AN EVALUATION OF LARGE CARNIVORE TRANSLOCATIONS INTO FREE-RANGE ENVIRONMENTS IN NAMIBIA

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In memoriam

Lea Weise, 1938 – 2011
Hanni Bäzol, 1924 – 2015

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A - Acknowledgements

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B – Dissemination of study findings

B.1 Peer-reviewed articles

Based on this work, the following peer-reviewed articles have been published:

1. Weise, F.J., Stratford, K.J. and van Vuuren, R.J. (2014) 'Financial Costs of Large Carnivore Translocations – Accounting for Conservation.' *PLoS ONE,* 9(8) e105042.

2. Weise, F.J., Lemeris Jr., J., Stratford, K.J., van Vuuren, R.J., Munro, S.J., Crawford, S.M., Marker, L.L. and Stein, A.B. (2015) 'A home away from home: Insights from successful leopard (*Panthera pardus*) translocations.' *Biodiversity and Conservation,* 24(7) pp. 1755-1774.

3. Weise, F.J., Wiesel, I., Lemeris Jr., J. and van Vuuren, R.J. (2015) 'Evaluation of a conflict-related brown hyaena translocation in Central Namibia.' *African Journal of Wildlife Research,* 45(2) pp. 178-186.

4. Weise, F.J., Lemeris Jr., J., Munro, S., Bowden, A., Venter, C., van Vuuren, M. and van Vuuren, R.J. (2015) 'Cheetahs (*Acinonyx jubatus*) running the gauntlet: an evaluation of translocations into free-range environments in Namibia. *PeerJ*, 3, e1346.

B.2 Workshop presentations

On 28th-29th August 2014, the author presented the results contained in this thesis to the Department of Wildlife and National Parks of the Republic of Botswana. The aim of this Carnivore Translocation Workshop was to review translocation as one of the available management approaches in the country's Predator Management Strategy. The workshop was held at Khutse Kalahari Lodge in Botswana.

C – List of acronyms and abbreviations

AICc – corrected Akaike Information Criterion

AUC – Area Under the Curve

CBSG – Conservation Breeding Specialist Group

CI – Confidence Interval

DNA - Deoxyribonucleic Acid

GPS – Global Positioning System

GIS – Geographic Information System

ICC – Individual Conservation Cost

IUCN – International Union for Conservation of Nature and Natural Resources

KDE – Kernel Density Estimation

Max – Maximum

MCP – Minimum Convex Polygon

Min – Minimum

NGO – Non-Governmental Organisation

NP – National Park

NRNR – NamibRand Nature Reserve

PA – Protected Area

RMSE – Root Mean Square Error

SD – Standard Deviation of the Mean

SE – Standard Error of the Mean

UNICEF – United Nations International Children's Emergency Fund

US – United States of America

USD – U.S. Dollar

VHF – Very High Frequency

ZAR – South African Rand

D – Abstract

Around the world, large carnivores are involved in human wildlife conflict by killing livestock or compromising peoples' safety. This results in widespread lethal persecution that contributes to carnivore population declines. Alternatively, translocation of so-called 'problem animals' is an often-used approach to resolve conflict non-lethally. However, translocations are rarely assessed in terms of their capacity to reduce conflict or their biological and financial implications. This study evaluates the efficacy of this strategy by investigating 22 intensively monitored translocations that were carried out into free-range environments in Namibia between 2008 and 2012. Translocations involved 23 cheetahs (*Acinonyx jubatus*) (plus 10 dependent offspring), six leopards (*Panthera pardus*) and one brown hyaena (*Parahyaena brunnea*). Translocation objectives included conflict mitigation and the rehabilitation of confiscated, indiscriminately trapped or orphaned individuals. Study animals were released at an average distance of 404.3 km (47 – 816 km) following captive periods ranging from 1 – 1,184 days. Using survival, livestock predation and homing as key measures of translocation success one year post-release, out of 27 individuals with known outcomes, 44.4% were successful. Success was higher for leopards (67%) than for cheetahs (40%), which were particularly unsuccessful if they habituated to human presence during prolonged captivity. Human-induced mortality accounted for most deaths in year one (58%; 10 cheetahs, one leopard, and the hyaena). Translocation success did not differ significantly by sex (M: 39%; F: 50%) or between hard (47%) and soft releases (40%). Regardless of species, release mode and recipient area size, all carnivores displayed extensive post-release movements ranging beyond protected area boundaries, but only two cheetahs returned to their capture site. Most animals that survived the first year successfully reproduced (five leopard cubs, 14 cheetah cubs) and settled into permanent ranges. Following an initial period of orientation and exploration, the ecology of translocated large carnivores reflects that of resident conspecifics. Only three case studies resulted in post-release conflict, but translocations did not resolve conflict on source properties permanently, leading to repeat requests for carnivore removals by those land managers. A farmland survey (26,090 km^2, n = 221 respondents) demonstrated that conflict with, and persecution of, large carnivores remain widespread, suggesting a high potential demand for carnivore translocations. However, release area suitability modelling across Namibia's protected area network showed that only a few public or private reserves can potentially accommodate individual cheetahs and leopards. Translocations are also costly, with a total expense of \$80,681 in this study (\$269 – \$7,559 per individual). The main cost factor was tracking technology (56%). Adjusted to account for failed events, the successful translocation of one large carnivore cost $5,983 (adjusted median) and 65% of all costs were recuperated from public support. Translocations can successfully conserve individual carnivores and help supplement low-density populations locally. However, due to its limited success, associated costs, and a high degree of variability in terms of outcomes, the strategy is not a feasible standard response to human-carnivore conflict. It is best reserved as a last-resort tool for the selective management of few individuals from highly endangered species. Where it is necessary, rigorous candidate and recipient area selection can improve outcomes significantly. Wildlife managers should predominantly aim at improving tolerance of large carnivores in unprotected, multi-use landscapes, thereby reducing the number of indiscriminately captured animals.

Chapter 1 Introduction

1.1 Large carnivores in a transforming world

The world of large terrestrial carnivores, i.e. those with a mean adult body mass of at least 15 kg (Linnell et al., 2012; Ripple et al., 2014), is a changing one. Since the onset of industrialisation, humans have manipulated most of Earth's terrestrial ecosystems significantly (Vitousek et al., 1997). To meet the rising demands of an increasing human population, more and more landscapes are converted, and their value to wildlife reduced through agricultural and industrial developments, urbanisation, and pastoralism (Barbaut and Sastrapradja, 1995; Vitousek et al., 1997). Primarily due to their low reproductive rates and high spatial and energetic requirements (Woodroffe and Ginsberg, 1998; Ripple et al., 2014), large carnivores appear to be particularly vulnerable to, and disproportionately affected by, human-induced landscape changes (Gittleman et al., 2001; Karanth and Chellam, 2009; Carbone et al., 2011; Crooks et al., 2011).

As a consequence of habitat transformation, reduced prey availability, disease, consumptive utilisation, trade and direct persecution, most large carnivores have suffered dramatic population and range declines around the world (Weber and Rabinowitz, 1996; Ripple et al., 2014). For example, it is estimated that the jaguar's (*Panthera onca*) distribution contracted by ~46% in the 20th century alone (Sanderson et al., 2002). In Asia, the range of wild tigers (*P. tigris*) decreased by ~41% in only one decade (Dinerstein et al., 2007) and three subspecies have gone extinct in the last 70 years (Weber and Rabinowitz, 1996). Others, like the Amur leopard (*P. pardus orientalis*), remain critically endangered (Henschel et al., 2008). Similarly, in Africa, the distributions of the continent's eight indigenous large carnivore species (half of which are endemic) have been reduced by an average of more than 60% in the last 100 – 150 years and only generalists like the leopard (*P. p. pardus*) or the hyaenas retain more than 50% of their historic ranges (Ray et al., 2005). Even the most iconic of species, the African lion (*P. leo*), is not immune to these trends with several populations facing imminent local extinction (Riggio et al., 2013; Henschel et al., 2014; Bauer et al., 2015).

Researchers are only beginning to understand the consequences of continuing carnivore declines, but their large-scale disappearance is believed to have detrimental effects on a variety of biodiversity and ecosystem functions (Ripple et al., 2014). Although there is dispute over the regulatory effect of large carnivores on herbivore populations (reviewed by Gandiwa, 2013), it is widely agreed that they are integral parts of biological communities because their activities either directly or indirectly shape ecosystem processes through cascading effects that influence various trophic levels (Miller et al., 2001; Terborgh et al., 2001; Ale and Whelan, 2008; Colman et al., 2014). For instance, removal of large carnivores can induce meso-predator release (Brashares et al., 2010) and these causal relations may only become apparent after extirpations have occurred (Estes et al., 2011). It has also been proposed that substitute apex control mechanisms such as human harvest of wildlife probably cannot replace their effect on entire species-interaction networks (Ordiz et al., 2013; Ripple et al., 2014). Beyond their ecological significance, large carnivores also play important roles in human culture (Sekhar, 1998; Ikanda and Packer, 2008; Montgomery, 2008) and are appreciated for their consumptive and non-consumptive economic tourism value (Stander et al., 1997a; Sillero-Zubiri et al., 2004; Lindsey et al., 2006; Packer et al., 2009; Jorge et al., 2013; Braczkowski et al., 2015). As for any other wildlife, many of the benefits that large carnivores provide to humans and ecosystems may not have been recognised yet (Gascon et al., 2015), and whether or not they can be identified, there is also an intrinsic argument for large carnivore conservation (Ray et al., 2005).

