had lower tolerance scores to predators (Table 4.9). Tolerance scores were not related to the reported number ($R_s = -0.23$, $n = 44$, $p = 0.140$), proportion ($R_s = -0.11$, $n = 41$, $p = 0.511$) or financial cost ($R_s = -0.12$, $n = 54$, $p = 0.392$) of game-stock losses to carnivore depredation, nor to the ranching of exotic species (U(85), $z = -0.33$, $p = 0.742$). Comparisons of multiple regression models demonstrated that the presence/absence of cattle in conjunction with the proportion of buffer species ranched on the farm were the most important variables to significantly predict tolerance scores ($f = 19.24$, $df = 2$, $p < 0.001$, adj. $R^2 = 0.29$). The regression equation was:

y = (0 [**cattle absent**] - 0.91 [SE = 0.25; **cattle present**]) + 0.03 (SE = 0.01) * **buffer species proportion** + 2.79 (SE = 0.32)

4.5 Discussion

4.5.1 Limitations

The primary limitation of this study, common to those of other HWC interviews and questionnaires, was determining the internal validity of the data collected. This study was unable to verify the number of losses or the financial costs of the losses reported by farmers or their reported causes. It is often difficult and time-consuming to independently verify losses and facts (Dickman, 2008), consequently in a review of ecology questionnaires, less than 10% utilised ground-truthing (White *et al.*, 2005). Farmers often overestimate losses to predators (Dickman, 2008) and in Chapter 6 farmers perceptions of game-stock losses to cheetahs were 1.6 - 5.9 times greater than calculated from cheetah prey analysis through prey remains in scat. However, farmers' perceptions of losses can often be a more important driver of tolerance of predators than direct costs (Mishra, 1997; Madden, 2004). Therefore, it was appropriate for this study to examine these perceptions; however, a thorough cost-benefit analysis of the impact of predators on game-

Table 4.9. Variables shown to be potential drivers of human-predator conflict on game ranches in Botswana. Factors marked with an ‘α’ or ‘β’ were significant in post-hoc testing. Variables in blue are those factors which were included in the final regression model (i.e. the model which accounted for the most variation in tolerance scores, whilst all individual variables remained significant)

Variable		TS Score (n)		Statistic	df/n	Prob
District	North East: 3.8 (8)	Southern: 2.4 (7)	Ghanzi: 3.1 (21)α	χ^2 = 18.55	5	0.002
	Ngamiland: 4.4 (13)	Kgalagadi: 2.7 (6)	Central: 4.4 (30)α	$^{\alpha}$z = -3.46	51	0.001
Religion	Christian: 3.8 (52)	Muslim: 2.0 (4)α	None: 4.4 (12)α	χ^2 = 7.63 $^{\alpha}$z = −2.69	2 16	0.022 0.007
Cattle		Farmed: 3.3 (40)	Not Farmed: 4.4 (45)	z = −3.39	85	0.001
Cheetah		Present: 3.5 (52)	Absent: 4.3 (33)	z = −1.86	85	0.063
Vegetation		Sandveld: 3.4 (44)	Not sandveld: 4.0 (40)	z = −1.98	84	0.048
Farm income from photographic tourism		≥ 50%: 4.5 (14)	< 50%: 3.6 (71)	z = −2.27	85	0.023
Household income from farm		≤ 25%: 4.4 (46)	> 25%: 3.5 (30)	z = −2.37	76	0.018
Number of springbok owned				R_s = −0.40	83	< 0.001
Proportion of game-stock made up by buffer species ‡				R_s = 0.37	73	0.001
Farm size (km^2)				R_s = −0.25	81	0.022

ł One participant was removed as an outlier

‡ Kudu and impala

stock and livestock populations would be beneficial to validate these perceptions in the future (Winterbach *et al.*, 2012).

Similarly, it was not possible to determine the measurement validity of farmers’ tolerance scores in relation to when predators were actually removed from the farm. Tolerance scores to predators correlated with how happy farmers stated they were to share their farm with predators and with the size of the problem farmers scored predators to cause to their farm. However, there is not always a direct link between ‘attitudes, intent, knowledge and behaviour’ (McCleery, 2009). In Namibia studies have shown that farmers’ positive attitudes to predators and verbally expressed tolerance are not always reflected by a lower proportion of removals (Marker *et al.*, 2003d; Schumann *et al.*, 2008).

Non-response rate on game ranches was 13.8% which is less than the accepted margin of 15% (Lindner, 2002); therefore, the results can be considered to possess external validity for comparison with the Botswana game ranching industry as a whole. However, the use of 'snowball' sampling could have reduced the external validity of the livestock farm surveys and had the potential to introduce bias (Sadler *et al.*, 2010). Farmer's potentially suggested people they knew to be experiencing predator problems, which may have positively biased the recorded direct costs of carnivore predation on livestock and negatively biased farmers' tolerance to predators. However, overall farmers were asked to submit contact details for their neighbours, therefore, as contact lists for livestock farmers were not available and 'snowball' sampling has higher response rates than other sampling techniques, it was considered an appropriate method to identify and question the livestock farmers living adjacent to game ranches.

4.5.2 Direct costs of carnivore depredation

Farmers from all land uses rated predators as the next biggest problem to the farm after the lack of government support and unreliable markets. Predators were considered the biggest cause of livestock losses, followed by disease, both in this study and in Namibia (Stein *et al.*, 2010). As a proportion of the total livestock owned, 1.0% of cattle and 3.6% of smallstock were reported to have been killed by predators, which was at the lower end of the range stated for southern Africa (0.8 – 5.0%) (Butler, 2000; Marker *et al.*, 2003d; Schiess-meier *et al.*, 2007; Stein *et al.*, 2010; Thorn *et al.*, 2012).

The most influential factors affecting the number of cattle killed by predators was the total number of cattle owned and the presence/absence of African wild dogs, spotted hyaenas, and/or lions. Farms where these predators were present reported 2.3 times the number of cattle losses than farms where they were absent and African wild dogs and lions were the least tolerated of all large predator species. However, as these species were only present on 23.0 – 43.4% of farms, as opposed to leopards which were present on 92.9% of farms (Chapter 5), nationwide leopards were the predator species which were reported to cause the greatest overall proportion of cattle losses. Lions, African wild dogs and spotted hyaenas were most commonly located close to protected areas (Chapter 5). Potentially as a result, farms within 100 km of a protected area suffered 2.1 times greater cattle losses than farms located further than 100 km. Consequently, areas close to nationally protected areas are often considered ‘hot-spots’ of conflict (Hazzah, 2006; Schiess-meier *et al.*, 2007). In a review of the conservation of large African predators other factors that were thought to affect the frequency of livestock predation were predators’ prey preferences (discussed in Chapter 6), the presence of habitual livestock killers (‘problem’ animals), prey availability (discussed in this chapter and in Chapter 7), livestock husbandry (discussed in Chapter 7) and habitat differences (Winterbach *et al.*, 2012).

Predators were reported as the greatest cause of game-stock losses in Botswana. The median reported loss to carnivore depredation of 3.0% of a farm’s game-stock herd in the previous 12 month period prior to study was larger than the losses of 1.9% reported in South Africa (Swanepoel, 2008). Thorn *et al.* (2012) calculated that the costs of carnivore depredation on game-stock were equivalent to 0.22 - 0.29% of game ranching net profit in South Africa. In Botswana predators were perceived by farmers to cause a financial loss of US $6536 in game-stock losses per year. Data were not available on the financial profit derived from game ranching, however, 84.2% of ranchers farming only game-stock were deriving less than 25% of their household income from the ranch; therefore, carnivore depredation is potentially having a larger impact on profit margins in the developing Botswana industry than in the more developed South African one. However, in light of the evidence from Chapter 6 that game ranchers over estimate game-stock losses compared to results from scat analysis the extent of this impact it difficult to establish.

The number of game-stock reported by farmers to be lost to predators was associated with the presence of lions (reported losses were 6.4 times greater if lions were present) and the number of years of experience the farmer had with ranching game-stock. Game-stock carcasses were more difficult to find than livestock carcasses. Therefore, some farmers estimated the number and financial cost of game-stock lost to carnivore depredation based on perceived decreases or failures to increase in game-stock herd size (also observed in ranchers’ decision making in South Africa [Thorn *et al*., 2012]), in conjunction with their perceptions of predators. These perceptions included the number of predators present, the prey preferences of predators and the frequency with which predators kill. Hazzah (2006) found that the longer people were resident in an area the more negative their attitudes and perceptions to predators were. Farmers' increased experience and cumulative exposure to carnivore depredation, in association with entrenched attitudes to predators could, therefore, cause farmers to overestimate losses.

Farmers in Tanzania reported livestock losses to carnivore depredation 4.6 times greater than calculated during monthly data collection (Dickman, 2008). Similarly, in Chapter 6 farmers perceptions of game-stock losses to cheetahs were 1.6 - 5.9 times greater than calculated from cheetah scat analysis. Farmers potentially overestimate losses, especially to high profile species, due to misidentification of tracks and signs at the carcass (Lindsey *et al.*, 2013b), due to negative attitudes to predators or to put pressure on the government to pursue predator control or to increase compensation payments (Gillingham & Lee, 2003; Gusset *et al.*, 2008a). Predation can elicit more resentment than the cumulative damage caused by chronic problems like disease or drought (Dickman, 2008). For example, in western Botswana cattle were four times more likely to die of disease than carnivore depredation, yet in questionnaires with farmers predators were listed as the primary cause of losses (Muir, 2012). Although farmers are generally competent at identifying predator signs (Dickman, 2008), confirmation bias can cause farmers to subconsciously seek out information, consistent with their pre-existing beliefs regarding carnivore depredation and discard contradictory information (Nickerson, 1998; Hudenko, 2012). Consequently predators are often blamed for unexplained stock losses (Gusset *et al.*, 2009). Research into observed decreases in herd size of tsessebe, roan antelope *Hippotragus equinus* and cattle, which were initially blamed on predators, discovered climatic factors, positioning of water-points and poaching to have been the major causal influences (Harrington *et al.*, 1999; Rasmussen, 1999; Dunham *et al.*, 2003).

4.5.3 Tolerance scores and drivers of tolerance of predators

Farmers operating game-stock only ranches were the most tolerant of predators (TS = 4.4) followed by L/G farmers (TS = 3.5) then livestock farmers (TS = 2.6). This higher level of tolerance occurred despite game ranchers reporting up to seven times greater financial losses to carnivore depredation (per hectare) than livestock farmers. This disproportionate level of intolerance of predators on farms ranching cattle, relative to the financial losses experienced, could reflect the emotional value and cultural significance of livestock, which can often be greater than its economic value (Loveridge, 2005). Additionally, ranchers farming only game-stock were more likely to have alternative sources of household income other than the farm, than were L/G farmers or livestock farmers. A factor which has been associated with a greater tolerance of predators in Kenya (Romañach *et al.*, 2011). Lindsey *et al.* (2013b) found that the likelihood that farmers would want predators on their ranch was greater as the proportion of income earned from wildlife increased. Game-stock only ranches generated 87.5% of their income from wildlife related activities, the primary use of which was photographic tourism and trophy hunting, and farmers conducting these activities commonly report that predators have economic value to them (Kent, 2011). By comparison the majority of income on L/G farms came from livestock farming (63.0%) and agricultural crop production (2.6%); game ranching activities contributed the remaining 34.4% of income, primarily from hunting for meat or live game sales. These activities are considered less lucrative than trophy hunting or photographic tourism; they require a larger off-take of game-stock, predators have no economic value to the farm and any game-stock animals killed by predators are likely to be perceived as a loss. Consequently, in South Africa and Zimbabwe attitudes to African wild dogs were more negative on farms whose income was based on hunting for meat than on ecotourism (Lindsey *et al.*, 2005c).

However, farmers reported tolerance of predators varied between individuals; a minority of farmers on both livestock farms (25.9%) and game ranches (5.7%) reported overall tolerance scores of less than one, indicating they would be likely to remove most or all predator species on first sight. The main drivers of tolerance on livestock farms were the farmers' age and education level, in conjunction with perceived changes in small antelope numbers, accounting for 49% of variability in tolerance scores. The stocking of cattle and the proportion of buffer species owned as a proportion of the total game-stock (commonly kudu and impala [Chapter 7]) explained 29% of the variability in tolerance scores to predators on game ranches. These findings support previous studies that a farmer's attitudes and perceptions of predators, influenced by age, education and social background have a larger impact on HPC and tolerance of predators than the direct costs of carnivore depredation (Madden, 2004; Swanepoel, 2008; Dickman, 2010; Thorn *et al.*, 2012).

Contrary to the findings of other studies in which younger farmers had a greater tolerance of predators (Lindsey *et al.*, 2005c; Zimmerman *et al.*, 2005), this study found that tolerance scores on livestock farms increased with age. Although not significant in the final regression model, English speakers had greater tolerance scores and were generally older than Afrikaans or indigenous African language speakers. Therefore, it is believed the inclusion of age in the final model is actually related to the individual's cultural group. Negative attitudes can be deeply rooted within a culture and can influence tolerance and behaviour to predators (Naughton-Treves *et al.*, 2003; Lindsey *et al.*, 2005c; Zimmerman *et al.*, 2005; Goldman *et al.*, 2010). However, education programmes and increased knowledge about wildlife can (although do not always [Baruch-Mordo *et al*., 2011; Glikman *et al*., 2011]) alter these negative attitudes and increase tolerance of predators (Ericsson & Heberlein, 2003; Marker *et al.*, 2003d; Naughton-Treves *et al.*, 2003; Draheim *et al.*, 2011; Romañach *et al.*, 2011; Strande-Straube, 2013) and education at tertiary level was a significant predictor of tolerance scores in this study.

The availability of natural prey or buffer species on the farm significantly predicted tolerance scores on both livestock farms and game ranches. The buffer species or natural prey can provide inexpensive prey items for predators to eat. This diversionary feeding can potentially reduce carnivore depredation on more expensive game-stock or livestock (Power, 2002). Maintaining a diverse prey base is believed by many farmers to be essential to reduce losses to predators (Marker-Kraus *et al.*, 1996), and the number and associated costs of livestock losses are often greater in areas depleted of natural prey (Henschel, 1986; Hemson, 2003; Woodroffe *et al.*, 2005a). Therefore, measures to promote natural prey across both communal and commercial land could have overall benefits for wildlife and people.

4.5.4 Tolerance scores to specific predator species

African wild dogs and lions were reported as the least tolerated of all predator species both in the current study and on private land in South Africa (Lindsey *et al.*, 2005c). Farms where these species were present reported 6.4 times greater game-stock (lions present) and 2.3 times greater cattle losses to predators (lions, African wild dogs or spotted hyaenas present), than farms on which they were absent. Therefore, the lower tolerance is probably linked to the actual or perceived threat these species pose to livestock and game-stock.

Leopards and spotted hyaenas were more tolerated by farmers for causing game-stock losses relative to the other predator species, than for causing cattle losses. In Botswana, spotted hyaenas can be trophy-hunted under licence and leopards could be trophy-hunted on game ranches at commencement of the study (a moratorium was placed on leopard hunting in January 2013 [BWPA, 2012b]). Therefore, the greater reported tolerance of these species is potentially due to their economic benefits, which may offset the costs of carnivore depredation on game-stock (Lindsey *et al.*, 2005c). Relative tolerance scores to each predator species are also likely to be influenced by the cultural perceptions and ecology of the species (Kellert *et al.*, 1996; Roque De Pinho *et al.*, 2014). For example, African wild dogs are potentially less tolerated due to the perception that they chase wildlife making it skittish and are ‘cruel’ because of their method of killing (Lindsey *et al.*, 2005c; Romañach & Lindsey, 2008). More visible predators such as cheetahs, which are diurnal and wide-ranging (Marker *et al.*, 2007; Houser *et al.*, 2009a) are often considered more of a problem than less visible predators (Marker *et al.*, 2003e; Rust & Marker, 2014), such as leopards, which are nocturnal. Cheetahs do not generally scavenge or return to a kill (Phillips, 1993); as a result, they are often negatively perceived by farmers as being frequent killers and as being ‘wasteful’ (Kent, 2011); compared to leopards, which do scavenge and are perceived as killing less frequently and as a consumer of smaller prey items than cheetahs (pers. obs. and P. Lindsey, pers. comm.). Cheetahs were reported to cause 23.6% of game-stock losses, compared to leopards which caused 32.7% of losses. However, possibly due to the lack of cheetah trophy hunting in association with these perceptions of the species ecology, it is cheetahs, not leopards, which are perceived as the predator species which cause the ‘greatest economic losses’ on game ranches in Botswana (BWPA, 2005).

4.5.5 Conclusion: The impact of the Botswana game ranching industry on biodiversity conservation

Overall, game ranches contributed ca. 12,000 km^2 of land for wildlife use and stocked ca. 142,300 wild ungulates, accounting for ca. 15.2% of Botswana’s large herbivore wildlife biomass (DWNP, 2013). Factors which have been highlighted as potential threats to long-term biodiversity conservation on game ranches in South Africa and Namibia (Bothma, 2005; Cousins *et al.*, 2010), such as stocking species that have the potential to hybridise (e.g., blue- and black wildebeest were stocked together on 7.5% of farms), stocking exotic species (occurred on 20% of farms) and breeding rare colour variants (e.g. 'golden' wildebeest; occurred on 2.3% of farms) were relatively uncommon practises, and in the case of exotic species will be prohibited with the implementation of the draft game ranching regulations (DWNP, 2011b). Additionally, game rancher's reported higher tolerance of predators than livestock farmers, and the majority of farms were open to predators (i.e. only 10% of farms were electrified). Therefore, although Botswana's game ranching industry is younger (by ca. 25 years) and substantially smaller than the industries in South Africa and Namibia, it has the potential to play a valuable role in the region's conservation of wildlife populations, including predators.

However, HPC did still occur, especially on game ranches farming game-stock in association with cattle. These ranches generally relied on low income, high off-take game ranching activities (e.g. hunting for meat). Support of the game ranching industry to promote its expansion and reduce its reliance on these activities will improve its stability, financial viability and maximise its role in biodiversity conservation. Additionally, efforts should be made to develop, promote and implement conflict mitigation techniques for both livestock farming and game ranching, which aim to specifically address the drivers of HPC. In the current study, these drivers were predominantly farmers' attitudes and perceptions of predators, as opposed to the direct costs of carnivore depredation. Therefore, mitigation methods need to aim to alter these attitudes and change the perception of predators from being a cost to being an asset to the farm.

Potentially the biggest threat to the game ranching industry's role in biodiversity conservation is its possible instability in the long-term. Many game ranchers feared that the implementation of the draft game ranching regulations in Botswana will impose large financial costs on hunting operations. In conjunction with the termination of hunting on state land (from January 2014) (MEWT, 2013) and the moratorium on leopard hunting (from January 2013 [BWPA, 2012b]) many farmers stated that they were unsure of the future of the game ranching industry. A lack of support outside of the wildlife ministry and a top-down approach from government, in conjunction with a lack of consultation, inconsistent regulations and leadership, has been highlighted as detrimental to the industry across southern Africa (Barnes & Jones, 2009; Brink *et al.*, 2011; Lindsey *et al.*, 2013a). These governance issues have caused declines in the number of game ranches in operation in Zambia and Zimbabwe (Lindsey *et al.*, 2009a; Lindsey *et al.*, 2013a). Therefore, to expand or maintain its conservation impact, improvements in communication and support between game ranchers and regulatory bodies is likely to be necessary. Overall, the game ranching industry has the potential to conserve free-ranging predator populations, but development and growth of the industry will be necessary to maximise these benefits.

Chapter Five: Distribution and density of large predators on commercial farmland in Botswana

5.1 Chapter overview

In the previous chapter, the predator species reported to cause the majority of smallstock losses was black-backed jackals, and nationwide the majority of cattle and game-stock losses were reported to have been caused by leopards. Farms where lions, or lions, spotted hyaenas and/or African wild dogs were present reported losing 6.4 times more game-stock and 2.3 times more cattle to predators than farms where these species were absent. In light of the impact different predator species have on the direct costs of HPC, this chapter aims to map the distribution of predators on game ranches in Botswana using questionnaire data and to determine the density of large predators in the Ghanzi farming block using three spoor surveys and one camera-trap survey.

Predator distributions were consistent with previous distribution data (Winterbach, 2008; CCB, n.d.), but density estimates varied from those reported in the Botswana National Predator Strategy. Brown hyaenas and leopards occurred at higher and lower densities in the Ghanzi farming block, respectively, than previously assumed (Winterbach, 2008). Cheetahs, leopards and brown hyaenas were more likely to be present on farms with abundant game-stock, and lions, spotted hyaenas and African wild dogs were more likely to occur close to source populations in protected areas. Cheetahs were detected more frequently on game ranches than on adjacent livestock farms, emphasising the potential role game ranches could play in cheetah conservation and the importance of alleviating HPC to maximise this conservation benefit.

5.2 Introduction

As demonstrated in Chapter 4 the presence of different carnivore species can affect the frequency of HPC incidents (Winterbach *et al.*, 2012). Therefore, knowledge of predator distributions and densities is necessary to identify potential ‘hot-spots’ of HPC and to determine target areas for conservation and human–predator conflict mitigation (McDonald & Yalden, 2004). However, information to direct the Botswana National Predator Strategy is generally only available from protected areas, causing assumptions to be made as to predator densities outside of Botswana’s national parks and reserves. These assumptions, based on expert opinion and the extrapolation of available data (Winterbach, 2008), could be detrimental to predator conservation and HPC management if they are incorrect.

Increases in predator populations and consequently increased livestock losses to carnivore depredation are sometimes blamed on the introduction of game ranching (Brink *et al.*, 2011). Predator density is positively correlated with prey biomass (Hayward *et al.*, 2007b), consequently, the recovery of wildlife populations could, or is often perceived to increase predator densities. This perception has resulted in conflict between game ranchers and neighbouring livestock farmers in South Africa (Brink *et al.*, 2011) and in Botswana, and can create peer pressure from neighbouring farmers for game ranchers to remove predators (pers. obs.). To investigate this common perception that predators are more abundant on game ranches than on livestock farms, this study used camera-trap and spoor surveys to determine predator densities in the Ghanzi farming block and questionnaire data to investigate the distribution of predators across Botswana's commercial game ranches.

5.3 Study area and methodology

5.3.1 Study area

Questionnaire surveys as described in Chapter 4 were conducted with game ranchers nationwide and with livestock farmers in the principal game ranching regions, namely the Ghanzi farming block and the Tuli-block, Makalamabedi and Hainaveld farms (Fig. 2.13). The camera-trap and spoor surveys were conducted on the Ghanzi farming block in north−west Botswana. The first spoor survey was conducted between March 2007 and June 2007 on an area of livestock and game-stock farms east of Ghanzi town ('Mixed farmland survey') (Fig. 5.1). Two subsequent spoor surveys were conducted concurrently between February 2008 and July 2008 on farms ranching only game-stock (Game farmland survey) and on farms ranching only livestock (Livestock farmland survey) west of Ghanzi town (Fig. 5.1). The camera-trap survey was conducted in an area containing both livestock farms and game-stock only ranches between October 2009 and January 2010 (Fig. 5.2). A description of the study area's ecology and human population is given in Chapter 2.

5.3.2 Questionnaires

As part of the questionnaires described in Chapter 4, data were collected from 43 ranches farming only game-stock, 37 farms ranching livestock and game-stock (L/G farms) and 27 livestock farms. Data were also collected from six additional private wildlife reserves in the Tuli-block. Wildlife reserves (WR) were unfenced properties which allowed the free movement of natural game populations. In total, questionnaires were completed on 113 farms. As some participants represented multiple registered game ranches, at least one representative from 86.2% of the game ranches in existence was questioned (n = 109). A breakdown of the participant and farm characteristics is given in Chapter 4.

Farmers were asked how often they saw mammalian predator species on their farm, namely African wild dogs, brown hyaenas, caracals, cheetahs, black-backed jackals, leopards, lions and spotted hyaenas. A sighting of a predator included seeing spoor or a physical sighting of the species. A predator species was considered present if farmers recorded seeing the predator at least quarterly (every three months) or absent if the species was seen less than quarterly. On distribution maps, the absent category was further divided into transient if predators were seen less than quarterly but within every few years and absent if they were never seen. Participants were also asked if they agreed with the following statement ‘game ranches have more predators than neighbouring livestock farms’, based on a four-point Likert scale of ‘strongly agree’, ‘agree’, ‘disagree’ and ‘strongly disagree’, a ‘do not know’ option was also available. Lastly, farmers were asked if they had experienced any conflict with their neighbours related to their choice of land use and if so to explain why.

Farm locations were mapped in ArcView 3.2 (Environmental Systems Research Institute Inc. 1992–2000) using GPS locations collected at the time of interview, in addition to GIS data from the BWPA and DWNP (BWPA, 2012a; Department of Wildlife and National Parks, n.d.). The relationship between predator presence and ecological characteristics of the area or characteristics of the farm was investigated using questionnaire data (Chapter 4) and by reference to national GIS data (CIESIN, 2012; Department of Wildlife and National Parks, n.d.).The closest straight line distance from the boundary of each farm to the nearest protected area was calculated in kilometres.

5.3.3 Spoor survey

The spoor surveys were conducted on a fixed route along sand roads, known as a 'spoor transect'. Two transects were identified in each survey area and were designed to be nearly linear, to reduce the chances of double sampling (Fig. 5.1). The four transects within the 2008 surveys (total length = 78.4 - 83.0 km) and the two transects in the 2007 survey (total length = 70.4 km) were sampled an equal number of times, alternating between transects each day.

A San bushman tracker accompanied by a predator biologist located and identified the spoor from a vehicle driven at slow speeds (ca. 10 – 13 km/h). San bushman trackers have a demonstrated ability to reliably and accurately identify and age animal spoor (Stander *et al.*, 1997a), and trackers with demonstrated skills in spoor identification were employed for this study. Spoor tracking began at sunrise and was completed before the sun angle made the visualisation of spoor difficult. Any large predator spoor (defined as African wild dogs, brown hyaenas, cheetahs, leopards, lions or spotted hyaenas) was identified and recorded with the date, GPS location, number of animals and land use the spoor was first located on (cattle farm or game ranch). Only fresh spoor (< 24 hours old) were used in the analysis. Road surfaces obscured by vehicle traffic or rain in the 24 hours prior to the survey were not included. When multiple footprints from the same species were found on the same transect, judgments were made as to whether the spoor belonged to the same or a new individual, with the intention of only recording each individual once per day in accordance with Stander (1998). Spoor were recorded as individual spoor, not as a family group (e.g. spoor of five animals found together were counted as five individual spoor). Preparing the road surface before sampling has not been found to be beneficial (Smallwood & Fitzhugh, 1995) and, therefore, was not performed in this survey.

Road penetration, spoor frequency and density were defined and calculated using the same methods as Stander (1998). Spoor frequency was defined as the mean number of kilometres per individual spoor, and sufficient sampling effort to accurately determine spoor frequency was

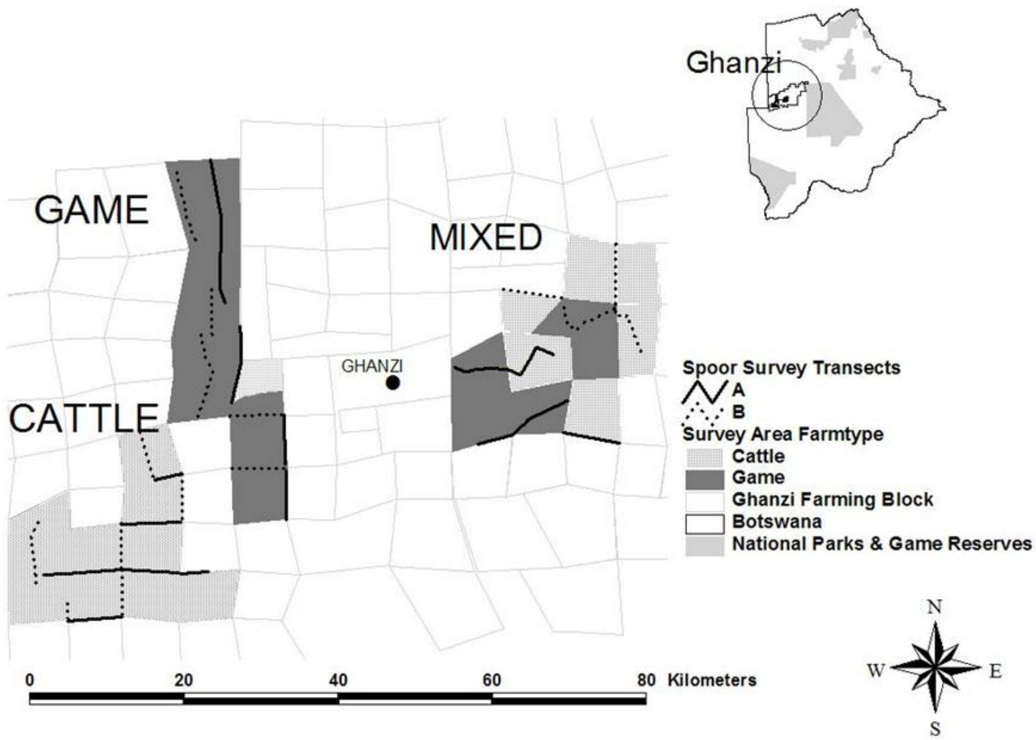


Figure 5.1. Location of three large predator spoor surveys on commercial farmland in the Ghanzi farming block, Botswana. The surveys were conducted in an area of livestock farms (Cattle), an area ranching game-stock only (Game) and a mixed area of livestock farms and game ranches (Mixed)

defined as the point when spoor frequency reached an asymptote (Stander, 1998; Houser *et al.*, 2009b). Spoor density was defined as the number of individual predator spoor per 100 km and is derived from the spoor frequency (Stander, 1998). Road penetration was defined as the sum of the distance surveyed expressed as a ratio of the sample area, e.g. 1 km surveyed : χ km^2 survey area (Stander, 1998; Houser *et al.*, 2009b). The spoor density estimate is a relative estimate of species density. To improve the accuracy of the estimate a calibration equation calculated by double sampling predator populations with a direct (e.g. capture-mark and recapture) and an indirect (e.g. spoor surveys) sampling technique can be applied. In this study spoor density was calibrated using the Funston *et al.* (2010) calibration equation, believed to be suitable for all large predators on sandy soils. However, because this equation generates negative predator density estimates below spoor densities of 0.32/ 100 km (Funston *et al.*, 2010), the Stander (1998) equation was used to calibrate the detected spoor density if it was below this threshold, however, the equation was only applicable for leopards.

5.3.4 Camera-trap survey

Survey design

The camera-trap survey was conducted between October 2009 and January 2010; a period of 84 days was chosen to maintain the assumption of demographic closure (Karanth & Nichols, 1998) (Table 3.1). Pairs of Cuddeback Capture cameras (Non Typical Inc., Green Bay, WI, USA) were placed on opposite sides of roads to photograph both flanks of the animal for identification and to provide redundancy in case of camera failure (Silver *et al.*, 2004). Cameras were offset by 1 – 2 m to prevent interference from the opposing camera flash. No bait or lures were used, and obstructions such as grass or branches were removed from the site. Cameras were mounted on

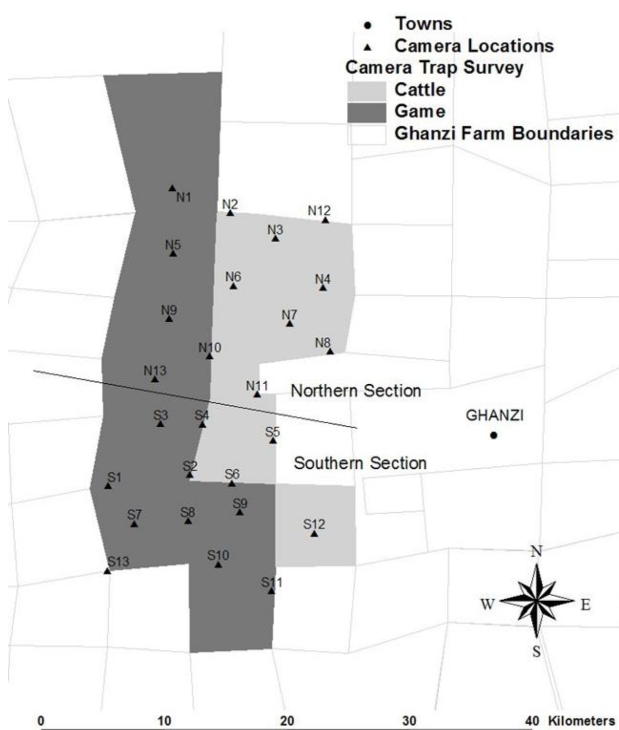


Figure 5.2. Location of cameras in a large predator camera-trap survey in the Ghanzi farming block, Botswana

wooden poles, fence posts or trees ca. 50 cm above ground level at 26 sites (camera-stations). Branches from *Acacia* species were placed around the camera to prevent animal interference.

Cameras should be positioned to maximise the probability of capture of a target species (Carbone *et al.*, 2001). As cheetahs are considered to be the predator species which cause the biggest economic losses to game ranchers in Botswana (BWPA, 2005) and are the most threatened felid within southern Africa (IUCN, 2013), the camera-trap survey was primarily designed for cheetahs. Cheetah signs (photographs, sightings of cheetahs or cheetah spoor) had been previously detected at 54% of the chosen camera-stations. All cameras were placed along sand roads, with the exception of two sites (a cheetah marking-tree and a water trough). Because all large predators use roadways, the camera-trap survey was considered applicable to estimate the density of all large predator species. The mean distance between camera-stations was 4.3 km (SD = 4.1; range 2.6 - 5.8 km). This distance was 2.0 times smaller than the radius of a female cheetah's home range in nearby habitat (241 km^2; radius = 8.7 km; [Houser *et al.*, 2009a]), 1.8 times smaller than a female leopard's home range radius (179 km^2; radius = 7.6 km; [Marker & Dickman, 2005]) and 2.7 times smaller than a female brown hyaena's home range radius (419 km^2; radius = 11.5 km [Maude, 2010]). Therefore, the requirement of capture-recapture analysis, that every individual has an opportunity for capture, was upheld. The size of the sampling area, formed by drawing a polygon through the outer camera-stations, should be equivalent to at least the size of one home range of the wider ranging sex, generally males (Sollmann *et al.*, 2012; Tobler & Powell, 2013). The sampling area size was 475 km^2, equivalent to 1.2 times a male cheetah's home range size within the geographical study area (Houser *et al.*, 2009a), 1.0 times a male brown hyaena's home range (Maude, 2010) and 2.1 times a male leopard's home range (Marker *et al.*, 2007), therefore, meeting the minimum requirements.

A 'blocked' survey design, as developed by Karanth (1995) was utilised. Twenty-six cameras at 13 sites were placed in the northern section of the sampling area for 42 days; the cameras were then moved to an additional 13 sites in the southern section of the sampling area for an additional 42 days (Fig. 5.2). This resulted in a camera density of one camera per 18 km^2. Camera-stations were located on ranches farming only game-stock (n = 14), on livestock farms (n = 10) or on the border between livestock and game-stock only ranches (n = 2), with one camera on either side of the fence. When a predator was captured on a farm border, the side of the fence the predator was located on (livestock farm or game ranch) was recorded.

Cameras were active continuously, with a 30 second delay between photos and were checked every five to 12 days. Photos of all species captured were analysed on Camerabase 1.3 (Tobler, 2007). The number of times a species was photographed (captured) per 100 camera-trap nights was recorded. Captures were included if recorded more than one hour after the previous capture of that species. Large carnivores with individually recognisable coat patterns, namely cheetahs, leopards, African wild dogs, spotted hyaenas and brown hyaenas, were identified manually, by looking for identical spot or stripe patterns on the fore- and hind legs (Fig. 5.3). The fore- and hind legs have the most stable patterns (Chelysheva, 2004; Wiesel, 2006); however, other areas of the body or characteristic scars were used when necessary (Fig. 5.3). To reduce observer bias, identification was conducted on two separate occasions by the same observer (Foster & Harmsen, 2012), and unidentified photos were excluded from the data analysis. Future studies would benefit from using two different observers. The sex and age of the individual was identified when possible, based upon external genitalia, social grouping and published aging criteria (Kruuk, 1972; Caro, 1994; Marker *et al.*, 2003b; Balme *et al.*, 2012). In line with the assumptions of capture-recapture analysis and the IUCN guidelines for defining population size, only mature individuals deemed to be capable of reproduction (referred to as 'adults') with independent movements were included in the data analysis (Foster & Harmsen, 2012; IUCN Standards and Petitions Subcommittee, 2013; Tobler & Powell, 2013). A capture occasion was considered independent for capture-recapture analysis if it was a different individual/family group or if the same individual/family group was captured more than 24 hours apart.

Camera-trap surveys utilise capture-recapture analysis to calculate species abundance. However, to be biologically relevant, abundance estimates must be converted to densities, either by utilising non-spatial buffer-strip methods to estimate the size of the surveyed area, or by using SECR analysis. SECR analysis can be performed using the maximum likelihood model

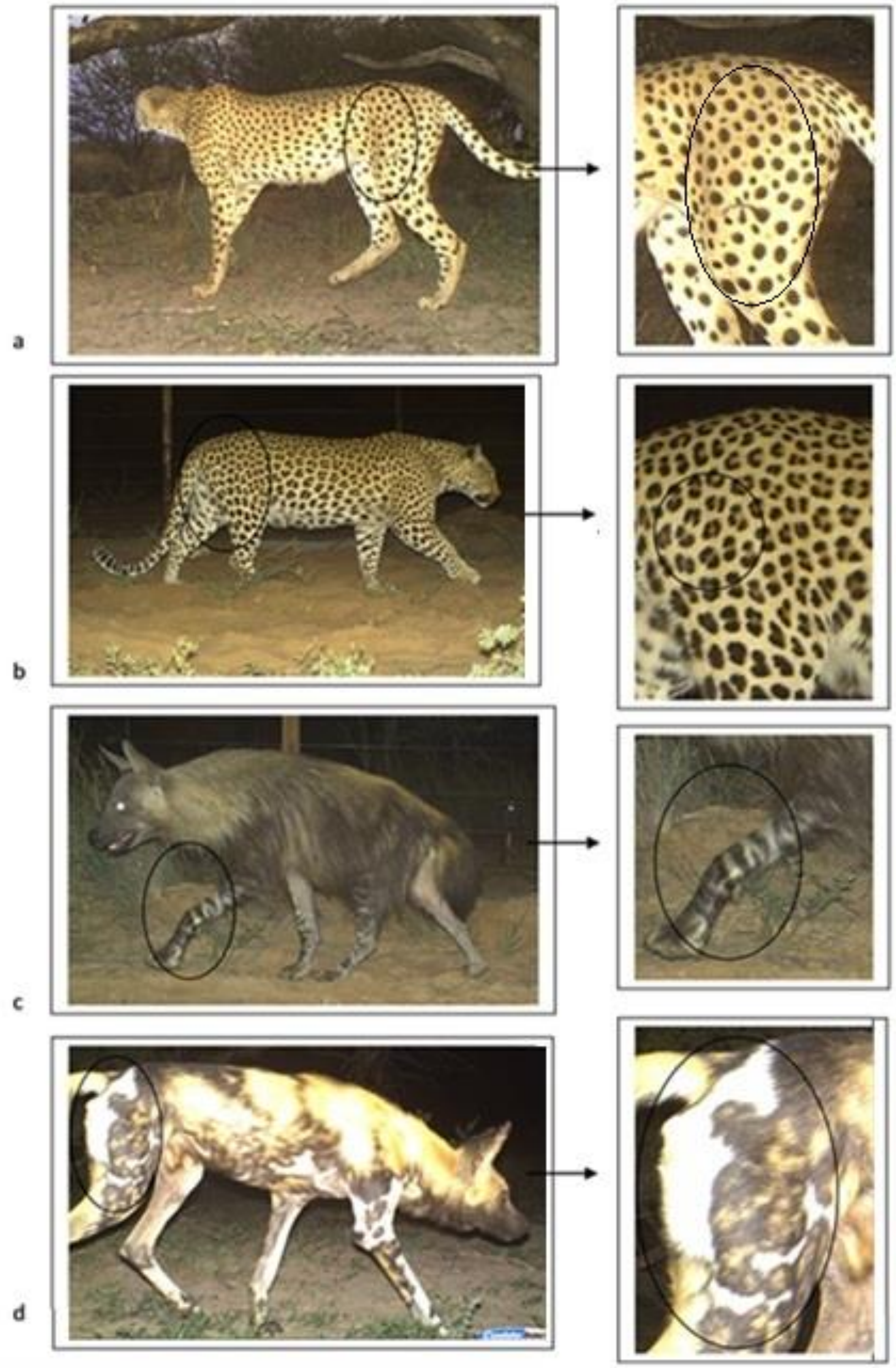


Figure 5.3. Examples of individually recognisable spot and stripe patters used to identify cheetahs, leopards, brown hyaenas and African wild dogs. Distinctive scars as shown in figure “a” were also used in identification. Camera-trap photos are courtesy of Cheetah Conservation Botswana

(Efford, 2004; Efford & Fewster, 2012) or the B-SECR model (Royle & Young, 2008; Royle *et al.*, 2009a; Foster & Harmsen, 2012). This study used the B−SECR model which is considered more appropriate for the small sample sizes commonly obtained with species which naturally occur at low densities (i.e. large carnivores) (Gros *et al.*, 1996; Kery *et al.*, 2011; Gopalaswamy *et al.*, 2012b). Density was also calculated using buffer-strip methods to enable comparison with past studies (Chapter 3 provides further explanation of the B-SECR model and buffer-strip methods).