1.2 Human-carnivore conflicts

Around the world, there is growing recognition of the fact that protected areas (PAs) alone will not be enough to maintain or contain sufficient numbers of large carnivores to ensure their long-term perseverance (e.g. Woodroffe and Ginsberg, 1998; Breitenmoser et al., 2001, 2005; Groom et al., 2014). If they are to persist as connected, self-sustaining and largely un-managed populations, many species will in the future have to be conserved in developed, human-dominated, multi-use landscapes (e.g. Breitenmoser et al., 2012; Athreya et al., 2013; Lindsey et al., 2013a; Swanepoel et al., 2013). However, it is well established that carnivores frequently pose problems to people (in terms of safety – e.g. Treves and Naughton-Treves, 1999; Karanth and Gopal, 2005; Packer et al., 2005) and their livestock (in terms of depredation – e.g. Baker et al., 2008) in areas where their activities overlap (Woodroffe et al., 2005a; Loveridge et al., 2010). Such conflicts occur world-wide and involve all of the extant terrestrial large carnivore species (reviewed by Inskip and Zimmermann, 2009; Linnell et al., 2012). Whilst livestock losses may generally be low in relation to livestock holdings (Graham et al., 2005; Baker et al., 2008), they vary locally and can become economically significant in rural areas and where people primarily depend on income from domestic animals (e.g. Mishra, 1997; Jackson and Wangchuk, 2001; Woodroffe et al., 2005b; Wang and Macdonald, 2006; Li et al., 2013; Aryal et al., 2014). The attitudes of both commercial and communal land users outside PAs are particularly negative towards carnivores when compared with other damage-causing wildlife (Kansky et al., 2014). Unsurprisingly, human intolerance of perceived and actual threats results in retaliatory and preventative persecution of large carnivores, which is one of the main drivers of their global demise (Woodroffe, 2000; Gittleman et al., 2001; Ray et al., 2005).

During much of the 20th century large carnivores were still classified as vermin in many countries and hence were lethally controlled in substantial numbers (i.e. in their hundreds or thousands annually) to protect lives and property (Mascarenhas, 1971; Childes, 1988; Herne, 2001; Creel and Creel, 2002; Berger, 2006; Hughes, 2013). At the extreme end of the scale, Athreya (2007) reports a cumulative removal of about 65,000 tigers (1875-1925) and over 100,000 wolves (*Canis lupus*) (1871-1916) in only 50 years in India. Government-supported eradication campaigns entailed use of trapping mechanisms, hounds, rifles, as well as poison to kill carnivores and in many countries head-bounties were paid to stimulate public participation (Hazell, 2001; Allen and Fleming, 2004; Riley et al., 2004; CBSG of the IUCN et al., 2006; Lindsey et al., 2013a). Despite these drastic measures and declining carnivore populations, human – carnivore conflicts have continued to exist (Berger, 2006; Zimmermann et al., 2010; McManus et al., 2014), may in fact be increasing in many areas (Treves and Karanth, 2003; Dickman, 2010; Pettigrew et al., 2012), and remain one of the most difficult conservation challenges of our time.

1.3 Large carnivore translocation as a conservation tool

In response to persistent conflicts, conservation managers have been tasked with the development of suitable mitigation strategies in order to increase tolerance of potentially harmful carnivores and to prevent further unsustainable removal (Treves and Karanth, 2003). With growing public sentiment towards wildlife (particularly for threatened and charismatic species), there has also been a paradigm shift away from lethal control methods and the responsible agencies are increasingly forced to adopt what is considered more humane management of so-called ‘problem animals’ (May, 1996; Craven et al., 1998; Linnell et al., 1999; Witmer and Whittaker, 2001; Massei et al., 2010).

As one of the available options, live translocation of endangered animals and those involved in conflict has rapidly become a standard mitigation strategy in Africa, North America and Asia (Hamilton, 1981; Gunther, 1994; Linnell et al., 1997; Athreya, 2006; Fontúrbel and Simonetti, 2011). Today the translocation of large carnivores is a commonly used conservation tool. Translocation (or relocation) describes the technical process of deliberately moving an organism from its source site to a recipient site that may be within its extant or historic range (Miller et al., 1999; Fontúrbel and Simonetti, 2011), a novel but suitable environment (Müller and Eriksson, 2013), or captivity. The specific objectives for its strategic use in large carnivore management differ (Massei et al., 2010). They typically include, but are not limited to, augmentation of isolated genetic stock (Trinkel et al., 2008), local recovery of critically endangered populations (Belden and Hagedorn, 1993), range expansion (Skinner and van Aarde, 1987), restoration of populations through reintroduction (Hayward et al., 2007a; Gusset et al., 2008), mitigation of attacks on people or livestock (Goodrich and Miquelle, 2005), rehabilitation of individuals (Houser et al., 2011; Pop et al., 2012), control of confined populations (Davies-Mostert et al., 2009; Trinkel et al., 2010), or any combination thereof. In South Africa, translocation has become the key mechanism to manage an entire African wild dog (*Lycaon pictus*) meta-population (Gusset et al., 2009a; Davies-Mostert et al., 2015).

In terms of conflict reduction, the rationale is that offending animals would either be deterred enough by a negative experience to refrain from future offences or be moved sufficiently far into an area where conflict is less likely to happen, thus potentially alleviating friction in a non-lethal and publicly acceptable manner (Linnell et al., 1997; Craven et al., 1998). Although the technical details of translocations carried out for different purposes do not significantly differ, the conceptual framework and considerations involved often do (see Table 1.1 from Massei et al., 2010). Ad hoc conflict translocations usually involve smaller biological units (i.e. individuals or small groups) than do reintroduction efforts and they are also frequently carried out into uncontrolled/unobstructed environments that are selected when the need arises as opposed to strategically identified recipient areas. Regardless of their purpose, a wide range of methodological, environmental, species-specific and population-level factors influence translocation outcomes (Wolf et al., 1998).

Particularly in a context of human – carnivore conflict, the effectiveness of translocations has attracted serious criticism and has been reason for much debate (Miller et al., 1999; Fontúrbel and Simonetti, 2011). The main issues of concern include extensive exploratory movements resulting in low site fidelity or successful homing, compromised survivorship and reproduction, resumption of conflict at source and recipient sites, an individual's response to trapping and associated stresses, high cost and resource demands, social effects on source and recipient populations as well as potential genetic dilution and pathogen transfer (summarised from Linnell et al., 1997; Miller et al., 1999; Massei et al., 2010; Fontúrbel and Simonetti, 2011). These are further confounded by the question as to whether the responsible offender has indeed been moved.

Table 1.1 - Main differences between conflict and conservation translocations.
Source: Massei et al. (2010).

Translocation for conservation	**Translocation to solve human–wildlife conflicts**
Endangered species or populations	Common or abundant populations
Limited number of individuals	One individual up to large numbers of animals
Receiving area with or without conspecifics	Receiving area often with conspecifics
Little competition from conspecifics	Likely competition from conspecifics
Potential increase of biodiversity of receiving area	Nil or minimal impact on the biodiversity of the receiving area
Modelling population viability is essential	Modelling population viability not essential
Genetic implications essential	Genetic implications less important
Animals must establish self-sustaining population	Animals should integrate with local population
Few alternative population management options	Several alternative population management options (including lethal control)

Despite its widespread use, there appears to be little consensus on the efficacy of this strategy (Fontúrbel and Simonetti, 2011). In some situations, translocation has achieved the desired objective of rescuing threatened large carnivores from conflict situations (e.g. tiger, in Goodrich and Miquelle, 2005; Miller et al., 2011) and occasionally has been proposed as a suitable management strategy for conflict carnivores (e.g. lion, Stander, 1990a; black bear (*Ursus americanus*), Armistead et al., 1994; Alldredge et al., 2015). Other scenarios only temporally delayed conflict (leopard, Stander et al., 1997a) or lethal control (leopard, Weilenmann et al., 2010) and, in extreme cases, may have resulted in increased human mortalities around recipient areas (leopard, Athreya et al., 2007, 2011). Overall success rates (estimated at <50%) are considered low (Linnell et al., 1997; Fontúrbel and Simonetti, 2011) leading some reviewers to suggest that translocation merely has cosmetic value (Craven et al., 1998; Miller et al., 1999) and may be similar to lethal control in terms of its ultimate outcomes (Miller et al., 1999; Fontúrbel and Simonetti, 2011).

Rigorous assessment of ad hoc conflict and rehabilitation translocations (not those carried out for reintroduction purposes), however, usually is hindered by a significant lack of monitoring, reporting of events, and published results (Linnell et al., 1997; Fischer and

Lindenmayer, 2000; Massei et al., 2010). The disparity between management activities (or reality) and monitoring effort is probably best illustrated by a review of conflict leopard translocations in Kenya where over 100 problem animals were moved prior to any follow-up of individuals (Hamilton, 1981). In comparison with reintroduction programmes, the common occurrence of ad hoc translocations implemented under uncontrolled free-range conditions remains under-represented in the available literature (Fischer and Lindenmayer, 2000). Whilst an entire conservation discipline has evolved to research and improve reintroduction biology, resulting in a sharp increase in peer-reviewed publications since the early 1990s (Seddon et al., 2007), peer-reviewed conflict translocation evaluations are few and far between (see Table 1.2 for African large carnivores). The only notable exception to this rule is the study of nuisance bear management in North America, totalling over 100 published accounts in the last 40 years (e.g. McArthur 1981; Miller and Ballard, 1982; Rogers, 1986a; Brannon, 1987; Fies et al., 1987; Armistead et al., 1994; Riley et al., 1994; Shivik et al., 2011; Hopkins and Kalnowski, 2013; Alldredge et al., 2015). In the meantime, reintroduction biology has moved from retrospective, deductive studies towards progressive and hypothesis-driven science (Wolf et al., 1998; Seddon et al., 2007; Hayward and Somers, 2009), but the general understanding of conflict translocations is still in its infancy.

Table 1.2 - Key publications with detailed biological and conflict outcomes of free-range translocations involving African large carnivores.

Table does not consider structured reintroduction/rehabilitation studies.

Species	Country	Sample size	Source
Lion			
	Rhodesia (Zimbabwe)	7	Van der Meulen, 1977
	Namibia	69	Stander, 1990a
Leopard			
	Kenya	10	Hamilton, 1981[a,b]
	Namibia	6	Stander et al., 1997a
	Botswana	4	Weilenmann et al., 2010
Cheetah			
	Botswana	39 (11 monitored)	Boast et al., 2015

[a] Leopards referred to in Cobb (1981) are those reported in Hamilton (1981).