Buffer-strip method: Abundance

The camera-trap survey was divided into sampling occasions. Each sampling occasion was derived from two or three days of data from both the northern and southern section of the sampling area e.g., Day 1 and 2 from the northern section and Day 1 and 2 from the southern section constituted the first sampling occasion (Fig. 5.2). A matrix of capture histories for each individually recognisable species was created, consisting of 0 (no capture) and 1 (capture) for each individual on each sampling occasion. The Stanley and Burnham (1999) closure test was used to test for population closure (i.e*.,* no immigrations or emigrations) using CLOSETEST (Stanley & Richards, 2005).

The capture-history-matrix was used in capture-recapture analysis with the web-based version of the programme CAPTURE (Rexstad & Burnham, 1991b; a). CAPTURE applies seven models to the data set and recommends the most appropriate model (scored between zero and one) to calculate the abundance ($\hat{N}$) and standard error (SE) (Otis *et al.*, 1978). Confidence intervals were calculated using the equation $\hat{N}$ ± 1.96 * SE (White *et al.*, 1982).

Buffer-strip method: Density

To account for possible animal movements outside of the sampling area (i.e. beyond the outer camera-stations) the size of the surveyed area was calculated by adding a circular buffer to each camera-station and dissolving the boundaries to form one overall area (Silver *et al.*, 2004). The radius of the buffer-strip was equal to the radius of the home range size for each species specific to that geographic area or habitat (Karanth & Nichols, 2000; Soisalo & Cavalcanti, 2006). Alternatively, if home range data was unavailable, the buffer-strip radius was estimated as the

MMDM of all individuals captured at different camera-stations during the camera survey (Soisalo & Cavalcanti, 2006; Dillon & Kelly, 2008; Maffei & Noss, 2008). The half MMDM was not included as it is generally considered to overestimate species density (Noss *et al.*, 2003; Parmenter *et al.*, 2003; Soisalo & Cavalcanti, 2006; Dillon & Kelly, 2008). Home ranges sizes were taken from published data from GPS or VHF radio collared individuals. Home range sizes were calculated using the 95% Minimum Convex Polygon method; home range areas were assumed to be circular in order to estimate the radius of the home range.

Bayesian Spatially Explicit Capture-Recapture model: Density

SPACECAP version 1.0.5 (Singh *et al.*, 2010) in 'R' version 2.12.0 was used to calculate a density estimate using the B-SECR model (Royle *et al.*, 2009a; Royle *et al.*, 2009b; Gopalaswamy *et al.*, 2012a). Three input files consisting of animal capture details, trap deployment details and state-space details (Singh *et al.*, 2010) were derived from the camera-trap data. The state-space data were derived from a grid of points (pixels), spaced 2 km apart within the sampling area plus a buffer zone of 50 km. Therefore, each pixel represented a 4 km^2 area.

The data augmentation (DA) value is the maximum number of potential individuals of each species that could occur in the capture area. The DA value needs to be sufficiently large to avoid biasing the density estimate (Marques *et al.*, 2011; Noss *et al.*, 2012) and was checked by ensuring the distribution of 'NSuper' was not bound above the specified DA value (Marques *et al.*, 2011; Noss *et al.*, 2012). Convergence of the 'Markov chain Monte Carlo' chains was checked by using the Geweke statistic (Geweke, 1992; Noss *et al.*, 2012), in the 'boa' package in the 'R' statistical computer programme (Smith, 2007; 2012). Various combinations of grid spacing, DA values, iterations, burn-in and thinning were tried before obtaining convergence. Density is quoted with the standard deviation and the 95% high posterior density interval (the Bayesian analogue of confidence intervals). Trap response behaviour was included for all species.

One of the assumptions of SECR analysis is that all captures are independent and that all individuals have independent activity centres (Table 3.1). In social species or groupings which hunt together, for example male cheetah coalitions, their movements and, therefore, their activity centres are not independent. To maintain this assumption covariates such as sex and social grouping of individuals captured can be included in the model, if the minimum required sample size of at least 10, but ideally 30 individuals is obtained (Sollmann *et al.*, 2011; Tobler & Powell, 2013). However, in a naturally low-density species, like cheetahs, this sample size is unlikely to be achieved, therefore, a different approach was required. In this study, male coalitions were counted as one group in density estimates to uphold the assumption of independent activity centres. When any of the group's members were photographed it was assumed all members were present and the sighting was counted as a recapture. To account for additional adults in coalitions and determine the total density estimate, the initial group density estimate (D_g) as calculated by capture-recapture analysis was adjusted on the assumption that the average group size in a male cheetah coalition is 2.3 adult males (CCB unpubl. data.) and 26% of adults in the Botswana cheetah population form coalitions (based on the assumption that 49% of adult cheetahs in a population are male [Berry *et al.*, 1996] and 52.6% of adult males are a member of a coalition [CCB, unpubl. data.]). Therefore:

Total cheetah density = (*density of individuals in a male coalition*) + (*density of individuals not in coalitions* [*females and solitary males]*)

$$\textit{Total cheetah density} = (D_g * 0.26 * 2.3) + (D_g * (1 - 0.26))$$

5.3.5 Statistical analyses

All data were coded for use in the statistical computer package SPSS (version 11.0.1; SPSS Inc.). The Kolmogorov-Smirnov or Shapiro Wilk tests were used to determine if continuous variables were normally distributed. T-tests or Mann-Whitney U-tests were used when comparing continuous variables with a categorical variable with two states. Chi-squared tests, using Yate's correction factor for tests with one degree of freedom were used to compare proportions. If the test was significant, the categories which gave rise to that significance were taken as those with the largest standardized residuals, with 1.96 being taken as a guide to the minimum standardized residual that could be treated as significant. Multi-nominal regressions were used to determine significant factors associated with the presence/absence of each predator species. A Spearman's correlation matrix was used to identify and exclude any inter-correlated predictor variables, defined as pairs of variables with a Spearman's correlation of more than 0.7. All variables expected to influence presence were included in the model and removed following a backwards stepwise procedure until all remaining variables were statistically significant as conducted by Lindsey *et al.* (2013b). All tests were two-tailed.

5.4 Results

5.4.1 Predator distribution on game ranches and nearby livestock farms

African wild dogs and lions were the least commonly sighted species, present on 26.5% and 23.0% of farms, respectively (Table 5.1, Fig. 5.4). Black-backed jackals were the most commonly sighted species, present on 98.2% of farms (Table 5.1). There was no significant correlation between the proportion of farms each species was reported to be present upon and farmers' mean reported tolerance score to the species ($R_s = 5.95$, $n = 8$, $p = 0.120$; Fig. 5.5).

Farms on which lions and African wild dogs were present had a median of four other large predator species present; farms with cheetahs or spotted hyaenas had a median of three other large predator species present, and farms with leopards had a median of two other large predator species present. Leopards, black-backed jackals and brown hyaenas were widely distributed across all sites (Table 5.1; Fig. 5.4). Cheetahs were present at the majority of sites (61.9%), excluding the Francistown and Makalamabedi area (Fig. 5.4). Lions were restricted to farms bordering nationally protected areas (median distance to a protected area = 12.5 km, range 0 – 65 km, n = 26), namely the CKGR, the Northern Tuli Game Reserve and Nxai/Makgadikgadi Pans National Park (Fig. 5.4). African wild dogs and spotted hyaenas were reported present on farms in the Tuli-block, Ghanzi farming block and Makalamabedi farming areas (Fig. 5.4).

Predators, with the exception of black-backed jackals which were widely distributed, were statistically more likely to be present in areas of low human population density (cheetahs), close to protected areas (lions, African wild dogs and spotted hyaenas), on game ranches with a larger number of game-stock animals (cheetahs, brown hyaenas and leopards), further than 10 km from a human settlement (lions) and on larger farms (all species except spotted hyaenas and brown hyaenas). Cheetahs were present on 68.8% of farms where spotted hyaenas were uncommon

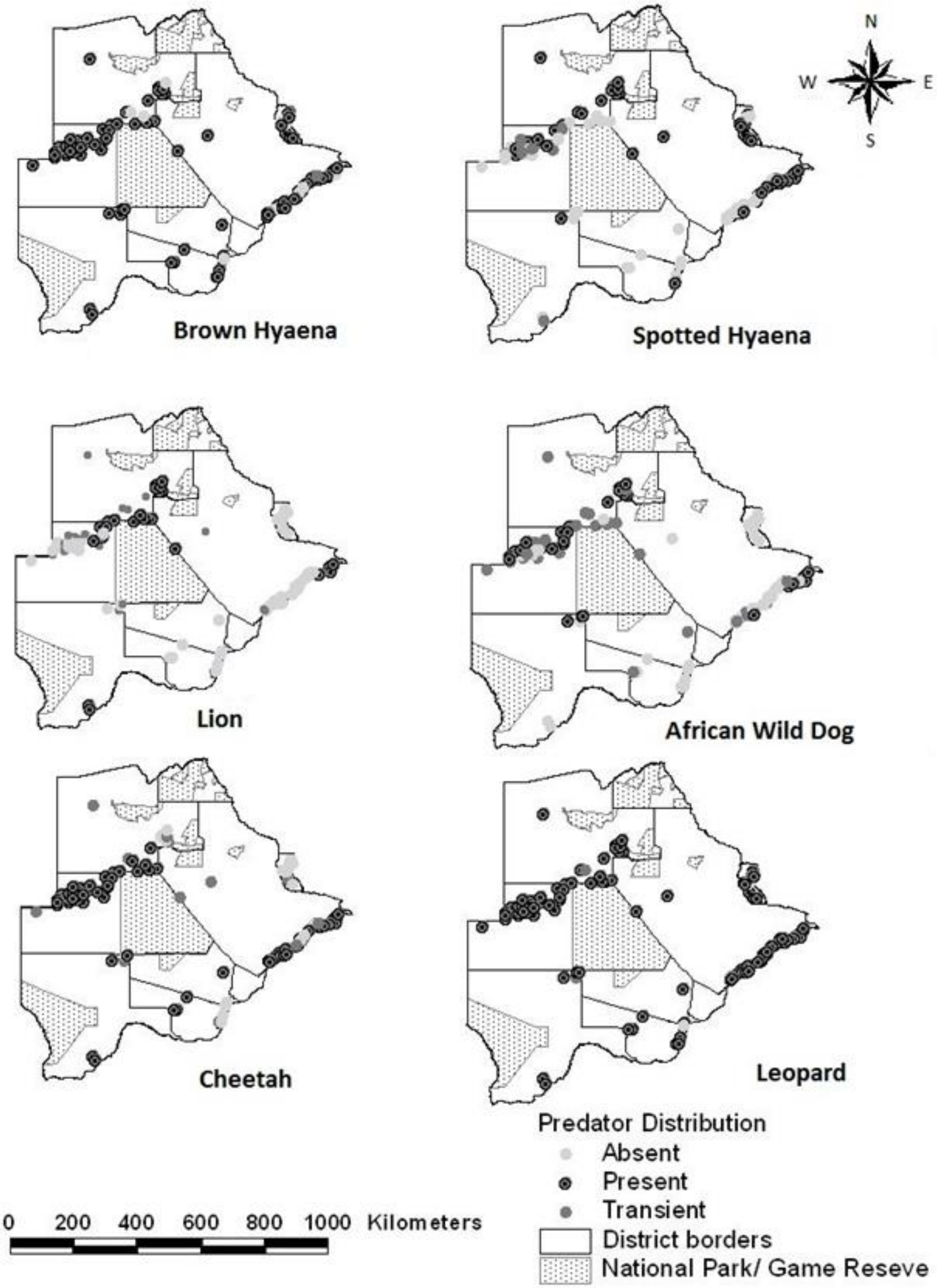


Figure 5.4. Distribution of large predators on commercial game ranches and livestock farms interviewed in Botswana in 2012/2013 (n = 109 - 113)

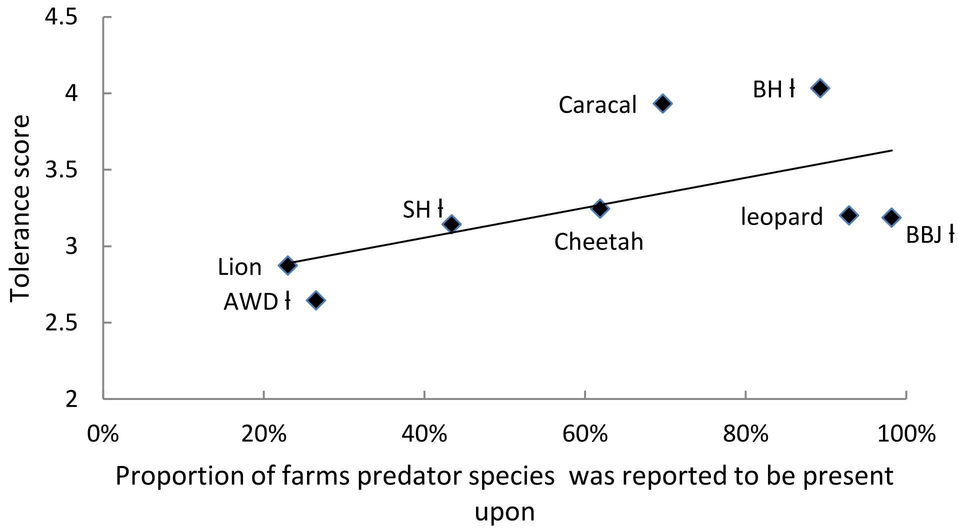


Figure 5.5. Correlation between farmers' reported mean tolerance scores to each predator species (Chapter 4) and the proportion of commercial farms that each predator species was reported to be present upon. Tolerance scores ranged from least to most tolerant (0 – 5). A predator species was considered present if farmers recorded seeing the predator (spoor or physically) at least once every three months

ł AWD = African wild dog, SH = Spotted hyaena, BH = Brown hyaena, BBJ = Black-backed jackal

(seen less than weekly) and 45.5% of farms where spotted hyaenas were common (seen weekly or more; $\chi^2 = 4.44$, df = 1, p = 0.035). Cheetah presence was not related to the presence of lions ($\chi^2 = 1.21$, df = 1, p = 0.270). Logistic regression models using the aforementioned variables reliably predicted the presence/absence of each predator species (Table 5.1). There was no significant relationship between the presence of each predator species and the income the farm earned from game-stock or specifically from photographic tourism.

5.4.2 Predator density: spoor survey

The total distance covered by the three spoor surveys was 3535 km. Spoor from cheetahs, leopards, brown hyaenas and African wild dogs was detected. However, the latter was not located on a specified spoor transect and estimates of density were not possible. Lion and spotted hyaena spoor were not detected. The cheetah and brown hyaena spoor survey results reached an asymptote, however, the leopard results did not, indicating that further sampling was necessary.

Table 5.1. The proportion of commercial farms interviewed (n = 113 ł) upon which each predator species was present and the factors associated with the presence (P) or absence (A) of each species ǂ. Factors significant in univariate tests are marked with a "*"; factors significant in multi-nominal regression models are in blue

	Black-backed jackal		Leopard		Brown hyaena		Caracal		Cheetah		Spotted hyaena		African wild dog		Lion	
% of farms present on	98.2%		92.9% (ł112)		89.3%		69.7% (ł109)		61.9%		43.4%		26.5%		23.0%	
Category	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
Human population density §	97.9% ¶	100%	94.7% ¶	82.4%	92.6% ¶	70.6%	73.9% χ^2 = 3.71, (1) p = 0.054	47.1%	70.8% χ^2 = 18.95, (1) p < 0.001*	11.8%	41.7% χ^2 = 0.75, (1) p = 0.549	52.9%	31.3% ¶	0.0%	27.1% ¶	0.0%
Human settlement λ	94.9% ¶	100%	94.7% ¶	90.9%	92.3% χ^2 = 0.169, (1) p = 0.681	87.8%	78.4% χ^2 = 1.27, (1) p = 0.260	65.6%	71.8% χ^2 = 1.59, (1) p = 0.212	57.6%	35.9% χ^2 = 0.82, (1) p = 0.366	47.0%	30.8% χ^2 = 0.11, (1) p = 0.742	25.8%	43.6% χ^2 = 10.25, (1) p = 0.001*	13.6%
Presence	P	A	P	A	P	A	P	A	P	A	P	A	P	A	P	A
Median distance to protected area (km); U(100-104)	*94* z = −1.11 p = 0.266	*38*	*92* z = −1.21 p = 0.225	*142*	*92* z = −0.28 p = 0.781	*107*	*88* z = −0.36 p = 0.721	*113*	*92* z = −0.66 p = 0.508	*107*	*48* z = −2.74 p = 0.006*	*121*	*26* z = −4.56 p < 0.001*	*121*	*13* z = −6.72 p < 0.001*	*128*
Median number of game-stock U(71-73)	*465* z = −1.04 p = 0.417	*1219*	*1308* z = −2.90 p = 0.004*	*164*	*1271* z = −2.34 p = 0.017*	*421*	*1479* z =−1.94 p = 0.052	*872*	*1412* z = −2.56 p = 0.011*	*687*	*1809* z = −1.27 p = 0.203	*1115*	*923* z = −0.36 p = 0.723	*1266*	*1277* z = −0.73 p = 0.466	*1214*
Median farm size (km^2); U(107-111)	*7075* z = −0.17 p = 0.867	*16,100*	*7999* z = −3.05 p = 0.002*	*2775*	*7500* z = −1.74 p = 0.082	*4047*	*8600* z = −2.59 p = 0.010*	*4757*	*4098* z = −3.76 p < 0.001*	*8600*	*8200* z = −0.91 p = 0.363	*6477*	*12,121* z = −3.86 p < 0.001*	*5000*	*11,500* z = −3.10 p = 0.002*	*5410*
Regression model	-		χ^2 = 11.76, (1) p = 0.001		χ^2 = 11.79, (2) p = 0.003		χ^2 = 7.83, (1) p = 0.005		χ^2 = 29.20, (2) p < 0.001		χ^2 = 18.56, (3) p < 0.001		χ^2 = 25.51, (1) p < 0.001		χ^2 = 70.35, (2) p < 0.001	

ł Sample size is shown in brackets; differs due to non-response

ǂ A species was deemed present if the animal or its spoor were sighted at least once every three months and absent if the species was never seen or was seen less than once every three months

§ 1 = low human population density (≤ 5 people per km^2, n = 92-96); 2 = high population density (> 5 people per km^2, n = 17)

λ 1 = no human settlement within 10 km (n = 37-39); 2 = human settlement within 10 km (n = 64-66)

¶ Violate assumptions of Chi-squared test

Based on spoor surveys, brown hyaenas were the most abundant predator species in the Ghanzi farming block (2.18 brown hyaenas/ 100 km^2), followed by cheetahs (0.21 cheetahs/ 100 km^2), then leopards (0.10 leopards/ 100 km^2; Table 5.2).

5.4.3 Predator density: camera-trap survey

Cameras were active for a total of 1063, out of a possible 1092, trap nights. A total of 37 mammalian wildlife species (including game-stock) were photographed during the survey (Table 5.3). The most abundant predator species based on capture rates was black-backed jackals (36.7 capture per 100 trap nights), captured at all camera-stations. Brown hyaenas were the next most frequently captured (7.0), followed by caracals (1.9), cheetahs (1.6) and leopards (0.9; Table 5.3). The most abundant ungulate species captured was kudu, photographed at 80.8% of camera-stations, on both game ranches (19.94 kudu/ 100 trap nights) and livestock farms (30.20 kudu/ 100 trap nights; $\chi^2 = 11.74$, df = 1, $p < 0.001$). Of the individually recognisable predator species density estimates were only possible for cheetahs, leopards and brown hyaenas; the number of African wild dog and spotted hyaena captures was too small for analysis (Table 5.4).

Five adult cheetahs (including a female with three adolescent cubs and a coalition of two adult males), three adult leopards (one male, one female and one of unknown sex) and 15 brown hyaenas (four males, four females and seven of unknown sex) were photographed during the survey (Table 5.4). All of the cheetah and leopard captures and 75.3% of brown hyaena independent captures were identifiable as individuals. Individual capture frequencies varied from one to eleven captures for cheetahs and brown hyaenas and from one to eight captures for leopards (Table 5.4).

The populations were demographically closed (cheetah: $\chi^2 = 13.8$, df = 10, $p = 0.180$; leopard: $\chi^2 = 2.90$, df = 6, $p = 0.82$; brown hyaena: $\chi^2 = 25.64$, df = 17, $p = 0.081$). Buffer-strip density estimates were based on 21 (two-day) sampling occasions for brown hyaenas and 14 (three-day) sampling occasions for cheetahs and leopards. The final parameters for B−SECR analysis are shown in Table 5.5.

Table 5.2. Results of three spoor surveys for large predators on commercial farms in the Ghanzi farming block, Botswana. Surveys were conducted on farms ranching only game-stock (Game survey), ranching only livestock (Cattle survey) and in an area of game ranches and livestock farms (Mixed survey). Lion, spotted hyaena and African wild dog spoor were not detected

		Mixed survey	**Game survey**	**Cattle survey**	**Surveys combined**
Total Area (Game/Cattle area) (km^2)		494.0 (201.1 / 292.9)	408.1 (382.5 / 25.6)	434.9 (0.0 / 434.9)	1337.0
Transect distance (Total distance travelled) (km)		70.4 (1026.2)	83.0 (1268.8)	78.4 (1240.1)	231.8 (3535.1)
Road penetration ł		7.02	4.92	5.55	-
Number of spoor counted	Cheetah	23	13	2	38
	Leopard	1	6	0	7
	Brown hyaena	81	78	98	257
Spoor frequency (km ± SE) (C. of V. ǂ)	Cheetah	44.62 ± 18.70 (42%)	97.60 ± 23.30 (24%)	620.05 ± 349.70 (56%)	93.03 ± 28.18 (30%)
	Leopard	1026.20 §	211.47 ± 25.00 (12%)	λ	505.01 ± 294.31 (58%)
	Brown hyaena	12.67 ± 1.69 (13%)	16.27 ± 2.60 (16%)	12.65 ± 1.90 (15%)	13.76 ± 1.18 (9%)
Spoor density (per 100 km) (95% CI)	Cheetah	2.24 (1.20 - 17.49)	1.02 (0.67 - 2.14)	0.16 §	1.07 (0.67 - 2.78)
	Leopard	0.10 §	0.47 (0.36 - 0.68)	λ	0.20 §
	Brown hyaena	7.89 (6.23 - 10.75)	6.15 (4.68 - 8.96)	7.90 (6.11 – 11.17)	7.27 (6.22 - 8.74)
Predator density (per 100 km^2) (95% CI)	Cheetah ¶	0.58 (0.25 - 5.43)	0.20 (0.09 - 0.55)	łł	0.21 (0.08 - 0.76)
	Leopard ǂǂ	0.05 §	0.25 (0.19 - 0.36)	λ	0.10 §
	Brown hyaena ¶	2.38 (1.85 - 3.29)	1.82 (1.36 - 2.72)	2.38 (1.81 – 3.42)	2.18 (1.85 - 2.65)

ł Sum of the distance surveyed expressed as a ratio of the sample area (1 km surveyed : χ km^2 survey area)

ǂ Coefficient of variation = 100 × standard error/mean

§ Insufficient data to calculate standard error (SE), 95 % confidence intervals (CI) and/or coefficient of variation (C. of V.)

λ ND = spoor not detected

¶ Using spoor density calibration equation from Funston *et al.* (2010)

łł Spoor density estimate was too small to apply a cheetah calibration equation

ǂǂ Using spoor density calibration equation from Stander (1998)

Table 5.3. The capture rate and proportion of camera stations mammalian species were captured at during a camera-trap survey in the Ghanzi farming block, Botswana

Common name	**Scientific name**	**Capture rate (per 100 trap nights)**	**Camera stations n = 26**
Aardvark	*Orycteropus afer*	3.29	42.3%
Aardwolf	*Proteles cristata*	3.76	50.0%
African wild dog	*Lycaon pictus*	0.28	11.5%
African wildcat	*Felis silvestris lybica*	1.41	46.2%
Banded mongoose	*Mungos mungo*	0.28	15.4%
Bat-eared fox	*Otocyon megalotis*	1.41	19.2%
Black wildebeest	*Connochaetes gnou*	1.69	7.7%
Black-backed jackal	*Canis mesomelas*	33.68	100.0%
Blue wildebeest	*Connochaetes taurinus*	9.31	46.2%
Brown hyaena	*Hyaena brunnea*	6.96	80.8%
Cape fox	*Vulpes chama*	2.63	38.5%
Cape porcupine	*Hystrix africaeaustralis*	6.96	65.4%
Caracal	*Felis caracal*	1.88	38.5%
Cheetah	*Acinonyx jubatus*	1.60	23.1%
Common duiker	*Sylvicapra grimmia*	20.04	84.6%
Common eland	*Tragelaphus oryx*	6.21	50.0%
Common genet	*Genetta genetta*	1.32	23.1%
Gemsbok	*Oryx gazella*	7.43	50.0%
Giraffe	*Giraffa camelopardalis*	2.35	23.1%
Honey badger	*Mellivora capensis*	0.28	15.4%
Impala	*Aepyceros melampus*	3.39	38.5%
Kudu	*Tragelaphus strepsiceros*	26.25	80.8%
Leopard	*Panthera pardus*	0.85	23.1%
Plains zebra	*Equus burchelli*	1.51	30.8%
Red hartebeest	*Alcelaphus buselaphus caama*	5.36	50.0%
Scrub hare	*Lepus saxatilis*	20.23	50.0%
Slender mongoose	*Herpestes sanguineus*	0.94	11.5%
South African ground squirrel	*Xerus inauris*	0.66	7.7%
Southern African hedgehog	*Atelerix frontalis*	0.09	3.8%
Spotted hyaena	*Crocuta crocuta*	0.09	3.8%
Springbok	*Antidorcas marsupialis*	0.38	7.7%
Springhare	*Pedetes capensis*	0.19	7.7%
Steenbok	*Raphicerus campestris*	7.15	65.4%
Warthog	*Phacochoerus africanus*	20.23	88.5%
Waterbuck	*Kobus ellipsiprymnus*	1.60	11.5%
Zorilla	*Ictonyx striatus*	1.69	34.6%

Table 5.4. Demography and capture frequencies of predators photographed during a camera-trap survey in the Ghanzi farming block, Botswana

Predator ID	Gender	Age	Social Grouping	No. of captures
Cheetah #1	Male	Adult	Single	11
Cheetah #2a	Male	Adult	2 males	3
Cheetah #2b	Male	Adult	2 males	3
Cheetah #3	Female	Adult	Female & 3 adolescents	1
Cheetah #3a	Unknown	Adolescent	Female & 3 adolescents	1
Cheetah #3b	Unknown	Adolescent	Female & 3 adolescents	2
Cheetah #3c	Unknown	Adolescent	Female & 3 adolescents	1
Cheetah #4	Unknown	Adult	Unknown	1
Leopard #1	Male	Adult	Single	5
Leopard #2	Female	Adult	Single	8
Leopard #3	Unknown	Adult	Single	1
Brown Hyaena #1	Male	Adult	Single	1
Brown Hyaena #2	Female	Adult	Single	4
Brown Hyaena #3	Unknown	Adult	Single	5
Brown Hyaena #4	Male	Adult	Single	8
Brown Hyaena #5	Unknown	Adult	Single	4
Brown Hyaena #6	Male	Adult	Single	11
Brown Hyaena #7	Male	Adult	Single	5
Brown Hyaena #8	Unknown	Adult	Single	4
Brown Hyaena #9	Unknown	Adult	Single	1
Brown Hyaena #10	Unknown	Adult	Single	6
Brown Hyaena #11	Female	Adult	Single	2
Brown Hyaena #12	Unknown	Adult	Single	3
Brown Hyaena #13	Female	Adult	Single	2
Brown Hyaena #14	Female	Adult	Single	3
Brown Hyaena #15	Unknown	Adult	Single	1
Spotted Hyaena #1	Unknown	Adult	Unknown	1
African Wild Dog #1	Male	Adult	Pack size ≥ 2	3
African Wild Dog #2	Male	Adult	Pack size ≥ 2	2

Table 5.5. Number of captures, abundance, and estimated density of cheetahs, leopards and brown hyaenas photographed during a camera-trap survey on commercial farmland in the Ghanzi farming block, Botswana

		Cheetah	**Leopard**	**Brown Hyaena**
Number of independent individuals/ family groups captured		4	3	15
Number of independent captures §		17	9	58 (+ 19 λ)
Model chosen by CAPTURE ƚ		M_o (M_h)	M_o (M_h)	M_h
Score for the chosen model in CAPTURE		1.00 (0.83)	1.00 (0.83)	1.00
Number of independent captures after data condensed into sampling occasions for CAPTURE		15	9	49
Abundance ($\hat{N}$) ± SE		5 ± 1.4	4 ± 1.55	18 ± 2.52
95% CI range		4 - 8	4 - 11	15 - 23
Probability of capture ($\hat{p}$)		0.21	0.16	0.13
Buffer-strip method (buffer radius) ǂ		A (11.42 km)	B (9.65 km)	B (14.94 km)
Buffer-strip method density estimate per 100 km^2 (lower and upper estimate)		0.36 (0.29 - 0.58)	0.26 (0.19 - 0.52)	0.74 (0.61 - 1.10)
B-SECR model parameters:	DA	130	130	150
	Iterations	200,000	200,000	50,000
	Burn-in	30,000	40,000	1000
	Thinning	10	1	1
B-SECR density estimate per 100 km^2 (95% HPD interval)		0.32 (0.04 - 0.77)	0.37 (0.06 - 0.75)	0.49 (0.12 - 0.87)

ƚ The null model (M_o) is considered to be unsuitable to analyse species with complex social behaviour and territoriality (Karanth *et al.*, 2004) and was rejected in favour of the heterogeneity model (M_h) with the jack-knife estimator

ǂ A. Radius of geographically specific home range calculated with the 95% Minimum Convex Polygon method (Houser *et al.*, 2009a)

B. Mean maximum distance moved, excluding individuals that were captured repeatedly but at the same camera station i.e. individuals that had moved “zero” distance

§ Number of captures of socially and temporally independent groups/individuals

λ Number of independent captures of unknown individuals

SECR density estimates were 1.1 and 1.5 times lower than buffer-strip density estimates for cheetahs and brown hyaenas, respectively, but 1.4 times greater for leopards (Table 5.5). Based on B-SECR model estimates, brown hyaenas were the most abundant predator species on Ghanzi farmland (0.49 brown hyaenas/ 100 km^2), followed by leopards (0.37 leopards/ 100 km^2), then cheetahs (0.32 cheetahs/ 100 km^2; Table 5.5).

5.4.4 Comparison of density estimates between different land uses

From the questionnaire survey, the majority of farmers agreed (30.5%, n = 118) or strongly agreed (34.7%) that game ranches had more predators than neighbouring livestock farms. Twelve percent (12.2%) of game ranchers reported neighbouring livestock farmers were unsupportive of the game ranch, primarily due to the perceived threat predators or game-stock caused to livestock through predation or disease transmission. Cheetah spoor was 2.2 times more likely to be counted on game ranches than on livestock farms and cheetah camera captures only occurred on game ranches (i.e. no captures on livestock farms) (Table 5.6). From the spoor and camera-trap surveys, distribution across land uses was not significantly different for brown hyaenas, black-backed jackals, caracals, leopards, spotted hyaenas or African wild dogs (Table 5.6).

5.5 Discussion

5.5.1 Limitations

The distribution of predators on farmland was based on farmers' assessments which are potentially biased by misidentification of predator signs (Lindsey *et al.*, 2013b) or due to preconceived attitudes of predator populations (Marker *et al.*, 2007). Farmers often overestimate

Table 5.6. Comparison of the number of cheetah, leopard and brown hyaena spoor counted and the number of predator photos captured on livestock farms and game ranches using the Chi-squared test, in a spoor survey and camera-trap survey in the Ghanzi farming block, Botswana

	Capture rate ł				
	Game ranch	**Livestock farm**	χ^2	**df**	**P value**
Cheetah spoor	1.43	0.64	4.36	1	0.037
Cheetah camera captures	3.20	0.00	13.96	1	< 0.001
Brown hyaena spoor	6.47	7.95	2.48	1	0.115
Brown hyaena camera captures	5.63	7.96	1.96	1	0.162
Leopard spoor	0.36	0.00	3.95	1	0.047 ǂ
Leopard, spotted hyaena, and African wild dog camera captures	0.76	1.63	0.94	1	0.332
Caracal camera captures	2.28	1.02	1.89	1	0.169
Black-backed jackal camera captures	29.53	33.27	1.14	1	0.286

ł Per 100 camera-trap nights for camera data or per 100 km of transects sampled for spoor data

ǂ Number of leopard spoor violated the assumptions of the chi - square test the abundance of species which cause HPC, or deliberately inflate abundance estimates in an attempt to increase trophy-hunting quotas or to stimulate lethal control of predator numbers (Gillingham & Lee, 2003; Gusset *et al.*, 2008a). Conversely, low-conflict species can be underestimated, for example many farmers in the current study were unsure if caracals were present. Ground-truthing of reported predator presence would be necessary to ensure reliability, but would be impractical to obtain over a wide study area. However, the resulting predator distribution maps were similar to previous predator distribution data (Winterbach, 2008; CCB, n.d.), therefore, the overall evaluation of predator presence is thought to be accurate.

The primary difficulty when working with low-density predator species is to obtain a large enough sample size for analysis. The leopard spoor frequency results did not reach an asymptote, indicating that further sampling was necessary. Although cheetah spoor frequency did reach an asymptote, the large coefficient of variance values compared to the brown hyaena data, indicates that the precision of the cheetah density estimate was potentially reduced due to the small sample size. Similarly, capture-mark-recapture analysis used in camera-trap surveys was designed for use with large data sets (Linkie *et al.*, 2008; Foster & Harmsen, 2012). However, the number of leopards and cheetahs detected in the camera survey fell short of the recommendations of more than 20 individuals for buffer-strip methods and 10 , ideally 30 individuals to include covariates in SECR models (White *et al.*, 1982; Jackson *et al.*, 2006; Efford *et al.*, 2009; Sollmann *et al.*, 2011; Tobler & Powell, 2013). Although the B-SECR model is able to deal with small sample sizes more accurately than the buffer-strip methods or maximum likelihood SECR model (Singh *et al.*, 2010), the precision of the density estimate was likely reduced by the small sample size and is reflected by the large 95% high posterior density intervals.

Large carnivores are naturally wide-ranging and occur at low densities, so although numerous camera-trap surveys have estimated species abundance and density based on a small sample size, few published surveys of felids have reached the aforementioned requirements (Foster & Harmsen, 2012). Sample size in spoor surveys could be maximised by increasing the road penetration (proportion of roads sampled in the study area), or by increasing the sampling area size and study duration, whilst maintaining demographic closure. However, spoor surveys are labour intensive and less accurate than camera-trap surveys (Balme *et al.*, 2009a). For individually identifiable species, further investment into camera-trap study design to increase the number of individuals caught would be beneficial for large predator density estimates. Sample size could be maximised by increasing the sampling area size, by altering camera placement, by increasing the distance between cameras, by increasing the length of the study and camera density or by combining multiple data sets (Efford & Fewster, 2012; Sollmann *et al.*, 2012; Tobler & Powell, 2013; Royle & Converse, 2014).

To optimise the probability of capture, cameras were positioned in relation to previous sightings of the target species, cheetahs. If the cameras had been positioned specifically at sites known for leopard or brown hyaena activity, or had used scent lures, which brown hyaenas are known to respond to, higher capture rates for these species could potentially have been obtained (Carbone *et al.*, 2001; Thorn *et al.*, 2009; Gerber *et al.*, 2012). However, cheetahs, leopards and brown hyaenas have similar ranging behaviour and the cameras were placed on roadways which all large predators are known to use (Kutilek *et al.*, 1983), so the camera locations were thought to be appropriate to provide a baseline density estimate for all three species.

5.5.2 Predator distribution

Lions, African wild dogs and spotted hyaenas were generally restricted to farms near source populations in protected areas. Farms where these species were present reported 2.3 times greater cattle and 6.4 times greater game-stock losses than farms where they were absent (Chapter 4). Consequently, farms bordering protected areas could be considered 'hot-spots' of HPC (Hazzah, 2006; Schiess-meier *et al.*, 2007). These species were reported by farmers as the least tolerated of the predator species (Chapter 4), therefore, their distribution was most likely restricted because of HPC (Ray *et al.*, 2005; Lindsey *et al.*, 2013b). Lions, in particular, are largely restricted to protected areas, due to HPC, across their range (Bauer & Van der Merwe, 2004; Ogutu *et al.*, 2005). In contrast, the species with the highest tolerance score, brown hyaenas, was present on 89% of farms. Leopards and black-backed jackals, which were moderately tolerated relative to the other predator species, but behaviourally are highly adaptive (Eaton, 1978; Loveridge & Nel, 2008), were reported as present on 93% and 98% of farms, respectively. These species were also the predator species most commonly reported as present on farms in Namibia (Stein *et al.*, 2010). Farms on which leopards, caracals, African wild dogs and lions were reported present were significantly larger than farms upon which they were absent. In contrast cheetahs were more commonly reported as present on smaller farms, in this study and in Namibia (Lindsey *et al.*, 2013b). Lindsey *et al.* (2013b) hypothesised that signs of predators were more likely to be noticed on these smaller properties.

Brown hyaenas, cheetahs and leopards were more likely to be present on farms with large numbers of game-stock (also found on farms in Namibia [Lindsey *et al.*, 2013b]) and density estimates using camera-trap and spoor surveys found that cheetahs (and potentially leopards; sample size was too small to determine significance) were more commonly sighted on game ranches than neighbouring livestock farms. This supports the perception, shared by 65.2% of farmers', that game ranches have more predators than neighbouring livestock farms. Many of the game ranches were originally overgrazed by cattle; the natural succession of vegetation following the introduction of game-stock results in dense vegetative cover (Hejcmanova *et al.*, 2010) and abundant natural prey in addition to game-stock populations (Thorn *et al.*, 2012). Thorn *et al.* (2012) hypothesised that game ranches may be able to support greater carnivore densities than livestock farms because of this increased abundance of prey.

As a result, the introduction of game ranching in livestock farming areas is often blamed for increased predator populations, and in the current study, resulted in conflict between game ranchers and neighbouring livestock farmers on 12.2% of game ranches. This conflict has also been reported on farms in South Africa (Brink *et al.*, 2011). Data were not available to determine if predator populations had increased since the introduction of game ranching. However, the presence of natural prey, including small antelope species and buffer species (e.g., kudu and impala) was associated with reduced livestock losses (Henschel, 1986; Hemson, 2003; Woodroffe *et al.*, 2005a) and increased tolerance of predators (Chapter 4). Therefore, despite a potential increase in predator density, the presence of game ranches in livestock farming areas could have the potential to reduce livestock losses by providing an alternative, natural source of prey for predators.

5.5.3 Differences in predator density between methodologies

Camera-trap and spoor survey data demonstrated that all large predator species (except lions) were present within the surveyed area. Based on relative camera-trap capture rates, black-backed jackals, followed by brown hyaenas, were the most common predator species.

Cheetah and brown hyaena density estimates were 1.1 and 1.5 times lower with the B-SECR model, respectively, than buffer-strip methods. The latter determines the size of the surveyed area based on assumptions as to the distances that individuals move (from home range data or MMDM). These assumptions often underestimate the size of the surveyed area, resulting in the potential inclusion of transient individuals in the buffer-strip population estimate. In contrast, SECR models are based upon observed animal movements in a geographically open system (Efford, 2004; Borchers & Efford, 2008; Royle *et al.*, 2009a). Therefore, in conjunction to the advantages of the SECR model discussed in Chapter 3, SECR models are considered more accurate than buffer-strip methods and are thought to be less likely to overestimate species density (Obbard *et al.*, 2010; Sollmann *et al.*, 2011; Athreya *et al.*, 2013). However, buffer-strip estimates of leopard density were 1.4 times lower than the B-SECR model estimate; this inconsistent pattern could be due to a loss of precision due to the small number of leopards detected (i.e. three adults).

The camera-trap survey using the B-SECR model for data analysis estimated cheetah and leopard density to be 1.5 and 3.7 times greater than the spoor survey. This is consistent with Balme *et al.* (2009a), who showed that leopard densities were underestimated to a greater extent by spoor surveys than by camera-trap surveys, compared to a known population of leopards. By comparison, brown hyaena density was estimated to be 4.4 times lower by the camera survey compared to the spoor survey. Brown hyaenas exhibit wider ranging daily movements than the other carnivore species (Maude, 2010). These wide movements could increase the detectability of spoor and potentially increase the risk of double counting (Kent & Hill, 2013). Additionally, the Funston *et al.* (2010) calibration equation was designed based on lion movements, and although extrapolated to brown hyaenas, it may not adequately account for the greater spoor detectability and, therefore, could cause the calibration equation to overestimate brown hyaena density. Although it was not possible to calculate the accuracy of the B-SECR model, because the true-density of predators in the study area was unknown, in light of the methodological advantages of the B-SECR model (Foster & Harmsen, 2012; Tobler & Powell, 2013) and the accuracy of camera-trap surveys compared to spoor surveys (Balme *et al.*, 2009a), the B-SECR model is believed to be the best approximation of density available.

5.5.4 Predator density estimates compared to previous assessments

The cheetah density detected with the B-SECR model of 0.32 cheetahs/ 100 km^2 was within the previously assumed range for the Ghanzi farmlands (0.15 – 0.56 cheetahs/ 100 km^2), as documented in the National Predator Strategy (Winterbach, 2008). This calculated density was 1.2 - 2.1 times higher than estimated in the adjacent protected area, the CKGR (Senior Wildlife Biologist, 2000), thereby highlighting commercial farmland as an important area for cheetah conservation (Klein, 2007).

Brown hyaena densities (B-SECR: 0.49 brown hyaenas/ 100 km^2) were up to 49 times greater than the previously assumed density of 0.01 − 0.1 brown hyaenas/ 100 km^2 (Winterbach, 2008), but were 4.7 times lower than estimates from camera-trap surveys also conducted on Ghanzi farmland by Kent and Hill (2013) (2.3 brown hyaenas/ 100 km^2). In the current study, it was not possible to identify individual brown hyaenas in 24.7% of independent captures due to poor picture quality. If these unidentified photographs were 'new' individuals, this could have resulted in an underestimate of species density. Alternatively, the camera-trap estimate by Kent and Hill (2013) was conducted in a sampling area approximately 2.5 times smaller than is recommended to avoid positively biasing SECR density estimates (based on species home range size [Maude, 2005; Sollmann *et al.,* 2012]). Therefore, it is also possible that the Kent and Hill (2013) camera-trap survey overestimated brown hyaena density. Either way, brown hyaenas occur at much greater densities on Botswana farmland than previously thought, and this study agrees with the conclusions of Kent and Hill (2013) that commercial farmland is an important area for brown hyaena conservation, with potentially comparable densities to protected areas. For example camera surveys in Pilanesberg National Park, South Africa detected 2.8 brown hyaenas per

100km2 Thorn, 2009 #98}) and spoor surveys in the Kgalagadi Transfrontier Park, South Africa detected 1.82 brown hyaenas / 100 km^2 (Funston *et al.*, 2001) compared to 2.18 brown hyaenas / 100km^2 in spoor surveys from this study.

In contrast, detected leopard densities (B-SECR: 0.37 leopards/ 100 km^2) were up to 4.1 times lower than the assumed estimate in the Botswana National Predator Strategy (0.95 – 1.5 leopards/ 100 km^2 (Winterbach, 2008) and lower than stated in a review of leopard densities outside of protected areas (2.1 leopards/ 100km2; [Marker & Dickman, 2005]). This low density contrasted with local landowners' perceptions that leopards are abundant (Kent, 2011). The majority of farmers interviewed by Kent (2011) in the Ghanzi farming block perceived cheetah (75%), black-backed jackal (60%) and leopard populations (35%) to be increasing (n = 20) and 22.9% of farmers in this study reported that predator numbers are unnaturally large and require control (n = 48; Chapter 7). Farmers' perceptions of the abundance of wildlife are closely linked to their perceptions of the impact wildlife will cause to their livelihoods (Dandy *et al.*, 2012). Consequently, if farmers' perceive predators as overabundant they are less likely to be in favour of their conservation. Leopards are ranked as the most problematic predator by farmers and cause over two times more government 'problem' animal reports in the Ghanzi district than any other predator species (Selebatso *et al.*, 2008). This intense level of conflict in conjunction with a failure to account for the wide-ranging movements of carnivores (Marker *et al.*, 2007), could inflate local farmers' perceptions of leopard abundance.

Two leopards were reported to have been killed in the cattle spoor study area within the three months prior to starting the survey, and one leopard was shot in the 'mixed farmland' spoor survey during the study. This high level of HPC implies additional removals could have occurred in adjacent areas during the spoor and camera-trap surveys, violating the assumption of demographic closure. Additionally, leopards have been shown to avoid roadways in areas of persecution or hunting, and this potential avoidance behaviour could have reduced capture rates (Ngoprasert *et al.*, 2007). Despite these potential violations to the survey assumptions, the precautionary principle should be applied and it should be conservatively assumed that leopard populations are lower than previously thought, and the management of 'problem' leopards and the distribution of leopard trophy-hunting quotas should be adjusted accordingly.

5.5.5 Conclusion

This study suggests that cheetahs, and potentially leopards, are more likely to be present on game ranches than on adjacent livestock farms, emphasising the important role the game ranching industry can play in cheetah and leopard conservation. However, the impact of HPC on predator populations was demonstrated by the recorded predator removals during the spoor surveys. In addition, the distribution of the least tolerated predator species (namely lions, spotted hyaenas and African wild dogs) was largely restricted to farms near source populations in protected areas. Therefore, a continued investment into the promotion and development of HPC mitigation techniques including livestock management techniques which decrease carnivore depredation, financial benefits from predators and community education is likely to be necessary to enable coexistence with predators.

Chapter Six: Prey preferences of free-ranging cheetahs on Botswana farmland: scat-analysis versus farmers' perceptions

6.1 Chapter overview

In Chapter 4 farmers reported that cheetahs caused 15.8% of the cattle and 23.6% of the game-stock losses reported to have been caused by carnivore depredation. In Botswana cheetahs are perceived as the biggest economic threat to game ranches (BWPA, 2005). However, many game ranchers were unsure of the number of game-stock animals killed by predators and estimates of losses were often based on farmers' perceptions of predator abundance and on their perceptions of the prey species preferred by predators (Chapter 4). This chapter aims to examine the prey preferences of cheetahs in the Ghanzi farming block and the relative contribution of livestock and game-stock to the cheetahs' diet.

Cheetahs' prey preferences and the percentage occurrence of prey species in the diet was determined through the cross-sectional analysis of prey remains, primarily hair, found in cheetah scat. Cheetahs were found to predominantly prey on free-ranging abundant game species, primarily kudu, steenbok, duiker, springhares *Pedetes capensis* and scrub hares *Lepus saxatilis*. Based on scat-analysis results, in the Ghanzi farming block cheetahs were estimated to kill 0.1 − 0.2% of a farm's total cattle stock, 0.2 – 0.8% of a farm's total smallstock and 0.3 – 1.1% of a ranch's total game-stock per year. Game ranchers overestimated the prominence of game-stock to the cheetahs' diet, and farmers' perceptions of preferred prey species differed from the results of the scat-analysis. Potential reasons for these discrepancies are discussed, and the importance of natural prey as a potential coexistence strategy, are discussed in Chapters 4 and 7.

6.2 Introduction

Human-predator conflict is often associated with the perception of the threat predators pose, rather than actual stock losses (Marker *et al.*, 2003d; Zimmerman *et al.*, 2005; Dickman, 2008). The wide-ranging behaviour of large predators can cause farmer's to overestimate the abundance of predators and the damage they cause to livestock and game-stock (McIvor & Conover, 1994). Botswana farms are typically large, with large paddocks, therefore, the carcasses of livestock or game-stock are often not located, or are located too late to identify the cause of death. This is a particular problem on game ranches; farmers estimated they located 50% of game-stock carcasses and found it difficult to monitor game-stock losses (Chapter 4). Additionally, when young game-stock or livestock die, the entire carcass including bones is frequently consumed by the predator (if applicable) and/or by scavengers (Phillips, 1993), preventing identification of the cause of death and potentially resulting in an underestimation of the numbers killed (Eaton, 1970a). Alternatively, signs that a predator has scavenged from a carcass can lead to the potentially erroneous assumption that the carnivore species caused the death of the stock (Ray *et al.*, 2005). Consequently predators are often blamed for unexplained stock losses (Rasmussen, 1999; Gusset *et al.*, 2009), and farmers' perceptions of stock losses can exacerbate HPC and lead to the indiscriminate removal of predators (Rust & Marker, 2014).

Farmers estimated cheetahs had caused 15.8% of the cattle and the 23.6% of game-stock losses that had been attributed to carnivore depredation in the 12 months prior to the survey (Chapter 4), and cheetahs are thought to be the biggest cause of economic loss on game ranches in Botswana (BWPA, 2005). Game ranchers often blame cheetahs for the decline of springbok populations (pers. obs.), and farmers' tolerance of predators was significantly lower on game ranches that stocked springbok (Chapter 4). However, it is unclear if this perception of conflict corresponds with actual losses. Due to their wide-ranging movements and diurnal nature (Marker *et al.*, 2007; Houser *et al.*, 2009a), cheetahs are commonly seen near prey, are often blamed for livestock or game-stock losses, and are considered more of a problem than predators that are seen less frequently (Marker *et al.*, 2003e; Rust & Marker, 2014). Cheetahs opportunistically prey on a diverse range of species from rodents to large ungulate species; however, they most commonly target small- to medium-sized ungulates (23 − 56 kg), preferentially preying on impala, springbok, Grant's gazelle *Nanger granti*, Thomson's gazelle, Kob *Kobus kob* or Dorcas gazelle *Gazella dorcas* depending on regional availability (Kingdon, 1997; Hayward *et al.*, 2006). It is believed that cheetahs prefer small- to medium-sized prey due to ease of capture, reduced risk of injury and less chance of kleptoparasitism (Hayward *et al.*, 2006; Hilborn *et al.*, 2012); however, bigger sized prey are occasionally killed. Cheetahs are able to kill cattle up to six months of age and smallstock of all ages (Marker-Kraus *et al.*, 1996). Cheetah prey analyses on Namibian farmland have found that livestock contributes 4 – 7% of the cheetah's diet, with common indigenous game species accounting for the majority of the diet (Marker *et al.*, 2003e; Wachter *et al.*, 2006).

This study aims to identify the species most commonly preyed upon by cheetahs on Botswana farmland and to determine prey preferences in relation to abundance. These preferences are compared with farmers' perceptions of common prey items and reported stock losses. Prey species can be identified using morphological or genetic analysis of prey remains in scat (Klare *et al.*, 2011; Koirala *et al.*, 2012), or through predator kill identification either opportunistically or by using data from collared individuals (Bissett & Bernard, 2007; Martins *et al.*, 2011b; Tambling *et al.*, 2012). Due to the difficulties in locating kills on farmland in dense bush (Marker *et al.*, 2003e), the rarity of scavenging behaviour in cheetahs (Caro, 1982), and the large costs of genetic analysis, this study used the morphological analysis of prey remains in scat, specifically hair to identify the consumed prey species.

6.3 Study area and methodology

6.3.1 Study area

Cheetah scats and questionnaire data were collected from game ranches and livestock farms in the Ghanzi farming block (Fig. 2.15), within the Ghanzi district of Botswana (Chapter 2). Game-stock species ranched on game ranches in Ghanzi included giraffe, plains zebra and a variety of medium- to large-sized antelope species, including blue- and black wildebeest, waterbuck *Kobus ellipsiprymnus*, gemsbok, eland, springbok, impala, red hartebeest and blesbok. Free-ranging prey species available in Ghanzi were steenbok, duiker, springhares, scrub hares, jackals, rodents, birds and warthog. Kudu were present on game ranches and also as free-ranging game on cattle farms. From camera-trap data kudu were 1.5 times more abundant on livestock farms (i.e. free-ranging) than farms ranching only game-stock (i.e. stocked; Chapter 5).

6.3.2 Questionnaires

As part of a questionnaire on farmers' attitudes and perceptions of predators, stock losses and potential methods to mitigate HPC (Chapter 4), data were collected from 80 game ranches and 27 livestock farms across Botswana. Farmers were asked which prey species they considered was the primary item consumed by predators on their farm. Responses were compared with farmers' reported tolerance scores to predators, scored zero (low tolerance) to five (high tolerance), based on how many stock losses a farmer would tolerate losing to predators before removing the predator from the farm (described in Chapter 4).

From the questionnaire data, 31 farms were in the Ghanzi district, 28 of which were within the Ghanzi farming block and 26 of which reported seeing tracks or signs of cheetahs on their farm at least yearly. This included 10 livestock farms, four game-stock only ranches and 12 L/G farms (where both livestock and game-stock were ranched). In total, the questioned farmers owned ca. 43.4% (5709 km^2) of the farming block (13,152 km^2) and all farmers ranching with game-stock in the area were among those questioned. Farmers were asked what they considered the primary prey species consumed by cheetahs on their farm. Additional data from the questionnaire regarding the number of livestock and game-stock present on the farm and the number of livestock and game-stock farmers estimated that had been killed by cheetahs in the 12 months prior to study were compared with the scat-analysis results.

6.3.3 Scat collection and analysis

Cheetah scats were collected between February 2007 and May 2012. The majority of samples were collected from cheetah marking-sites, primarily trees. Scats were also collected during physical health checks directly from the rectum, at kill-sites or opportunistically throughout the research period. Scats were collected from cheetah marking-sites every 2 - 6 weeks. The mean time between cheetahs visiting marking-sites in the Northern Transvaal was 6.0 days (Marnewick *et al.*, 2006), compared to the 2.0 – 4.6 days that presented food items were observed in cheetah scats during feeding trials (Marker *et al.*, 2003e). Additionally, camera-trap data revealed that cheetahs generally stayed at a marking tree for a maximum of 10 minutes (J. Horgan pers. comm.). Therefore, any bias introduced by potentially collecting multiple scats from the same meal (Davies-Mostert *et al.*, 2010) although possible (as multiple members of male coalitions may have defecated at the site at the same time) is thought to be minimal.

A sample of each scat (ca. 50%) was removed from marking-sites; the remainder of the scat was marked with a dot of white correction fluid, to identify it as already having been sampled. In wolf scats bone and feather quills were commonly located at the centre of the scat; by sampling only a proportion of the scat these could have been missed (Spaulding *et al.*, 2000). However, it was considered important to avoid altering cheetah-marking behaviour which could occur if the whole scat was removed. The scat was identified as cheetah scat based on shape, size, colour and odour. Identity was confirmed by the presence of cheetah spoor, cheetah kills or sightings of cheetahs at the collection site or by the presence of cheetah hair (from grooming) in the scat.

Scat sample analysis was based on the method used by Davies-Mostert *et al.* (2010). Scats were air dried then crushed before analysis. In the case of large scats, a random sample of approximately 30 ml was crushed. Approximately 15 hairs were selected from each scat sample, ensuring all hair types (based on macroscopic analysis of length, diameter, colour and banding patterns) were included. In the case of a large variation in the physical appearance of the hair, a larger number of hairs were selected. The crushed scat or remainder of the scats were either frozen or stored in ethanol for future analysis. All feathers and bone fragments from the scat sample were collected and identified to the species level when possible.

Collected hairs were cleaned of any visible adhesions and placed inside a medium-tipped pasteur pipette for cross-sectional analysis. Cross-sections have been found to be more reliable than scale imprints for identification of species (Keogh, 1983; Henschel & Skinner, 1990). Hairs were placed so the middle length of the hair (which has the most consistent and species-distinct pattern [Keogh, 1983]) was in the sampled piece of the pipette. The pipette was filled with molten bees-wax. Upon solidifying, the pipette was cut into several thin cross-sections (ca. 0.2 mm) using a surgical blade. The cross-sections were glued to a microscope slide using wood glue that dries clear. Hairs were identified at 400 x magnification with a standard light microscope, by comparing the cross-sectional appearance of the hair with a reference collection of hair samples. The reference collection was taken from Kent (2004) and Davies-Mostert *et al.* (2010), and was supplemented by hair samples collected from livestock and various ungulate, mustelid and rodent species resident in the study area. Hair type varies across the body (Davies-Mostert *et al.*, 2010), therefore, hair samples were collected from across the body of small prey and from the rump and inner-leg of large species, the regions first consumed by cheetahs (Phillips, 1993). A total of 33 species were included in the reference collection (Appendix 6), including cheetah and leopard hair samples which were collected for comparison purposes. To minimise observer bias, all samples were identified by a sample number. Prey remains were identified by the same observer on two separate occasions; any samples for which the identified species differed between occasions were re-examined.

Due to the differential digestibility of different prey items and the resulting variation in the number of scats produced, a correction factor is necessary to improve the accuracy of quantifying the contribution different prey species make to the cheetahs' diet (Floyd *et al.*, 1978; Hiscocks & Bowland, 1989; Klare *et al.*, 2011). A correction factor from feeding trials in cheetahs was applied, i.e. $y = 0.0098x + 0.3425$, where y = the kg of prey consumed per scat and x = the average mass of an individual of a given species in kg (Marker *et al.*, 2003e). Among large species, such as cattle, waterbuck, eland, gemsbok, kudu, blue- or black wildebeest, red hartebeest or warthog, cheetahs most commonly prey on calves or young animals (Marker-Kraus *et al.*, 1996); therefore, calculations for these species were based on the upper limit of new-born body mass quoted in Bothma (1989). For all small- to medium-sized prey, (including smallstock, but excluding birds, springhares and scrub hares, for which adult body mass was assumed) body mass was calculated as equal to 0.75 times of the adult female body mass, thereby accounting for the consumption of all age classes, in accordance with Hayward *et al.* (2006). Prey body masses were primarily cited from Bothma (1989); additional species were cited from Marker *et al.* (2003e) or Kingdon (1997), and the average mass of domestic stock in the Ghanzi farming block were estimated by members of the farming community (C. Woolcott pers. comm.). The frequency of occurrence (defined as the proportion of the total number of scats in which a prey species was found [Ackerman, Lindzey & Hemker, 1984]) and the percentage occurrence after correction (the number of times a specific prey species was found as a percentage of all items found [Ackerman *et al*., 1984]) of each prey species was calculated. The percentage occurrence of livestock and game-stock species in the diet was used to estimate the number of livestock and game-stock killed by cheetahs per farm, per year in accordance with methods used by Marker *et al.* (2003e) and was compared with farmers' estimates from questionnaires.

6.3.4 Prey preference index

For each mammalian prey species consumed a prey preference index (*D*) was calculated using the Jacobs' Index (Jacobs, 1974):

$$D = \frac{(r-p)}{(r+p-(2rp))}$$

where, r is the proportion of occurrences in the diet, based on the number of individuals of a specific species consumed after use of the correction factor, relative to the total number of individuals consumed of all species, and p is the abundance of the prey species relative to the abundance of all prey species.

Relative abundance was calculated from the number of times a species was independently photographed in a camera-trap survey on livestock and game-stock farms in the Ghanzi farming block. The camera survey was conducted between October 2009 and January 2010 and pairs of cameras were placed at 26 camera-stations located along sand roads, for a total of 1063 camera-trap nights (see Chapter 5 for further details). The number of captures (i.e. photographs) of each species was recorded and calculated per 100 trap nights, in accordance with Stein *et al.* (2008). A capture was included if it occurred more than one hour after the previous capture of the same species. Small r or p values (less than 0.01) indicate the sample size is too small to reliably estimate the prey preference index and preference indices for these species should be considered with caution (Tjorve *et al.*, 2005). The prey preference index is constrained to lie between '−1' and '+1'. A preference for a species is indicated by an index above '0' and an avoidance of a species is designated by an index below '0' (Jacobs, 1974). Preference indices close to the extremes ('−1' and '+1') are considered to show a strong avoidance or preference to the prey item. As livestock were constrained to paddocks and did not have an equal opportunity for capture during the camera survey, the abundance of livestock was extrapolated from the density of cattle, sheep and goats on the questioned farms (n = 24) to estimate the abundance of livestock in the Ghanzi farming block (13,152 km^2).

6.3.5 Birthing season of prey

The birthing season and/or peak birthing months for each prey species were cited from the BWPA handbook (BWPA, 2005). The number of scats containing identifiable remains of each prey species, that were collected during this birthing period plus the subsequent month, was compared with the number of identified scats for each prey species collected during the remainder of the year, whilst compensating for differences in sample effort. For species with perennially breeding (including livestock) comparisons were drawn between the number of scat samples collected in each quarter of the year (1^{st} – January-March, 2^{nd} – April-June, 3^{rd} – July-September and 4^{th} – October-December).

6.3.6 Statistical analyses

All statistical tests were conducted using the statistical computer package SPSS version 11.0.1. Differences in the proportions of prey species identified in scats collected at different sites or in different years were analysed using Chi-squared tests using Yate's correction factor for tests with one degree of freedom. Prey species with a low incidence in the diet were collapsed into fewer categories to avoid sample size violation. Spearman's correlation was used to examine the relationship between farmers' perceptions of the main prey species killed by cheetahs and the percentage occurrence of prey species. Means are presented with their standard deviation (SD). All tests were two-tailed.

6.4 Results

6.4.1 Prey analysis of the cheetahs' diet

Scat samples were collected from the Ghanzi farming block (game ranches n = 162; livestock farms n = 95, unknown land use n = 2) between February 2007 and May 2012. A larger proportion of samples were collected in 2011/12 (49.0%) compared to 2007/08 (15.2%) or 2009/10 (35.8%); $\chi^2 = 44.88$, df = 2, $p < 0.001$). However, there was no significant difference in the overall number of samples collected during the wet or dry season ($\chi^2 = 2.10$, df = 1, $p = 0.147$) or between each quarter of the year ($\chi^2 = 6.41$, df = 3, $p = 0.093$). Prey remains were identified from 94.2% of scat samples. All prey remains were identified to a species, with the exception of one scat sample, which contained feathers from an unidentified bird species.

Livestock remains were found in 11.5% of scat samples, which equated to 5.9% of the cheetahs' diet after data were adjusted for differential digestibility (Table 6.1). Free-ranging species were found in 45.9% of scats which equated to 58.9% of the diet and game-stock species were found in 42.6% of scats accounting for the remaining 35.3% of the cheetahs' diet. The game-stock species that accounted for the largest proportion of the cheetahs' diet were waterbuck (accounting for 12.9% of the diet), kudu (7.7%) and blue wildebeest (3.8%; Table 6.1).

6.4.2 Location of scat samples: marking-sites

The majority of scat samples (88.8%) were collected at cheetah marking-sites, predominantly marking-trees. Samples were collected from seven different marking-trees; the median number of scats collected per tree was 19, (range 4-95). The number of scats collected per tree varied depending on the duration of time each tree had been sampled. The number of scat samples containing the remains of large prey species (cattle, kudu, blue wildebeest, eland, gemsbok, red hartebeest, waterbuck and plains zebra) or small- to medium-sized prey species (springhare, scrub hare, birds, steenbok, duiker, springbok, impala, blesbok and smallstock) collected at marking-sites (large: 62.2% and small-medium: 30.9%, $n = 233$) were statistically similar to the number of scat samples containing large (39.1%, $n = 23$; $\chi^2 = 1.49$, $df = 1$, $p = 0.222$) or small- to medium-sized prey remains (56.5%, $n = 23$; $\chi^2 = 3.40$, $df = 1$, $p = 0.065$) collected at non-marking-sites.

6.4.3 Prey availability and prey preference

Kudu (PI = 0.17), duiker (PI = −0.03)and plains zebra (PI = 0.04) had preference indices (PI) close to zero, showing they were consumed in relation to their abundance in the Ghanzi farming block. Waterbuck (PI = 0.86) and to a smaller extent steenbok (PI = 0.43) were preferentially preyed upon in relation to their abundance. Blue wildebeest (PI = −0.25), gemsbok (PI = −0.22), scrub hares (PI = −0.76), red hartebeest (PI = −0.34) and black-backed jackals (PI = −0.92) were avoided (Table 6.2). Despite the large abundance of warthog in the Ghanzi farming block, warthog remains

Table 6.1. Proportions of prey consumed by cheetahs in the Ghanzi farming block, Botswana, from scat-analysis using the Marker *et al.* (2003e) correction factor to compensate for the differential digestibility of prey

λ	Species	Scientific name	Assumed prey mass(kg) ł	Prey per scat (kg) ‡	No. of scats with prey remains	Frequency of occurrence	Mass of prey consumed	No. of individuals consumed	Percentage occurrence
F	Kudu (free-ranging) §	*Tragelaphus strepsiceros*	16.00	0.50	45	18.44%	22.47	1.40	15.67%
G	Waterbuck	*Kobus ellipsiprymnus*	13.00	0.47	32	13.11%	15.04	1.16	12.90%
F	Common duiker	*Sylvicapra grimmia*	13.88	0.48	33	13.52%	15.79	1.14	12.70%
F	Springhare	*Pedetes capensis*	3.50	0.38	10	4.10%	3.77	1.08	12.01%
F	Steenbok	*Raphicerus campestris*	8.48	0.43	20	8.20%	8.51	1.00	11.20%
G	Kudu (stocked) §	*Tragelaphus strepsiceros*	16.00	0.50	22	9.02%	10.98	0.69	7.66%
L	Cattle	*Bos taurus*	37.50	0.71	24	9.84%	17.04	0.45	5.07%
F	Bird ¶		1.00	0.35	1	0.41%	0.35	0.35	3.93%
G	Blue wildebeest	*Connochaetes taurinus*	16.00	0.50	11	4.51%	5.49	0.34	3.83%
G	Gemsbok	*Oryx gazella*	15.00	0.49	9	3.69%	4.41	0.29	3.28%
G	Springbok	*Antidorcas marsupialis*	27.75	0.61	13	5.33%	7.99	0.29	3.21%
F	Scrub-hare	*Lepus saxatilis*	1.90	0.36	1	0.41%	0.36	0.19	2.12%
G	Red hartebeest	*Alcelaphus buselaphus*	15.00	0.49	5	2.05%	2.45	0.16	1.82%
F	Black-backed jackal	*Canis mesomelas*	7.50	0.42	2	0.82%	0.82	0.11	1.24%
G	Plains zebra	*Equus quagga*	35.00	0.69	5	2.05%	3.43	0.10	1.09%
G	Common eland	*Tragelaphus oryx*	36.00	0.70	4	1.64%	2.78	0.08	0.86%
L	Goat	*Capris hircus*	45.00	0.78	3	1.23%	2.35	0.05	0.58%
G	Impala	*Aepyceros melampus*	37.50	0.71	2	0.82%	1.42	0.04	0.42%
L	Sheep	*Ovis aries*	37.50	0.71	1	0.41%	0.71	0.02	0.21%
G	Blesbok	*Damaliscus dorcas*	45.75	0.79	1	0.41%	0.79	0.02	0.19%

ł Prey body masses cited from Bothma (1989), Marker *et al.* (2003e), Kingdon (1997) or C. Woolcott pers. comm.

‡ Calculated with Marker *et al.* (2003e) correction factor

§ A total number of 67 scats containing kudu remains were collected. The proportion of free-ranging and stocked kudu was based on the assumption that kudu are killed by cheetahs in relation to their abundance (Jacobs' preference index = 0.17) and that kudu are 1.5 times more abundant on livestock farms compared to game ranches, from camera-trap data (Chapter 5)

λ Species unidentified – mass based on helmeted guinea fowl *Numida meleagris* (Marker *et al.*, 2003e) were not found in any of the cheetah scats (Table 6.2). The relative abundance of springbok (PI = 0.86) and springhare (PI = 0.98), and the relative occurrence of eland (PI = −0.67) and impala (PI = −0.69) were less than 0.01; so the PI values are considered unreliable.

Estimates of livestock abundance from camera-trap data were not available, so livestock was not

Table 6.2. Abundance and prey preference index of mammalian prey species in the cheetah's diet determined by scat-analysis, on the Ghanzi farming block, Botswana (13,152 km^2)

Species	**Relative occurrence in diet from scat-analysis**	**Abundance (numbers) ł**	**Abundance (per 100 camera-trap nights) ǂ**	**Relative abundance §**	**Preference index § λ**
Kudu	23.33%	(8195 ¶)	26.25	0.18	0.17
Waterbuck	12.90%	963	1.60	0.01	0.86
Common duiker	12.70%	-	20.04	0.14	−0.03
Springhare	12.01%	-	0.19	< 0.01	0.98
Steenbok	11.20%	-	7.15	0.05	0.43
Cattle	5.07%	119,353	-	-	-
Blue wildebeest	3.83%	8267	9.31	0.07	−0.25
Gemsbok	3.28%	4354	7.43	0.05	−0.22
Springbok ħ	3.21%	1745	0.38	< 0.01	0.86
Scrub hare	2.12%	-	20.23	0.14	−0.76
Red hartebeest	1.82%	1469	5.36	0.04	−0.34
Black-backed jackal	1.24%	-	33.68	0.24	−0.92
Plains zebra	1.09%	2782	1.51	0.01	0.04
Common eland	0.86%	6330	6.21	0.04	−0.67
Goat	0.58%	4182	-	-	-
Impala	0.42%	3452	3.39	0.02	−0.69
Sheep	0.21%	2410	-	-	-
Blesbok	0.19%	114	-	-	-
Warthog	0.00%	-	20.23	-	-

ł Estimated species abundance extrapolated from questionnaires with game ranchers and livestock farmers in the Ghanzi farming block, conducted between November 2012 and April 2013 (n = 28)

ǂ Estimated species abundance based on the number of times a species was photographed during a camera-trap survey conducted in the Ghanzi farming block, between October 2009 and January 2010 (Chapter 5)

§ Based on abundance (per 100 camera-trap nights)

λ Preference index was calculated using Jacobs' Index (Jacobs, 1974)

¶ Number of kudu stocked on game ranches; not does include free-ranging kudu

ħ Springbok were stocked on 11 farms (median = 40 springbok per farm, range 1 – 700) included in the preference index. However, livestock were 3.3 times more abundant than game-stock in the Ghanzi farming block (total number of livestock = 125,945; total number of game-stock = 37,671), yet accounted for a disproportionately small fraction of the diet, compared to game-stock (livestock = 5.9% of the cheetahs diet; game-stock = 35.3% of the cheetahs diet), so livestock are considered to be avoided.

6.4.4 Seasonal availability of prey

The date of collection was unknown for seven samples, therefore, analysis was based on 237 scats with identifiable prey remains. Although springbok breed all year round, the number of births peak during the September-January period (BWPA, 2005). During this peak birthing period 10.7% ($n = 103$) of scats collected contained springbok remains compared to 1.5% of scats collected outside of this period ($n = 134$; $\chi^2 = 7.36$, $df = 1$, $p = 0.007$). In contrast, 12.3% of scats collected during the kudu birthing season December-May contained kudu remains ($n = 122$) compared to 44.3% of scats collected outside of the birthing season ($n = 115$; $\chi^2 = 20.70$, $df = 1$, $p < 0.001$), and steenbok remains were equally likely to be detected during the steenbok peak breeding period (September−October; 4.5% of scats collected contained steenbok, $n = 66$) compared to rest of the year (9.9%, $n = 171$; $\chi^2 = 1.07$, $df = 1$, $p = 0.301$). Sample sizes were too small to individually test the remaining seasonal breeding species. However, 100% of scats containing blue wildebeest and red hartebeest remains, both large prey species whose birthing season is between September and February, were detected outside of this peak birthing period ($n = 14$; $\chi^2 = 13.60$, $df = 1$, $p < 0.001$).

For species with perennial breeding, duiker remains were more likely to be present in samples collected in the 2nd quarter of the year (April to June; 25.9% of scat collected contained duiker, $n = 58$; Z score = 2.61) compared to the 1st (7.1%, $n = 56$), 3rd (9.2%, $n = 76$) or 4th quarters of the year (12.0%, $n = 50$; $\chi^2 = 9.47$, $df = 3$, $p = 0.024$). Waterbuck remains were found equally throughout the year ($\chi^2 = 3.56$, $df = 3$, $p = 0.313$). Sample sizes were too small to test for quarterly patterns in the remaining perennial breeding species.

6.4.5 Farmers' perceptions of the primary prey items killed by predators

The species most commonly listed by farmers as being killed by cheetahs on Ghanzi farmland was springbok (26.9% of farmers, n = 26), followed by kudu (19.2%; Table 6.3). Livestock was listed as the cheetahs' primary prey by 13.6% of farmers farming with livestock (livestock and L/G farmers, n = 22), and game-stock (excluding kudu) was listed as the primary prey by 50.0% of game ranchers (game-stock only ranchers and L/G farmers, n = 16). There was no correlation between the proportion of farmers who ranked each prey species as the cheetahs' primary prey item and the proportional contribution each species made to the cheetahs' diet ($R_s = 0.284$, n = 18, p = 0.253; Fig. 6.1). Springbok were perceived to be more heavily predated than the empirical data showed, and the importance of waterbuck and springhares to the cheetahs' diet were underestimated (Fig. 6.1). Farmers questioned as part of the nationwide surveys who perceived livestock as the primary prey item killed by predators (19.0%, n = 100 [data was not available for seven farmers]) had a lower reported tolerance of predators (TS = 1.3), than farmers who

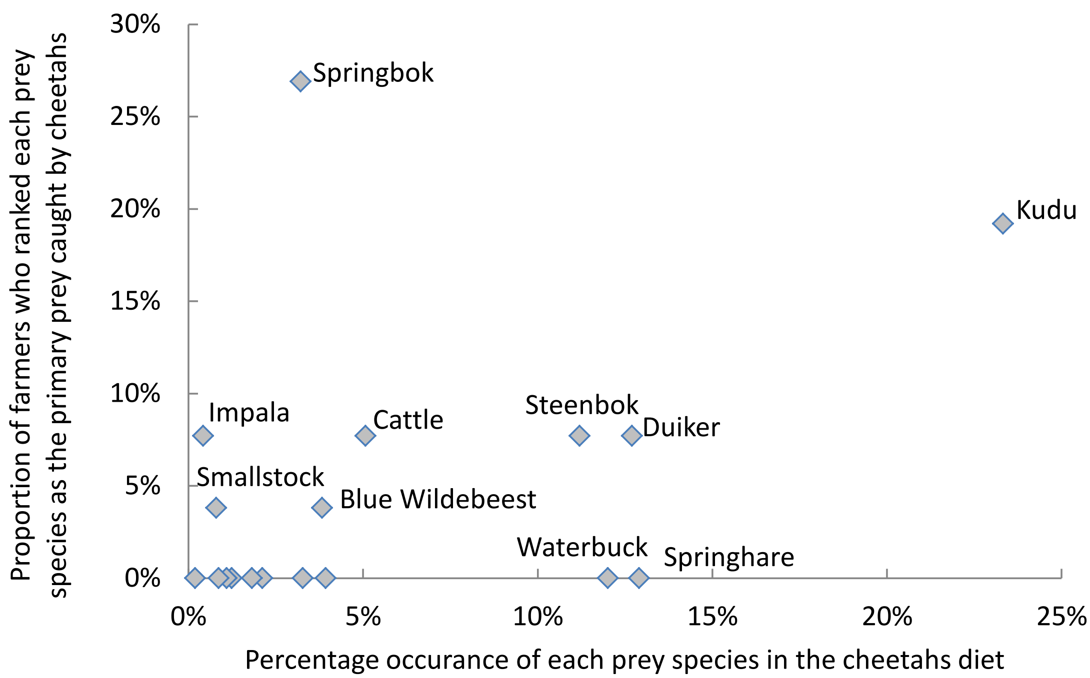


Figure 6.1. The relationship between farmers' perceptions of the primary prey item killed by cheetahs on their farm and the proportion contribution each prey item was found to make to the cheetah's diet from scat-analysis

perceived free-ranging wildlife species including kudu (26.0%; TS = 3.8; U(45), $z = -3.58$, $p < 0.001$) or game-stock (55.0%; TS = 3.2; $z = -3.09$, $p = 0.002$) as the primary prey item.

6.4.6 Estimates of livestock and game-stock depredation by cheetahs from scat-analysis

Free-ranging cheetahs have been reported to consume 1.4 - 5.3 kg of meat per cheetah, per day (Schaller, 1972; Mills *et al.*, 2004). This is equivalent to 511 – 1936 kg of meat per year (365.2 days in a year). In cheetah feeding trials Marker *et al.* (2003e) calculated a cheetah produces 1.87 scats for every kg of food ingested (Marker *et al.*, 2003e), resulting in 956 - 3620 field collectable scats per cheetah, per year.

Livestock

Livestock remains were found in 11.5% of scats (cattle 9.8%, goats 1.2%, sheep 0.4%, Table 6.1). Based on a cheetah producing 956 - 3620 field collectable scats per year, 94.0 - 356.1 scats per year would contain cattle remains, 11.8 - 44.5 scats would contain goat remains and 3.9 - 14.8 scats would contain sheep remains. After applying the correction factor described by Marker *et al.* (2003e) in conjunction with the assumed mass of prey species (listed in Table 6.1), the number of individuals consumed can be calculated as 1.8 - 6.7 cattle, 0.2 - 0.8 goats and 0.1 - 0.3 sheep per cheetah, per year. It is estimated in the Serengeti cheetahs lose 12.9% of their kills to scavengers

Table 6.3. Species identified by commercial farmers as the most common prey species killed by cheetahs on farmland in the Ghanzi farming block, Botswana, on which tracks or signs of cheetahs were seen at least yearly

Prey Species	Proportion of farmers n = 26
Springbok	26.9%
Kudu	19.2%
Steenbok	7.7%
Impala	7.7%
Cattle	7.7%
Common duiker	7.7%
All game	7.7%
Blue wildebeest	3.8%
Smallstock	3.8%
Do not know	7.7%

(Hunter *et al.*, 2007a), therefore, assuming cheetahs kill 12.9% more animals than they consume ; the number of livestock killed by cheetahs is estimated as 2.0 – 7.6 cattle, 0.2 – 0.9 goats and 0.1 - 0.3 sheep per cheetah, per year. Based on an estimated cheetah density of 0.32 cheetahs per 100 km^2 (Chapter 5) and a median livestock farm size per owner of 90 km^2 in the Ghanzi farming block (n = 24) this amounts to each farm losing 0.6 - 2.1 cattle, 0.1 - 0.2 goats and 0.0 - 0.1 sheep to cheetahs per year. Equivalent to 0.1 – 0.2% of the median number of cattle stocked per farm (median = 1000, n = 24) and 0.2 – 0.8% of the median number of smallstock stocked per farm (median = 31, n = 24). These estimated losses are greater than the median loss reported by farmers in the Ghanzi farming block with resident cheetah populations, who reported losing a median of 0.2 cattle (range 0 – 14, n = 24) and 0.0 smallstock (range 0 – 1, n = 7) per farm, per year to cheetahs.

Game-stock

Game-stock species preyed upon by cheetahs were kudu, waterbuck, blue wildebeest, gemsbok, springbok, red hartebeest, plains zebra, eland and impala. Remains of game-stock species were identified in 42.6% of scats (Table 6.1), equating to 407.4 – 1543.0 scats per year. After applying the correction factor (Marker *et al.*, 2003e) and assuming that cheetahs kill 12.9% more animals than they consume (Hunter *et al.*, 2007a), the total number of game-stock killed per cheetah, per year is 14.0 − 53.0. Based on a cheetah density of 0.32 cheetahs per 100 km^2 (Chapter 5) and a median farm size of 89.0 km^2 in the Ghanzi farming block (based on the area of ranches farming only game-stock and the area of the game ranching section of L/G farms reported in questionnaire data, n = 17), this amounts to each farm losing 4.0 – 15.1 game-stock to cheetah per year (0.3 − 1.1% of the median number of total game-stocked [median = 1338, n = 16]). These estimated losses were 1.6 – 5.9 times less than the losses reported by Ghanzi farmers with resident cheetah populations (seen at least yearly), who reported losing a median of 23.6 game-stock (range 0 − 160, n = 6) per farm, per year to cheetahs.

6.5 Discussion

6.5.1 Limitations

The accuracy of determining a carnivore's diet from scat-analysis is limited by the accuracy of the feeding trials upon which the correction factor and, therefore, results are based. Studies have shown that the correction factor can vary with the sex and age of the carnivores used in the feeding trial, the time they have spent in captivity and in relation to the prey species used (Reynolds & Aebischer, 1991; Klare *et al.*, 2011). Even in nearly identical feeding trials in wolves, the correction factor differed between studies (Floyd *et al.*, 1978; Weaver, 1993; Ruehe *et al.*, 2003). Additionally, the use of linear regression to calculate correction factors is potentially inherently biased, because it incorrectly assumes that the consumed prey mass to excrete one scat will not reach an asymptote (Wachter *et al.*, 2012).

As many ungulates exhibit sexual dimorphism and size differences across age classes (Bothma, 1989), the correction factors are also limited by the lack of information regarding the size, sex, age and proportion of the prey consumed (Mills, 1992; Klare *et al.*, 2011). Floyd *et al.* (1978) suggested that the mass of rapidly growing prey could be adjusted for the time of year the scat was collected and Wachter *et al.* (2012) suggested the use of a second correction factor to compensate for the proportion of the carcass consumed. Future studies should consider using these adjustments to improve the accuracy of correction factors and prey analysis results.

Manual identification of prey remains can be subjective (Mills, 1992); there is a possibility for observer bias (Spaulding *et al.*, 2000), and ideally two independent observers should identify prey remains in the future. Alternatively, the use of pattern recognition technology (Moyo *et al.*, 2006; Foster *et al.*, 2011) or genetic analysis (Casper *et al.*, 2007; Koirala *et al.*, 2012) could potentially remove or limit the degree of subjectivity involved. Genetic analysis would also enable the species from which the scat sample originates to be confirmed (Janecka *et al.*, 2011) and to determine the identity and sex of the cheetah which would enable a more accurate analysis of the cheetah's diet.

Lastly, the sample size remains a limitation of this study; Reynolds and Aebischer (1991) calculated 9600 scats with identifiable prey remains would be required to accurately establish the relative proportions of prey consumed by red foxes *Vulpes vulpes*. It is impractical and likely to be impossible, to collect such a large number of scats from a low-density species, such as the cheetah (Marker *et al.*, 2003e). However, a larger sample size would have enabled further comparisons to be drawn with covariates such as the location and season of scat collection.