[b] Report is not publicly accessible.

Further impeding a meaningful evaluation of efficacy, very few studies have attempted to document translocation costs (Fischer and Lindenmayer, 2000; Fontúrbel and Simonetti, 2011) or determined whether the core problem of conflict has been addressed (Linnell et al., 1997; Massei et al., 2010). Where efforts have been published, detailed accounts are usually limited to a few case studies for each species (e.g. tiger, Goodrich and Miquelle, 2005; leopard, Weilenmann et al., 2010; Table 1.2) and monitoring rarely lasted longer than a few weeks or months post-release, thus probably masking both failures and successes (Hamilton, 1981). Frequently, studies report dozens of events but only provide partial or summarised assessments that lack crucial biological or conflict parameters (e.g. southern African cheetahs (*Acinonyx jubatus jubatus*), Marker et al., 2003a; Purchase et al., 2006; Boast et al., 2015). Other accounts mention translocations but provide no detail of their outcomes (e.g. Balme et al., 2009). In many cases the true outcomes are unknown. Our lack of knowledge can in part be attributed to the high failure rates of early monitoring technologies and the fact that tagged carnivores could not be followed reliably after release into unobstructed landscapes (Hamilton, 1981). With recent advances in monitoring technology though, notably through GPS-technology (Kays et al., 2015; Wilmers et al., 2015), more reliable investigations are possible (e.g. Weilenmann et al., 2010).

The variable circumstances, non-standardised objectives, and small sample sizes inherent to ad hoc carnivore translocations make it difficult to assess events as structured scientific experiments. Due to their symptomatic and often haphazard implementation, ad hoc translocation scenarios almost by default do not satisfy the desired research standards in terms of survey and sampling designs (Hayward et al., 2015). This means that evaluation frameworks need to be realistic and reflect the conditions that influence assessment criteria, as has been proposed for reintroduction translocations (Hayward et al., 2007b). In these situations, thorough documentation and evaluation of case studies can contribute important insights (Weilenmann et al., 2010) and help improve translocation protocols (Stander, 1990a; Houser et al., 2011). Moreover, pooled experiences from exercises carried out at different sites and times can potentially provide sufficient cumulative data to inform conservation management decisions for a variety of carnivores (e.g. synopsis of large carnivore reintroductions - Hayward and Somers, 2009), thus maximising the value from opportunistic observations and experience-based information (Fazey et al., 2006; Gusset et al., 2008). With a specific view on ad hoc translocations into free-range environments, and considering that a range of highly adaptable and opportunistic species are involved, the question arises as to whether the existing amount of data permits a representative judgement of the strategy's efficacy. In a recent analysis of the subject, there was a 4:1 ratio of original research articles to reviews (Fontúrbel and Simonetti, 2011).

1.4 Outlook and opportunities

The prospects of large carnivore conservation hinge upon an interdependent network of political, socio-economic and ecological factors (Winterbach et al., 2013; Ripple et al., 2014). Where conflict occurs, the rationale of large carnivores as flagships for biodiversity conservation by itself will not suffice to justify and motivate their free-range conservation (Linnell et al., 2000). Therefore, wildlife managers depend on a variety of available strategies to mitigate conflict in a case-sensitive as well as pragmatic fashion (Treves and Karanth, 2003; Bangs et al., 2006). Improving human coexistence with carnivores often stands out as one of most immediate needs for maintaining viable free-ranging populations (Woodroffe et al., 2005a; Inskip and Zimmermann, 2009). In Namibia, where the research for this thesis took place, and in other parts of southern Africa, this endeavour inevitably entails securing private farmlands as a mainstay habitat for large carnivores (Marker et al., 2003a; Lindsey et al., 2009a, 2013a; Kent and Hill, 2013; Thorn et al., 2013; Winterbach et al., 2014) and conflict mitigation strategies have become ever more important conservation priorities.

Due to their large public appeal and government-recommended use in various countries (e.g. Botswana: M. Flyman, pers. comm. 2014, India: Ministry of Environment and Forests, 2011), ad hoc translocations for conflict reduction and rehabilitation purposes will likely continue to be employed in the foreseeable future. In addition, translocations currently find expanding use in other contexts such as moving wildlife away from development projects (Sullivan et al., 2014; Germano et al., 2015). Thus, the strategy needs to be scrutinised critically to identify its potential merits and disadvantages, costs and benefits, and ultimately its appropriate application. There is much room for improvement in terms of translocation reporting and evaluation, and an evident need for additional information to enable a more comprehensive assessment of a strategy that is frequently implemented under highly variable circumstances. After all, the decisions that carnivore managers take will only be as good as the information that they are based upon.

1.5 Aims, objectives and thesis outline

Addressing the significant paucity of translocation reporting, the present study analyses the outcomes of 22 intensively monitored events carried out into free-range environments in Namibia between 2008 and 2012. Focal large carnivores were southern African cheetah

(Schreber, 1775), African leopard (*P. p. pardus* - Linnaeus, 1758) and brown hyaena (*Parahyaena brunnea* - Thunberg, 1820), for which very limited translocation data were available at the time of this work, and specifically in a context of free-range systems as opposed to structured re-introductions or meta-population management in fenced PAs (*cf.* Hayward et al., 2007a, 2007b; Hayward and Somers, 2009).

The main focus of this study was to assess the efficacy of large carnivore translocations into free-range environments, and therefore to provide conservation managers with an enhanced understanding of the potential outcomes and implications of ad hoc translocations. Considering the severe monitoring constraints that have occurred during previous carnivore translocation efforts (e.g. Ebedes, 1970; Hamilton, 1981), it was one of the core purposes of this work to follow translocated subjects for as long as possible in order to document results beyond a few weeks or months post-release. The key objectives of this study were:

1) To determine post-release biology and conflict following *in situ* translocation;
2) To evaluate translocation success using both biological and technical parameters; and
3) To determine the financial costs of large carnivore translocations.

Following a description of the general methods and translocation circumstances in chapter 2, existing carnivore conflicts in Namibia (and the scale of persecution ensuing from these) are outlined in chapter 3 to enable an overview of the current demand for ad hoc translocations as well as a brief summary of their use in the past. Chapters 4 - 6 report translocation information for the focal species. In recognition of species ecology, conservation status and the different circumstances of the reported events, the results are evaluated case- and species-specifically using a combination of biological parameters (e.g. survival, reproduction, feeding ecology, and post-release movements) and technical considerations (e.g. release protocols, livestock conflict, and homing). Where possible, the results are also evaluated against those from resident (non-translocated) conspecifics and in light of available recipient areas in Namibia. Chapter 7 documents detailed large carnivore translocation costs and introduces the concept of Individual Conservation Cost (ICC) as a measure of cost-efficiency by adjusting expenses to observed translocation success. In chapter 8, the results obtained in this study are reviewed critically to discuss the role and usefulness of translocation as part of a broader carnivore management repertoire.

Chapter 2 General Methods

2.1 Subjects

Translocation data comprise information from monitoring of 23 cheetahs (with ten dependant offspring), six leopards and one brown hyaena. All subjects were trapped by private Namibian land managers between March 2008 and December 2012, either purposefully as suspected or confirmed livestock/game raiders or indiscriminately as part of routine carnivore removal campaigns. Animals were reported as problem-causing or were confiscated by the state wildlife authority. Depending on their specific background and the supporting evidence available in each case, study animals were classified as either livestock raiders (captured at livestock kills within 24 h of the incident), indiscriminate captures (including predators of expensive game species and those with doubtful conflict involvement), orphans (trapped at pre-mature age without mothers) or confiscations. Therefore, case-specific translocation objectives varied and included elements of carnivore rehabilitation (for orphans and confiscations) as well as conflict reduction. For all subjects, standard translocation objectives were:

1) To return subjects into free-range environments with minimal potential for post-release conflict;
2) To enable the contribution of these individuals to the wild gene pool;
3) To alleviate conflict at source sites in cases where livestock depredation had occurred; and
4) To investigate the factors that influence translocation success in ‘uncontrolled’ environments for future conservation planning.

Since responses to translocation vary for different carnivores (Linnell et al., 1997; Stoskopf, 2012), this study presents results on a species-specific basis (chapters 4-6). The biological and management details of all subjects are provided in respective chapters.

2.2 Immobilisation

Study carnivores were immobilised to enable health assessments, disease screening, fitting of tracking technology, recording of body measurements and collection of biological samples, including hair, blood, saliva, faecal matter and ecto-parasites. Immobilisations were carried out and supervised by licensed veterinary personnel who were registered with the Namibian Veterinary Council. Anaesthetics were administered intramuscularly via pneumatic darts propelled from a CO_2-powered rifle (Dan-Inject®, Barkøp, DK) or standard blow pipe and included suitable combinations of cyclohexamines and α2-agonists.

Until October 2010, translocated carnivores were immobilised with Hellabrunn mixture (Wiesner, 1998), a combination of 100.0 mg/ml Ketamine (Ketamine®, Kyron Laboratories, Benrose, RSA) with 125.0 mg/ml Xylazine (Xylazine®, Kyron Laboratories, Benrose, RSA) at a mixed dose of 0.2 ml per estimated 5.0 kg body mass for cheetahs, and 100.0 mg/ml Ketamine with 100.0 mg/ml Xylazine at a mixed dose of 0.2 ml per estimated 5 kg body mass for leopards. These protocols (i.e. the Xylazine component) were reversed with an intramuscular or intravenous administration of Yohimbine (Yohimbine®, Kyron Laboratories, Benrose, RSA) at a dose of 0.1 mg/kg body mass. From October 2010 onwards, a combination of Ketamine with Medetomidine (Medetomidine®, Kyron Laboratories, Benrose, RSA) was used to induce immobilisation. Cheetahs were injected with 10.0 mg Ketamine (100.0 mg/ml solution) and 0.25 mg Medetomidine (10.0 mg/ml solution) per estimated 5.0 kg body mass. The Medetomidine component was reversed with 0.5 – 0.75 mg Atipamezole (Antisedan®, Pfizer Laboratories, Sandton, RSA) per estimated 5.0 kg body mass. For leopards, 15.0 mg Ketamine and 0.4 mg Medetomidine were administered per estimated 5.0 kg body mass to achieve immobilisation. The Medetomidine component was reversed with 0.8 – 1.2 mg Atipamezole per estimated 5.0 kg body mass. Protocols differed for resident study leopards and brown hyaenas and are presented in chapters 4 and 6 respectively.