6.5.2 Prey items and preferences

Cheetahs preyed upon 19 species in the Ghanzi farming block (Table 6.1); however, the majority of the cheetahs' diet consisted of a limited number of prey items, namely kudu (23.3%), small antelope (23.9%; duiker and steenbok) and springhare (12.0%). This pattern of consuming a broad range of species, but predominantly focused on a few selected species, has also been observed in cheetahs in protected areas (Caro, 1994; Hunter, 1998; Mills *et al.*, 2004). Cheetahs from the Ghanzi farming block predominantly fed upon free-ranging wildlife species (58.9% of diet). Game-stock made up 35.3% of the cheetah's diet, and livestock made up 5.9% of the diet despite livestock species being 3.3 times more abundant than game-stock. Cheetahs on Namibian farmland also primarily preyed upon abundant indigenous wildlife species despite a greater abundance of livestock (Marker *et al.*, 2003e). A review of cheetahs' prey preferences across southern and East Africa showed that cheetahs' primarily prey upon small- to medium-sized animals (23 – 56 kg) based on their relative abundance (Bissett, 2004; Hayward *et al.*, 2006). Kudu was the most abundant game species available on the Ghanzi farmland, occurring both on game ranches and free-roaming on livestock farms, and this correlates with its status as the primary prey species consumed. Kudu, where locally abundant, were also the primary prey of cheetahs in Namibia and of reintroduced cheetahs in South Africa (Hunter, 1998; Marker *et al.*, 2003e).

Kudu, duiker and plains zebra remains were present in scat samples in relation to the species relative abundance in the study area, while waterbuck, springhare, steenbok and springbok were preferentially preyed upon. Abundance data were based on a camera-trap survey conducted from October 2009 to January 2010, prior to the peak period of scat collection in 2011 – 2012 and the camera survey only covered an area of 475 km^2. Consequently, the calculated prey abundance may not be a true representation of prey availability for all species throughout the Ghanzi farming block. Springhares had the highest prey preference index (0.98); however, their relative abundance was possibly underestimated as the camera survey was not conducted in prime springhare habitat, and springhares were photographed at only two of the 26 camera locations (Chapter 5). Additionally, due to differences in body size and species specific behaviour, including habitat use and animal movement, camera capture frequencies are considered a poor estimator of relative abundance (Jennelle *et al.*, 2002; Tobler *et al.*, 2008). However, camera trap rates have been previously used to determine the relative abundance of species (Carbone *et al.*, 2001; Stein *et al.*, 2008), and were considered the best information available at the time of study.

Waterbuck were also preferentially preyed upon and were the most commonly consumed game-stock species. Waterbuck are native to the wetland areas of northern Botswana, but do not naturally occur in the semi-arid environment of Ghanzi (IUCN SSC, 2008). Species that are not native to an area are less likely to be adapted to local environmental conditions and are potentially more susceptible to predation (Marker-Kraus *et al.*, 1996; Marker & Schumann, 1998); this could have contributed to the observed prey preference for waterbuck. However, waterbuck were not listed as the main prey killed by cheetahs by any of the farmers questioned; therefore, this level of predation is not thought to be negatively affecting herd growth or contributing substantially to human-cheetah conflict.

The majority of scat samples were collected from cheetah marking-sites and although females are known to visit marking-sites, these locations are predominantly used by males (Marnewick *et al.*, 2006). Male cheetahs are more likely to kill large antelope species such as kudu, waterbuck, eland and gemsbok than are females, who predominantly feed on smaller prey such as steenbok and duiker (Caro, 1994; Hunter, 1998; Marker *et al.*, 2003e). When killing large species, cheetahs predominantly target calves (Marker-Kraus *et al.*, 1996), however, male cheetahs, especially in coalitions are capable of killing sub-adults and potentially adults of these larger prey species (Caro, 1994; Mills *et al.*, 2004; Bissett & Bernard, 2007). The bias to collecting scats from males could, therefore, explain why the scats containing kudu, blue wildebeest and red hartebeest (i.e. large antelope species) were commonly collected outside of the relevant peak birthing periods, implying predation on these species was not restricted to calves. However, no significant difference was seen in the proportion of scat samples containing large prey collected at marking-sites compared to non-marking-sites. Therefore, although the mass of large species used in calculations (which was presumed to be that of a new-born calf [Bothma, 1989]), could have been underestimated, any resulting bias is thought to be negligible. Additionally, any bias is likely to have increased the proportion of large game-stock species in the diet; therefore, the contribution of game-stock species to the cheetahs' diet is still predicted to be minor compared to free-ranging prey species.

Springbok was the prey species most commonly listed by farmers as being the cheetahs' primary prey item in the Ghanzi farming block, and the BWPA handbook refers to 'cheetahs in particular causing heavy losses to springbok' (BWPA, 2005). The relative abundance of springbok was below the threshold considered necessary to ensure the reliability of prey preference indices, however, if correct, it does appear that springbok were preferentially preyed upon by cheetahs (PI = 0.86). In protected areas within the Kalahari, springbok are the most common prey item killed by cheetahs (Mills, 1984) and, under natural conditions, springbok should be the most abundant antelope species in the Ghanzi district, within the 23 – 56 kg weight category, identified by Hayward *et al.* (2006), as being preferred by cheetahs. However, springbok populations are declining throughout Botswana's protected areas and farmlands (Skinner Frssaf & Moss, 2004; Ghanzi farming community member, pers. comm.), and the national population is estimated to have declined by 71% between 1992 and 2012 (DWNP, 2013). The reasons for the decline are unknown (Skinner Frssaf & Moss, 2004), but habitat loss due to human encroachment and climate change, in conjunction with poaching and predation are thought to play a role (BWPA, 2005). The low relative abundance may, therefore, explain why springbok accounted for a relatively small proportion of the cheetahs' diet (3.2%).

Springbok favour dry, open plains with short grass cover and often congregate in large dense herds (Kingdon, 1997). However, many of the game ranches in the Ghanzi farming block were previously livestock farms and are often bush encroached with reduced grass cover, because of poor veldt management (Moleele *et al.*, 2002; Hejcmanova *et al.*, 2010). The increased bush cover and, therefore, camouflage for predators could make springbok more vulnerable to carnivore depredation (Bednekoff & Ritter, 1994). A larger proportion of springbok remains were observed during the peak breeding months as compared to the rest of the year. Springbok hide their lambs during the first two days after birth (Apps, 2000), and a reduction in grass cover has been associated with increased susceptibility to predation in roan antelope calves, which also remain hidden after birth (Harrington *et al.*, 1999). Therefore, cheetahs could be impacting upon the springbok's breeding success by targeting young lambs, and the assumed mass of springbok used in calculations (assumed to be sub-adult to adult) was potentially overestimated, with a larger proportion of juveniles consumed than accounted for. If this was the case, the proportion of springbok in the cheetahs' diet could have been underestimated.

The degradation of vegetation negatively affects springbok feeding behaviour (Stapelberg *et al.*, 2010) and species living in sub-optimal habitat often have lower population growth rates, lowered fecundity and increased mortality from all factors (Smuts, 1978; Dunham *et al.*, 2003). Springbok on game ranches could also be more susceptible to carnivore depredation, because they are often stocked in small numbers (median = 40 on game ranches in the Ghanzi farming block). Larger herds have increased vigilance and are able to spot predators from further away (Hunter & Skinner, 1998), demonstrated by cheetahs greater hunting success in game herds of less than 30 individuals (Eaton, 1970b). Additionally, springbok on game ranches are potentially naive to predators, having been sourced from farms in Namibia that are generally devoid of large carnivores (farming community member pers. comm.). Prey species that are naive to predators are less vigilant and exhibit lower survival rates when introduced to areas containing predators than individuals with experience of carnivores (Hunter & Skinner, 1998; Griffin *et al.*, 2000; Shier & Owings, 2006).

Overall, farmers' perceptions of the species primarily preyed upon by cheetahs' did not correlate with the proportional contribution each prey species made to the diet from scat-analysis. Although the importance of kudu, steenbok and duiker in the cheetahs' diet was recognised by some farmers, cheetahs' reliance on springbok was grossly overestimated. Conversely the importance of springhares and scrub hares were vastly underestimated. Springhare and scrub hare combined, were the second most common prey items, contributing to 14.1% of the cheetahs' diet in the current study and 41 – 43% in similar studies in Namibia (Marker *et al.*, 2003e; Wachter *et* *al.*, 2006), yet neither species was listed as a top prey item consumed by cheetahs by any of the interviewed farmers.

Although cheetahs are active on moonlit nights, they are primarily a diurnal species (Cozzi *et al.*, 2012) and are, therefore, more frequently sighted on farms than are other nocturnal predators, such as leopards (Marker *et al.*, 2003e). Farmers' perceptions of the cheetahs' main prey is sometimes based on sightings of cheetahs near specific livestock or game-stock. However, the number of times cheetahs are seen near livestock or game species is not representative of the prey species consumed (Marker *et al.*, 2003e). Farmer's perceptions may also be based on the opportunistic finding of kills which is biased to larger prey species, and may explain why the importance of springhares and scrub hares in the cheetahs' diet was not recognised by farmers (Mills, 1992; Marker *et al.*, 2003e). Additionally, kills reported by farmers are often biased to diurnal, larger game species which can be easily monitored, typically savannah species (Swanepoel, 2008). Population declines or failure to increase of a game-stock species is often taken as evidence of predation (Thorn *et al.*, 2012), potentially explaining why springbok were the prey species most commonly reported by farmers as the cheetahs' primary prey.

6.5.3 Livestock and game-stock depredation

Overall, livestock contributed 5.9% to the cheetahs diet, within the 4 – 7% range previously reported from scat-analysis in Namibia (Marker *et al.*, 2003e; Wachter *et al.*, 2006). Based on the percentage occurrence of livestock in the diet from scat-analysis, it was estimated that an average farm loses 0.6 − 2.1 cattle, 0.1 − 0.2 goats and 0.0 − 0.1 sheep to cheetahs per year. However, these results were based on a kill-to-consumption rate of 1.13 : 1 as calculated in the Serengeti National Park (Hunter *et al.*, 2007a), where kleptoparasitism from lions and hyaenas is likely to be greater than on Ghanzi farmland, where lions and spotted hyaenas are rare (Chapter 5). Additionally, the upper limit of stock lost is likely to be an overestimate as it is based on the highest reported meat consumption of 5.3 kg per day. Therefore, it is not surprising that the median livestock losses reported by farmers of 0.2 cattle and 0.0 smallstock per farm in the 12 months prior to the survey, was lower than the scat-analysis results.

Despite this tendency for scat-analysis results to overestimate the number of stock losses, game ranchers estimated losing 1.6 – 5.9 times more game-stock per farm, per year than estimated through scat-analysis. Farmers frequently overestimate losses to carnivore depredation (Dickman, 2008), and it can be more difficult for ranchers to monitor game-stock numbers, births and deaths than for livestock (Chapter 4). Farmers' estimates of the number of game-stock killed by predators was correlated with the number of years of game ranching experience the farmer had (Chapter 4), implying estimates were often based on farmers' attitudes to predators including assumptions as to the number of predators present, perceived prey preferences and the frequency with which predators kill (Chapter 4). Data from this study showed that the species farmers' perceive as predators primary prey items are often incorrect. These perceptions can affect farmers' tolerance of predators; approximately, 19.0% of farmers questioned nationwide considered livestock to be predators' primary prey and this perception was associated with significantly lower tolerance scores (TS =1.3) to predators than farmers who thought that free-ranging wildlife (TS = 3.8) or game-stock (TS = 3.2) were the primary prey.

6.5.4 Conclusion

Overall, this study showed cheetahs on Botswana farmland to predominantly prey upon free-ranging wildlife species, namely kudu, steenbok, duiker and springhares. Game-stock species, especially those outside of their natural habitat such as waterbuck and springbok, may be preferentially preyed upon compared to abundance, potentially increasing HPC on game ranches. Livestock was shown to make up a small proportion of the cheetah's diet, but even relatively low predation rates, in conjunction with negative perceptions and attitudes toward predators, can result in HPC and have impacts on local predator populations (Oli *et al.*, 1994; Madden, 2004). Therefore, conflict mitigation and education programmes which alter farmers' perceptions of predators feeding habitats are likely to increase tolerance of predators.

Chapter Seven: The utilisation and perceived effectiveness of methods to prevent or mitigate the conflict between humans and predators on farmland

7.1 Chapter overview

In the previous chapters the drivers and direct costs of HPC on Botswana game ranches and livestock farmland were discussed. Farmers' perceptions of the species predominantly preyed upon by predators (Chapter 6), the predators present on the farm (Chapter 5), the farmers' game ranching experience, first language spoken (in association with age) and education level (Chapter 4) were found to be significant factors associated with the reported number of stock losses to carnivore depredation and farmers' tolerance of predators. The current chapter examines the mitigation methods available to farmers to reduce HPC. It addresses the animal husbandry methods available for the prevention of livestock and game-stock losses to carnivore depredation, compensation for these losses and methods to increase the financial benefits derived from predators. Questionnaires were used to investigate the use of conflict mitigation methods by game ranchers and livestock farmers, the reasons why methods were not utilised and the perceived effectiveness of these methods to enable predators and humans to coexist.

Hunting of 'problem' predators, trophy hunting of predators under quota and photographic tourism, all methods which change predators from being perceived as an economic cost to an asset, were considered the most effective methods of promoting predator-human coexistence. Translocation of 'problem' predators (Chapter 8) and the utilisation of livestock management techniques, which aim to reduce carnivore depredation, were considered the next most effective methods to enable coexistence. Smallstock were more actively protected from predators than cattle, and farmers stated that they were reluctant to adopt livestock husbandry techniques due to limited farm infrastructure or because their implementation would result in lower cattle productivity, continued predation or a higher prevalence of livestock diseases. The utilisation of livestock management techniques were considered significantly more effective than game-stock management techniques, which were generally thought to be too expensive or too time-consuming to implement and to be ineffective at reducing game-stock losses to carnivore depredation. Compensation and insurance were considered the least effective methods.

Promotion of conservancies or eco-certification of hunting, photographic tourism and meat products were concepts which could potentially promote predator conservation within the game ranching industry.

7.2 Introduction

Human-predator conflict can be driven by the direct costs from carnivore depredation on livestock and game-stock and, as shown in Chapter 4 often to a greater extent, to farmers' attitudes and preconceptions of predators (Madden, 2004). Education and awareness initiatives directly address these potential misconceptions and negative attitudes (Marker *et al.*, 2003d; Strande-Straube, 2013), whilst mitigation and conflict prevention programmes indirectly affect attitudes by reducing or offsetting the costs of carnivore depredation (Mishra *et al.*, 2003). Chapter 4 showed that tolerance of predators was greater on game ranches than livestock farms, but that HPC did still exist. Mitigation methods should be tailored to address different cultures, industries and communities (Manfredo & Dayer, 2004), yet conflict mitigation and education initiatives which are directly aimed at HPC on game ranches are poorly developed (pers. obs.).

The Botswana government provides financial compensation for livestock killed by cheetahs, African wild dogs, leopards and lions (CAR, 2011), and cattle can be insured for losses from all causes, including predation, through the Botswana Insurance Company (N. Mtunzie pers. comm.). Farmers ranching livestock are permitted to kill predators if they have 'caused, are causing or threaten to cause damage to any livestock' or 'to human-life' (Part 4, no. 46, Botswana Wildlife and Conservation Act, 1992; [DWNP, 1992]). Farmers ranching both livestock and game-stock can also translocate a 'problem' predator with the assistance of the DWNP. However, compensation, insurance through the Botswana insurance company and lethal control of predators are not applicable to game-stock lost to carnivore depredation (DWNP, 1992; N. Mtunzie pers. comm.).

An alternative method to financially compensate for the direct costs of carnivore depredation is through eco-certification initiatives. These initiatives follow the trend for today's consumers to be concerned about the origins and ecological impact of their food or holiday (Aquino & Falk, 2001). Businesses can use a marketable logo that assures consumers that the assessed product meets a set of environmental (and often social) standards (Discover Limited, 2008). This serves to allow consumers to distinguish between companies and enables businesses to increase their patronage by environmentally concerned consumers and to charge a higher price for their product or services (Lewis & Alpert, 1997). Although not available in Botswana at the time of study (2012/2013), the implementation of an eco-certification initiative and the marketing of game-stock and livestock meat products as predator-friendly could result in a higher sale price for these products, as occurs in similar initiatives in South Africa and in the USA (Badgley, 2003; Tjaronda, 2006; Conservation International, 2014).

Husbandry techniques that reduce livestock or game-stock losses to carnivore depredation can also be utilised (Breitenmoser *et al.*, 2005). Traditional livestock management techniques such as livestock guarding dogs (LSGDs), kraaling of livestock (placing livestock in fenced enclosures), herding and synchronised breeding of livestock with wildlife calving times, have been associated with reduced livestock losses to predators (Ogada *et al.*, 2003; Woodroffe *et al.*, 2007; Stein *et al.*, 2010). More innovative techniques such as disruptive stimuli (e.g. electronic guards), aversive stimuli (e.g., electric shock or toxic collars), bio-fences and taste aversion (using emetic meat) are in development and have been used on commercial farms on a small scale (Gill *et al.*, 2000; Shivik, 2006; Jackson *et al.*, 2012).

However, due to the large size of game ranches and the non-domesticated, wide-ranging habits of game-stock, protection methods on game ranches are largely restricted to preventing the entry of predators to breeding areas or to the whole farm (Schumann *et al.*, 2006). Predators can gain entry to a farm through the wires in the fence or through holes under the fence-line, dug by digging species such as aardvark *Orycteropus afer* and warthog (du Toit, 1996; Van Rooyen *et al.*, 1996). Therefore, the use of electric fencing, underground or bonnox (diamond mesh) fencing, the filling of holes under the fence-line or the use of swing gates can prevent their entry. Swing gates are placed in holes under the fence similar to a cat flap. Warthog and other digging species quickly learn to use the gate (Fig. 7.1), reducing the number of holes dug under the fence-line, whilst predators are thought to be less likely to use the gate (Schumann *et al.*, 2006). Alternatively, some farmers provide carrion or more commonly stock inexpensive game-stock species, known as

Figure 7.1 Swing gate being used by warthog. Source: (CCB, n.d.)

'buffer species', to divert the attention of predators and satisfy their food requirements, thereby reducing predation on more valuable species (Cousins *et al.*, 2008). These methods are potentially more difficult and expensive to implement than traditional livestock husbandry. As a result, game-stock can be more difficult to protect from carnivore depredation than livestock (Marker *et al.*, 2003a), and farmers reported double the financial losses from carnivore depredation on game-stock compared to livestock (Chapter 4).

Despite the greater financial losses, game ranchers reported a greater tolerance of predators than livestock farmers (Chapter 4). Game ranchers have more opportunities to benefit ecologically and financially from predators than livestock farmers, especially when operating as part of a conservancy, and these benefits can potentially increase tolerance of predators (Lindsey *et al.*, 2009b; Lindsey *et al.*, 2013b). Ecologically, predators play a pivotal role in trophic cascades, and the presence of large predators is associated with healthier game-stock and vegetation communities and the control of meso-predator populations, such as jackals and caracals (Crooks & Soulé, 1999; Schmitz *et al.*, 2000; Ripple & Beschta, 2004; Ritchie & Johnson, 2009). Ranchers can also benefit financially from predators through photographic tourism, eco-certification of tourism or hunting operations as predator-friendly and through sustainable predator trophy hunting. From 2013, only non-protected predator species, namely spotted hyaenas, caracals and black-backed jackals could be trophy-hunted on private land in Botswana, within a quota (DWNP, 1992). However, they are considered unattractive trophy species, therefore, are generally hunted in small numbers (Mills & Hofer, 1998; Ray *et al.*, 2005). Trophy-hunting quotas for cheetahs and African wild dogs have never existed in Botswana. Lions were trophy-hunted in Botswana in the past, but this ceased in 2003 - 2005 and since 2007 to present (2013), due to international concern about lion conservation (Packer *et al.*, 2006). Leopard trophy hunting occurred until a moratorium was implemented in January 2013 (BWPA, 2012c), based on the precautionary principle that there was insufficient evidence on the sizes of leopard populations (DWNP director pers. comm.). The moratorium on leopard hunting led to requests from the BWPA that if game ranchers can no longer be compensated for their game-stock losses to carnivore depredation from leopard trophy hunting, then game-stock should be treated as an equal asset as cattle and should be applicable to the same compensation and protection laws as livestock (BWPA, 2012c).

The main drivers of conflict on Botswana's game ranches were not related to the economic direct costs of carnivore depredation (Chapter 4); therefore, mitigation methods designed for purely financial compensation could be less successful than measures which aim to change the image of predators from a cost to an asset (Montag, 2003). This chapter aims to discuss the use and applicability of the available conflict mitigation methods on Botswana commercial farmland, in relation to farmers' perceived effectiveness of the methods at enabling farmers and predators to coexist.

7.3 Study area and methodology

7.3.1 Study area and sampling

Questionnaire surveys were conducted nationwide with game ranchers in Botswana, as described in Chapter 4. For comparative purposes, questionnaires were completed with livestock farmers from the largest game ranching areas, primarily Tuli-block farms, Ghanzi farming block, and Hainaveld and Makalamabedi farms (Fig. 2.13). Livestock farmers were primarily selected using 'snowball' sampling, the limitations and advantages of which were discussed in Chapters 3 and 4. The study area and the status of the game ranching and cattle farming industries in Botswana at the time of study (2012/2013) were discussed in Chapter 2.

7.3.2 Questionnaires

As described in Chapter 4, sections E–H of the questionnaires referred to trophy hunting of predators, livestock and game-stock management techniques and photographic tourism (Appendix 3). Section I referred to the translocation of 'problem' predators, the results of which will be discussed in Chapter 8. Farmers were asked which farming techniques they used on the farm and, if appropriate, the reasons why they were not used. They were given a series of options: too expensive, too time-consuming, not needed (only game-stock), unsure how to do, decreases herd productivity, ineffective at reducing carnivore depredation or other (if other they were asked to state why). All of the farmers' additional comments were recorded; farmers often gave multiple comments, so the number of comments was potentially larger than the number of participants. In cases where questionnaire surveys were conducted with multiple representatives from the same farm, the farming techniques used were recorded once. The use of farming techniques was compared with the number of livestock and/or game-stock killed by predators, the predators present and the location and characteristics of the farm using data from Chapter 4. Expensive ungulate species stocked in Botswana were nyala *Tragelaphus angasii*, 'golden' wildebeest, black-faced impala *Aepyceros melampus petersi*, roan antelope, sable antelope, tsessebe, blesbok and black wildebeest.

Section J asked participants to rate how effective they perceived potential mitigation methods to be at enabling farmers and predators to coexist. The mitigation methods included the translocation of 'problem' predators (to be discussed in Chapter 8), livestock and game-stock management techniques, government compensation or private insurance for stock losses to predators, financial benefits from predators through trophy hunting, photographic tourism and eco-certification initiatives, membership of conservancies and the lethal control of predators. The perceived effectiveness of mitigation methods was graded on a four-point Likert scale of 'very effective', 'effective', 'ineffective' to 'very ineffective'. A four-point scale was used to avoid neutral responses; however, a 'do not know' option was available.

7.3.3 Randomised response technique (RRT)

Section K of the questionnaire consisted of yes/no questions regarding the lethal removal of predators and was conducted using the 'forced response' version of the RRT (Boruch, 1971; Lensvelt-Mulders *et al.*, 2005a), based on the methodology described by St John *et al.* (2010). Consequently, participation was restricted to farmers who were questioned in face-to-face interviews. Farmers were given written instructions explaining the technique and a copy of the questions in English or Afrikaans (Appendix 4). The interviewer then discussed the technique with the farmer to ensure they understood, before conducting a practice question. If the participant was thought to understand the technique the interviewer continued with the questions, if not, further explanation was provided. Farmers were instructed to roll two dice before answering each question and the sum of the dice determined how they should answer the question. If the sum of the dice was two, three or four farmers were instructed (not 'forced' as the name applies) to answer 'Yes', if the sum of the dice was 11 or 12 farmers were instructed to answer 'No'. If the sum of the dice was five to 10, farmers were instructed to answer truthfully, 'Yes' or 'No' (St John *et al.*, 2010). The interviewer did not see the dice roll and, therefore, was unaware if the answer was truthful or 'forced'. In some cases farmers insisted on answering the questions without the dice, in these cases the answers were analysed separately from those answered using the RRT. The Hox and Lensvelt-Mulders (2004) model was used to equate the proportion of people who had performed the questioned behaviour (e.g. killing a cheetah) from the RRT responses, using the equation:

$$\pi = \frac{\lambda - \vartheta}{\varsigma}$$

Where π = the proportion of people who have performed the behaviour, ς = the probability of answering truthfully, ϑ = the probability of a forced 'Yes', and λ = the number of 'Yes' responses. The probability of rolling a two, three or four with two dice is $^1/_6$ hence $\vartheta = {}^1/_6$; the probability of rolling an 11 or 12 is $^1/_{12}$ and the probability of rolling five to 10 is $^3/_4$ hence $\varsigma = {}^3/_4$ (St John *et al.*, 2010). Farmers were told to exclude predators legally trophy-hunted in their answers.

Due to the use of the RRT the behaviour of killing a predator could not be linked to a specific individual, therefore, potential links between lethal control and related variables could not be examined. The incidence of lethal removal was estimated for each predator species on farms where the species was reported to be resident (defined as a sighting [physical sighting or spoor] at least once every 12 months.

7.3.4 Statistical analyses

All data were coded for use in the statistical computer package SPSS version 11.0.1. Open-ended questions were coded based on content analysis to identify consistent themes within the answers. The Kolmogorov-Smirnov or Shapiro Wilk tests were used to determine if continuous variables were normally distributed. The statistical tests used included t-tests, Mann-Whitney U-tests, ANOVA, Kruskal-Wallis ANOVA (Turkey HSD for *post hoc* testing), Wilcoxon signed rank test and Spearman's correlation. Chi-squared tests using Yate's correction factor for tests with one degree of freedom were used to compare proportions. Means are presented with their standard deviation (SD). All tests were two-tailed. Sample sizes vary due to the experience farmers had or due to non-response.

7.4 Results and discussion

Data were collected from 44 ranchers who farmed only game-stock (based on 43 separate ranches), 44 ranchers who farmed both livestock and game-stock (based on 37 separate farms, referred to as L/G farms) and 27 livestock farmers (based on 27 separate livestock farms) (total: 115 farmers, 107 farms). In total, 61 farms had cattle, 43 farms had smallstock and 80 ranches had game-stock. As some participants represented multiple registered game ranches, at least one representative from 86.2% of the game ranches in operation was questioned (n = 109).

7.4.1 Limitations

The primary limitation of this study as described in Chapter 4 is in establishing the external validity of the answers provided. Rating the effectiveness of mitigation methods to enable predator-human coexistence would be best performed by demonstrating a human behaviour change in response to mitigation e.g. reduced incidence of lethal control of predators (Muir, 2012). However, such evidence can be difficult to obtain and, therefore, is rarely collected (Muir, 2012). The current study used farmers' perceptions of effectiveness. This is a subjective scale and the term 'effective' has been criticised as ambiguous (Websurveyor, 2003). However, often feelings are more predictive of support for management options than direct knowledge (Glikman *et al.*, 2011), therefore, the four-point scale offered an appropriate way to establish farmers' perceptions of conflict mitigation methods.

The long length of the questionnaire was also a potential limitation and in some cases may have reduced participants' willingness to expand on answers in open-ended questions, but the length of the questionnaire was felt to be necessary to gain an overview of the game ranching industry's role in predator conservation. The use of open-ended questions and content analysis could also have potentially resulted in observer bias in relation to the interpretation and categorising of comments. The use of a second researcher to independently conduct the content analysis would have reduced this potential source of bias and, therefore, should be conducted in the future (Fink & Kosecoff, 1996). The number of individuals who made each comment is stated in the relevant appendices, however, due to the open nature of the question it is possible the comments were relevant to additional respondents, but were not mentioned at the time of questioning (O'Cathain & Thomas, 2004).

The use of the RRT technique has been shown to increase the honesty of answers to sensitive questions, which would otherwise suffer from non-response bias or social desirability bias (Warner, 1965; Fisher, 1993), such as the lethal control of predators (St John *et al.*, 2012). However, some farmers insisted on answering questions without the use of the dice. The proportion of farmers who reported killing a specific predator species was greater (although non-significantly) for those farmers who used the RRT than those who did not. As a result, the reported incidence of removals may be an underestimate. Researchers should invest substantial effort in explaining the RRT technique, to encourage its utilisation. This study used one trial question before commencement of the test questions, but recommends the use of three of four trial questions to establish the farmers' level of understanding of the technique and to provide opportunities for further explanation.

7.4.2 Effectiveness of conflict mitigation methods at enabling predators and farmers to coexist on farmland

Introducing hunting of ‘problem’ predators was considered significantly more effective at enabling farmers and predators to coexist on farmland than all other mitigation methods, except quota-based predator trophy hunting (Table 7.1). ‘Problem’ predator hunting was considered effective or very effective [herein collectively referred to as ‘effective’] by 81.7% of farmers, compared to 78.3% of farmers who considered that quota-based predator hunting was effective (Table 7.1). Photographic tourism (66.7%), the translocation of ‘problem’ predators (61.9%) and the utilisation of livestock management techniques which aim to reduce carnivore depredation (60.2%) were considered the next most effective. The utilisation of game-stock management techniques which aim to reduce carnivore depredation (24.6%) were considered significantly less effective than all other mitigation methods except the eco-certification of meat products (48.5%), compensation for game-stock losses to carnivore depredation (32.8%) and insurance for game-stock or livestock losses (17.0%; Table 7.1). There were no significant difference in how farmers ranching livestock or game-stock rated the effectiveness of each method, with the exception of predator-friendly meat, which was rated significantly more effective by livestock farmers (61.5% of farmers thought it was effective, $n = 26$) than game ranchers (44.2%, $n = 77$; $U(103)$, $z = -2.74$, $p = 0.006$). Each conflict mitigation method and its potential applicability to HPC in Botswana are discussed in the following sections in the order perceived by farmers as the least to most effective at enabling farmers and predators to coexist (13^{th} – 1^{st}; Table 7.1) .

7.4.3 Insurance for livestock and game-stock losses – (Effectiveness rank: 13th)

Insurance of livestock or game-stock for losses incurred due to carnivore depredation was considered an effective solution to enable farmers and predators to coexist by 17.0% of interviewed farmers and was, therefore, perceived as the least effective solution (Table 7.1). Few farms insured their livestock (1.7%) or game-stock (1.3%) for accidental losses (Table 7.2). Farmers believed insurance was only appropriate for stud cattle, bulls, expensive game-stock species or animals in transport (commented by 25.7% of farmers, $n = 35$), that it was too expensive (34.3%), that it would be too difficult to prove losses (17.1%), that game-stock insurance was not available

Table 7.1. Botswana commercial farmers' perceptions of the effectiveness (very effective [VE], effective [E], ineffective [IE] or very ineffective [VIE]) of conflict mitigation methods at enabling predators and farmers to coexist on farmland. Significance testing was conducted with the Wilcoxon Signed Ranks Test.

Rating	Mitigation method	Proportion of farmers who rated method E or VE (n)	Rated significantly more effective than	z	n	Prob	Rated significantly more effective than	z	n	Prob
1st	Trophy hunting of 'problem' predators	81.7% (104)	Photographic tourism	−3.36	72	0.001	Eco-certification of meat	−4.68	96	< 0.001
			Translocation	−3.68	99	< 0.001	Compensation (livestock)	−5.88	76	< 0.001
			Livestock management	−2.56	76	0.010	Compensation (game-stock)	−5.29	61	< 0.001
			Eco-certification of tours	−4.44	74	< 0.001	Game-stock management	−5.21	59	< 0.001
			Conservancies	−3.66	70	< 0.001	Insurance	−6.51	88	< 0.001
			Reduce predator numbers	−5.17	100	< 0.001				
2nd	Trophy hunting of predators under quota	78.3% (106)	Photographic tourism	−2.67	74	0.008	Eco-certification of meat	−4.48	99	< 0.001
			Translocation	−3.22	101	0.001	Compensation (livestock)	−5.66	64	< 0.001
			Livestock management	−2.69	78	0.007	Compensation (game-stock)	−5.47	63	< 0.001
			Eco-certification of tours	−4.32	77	< 0.001	Game-stock management	−5.25	60	< 0.001
			Conservancies	−3.79	73	< 0.001	Insurance	−6.45	89	< 0.001
			Reduce predator numbers	−5.02	102	< 0.001				
3rd	Photographic tourism ł	66.7% (75)	Eco-certification of tours	−2.38	72	0.017	Compensation (game-stock)	−3.40	59	0.001
			Eco-certification of meat	−3.28	70	0.001	Game-stock management	−3.96	58	< 0.001
			Compensation (livestock)	−3.49	64	< 0.001	Insurance	−4.82	66	< 0.001
4th	Translocation of 'problem' predators	61.9% (105)	Compensation (livestock)	−3.87	93	< 0.001	Game-stock management	−3.33	59	0.001
			Compensation (game-stock)	−2.69	63	0.007	Insurance	−5.37	90	< 0.001
5th	Livestock management	60.2% (83)	Eco-certification of tours	−2.24	55	0.025	Compensation (game-stock)	−2.93	44	0.003
			Eco-certification of meat	−2.82	78	0.005	Game-stock management	−3.22	41	0.001
			Compensation (livestock)	−4.44	77	< 0.001	Insurance	−5.32	72	< 0.001
6th	Eco-certification of tours ł	56.4% (78)	Eco-certification of meat	−2.33	75	0.020	Game-stock management	−2.39	58	0.017
			Compensation (game-stock)	−2.15	60	0.032	Insurance	−3.78	73	< 0.001
7th	Conservancies ł	52.7% (74)	Eco-certification of meat	−2.20	70	0.028	Game-stock management	−3.42	55	0.001
			Compensation (livestock)	−2.29	62	0.022	Insurance	−3.81	67	< 0.001
			Compensation (game-stock)	−2.70	56	0.007				
8th	Reduce predator numbers	51.9% (106)	Compensation (livestock)	−2.32	92	0.021	Game-stock management	−2.37	60	0.018
			Compensation (game-stock)	−2.70	63	0.007	Insurance	−3.56	89	< 0.001
9th	Eco-certification of meat	48.5% (103)	Insurance	−3.51	88	< 0.001				
10th	Compensation (livestock)	32.7% (98)	Game-stock management	−2.12	51	0.034	Insurance	−2.41	82	0.035
11th	Compensation (game-stock) ł	32.8% (64)	Insurance	−2.24	56	0.025				
12th	Game-stock management ł	24.6% (61)	None							
13th	Insurance	17.0% (94)	None							

ł Only included ranchers who were currently farming game-stock or had farmed game-stock in Botswana in the past in Botswana (14.3%) or that it would only benefit the insurance companies (8.6%; Appendix 7.1). The Botswana insurance company offers insurance for livestock losses to all causes including carnivore depredation; however, the claims procedure and regulations are still in development (N. Mtunzie pers. comm.). Subsidised insurance initiatives in which part of the insurance premium is covered by donor funding or income from wildlife tourism or hunting activities have been trialled in communities with some success (Khan & Waseem, 2007; Kasaona, 2009; Rodricks, 2010). These initiatives have benefits over traditional insurance or compensation programmes as farmers have a vested interest in the legitimacy of claims and 'no claims' bonuses can encourage the improvement of livestock husbandry (Sillero-Zubiri & Stwizer, 2004). However, farmers are often concerned about the affordability of premiums and it is often difficult for these initiatives to become economically self-sustaining (Kasaona, 2009). Taking these limitations into consideration in conjunction with difficulties in establishing game-stock losses (Chapter 4) and the negative impression of insurance expressed by farmers, it is unlikely private insurance or subsidised insurance initiatives would be successful in reducing HPC on commercial game ranches.

7.4.4 Game-stock management techniques (Effectiveness rank: 12th)

The majority of farmers (75.4%) considered that altering game-stock management as a protection method against predators would be ineffective at enabling farmers and predators to coexist. Available methods included 'predator-proof' fencing of breeding camps or the perimeter of the farm, filling holes under the fence-line or diversionary feeding with buffer species. Some game-stock management techniques, such as electric fencing, were considered effective by some farmers (commented by 16.7% of farmers, n = 30; Appendix 7.2). However, overall, game-stock management techniques were considered significantly less effective than all techniques except the eco-certification of meat products, insurance for stock losses and compensation for game-stock losses (Table 7.1).

'Predator-proof' fencing of the farms perimeter

Farmers rarely used external 'predator-proof' fencing to prevent predators from entering the farm (bonnox 23.4%; electric 10.0%; underground fencing 16.4%) (Table 7.2), primarily because of the large set-up and maintenance costs (commented by 46.3% of farmers, n = 41; Appendix 7.3).

Farms which did use ‘predator-proof’ fencing were generally 1.5 times smaller (median area = 55km^2) than farms without ‘predator-proof’ fencing (area = 85km^2, U(78), z = −2.31, p = 0.021) and had a median of three human settlements within a 10 km radius of the farm compared to one settlement on farms without such fencing (U(79), z = −4.09, p = <0.001). The small size coupled with the close proximity to human settlements is likely to have reduced the installation and maintenance costs and potentially increased the risks of poaching, therefore, making fencing a more desirable and economical option. Additionally, it is likely that other non-financial reasons could influence decision making towards the use of ‘predator-proof’ fencing. For example, many farmers commented on the potential instability of the game ranching industry and listed a lack of government support as the primary problem affecting their farm (Chapter 4). Therefore, it is possible that farmers are reluctant to invest in farm infrastructure at this time. It has also been suggested that farmers could be reluctant to electrify the fence-line as it would exclude all predator species including leopards (McNutt, 2005) which, at the commencement of this study (2012), could be commercially trophy-hunted and, therefore, ranchers may wish to keep leopards on their farms.

Farmers were also concerned that predators would still gain access to the farm and that carnivore depredation on game-stock would not reduce (commented by 22.0% of farmers, n = 41; Appendix 7.3). However, farms located within a HPC ‘hot-spot’ (defined as within 100 km of a protected area; Chapter 5), that used electric or bonnox fencing, reported losing 6.8 times less game-stock to carnivore depredation (median: 12.5, n = 4) than farms without electric or bonnox fencing (median: 85.5, n = 20; U(24), z = −2.05, p = 0.040). Previous studies have also shown that electric fencing reduced sheep losses to coyotes in America (Linhart *et al.*, 1982) and excluded large predators and reduced holes dug by black-backed jackals in South Africa (du Toit, 1996; Van Rooyen *et al.*, 1996). However, although ‘predator proofing’ the perimeter fence is likely to reduce HPC, its use on a wide scale is undesirable as fences, especially electric fences, effectively split the land into small fragmented pockets, reduce the available habitat for wildlife by preventing access to key resources and disrupt the immigration and emigration of wildlife species (Cousins *et al.*, 2008; Lindsey *et al.*, 2012d; Woodroffe *et al.*, 2014).

Breeding camps

A potential alternative to 'predator-proofing' the perimeter of the farm is the use of small electric fenced breeding camps to protect specific game-stock animals. Overall, 16.3% of game ranches used breeding camps (Table 7.2). Being smaller, the installation and maintenance costs of breeding camps are lower than for perimeter-fencing. Additionally, predators can maintain access to the rest of the farm to aid their conservation and could, therefore, be utilised for photographic tourism and hunting. Generally, farmers thought that breeding camps were only necessary for expensive game-stock species (commented by 14.8% of farmers, n = 54 comments). Forty-seven percent (47.4%, n = 19) of game ranches with expensive ungulate species were using breeding camps, compared to 6.6% (n = 61) of farms without these species ($\chi^2 = 12.44$, df = 1, $p < 0.001$). Breeding camps that were not used for expensive animals were used to protect and maintain

Table 7.2. Comparison between the number of farmers using livestock and game-stock management techniques on Botswana farms using the Chi-squared test. Significant results are marked with '*'

Management Technique	**Proportion of farms using technique**				
	Cattle (n)	**Smallstock (n)**	**χ^2**	**df**	**Prob**
Kraal livestock at night	23.0% (61)	100.0% (39)	25.21	1	< 0.001*
Kraal livestock when calving/lambing	53.4% (58)	92.5% (40)	4.66	1	0.031*
Kraal young calves/smallstock	67.8% (59) ł	92.9% (42)	1.66	1	0.198
Livestock guarding dog	8.3% (60)	46.5% (43)	13.51	1	< 0.001*
Herders	8.5% (59)	42.9% (42)	11.57	1	< 0.001*
Breeding seasons	48.3% (60)	31.0% (42)	1.42	1	0.233
Promote wildlife populations	83.3% (24) §	-	-	-	-
	Livestock (n)	**Game-stock (n)**			
Keep records of stock numbers	85.5% (55)	72.7% (77)	0.52	1	0.471
Insure against losses from predators	1.7% (59)	1.3% (78)	ǂ	-	-
Fenced breeding camps	-	16.3% (80)	-	-	-
Electrified perimeter fence	-	10.0% (80)	-	-	-
Bonnox fencing of perimeter	-	23.4% (77)	-	-	-
Perimeter fence buried underground	-	16.4% (73)	-	-	-
Patrol perimeter fence at least weekly	-	84.2% (76)	-		-
Fill in holes under perimeter fence	-	67.1% (79)	-	-	-
Swing gates in holes under fence	-	2.6% (78)	-	-	-
Stock a buffer-prey species	-	43.4% (76)	-	-	-

ł Calves kraaled for a median of 3.6 months (SD = 1.8 months, n = 21)

ǂ sample size too small to test significance

§ Farms ranching only livestock

breeding herds of species which were particularly vulnerable to carnivore depredation, such as springbok (Chapter 6), the adults of which were released onto the farm. Tolerance scores to predators were negatively correlated with the number of springbok owned (Chapter 4); therefore, extending the use of breeding camps to vulnerable species could increase farmers' tolerance of predators.