During immobilisation, heart rate, body temperature and respiration were monitored at least every 5 minutes. Age was estimated using dental conditions (Stander, 1997) and other morphological features (Marker and Dickman, 2005) for leopards and followed body measurements and morphological features outlined in Caro, (1994), Eaton (1969) and Thalwitzer (2008) for cheetahs. The hyaena was aged according to tooth eruption and wear (I. Wiesel, pers. comm.). Precautions were taken to avoid zoonotic disease transmission, undue disturbance and stress of animals. Carnivores were not exposed to trivial display or handling. To prevent dehydration and irritation of the cornea, eye ointment (ISEE®, Virbac, Centurion, RSA) or artificial tear fluid were administered and eyes were covered with blindfolds. Capture-related facial abrasions were disinfected and treated with Terramycin® wound spray. No serious capture-related injuries were diagnosed in any of the individuals.

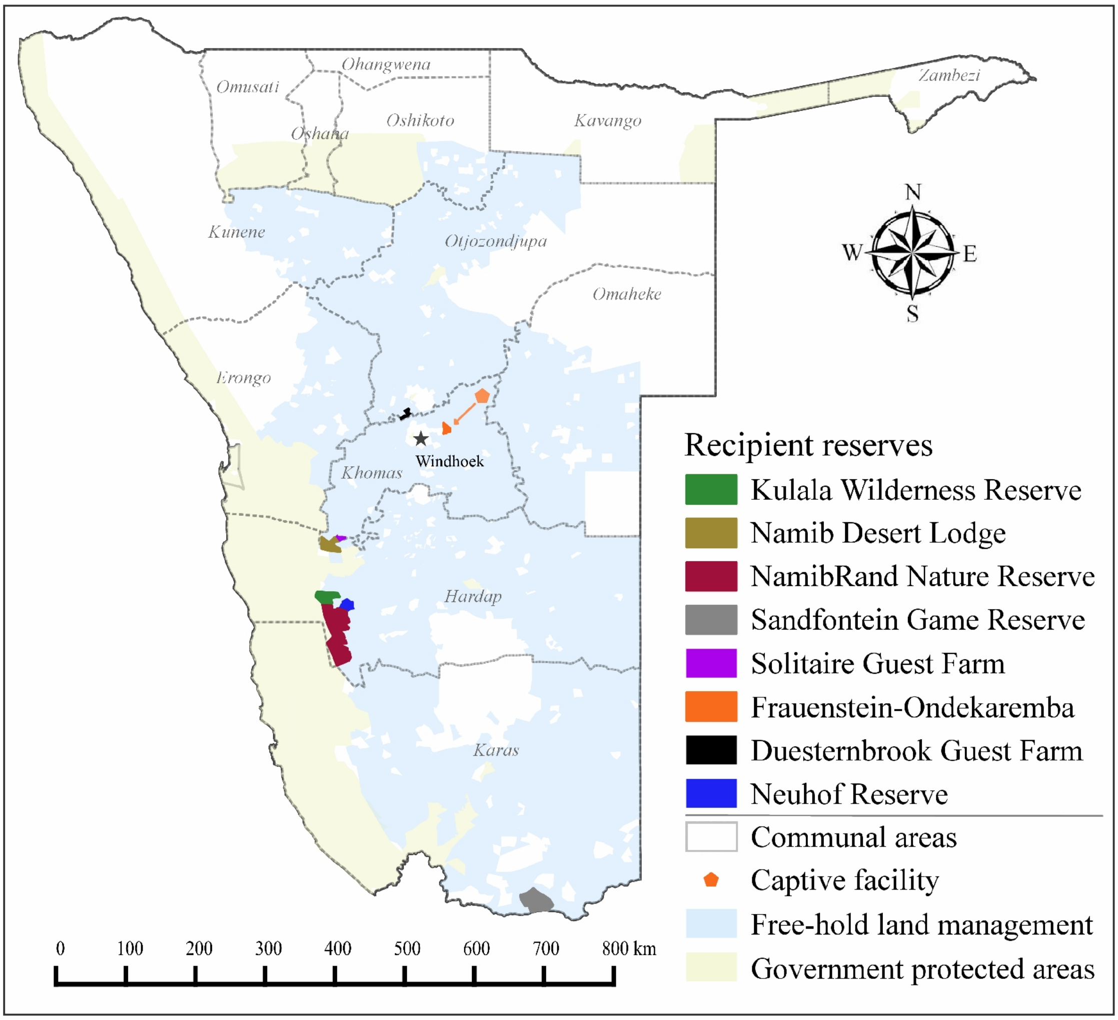


Figure 2.1 – Location of large carnivore recipient reserves and the carnivore housing facility in Namibia. *Sites are shown in relation to government protected areas, free-hold commercial farms and communal areas.*

2.3 Captivity

Prior to release, all translocated subjects were moved from their source sites into a government-approved captive facility (GPS: -22.3758, 17.4116) (Fig. 2.1). In a few cases, when government permission for translocation could be obtained immediately, carnivores were only admitted to the facility to carry out medical examination and for fitting of tracking technology, followed by immediate transfer to their release site. During temporary captivity, the animals were maintained in enclosures that reflected their natural habitat (*Acacia*-encroached bush savannah) in central Namibia. All captive facilities were in compliance with the national guidelines for the captive keeping of large carnivores (Ministry of Environment and Tourism, 2012).

Enclosure sizes varied between 72 m^2 – 235 m^2 for leopards (housed individually), and 90 m^2 – 4.3 ha for cheetahs depending on group size, social composition and length of temporary captivity. All enclosures allowed permanent access to water and provided sufficient shade and opportunities to seek full cover. Enclosures often had to be custom-built for specific animals. For example, to prevent accidental escapes (Hamilton, 1981; Hayward et al., 2006a), leopard holding facilities were roofed and constructed with galvanised, re-enforced wire mesh sections (Appendix 1) whereas large cheetah enclosures necessitated electric fencing (max. 9000 V with low amperage) for safekeeping (Appendix 2). The enclosures did not permit direct interaction with wild conspecifics that occurred around the captive facility.

As a general rule, time in captivity was minimized as much as was practicable and depended on case-specific circumstances such as the sourcing of suitable recipient sites, manufacture of suitable tracking devices, physical fitness, presence of offspring, or the issuing of permit documents. The animals were fed six days per week with horse (*Equus spp.*), donkey (*E. africanus asinus*) or game meat including warthog (*Phacochoerus africanus*), springbok (*Antidorcas marsupialis*), oryx (*Oryx gazella*), red hartebeest (*Alcelaphus caama*), blue wildebeest (*Connochaetes taurinus*), common impala (*Aepyceros melampus*), plains zebra (*E. quagga*) and greater kudu (*Tragelaphus strepsiceros*). Diets included aliquots (approximately 2.0 kg per cheetah per feed and 2.5 kg per leopard per feed) of red meat, bone material, organs, offal and fur. Whenever possible, feeds were provided as whole carcasses. During the first weeks after admission into captivity, feeding and drinking behaviour was monitored with motion-triggered wildlife cameras positioned approximately 5 m from feeding/watering stations. Veterinary care for all captive subjects was provided by Windhoek Animal Hospital, Windhoek, Namibia.

During confinement, all efforts were made to minimize stress and disturbance of study animals. Although carnivores were kept with minimal human contact to avoid habituation to human presence and loss of natural fear, prolonged captivity is known to influence a subject's behavioural response to human proximity and, consequently, its suitability for release (Kleiman, 1989; Jule et al., 2008). Therefore, the degree of habituation was evaluated for all subjects prior to release using discrete responses as classification criteria (Table 2.1).

Table 2.1 – Classification used to determine degree of habituation of translocated large carnivores.

Degree of habituation	**Specific behavioural response**
Wild	Subject does not tolerate human presence in any context; Subject exhibits typical natural avoidance behaviour and flight response – no direct observations possible; Subject exhibits strong signs of stress/discomfort when confined – hissing, spitting, stomping, attempt to escape.
Semi-habituated	Except for feeding context, subject always avoids human presence; Subject exhibits expected flight response after food is obtained and seeks cover; Subject exhibits obvious signs of stress/discomfort when confined – hissing, spitting, stomping.
Habituated	Subject tolerates human presence beyond feeding events; Subject does not exhibit typical flight or avoidance behaviour when approached; No direct/physical contact possible (contrary to tame); Subject exhibits reduced signs of stress/discomfort when confined (e.g. only mild irritation).

2.4 Transportation and release mode

In this study, translocation distance was defined as the linear distance from source site to final recipient site. In general, long distances (>200 km) were preferred to prevent successful homing (Linnell et al., 1997; Marker, 2002), but values were influenced by the case-specific availability of recipient areas. During transportation, animals were not under the influence of immobilising agents, nor were they exposed to public viewing. Carnivores were moved in grass-padded, covered/closed crates with minimum dimensions of 86 cm (width), 80 cm (height) and 123 cm (length) for a single adult subject (Appendix 3). Female cheetahs with dependent juveniles were moved in the same crate. The ventilated crates allowed safe transportation whilst limiting the possibility of injury.

At recipient sites, subjects were released using hard releases, i.e. directly from the transport crate after being placed at a permanent water source for several hours, or through soft releases, i.e. following periods of acclimatisation in a holding pen at the recipient site lasting from 10 days to 9.5 months (specific details in species chapters). During acclimation, managers of recipient areas were responsible for the care of carnivores and feeding was reduced to 2-3 events per week with provision of large food portions (e.g.

entire legs) for each animal or whole carcasses (Appendix 4). All holding pens provided permanent access to water, sufficient shade and cover and unobstructed views of the recipient reserve. The leopard pen measured approximately 144 m^2 with an additional feeding partition whereas cheetah pens ranged between 1.2 ha – 500 ha in size. During the first weeks of pen acclimatisation, successful feeding of each individual was observed from a distance using binoculars (in the case of cheetah groups) or with wildlife cameras positioned near feeding spots. Due to their potential danger, leopards were released using a rope-pulley system that was operated from a nearby vehicle (Appendix 5).

2.5 Recipient areas and release considerations

All recipient areas were privately owned nature reserves (Fig. 2.1; Appendix 6) and formed part of Namibia's free-hold commercial farming and tourism landscape (Mendelsohn, 2006). The reserves were established in the last 30 years in former areas of livestock production. All sites promoted large carnivore population recovery as one of their management objectives after the cause of original carnivore declines (persecution to protect livestock farming) had been eliminated. At the recipient sites, carnivores were valued as additional contributors to existing wildlife tourism efforts. Therefore, the reserves presented potential recipient habitat for individuals that would otherwise be destroyed or admitted into permanent captivity.

Similar to previous translocation studies (e.g. Purchase et al., 2006; Boast et al., 2015) the events reported here were experimental, e.g. using variable translocation distances, release modes and captive times, thus reflecting the general, non-standardised conditions under which ad hoc translocations typically occur. The presence of game-proof fencing at several recipient sites and recipient area size (Appendix 6) was not considered effective to constrain carnivore movements in all cases. Hence, and although this was not always possible, preference was given to recipient areas bordering, or in the immediate vicinity of, large government PAs (Fig. 2.1). Site fidelity, however, could not be expected for all events and therefore a simple data sharing system was employed (description in section 2.8) to inform land managers of the movements of released carnivores. There were no formal feasibility studies available for the recipient reserves and areas were mainly selected according to discussions with and recommendations made by reserve management and the state wildlife authorities, including the following minimum criteria:

1) Located within suitable species habitat (extant species range) with estimated low to medium density of conspecifics (according to Hanssen and Stander, 2004; Stein et al., 2012);

2) Providing regular access to permanent water, adequate prey compositions and known intra-guild competition (Appendix 6); and
3) Supported land uses with minimum potential for post-release conflict.

Managers of recipient reserves actively supported monitoring efforts (mainly through their own observations and camera trapping) and provided important logistic support, especially in terms of carnivore management in pre-release enclosures. Reserves informed surrounding land managers of translocation events and facilitated conflict mitigation through compensation agreements. In the case of cheetah, attempts were made at several sites to restore historic populations by supplementing very low local densities (estimated at <1.0 cheetahs per 100 km^2) with translocated individuals.

To minimise the likelihood of negative effects such as population overspill and intra-specific aggression associated with sustained carnivore translocations into the same recipient area (*cf.* Hamilton, 1981; Athreya et al., 2007, 2011), releases were carried out as staggered events in order to allow for sufficient settling and acclimation periods of previously translocated subjects. A minimum inter-release interval of over six months was maintained for leopards and over four months for cheetahs (except one event) if animals were released on the same reserve. Most study animals were translocated into areas where they experienced novel intra-guild competition from spotted hyaena (*Crocuta crocuta*) but none of the recipient areas had lions. Except for the recapture of one unsuccessful individual and the re-release of two subjects after accidental capture on farms outside of recipient reserves, there was no supportive post-release management of translocated carnivores such as supplementary feeding.

2.6 Success

With regard to previous shortcomings in translocation reporting (Linnell et al., 1997; Fischer and Lindenmayer, 2000; Massei et al., 2010; Fontúrbel and Simonetti, 2011), minimum success conditions in this study included:

1) Survival for at least one year post-release;
2) No homing to the capture site and/or captive facility; and
3) Minimal livestock conflict (within levels considered tolerable or appropriate for compensation).

Based on average landowner tolerance toward livestock predation (chapter 3; Stein et al., 2010), five livestock units per year per carnivore was defined as the maximum acceptable conflict threshold to qualify a translocation as successful. To demonstrate accountability for translocations and to prevent immediate persecution of individuals when conflict occurred, this threshold was linked to the proviso that landowners were compensated for any verifiable damage from translocated carnivores (as determined by data sharing and monitoring information). Reproduction, the ultimate indication of biological success (Reading and Clark, 1996; Hayward et al., 2007a, 2007b), was not a minimum success condition as mating events were difficult to confirm for male subjects. Nevertheless, it remained a key criterion in the assessment of female case studies and occasionally for males too. In contrast with other studies (e.g. Hamilton, 1981; Weilenmann et al., 2010), and although generally desirable, site fidelity was also not considered a prerequisite for translocation success here because all subjects were released into environments permitting free choice of movement. Furthermore, a carnivore moving out of its designated recipient area may not necessarily constitute a failure if the animal has little or no negative impact on its biological and anthropogenic environment.

2.7 Monitoring technology and protocols

Except for sub-adult cheetahs released alongside their mothers, all subject animals were fitted with external tracking devices to enable intensive post-release monitoring. Tracking unit details are provided in results chapters but all devices adhered to recommended transmitter weight to body weight ratios of less than 1.5% (Kenward, 2000). Tracking collars were fitted to accommodate expected neck growth in pre-prime individuals. External transmitters included Very High Frequency (VHF) radio-collars with activity and mortality modes as well as dual VHF-GPS (Global Positioning System) satellite transmission collars equipped with drop-off mechanisms. Battery life spans were expected to be approximately 36 months for all VHF units and between 12 - 30 months for GPS trackers. In addition to tracking collars, ID-microchips (BackHome®, Virbac, Centurion, RSA) were implanted subcutaneously for permanent identification.

Positional data were collected with standard VHF telemetry methods (Amlaner Jr. and Macdonald, 1980; Kenward, 2000) using tracking receiver model R4000 (Advanced Telemetry Systems™, Insanti, USA) in combination with a foldable yagi-antenna, and via remote GPS satellite telemetry. Tracking units occasionally enabled direct observations and care was taken not to disturb or displace animals during monitoring activities. Depending on the total number of study animals that were monitored simultaneously, a team of 2-6 field researchers attempted to locate VHF-tagged individuals at least once per week (range = 1-7 days) and telemetry efforts generally concentrated on these subjects. All team members had extensive experience in telemetry tracking and used standardised recording protocols. Field monitoring was supervised and coordinated by the author.

GPS satellite transmitters recorded and relayed 1-6 daily locations. Daily detection probability was <1.0 for VHF-tagged subjects and ~1.0 for subjects equipped with active GPS trackers. GPS sampling regimes were adjusted based on specific research and management objectives at and around the recipient sites. For example, when carnivores left recipient reserves daily GPS fix schedules often were upgraded to provide land managers with sufficient information to assess conflict involvement of translocated subjects or to predict animal movements in relation to livestock areas in order to prevent conflict.

If a carnivore could not be located for about two weeks or GPS units failed early, aerial telemetry from a Cessna 182 aircraft was employed to ascertain the fate of the individual. Depending on local terrain conditions and last known locations, tracking flights were carried out at altitudes of approximately 1,500 m above ground to enable signal detection, and at lower altitudes to enable direct observations. If tracking flights resulted in the detection of a mortality signal, the GPS location was recorded and the site was later investigated from the ground. Locations of animals that could not be observed during ground or aerial VHF telemetry were calculated through triangulation of the strongest signal bearings using software *LOCATE II* (Nams, 1990). Estimated positions with an accuracy error >500 m were not considered for analyses.

One leopard male was also monitored using his unique spoor that resulted from an old front-foot injury and enabled discrete identification. In this particular case, spoor identification was confirmed by experienced San trackers (also see Stander et al., 1997b).

Prey was located by investigating GPS position clusters (after the animal had moved away) that indicated potential kill sites (Knopff et al., 2009) or by investigating kill remains after hunts were observed directly (only for cheetahs).

Camera trap photos as well as observation reports further supplemented monitoring information at the different study sites. Cameras were usually placed at artificial water holes, prey items, marking sites (e.g. play trees for cheetahs) or along naturally confined movement corridors such as narrow canyons. However, camera trapping efforts varied significantly in terms of sampling effort and equipment used across recipient sites. Altogether, data included photographic records from over 20 camera models from a range of manufacturers: Reconyx^{TM} (Holmen, USA), Bushnell^{TM} (Cody, USA), Stealth Cam^{TM} (Grand Prairie, USA), Moultrie^{TM} (Alabaster, USA), Scoutguard^{TM} (Norcross, USA), Cuddeback^{TM} (De Pere, USA) and Covert^{TM} (Lewisburg, USA). Due to the geographic extent of this study, standardised camera trapping was neither practical nor feasible, but opportunistic records provided valuable information, particularly in terms of reproduction and physical condition. Subjects were identified by their tracking units or distinct coat patterns.

Where this was necessary, research work was carried out beyond the boundaries of recipient areas. Prior to accessing government PAs or privately owned farms for monitoring purposes, permission to do so was obtained from wildlife authorities or land managers respectively. Research activities and management of animals were endorsed by the Ministry of Environment and Tourism in Namibia. Translocated and resident carnivores were studied under permit numbers: 691/2003; 914,916,990/2005; 1025/2006; 1134/2007; 1233,1254,1330/2008; 1354,1451/2009; 1330,1459,1550/2010; 1459,1550/2011; 1748/2012; 1782/2013; 1843/2013; and 1888/2014.

2.8 Data sharing protocol

One key element of monitoring these translocations was on-going interaction with the managers of the areas where carnivores were released or on the properties to where animals moved. For this purpose, carnivore locations and property features were projected with Google Earth high-resolution imagery on a regular basis and were subsequently distributed via email (Fig. 2.2). In addition, Google Earth enabled meta-data from GPS satellite units were supplied to allow managers to view position data at any required scale and resolution. Emails further included a description of the animal's movements and activities. Information was emailed daily during the first months post-release and less regularly after animals had settled into definable ranges. When internet was not available, information was distributed via phone.