Patrolling and filling holes in the perimeter fence-line

An alternative and cheaper method of preventing predators from gaining entry to the farm is to fill the holes under the game-fence. Farmers from the majority of game ranches reported that they patrolled their perimeter fence-lines at least weekly (84.2%) and farmers on 67.1% of ranches filled the holes under the fence-line (Table 7.2). However, farmers' primary motivation for filling holes was often to prevent game-stock species from leaving the farm, therefore, smaller holes, which could still allow predators access to the farm, were often left unfilled (pers. obs.). Farmers stated that filled holes were frequently reopened or new holes appeared (commented by 36.4% of farmers, n = 22); therefore, predators were still able to gain access to the farm (18.2%; Appendix 7.4). The majority of Botswana has sandveld soils (70%) (Central Statistics Office, 2008); consequently, digging species, primarily warthog, can quickly reopen holes in the soft sand (Schumann *et al.*, 2006). One farmer had tried placing water points outside his fence-line; because warthogs' movements are often water dependent (Skinner & Smithers, 1990) it was hoped this would reduce the likelihood of warthogs entering the farm. Other farmers persecuted warthog in an attempt to reduce their abundance (pers. obs.) and such persecution of digging species has also been reported in Namibia (Schumann *et al.*, 2006). In a study in the Khutse game reserve and Makgadikgadi Pans National Park in Botswana, predators as large as lions were able to squeeze through relatively small holes (500 – 725 cm^2), and it was recommended that fence maintenance should occur daily (Kesch *et al.*, 2014); a significant investment of time and resources by landowners.

An alternative to reduce the necessary fence maintenance is to place swing gates in the fence-line. Digging species quickly learn to use the gates reducing the number of new holes dug under the fence-line. Swing gates cost 18 times less per kilometre than electric fencing to install and

2.0 times less per kilometre, per year, to maintain (Schumann *et al.*, 2006). However, few farmers were using swing gates (2.6%), primarily due to a lack of knowledge regarding their installation (commented by 46.2% of farmers, n = 13; Appendix 7.4). Therefore, educational materials regarding their use should be distributed.

Diversionary feeding: Buffer species

Farmers on 43.4% of game ranches reported stocking buffer species (most commonly impala and kudu [81.8%, n = 22]) in an attempt to buffer the extent of carnivore depredation on more expensive game-stock animals (Table 7.2). Predators generally consume prey species in relation to their relative abundance (Hayward *et al.*, 2006), for example in the Ghanzi farming block, cheetahs commonly preyed on kudu, which was the most abundant antelope species (Chapter 6). However, cheetahs preyed on springbok more than predicted by their relative abundance (Chapter 6) and lions continued to predate on eland, red hartebeest and gemsbok despite their low relative abundance on a game ranch in South Africa (Lehmann *et al.*, 2008). Due to this preference Lehmann *et al.* (2008) concluded the utility of buffer species may be minimal. It is also possible that if the overall prey biomass increases, it could result in an increase in predator density and, therefore, potentially increased losses (Reynolds & Tapper, 1996; Hayward *et al.*, 2007b). Few studies have empirically tested this diversionary feeding theory (Graham *et al.*, 2004), but the stocking of buffer species is recommended in the Zambian game ranching handbook (Zieger & Cauldwell, 1998), and the provision of carrion or alternative sources of prey has been associated with reduced predation on expensive game-stock (Power, 2002), moose (Gasaway *et al.*, 1992) and game birds (Redpath *et al.*, 2001).

Buffer species made up a median of 26.5% (n = 21) of the game-stock on game ranches where farmers perceived predators to primarily prey on livestock or game-stock (excluding bufffer species), compared to a median of 57.4% (n = 46) on ranches where farmers perceived free-ranging wildlife or buffer species as the primary prey consumed ($U(67)$, $z = -3.11$, $p = 0.002$). Therefore, although data were not available to determine if the stocking of buffer species does reduce predation on more expensive game-stock, farmers perceived this to be the case. Tolerance scores to predators were positively correlated with the proportion of buffer species stocked on

Table 7.3. Association between farmers' tolerance scores ł to predators and their use of methods to mitigate conflict between humans and predators. The Mann-Whitney U-test and the Kruskal-Wallis ANOVA test were used to determine significance; significant results are marked with '*'

Conflict mitigation method	Median tolerance score (n) Use	Do not use	Statistic	df/n	Prob
Trophy hunted a leopard ǂ					
Tolerance of predators	Once: 3.25 (21) Multiple: 3.81 (13)	3.13 (33)	$\chi^2 = 1.00$	2	0.607
Tolerance of leopard	Once:4.00 (19) Multiple: 3.50 (13)	3.00 (30)	$\chi^2 = 0.60$	2	0.740
Member of conservancy or wish to join a conservancy in future	4.56 (35)	3.20 (49)	z = −3.12	84	0.002*
Conduct photographic tourism	4.17 (41)	3.80 (44)	z = −0.59	85	0.557
Earn > 50% of farm's income from photographic tourism	5.00 (14)	3.75 (71)	z = −2.27	85	0.023*
Claim compensation	2.97 (36)	3.25 (29)	z = −1.74	65	0.081
Stock a buffer species	4.00 (36)	3.23 (44)	z = −2.06	80	0.040*
'Predator-proof' fencing	4.00 (30)	3.75 (54)	z = −0.01	84	0.990
Breeding camps	4.00 (15)	3.75 (69)	z = −0.84	84	0.401
Kraal cattle	3.00 (49)	3.19 (17)	z = −0.73	66	0.468
Guard livestock (herd or LSGD)	3.00 (35)	3.31 (32)	z = −1.16	67	0.245

ł Tolerance scores ranged from zero to five and were scored according to when a farmer would remove a predator from their farm in relation to the number of livestock or game-stock losses the predator had caused. A higher score indicated a greater tolerance of predators

ǂ Only included game ranchers who were pro-leopard hunting and who had or had not hunted a leopard in the five years prior to the survey

the game ranch ($R_S = 0.37$, n = 73, p = 0.001; Chapter 4), and farmers that reported they stocked buffer prey animals had greater tolerance scores to predators (TS = 4.0) than farmers without these animals (TS = 3.2; U(80), z = −2.06, p = 0.040; Table 7.3). However, some farmers stated that buying buffer species was too expensive (commented by 36.4% of farmers, n = 11) or that they were unwilling to accept predators killing any game-stock species (18.2%; Appendix 7.4). In these cases, although excluding predators from habitat is not ideal, preventing the predators' entry to the farm is likely to be the preferable option (Schumann *et al.*, 2006).

A cost-benefit analysis and testing of the various game-stock management techniques is needed. However, preliminary information from this study showed that game-stock management, primarily 'predator-proof' fencing, breeding camps for vulnerable species and the stocking of buffer species were associated with reduced game-stock losses to carnivore depredation and were linked to increased human tolerance of predators and, therefore, could be more successful at promoting predator-human coexistence than farmers perceived. Farmers often commented that in terms of game-stock management there was nothing you could do to stop predators (commented by 37.5% of farmers, n = 24; Appendix 7.2). This perception of a lack of control can increase the level of risk a threat is perceived to cause (McKenna, 1993) and can increase the intensity of HPC (Dickman, 2008). Changing this perception and restoring farmers' feeling of control over HPC situations is likely to increase farmers' tolerance (Dickman, 2008).

7.4.5 Compensation for livestock and game-stock losses to predators (Effectiveness rank: 10th/11th)

Botswana is the only country in southern Africa that is still using a government-run compensation programme and annually spends BWP 4.9 million (ca. US $640,500) on compensation pay-outs, plus an additional BWP 30 million (ca. US $3,920,000) on administration costs (Bowie, 2009; CAR, 2011). Despite this large financial outlay, farmers rated compensation as significantly less effective at enabling predator-human coexistence than all other suggested conflict mitigation methods, except insurance and game-stock management; 32.7% of farmers thought that compensation for livestock losses was effective and 32.8% of farmers thought compensation for game-stock losses could be effective (Table 7.1). Farmers stated that the payment was too low (commented by 66.0% of farmers, n = 47), that it was too difficult (10.6%) or time-consuming (8.5%) to find carcasses or report claims, that reimbursements were not dealt with efficiently (14.9%) or that compensation was inapplicable to their farm (e.g. dairy farms are not included; 2.1%) or did not cover all predator species (2.1%; Appendix 7.5). When conflict management is viewed to be handled improperly it can reduce the credibility of the organisations involved and detract from the long-term conservation goal of reducing HPC (Hewitt & Messmer, 1997). Users of compensation (55.0% of interviewed farmers ranching livestock, n = 60) had lower tolerance scores to predators (TS = 3.0) than non-users (TS = 3.3, appr. sig.; Table 7.3). Other studies have also concluded that compensation programmes including the Botswana one do not increase tolerance or reduce HWC and are unlikely to provide substantial benefits in terms of poverty alleviation or long-term conservation (Gusset *et al.*, 2009; Dickman *et al.*, 2011; CAR, 2012).

The Botswana compensation programme has previously been reviewed with suggestions that there should be penalties for false claims or for killing predators, a less restricted and bureaucratic reporting system and decentralisation of control of the programme to the community level. Additionally, it has been suggested all predator species should be compensated for at a higher rate and that compensation should only be given if farmers have adhered to a strict minimum standard of livestock husbandry (Okavango Delta Management Plan, 2006; Gusset *et al.*, 2009; CAR, 2011; Kent, 2011). In principle farmers must be using livestock management techniques, such as kraaling, to claim compensation and DWNP officers should give advice on livestock management, however, often this does not occur (Okavango Delta Management Plan, 2006). Compensation programmes in general have been associated with increased stocking rates and reduced incentives for farmers to take precautionary measures and improve their agricultural management practices (Nyphus *et al.*, 2005; CAR, 2011; Dickman *et al.*, 2011). In Sweden, compensation at or above market value of stock with requirements for strict adherence to livestock husbandry guidelines was shown to increase tolerance and reduce the number of livestock being killed by predators (Swenson & Andren, 2005). However, livestock management did not improve on farms in southern Kenya, despite making it a requirement to receive compensation, and farmers may lack the knowledge, income or materials to improve husbandry (Maclennan *et al.*, 2009). Additionally, compensation does not address the attitudinal and cultural aspects of HPC and is unlikely to be effective when HPC is driven by factors other than economics (Montag, 2003). Despite these failings the BWPA submitted a proposal in 2004 for game-stock lost to carnivore depredation to qualify for compensation (BWPA, 2004), but the proposal was rejected (BWPA, 2006a). However, in light of the 2013 restrictions on leopard hunting the BWPA has re-submitted its request (BWPA, 2012c). Many game ranchers feel game-stock should be treated like cattle in terms of its protection from predators (BWPA, 2012c), but farmers in this study recognised it would be difficult to verify claims (commented by 30.8% of farmers, n = 26) as an estimated 50% of game-stock carcasses are never located (Chapter 4).

In light of the general dissatisfaction with the livestock compensation programme, it is unlikely that compensation for game-stock losses would increase tolerance of predators. The money spent on compensation programmes could be more effectively spent on subsidising improvements in livestock and game-stock husbandry or to provide performance payments for tolerance of predators (Bulte & Rondeau, 2005; Nyphus *et al.*, 2005; Nelson, 2009). For example, in Sweden, farmers are paid for every grey wolf or wolverine *Gulo gulo* den present on their property (Zabel & Holm-Müller, 2008) or in Mexico ranchers are paid if camera-traps record a jaguar *Panthera onca*, puma or ocelot *Leopardus pardalis* on their land (Nelson, 2009). A similar approach could be adopted for camera-trap pictures of African carnivores or the presence of African wild dog dens or cheetah marking-trees. These conservation payments create a direct incentive to maintain carnivores and there is increasing evidence that they can be a valuable tool for promoting human-predator coexistance (Zabel & Holm-Müller, 2008; Dickman *et al.*, 2011).

7.4.6 Eco-certification of meat products (Effectiveness rank: 9th)

The eco-certification and marketing of predator-friendly meat was not available in Botswana at the time of study (2012/2013), but was considered as a potentially effective means of enabling predator-human coexistence by 61.5% of farmers interviewed who owned livestock and by 44.2% of farmers ranching only game-stock (Table 7.1). Farmers' primary concerns for the success of predator-friendly meat were the lack of a consumer market (commented by 34.1% of farmers, n = 41), policing of the programme (12.2%) and whether the extra premium would cover the costs of living with predators (2.4%; Appendix 7.6). Predator-friendly meat, sourced from certified predator-friendly farms, is available in the USA and initiatives are underway in both South Africa and Namibia (Badgley, 2003; Tjaronda, 2006; Conservation International, 2014). In Italy consumers were willing to pay more for organic beef than its commercial value (Napolitano *et al.*, 2009), and in Georgia, USA, 66% of consumers expressed a willingness to pay above the normal premium for environmentally friendly beef (Wong, 2009); therefore, a potential export market is available. However, producers in Montana, USA were not gaining a higher price for their certified meat products (Early, 2012), and Treves and Jones (2010) concluded that proving producers are conserving wildlife can be time-consuming, costly and technically challenging. Therefore, farmers' concerns about the feasibility of predator-friendly meat could be valid and further research on existing initiatives is required.

Botswana has a long history of exporting beef to the European Union (Darkoh & Mbaiwa, 2002), where the market for eco-certified products is likely to be greater than the local market. In contrast, game meat production and export is often hindered by the costs of harvesting, processing, stringent hygiene and veterinary control (Bond *et al.*, 2004), and by poor, often incorrect, commercial labelling of the game-stock species from which the meat is sourced (D'Amato *et al.*, 2013). Additionally, farmers who are reliant on low income, high off-take wildlife uses, such as meat production, tend to have a lower tolerance of predators (Lindsey *et al.*, 2005c; Lindsey *et al.*, 2009c). As a result, the introduction of eco-certification of game-stock meat is unlikely to be practical or productive, in terms of predator-human coexistence in Botswana, at the current time (2013).

7.4.7 Lethal control of predators (Effectiveness rank: 8th)

Fifty-two percent (51.9%) of farmers considered reducing predator populations to a more 'tolerable' level as an effective way to enable them to coexist with the remaining predators (Table 7.1). Some farmers believed that predator populations were out of balance and it was necessary to reduce numbers in order to restore this balance (commented by 22.9% of farmers, n = 48). However, opponents argued that nature controls predator numbers (12.5%) and an individual farmer's measure of what is 'tolerable' or in balance can vary and is likely to be too few (20.8%; Appendix 7.7). Similar opinions were expressed by member of the public in relation to the culling of deer in the UK and beliefs about overabundance of wildlife can be powerful drivers of support for wildlife management through lethal control (Dandy *et al.*, 2012).

Questions regarding the lethal control of predators were answered by representatives from 91 farms; an additional two farmers refused to take part. The RRT was used by 67 farmers while 24 farmers insisted on answering questions without the use of the dice. The proportion of farmers who reported killing a specific predator species was higher for those farmers who used the RRT than those who did not; however, this difference was not statistically significant (p values = 0.39 − 1.00).

Overall, farmers reported that in the 12 months prior to the survey they had killed lions on an estimated 87.0% of farms on which they were resident. The next highest persecuted species was black-backed jackals, killed on 52.5% of farms with resident populations, followed by African wild dogs (34.0%) and spotted hyaenas (25.2%; Table 7.4). Such a high incidence of removals correlates with African wild dogs (TS = 1.7), lions (TS = 2.0) and spotted hyaenas (TS = 2.0) being the least tolerated of the predator species (Chapter 4). Farmers reported relatively higher tolerance to black-backed jackals (TS = 2.6), but the removal of jackals was generally considered as routine and necessary (Kent, 2011) and they were the most frequently killed predator species on farms in South Africa (Thorn *et al.*, 2012). Despite this high incidence of lethal control, black-backed jackals are considered one of the least threatened carnivore species in Africa (Ray *et al.*, 2005). Lethal control is potentially counter-productive, having been linked to compensatory breeding, reduced emigration and decreased natural mortality, which can result in actual or perceived population growth (Prugh *et al.*, 2009).

Table 7.4. Proportion of interviewed commercial farmers who reported they had killed predators on their farm. Each farmer represented a separate ranch

Species killed	**Proportion of farms species resident on (n) ǂ**	**Proportion of farmers who had killed predator species (n)**	**Proportion of farms with resident predator populations, upon which the predator species was killed**
African wild dog ƚ	38.5% (91)	13.1% (90)	34.0%
Brown hyaena ƚ	90.0% (90)	17.0% (90)	18.9%
Caracal ƚ	78.7% (89)	12.0% (90)	15.2%
Cheetah ƚ	70.3% (91)	14.0% (90)	19.9%
Black-backed jackal ƚ	97.8% (90)	51.3% (91)	52.5%
Lion ƚ	25.3% (91)	22.0% (90)	87.0%
Leopard ƚ	94.4% (90)	20.2% (90)	21.4%
Spotted hyaena ƚ	47.3% (91)	11.9% (90)	25.2%
Protected predator § without reporting it		18.7% (89)	-
Used poison to kill a predator		21.5% (90)	-

ƚ Within the 12 months prior to study

ǂ Based on farmers' reports of a sighting of the predator (physical sighting or spoor) at least once every 12 months

§ African wild dog, brown hyaena, cheetah, leopard or lion

In contrast, the lethal removal of large carnivores can affect the long-term viability of predator populations and high mortality rates in farms near protected areas can weaken the ability of such areas to conserve predator populations (Woodroffe & Ginsberg, 1998; Loveridge *et al.*, 2007; Balme *et al.*, 2010). For example, lion removals surrounding the KTP, in Botswana have resulted in a decline in population and the doubling of the female to male ratio, which could affect the populations long-term viability (Castley *et al.*, 2002). Additionally, increased intraspecific aggression and infanticide associated with territorial disputes, after the removal of predators, can impact on predator populations beyond the removed individual (Treves & Karanth, 2003; Balme *et al.*, 2009b). Lethal removal of large predators has also been associated with increased conflict with meso-predators (e.g., jackals and caracals), immigration of younger, non-territorial predators (which could be more likely to be 'problem' animals) and in many incidences has failed to reduce stock losses (Marker *et al.*, 2003d; Loveridge *et al.*, 2007; Bailey & Conradie, 2013).

In Botswana, farmers can legally kill a predator to protect their livestock (but not game-stock) but must report the incident (and return the skin) to the DWNP within seven days (DWNP, 1992). However, many individuals do not report predators they have killed, possibly due to the fear of retribution or due to the inconvenience of reporting (Okavango Delta Management Plan, 2006). In Namibia the number of farmers reporting killing predators in questionnaire surveys was much greater than official records (Stein *et al.*, 2010). Some farmers referred to the term 'SSS' - 'shoot, shovel and shut up' (Treves, 2009a; farming community member pers. comm.) and 18.7% of farmers questioned in the current study stated that they had illegally killed a protected predator species without reporting it to the DWNP (Table 7.4).

Shooting is the only legal method for lethal control in Botswana (DWNP, 1992) and the discriminate removal of specific 'problem' predators can enable farmers to tolerate those that remain (Linnell, 2011). However, some farmers also referred to using indiscriminate methods such as gin-traps, cage-traps, snaring and poison bait (used by 21.5% of farmers; Table 7.4). Frequently, non-target individuals or other mammalian scavengers, birds of prey and vultures can be killed with these methods. For example, Mateo-Tomás *et al.* (2012) found in a review of poisoning events, primarily targeted at wolves in Spain, that 52.6% of the animals killed were birds of prey and 36.4% were non-target mammalian scavengers. As a result of the indiscriminate nature of poison bait, poisoning is considered one of the major threats to vulture populations worldwide (Ogada *et al.*, 2012).

Hachileka (2003) suggested farmers in Botswana should be required to seek permission to kill 'problem' animals, as is the case in South Africa (Cousins *et al.*, 2010). However, legislation banning lethal control does not address the social and psychological reasons for its use (Marchini & Macdonald, 2012), and prohibiting the killing of wildlife can be impossible to enforce without methods to change the motivations behind these behaviours (Treves *et al.*, 2009). Stricter regulation of lethal control has been associated with illegal hunting and can intensify social and political conflict with wildlife organisations and government (Woodroffe *et al.*, 2005b; Linnell *et al.*, 2010). Therefore, legislation alone is unlikely to be effective.

7.4.8 Conservancies (Effectiveness rank: 7th)

Conservancies defined as a 'group of adjoining private commercial farms operating under a cooperative management agreement' (ABSA, 2003) are generally considered one of the best approaches to encourage tolerance of predators and ecological friendly practices on game ranches and livestock farms (Schumann *et al.*, 2008; Lindsey *et al.*, 2009b; Brink *et al.*, 2011). Conservancies in Botswana were rare; 5.0% (n = 80) of game ranches and none of the livestock farms operated as part of a conservancy (Chapter 4). However, 52.7% of farmers thought that conservancies could be effective at enabling predator-human coexistence (Table 7.1) and 36.2% of farmers wanted to join a conservancy in the future.

Farmers operating as part of a conservancy or those who wished to join a conservancy in the future had greater tolerance scores to predators (TS = 4.6) than farmers who did not want to join a conservancy (TS = 3.2; $U(84)$, $z = -3.12$, $p = 0.002$; Table 7.3). Similarly, in South Africa and Namibia members of conservancies had more positive attitudes to predators than non-conservancy members (Schumann *et al.*, 2008; Swanepoel, 2008; Lindsey *et al.*, 2009b). In general conservancies are more likely to use livestock management techniques (Schumann *et al.*, 2008), require less management of game-stock, suffer less impact from predation (Romañach & Lindsey,

2008) and due to economies of scale incur less costs (Lindsey *et al.*, 2009b). Conservancies are more politically land stable, reduce the effect of habitat fragmentation and are more likely to introduce low off-take, high income game-stock uses such as photographic tourism and trophy hunting (Lindsey *et al.*, 2009c). Therefore, they are thought to be more efficient financially to the individual and economically to the national economy than other land uses (Barnes & De Jager, 1996).

However, the majority of farmers did not wish to join a conservancy (58.5%). Some farmers believed conservancies were not a legally recognised land use (commented by 5.0% of farmers, n = 40), they were concerned about the legal status and ownership of game-stock within a conservancy (10.0%) and believed there was more money to be made farming separately (2.5%; Appendix 7.8). Increased dissemination of information about the legalities and benefits of conservancies, about the problems affecting single game ranches and examples of successful models for conservancy constitutions could help to promote their adoption (Lindsey *et al.*, 2009b). The introduction of economic incentives to encourage the formation of conservancies such as tax reductions, discounts on state bought wildlife, performance payments for predator species present (e.g. for African wild dog dens) or custody over threatened species, such as black rhinoceros *Diceros bicornis* have either been proposed or are utilised in some areas of Namibia and South Africa (Langholz *et al.*, 2000; Nyphus *et al.*, 2005; Spenceley & Barnes, 2005; Schumann *et al.*, 2008; Zabel & Holm-Müller, 2008; Cousins *et al.*, 2010). It is possible that their application in Botswana could increase conservancy membership.

However, the primary reasons farmers gave for their reluctance to join conservancies were the lack of suitable neighbours (85.0%, n = 40) and a preference for working alone (12.5%). Lindsey *et al.* (2009b) described ranchers as 'fiercely independent' and McGranahan (2008) thought that fences were essential as people were not ready to share land or stock. Changing these perceptions could be the hardest obstacle to overcome.

7.4.9 Eco-certification of hunting and photographic tourism operations (Effectiveness rank: 6th)

Eco-certification of hunting or photographic tourism was considered effective at enabling predator-human coexistence by 56.4% of farmers, and was considered significantly more effective than the eco-certification of meat products (48.5% of farmers, $U(75)$, $z = -2.33$, $p = 0.020$). However, these initiatives were rarely used in Botswana; two farms were members of Fair Trade and one farm was part of the Botswana Ecotourism Certification System, the national accreditation system introduced in 2009. This system awards tourism outlets with a 'green', 'green+' or 'ecotourism' status based on their environmental impact, focusing on aspects such as clean energy, solid waste management and waste water (Botswana Tourism Board, 2008). However, the scheme is not applicable to tourism operations that involve the consumptive use of game-stock (Botswana Tourism Board officer pers. comm.).

Farmers commented that tourists would always pick the cheapest option (commented by 4.3% of farmers, $n = 23$), or that the premium would not cover the losses (13.0%; Appendix 7.6). However, 88% of travellers interviewed from 10 countries ($n = 5000$) reported they would choose an environmentally friendly tourism option if it was available (VISA, 2007), and 25% of travellers would pay up to 5 − 10% more to stay in eco-friendly accommodation (TripAdvisor, 2010). One of the primary limitations to greener tourism is a lack of information from tourism operators advertising eco-friendly initiatives (Baker *et al.*, 2014). Therefore, allowing all game ranches to join the Botswana Ecotourism Certification system, or encouraging them to join international initiatives like the 'Baobab Certification Program', would enable tourists to distinguish between game ranches and make an informed choice. The increased business from eco-minded tourists and the introduction of a predator-friendly clause within the ecotourism standards, could encourage the adoption of both predator and environmentally friendly practices.

The introduction of an independent regulatory body and accreditation initiative for hunting operators would also be potentially beneficial (Lewis & Alpert, 1997). Lindsey *et al.* (2006) demonstrated that hunters were more averse to hunting in areas with poor conservation objectives than hunting operators thought. Thirty-two percent of hunters were happy to shoot on farms where cheetahs were being killed as 'problem' animals and 20% of hunters were happy to shoot on farms where African wild dogs were being killed. The most common reason farmers stated as to why the eco-certification of trophy hunting would be ineffective at promoting coexistence was that hunters 'do not care' and people would come to the farm regardless of their eco-rating (commented by 30.4% of farmers, n = 23; Appendix 7.6). The results presented by Lindsey *et al.* (2006) suggest that this is not the case, with hunters and their non-hunting family members voicing concern about the conservation principles and social benefits of their hunt. If hunters were able to distinguish between operators, their preferences could move the industry to be more environmentally and predator-friendly (Lindsey *et al.*, 2006). With the increasing public negativity towards hunting (Peterson, 2004) adopting an international accreditation and regulatory body on Botswana's game ranches to promote hunting's role in conservation would potentially be beneficial.

7.4.10 Livestock management techniques (Effectiveness rank: 5th)

Smallstock were kraaled at night on 100% of farms and 92.9% of farms kraaled their small lambs or kids, compared to 23.0% of farms kraaled cattle at night and 67.8% of farms kraaled their young calves (Table 7.2). Smallstock are considerably smaller than cattle and range over shorter distances, therefore, they are often perceived as easier to manage than large cattle herds, and overall, smallstock were more intensively protected from carnivore depredation than cattle (Table 7.2). Farmers stated that they were reluctant to kraal calves because it increased the risk of disease (commented by 38.9% of farmers, n = 18), and kraaling of cattle has been associated with additional costs to provide extra health care (Andelt, 1996). Farmers also stated that they preferred not to kraal adult cattle because cattle fed better at night when it was cool than during the day (37.8%, n = 37) and that kraaling reduced the cattle's grazing potential and herd productivity (18.9%, n = 37; Appendix 7.9). However, on farms where high conflict species were present, such as spotted hyaenas, farmers reported the risks of carnivore depredation forced them to kraal cattle. Farmers were 5.6 times more likely to kraal their cattle herd at night (44.0% of farms, n = 25), and 1.6 times more likely to kraal calves during the day/night (88.0%) if spotted hyaenas were present than if they were absent (night: 7.9%, n = 38; $\chi^2 = 9.38$, df = 1, p = 0.002; calves: 55.6%, n = 36; $\chi^2 = 5.81$ df = 1, p = 0.016). Similarly, farms were 3.4 times more likely to kraal cattle at night, if lions and/or African wild dogs were present (40.9%, n = 22) than if they were absent (12.2%, n = 41; $\chi^2 = 5.27$, df = 1, $p = 0.022$). Therefore, farmers' management decisions are often based on their perceptions of the risks and costs imposed by different farming techniques in conjunction with the risks of carnivore depredation.

However, some farmers did not kraal cattle as they thought that it would increase carnivore depredation (commented by 44.4% of farmers, n = 18), believing that it was easier for predators to kill calves inside the kraal (Appendix 7.9). Fears of increased predation are most likely related to incidences of surplus killing when predators have gained access to the kraal, and the fleeing movements of the livestock repeatedly triggers the carnivores killing instinct (Nowell & Jackson, 1996; Ogada *et al.*, 2003). For example, a single caracal killed 21 young goats in one event (Stuart, 1981) and two lions were reported to kill 43 goats in one night (Hemson, 2003). The majority of cattle kraals on commercial farms are made of poles and wire, which allow predators access (pers. obs.). If kraals are reinforced with branches of *Acacia* tree species or a similar deterrent, which prevents predators gaining access to the kraal or livestock breaking out of the kraal, predation is reduced (Butler, 2000; Ogada *et al.*, 2003; Gusset *et al.*, 2009). In the current study, farms within 'hot-spots' of HPC (within 100 km of a protected area; Chapter 5) which kraaled their calves until they were at least three months old (the period during which calves are most vulnerable [Swanepoel, 2008]) reported losing 3.8 times less cattle to carnivore depredation (median = 4, n = 7) than farms which did not kraal calves (median = 15, n = 17; U(24), $z = -2.23$, $p = 0.026$).

When grazing, smallstock herds were also more commonly protected from carnivore depredation than cattle. Smallstock were accompanied by a LSGD on 46.5% of farms and by a herder on 42.9% of farms compared to cattle on 8.3% and 8.5% of farms (Table 7.2). Farmers' reasons for not using herders (57.1%, n = 14) or LSGDs (23.8%, n = 21) were largely due to limitations in the farm infrastructure, management and staffing (Appendix 7.9 – 7.11). Farmers thought that their employees were too unreliable to guard livestock (7.1%, n = 14), the farms were too large for herding to be effective (14.3%, n = 14) and they were concerned that staff would use the dogs to hunt with (9.5%, n = 21). Disciplinary problems with LSGDs were also mentioned, with farmers reporting problems with the dogs staying with the herd, or playing with and injuring livestock (50.0%, n = 8; Appendix 7.10). Although such disciplinary problems are commonly reported in juvenile dogs, with proper training adult LSGDs should not disturb the herd or local wildlife (Landry, 2001). Educational materials instructing farmers on appropriate training for LSGDs, including the benefits of using LSGDs in conjunction with herding (Breitenmoser *et al.*, 2005), would be beneficial, particularly in the case of cattle farmers, who stated they were unsure about how to use LSGDs with cattle herds (14.3%, n = 21; Appendix 7.10 and 7.11). As a result, some farmers considered LSGDs (20.7%, n = 29) and herders (42.9%, n = 21) to be ineffective at reducing predation (Appendix 7.10 and 7.11). However, the utilisation of LSGDs and herders reduced losses to predators in Namibia and Kenya (Ogada *et al.*, 2003; Marker *et al.*, 2005; Woodroffe *et al.*, 2007; Stein *et al.*, 2010), and the use of LSGDs resulted in financial savings of ca. US $3000 per farm in South Africa (Rust *et al.*, 2013). Other guarding animals such as donkeys, lamas and baboons have also been reported to successfully deter predators (Breitenmoser *et al.*, 2005).

Another technique used by the majority of livestock farmers (83.3%; Table 7.2) was the protection of wildlife populations on their farm to serve as an alternative prey source for predators. Large carnivores including cheetahs in this study (Chapter 6) will preferentially kill wild prey, despite a larger abundance of livestock (Marker *et al.*, 2003e; Woodroffe *et al.*, 2005a). Some farms provided water or salt licks specifically for wildlife (commented by 42.9% of farmers, n = 14). Conversely, farmers not promoting wildlife said it was not necessary as wildlife species were naturally there (7.1%), that some ungulate species carry diseases which can be passed on to cattle (14.2%) or that wildlife can damage fences or eat too much grazing (7.1%).

Historically, herding, kraaling and LSGDs were routinely used by farmers to reduce carnivore depredation on livestock (Breitenmoser *et al.*, 2005). The commercialisation and increased scale of livestock farming in conjunction with introduced religions, compulsory education (children used to be the herders), relief food and compensation has resulted in many farmers no longer using traditional protection methods (Hazzah, 2006; Woodroffe *et al.*, 2006). The adoption of livestock husbandry techniques is often related to social norms, in conjunction with the perception of how easy or difficult it will be to change management and the perceived outcome (Beedell & Rehman, 1999). Twenty percent (20.0%) of farmers commented that there was nothing they could do and that you cannot stop predators (n = 30; Appendix 7.2). In these cases it is often perceived to be easier and less expensive to trap or poison predators than to change management (Frank, 2004).

Overall, the primary reasons stated by farmers for not using livestock management techniques to protect against predators were a lack of farm infrastructure and preconceptions of negative outcomes. In contrast, the primary reason stated for not using similar techniques in communities in Tanzania was cost (Dickman, 2008). Therefore, continued development of education programmes and demonstration farms to promote the use of livestock management techniques should be tailored to commercial or subsistence farmers and should promote appropriate materials and solutions (Dickman, 2008).

Translocation of 'problem' predators (Effectiveness rank: 4th)

Discussed in Chapter 8.

7.4.11 Photographic tourism (Effectiveness rank: 3rd)

Photographic tourism was conducted on 50.0% of game ranches (Chapter 4) and was considered the third most effective conflict mitigation method to enable predator-human coexistence. However, farmers thought that photographic tourism was limited to prime viewing areas (commented by 27.3% of farmers, n = 22) and was often financially unviable outside of these areas (18.2%; Appendix 7.12).

The proportion of income earned from photographic tourism on wildlife ranches was negatively correlated with the distance the farm was located from protected areas ($R_s = -2.89$, $n = 77$, $p = 0.011$). Game ranches are generally thought to have a greater potential for photographic tourism in areas close to the tourist route, with beautiful scenery and in areas with little livestock production (Barnes & Jones, 2009). Therefore, photographic operations on remote farms, in the flat Kalahari landscape, or on farms ranching livestock and game-stock, are likely to be harder to establish.

The majority of farmers operating photographic tourism considered predators to be important to their financial success (78.4%, $n = 37$), and tolerance of predators was greater in the current study and in South Africa and Zimbabwe (Lindsey *et al.*, 2005c) when photographic tourism was the main income on the farm (income from photographic tourism > 50%, TS = 5.0); ≤ 50%, TS = 3.8; $U(85)$, $z = -2.27$, $p = 0.023$; Table 7.3). Although sightings of free-ranging predators cannot be guaranteed, their presence adds an attraction or 'mystical atmosphere' to the farm (Bothma, 1989) and one farmer commented on the value of seeing signs of predators (e.g. spoor) or hearing species such as spotted hyaenas at night ($n = 22$; Appendix 7.12). However, the main limitation to marketing predators on game ranches cited by 36.4% of farmers was the difficulty in seeing them ($n = 22$). The mammal species most tourists want to see are leopards, lions and cheetahs (Lindsey *et al.*, 2007a; Stein *et al.*, 2010; Maciejewski & Kerley, 2014). Farmers thought that habituating predators to people and cars (commented by 20.0% of farmers, $n = 40$), walking safaris to track predators (15.0%), viewing predators at dens or marking-sites (15.0%) or owning captive predators (12.5%) could increase the role predators play in photographic tourism (Appendix 7.13).

The income from 'leopard tours' in Namibia, in which guests viewed leopards at a kill-site, exceeded the economic costs of carnivore depredation on livestock in the surrounding villages by 2.6 times (Stander *et al.*, 1997b). Similarly, based on tourists' willingness to pay, income from potential African wild dog tours to view the animals at a den-site, was sufficient to compensate for the costs of carnivore depredation on livestock farms (Lindsey *et al.*, 2005b), and several tour operators in Namibia were marketing the viewing of cheetahs at marking-trees (Marker *et al.*, 2003d). However, the habituation of predators for tourism viewing can be difficult to achieve in areas where hunting occurs or where predators are persecuted due to HPC (Swanepoel, 2008). Animals often remain wary of people and avoid roadways (Caro, 1999; Ngoprasert *et al.*, 2007) and habituated predators can become easy targets of persecution (farming community member pers. comm.). Therefore, the habituation of predators and related tourism opportunities should be further explored in Botswana, but their potential benefits should be weighed against their feasibility and the potential danger of disturbing predator behaviour.

7.4.12 Trophy hunting of predators (Effectiveness rank: 2nd)

The role of trophy hunting in conservation is controversial; conservationists' views are polarised from those that oppose hunting in all circumstances to those who think hunting is a way to create incentives for the protection of species and habitat (Hutton & Leader-Williams, 2003). Interviewed farmers viewed trophy hunting of predators, under a quota system, as the second most effective way to promote predator-human coexistence on commercial farmland and significantly more effective than photographic tourism, often the suggested alternative (Table 7.1). Farmers are likely to favour trophy hunting over photographic tourism as it often has lower set-up costs, is more suitable for farms with livestock, can generate greater revenue and has a wider applicability to political and geographical environments (Bond *et al.*, 2004; Lindsey *et al.*, 2006; Lindsey *et al.*, 2007b).

Income from predator trophy hunting can act as a financial incentive to landowners to operate a game ranch and acts as compensation for game-stock lost to carnivore depredation (Hristienko & McDonald, 2007; Lindsey *et al.*, 2012c). A moratorium on leopard hunting was announced in Botswana whilst the questionnaires were being conducted and with the loss of this incentive, in conjunction with proposed changes to the government's game ranching regulations, many farmers stated that they would have to return to cattle farming as game ranching would no longer be profitable (Chapter 4). Similarly, if lion trophy hunting was banned across all countries it has been argued that many hunting operations would become unviable and habitat reserved for wildlife would be converted to alternative land uses, resulting in an approximate 16% loss of habitat for lions (Lindsey *et al.*, 2012c).

A leopard trophy hunt had been conducted by 45.0% (n = 80) of the questioned farms in the five years prior to the survey (2008 − 2012). Trophy hunting is thought to increase farmers' tolerance of predators due to both the financial gain from the trophy fee and the attitudinal change that ownership and control over predators gives (Linnell *et al.*, 2001; Hristienko & McDonald, 2007). In Botswana (Chapter 4) and in South Africa, game ranchers reported they were more tolerant or had more positive attitudes towards species which could be hunted (leopards and spotted hyaenas) than species which cannot be commercially hunted (e.g., cheetahs and African wild dogs)

(Lindsey *et al.*, 2005c). Myers (1981) states that 'in emergent Africa you either use wildlife or lose it; if it pays its own way some of it will survive', a sentiment expressed by many of the farmers questioned (commented by 20% of farmers, n = 20; Appendix 7.14). In Zimbabwe, the only species believed to have declined in numbers after the introduction of private ownership of game-stock was cheetah, which was still in state control (Child, 1995; Child, 2009b).

Increases in tolerance of predators can extend to livestock farmers or community members who allow trophy hunting to occur on their land in exchange for financial benefits. Farmers stopped killing jaguars in Mexico, after receiving income from the trophy hunting of white-tailed deer *Odocoileus virginianus* (Rosas-Rosas & Valdez, 2010), and attitudes towards lions and wildlife conservation in communities in the Okavango Delta in Botswana were reported to be more positive due to the financial and employment benefits of hunting (Thakadu *et al.*, 2005; Mbaiwa *et al.*, 2011). However, these findings are not universal; residents in Slovenia expressed similar attitudes to brown bears in areas with or without bear hunting (Kaczensky *et al.*, 2004). In the current study, game ranchers who had trophy-hunted a leopard once or multiple times in the last five years reported statistically similar tolerance scores to all predators (once TS = 3.3; multiple TS = 3.8) and specifically to leopards (once TS = 4.0; multiple TS = 3.5) as farmers who were pro-leopard hunting but had not taken part in a hunt (TS = 3.1, 3.0; Table 7.3). Some farmers or their neighbours reported leopards were still being killed as 'problem' animals on farms conducting leopard trophy hunting. Similarly, the number of wolves, Eurasian lynx or cheetahs illegally killed does not reduce in relation to the numbers being legally hunted (Andrén *et al.*, 2006; Adams *et al.*, 2008; Person & Russell, 2008; Gros & Caro, n.d.).

Two farmers commented that more frequent trophies were necessary to promote tolerance (n = 89). The leopard trophy-hunting quota for game ranches had declined from 46 in 2007 to 15 in 2012 (BWPA, 2010b; 2012b), partially due to the introduction of an age-based leopard trophy hunting system in 2009. It was hoped this sustainable system would increase the available quota, but its introduction resulted in further declines in quota size, largely due to non-compliance with the new system (BWPA, 2010b; 2012b). When people do not receive the expected benefits from a wildlife initiative, attitudes can often become worse (Romañach *et al.*, 2011). It is, therefore, possible that the benefits from leopard trophy hunting were too infrequent to promote tolerance and, as some farmers, claimed tolerance of leopards due to trophy hunting was greater in the past.