Land managers were encouraged to utilise tracking information to adjust their livestock and game husbandry practices or for non-consumptive tourism activities. Although offending carnivores can legally be removed from private lands in Namibia, data were provided on the condition that they could not be utilised for trophy hunting purposes or as part of indiscriminate persecution campaigns. Using the same stipulation, managers shared information within their communities (livestock farming associations and local commercial conservancies) when carnivores moved across multiple properties. Managers frequently provided valuable feedback on prey, reproduction and conflict involvement.

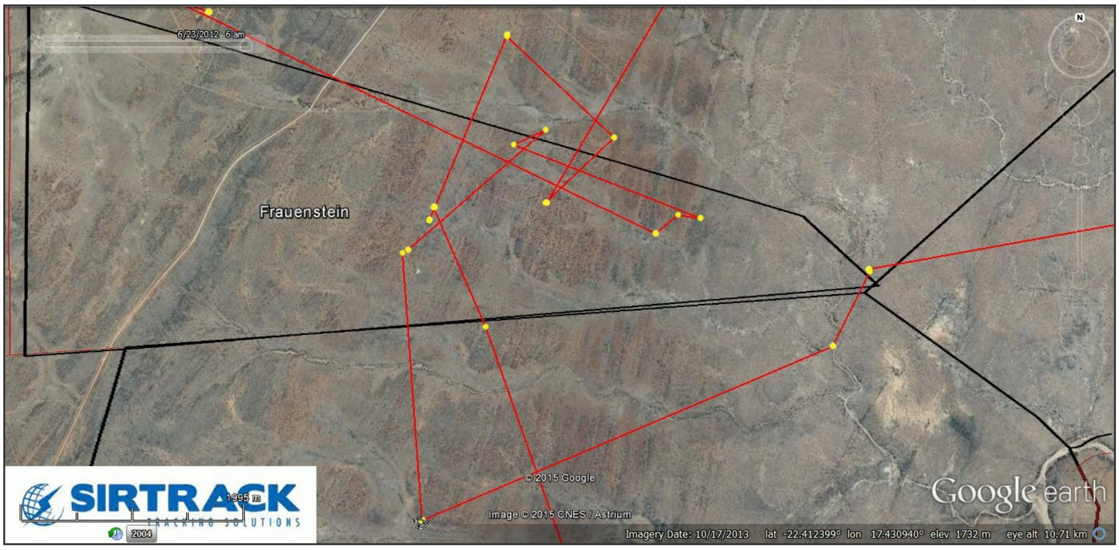


Figure 2.2 – Example of data sharing display used to inform land managers about the movements of translocated subjects. *Yellow markers indicate GPS fixes; red lines indicate direct connections between subsequent GPS fixes; black lines indicate property boundaries.*

2.9 Data analyses

2.9.1 Software

Statistics were calculated with software *R v.3.1.0* (R Core Team, 2014) and *JMP v.11.0* (SAS Institute, 2013), and spatial analyses were computed with *ArcGIS v.10.2* (ESRI, 2013) and *QGIS v.2.6-2.8.1* (QGIS Development Team, 2014).

2.9.2 General considerations

To identify individual factors influencing response variables, multi-variable data sets were first interrogated with regression partitioning using Decision-Tree models (Breiman et al., 1984) in *JMP* software. This approach was used because data sets were generally too small, or significantly deviated from normal distributions, to permit analyses with more sophisticated Bayesian Networks, K-means cluster analysis or Generalised Linear Models. Prior to analyses, data were tested for normality and parametric or non-parametric analysis methods were selected accordingly.

2.9.3 Movements, homing and site fidelity

After removing outliers and low-quality GPS fixes, positional data were standardised to one location (closest to 12:00 GMT) per day for all tagged subjects (Hunter, 1998a; Mizutani and Jewell, 1998). While this selection cannot entirely compensate for auto-correlation between subsequent positions (especially for regular GPS satellite data) (Swihart and Slade, 1985; Rooney et al., 1998), it increases sample independence significantly (De Solla et al., 1999). In addition, it provided a useful approach to enable comparisons between VHF- and GPS-tagged individuals, without omitting large amounts of biologically relevant data (De Solla et al., 1999; Weilenmann et al., 2010). Standardisation was also considered necessary because GPS fix regimes varied between GPS-tagged subjects, mainly because of data sharing protocols with land managers outside of recipient areas.

Carnivore movements were calculated with ArcGIS using the *Spatial Analyst Movement Ecology Tools Extension v.10.2.2* (Wall, 2014). Map projections were created with the WGS84 grid reference whereas fine-scale movements were analysed with the Africa Albers Equal Area Conic grid reference. Home ranges were measured as 50% and 100% Minimum Convex Polygons (MCP) (Mohr, 1947) and as individually smoothed movement-based 50% and 95% kernel density estimations (KDE) (following AUC-methods in Cumming and Cornélis, 2012). Home ranges were calculated excluding periods of orientation and exploration. The 100% upper MCP threshold was chosen to document the true extent of post-release movements (and its associated influence on site fidelity - see site fidelity definition below) as this is an important element of translocation evaluations (e.g. Linnell et al., 1997), while simultaneously enabling a reliable assessment of the duration of post-release explorations as indicated by range use area progression (see settling behaviour definition below). Moreover, the 100% MCP threshold represents the most commonly used metric in spatial ecology studies on the focal species (e.g. see range-wide leopard home range summary in Mondal et al., 2013). The 95% upper KDE threshold was selected to enable range metric comparison of settled, post-exploration home ranges with the most recent spatial ecology studies on the focal species in Namibia (e.g. brown hyaena, Wiesel, 2006; Kuhn et al., 2008; cheetah, Marker et al., 2008a; leopard, Stein et al., 2011) and those of other translocation studies in similar environments in southern Africa (e.g. Weilenmann et al., 2010; Houser et al., 2011).

To determine the duration of exploratory movements for cheetahs and leopards, 100% MCP values were calculated for 10-day periods (with a minimum of five available data points) starting from the day of release. This 10-day increment was shifted repeatedly by one day, creating a progressive estimate of range area utilised post-release. Settling behaviour was assumed when MCP area-plots reached stable asymptotes as determined by the date when percentage increase between progressive MCP increments was no longer statistically significant (following Gautestad and Mysterud's (1995) methodology) and no further significant increase occurred for at least 90 days.

Percentage overlap of pruned daily locations with the recipient reserve (and connected complexes of PAs) was used as a measure of site fidelity for the first 12 months after release. Hence, site fidelity is here expressed as a measure of occupancy of the targeted recipient area (or other PAs) rather than as a measure of association between individual locations and an activity centre (*cf.* Spencer et al., 1990) or the specific release locality.

Homing behaviour was assessed using package *Circular v.0.4-7* (Lund and Agostinelli, 2013) in *R* by calculating bearing angles and distances between a subject's last known location and release location relative to the capture site. Bearing angles were adjusted to set an individual's 'home' direction (i.e. the original capture site) to 0°. All distances were normalised on a scale from zero to one, representing the distance between a subject's capture and release sites. An animal was considered homing if it returned to its capture property or had moved its entire translocation distance towards the capture location within 22.5° on either side of the true 'home' direction (Fies et al., 1987). In addition, successful homing was defined to include an animal returning to the captive facility.

2.9.4 Survival

Following the conclusion of the study period, Kaplan-Meier survivorship estimates (product limit estimator, Kaplan and Meier, 1958) were calculated for all treatment groups using a staggered entry design (Pollock et al., 1989) that allowed for inclusion of subjects from separate events conducted across several years. This survivorship method is particularly useful for studies with unpredictable design, e.g. animals entering the sample at irregular intervals (Marnewick et al., 2009), and it accommodates small, fluctuating sample sizes without discrimination against individuals with unknown fate when collars fail or deplete. These subjects are removed ('censored') from the sample at the time they disappear and subsequently no longer influence survivorship results.

2.9.5 Potential recipient areas

To determine area suitability for future cheetah and leopard translocations throughout Namibia, the ArcGIS-based geo-processing package *CaTSuiT* (Lemeris Jr., 2013, 2015) was modified for those species. With the tool, conservation managers have maximum flexibility to inform the selection process according to species- or area-specific conditions, thus accounting for variable ecological requirements and conservation goals. The user defines minimum translocation suitability criteria (e.g. according to specific land-use parameters, habitat characteristics, wildlife population data, minimum desired translocation distance) and each variable can be assigned with a relative influence towards the overall selection process. Based on the provided input information, the tool then works by a step-wise elimination process and favours (or maintains) areas that fulfil suitability criteria (e.g. protected status with high prey density) while eliminating those that do not meet minimum suitability conditions (e.g. unprotected land devoid of wildlife).