More farmers were in favour of hunting leopards (76.4%, n = 106, z score = 3.23,) than hunting cheetahs (45.1%, n = 102, z score = −1.16), lions (60.8%, n = 97, z score = 0.99) or African wild dogs (30.3%, n = 99, z score = −3.15; $\chi^2 = 22.69$, df = 3, $p < 0.001$). Farmers often commented that African wild dog populations were too small to allow trophy hunting to occur. Previous overexploitation of lions and leopards has led to population declines in Tanzania and South Africa (Loveridge *et al.*, 2007; Packer *et al.*, 2011) and trophy hunting can detrimentally affect predator populations at an interspecific, social grouping, reproductive behaviour and genetic level, if not conducted in a sustainable manner (Loveridge *et al.*, 2007). However, the introduction of minimum age restrictions to the hunting of predators would increase the sustainability of trophy hunting and, with strict regulations and enforcement, could enable the hunting of all species, including African wild dogs (Loveridge *et al.*, 2007; Balme *et al.*, 2009b; Packer *et al.*, 2009; Packer *et al.*, 2011; Lindsey *et al.*, 2012c). However, with declining budgets for species protection (Cousins *et al.*, 2010) the required level of regulation may be difficult to achieve.

Botswana leopard populations were lower than previously assumed in the Ghanzi faming block (Chapter 5). However, the continuation of an age-restricted hunting programme for leopards, if restricted to the hunting of males over seven years old (the age by which the hunted male's offspring are thought to have reached independence) and conducted in conjunction with an international eco-certification initiative for hunting operators, is unlikely to be of detriment to Botswana's leopard populations (Balme *et al.*, 2012). The recommencement of leopard trophy hunting could benefit predators by providing an incentive to operate a game ranch, by reducing conflict between government and the game ranching industry and potentially by increasing tolerance of predators.

7.4.13 'Problem' predator hunting (Effectiveness rank: 1st)

Commercial 'problem' animal hunting was suggested as a potential solution to PAC in the draft HWC 'Green Paper' for Botswana (CAR, 2012) and 'problem' predator hunting was rated by questioned farmers as the most effective method to promote predator-human coexistence on farmland (considered effective by 81.7% of farmers; Table 7.1). In Namibia farmers can call a 'predator hotline' to report when 'problem' predators kill livestock. Evidence is necessary to prove the predator is a problem and, if a hunter is available, the predator is hunted and the trophy fee is split between the hunting company and the farmer (Hein, 1995). In theory this will lead to farmers being more tolerant of leopards taking livestock as they will be able to benefit from the species in the long run (Linnell *et al.*, 2001; Hristienko & McDonald, 2007).

The majority of farmers were in favour of hunting 'problem' predators (78.0%, n = 100) and in allowing trophy-hunted leopards within the quota system to be shot on livestock farms (74.3%, n = 105). Farmers thought that only 'problem' animals should be hunted as opposed to general trophy leopards (commented by 21.3% of farmers, n = 89), that hunting would compensate farmers for their stock losses (24.7%), control predator numbers (10.1%) and increase farmers' tolerance to leopards by enabling them to gain financially from a leopard hunt (23.6%; Appendix 7.14). In Namibia trophy hunters stated they would pay up to 50 – 60% of their trophy fee (ca. US $1682) to farmers to shoot a leopard on their farm (Stein *et al.*, 2010). An equivalent fee in Botswana would compensate farmers for 49% of their annual reported losses ($3399; Chapter 4). Farmers stated that predators were likely to be killed as 'problem' animals anyway, so may as well be hunted (18.0%) and, therefore, killed in a more humane way (7.9%; Appendix 7.14). However, not everyone was in agreement; one farmer commented that additional predators would still be killed as 'problem' animals (n = 22) and 21.1% of farmers thought the system would be abused (n = 57; Appendix 7.14).

Few farmers (12%, n = 23) in Namibia use the 'predator hotline' to organise hunts (Stein *et al.*, 2010) and the scheme has been criticised for the delay between reporting the problem and the hunt. Lindsey *et al.* (2006) found that hunting operators underestimated the amount that clients were willing to pay for hunting a 'problem' animal relative to a regular trophy hunt, potentially contributing to its lack of utilisation. ‘Problem’ predator hunting has also been criticised for providing an incentive for poor husbandry and for the difficulties involved in defining a ‘problem’ animal and ensuring the correct individual is targeted (Lindsey *et al.*, 2006; Packer *et al.*, 2006; Balme *et al.*, 2009b). Additionally, as mentioned by farmers in this study, ‘problem’ predator hunting can potentially encourage farmers to report fictitious ‘problem’ animals (21.1%, n = 57; Appendix 7.14) (Lindsey *et al.*, 2006; Packer *et al.*, 2006; Balme *et al.*, 2009b). On farms near the Kruger National Park in South Africa, the introduction of professional hunting resulted in farmers luring lions onto their property to be hunted and the scheme had to be stopped (Anthony *et al.*, 2010). Similarly, the hunting of ‘problem’ predators was trialled in Botswana in 2005 when five licences for ‘problem’ lions were given to farmers, however, it was described by DWNP as an ‘unsuccessful experiment’ due to abuse of the system (BWPA, 2006a, farming community member pers. comm.). The re-instigation of ‘problem’ animal hunting in Botswana would require a thorough assessment of its applicability and the implementation of strict regulations to prevent its abuse and meaningful distribution of the funds (Jorge *et al.*, 2013). However, revenue from ‘problem’ animal hunts has reduced the number of animals killed in Zimbabwe (Child, 2009b) and the approach should be evaluated further as a potential means of reducing indiscriminate lethal control of predators.

7.4.14 Conclusion

Farmers perceived the hunting of ‘problem’ predators or predator trophy hunting as the most effective methods of enabling farmers and predators to coexist on farmland. However, in light of the Botswana government’s recent negative stance on hunting (with the total ban of trophy hunting on state land from January 2014 [MEWT, 2013] and the moratorium on leopard trophy hunting from January 2013 [BWPA, 2012c]) it is unlikely this method of conflict mitigation will be easily adopted. Nevertheless, there is no clear evidence that PAC methods in Botswana at the time of study are reducing HPC and they could in fact be causing other areas of wildlife management and conservation to be neglected (CAR, 2011; M. Selebatso pers. comm. cited in UNDP, 2007). Therefore, conducting a feasibility study into ‘problem’ predator hunting, as a potential alternative would be worthwhile.

Encouragingly, livestock management techniques were rated favourably by farmers and the kraaling of young calves (67.8%) and kids/lambs (92.9%) was practised by the majority of farmers. However, there were opportunities for substantial improvement. The development of contact lists, which enable farmers experiencing losses to contact and receive advice from farmers who are successfully utilising livestock husbandry techniques, would be beneficial. In contrast, game-stock management techniques were negatively perceived, as they were believed to be largely ineffective or too expensive to implement. Promoting the introduction of buffer species onto game ranches and the use of breeding camps for vulnerable species, in particular, could be affordable solutions which could alter farmers' tolerance of predators.

As found in previous surveys, farmers had negative attitudes towards the Botswana compensation programme and its application to game-stock is unlikely to be possible or beneficial. However, revision of the compensation programme as suggested in the draft HWC 'Green Paper' (CAR, 2012) is likely to benefit farmers ranching livestock and government resources.

Ways to diversify income on game ranches and innovative approaches to maximise the role of predators in photographic tourism, such as the tracking and viewing of predators at kill- or den-sites, would increase the benefits of predators to farmers. Education campaigns on the advantages of conservancies and disadvantages of single game ranches are needed to promote conservancy membership. Government-introduced incentives to encourage the formation of conservancies could assist in changing negative attitudes towards conservancy membership and would assist the 36.2% of game ranchers who wish to join such associations to achieve this goal.

Overall, no single conflict mitigation technique is likely to be successful and a combination of techniques will need to be applied (Madden, 2004; Distefano, 2005). For example, Balme *et al.* (2009b) found that restricting lethal control of leopards to habitual livestock killers, providing education on husbandry techniques and redistributing leopard trophy permits more fairly across the farming area, resulted in increases in the local leopard population. Education about carnivores in the area and the dissemination of information about the available conflict mitigation techniques is likely to result in reduced conflict (Ericsson & Heberlein, 2003; Marker *et al.*, 2003d; Lindsey *et* *al.*, 2009c). Additionally, information sharing with landowners to discuss perceptions, feelings and knowledge of mitigation methods through a participatory approach is likely to be beneficial to optimise techniques (Messmer, 2000; White & Ward, 2010; Brink *et al.*, 2011).

Chapter Eight: Translocation of 'problem' predators: an effective way to mitigate conflict between farmers and predators?

8.1 Chapter overview

Chapters 4 and 7 showed that conflict between farmers and predators can lead to the indiscriminate removal of predators on farmland through lethal control. For threatened species, lethal removal is often undesirable and the alternative, predator translocation, was perceived by farmers as the fourth most effective method to enable farmers and predators to coexist on farmland (after 'problem' predator hunting, trophy hunting of predators and photographic safaris; Chapter 7). The Botswana DWNP translocates 'problem' predators as their preferred method of removal for protected species. However, the survival of these predators and the effectiveness of the translocation at reducing conflict at the removal site have not been documented. This chapter estimated the incidence of predator translocation in Botswana and examined farmers' perceptions of the effectiveness of translocation at reducing stock losses. It also determined the post-release survival of translocated cheetahs, the predator perceived, in this region, to cause the greatest economic losses on game ranches (BWPA, 2005).

Between 2010 and 2012, 103 predators were reported as having been translocated due to HPC. Survival rates for translocated cheetahs (18.2% survived one year) were 2.1 times less than reported for large felids in a review of HPC translocations (Fonturbel & Simonetti, 2011); poor survival was thought to be related to homing behaviour and wide-ranging movements post-release. The majority of farmers who had translocated a 'problem' predator from their farm within the last 12 months thought that the translocation was ineffective at reducing stock losses, both in the short-term (59.1%) and in the long-term (63.6%). Additionally, at least one of the monitored cheetahs continued to predate on livestock after release. In relation to the poor survival, large costs and failure to reduce HPC, this study concludes that the translocation of 'problem' predators in Botswana should no longer be conducted and that conflict mitigation methods should focus on techniques which promote predator-human coexistence.

8.2 Introduction

Translocation is defined as 'a deliberate and mediated movement of wild individuals or populations from one part of their range to another' (IUCN, 1998). It can be carried out: commercially, during the establishment of a game ranch, for conservation purposes (to reintroduce or supplement a species in its former range) or to reduce HWC (Massei *et al.*, 2010; Seddon *et al.*, 2012). Reintroductions and supplementation are the most commonly practised forms of translocation, followed by translocations to reduce HWC (Fischer & Lindenmayer, 2000). The translocation of 'problem' animals is popular with the public as an alternative to lethal control; it is often regarded as a humane method (Massei *et al.*, 2010), with the undocumented belief that translocated 'problem' animals will 'live happily ever after' (Craven *et al.*, 1998). As a result, there is often pressure on conservation organisations to translocate a 'problem' animal in order to prevent it being killed and since the 1980s, translocation has become, in many areas, a standard method for dealing with 'problem' predators (Rogers, 1988; Linnell *et al.*, 1997; Massei *et al.*, 2010). The majority of HPC translocations have taken place in North America and Africa, involving nearly all large carnivores (Fonturbel & Simonetti, 2011). Black bears *Ursus americanus* are the most frequently relocated species (Linnell *et al.*, 1997), however, HPC translocations of tigers, jaguars, leopards, wolves, brown bears and cheetahs have also been conducted (Rogers, 1986; Riley *et al.*, 1994; Bradley *et al.*, 2005; Goodrich & Miquelle, 2005; Marnewick *et al.*, 2009; Athreya *et al.*, 2010; Isasi-Catala, 2010; Weilenmann *et al.*, 2010).

The rationale behind HPC translocations is to reduce stock losses by removing the specific 'problem' predator and to increase farmers' tolerance towards predators by offering the farmer an opportunity to obtain help if a predator's presence can no longer be accepted (Marnewick *et al.*, 2009). For HPC translocations to be deemed successful, the reduction of conflict at the capture area should be the primary requirement (Linnell *et al.*, 1997; Massei *et al.*, 2010), but often the effect that specific translocations have upon conflict resolution is not documented (Linnell *et al.*, 1997; Massei *et al.*, 2010). Translocation should also be cost-effective in relation to other available conflict mitigation methods (Massei *et al.*, 2010), yet costs are rarely reported. For example, in a review of 180 translocation publications only 3% specified the costs involved (Fischer &

Lindenmayer, 2000). Lastly, translocated individuals should have an 'acceptable' chance of survival, herein defined as surviving one year post release (Fonturbel & Simonetti, 2011).

The successful reintroduction and establishment of healthy populations of wolves, cheetahs and lions have been described (Bradley *et al.*, 2005; Purchase *et al.*, 2006; Hayward *et al.*, 2007a). However, high mortality rates and incidences of homing behaviour and returning to livestock predation have also been reported (Linnell *et al.*, 1997). The Botswana DWNP operates a policy to translocate 'problem' predators from farmland into protected areas. Translocations are conducted with limited to no follow-up (pers. obs.) and the evaluation of the effectiveness of translocation is highlighted 'as needed' in the Botswana National Predator Strategy (Winterbach, 2008).

One of the species commonly translocated are cheetahs, the predator species perceived, in this region, to cause the greatest economic losses on game ranches (BWPA, 2005). Cheetahs' display all of the characteristics necessary for translocation, they consume a broad range of prey species, can tolerate a wide variety of habitats and have an exploratory nature (Caro, 1994). Populations of cheetahs have been re-established in Matusadona National Park (Purchase, 1998; Purchase & Vhurumuku, 2005; Purchase *et al.*, 2006), Phinda Game Reserve (Hunter, 1999) and in other fenced reserves in South Africa (Hayward *et al.*, 2007a; Marnewick *et al.*, 2009). However, cheetahs generally have lower survival rates than other translocated carnivores (Hayward *et al.*, 2007a) and mortalities have been attributed to the presence of resident cheetahs and lions (Hofmeyr & van Dyk, 1998), or to translocated animals leaving protected areas and being killed by humans (Du Preez, 1970; Purchase, 1998).

Between 2003 and 2011, CCB operated a cheetah translocation programme in association with the DWNP, to monitor the outcome of these translocations and determine if the programme should be continued. This chapter aims to discuss the success of translocation as a conflict mitigation technique in relation to cheetah survival data from the CCB translocation programme. Additionally it will document all predator translocations conducted in Botswana between 2010 and 2012 and it will assess farmers' perceived effectiveness of translocation at reducing stock losses to carnivore depredation through questionnaires.

8.3 Study area and methodology

8.3.1 Study area

Data regarding the incidence of predator translocations were collected nationwide using DWNP resources (records and interviews with personnel). Questionnaires with farmers regarding their attitudes and experiences of predator translocation were also conducted nationwide. Translocated 'problem' cheetahs in the CCB programme originated from the Ghanzi, Southern or Kgalagadi districts, in western Botswana, where CCB has an active presence. Cheetahs were translocated to protected areas, WMAs or farmland within this region.

8.3.2 Predator translocations conducted in Botswana, 2010 to 2012

All DWNP offices in Botswana (n = 25) were contacted in person or via the telephone and the Problem Animal Control (PAC) department at each office was asked to submit records of predator translocations conducted between 2010 and 2012. The DWNP does not have a standardised method for the collection of predator translocation data and the majority of offices (76%) had no formal records. In these cases, data collection relied on DWNP officers' memory of recent events and efforts were made to contact multiple members of staff. Officers were asked to provide the date, species, sex and number of individuals translocated, the location of the release site and contact details for the farm from which the predator was captured. In addition, the DWNP chief veterinarian and predator conservation organisations in Botswana were asked to submit a list of the translocations in which they had been involved. This list was cross-referenced with the individual DWNP office reports, as was data from questionnaires with farmers.

8.3.3 Interviews: Farmers' experiences and attitudes to predator translocation

As part of the questionnaire survey described in Chapter 4, both livestock and game-stock farmers were asked if they had ever translocated a predator from their property. If applicable, farmers were asked if they viewed the translocation as 'very effective' (VE), 'effective' (E), 'ineffective' (IE) or 'very ineffective' (VIE) at reducing stock losses to carnivore depredation on their farm. Farmers were also asked how likely they would be to translocate a predator in the future on a four-point scale of 'very likely', 'likely', 'unlikely' or 'very unlikely'. A 'do not know' option was available. Any additional comments regarding predator translocations were recorded, as were details of the most recent event, if applicable. The DWNP records of predator translocations were used to identify additional farmers who had translocated a predator in the past. These farmers were contacted by telephone and asked the same questions. In some cases, farmers had experience of translocating multiple species or for multiple reasons; therefore, reported percentages do not always add up to 100%. If a farmer was unable to remember the year the translocation took place it was assumed to have occurred more than 12 months prior to the survey.

8.3.4 Cheetah capture and translocation

Cheetahs were live-captured in double ended box traps (2 m x 0.8 m), between January 2003 and May 2011 using limited access or bait trap-sets as described in Houser *et al.* (2009a). The majority of cheetahs were caught by farmers using their own trap-set, most commonly a limited access trap. Limited access traps use cuttings from *Acacia sp*. to block access to a waterhole, a marking-site or along fence-lines, with the exception of passage through the trap. Cheetahs were transported in wooden squeeze boxes (1.2 m x 0.8 m) to CCB research bases and held temporally in holding pens (20 m x 40 m) until their translocation release. Cheetahs were held for a median of 4 days (range 0 − 16 days, n = 21) between capture and release.

Adult cheetahs underwent a physical health check, using methods adapted from Marker (2002). Cheetahs were anaesthetised using medetomidine (Domitor; Pfizer Inc., New York City, New York; 30 − 40 μg/ kg) and tiletamine–zolazepam (Zoletil; Virbac, Carros, France; 1 mg/kg), using a hand syringe in the squeeze boxes or by dart gun in the holding pens. Cheetahs were evaluated to be in ‘excellent’, ‘good’, ‘fair’ or ‘poor’ health using methods adapted from Marker (2002). Superficial trap-cage injuries were noted but were not considered in the assessment of health status as they do not reflect on wild cheetahs’ health (Marker & Dickman, 2003).

A VHF radio, GPS-cell or GPS-satellite collar was fitted to cheetahs during the physical health check. In male coalitions, the collar was placed on only one member of the coalition. VHF radio collars from Africa Wildlife Tracking (Africa Wildlife Tracking cc, Pretoria, South Africa), weighing 100 g, were monitored daily by car or weekly by plane. Cellular telephone collars from the same company weighing 450 g, or Sirtrack (Sirtrack Limited, New Zealand) GPS-satellite collars weighing 310 g, were set to record GPS locations two to four times a day, as conducted by Houser *et al.* (2009a). Drop-off collars were not used, because the technology was deemed too unreliable and too heavy to place on cheetahs at the time of study (2003 – 2011). Visual follow-up was not conducted on any of the cheetahs.

Release sites were determined through discussion between DWNP and CCB staff at the time of capture and depended upon vegetation, the availability of water and prey, the presence/absence of larger competitors (lions and spotted hyaenas), the cheetahs' social grouping and the threat the cheetah was thought to pose to livestock. Translocation release sites included protected areas, WMAs and farmland. Cheetahs were translocated to their release site in the wooden squeeze boxes and were hard-released at the chosen site. The cheetahs released onto farmland were not thought to be habitual livestock killers.

8.3.5 Survival data and post-release movements

Survival time was calculated as the number of days between the release of the animal (day 0) and death or collar failure (day χ). The success rate was defined as the 'proportion of individuals that survived one year', in accordance with Fonturbel and Simonetti (2011). The survival of un-collared cheetahs or of un-collared individuals within a collared coalition was unknown. It was presumed that in incidents involving the death of a female cheetah with dependent cubs, the cubs also died.

Spatial analysis was performed in Arcview GIS 3.2. (ESRI Software, Redlands, CA). Cheetahs' daily movements were monitored and, if applicable, the time taken for cheetahs to return to their capture site was recorded. A return to capture site was defined as a recorded GPS location within 23.4 km of the site at which the cheetah was trapped. This is equivalent to the radius of the average cheetah home range on farmland in Namibia, calculated from 41 cheetahs (Marker *et al.*, 2007). The Namibian home range was chosen over the smaller Botswana range (radius: 11.4 km; [Houser et al., 2009a]) for two reasons. Firstly, the Botswana range was rejected as it was based on a limited sample of 5 cheetahs; secondly the Botswana range was much smaller than the Namibian range, as it was unknown if cheetahs were caught towards the edge or centre of their home range utilising the larger Namibian home range was more likely to detect animals which had returned to their capture site. In cases where the specific trapping location was not known, the home range radius was drawn around the central point of the farm or village of capture. Release-site fidelity has been defined as when an animal establishes a territory that encompasses the release site (Bradley *et al.*, 2005). In this study this was specified as when an individual utilises an area within 23.4 km of the release site more than once, post 60 days after release.

8.3.6 Statistical analyses

Questionnaire data were coded for use in the statistical computer package SPSS version 11.0.1. Open-ended questions were coded based on content analysis to identify consistent themes. The Mann-Whitney U-test and the Wilcoxon signed rank test were used to determine the significance of differences in how effective farmers rated translocation at reducing stock losses or how likely farmers were to translocate a predator in the future. Chi-squared tests were used to test differences in proportions using Yates' correction factor for tests with one degree of freedom. Means are presented with their standard deviation (SD). All tests were two-tailed.

8.4 Results

8.4.1 Incidence of predator translocation in Botswana

Between 2010 and 2012 large predators were translocated as 'problem' animals on 74 reported occasions, involving 103 individuals. Lions (41.9%) and leopards (47.3%) accounted for the majority of translocation incidents (Table 8.1). The CKGR (including Khutse Game Reserve) and the KTP (including Mabuasehube Game Reserve) were the most commonly used release sites (70.2% of translocation incidents); other locations included WMAs, farmland and other protected areas (Fig. 2.10 and 2.11). Eighteen lions and 11 leopards were released into the CKGR and 7 lions and 16 leopards into the KTP during this period. Follow-up was reported to have occurred on 12.2% of translocations. The majority of translocations were conducted by the DWNP (80.0%), the remainder being conducted by conservation NGOs (9.1%) or by farmers without any additional assistance (10.9%).

Table 8.1. Incidence of large predator translocations in Botswana between 2010 and 2012; the number of individuals moved is shown in brackets

	2010	2011	2012	TOTAL
African wild dog	1 (9)	0	1 (7)	2 (16)
Brown hyaena	1 (1)	2 (2)	1 (1)	4 (4)
Cheetah	1 (3)	1 (2)	0	2 (5)
Leopard	10 (11)	11 (11)	14 (15)	35 (37)
Lion	5 (10)	17 (21)	9 (10)	31 (41)
Spotted hyaena	0	0	0	0
TOTAL	18 (34)	31 (36)	25 (33)	74 (103)

8.4.2 Farmers' experiences and attitudes to predator translocation

During the questionnaire survey described in Chapter 4, 26.1% (n = 115) of farmers had been involved in the translocation of a 'problem' predator in the past. Predators were equally likely to have been translocated from ranches farming only game-stock (16.3% of ranches had translocated a predator, n = 43), livestock farms (29.6%, n = 27) or L/G farms (i.e. farms stocking livestock and game-stock; 27.0%, n = 37; $\chi^2 = 1.48$, df = 2, p = 0.477). An additional 24 questionnaire surveys were conducted with farmers known to have participated in a predator translocation from DWNP records; in total, 54 people who had previous experience of predator translocations were questioned. The median time between translocation and interview was two years (n = 48, range 0.25 – 28.00 years), an additional six farmers could not remember the year the translocation took place. Farmers had translocated leopards (55.5%), lions (25.9%), cheetahs (16.7%), brown hyaenas (9.3%), black-backed jackals (1.9%), African wild dogs (1.9%) and spotted hyaenas (1.9%).

The predators were removed for causing losses to livestock (83.3%) or game-stock (14.8%), or simply because the predator was on the farm (5.5%). Farmers were equally likely to think the translocation was effective at reducing stock losses in the short-term (VE and E: 57.1% of farmers, n = 49) or in the long-term (46.9%, n = 49, (U)48 z = −1.63, p = 0.102). Farmers' perceptions of effectiveness did not vary with the species farmed, the reason for translocation or the predator species translocated. However, farmers who had translocated a predator within the 12 months prior to study were up to 1.7 times less likely to rate the translocation as effective at reducing stock losses (short-term: VE and E: 40.9% of farmers; long-term: 36.4%; n = 22) than farmers who had translocated a predator more than 12 months ago (short-term: 70.4%, n = 27; (U)49, z = −2.15,

$p = 0.032$; long-term: 55.6%; $n = 27$; $U(49)$, $z = -1.63$, $p = 0.104$). An additional five farmers answered they 'did not know' how effective the translocation was.

Eighty-five percent (84.6%) of farmers who had translocated a predator in the past were 'very likely' or 'likely' to consider translocating predators in the future ($n = 52$), compared to 50.0% of farmers who had never been involved in predator translocation ($n = 80$; $\chi^2 = 18.85$, $df = 1$, $p < 0.001$). Twenty percent (20.2%) of farmers who commented on predator translocation did not want to translocate predators in the future as they wanted to have predators on the farm ($n = 129$; Appendix 8). Other reasons farmers stated as to why they would not translocate a predator in the future, or why they thought that translocation was an ineffective method to promote farmers and predators to coexist on farmland were that predators are not moved far enough away (6.2%) and come back to the farm (16.3%) and that predators do not survive after release (12.4%). Farmers also cited that other predators (6.2%) or new individuals moved into the area (4.7%) and continued to kill livestock or game-stock and that in some cases stock losses to predators increased after the predator translocation (3.9%; Appendix 8). Farmers were also concerned that there was nowhere to move the predators to, because protected areas were unsuitable or at full capacity (7.8%) and that DWNP and conservation NGOs were slow to respond, did not treat animals humanely (10.1%) or were unwilling to assist in translocating predators (7.8%). The primary reason for translocating predators was to reduce livestock or game-stock losses (17.8%), whilst giving the predators a chance at survival, as it was thought that translocation was better than shooting the predator (16.3%, Appendix 8).

8.4.3 Translocated cheetahs

CCB took part in the translocation of 21 social groups of cheetahs (comprising 39 individuals) which, with one exception, were caught as 'problem' animals on farmland between 2003 and 2011. The exception was a male cheetah confiscated from poachers by the Botswana Anti-Poaching Unit. Male coalitions were the most commonly translocated social group (47.6%), followed by single males (28.6%), females with cubs (19.0%) and single females (4.8%). Of the

Table 8.2. Post-release survival and movements of 12 collared cheetahs translocated as 'problem' animals by Cheetah Conservation Botswana between 2003 and 2011

ID #	Health status	Trap cage injuries ƚ	Grouping/ Sex	Release Site §	Distance from capture to release site (km)	Site fidelity λ	Return to capture site ¶	Survival (days)	Outcome	Cause of death
1	Excellent	Y	2 males	WMA	142	-	N	> 7	Collar failure	-
2	Excellent	Y	1 male	Farmland	48	-	Y	31	Died	Shot
3	Good	Y	1 male	Farmland	28	-	Y	46	Died	Unknown
4	Good	Y	2 males ǂ	Farmland	29	Y	N	145	Died	Shot
5	Good	Y	2 males	Farmland	31	N	Y	64	Died	Shot
6	Good	N	1 male	Farmland	191	N	N	633	Died	Unknown
7	Fair	Y	1 male	CKGR	215	Y	N	66	Died	Unknown
8	Good	N	1 female & 5 cubs	CKGR	200	-	N	21	Died	Unknown
9	Good	Y	1 male	KTP	79	N	N	> 981	Collar failure	-
10	Good	Y	1 female & 3 cubs	CKGR	90	-	N	31	Died	Unknown
11	Fair	Y	1 female & 2 cubs	WMA	278	N	N	95	Died	Unknown
12	Excellent	Y	2 males	KTP	170	N	Y	347	Unknown	Unknown

ƚ Minor to moderate trap cage injuries present

ǂ Were released separately

§ WMA = Wildlife Management Area; CKGR = Central Kalahari Game Reserve; KTP = Kgalagadi Transfrontier Park

λ Release site fidelity was defined an individual utilising an area within 23.4 km of the release site more than once, post 60 days after release

¶ Return to capture site was defined as a recorded GPS location within 23.4 km (radius of mean cheetah home range size [Marker *et al*., 2007]) of the site at which the cheetah was trapped

Table 8.3. Estimated costs of translocating a single 'problem' cheetah from a farm 100 km from the research camp to a release site 250 km away including the costs of a GPS-satellite collar and data retrieval (distances are driving distance). Staff costs were estimated at approximately USD $300 for veterinary staff and USD $ 4.5 per hour for project staff, based on local Botswana wages. The stated costs do not include the initial material costs such as traps, holding pens, squeeze boxes and darting and medical equipment

Activity	**Cost (USD)**
Setting traps to catch cheetah (travel and staff)	$634
Collecting cheetah (travel and staff)	$107
Medical work up	$1237
Release of cheetah (travel and staff)	$378
Collection of traps (travel and staff)	$421
Monitoring (GPS-Satellite collar and data retrieval)	$4340
TOTAL	$7117

cheetahs that underwent a physical health check, 88% (n = 26) were deemed to be in 'excellent' or 'good' health; the remaining cheetahs were in 'fair' health. Minor to moderate trap-cage injuries including abrasions to paws, face, shoulders, base of tail and hips were received by 60% (n = 25) of adult cheetahs. The mean linear distance between capture and release sites was 138 km (SD = 75, n = 21).

8.4.4 Survival and post-release movements of translocated cheetahs

A VHF radio collar (n = 1), GPS-cell (n = 4) or GPS-satellite collar (n = 7) were fitted to 12 adult cheetahs. Survival data were available for only 11 of the 12 collared cheetahs, due to the inability to locate the VHF collared cheetah beyond seven days post-release. Median survival time was 106 days (range 46 – 981) for males and 31 days (range 21 – 95) for females (Table 8.2). Only two of the cheetahs survived for more than one year, resulting in a success rate of 18.2% (n = 11), however, a third cheetah survived 347 days (Table 8.2). Three of the four cheetahs whose release sites were less than 50 km from their point of capture returned to the capture site and one cheetah showed site fidelity to the release site. Of the seven cheetahs translocated farther than

50 km from the capture site, one returned to the capture area and one cheetah showed site fidelity to the release site (Table 8.2).

8.4.5 Costs of cheetah translocations

The costs of translocation varied depending on how and where the animal was captured, the social grouping, veterinarian fees, the release point and the follow-up conducted. The estimated cost to translocate a single ‘problem’ cheetah was US $7117; seventy-eight percent (78.4%) of these costs related to post-release monitoring and the physical health check (Table 8.3).

8.5 Discussion

8.5.1 Limitations

The primary limitation in this study was the time delay between the translocation event and the collection of data from farmers; questionnaires were completed a median of two years and in one case 28 years after the incident. This time delay is likely to have affected farmers’ ability to recall the details and outcome of the predator translocation. Individuals who had translocated a predator within the 12 months prior to study were less likely to think it was effective (40.9%) than individuals that had conducted the translocation longer than 12 months ago (70.4%). Memories can be biased by what is known as ‘rosy retrospection’, in which individuals rate past events more positively than they would have rated them when the event occurred (Mitchell & Thompson, 1994). It is possible this bias accounts for the observed differences and, therefore, perceived effectiveness would likely have been lower if follow-up had been conducted nearer to the time of translocation. This study was also limited due to its reliance on DWNP officers’ memories to determine the frequency of which translocation was used as a management tool in Botswana. If the translocation of ‘problem’ predators is to continue a national monitoring system should be instigated to monitor the number of predators being translocated and the potential impact multiple introductions could have on resident carnivore populations. Data quantifying the effect translocations have on stock losses and farmers’ perceived effectiveness of the technique should be routinely collected, potentially at three, six and 12 months post translocation.

8.5.2 General discussion

Between 2010 and 2012 an estimated 74 large predator translocations took place in Botswana, involving 103 individual predators. Records were largely based on DWNP officers' memories and data were often incomplete or missing. Of these translocations, 10.9% were conducted by farmers without the assistance of any government or private organisation, so no formal records of these occurrences exist. Therefore, it is assumed that other unrecorded translocations could have occurred and 74 is the minimum number of translocation incidents that took place during the three year period. Follow-up was conducted on approximately 12.2% of translocations; the outcome of the remaining 87.8% is unknown.

For 'problem' animal translocations to be deemed successful the primary aim should be a reduction of conflict at the capture site (Massei *et al.*, 2010). Additionally, the translocated individual should survive to reproduce (herein estimated as at least one year; [Fonturbel & Simonetti, 2011]), should not cause stock losses at the release site and the translocation should be a cost-effective method of mitigating conflict (Massei *et al.*, 2010). Each of these factors in relation to the current study, are discussed.

8.5.3 Reduction of conflict at capture site

Quantitative data on livestock or game-stock losses, that occurred before and after a predator was translocated, were not available. However, the drivers of conflict are often related to farmers' perceptions of predators and the threat and fear of economic losses rather than actual stock losses (Chapter 4), and often feelings are more predictive of support for management options than direct knowledge (Marker *et al.*, 2003a; Glikman *et al.*, 2011). Therefore, farmers' attitudes to predator translocation are an applicable measure to gauge its effectiveness at reducing stock losses and as a HPC mitigation technique. Fifty percent (57.1%) of the farmers who had previously translocated a predator thought that the translocation reduced their stock losses in the short-term, however, this reduced to 40.9% of farmers if including only those who had translocated a predator within the 12 months prior to study. Translocation or lethal removal of wolves temporarily reduced livestock predation at the capture site in north-western USA (Bradley *et al.*, 2005). Similarly, translocation of golden eagles *Aquila chrysaetos* reduced predation on lambs in

South Dakota (Waite & Philips, 1994), but had no effect on lamb mortality in Montana (Matchett & O'Gara, 1987). Forty-three percent (46.9%) of farmers in the current study thought that translocation reduced their losses in the long-term, consequently, long-term benefits to farmers cannot be assured (Jewell, 1982).

The translocation of 'problem' predators relies on the assumption that 'problem' individuals, ones that repeatedly kill livestock, exist and can be identified and captured (Linnell *et al.*, 1997; Massei *et al.*, 2010; Linnell, 2011). However, if 'problem' animals do not exist or if the 'wrong' individual is captured, livestock depredation is likely to continue and farmers commented that translocation was 'ineffective' as other predators continued to prey on livestock. The removal of territorial individuals has been associated with an increase in the number of sub-adult or transient individuals in the area (Philips *et al.*, 1991; Athreya, 2006). This corresponds with farmers' comments that new individuals moved into the area after translocation and, in some cases, stock losses or predator problems increased after the translocation (Appendix 8). Linnell *et al.* (1999) found no evidence that dispersing juveniles or infirm adults killed more livestock than resident adults. However, 'problem' cheetahs often show physical or behavioural abnormalities and there is a general assumption that these animals are weaker or less dominant than other predators (Marker *et al.*, 2003c), so an influx of transient individuals could increase stock losses.

In the current study, predators had been translocated from 21.3% of the surveyed game ranches in Botswana and some ranches had translocated multiple cheetahs or leopards, without noticing a change in their game-stock numbers (farming community member pers. comm.). On game ranches where specific 'problem' individuals do not exist, because predators are eating their natural prey, it could be necessary to remove multiple individuals of all predators species to successfully reduce HPC and noticeable improve game-stock herds (Massei *et al.*, 2010). Such indiscriminate removal of predators on game ranches has been recorded in Namibia (Marker *et al.*, 2003c) and is detrimental to predator conservation (IUCN/SSC, 2007b).

Farmers also commented that predation continues when the translocated predators are not moved far enough away and are able to return to the capture site. Carnivores possess an intrinsic ability to navigate 'home' (Linnell *et al.*, 1997), and four of the 11 collared cheetahs returned to their capture site. It is generally believed that large carnivores are unlikely to return to their place of capture if they are moved over 100 km (Linnell *et al.*, 1997). However, instances of long distance homing such as cheetah #12, which returned to its capture site from 170 km away (Table 8.2), have also been observed in pumas, leopards, wolves and bears (Linnell *et al.*, 1997; Weilenmann *et al.*, 2010). Two cheetahs showed site fidelity to the release site, whilst the majority of cheetahs exhibited wide-ranging movements after release. If the cheetahs' survival time had been longer (median survival time 106 days in males and 31 days in females) it is possible more of the cheetahs would have returned to their capture site.

8.5.4 Translocated cheetahs' survival

The post-release success rate of 18.2% was lower than the 36% success rate for 'problem' cheetahs translocated into the unfenced Matusadona National Park (Purchase, 1998), or the 83% success rate for cheetahs translocated into fenced reserves in South Africa (Marnewick *et al.*, 2009). In a review of felid translocations, success rate was 39% (SD = 21%), although the authors of the review thought that this rate was potentially overestimated due to the tendency for publications to be biased towards those with positive outcomes (Fonturbel & Simonetti, 2011). The higher survival rate of cheetahs in fenced areas such as the South African reserves cited above can be attributed to the increased availability of post-release monitoring and veterinary treatment, which enabled human intervention to quickly treat injuries or problems. The predators' inability to leave the reserve will have also strongly influenced survival rates. Post-translocation roaming behaviour increases exposure to human conflict areas and has been linked to mortalities in many translocated species, including cheetahs (Linnell *et al.*, 1997; Millar *et al.*, 1999). Soft release, when animals are held in a temporary holding enclosure at the release site to acclimatise to the area before their release, can reduce post-release movements (Linnell *et al.*, 1997). This has been associated with increased survival rates (Massei *et al.*, 2010) and could explain the higher success rate experienced in the soft release of cheetahs into Matusadona National Park (Purchase & Vhurumuku, 2005), as compared to this study. However, soft release is expensive and despite the potential survival benefits, most predator translocation programmes are based on hard release (Linnell *et al.*, 1997; Fonturbel & Simonetti, 2011).

The median survival time for the translocated cheetahs in this study was 106 days in males and 31 days in females. Low median survival time has been demonstrated in many predator translocation studies. A review of HPC-driven predator translocations concluded 47% of translocated individuals died within 110 days of release (Fonturbel & Simonetti, 2011), compared to 67% of cheetahs in this study. During the initial 110 days, predators are often trying to return to the capture site or to establish themselves in the new area, and this early high mortality rate suggests that increased monitoring and assistance during this time could increase survival rates. For example, recorded median survival time on South African ranches where such assistance was given, was 38 months for male and 53 months for female cheetahs.

Reviews of predator translocations have shown human-mediated death, namely road accidents, snaring or persecution due to HPC as the leading causes of mortality (Linnell *et al.*, 1997; Bradley *et al.*, 2005; Jule *et al.*, 2008; Massei *et al.*, 2010; Fonturbel & Simonetti, 2011). In the current study, three out of nine cheetahs that were confirmed dead were shot on farmland and human related causes were suspected in three other cases. Other potential causes of death are starvation, disease and inter- and intra-species conflict (Massei *et al.*, 2010). Unfortunately, the lack of visual follow-up and the time delay in reaching deceased cheetahs prevented the confirmation of the cause of death in 70.0% of the cheetahs; this is a common problem when animals are released into unfenced areas (Wolf *et al.*, 1996).

The quality of the habitat in terms of food and shelter and a low presence of competitors including humans, are thought to be the most important factors determining survival of translocated animals (Wolf *et al.*, 1996; Fischer & Lindenmayer, 2000; Moehrensclager & Macdonald, 2003; Johnson *et al.*, 2010; Massei *et al.*, 2010). The poor survival (21 – 66 days) of the three cheetahs released into the CKGR was possibly related to the high density of lions at the release sites compared to the capture sites, where lions have been largely extirpated (Winterbach, 2008). Female cheetah survival in the Serengeti was influenced by number of lions in the study area (Durant *et al.*, 2004) and it is, therefore, possible the translocated cheetahs' naivety of lions contributed to their low survival rate (Bissett & Bernard, 2007; Hayward *et al.*, 2007a; Marnewick *et al.*, 2009).

Other competitors, such as resident cheetahs, are widely distributed across Botswana and it is likely that resident cheetahs were present at all release sites (Klein, 2007). Fatal territorial fights between male cheetahs have been recorded during their reintroduction into electric fenced game ranches in South Africa (Hofmeyr & van Dyk, 1998; Bissett & Bernard, 2007) and between territorial males in the Serengeti (Caro, 1994). Therefore, territorial fights could have contributed to the poor survival rate of males, either directly or due to individuals exhibiting increased ranging behaviour in order to avoid conflict (Jewell, 1982). This ranging behaviour could have increased the risks of human-mediated death, starvation and disease (Linnell *et al.*, 1997; Millar *et al.*, 1999). Chronic stress caused by the translocation process has been hypothesised to exacerbate homing behaviour and increase the translocated individual's risk of starvation, competition and disease (Teixeira *et al.*, 2007; Dickens *et al.*, 2010). Inappropriate trap design or care of translocated predators, leading to trap-cage injuries, could also contribute to the high early mortality rate (Athreya, 2006), and 60.0% of cheetahs translocated in the current study had trap cage injuries. Consequently, translocated individuals often have lower survival rates than resident predators. Translocated cheetahs in this study had substantially shorter survival times than recorded for resident collared cheetahs in Namibia (eight and 17 months in females and males) (Marker *et al.*, 2003b). Similarly, survival times in translocated brown bears and wolves were shorter than for resident predators (Riley *et al.*, 1994; Blanchard & Knight, 1995; Bradley *et al.*, 2005).

Many of the farmers were aware of translocation success rates and commented on the low survival rates as a reason why they would not want to translocate predators in the future, and some farmers commented on the importance of releasing predators in a suitable habitat to improve survival. Although a qualitative assessment of habitat suitability was conducted before each of the cheetah releases, a quantitative analysis of food and water availability, shelter, prey and competitors was not available and can be difficult to obtain (Millar *et al.*, 1999; Massei *et al.*, 2010). Bayesian and GIS-based models have been developed to assist in the selection of release sites for translocated cheetahs and could assist in future releases (Johnson *et al.*, 2010; Lemeris, 2013).