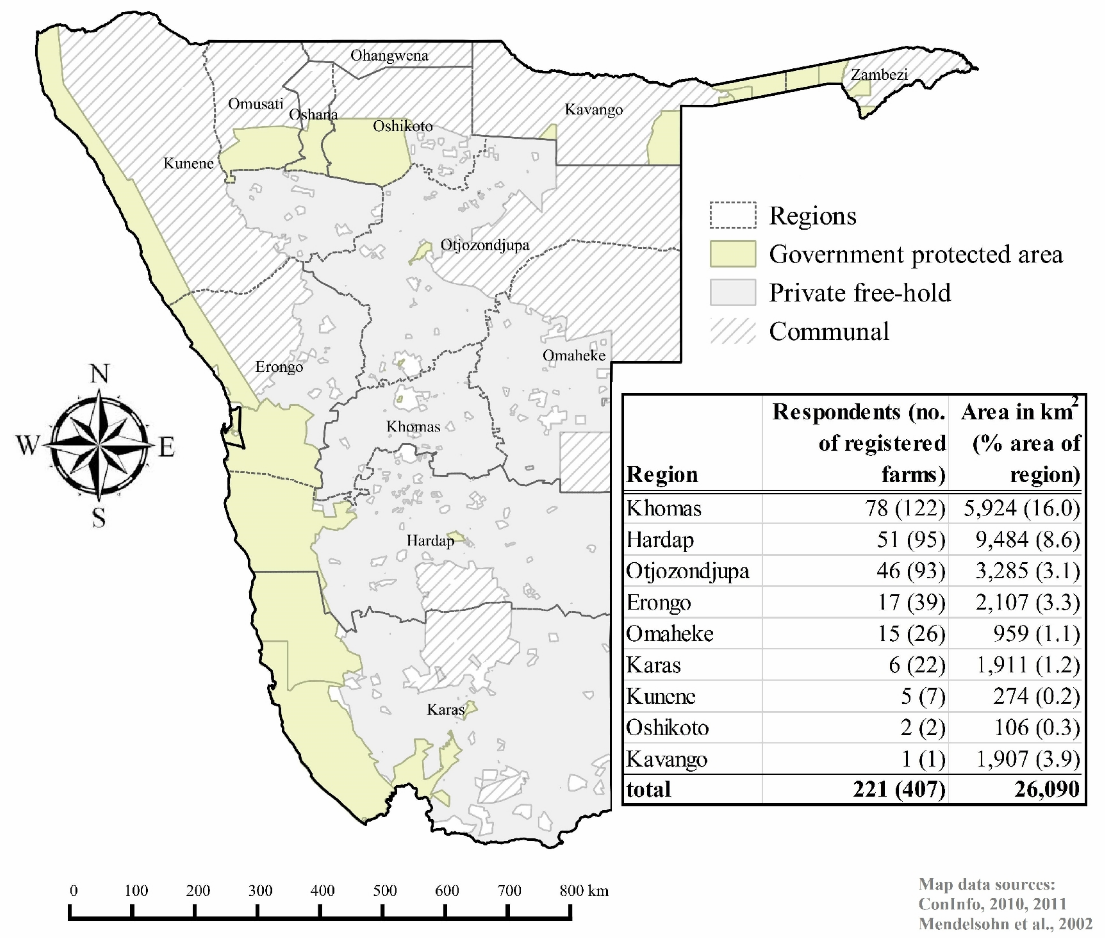
Chapter 3 Large Carnivore Management in Namibia

3.1 Introduction

For any conservation management intervention to have a chance of success it is imperative to understand the issue it aims to address in terms of its origin and scale. Only once the 'problem' has been defined can wildlife managers and researchers proceed and test specific activities in order to solve it. As regards the translocation of carnivores this requires at least a summary assessment of the situations that lead to conflict between land managers and large carnivores and the resulting persecution of the latter.

As elsewhere in southern Africa (e.g. Gusset et al., 2009b; Thorn et al., 2013), conflict with carnivores in Namibia mainly arises outside PAs in landscapes dominated by livestock production (Morsbach, 1986a; Marker et al., 2007; Stein et al., 2010). Despite its relatively low contribution to the Gross National Income (e.g. commercial cattle ranching = USD 235 million or 3.3% of GNI in 2009) livestock farming remains the prevalent land use in Namibia (Barnes et al., 2012) and is practised in various forms for subsistence or commercial purposes across ~78% of the country (Mendelsohn, 2006). The predominant types are cattle ranching (on approximately 315,000 km^2 or 38% of Namibia) and smallstock production (on approximately 270,000 km^2 or 33% of Namibia), or a local mix thereof (Appendix 7, from Mendelsohn, 2006). Farms are managed privately with free-hold land tenure (43% of Namibia) or communally on open access pastures (32% of Namibia) (Fig. 3.1) and a minority of about 3,500 mainly white free-hold commercial farmers own more than 50% of the livestock landscapes in the country (Mendelsohn, 2006; Lindsey et al., 2013a).

Namibia's highly variable bio-climatic conditions, i.e. soil conditions and erratic rains influencing local graze and browse yields, and the low primary productivity of desert and semi-desert environments make commercial farming a challenging task that requires large areas (up to 150 km^2 for cattle and 350 km^2 for smallstock) to support sufficient herd sizes (Mendelsohn, 2006; Barnes et al., 2012). Livestock production, therefore, is extensive with an average of 5.1 large stock units/km^2 on cattle ranches (Lindsey et al., 2013b). To facilitate recovery of sparse vegetation, domestic animals are rotated through fenced camp systems (year-round) arranged around artificial water holes. In most farming areas ordinary five-strand stock fences do not obstruct wildlife movements (Lindsey et al., 2013b) and livestock share camp systems with free-ranging ungulates, and with a number of large carnivores. Depending on the location of properties, these include lion, spotted hyaena, brown hyaena, cheetah, leopard and/or African wild dog (Hanssen and Stander, 2004; Stein et al., 2012; Lindsey et al., 2013a).



Region	Respondents (no. of registered farms)	Area in km^2 (% area of region)
Khomas	78 (122)	5,924 (16.0)
Hardap	51 (95)	9,484 (8.6)
Otjozondjupa	46 (93)	3,285 (3.1)
Erongo	17 (39)	2,107 (3.3)
Omaheke	15 (26)	959 (1.1)
Karas	6 (22)	1,911 (1.2)
Kunene	5 (7)	274 (0.2)
Oshikoto	2 (2)	106 (0.3)
Kavango	1 (1)	1,907 (3.9)
total	**221 (407)**	**26,090**

Map data sources:
ConInfo, 2010, 2011
Mendelsohn et al., 2002

Figure 3.1 – Distribution of land tenure systems in Namibia in relation to land manager sample contribution by administrative region. *The map displays the distribution of Namibia's three main land tenure types across the country. The table inset shows free-hold manager sample representation per region in terms of respondents and area contribution, with bias towards the central farming regions.*

Prior to a legislative change in 1967 (Nature Conservation Ordinance No. 31 of 1967), which granted first user rights over wild ungulates to free-hold land managers (Joubert, 1994; Barnes and de Jager, 1996), most farmers reduced food competition and disease risk for their livestock by eradicating indigenous ungulates from their properties. At the same time, they heavily persecuted large social carnivores (i.e. lion, spotted hyaena, and African wild dog) to prevent attacks on livestock, and by the 1950s these species had largely disappeared from free-hold stock farming areas (Marker, 2002), a situation that has changed little until present day (Stein et al., 2012; Lindsey et al., 2013a). However, with new wildlife use incentives and rising international markets for wildlife tourism and safari hunting in the 1980s (Bond et al., 2004) wild ungulates returned to private livestock lands (Barnes and de Jager, 1996; Erb, 2004). Since then game populations have recovered with recent estimates varying between 1.8 million (Barnes et al., 2009) and 2.8 million heads (Lindsey et al., 2013b). Today, ungulate populations on free-hold farms possibly outnumber those in PAs by a factor of 21 - 33 (Lindsey et al., 2013b).

Benefiting from this restoration of natural prey, and in part due to the absence of competing large carnivores (Marker-Kraus and Kraus, 1994; Marker and Dickman, 2005; Lindsey et al., 2013a), cheetahs and leopards have also proliferated on free-hold farms and filled the role of apex predators. Due to the size, difficult terrain and often dense, bush-encroached vegetation on farmlands, protective anti-predator measures are difficult to implement and active herd management is usually limited to day-time hours, thus exposing livestock to predation from free-ranging carnivores. Previous studies showed that cattle predation by leopards alone can result in annual financial damage from $1,370 (3.5 calves/year, Stein et al., 2010) to $2,644 per property (Lindsey et al., 2013a), exceeding the damage that commercial ranchers are willing to accept (estimated at 3.0 calves farm/year or $1,190, Stein et al., 2010). Depending on their local density, cheetahs may account for an additional annual predation of 0.76 – 10.3 cattle calves and 0.32 – 4.4 sheep per unit (Marker et al., 2003b), providing sufficient motivation for farmers to persecute both species and regard them as a nuisance. The economic interests of Namibian farmers are legally protected (Nature Conservation Ordinance No. 4 of 1975), including provisions for legal removal of offending carnivores with government permission.

In recent years, free-hold land managers increasingly farm livestock and wildlife in mixed production systems (Mendelsohn, 2006; Lindsey et al., 2013a) and supplement their incomes with wildlife-based land uses such as trophy hunting and ecotourism (Lindsey et al., 2013b). This change has brought about a new and more complex set of attitudes towards large carnivores (Lindsey et al., 2013a). On mixed livestock-wildlife and exclusive wildlife farms, large carnivores may still pose a serious risk to domestic animals and/or valuable game species (e.g. Schumann et al., 2006; Marker et al., 2007) whilst simultaneously offering income opportunities from consumptive or non-consumptive tourism activities. For example, with bans imposed in many African countries, Namibia remains one of the few destinations where lion, leopard, cheetah, brown and spotted hyaenas can be sports-hunted. The option of economic exploitation (both consumptively and non-consumptively) provides an impetus for land managers to coexist with carnivores if income can offset the financial damage incurred from stock depredation (Stein et al., 2010). On private ecotourism reserves, large carnivores have little potential for conflict and are mainly desired for their intrinsic, ecologic, and viewing values. In Namibia's scenic South and Southwest, for instance, large private reserves (>750 km^2) have been established in areas of former sheep and goat farming (e.g. Gondwana, NamibRand and Sandfontein in Fig. 3.2) with an aim of restoring historic wildlife populations (including large carnivores) and ecologic integrity through exclusion of livestock. Across the country, there are approximately 153 private game parks and nature reserves which promote conservation and tourism objectives on ~1.6% of Namibia's land (Zimmermann et al., 2012). Thus, a shift in land use reflects a change in the traditional view of large carnivores as pests, at least in some free-hold areas (Lindsey et al., 2013a).