Despite the risks of intraspecific conflict, it is recommended to release translocated predators into areas where the species are known to occur, as a marker of suitable habitat (Lemeris, 2013). However, the introduction of translocated predators could potentially endanger resident predator populations through disease exposure, genetic outbreeding or competition (Wolf *et al.*, 1996). The risks to resident cheetah populations in this study were thought to be minimal because translocated cheetahs underwent disease screening (CCB unpubl. data.), Botswana cheetah populations are genetically similar (Dalton *et al.*, 2013) and cheetahs are not thought to commit infanticide (Hunter & Skinner, 2003). Additionally, competition is generally thought to be more detrimental to the translocated individual than to residents (Massei *et al.*, 2010). However, the repeated translocation of lions and leopards into Botswana's two largest national parks (at least 27 leopards and 25 lions were translocated into the CKGR and KTP between 2010 and 2012) could result in territorial disputes with resident predators, potentially resulting in infanticide. These disputes could disrupt lion and leopard populations (Treves & Karanth, 2003; Balme *et al.*, 2009b), and future studies are needed to investigate the impact of the repeated translocation of these species into Botswana's national parks.

8.5.5 Avoidance of conflict at release site

Follow-up data regarding conflict at the cheetahs' release sites were not collected systematically; however, a calf carcass and cheetah spoor were found at the GPS-point of a translocated male cheetah, so it is known that at least this one individual continued to prey on livestock. The introduction of cheetahs into Matusadona National Park did not increase conflict with cheetahs on the reserve's borders (Purchase & Vhurumuku, 2005). However, other carnivore studies have shown that habitual stock-raiding lions continued killing livestock after translocation (Funston, 2001; Frank *et al.*, 2006), and 25% of translocated wolves and 40% of translocated brown bears continued to prey on livestock or were involved in a second conflict event within two years of release (Blanchard & Knight, 1995; Bradley *et al.*, 2005). The release of translocated leopards in India was associated with an increase in fatal leopard attacks on humans and livestock (Athreya *et al.*, 2010). The authors believed the introduced density of leopards alone could not account for the increase in attacks and hypothesised that the translocation process and the associated stress could have altered leopard behaviour resulting in less fear towards humans and increased levels of aggression (Athreya *et al.*, 2010).

To reduce the potential for animals to continue to prey on livestock, the DWNP's policy is to only release 'problem' predators into protected areas (i.e. away from livestock). However, protected areas in Botswana are unfenced and, as seen in this and other studies, very few translocated predators remain at their release site (Linnell *et al.*, 1997). Therefore, the repeated translocation of predators into nationally protected areas could be increasing HPC on farms bordering national parks and reserves. An alternative suggested in the draft 'Green Paper' on HWC in Botswana is to release predators into game ranches with electrified, 'predator-proof' fencing, thereby preventing the animal from leaving (CAR, 2012). Such reserves with meta-populations of cheetahs and African wild dogs occur in South Africa (Hayward *et al.*, 2007a; Marnewick *et al.*, 2009; Gusset *et al.*, 2010). However, only 10.0% of game ranches in Botswana had electric fencing at the time of survey (Chapter 7) and there are no policies in place as to how 'fenced' predators would be managed. In fenced systems, the predator populations must be intensively controlled to maintain prey numbers and to avoid genetic inbreeding (Lindsey *et al.*, 2011) and the removal of predators from the wild genetic pool is not thought to be desirable for their overall conservation in Botswana.

8.5.6 Cost-effectiveness

The cost to capture, translocate and monitor one cheetah in this study was estimated at US \$7117. Fonturbel and Simonetti (2011) reviewed the costs of felid translocations, finding the mean cost to translocate and monitor an individual was \$3941 (SD = 3286, n = 7). The larger costs in the current study are most likely due to the large costs involved in using satellite collars ($4340 of the reported costs in this study were attributed to the satellite tracking collar and GPS downloads). Fonturbel and Simonetti (2011) compared the cost of translocation with that of compensating farmers for stock losses and estimated the cost of one felid translocation could compensate farmers for seven head of livestock (based on a combined compensatory value for cattle, goats and sheep, F. Fonturbel pers. comm.). In addition to financial costs, the translocation of predators are likely to impact on NGO and DWNP's limited resources, diverting personnel and equipment away from other conflict mitigation activities.

8.5.7 Conclusion

With the substantial costs of translocating 'problem' predators, the method should be successful compared to other mitigation techniques (Jewell, 1982; Massei *et al.*, 2010). In the current study, farmers rated translocation as the fourth most effective method at enabling predator-human coexistence (Chapter 7). However, predator survival rates were low and stock losses continued both at the capture site and potentially at the release site. Many farmers spoke negatively of translocation in terms of predator survival, the efficiency of the organisations involved or the ability of the method to reduce stock losses. Yet farmers believed it gave the predator a chance of survival, compared to the alternative of lethal control, and 84.6% of farmers who had previously translocated a predator stated that they would be likely to do so again in the future. The choice between lethal control and attempting translocation is a dilemma commonly faced by predator conservation organisations and potentially explains why many translocation programmes continue despite the low success rate (Anderson, 1992; Athreya *et al.*, 2010). It is possible that predator translocation programmes give the farmer an opportunity to obtain help if a predator can no longer be accepted (Marnewick *et al.*, 2009). This availability of a coping strategy and perceived control over a risk is associated with reduced perceptions of the threat the risk causes (Dickman, 2008), which could increase tolerance of predators. However, it is difficult to ascertain if this potential benefit justifies the costs.

Evidence from the current study and from Weilenmann *et al.* (2010) suggest that the success of cheetah and leopard translocations in Botswana do not justify the costs. Hence, this study concurs with the conclusion of Linnell *et al.* (1997) that, for carnivore species such as cheetahs, where populations as opposed to individuals are the management units, translocation is unlikely to be justified and the money and time would be better spent on mitigation methods such as livestock compensation, education programmes or in improving farm management (Linnell *et al.*, 1997; Massei *et al.*, 2010; Fonturbel & Simonetti, 2011). The proactive prevention of HWC is likely to be more effective than the reactive use of translocation as a temporary solution.

Chapter Nine: General discussion

9.1 Introduction

Since European colonisation of southern Africa in the 17th century wildlife conservation in the region has undergone numerous stages. Uncontrolled hunting and persecution of wildlife for the development of the cattle industry resulted in dramatic declines in wildlife distribution and abundance (Cumming & Bond, 1991). The subsequent designation of national protected areas for wildlife attempted to, and partially succeeded in, conserving habitat and wildlife biodiversity (Bruner *et al.*, 2001; Krug, 2001). However, this 'protectionist' theory often resulted in wildlife being restricted to protected areas. Without incentives for species conservation outside of these areas, wildlife populations on non-protected land diminished due to neglect or due to persecution as a result of competition for land and resources (Bond *et al.*, 2004; Lindsey *et al.*, 2009c). The introduction of game ranching on private land in southern Africa gave the ownership and control of wildlife populations to landowners. The industry has been reported to protect vast areas of land for habitat and ungulate conservation and was associated with the well-publicised recovery of white rhinoceros, bontebok and mountain zebra populations (Flack, 2003; Lindsey *et al.*, 2009b; Lindsey *et al.*, 2013c). However, environmentally damaging practices also developed on some farms including the stocking of exotic species, land fragmentation through fencing and the potential persecution of predators (Cousins *et al.*, 2008). As a result, the overall impact of game ranches on predator conservation was considered a 'gap in knowledge' (Inskip & Zimmerman, 2009; Balme *et al.*, 2014).

Predators worldwide have declined faster than species at other trophic levels (Millar *et al.*, 1999) and until the 1970s predators were killed within national parks in order to 'protect the game' (Woodroffe *et al.*, 1997; Breitenmoser, 1998; Schwartz *et al.*, 2003). This view of predators as pests still persists (Sillero-Zubiri & Laurenson, 2001), and the main threat to large carnivore conservation outside of protected areas is conflict with humans, because of the threat, either real or perceived, predators pose to livestock or game-stock. Game-stock species are predators' natural prey, therefore, potential losses can be substantial and difficult to prevent (Lindsey *et al.*, 2013b). However, game ranches have the opportunity to benefit from predators through photographic tourism and trophy hunting, and farmers in Namibia were more likely to want predators on their farm as the proportion of income earned from wildlife increased (Lindsey *et al.*, 2013b). However, in cases when HPC on game ranches does occur, it has been described as potentially more intense and more damaging to carnivore biodiversity than other forms of HPC (Marker *et al.*, 2003d; Cousins *et al.*, 2008).

The game ranching industry in Botswana was introduced in 1992 and has grown rapidly, with the potential for further expansion. Botswana is geographically the centre of southern Africa carnivore populations and hosts an estimated 23.6% of the cheetah, 25.1% of the African wild dog, 6.0% of the lion, up to 10.5% of the spotted hyaena and 52.7% of the brown hyaena remaining global populations (Hofer & Mills, 1998a; Durant *et al.*, 2008; Höner *et al.*, 2008; Winterbach, 2008; Woodroffe & Sillero-Zubiri, 2012; Riggio *et al.*, 2013). As a result, Botswana is a vitally important area for predator conservation across the region.

This research aimed to examine the direct costs and drivers of HPC on game ranches in Botswana. It aimed to discuss the influence of predator distributions and densities and farmers' perceptions of predators' primary prey items on HPC. It also aimed to examine the use and perceived effectiveness of available conflict mitigation methods. This chapter will summarise the main results from the research, their limitations and their potential relevance to the wider field of HWC and predator ecology and conservation. Management recommendations and suggestions for further research will be made.

9.2 Summary of findings

Chapters 1 - 3 provided background to the research topic, the study area and methodology. Chapters 4 - 8 were the data chapters; the primary results of which will be discussed.

The number of cattle reported to have been killed by predators in the 12 months prior to the study was positively correlated with the total cattle herd size and was 2.3 times greater if lions, African wild dogs or spotted hyaenas were present on the farm, compared to where they were absent (Chapter 4). Farmers were 27 times less likely to know the number of game-stock killed by predators than livestock, and ranchers reported they found 50.0% of game-stock carcasses compared to 92.5% of livestock carcasses. Consequently, the number of game-stock reported to have been killed by predators in the 12 months prior to the study was largely based on perceptions and was primarily associated with the number of years of farming experience the rancher had (positively correlated) and the presence/absence of lions (6.4 times greater losses if lions were present; Chapter 4).

The reported number of livestock lost to carnivore depredation was correlated with tolerance scores to predators, but was not selected in the regression model as one of the primary correlates of conflict. The reported number or financial costs of game-stock lost to carnivore depredation were not related to farmers' reported tolerance of predators (Chapter 4). Overall, ranchers farming only game-stock reported seven times greater financial losses to carnivore depredation (US $2.1 per hectare) than livestock farmers (US $0.3 per hectare). However, ranchers farming only game-stock reported a significantly greater tolerance of predators (TS = 4.4; scored from 0 – 5, lowest to highest tolerance) than L/G farmers (TS = 3.5) and livestock farmers (TS = 2.6; Chapter 4). African wild dogs (TS = 1.7) and lions (TS = 2.0) were the least tolerated of the predator species, and brown hyaenas (TS = 3.5) and caracals (TS = 3.3) were the most tolerated. On livestock farms, education level (tertiary educated more tolerant), first language spoken (English or Afrikaans speakers more tolerant than indigenous African language speakers) and whether small antelope abundance was perceived by farmers to be increasing, stable or decreasing (increasing more tolerant) were the primary drivers of tolerance. On game ranches the presence of cattle (not farming cattle more tolerant) and the proportion of buffer species stocked (larger proportion more tolerant) were the primary drivers of tolerance of predators (Chapter 4).

The presence of lions, African wild dogs and spotted hyaenas reported by farmers (the carnivores associated with the greatest number of cattle lost to predation, Chapter 4), was largely restricted to farms near protected areas, resulting in 'hot-spots' of HPC (Chapter 5). In contrast cheetahs, leopards and brown hyaenas were widespread. Inconsistences were detected between the assumed densities of predators documented in the Botswana National Predator Strategy and the density of predators detected from spoor and camera-trap surveys on the Ghanzi farming block.

The density of leopards, a species frequently involved in HPC was overestimated by 2.6 − 4.1 times, whilst brown hyaena density, a low-conflict species, was underestimated by 4.9 − 49 times (Chapter 5).

Farmers who perceived livestock to be the primary prey item killed by predators on their farm reported lower tolerance scores to predators (TS = 1.3) than farmers who perceived predators to be consuming free-ranging wild prey species (TS = 3.8; Chapter 6). Cheetahs were thought by the BWPA to cause the largest economic losses on Botswana game ranches (BWPA, 2005) and were reported to cause 23.6% of the game-stock, 15.8% of the cattle losses and 4.3% of the smallstock losses reported to have been caused by carnivore depredation (Chapter 4). From scat-analysis cheetahs were found to prey primarily upon kudu, duiker, steenbok and springhares. Scat-analysis results demonstrated that cheetahs preyed upon game-stock (especially springbok) less frequently than was perceived by farmers. The maximum number of game-stock estimated to be killed by cheetahs based on scat-analysis (4.0 – 15.1 game-stock animals per farm, per year) was 1.6 − 5.9 times less than perceived by game ranchers (Chapter 6).

Financial payments for livestock losses to carnivore depredation through insurance or compensation plans were poorly adopted on commercial farms and were considered the least effective methods of promoting predator-human coexistence on farmland (Chapter 7). Hunting of ‘problem’ predators, trophy hunting of predators under quota and photographic tourism, all methods which change predators from being perceived as an economic cost to an asset, were considered the most effective methods of promoting coexistence. The utilisation of game-stock management techniques was considered significantly less effective than livestock management techniques (Chapter 7). However, the use of electric and bonnox perimeter fencing was associated with fewer stock losses in HPC ‘hot-spots’ than farms with standard game fencing and farmers who stocked a buffer-prey species on their game ranch reported a greater tolerance of predators than farmers without buffer species (Chapters 4 and 7).

Farmers rated the translocation of ‘problem’ predators as the fourth most effective method at enabling coexistence (Chapter 7). However, predator translocations were shown to be expensive, ineffective at reducing stock losses and the survival of translocated cheetahs was poor (18.2% survived one year post-release; Chapter 8).

9.3 Limitations

A brief summary of the primary limitations of this study is below; additional details and further limitations are discussed in the relevant data chapters.

No ground-truthing was conducted to validate the number of livestock and game-stock losses reported by farmers or the farmers' reported tolerance of predators (Chapter 4). Therefore, questionnaire data should be considered as a perception of the direct costs of carnivore depredation and may not be a true reflection of when predator removals would occur (Marker *et al.*, 2003d; Schumann *et al.*, 2008). Farmers' responses can be affected by misinformation, social norms and social-desirability bias or with aims to influence governmental policy on HPC and predator control (Fisher, 1993; Gillingham & Lee, 2003; Gusset *et al.*, 2008a; Lindsey *et al.*, 2013b). Future studies to validate the actual costs of carnivore depredation on game-stock, would be beneficial (White *et al.*, 2005; Dickman, 2008; Winterbach *et al.*, 2012). However, understanding farmers' perceptions of the costs of carnivore depredation can be equally important as actual costs for the development of conflict mitigation methods (Mishra, 1997; Madden, 2004; Ray *et al.*, 2005), therefore, the lack of ground-truthing did not affect the validity of the conclusions presented.

'Snowball' sampling was conducted due to the lack of available contact lists for livestock farms in the primary game ranching regions. Although this reduced the external validity of the data, 'snowball' sampling has the advantage of higher response rates (Sadler *et al.*, 2010). As farmers primarily suggested their neighbours, it was, therefore, considered the most appropriate of the available techniques to identify and compare HPC on livestock farms and nearby game ranches. However, future surveys should aim to use randomised sampling.

To minimise observer bias the identification of prey remains in scat-analysis (Mills, 1992) and the identification of individuals in camera-trap data (Foster & Harmsen, 2012) was conducted by the same observer on two separate occasions. In the future, conducting questionnaire content analysis, scat-analysis and predator identification with two observers or by using computer recognition technology would be recommended to remove potential observer bias (Fink & Kosecoff, 1996; Kelly, 2001; Moyo *et al.*, 2006; Foster *et al.*, 2011).

9.4 Advancing knowledge of large predator ecology and conservation

Research to monitor and evaluate the distribution and status of predator populations, especially outside of protected areas, is recommended as a range-wide conservation priority for African large carnivores (Ray *et al.*, 2005). In Botswana, monitoring predator populations, especially cheetahs and leopards, is one of the key targets of the National Predator Strategy (Winterbach, 2008). The spoor surveys conducted in Chapter 5 were part of this strategy to establish trends in cheetah populations in agricultural zones (Winterbach, 2008). The detected cheetah density of 0.32/ 100 km^2 on the Ghanzi farming block was within the previously assumed range documented in the National Predator Management Strategy (Winterbach, 2008). Detected leopard densities (0.37/ 100 km^2) were 2.6 – 4.1 times lower and detected brown hyaena densities (0.49/ 100 km^2) were 4.9 – 49 times higher than assumed in the National Predator Strategy (Winterbach, 2008). These discrepancies impact on management decisions, such as identifying target areas for conservation and, in the case of leopards, for trophy-hunting quotas.

Understanding the feeding ecology of cheetahs in Botswana was identified as a key target in the National Conservation and Action plan for Cheetahs (DWNP, 2009). By analysing scat the current study established free-ranging wildlife species as the primary prey items consumed by cheetahs' and demonstrated their preference for wild game despite the higher availability of livestock (Chapter 6). The apparent avoidance of livestock in the cheetah's diet emphasises the potential for cheetahs and livestock farmers to coexist if perceptions and tolerance of cheetahs can be increased. However, game ranchers perceived cheetahs to kill more game-stock than calculated by scat-analysis and farmers who perceived that predators primarily fed on livestock were significantly less tolerant of predators (Chapter 6). Therefore, changing these perceptions should be of the upmost priority to promote coexistence.

9.5 Advancing knowledge of human-wildlife conflict and conflict mitigation

Human-predator conflict is considered the primary threat facing large African carnivores (Ray *et al.*, 2005), yet the extent and causes of HPC on game ranches in southern Africa is described as a gap in knowledge (Inskip & Zimmerman, 2009; Balme *et al.*, 2014). Overall, game ranchers were reported to be more tolerant towards predators than livestock farmers in Namibia (Lindsey *et al.*, 2013b) and South Africa (Swanepoel, 2008; Lindsey *et al.*, 2009c; Thorn *et al.*, 2012). In Botswana, conflict between predators and humans on game ranches had been reported in several studies (Swarner, 2004; BWPA, 2006b; Selebatso *et al.*, 2008; Kent, 2011), but the extent and drivers of this conflict was largely unknown (McNutt, 2005). The Botswana National Predator Strategy aims to 'monitor and evaluate the level of people-predator conflict' and to 'investigate communities' values and attitudes toward wildlife with the aim of increasing their tolerance' (Winterbach, 2008). Similar targets in the Botswana Action Plan for Cheetah and African Wild Dog Conservation aim to identify the extent of conflict, conflict areas and the perceived and economic value of cheetahs and African wild dogs (DWNP, 2009) The analysis of the direct costs and drivers of HPC on game ranches in this study (Chapter 4) contributed towards fulfilling these aims and addressing this gap in knowledge (Inskip & Zimmerman, 2009; Balme *et al.*, 2014).

Many of the causes and effects of HWC are similar in conflict cases around the world and are often applicable across taxonomic groups (Madden, 2004; Treves, 2009a). Dickman (2013) identified eight key factors associated with HWC, of which six were representative of conflict with both carnivores and primates, namely economic or opportunity costs of HWC, visibility of species, fear and a lack of knowledge, wealth, cultural norms and social tensions. These factors were all thought to play a role in HPC in the current study. The direct costs of carnivore depredation on livestock were correlated with tolerance scores (Chapter 4) and the inability to trophy hunt specific predator species, such as cheetahs, (i.e. an opportunity cost) was associated with reduced tolerance to those species (Chapters 4 and 7). The abundance and damage caused by more visible species, for example diurnal species, were often overestimated (Chapters 4 – 6). Tertiary educated farmers, those with alternative sources of wealth or those from an English or Afrikaans speaking culture were more tolerant of predators (Chapter 4). Lastly, social tensions were detected between game ranchers and livestock farmers (Chapter 5) and between farmers and government bodies (Chapter 4). Therefore, the findings from the current study fit within the general picture of HWC and impact on the global understanding of the topic. Similarly, the game ranching model is being considered as a potential model for natural resource management in North America (Licht *et al.*, 2014). Therefore, the results of this study have a potential impact beyond the southern African region.

Conflict between the game ranching community and wildlife (including predators and elephants) was originally overlooked in the 2011 draft of the HWC 'Green Paper' for Botswana (but was later incorporated in the 2012 draft) (CAR, 2011; 2012). This failure to acknowledge stakeholders concerns can lead to resentment of governing bodies and increased HWC (Madden, 2004). For example, in Kenya in 2003, local Maasai tribesmen killed over half the lions in Nairobi National Park after feeling the government was taking no action to help them with their livestock losses to carnivores (Anon, 2003). Often the first step in relieving HPC and promoting predator-human coexistence is to acknowledge the problem and to acknowledge the different stakeholders' interests and concerns in an objective fashion (Weber & Rabinowitz, 1996; Sillero-Zubiri & Laurenson, 2001; Madden, 2004). The current study gave game ranchers an opportunity to voice their concerns about carnivore depredation on game-stock. These concerns can now be included and addressed in future management plans.

Overall, game ranching is likely to contribute to predator conservation by providing habitat outside of nationally protected areas, where predators will be tolerated to a greater extent than on livestock farmland (Chapter 4). Tolerance was greatest on game ranches without cattle, therefore, the tolerance of predators and conservation benefits of game ranches across southern Africa could be maximised by promoting game-stock only ranches and reducing L/G farmers' economic reliance on livestock (Chapter 4). Reducing farmers' reliance on low income, high off-take wildlife uses such as game-stock meat and live game sales and increasing their use of high-income wildlife uses such as trophy hunting and photographic tourism is also likely to increase tolerance (Chapter 4) (Lindsey *et al.*, 2005c; Lindsey *et al.*, 2009b). A combination of both photographic tourism and trophy hunting will be necessary as the photographic tourism market is potentially limited on remote farms, or on farms ranching livestock in conjunction with game-stock (Chapter 7) (Barnes & Jones, 2009).

One option to enable the adoption of high-income wildlife uses is to encourage farms to become part of a conservancy. Conservancies have less management costs compared to single owner farms. They increase the ecological stability of the farm and enable the reintroduction of the full range of species native to the area, thus promoting photographic tourism and trophy hunting operations (Lindsey *et al.*, 2009b). Conservancies were rare in Botswana (5.0% of interviewed farms), but 36.2% of game ranchers wished for their ranch to join a conservancy in the future, and these ranchers reported greater tolerance scores to predators than farmers who did not wish to join a conservancy (Chapters 4 and 7). Conservancy members in Namibia, South Africa and Zimbabwe also had greater tolerance of predators compared to farmers on other land uses (Schumann *et al.*, 2008; Lindsey *et al.*, 2009b), therefore, government initiatives to promote the formation of conservancies across the region would be beneficial (Lindsey *et al.*, 2009b; Lindsey *et al.*, 2013b).

In particular, game ranches have been highlighted as having potential for conserving cheetahs and African wild dogs. Therefore, monitoring the development of game ranches and their influence on predator conservation is one of the targets of the Botswana National Action Plan for Cheetahs and African Wild Dogs (DWNP, 2009). Game ranches' potential suitability as a land use for cheetah conservation was demonstrated by the greater detection of cheetahs on game ranches than on livestock farms in the Ghanzi farming block (Chapter 5). This finding is most likely due to the greater availability of prey on game ranches and due to farmers' greater tolerance of cheetahs for killing game-stock (TS = 3.8) than for killing cattle (TS = 2.6; Chapter 4). African wild dogs, however, were the least tolerated of all of the predator species for killing game-stock (TS = 3.2) and cattle (TS = 1.67) in this study (Chapter 4) and in South Africa (Lindsey *et al.*, 2005c). Without increased benefits possibly from photographic tourism (for example the viewing of African wild dogs at dens as suggested by Lindsey *et al.* [2005b]), it is unlikely game ranches will positively impact on free-ranging African wild dog conservation to the same extent as the other predator species.

The use of conflict mitigation techniques, the reasons why individuals choose not to adopt mitigation methods and farmers' perceptions of the effectiveness of these methods was discussed in Chapter 7. This data contributed to the fulfilment of the aims of the Botswana National Predator Strategy (Winterbach, 2008) and the Southern Africa Action Plan for Cheetah and African Wild Dog Conservation (IUCN/SSC, 2007b) to 'monitor and evaluate ……the effectiveness and impact of problem animal control measures'. Interviewed farmers thought that the introduction of 'problem' predator hunting was the most effective method at enabling humans and predators to coexist, followed by trophy hunting of predators and photographic tourism. There was no clear evidence that farmers that took part in leopard trophy hunting had a greater tolerance of leopards than farmers who were pro-leopard hunting but had not personally benefited from a hunt. However, game ranchers' in general reported a greater tolerance of predator species which could be hunted than species which could not be hunted, a trend that was also observed in South Africa (Lindsey *et al.*, 2005c). Leopard trophy hunting may provide an incentive to operate a game ranch and act as compensation for game-stock losses to carnivore depredation (Hristienko & McDonald, 2007; Lindsey *et al.*, 2012c). Therefore, the closure of leopard hunting in Botswana could have the potential to decrease farmers' tolerance of leopards and studies to investigate if tolerance has changed would be beneficial.

Aspects of livestock management were considered effective at enabling predator-human coexistence (Chapter 7). Farmers in 'hot-spots' of HPC who kraaled their calves reported losing fewer cattle than farmers who did not kraal calves. On farms where livestock management techniques which protect against carnivore depredation were not utilised, farmers stated the primary reasons were a lack of farm infrastructure and concerns about possibly negative outcomes (for example increased prevalence of disease). This contrasts with the reasons stated by farmers for not using game-stock management techniques (Chapter 7) and with the reasons stated by subsistence farmers for not utilising livestock management techniques in Tanzania; in these cases economic factors were of primary importance (Dickman, 2008). Overall, game-stock management techniques were thought to be ineffective at enabling predator-human coexistence (Chapter 7). However, preliminary information from this study implies 'predator-proof' fencing, breeding camps for vulnerable species and, in particular, the stocking of buffer species can reduce losses and increase tolerance of predators. Few studies have empirically tested the buffer species diversionary feeding theory (Graham *et al.*, 2004), however, the stocking of buffer-prey species was perceived by farmers to reduce predation on expensive game-stock items or livestock, therefore, the stocking of buffer species should be tentatively encouraged, whilst further studies are conducted.

Reducing predator numbers to a more 'tolerable' number was rated as equally effective at enabling predator-human coexistence as conservancies and livestock management techniques and was considered to be more effective than game-stock management techniques. The Botswana Action Plan for Cheetahs and African Wild Dogs aims to 'clarify, improve and standardise the monitoring of the cause and extent of intentional killing…[of] cheetah and African wild dog' (DWNP, 2009). The killing of predators, potentially illegally, can be a sensitive issue and questions are often left unanswered or are not answered truthfully in conventional interviews and questionnaires (Marker *et al.*, 2003d). Chapter 7 reported on the incidence of lethal control using the RRT. The RRT has been shown to increase the proportion of honest answers to sensitive questions, thereby enabling a more accurate estimate of the incidence of predator removals (St John *et al.*, 2012). Approximately, 18.7% of farmers had killed a protected predator without reporting it to the DWNP (a legal requirement). Official government registers of predator mortality are considered an underestimate of the true impact of HPC on predator populations (Kent, 2011). Therefore, this data can be used to form better estimates of the numbers of predators being killed as a result of HPC, in order to advise predator management plans.

The translocation of 'problem' predators was rated by farmers as the fourth most effective conflict mitigation method at enabling farmers and predators to coexist. The National Predator Strategy aims to 'develop and implement guidelines for the translocation of predators' and states an 'evaluation of the effectiveness of translocation for the different species is needed'. The survival rates of 'problem' cheetahs that had been translocated (18.2% survived one year) was lower than the 39% reported in a review of translocated felids (Fonturbel & Simonetti, 2011). This low survival rate was potentially due to hard releasing cheetahs into unfenced areas which enabled cheetahs to undertake wide-ranging movements post-release (Chapter 8). The primary reasoning for conducting HPC translocations is to reduce conflict at the capture site, however, the effect translocations have upon conflict resolution is generally not documented (Linnell *et al.*, 1997; Massei *et al.*, 2010). This study provided information on conflict reduction based on farmers' perceived effectiveness of translocation at reducing stock losses; fifty-nine percent (59.1%) and 63.6%% of farmers who had translocated a predator in the 12 months prior to study thought that the translocation was ineffective at reducing their stock losses in the short-term and in the long-term, respectively. The chapter concludes that survival rates and effectiveness were poor and efforts should be focused on mitigation methods which promote coexistence rather than on the removal of predators.

9.6 Management recommendations

9.6.1 Prey populations

Cheetahs were shown to primarily prey upon kudu, steenbok, duiker and springhares (Chapter 6), these are indigenous species which were common on both commercial livestock and game-stock farmland (Chapter 5). Eighty-three percent (83.3%) of livestock farms and 43.4% of game ranches reported encouraging natural prey or stocking a buffer-prey species on their farms (Chapter 7). The presence of natural prey and buffer-prey species was associated with increased tolerance of predators and reduced smallstock losses to carnivore depredation (Chapter 4). However, on community lands natural prey is thought to be declining due to land conversion to agriculture, habitat degradation and poaching (Mordi, 1989; Moleele & Mainah, 2003). Management options to promote veld management, to reduce feral domestic dogs (which often kill wildlife) and to diversify and improve rural livelihoods to discourage poaching could aid the recovery of natural prey populations, which could reduce HPC.

9.6.2 Predator population monitoring

The density of leopards on commercial farmland was substantially lower than presumed in the Botswana National Predator Strategy; in contrast, brown hyaena densities on commercial farmland were substantially higher (Chapter 5). Determination of predator densities in communal farming areas would be beneficial to predator management plans. The continued monitoring of predator populations to establish population trends is also necessary to optimise predator conservation.

9.6.3 Predator removal or coexistence

Translocated 'problem' predators including cheetah in this study (Chapter 8) often have low survival rates (Fonturbel & Simonetti, 2011). Translocation is expensive, ineffective at reducing stock losses (especially game-stock) and potentially increases stock losses near release sites (Chapter 8). In light of these limitations, it is recommended that the Botswana translocation programme for HPC is stopped for all species whose management occurs at a population rather than an individual level, as suggested by Linnell *et al.* (1997). Stopping the translocation of 'problem' predators would refocus efforts on predator-human coexistence and conflict prevention. Resources and funds spent on predator translocation could be invested in livestock and game-stock management techniques or to promote the generation of alternative incomes (Linnell *et al.*, 1997; Massei *et al.*, 2010; Fonturbel & Simonetti, 2011). Any change in translocation policy should be clearly explained to the farming community to avoid any confusion or mistrust as to why this option is no longer available.

Twenty-two percent (21.5%) of farmers reported they had used poison to kill predators (Chapter 7), potentially also killing mammalian and avian scavengers. Poisoning is considered one of the primary threats to vulture populations worldwide (Ogada *et al.*, 2012). Banning or restricting the sale of pesticides, such as Temik, which are commonly used to kill predators, could reduce incidences of indiscriminate lethal control.

9.6.4 'Hot-spots' of human-predator conflict

'Hot-spots' of HPC were identified close to protected areas, with farmers reporting 2.1 times more cattle losses if their farms were located within 100 km of a protected area, compared to those farms which were farther away. These 'hot-spots' of conflict were mainly due to lions (and to a lesser extent African wild dogs and spotted hyaenas) being present on farms in these areas. Farmers reported removing lions on 87.0% of farms upon which they were resident. Removals of lions on farms around the KTP in Botswana has resulted in a population decline within the reserve

(Castley *et al.*, 2002) and it is possible similar declines could be occurring in Botswana's other national parks and game reserves (Packer *et al.*, 2013). The draft HWC 'Green Paper' suggests creating buffer zones around nationally protected areas to reduce HWC by acting as a buffer from livestock farms (CAR, 2012), and Winterbach *et al.* (2012) suggested these buffer zones could be designated as game ranches. Although difficult to instigate with existing farms, land use planning to designate future farms near protected areas as game ranches could be beneficial. Ranchers whose main income was photographic tourism had the greatest tolerance to predators. Therefore, photographic tourism based game ranches should be promoted in these buffer zones, potentially by providing easier access for neighbouring farms to conduct guided safaris into the national parks.

9.6.5 Future development of the game ranching industry

Botswana's game ranching industry has grown and developed substantially since its introduction in 1992, but during the surveys many farmers voiced concerns about the financial viability and future of the industry saying that they would potentially need to return to cattle farming (Chapter 4). Overall, game ranchers, including dual-use livestock and game-stock ranches reported significantly greater tolerance of predators than farmers farming only livestock. Therefore, a return to cattle ranching is likely to be to the detriment of both Botswana's predators and its general wildlife population.

The primary problem game ranchers cited as affecting their farm was a lack of support from the government. Farmers felt that they had not been adequately consulted in regard to the proposed new game ranching regulations, and they were concerned for the future of the industry due to the negative stance the government had taken to hunting at the time of study (Chapter 4). A lack of support outside of the wildlife ministry and a top-down approach from government, in conjunction with a lack of governance, consultation and inconsistent regulations and leadership, has been highlighted as detrimental to the industry in South Africa, Zambia and Namibia (Barnes & Jones, 2009; Brink *et al.*, 2011; Lindsey *et al.*, 2013a; Lindsey *et al.*, 2013c). Often indigenous Africans are under-represented in the industry (Lindsey, 2011; Lindsey *et al.*, 2013a), and game ranches are sometimes perceived as wasting land that could be used for agriculture (Wolmer *et* *al.*, 2004). Efforts to change these attitudes, to improve communication between stakeholders and to promote local involvement and support for the industry, will be crucial to its success and to the continuation and development of its wildlife conservation benefits.

However, development of the game ranching industry and its role in predator conservation relies on farmers taking responsibility for its future. The BWPA reported poor correspondence and involvement from its members (BWPA, 2011a), which potentially encourages a top-down approach from government departments. Adoption of an independent regulatory body and eco-certification initiative to certify and regulate hunting and game ranching operations could potentially drive the industry into utilising more ecologically friendly practices (Lewis & Alpert, 1997; Lindsey *et al.*, 2006). This regulation of undesirable and potentially illegal practices, such as the capture and sale of predators, conducted by the minority, would help to improve attitudes towards the industry as a whole.

9.6.6 Communication

One of the clearest problems that was apparent during this research was the lack of communication and understanding between government officials, NGOs and the farming community, a problem also highlighted in a global IUCN HWC workshop and in the National Predator Strategy (Madden, 2004; Winterbach, 2008). Often conservation NGOs are seen as collaborating with government, potentially reducing the opportunities for trust and cooperation (Treves, 2006) and conversely, farmers often report they feel like part of the problem rather than partners in finding the solution (Rabinowitz, 2005). Listening to the views of stakeholders has in itself been shown to reduce HWC (Weber & Rabinowitz, 1996; Sillero-Zubiri & Laurenson, 2001). Similarly, increased participation of stakeholders in decision making can lead to increased public support, enhanced credibility of the programme and is more likely to achieve its goal (Hewitt & Messmer, 1997). The instigation of regular meetings involving multiple stakeholders would contribute to finding suitable solutions to HWC conflict (Brink *et al.*, 2011; Henle *et al.*, 2013).

9.6.7 Education

A farmers' level of education is an important driver of tolerance of predators (Chapter 4) (Naughton-Treves *et al.*, 2003; Romañach *et al.*, 2011), and environmental education and promotion of animal husbandry techniques has been shown to encourage positive attitudes to predators (Marker *et al.*, 2003d; Draheim *et al.*, 2011; Strande-Straube, 2013). Therefore, it is recommended that education programmes and the distribution of educational materials should be continued. The primary reasons stated by farmers in the current study for not adopting livestock and game-stock management techniques which aim to reduce carnivore depredation were due to limitations in the farm infrastructure (often due to the size of the farm), lack of perceived effectiveness of the technique, expense (particularly game-stock techniques) or limited knowledge (Chapter 7). Therefore, it is recommended that education materials should address these perceptions and concerns within the target community.

The Botswana game ranching handbook contains little information on predators or game-stock management techniques to reduce carnivore depredation (BWPA, 2005). Game-stock management techniques were generally negatively perceived by game ranchers, often with the belief that it was not possible to stop predators (Chapter 7). However, electric and bonnox perimeter fencing and the stocking of buffer species was associated with reduced game-stock losses and increased tolerance of predators (Chapters 4 and 7). Although, 'predator-proofing' of the farm's perimeter is not ideal as it reduces and fragments the available habitat for predators, the use of electrified or bonnox fencing of breeding pens for vulnerable game-stock species could be an effective alternative. Therefore, the use of buffer species and breeding pens should be promoted and should be added to the game ranching handbook when republished.

Farmers often stated that they were reluctant to join conservancies due to a lack of understanding of the rules, regulations and advantages of conservancies (Chapter 7). Relevant information should be disseminated to game ranchers, both through traditional routes such as media and information leaflets and also through the development of a 'farmer network'. Often farmers get their farming advice, and are more likely to listen to advice, from other farmers rather than government departments or NGOs (Sligo, 2005; Isaac *et al.*, 2007). Botswana's small human population and close ties to Namibia and South Africa would enable a contact network of farmers involved in conservancies or using certain livestock or game-stock management techniques to be developed and promulgated. This would enable farmers to discuss their experiences and promote the use of these techniques from within the industry.

9.7 Recommendations for further research

This research supported the general conclusion that there is no clear evidence that PAC methods in Botswana at the time of study are reducing HPC, and they could in fact be causing other areas of wildlife management and conservation to be neglected (CAR, 2011; M. Selebatso pers. comm. cited in UNDP, 2007). The government's vast investment in providing compensation for livestock losses of ca. US $4,560,500 a year and 60% of the DWNP departments time and resources (Bowie, 2009; CAR, 2011) is not enabling either commercial (Chapter 7) or subsistence farmers (Gusset *et al.*, 2009) to coexist with predators. Predators are being frequently killed as a result of HPC, for example lions had been killed on ca. 87.0% of farms and African wild dogs on 34.0% of farms upon which they were reported to be present in the 12 months prior to survey (Chapter 7). Additionally, the non-lethal alternative of 'problem' predator translocation is costly, generally not perceived to reduce stock losses and, in the case of cheetah, has a poor chance of survival for the translocated individual (Chapter 8). Therefore, alternative options for 'problem' predator control should be investigated. A suggested alternative, the hunting of 'problem' predators was viewed by farmers as the most effective method at enabling predator-human coexistence (Chapter 7) and has been recommended in the 2012 'Green Paper' on HWC (CAR, 2012). The hunting of 'problem' lions was trialled on the Ghanzi farmlands in 2005, but it was described by a DWNP representative as an 'unsuccessful experiment' (BWPA, 2006b), due to concerns the system was abused (farming community member pers. comm.). As a result, the government has implied it would be unlikely to reinstate 'problem' predator hunting (BWPA, 2006b). Often people-orientated approaches to HWC fail due to short comings in implementation rather than because of any fundamental incompatibility with biodiversity conservation (Abensperg-Traun, 2009). 'Problem' animal hunting has been successful in reducing the number of predators killed in Zimbabwe (Child, 2005). Consequently, it is recommended that a feasibility study into 'problem' animal hunting is undertaken. The study could review existing 'problem' hunting programmes, for example the ‘predator hotline’ in Namibia, to determine their limitations and potential advantages. This would enable the Botswana government and conservation organisations to make an informed choice on the potential application of this method. Such a feasibility study could be initiated by the BWPA in collaboration with the DWNP.

Farmers’ attitudes and perceptions of predators, which commonly drive HPC (Chapters 4 and 6) are often derived from friends, peers and media (Karlsson & Sjöström, 2007). Farming magazines and national newspapers often portray a negative image of predators and conservation programmes (Houston, 2009; Bhatia *et al.*, 2013). Studies have shown mass-media can influence peoples’ attitudes to environmental issues and conservation (Stamm *et al.*, 2000); however, little research has been conducted on the portrayal of predators in African media. It is recommended that a content analysis be carried out to examine the media’s portrayal of conservation and HWC and the influence the media has on HWC in southern Africa.

9.8 General conclusion

The current study contributed to answering the gap in knowledge identified by Inskip and Zimmerman (2009) and Balme *et al.* (2014), regarding the occurrence of HPC on game ranches and the game ranching industry’s role in predator conservation. The study demonstrated that HPC does occur, and game ranchers reported financial losses significantly greater than reported by livestock farmers. However, farmers overestimated the number of game-stock killed by cheetahs compared to scat-analysis data. Overall, game ranchers exhibited more positive attitudes to predators than livestock farmers and large predators, particularly cheetahs and leopards, were detected more frequently on game ranches than livestock farms. Therefore, the game ranching industry has the potential to positively impact predator conservation.

The industry within Botswana is undergoing development induced by changes in hunting policy within the country. The outcome of these changes upon the industry and HPC should be closely monitored. Ranchers described the lack of support from the government as the primary problem to their farms and issues of governance are thought to threaten the industry across southern

Africa. A more participatory approach between NGOs, government departments and the industry will be necessary for the continued development of its conservation benefits.

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Appendix 1: Information sheet for questionnaires

University of Cape Town.
Department of Zoology

Information Sheet - Questionnaire

This survey is part of a research project on the impact of predators on private game and cattle farms in Botswana, and will form part of a PhD with the University of Cape Town. As part of this research I would like to conduct a questionnaire, on your experiences and opinions of predator conservation on farmland and the potential options that would enable farmers and predators to coexist.

I hope this survey will give game-stock farmers and cattle farmers an opportunity to voice their opinions regarding predator conservation and conflict and may impact upon future government policy.

All of your answers will be made anonymous and will remain confidential. Participant's comments, opinions or actions will not be shared with anyone and individual participant's answers will not be reported to the Department of Wildlife and National Parks or to any other governing body. Participants may leave the study at any time.

There are no benefits or risks, such as costs or payments, associated with this research.

If you can spare some time to share your experiences, I would be extremely grateful.

Many thanks

Title of research project: Exploring the causes of and mitigation options for human-predator conflict on game ranches in Botswana: How is coexistence possible?
Names of principal researchers: Les Underhill, Lorraine Boast
Department/research group address: Department of Zoology, University of Cape Town
Private Bag, Rondebosch, 7701, South Africa
Telephone: +27216503227 (Les)
+27763600492 / +255 687138253 (Lorraine)
Email: les.underhill@uct.ac.za / lboast@yahoo.co.uk
Nature of research
Participant's involvement: Completion of questionnaire interview
What's involved: Completion of questionnaire interview
Risks: None   Benefits: None
Costs: None   Payment: None

Appendix 2: Consent form for questionnaires

University of Cape Town.
Department of Zoology

Consent Form

Title of research project: Exploring the causes of and mitigation options for human-predator conflict on game ranches in Botswana: How is coexistence possible?
Names of principal researchers: Les Underhill, Lorraine Boast
Department/research group address: Department of Zoology, University of Cape Town, Private Bag, Rondebosch, 7701, South Africa
Telephone: +27216503227 (Les)
+27763600492 / +255 687138253 (Lorraine)
Email: les.underhill@uct.ac.za / lboast@yahoo.co.uk
Nature of research
Participant's involvement: Completion of questionnaire interview and/or event diary
What's involved: Completion of questionnaire interview and/or event diary

Risks:	None	Benefits:	None
Costs:	None	Payment:	None

- I agree to participate in this research project.
- I have read and understood this consent form and the attached information sheet and I have had the opportunity to ask questions about them.
- I agree to my responses being used for education and research on condition my privacy is respected.
- I understand that my personal details will be used in aggregate form only, so that I will not be personally identifiable.
- I understand that I am under no obligation to take part in this project.
- I understand I have the right to withdraw from this project at any stage.
- I agree / disagree for the interview to be recorded (please delete as appropriate).

Signature of Participant: ______________________________
Name of Participant / Guardian: ______________________________

Signature of person who sought consent: ______________________________
Name of person who sought consent: ______________________________

Signatures of principal researchers: ______________________________

Date: ______________________________

Appendix 3: Questionnaire: Farmers ranching livestock and game-stock

Date: Interviewer: Begin time: End time: Location:
GPS: S _________ E __________ Predator ID correct: 1,2,3,4,5,6,7,8 People present:

A. GENERAL

1. Game and Livestock on the same farm ☐ Game and Livestock on different farms ☐

2. Ranch size ____________ ha

3. Are you: Owner (leasehold) ☐ Owner (freehold) ☐ Lessee ☐
Manager ☐ Other ☐ (please state)____

4. Gender: Male ☐ Female ☐ 5. Year of birth: ____________________________________

6. Religion: ____________________________ 7. Ethnic group: ____________________________________

8. Highest educational level: Did not complete secondary school ☐ Secondary school graduate ☐
Undergraduate degree ☐ Post-graduate degree ☐ Other ☐ (please state) ______________________

9. How long is your total farming/ranching experience? Livestock ______ yrs. Game ______yrs.

10. How long have you owned/ worked on this ranch? ___________ yrs.

11. How often are you on the ranch?
Daily ☐ Weekly ☐ Monthly ☐ Quarterly ☐ Other ☐ (please state) ________________________

12. Select which activities you conduct on your farm/ranch and the percentage contribution they make to the ranches income?
Livestock farming ☐ _____ % Culling - Meat production ☐ _____ % Live game sales ☐ _____ %
Trophy hunting ☐ _____ % Biltong hunting ☐ _____ % Client bow hunting ☐ _____ %
Photographic tourism ☐ _____ % Other ☐ (please state_____________________) _____ %

13. If you are the owner, what percentage contribution does the ranch make to your household income? ___ %

14.a Is the ranch currently part of a conservancy (Area of land where neighbouring landowners remove internal fencing and create larger, cooperatively managed wildlife areas surrounded by a single perimeter fence)?
Yes ☐ No ☐ No, But would like to join a conservancy in future ☐
b. If your ranch is currently not part of a conservancy please explain why? _________________________________
____

15. How many of the following livestock and stocked game species are on the farm/ranch? If you do not know the exact number, please provide your best estimate.

Cattle	___	Goats	___	Sheep	___				
Springbok	___	Impala	___	Bushbuck	___	Kudu	___	Blue Wildebeest	___
Plains Zebra	___	Eland	___	Gemsbok	___	Waterbuck	___	Black Wildebeest	___
Red Hartebeest	___	Giraffe	___	Blesbok	___	Ostrich	___	Other (please state) _______________	___

16. What are your two most valuable game species? 1. __________________ 2. ___________________________

17. How often do you see small antelope species on the ranch (for example steenbok or duiker)?
Daily ☐ Weekly ☐ Monthly ☐ Quarterly ☐ Yearly ☐ Never ☐ Do not know ☐

18. Have the numbers of small antelope on the farm/ranch changed in the last ten years?
Increased ☐ Stable ☐ Decreased ☐ Do not know ☐

B. PROBLEMS AND THREATS

19. Please score the factors below out of ten, in relation to the size of the problem they cause to the game ranch. (where 1 is a small problem and 10 is the big problem.)

Disease	Insufficient Grazing	Limited government support	Losses due to predators	Theft / poaching	Unreliable market	Other (please state)
____	____	____	____	______	________	____________________

20. Please score the factors below out of ten in terms of the economic losses they cause to your livestock and game. (where 1 is a small loss and 10 is a big loss).

	Calving	Disease	Fire	Poisonous plants	Predators	Starvation	Theft /poaching	Other (please state)
Livestock	___	___	___	___	___	___	___	__________________
Game	___	___	___	___	___	___	___	__________________

21. Are the neighbouring farms/ranches/community supportive of the game ranch?
All are supportive ☐ Most of farms are supportive ☐ Indifferent ☐
Most of farms are unsupportive ☐ All are unsupportive ☐
Can you explain why? ____

C. PREDATORS

22. In the table below please select how often you see signs of each predator and non-typical predator species on your ranch (spoor or visual).

	Never	Daily	Weekly	Monthly	Quarterly	Yearly	Every few years	Other (please state)
African wild dog	☐	☐	☐	☐	☐	☐	☐	________
Brown hyaena	☐	☐	☐	☐	☐	☐	☐	________
Caracal	☐	☐	☐	☐	☐	☐	☐	________
Cheetah	☐	☐	☐	☐	☐	☐	☐	________
Jackal	☐	☐	☐	☐	☐	☐	☐	________
Leopard	☐	☐	☐	☐	☐	☐	☐	________
Lion	☐	☐	☐	☐	☐	☐	☐	________
Spotted hyaena	☐	☐	☐	☐	☐	☐	☐	________
Baboon	☐	☐	☐	☐	☐	☐	☐	________
Stray domestic dog	☐	☐	☐	☐	☐	☐	☐	________

23. How do you feel about sharing the land with predators?
Very happy ☐ Happy ☐ Neutral ☐ Unhappy ☐ Very unhappy ☐

24. In the table below please select when you would remove a predator from the ranch, in relation to the number of livestock or game losses you would be willing to tolerate during a single calving season (assumed to be a 3-month period)?

I would remove the predator…….	Livestock Farm				Game Ranch			
	at the first sign of presence	after the first cattle loss	after multiple cattle losses (please state how many?)	never	at the first sign of presence	after the first game-stock loss	after multiple game-stock losses (please state how many?)	never
African wild dog	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Brown hyaena	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Caracal	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Cheetah	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Jackal	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Leopard	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Lion	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐
Spotted hyaena	☐	☐	☐ _______	☐	☐	☐	☐ _______	☐

25. How much do you agree or disagree with each of these statements?

	Strongly agree	Agree	Disagree	Strongly disagree	Do not know
Game ranches play an important role in wildlife conservation in Botswana	☐	☐	☐	☐	☐
Game ranches play an important role in predator conservation in Botswana	☐	☐	☐	☐	☐
Game ranches have more predators than neighbouring cattle farms	☐	☐	☐	☐	☐

D. STOCK LOSSES

26. Please estimate how many livestock and stocked game you lost to the following causes in the last 12 months?

	Calving	Disease	Fire	Poisonous plants	Predators	Starvation	Theft/ poaching	Other (please state)
Cattle	___	___	___	___	___	___	___	_________________
Smallstock	___	___	___	___	___	___	___	_________________
Game	___	___	___	___	___	___	___	_________________

27. For what percentage of your losses were you able to locate the carcass? _____________ %

28. What was the estimated economic value of your losses to **predators** in the last 12 months?
Cattle BWP _________ Goats/Sheep (N/A ☐) BWP _________ Game BWP ________

29. What are the two main prey species that are killed by predators on your ranch?
1. ___________________________ 2. ___________________________ Do not know ☐

30. What are the two main prey species that are killed by cheetahs on your ranch? N/A ☐
1. ___________________________ 2. ___________________________ Do not know ☐

31. What percentage of your losses to predators was caused by each predator species?

	African wild dog	*Baboon*	Brown hyaena	Caracal	Cheetah	Jackal	Leopard	Lion	Spotted hyaena	*Stray domestic dog*
Smallstock	____ %	____ %	____ %	____ %	____ %	___%	____ %	_ %	____ %	____ %
Cattle	____ %	____ %	____ %	____ %	____ %	___%	____ %	_ %	____ %	____ %
Game	____ %	___ %	____ %	____ %	____ %	___%	____ %	__%	____ %	____ %

32. How are losses to predators identified? (tick all that apply)

Herd not increasing ☐ Mothers seen without young ☐ Spoor/signs at carcass ☐
Stock missing ☐ Other ☐ (please state) ____

E. TROPHY HUNTING OF PREDATORS

33. Have you obtained a leopard trophy hunting permit or financially benefited from leopard trophy hunting in the last 5 years? Yes ☐ (If Yes, how many times ____________) No ☐ Do not know ☐

34. Should ranchers be able to use leopard trophy hunting permits on nearby cattle farmland if they have the landowner's permission?
Yes ☐ No ☐ Do not know ☐ Please explain why? _________________________________
____

35. How do you think the leopard trophy hunting system on game ranches could be improved? ________________
____

36. Should trophy-hunting quotas exist for the following predators?
Free-ranging…… Cheetah Yes ☐ No ☐ Leopard Yes ☐ No ☐
Lion Yes ☐ No ☐ African Wild Dog Yes ☐ No ☐
'Problem' predators on cattle farms Yes ☐ No ☐ If yes, should it include female 'problem' predators Yes ☐ No ☐

F**. LIVESTOCK MANAGEMENT**

37. Please indicate which of the following livestock management techniques you currently use, have used in the past, have never used or plan to use in the future. For each activity you do not currently use, please explain why you do not currently use it (tick all that applies).

		Usage					Why I do not currently use the technique…					
		Currently use	Used in past	Plan to use in future	Never used		Too expensive	Too time-consuming	Unsure how to do	Decreases herd productivity	Ineffective	Other (please state)
CATTLE	Kraal cattle at night	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Kraal cattle that are calving	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Kraal young calves	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Guarding animals (dogs)	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Herders (stay with cattle all day)	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Breeding seasons	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
SMALLSTOCK	Kraal smallstock at night	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Kraal smallstock that are lambing	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Kraal young kids / lambs	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Guarding animals (dogs)	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Herders (stay with stock all day)	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Breeding seasons	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
BOTH	Keep records of livestock numbers, births & deaths	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Claim compensation from DWNP	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Insure livestock against stock losses from predators	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________
	Promote wild game populations	☐	☐	☐	☐		☐	☐	☐	☐	☐	______________

G. GAME SPECIES MANAGEMENT

38. Please indicate which of the following game-stock management techniques you currently use, have used in the past, have never used or plan to use in the future. For each activity you do not currently use, please explain why you do not currently use it (tick all that applies).

		Usage					Why I do not currently use the technique…						
		Currently use	Used in past	Plan to use in future	Never used		Not needed	Too expensive	Too time-consuming	Unsure how to do	Decreases game productivity	Ineffective	Other (please state)
GAME	Electric fenced breeding pens for important game species	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Electrified perimeter fence	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Bonnox fencing of perimeter	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Patrol fence (please state how often ___________)	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Perimeter fence buried underground	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Fill in holes under perimeter fence	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Swing gates in holes under perimeter fence	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Keep records of stock numbers, births & deaths	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Stock a buffer-prey species for predators to eat	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________
	Insure game against stock losses from predators	☐	☐	☐	☐		☐	☐	☐	☐	☐	☐	____________

39. Have you used/tried any other methods to prevent stock losses to predators on the farm/ranch?
Yes ☐ No ☐ (If yes, please provide details) ____
____

H. PHOTOGRAPHIC SAFARIS **(If you do NOT conduct photographic safaris move to question 43)**

40. How important are predators for the **current economic success** of photographic safaris on **the game ranch**?
N/A☐ Very important Important ☐ Unimportant ☐ Very unimportant ☐ Do not know ☐

41. Are you conducting any tours or packages specifically targeted at predator tourism? N/A ☐ Yes ☐ No ☐
If yes, please provide details ____
____

42. How could free-ranging predators play a bigger role in photographic tourism on private game ranches? _______
____

43. Should ranchers/farmers be allowed to keep captive predators permanently in DWNP approved holding facilities?
Yes ☐ No ☐ Do not know ☐

44. If yes, under what restrictions? (Please tick all that apply)

Source of predators:	Captive bred ☐	Wild Caught ☐	Wild caught 'problem' animals ☐			Orphans ☐
How should they be utilised?	Not utilised ☐	Education ☐	Photographic tourism ☐	Hunting ☐	Sell alive ☐	Other ☐ (please state___________)
Should they be allowed to breed?	No, Animals sterilised ☐	No, Keep M/F separate ☐	Yes, but offspring may not be utilised ☐	Yes, offspring may be utilised ☐		Other ☐ (please state _________ ________________)

45. Are you a member of any certification schemes? e.g. Fair trade, Botswana tourism board eco-certification
No ☐ Yes ☐ (please state) ____

I. TRANSLOCATION

46. Have you ever captured and relocated (translocated) a 'problem' predator from the ranch?
Yes ☐ No ☐ Do not Know ☐ **(If 'No' please move to question 49)**

47. If Yes, please provide further details of the most recent event.

Year	Species translocated	Who assisted with the predator removal?	Reason for removal?
____	_________	No-one ☐ Friends/Family ☐ DWNP ☐ Conservation NGO ☐	____ ____

48. If Yes, how effective was the translocation at reducing your stock losses to predators?
In the short-term (0-3 months):
Very effective ☐ Effective ☐ Ineffective ☐ Very ineffective ☐ Do not know ☐
In the long-term (>3 months):
Very effective ☐ Effective ☐ Ineffective ☐ Very ineffective ☐ Do not know ☐

49. How likely would you be to translocate a predator in the future?
Very likely ☐ Likely ☐ Unlikely ☐ Very unlikely ☐ Do not know ☐
Please explain why: ____

J. EFFECTIVENESS OF METHODS TO ENABLE PREDATORS AND RANCHERS TO COEXIST

50. How effective could these potential solutions be at enabling farmers and predators to coexist on farmland/ ranchland? (Very effective VE, Effective E, Ineffective IE, Very ineffective VIE, do not know DNK)

	VE	E	IE	VIE	DNK
Government compensation schemes for livestock losses to predators	☐	☐	☐	☐	☐
Government compensation schemes for game losses to predators	☐	☐	☐	☐	☐
Insurance schemes for stock losses to predators	☐	☐	☐	☐	☐
Increase value of predators through trophy hunting	☐	☐	☐	☐	☐
Introduce trophy hunting of 'problem' predators on cattle farms	☐	☐	☐	☐	☐
Increase value of predators through photographic safaris	☐	☐	☐	☐	☐
Change ranch management to reduce livestock losses	☐	☐	☐	☐	☐
Change ranch management to reduce game losses					
Increased sale price for meat that comes from a ranch that is certified and marketed as predator-friendly (e.g., dolphin friendly tuna; predator-friendly beef/game)	☐	☐	☐	☐	☐
Increased sale price for hunting/tourism operators that are certified and marketed as predator-friendly	☐	☐	☐	☐	☐
Join a conservancy	☐	☐	☐	☐	☐
Reduce predator numbers to 'tolerable' level	☐	☐	☐	☐	☐
Translocation (live-capture and relocation) of 'problem' predators	☐	☐	☐	☐	☐

Appendix 4: Lethal removal of predators information and question sheet

This section will discuss the shooting/poisoning of predators on farmland. The questions will be posed using a technique known as the 'randomised response technique'. This technique has been commonly used when discussing sensitive topics for which people wish to remain anonymous. A roll of the dice before you answer each question will determine if you answer 'Yes', 'No' or truthfully. This element of chance makes your response anonymous, by preventing the interviewer from knowing if you are telling the truth. However, when answers are collected from a large number of people, the data will inform the research team of the overall incidence of predator removal on farmland.

You will be given two dice and a plastic beaker in which to shake the dice. Before answering each question shake the dice. DO NOT show the interviewer the dice roll.

If the sum of the dice equals five to ten answer truthfully.

If it equals two, three or four answer 'Yes'.

If it equals 11 or 12 answer 'No'.

Here is a practice question:

Roll the dice. Depending on the sum of the numbers, answer accordingly:

2	Yes	5	Truthfully	11	No
3	Yes	6	Truthfully	12	No
4	Yes	7	Truthfully		
		8	Truthfully		
		9	Truthfully		
		10	Truthfully		

Have you visited the Central Kalahari Game Reserve? Yes ☐ No ☐

The interviewer will now read four questions. Roll the dice before each question is read out loud. Then answer 'YES', 'NO' or truthfully according to the sum of the numbers on the dice.

K. ROLL THE DICE

1. Have you ever killed a predator on your ranch? Yes ☐ No ☐

2. Have you killed any of these predator species in the last 12 months?

ROLL THE DICE

African Wild dog Yes ☐ No ☐

ROLL THE DICE

Brown hyaena Yes ☐ No ☐

ROLL THE DICE

Caracal Yes ☐ No ☐

ROLL THE DICE

Cheetah Yes ☐ No ☐

ROLL THE DICE

Jackal Yes ☐ No ☐

ROLL THE DICE

Lion Yes ☐ No ☐

ROLL THE DICE

Leopard Yes ☐ No ☐

ROLL THE DICE

Spotted hyaena Yes ☐ No ☐

ROLL THE DICE

Stray Domestic dog Yes ☐ No ☐

ROLL THE DICE

3. Have you ever killed a protected predator without reporting it to DWNP? (i.e., cheetah, African wild dog, leopard, lion, brown hyaena) Yes ☐ No ☐

ROLL THE DICE

4. Have you ever used poison to kill a predator? Yes ☐ No ☐

Appendix 5: Predator identification sheet

Appendix 6: Species included in the reference collection for prey analysis

Species in the cross-sectional hair reference collection used to identify cheetah's prey species from hair found in scat samples. The reference collection was adapted from Davies-Mostert *et al.* (2010) and Kent (2004) supplemented with additional species as required.

Common name	Scientific name	Age and Sex	Area of body hair sample taken from
Aardvark	*Orycteropus afer*	Unknown	Belly, body
African wild dog	*Lycaon pictus*	Adult, male	Body
Black wildebeest	*Connochaetes gnou*	Adult, male	Leg
Black-backed jackal	*Canis mesomelas*	Adult	Belly, body, leg
Blesbok	*Damaliscus pygargus phillipsi*	Adult, male	Leg, rump
Blue wildebeest	*Connochaetes taurinus*	Unknown	Belly, body, mane
Bushbuck	*Tragelaphus scriptus*	Adult, male	Belly, body, head, leg, mane, tail
Cane rat (greater)	*Thryonomys swinderianus*	Adult	Body
Cattle	*Bos primigenius*	Juvenile, female	Leg, rump
Cheetah	*Acinonyx jubatus*	Adult, male	Back, leg
Common duiker	*Sylvicapra grimmia*	Adult	Back, belly, head, leg, tail
Common eland	*Tragelaphus oryx*	Adult	Dewlap, head, leg, mane, rump
Donkey	*Equus africanus asinus*	Adult, male	Leg, rump
Gemsbok	*Oryx gazella*	Juvenile	Leg, mane, tail
Giraffe	*Giraffa camelopardalis*	Unknown	Body, mane
Goat	*Capra hircus*	Juvenile	Leg, rump
Greater kudu	*Tragelaphus strepsiceros*	Adult, male	Back, belly, body, face, leg, mane, neck, tail
Impala	*Aepyceros melampus*	Adult & juvenile Male & female	Belly, body, face, head, face, fetlock, leg, rump, tail
Klipspringer	*Oreotragus oreotragus*	Adult	Back, belly, head, leg, tail
Leopard	*Panthera pardus*	Adult	Unknown
Mountain reedbuck	*Redunca fulvorufula*	Unknown	Belly, body
Nyala	*Tragelaphus angasii*	Unknown	Belly, body, mane
Plains zebra	*Equus quagga*	Adult	Belly, body, leg, mane
Red hartebeest	*Alcelaphus buselaphus caama*	Adult	Back, belly, head, leg, neck, tail
Scrub hare	*Lepus saxatilis*	Adult	Body
Sheep	*Ovis aries*	Adult, female	Leg, rump
South African ground squirrel	*Xerus inauris*	Adult	Back
Springbok	*Antidorcas marsupialis*	Adult	Back, belly, head, leg, tail
Springhare	*Pedetes capensis*	Adult	Unknown
Steenbok	*Raphicerus campestris*	Adult	Back, belly, head, leg, tail
Warthog	*Phacochoerus africanus*	Adult	Belly, body, foot, leg, mane
Waterbuck	*Kobus ellipsiprymnus*	Adult, female	Face, leg, neck/body, rump, tail
Yellow mongoose	*Cynictis penicillata*	Adult	Back, belly

Appendix 7: Farmers' comments regarding the use of conflict mitigation methods and their effectiveness at enabling farmers and predators to coexist on Botswana farmland ł

List of tables:

ł Farmers often gave multiple comments, so the number of comments was potentially larger than the number of participants.

7.1 Farmers' comments regarding the use of insurance initiatives for livestock and game-stock losses incurred due to carnivore depredation and the perceived effectiveness of insurance initiatives at enabling coexistence of predators and farmers

	Insure livestock/ game-stock	**Insurance effectiveness**
Number of farmers who commented on insurance	35	35
Effective	**0**	**1**
If it worked as advertised	0	1
Ineffective	**35**	**39**
Too expensive	12	15
Only needed for expensive species or stud breeding	9	3
Too hard to prove/ implement	6	10
Not available; who will insure food	5	7
Only benefit the insurance company	3	0
Open to abuse	0	3
Not worked in South Africa	0	1

7.2 Farmers' comments regarding the perceived effectiveness of livestock and game-stock management techniques at enabling coexistence of predators and farmers

	Manage livestock	**Manage game-stock**
Number of farmers who commented on management techniques	30	24
Effective	**13**	**6**
Should try to – need to look after stock	6	2
Some aspects effective e.g., kraaling and electric fences	5	4
If staff are well trained	1	0
People used to look after their stock in past, they no longer do, but they should be	1	-
Ineffective	**20**	**21**
Nothing you can do; cannot stop predators	5	9
Cannot change livestock management on an extensive system	4	1
Too expensive	4	7
Easier said than done; if it is possible	3	3
Use best management intentions; may not be best to reduce predation	2	0
People will not change ways; tradition to kill predators	2	0
Lions should not be excluded from habitat	-	1

7.3 Farmers’ comments regarding the use of breeding camps for game-stock and perimeter-fencing on game ranches

	Breeding camps	**Electric fencing**	**Bonnox fencing**	**Underground fencing**
Number of farmers who commented on breeding camps or perimeter-fencing	54	41	13	13
Ineffective	**8**	**17**	**3**	**4**
Ineffective at reducing predation (warthog make holes under fence and predators follow)	7	9	2	4
Elephants break down fence	1	7	0	0
Game species break down fence	0	1	0	0
Prevents small game/predator movement	0	0	1	0
Farm infrastructure/ management	**8**	**26**	**9**	**6**
Too expensive	5	19	9	2
Equipment stolen/damaged	1	4	0	0
Time-consuming or difficult management	2	3	0	2
Damaged too easily	0	0	0	1
Ground too rocky	0	0	0	1
Not necessary	**37**	**7**	**1**	**3**
Not needed	29	4	1	3
Only necessary for breeding expensive game-stock species	8	1	0	0
Only necessary if elephants are in the area	0	1	0	0
Not concerned about game-stock losses to predators	0	1	0	0

7.4 Farmers' comments regarding filling holes under the fence-line, the use of swing gates in the fence-line and the stocking of buffer-prey species on the game ranch

	Fill holes under the fence-line	**Use swing gates in holes under the fence-line**	**Stock a buffer-prey species**
Number of farmers who commented on methods	22	13	11
Ineffective	**12**	**4**	**4**
Holes get opened again or new holes are formed	8	3	-
Ineffective at stopping predators gaining access or at reducing predation	4	1	1
Not happy for predators to eat any game-stock	-	-	2
Buffer species have no value to farms	-	-	1
Farm infrastructure/ management	**11**	**4**	**5**
Use tyres in holes	8	-	-
Only fill big holes to stop crawling game-stock or lions	2	-	-
Only fill holes when bring in new game-stock	1	-	-
Time-consuming or difficult management	0	2	1
Too expensive	0	1	4
Gates were too heavy animals did not use them	-	1	-
Lacking knowledge	**0**	**6**	**0**
Not heard of	0	6	**0**
Not necessary	**6**	**5**	**3**
Need to let predators move out of farm again	4	0	-
Not needed	2	5	2
Not concerned about predator losses to game-stock	0	0	1

7.5 Farmers' comments regarding the use of compensation programmes for livestock and game-stock losses to carnivore depredation and the perceived effectiveness of compensation programmes at enabling coexistence of predators and farmers

	Claim compensation	**Livestock compensation effectiveness**	**Game-stock compensation effectiveness**
Number of farmers who commented on compensation programmes	18	47	26
Effective	**0**	**5**	**7**
Better than nothing; allows farmers to restock livestock/ game-stock	-	3	4
If properly instigated	-	2	1
If government are claiming jurisdiction for predators they need to be responsible for the damage they cause	-	0	1
Should treat game-stock like cattle	-	-	1
Ineffective	**19**	**59**	**22**
Too little money; pay the same for all breeds of livestock	8	31	2
Compensation programme inefficiently organised	6	7	2
Does not cover all land uses or all predator species	3	1	0
Too difficult to prove/ claim compensation e.g. need carcass	2	5	8
Does not stop people killing predators	-	5	0
Takes too long to receive compensatory payments	-	4	0
People abuse compensation programme	-	3	5
Does not encourage people to look after stock	-	3	0
Does not address problem; there is more to human-predator conflict than financial costs	-	0	2
Should expect predators to kill game-stock – part of game ranching	-	-	2
Returns responsibility to government; game industry already too tightly regulated	-	-	1

7.6 Farmers' comments regarding the perceived effectiveness of eco-certification initiatives at enabling coexistence of predators and farmers

	Predator-friendly meat effectiveness	**Predator-friendly tourism/hunting effectiveness**
Number of farmers who commented on eco-certification	41	23
Effective	**13**	**5**
Might work for exported meat	10	0
If well branded/ marketed	3	1
May help; clients care	0	3
Would only work for tourism	0	1
Ineffective	**50**	**22**
Will not work in Botswana/ Africa	16	3
No-one would buy it; people buy cheapest option	11	1
Difficult to police	5	1
Lacks government support and management	4	2
Hunters do not care; not enough people would care; would come anyway	-	7
No ranch is predator-friendly; people will still kill predators	2	1
Will not help predators as neighbours will still kill predators	2	0
Will not happen	2	0
Would not earn more money from it; would the premium cover the losses?	1	3
Will not help communal farmers/ livestock farmers	1	2
Cannot change people's view to predators; is part of culture to be intolerant	1	1
With veterinary restrictions will not work	1	-
Game meat currently sold on too small a scale	1	-
Do not see predators	-	1

7.7 Farmers' comments regarding the perceived effectiveness of 'reducing predator number to 'tolerable' levels' at enabling coexistence of predators and farmers

	Reduce predator numbers effectiveness
Number of farmers who commented on reducing predator numbers	48
Effective	**30**
Need to keep predators populations in balance; not destroy them all	11
If necessary; if cause too much damage; target 'problem' animals	7
As part of a management plan; to utilise predators	3
Farmers ranching livestock currently do it for game ranchers; many predators are killed and not reported	3
Should be no predators on farmland	2
By translocation only	1
Could be effective, but everyone would need to agree; some farms protect predators	1
Need to in communal areas; predators can kill total livelihood of people	1
Farmers are the best people to know how to keep predator numbers in balance	1
Ineffective	**34**
What is 'tolerable'; tolerable means different things to different people	10
Nature controls predator numbers	6
Predators not that big a problem	6
Killing predators causing them to breed more; other predators will move into the territory; can make problem worse	5
Not many or not enough predators	4
How monitor/ regulate it?	1
Too time-consuming	1
Would result in total destruction of all predators	1

7.8 Farmers' comments regarding joining a conservancy and the perceived effectiveness of conservancies at enabling coexistence of predators and farmers

	Join a conservancy	**Conservancies effectiveness**
Number of farmers who commented on conservancies	40	25
Do not have suitable neighbours	**34**	**9**
Different management interests to neighbours	18	7
Too much politics with neighbour; neighbours are difficult to work with	7	2
Neighbours not interested	7	0
Neighbours will only sell or lease farm for large sums of money	2	0
Prefer to work separately	**5**	**3**
Have a large area myself or plans to expand	2	1
Want to be my own boss, with no-one else to report to	2	0
Make more money farming separately	1	2
Rules and regulations	**8**	**1**
How do you define who owns game-stock	4	0
No legal status for a conservancy in Botswana; government restrictions prevent their formation	2	1
Too many rules and regulations	1	0
Limitations	**10**	**6**
Not feasible in area; limited by water availability/ location	4	1
Needs someone to drive the project	2	0
Limited by funding/ investors	2	0
Game industry in bad state/ has no future	2	0
Cattle on farm makes it difficult	1	1
Do not own the land	1	0
Would increase predators and predation	0	2
Needs to be a big area/ working well	0	2
Non-specific reasons	**4**	**7**
Not interested	3	1
Should be mandatory to join; should work together; will work	0	6

7.9 Farmers' comments regarding kraaling cattle at night, kraaling pregnant cows during calving and kraaling young calves

	At night	During calving	Young calves
Number of farmers who commented on the kraaling of cattle	37	25	18
Causes negative outcomes	**23**	**10**	**18**
Cattle graze better at night	14	-	-
Disrupts calf growth or decreases productivity of cattle herd	7	3	2
Ineffective at reducing carnivore depredation or increases depredation	3	4	8
Increases disease and stress in cattle	0	3	7
Causes mothers to abandon calves	-	-	1
Farm infrastructure/ management	**14**	**15**	**4**
Not possible; farm is too large – cattle must walk long distances	6	6	3
Not possible; necessary to work with and check the cattle during the day	5	-	-
Time-consuming or difficult management	2	5	1
Have too many cattle	1	0	0
Staff unreliable or too few	0	3	1
Do not have necessary farm infrastructure	0	1	0
Not necessary	**4**	**3**	**1**
Only necessary for stud breeders	2	2	0
Not losing enough livestock to predators to necessitate it	2	1	1
Non-specific reasons	**4**	**3**	**1**
Just the way we have been farming forever	1	0	0
Unnatural to kraal cattle	0	0	1

7.10 Farmers' comments regarding using livestock guarding dogs to guard cattle and smallstock

	Cattle	Smallstock
Number of farmers who commented on using livestock guarding dogs	21	8
Causes negative outcomes	**9**	**4**
Ineffective at reducing predation	6	0
Disturbs cattle/smallstock	3	0
Farm infrastructure/ management	**5**	**4**
Staff use dogs to hunt with	2	3
Time-consuming or difficult management	1	0
Suitable dog is not available	1	0
Difficult to distinguish livestock guarding dogs from stray dogs	1	0
Have herders instead	0	1
Lacking knowledge	**3**	**0**
Not sure how to do	2	0
Had not thought of using	1	0
Disciplinary problems	**2**	**4**
Dogs do not stay with the herd	2	1
Puppy plays with calves/ smallstock	0	1
Dogs killed calves/ smallstock	0	1
Dogs refused to guard sheep	-	1
Not necessary	**2**	**1**
Not losing enough livestock to predators to necessitate it	2	1

7.11 Farmers' comments regarding using herders to protect livestock or on using livestock breeding seasons

	Herders		Breeding seasons	
	Cattle	**Smallstock**	**Cattle**	**Smallstock**
Number of farmers who commented on using herders or livestock breeding seasons	14	7	14	5
Causes negative outcomes	**8**	**3**	**1**	**1**
Ineffective at reducing predation	7	2	1	1
Decrease herd productivity	1	1	1	1
Farm infrastructure/ management	**8**	**6**	**4**	**2**
Too expensive	3	0	0	0
Farm is too large	2	0	2	0
Staff unreliable or too few	1	5	0	0
Time-consuming or difficult management	1	0	1	0
Cattle graze at night	1	-	-	-
Nowhere to isolate bulls/ ram	-	-	1	1
Prefer to control breeding to 20 at a time	-	-	0	1
Not necessary	**0**	**1**	**5**	**1**
Have a livestock guarding dog instead	0	1	-	-
Smallstock is a hobby	-	0	-	1
Breeding peaks naturally anyway	-	-	4	0
Why waste the rest of the year	-	-	1	0

7.12 Farmers' comments regarding the perceived effectiveness of photographic tourism operations to enabling coexistence of predators and farmers

	Photographic tourism
Number of farmers who commented on photographic tourism	22
Effective	**3**
Captive predators might help	2
Predators are photographic tourism draw card, through visuals, tracks or sounds	1
If neighbours are photographic too (habituated predators get shot on nearby farms)	1
Ineffective	**22**
Too difficult to see predator; guests do not want to see tracks	8
Difficult to do in Botswana; only feasible in prime viewing areas	6
Not financially viable	4
Not worked on other ranches	1
Too expensive to keep predators	1
You will still have cattle losses to predators	1
Difficult to do tourism if you also have cattle	1

7.13 Farmers' comments regarding potential ways to improve predators' role in photographic tourism

	Predators' role in photographic tourism
Number of farmers who suggested methods	40
Habituate predators to improve sightings	8
Walking safaris to track predators	6
View predators at den-sites or marking-sites	6
Keep captive predators	5
Increase predator numbers	4
Bait predators	4
Form conservancies or bigger farms	4
Introduce predators to reserves with 'predator-proof' fencing	3
Involve tourists in predator research or volunteer program	3
Cultural tourism – educate guests about human-wildlife conflict and community projects	2
Increase advertising of predators	2
Use motion cameras to view predators	1
Collar or microchip predators to track them	1
Do not know, very difficult as cannot guarantee visitors will see predators	7

7.14 Farmers' comments regarding predator trophy hunting and 'problem' predator hunting and the perceived effectiveness of predator trophy hunting and 'problem' predator hunting at enabling coexistence of predators and farmers

Comments	**Trophy hunting on livestock farms**	**Hunt 'problem' animals**	**Trophy hunting of predators effectiveness**	**Hunting of 'problem' predators effectiveness**
Number of farmers who commented on trophy hunting of predators and hunting of 'problem' predators	89	57	20	22
FOR - Benefit predator populations	**47**	**20**	**5**	**9**
By giving predator a value it will increase peoples' tolerance of predators i.e. people will look after predator species, if they can gain money from that species in the long run	21	5	4	3
Should only hunt 'problem' animals, then it will not be necessary to shoot 'non-problem' predators on game ranches, but need proof it is a 'problem' animal	19	13	1	5
Predators will be killed more responsibly than if they were being killed for 'problem' animal control	7	2	0	0
Hunters aware of preserving wildlife; will not shoot too many; will maintain balance	0	0	0	1
FOR - Benefit people	**47**	**16**	**3**	**3**
Livestock farmer/ community benefit from predator too; provide financial compensation for stock losses	22	6	2	2
Need to control predator numbers, hunting is a way to do it	9	5	1	0
Would reduce cattle losses	5	0	0	0
Provide income to government	4	4	0	1
Going to keep being a 'problem' animal; therefore, need to remove it	3	1	0	0
Cannot find leopard on own farm so would improve chances of hunting a leopard	2	0	0	0
Would improve relations between cattle farmers and game ranchers	2	0	0	0
FOR - Other	**24**	**10**	**1**	**1**
Leopard is going to be killed as a 'problem' animal anyway, so may as well hunt it	16	9	1	1
Leopard utilise both livestock farms and game ranches within its territory	7	0	0	0
Neighbours shooting so many predators anyway, so why should I not shoot one	1	0	0	0
Current approach to 'problem' animal control is not working; should try hunting	-	1	0	0

	Trophy hunting on livestock farms	Hunt 'problem' animals	Trophy hunting of predators effectiveness	Hunting of 'problem' predators effectiveness
AGAINST	**23**	**33**	**6**	**9**
Against all leopard hunting; it may be the last one	6	1	0	0
Catching and selling live predators or translocation of 'problem' predators would be better	4	6	0	0
Open to corruption and abuse of system; every predator will become a problem	2	12	2	5
If remove a predator another one will move in	2	2	0	0
Shooting out good genetics in population	2	0	0	0
Trophy hunting does not increase tolerance of predators; still shoot leopard for 'problem' animal control	2	0	2	1
'Problem' animals do not exist; should improve farming techniques instead	1	5	0	0
As a last resort only	1	1	0	1
Need more information about leopard populations first	1	0	0	0
Leopards keep cheetahs off the farm, therefore, should value leopards	1	0	0	0
Leopard hunting is not fair; bating is not hunting	1	0	1	0
'Problem' animals should not be a business; only farmer or Department of Wildlife and National Parks officer should shoot a 'problem' animal; money and nature do not mix	0	4	1	2
Predators not that big a problem	0	2	0	0
Criticisms or suggestions to change hunting system	**7**	**15**	**5**	**5**
Not much money goes to community; money should go in a community kitty not to an individual	3	3	1	0
Need higher quota; to induce tolerance people need to benefit more regularly	2	0	1	0
Was not well regulated in past, would need strict controls	1	6	0	2
Difficult for livestock farmers to benefit; livestock farmers should be able to get permits	1	1	2	1
The quality of the hunt would be reduced	0	2	0	0
If get 'problem' animal licence then farmer should not claim compensation for stock losses	0	2	0	0
Difficult to find client; would need to be on standby	0	1	1	1
Should translocate 'problem' animals to game ranches to hunt	0	0	0	1
Need to be on community level as predators move onto neighbouring farms	0	1	0	0

Appendix 8: Farmers' comments regarding the translocation of 'problem' predators and its effectiveness at enabling farmers and predators to coexist on Botswana farmland.

	Translocation
Number of farmers who commented on translocation	129
Motivation for translocation	
Have to translocate predators as not allowed to kill them; no other way to stop predators	5
Department of Wildlife and National Parks (DWNP) decide whether to translocate	2
Predators should be conserved for the future/ tourism	8
Remove the problem; stop them killing cattle/ game-stock	23
Too many predators on farmland; Predators should not be on farmland	4
Better than shooting/ killing it; Gives the predator at least a chance to survive	21
Translocation good public relations campaign	1
Specific species only	5
Do not want to translocate; happy with predators on farm; do not have many predators	26
Happy with predators if there is a hunting quota	3
Effectiveness/ Limitations	
DWNP, conservation organisation refuses to help to translocate a predator; cannot capture yourself; Farmers should be able to translocate predators themselves	10
DWNP do not respond quickly; treat the translocated predator inhumanely	13
Too many predators for translocation to work	4
Predators not moved far enough away; must be moved far away	8
Nowhere to move predators to; territories already full; need to move predator to an area it will survive	10
Predators too hard to catch	5
Expensive to translocate	1
Outcome on farm	
Predators come back and continue killing livestock/ game-stock	21
Predators did not return to farm	6
After translocation more predators appear on farm but do not know if it is the same one or different ones	6
Another predator moves into the area	9
Stock losses continue by other predators	8
Stock losses or the predator problem increases due to translocating a predator	5
Reduces stock losses	4
Stock losses or the predator problem increases due to translocating a predator	5
Gives farmer a break - effective for a short while	3
Disrupts territory - minimising predators destructive behaviour	1
Outcome to predator	
Translocated predators do not survive; shot on farmland or killed by resident predators	16
Predator not collared so do not know what happens to them; should be collared	5