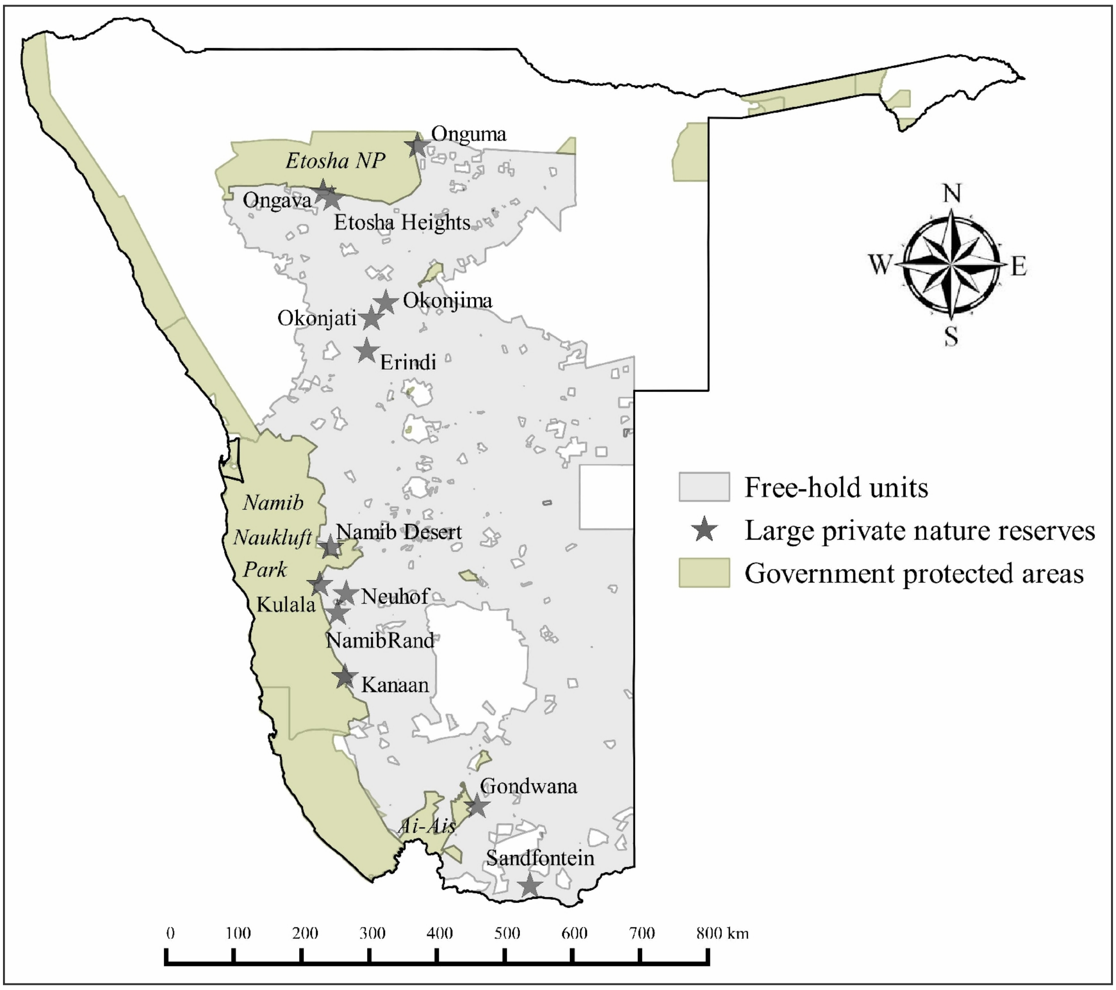


Figure 3.2 – Location of large private nature reserves (>200 km^2) with free-hold land tenure.

Conditions for carnivore conservation likely will continue to improve because the extent and profitability of livestock farming is expected to decline due to climate change, bush encroachment, range degradation, and international trade restrictions (Lange et al., 1998; van Schalkwyk et al., 2010; Barnes et al., 2012). This trend leads farmers to contemplate expansion of wildlife-based land uses. For example, 21.4% of managers consider wildlife-only production systems (Lindsey et al., 2013b). Following a five-fold increase in international tourist arrivals between 1990 and 2005 (Mendelsohn, 2006) income from wildlife tourism already exceeds that from livestock farming on many properties (Lindsey et al., 2013b) and projections predict a further rise in visitor numbers of 5.7% per year until 2022 (World Trade and Tourism Council, 2012). With large carnivore populations dwindling in many African countries (Ray et al., 2005) and other tourism destinations like Kenya facing continued unrest, Namibia presents a key carnivore market for recreational tourists and hunters alike. The country provides private land managers with increasing opportunities to benefit from the high tourism potential of large carnivores (e.g. viewing value in Lindsey et al., 2007; Maciejewski and Kerley, 2014; trophy values in Lindsey et al., 2012; Funston et al., 2013; Jorge et al., 2013; Rust and Marker, 2013a; Braczkowski et al., 2015).

The key element to increasing tolerance of large carnivores on private lands will be to reduce the costs associated with coexistence (Marker et al., 2007; Stein et al., 2010; Lindsey et al., 2013a). Private land managers in Namibia do not receive compensation for stock predation. Therefore, mitigation attempts typically entail preventative or symptomatic live- and lethal removals of perceived ‘problem’ carnivores. Numerous interview surveys have previously analysed the attitudes of different land manager groups towards large carnivores (e.g. Marker et al., 2003c; Schumann et al., 2008; Stein et al., 2010; Rust and Marker, 2013b); but few studies have done so at large spatial scales (e.g. Lindsey et al., 2013a) or with a focus on how persistent conflict translates into carnivore removals (Marker et al., 2003a). This information, however, needs to form part of the baseline for guiding conservation management activities and reviews of their efficacy to address existing conflicts, particularly if the objective is to mitigate indiscriminate carnivore removals non-lethally.

In this context, this chapter outlines land manager conflicts with large carnivores in a transforming multi-use landscape in Namibia. The aim of this chapter is to determine the extent and characteristics of conflicts, the levels of persecution ensuing from these, and the potential demand for translocation as a non-lethal management option. The main focus here is on free-hold commercial properties where about 3,500 managers exert control over large carnivores across 43% of the country.

3.2 Methods (supplemental to chapter 2)

3.2.1 Semi-structured interviews

As part of large carnivore consultations with Namibian land managers, interviews about carnivore conflicts and management were conducted between May 2008 and October 2014. The term "manager" as opposed to "land owner" is used because those responsible for property and carnivore management frequently do not hold property title deeds but act on a management mandate. Direct consultations involved elements of scientific research, education, joint carnivore monitoring, and experimental carnivore management on free-hold farms. It is important to note that, except for seven events, contact was sought by managers rather than the investigator.

A pilot study was conducted from March - May 2008, during which 76.5% of respondents (total $n = 17$) refused to partake in a formal questionnaire survey because they had participated in similar surveys in the past, but not received feedback. To allow for a guided process with maximum flexibility, semi-structured interviews (Brockington and Sullivan, 2003) were employed to collect information. Interviews focused on a defined set of carnivore-related topic areas and were administered through open and closed questions (Appendix 8). Information relevant to activities covering the 24 months prior to first contact with the respondent were recorded to determine manager and property characteristics, details of conflict with large carnivores, and associated management practices.

On management units (defined in section 3.2.2) where carnivores were controlled using lethal methods, objective evidence (e.g. photographs and pelts) was sought to verify removal statistics and reduce the possibility of exaggeration bias (Maddox, 2003). Although this was not always possible, attempts were made to carry out first interviews on-site whereas repeat interactions were often conducted via phone and email. During initial interviews, participants were limited to relevant managers (and sometimes their immediate families) and research team members. Contrary to similar previous studies (e.g. Schumann et al., 2008), interviews were not conducted in public group settings to limit the chance of bias resulting from social agendas and peer pressure. Depending on the manager's preference, interviews were conducted in German, Afrikaans, or English. Because data often involved sensitive information, managers participated on an anonymous basis and on the condition that they (or their management units) were not identifiable from published materials. Results were recorded during or immediately after interviews. If team members other than the author made initial contact, the author confirmed data via phone or email, or during a subsequent interview. There was no finite number or duration of interactions per manager and no discrimination against managers in remote areas. When consultations resulted in participatory carnivore monitoring a simple data sharing protocol (section 2.8 in chapter 2) was employed. The researchers had no influence on a manager's motivation to request help with or information on carnivore issues, and participation was not rewarded financially.

For reasons of consistency, interviews carried out during the pilot study are excluded from analyses. Results from interviews during which the respondent answered less than half of the questions are also excluded.

3.2.2 Definitions

A management unit is defined as the combined area over which a respondent had management control through ownership, lease agreements or other legal mandates.

Tolerance of respondents was based on observed large carnivore management and classified into three categories:

1) Tolerant (no lethal persecution);
2) Semi-tolerant (attempted selective persecution only after conflict had occurred, either lethal or non-lethal); and
3) Intolerant (continuous indiscriminate persecution).

Primary income was determined by priority contribution of different land uses to management unit income and was classed into five major categories, including:

1) Livestock (cattle and smallstock);
2) Wildlife (game farming and trophy hunting);
3) Recreation (non-consumptive wildlife and scenic tourism);
4) Business (majority of income not farm dependent and from own business, e.g. crafts and trade firms, consultancy/academic/medical professions); and
5) Others (retirement properties, agriculture and mineral extraction).

Age was categorised in 10-year increments: 31 – 40 years, 41 – 50 years and so forth because respondents often would not disclose their accurate age but agreed to a broader classification.

Persecution is here defined as the removal of a large carnivore from the free-ranging population through trapping, killing, or permanent captivity.

Predation impact was measured as the annual percentage loss of total livestock keepings for each category (smallstock, cattle) and cumulatively (total stock) that managers attributed to large carnivores. Financial impact was calculated by considering different annual sales values of livestock (Meat Board of Namibia, 2014) and fluctuating currency conversion rates during the study, using the mean annual conversion rate USD:ZAR in each year (The World Bank, 2015).

3.3 Results

3.3.1 Manager sample characteristics

A total of 574 consultations with 272 managers in nine regions of Namibia were conducted. Of these managers, 19 refused to participate in the study (7.0%); a refusal rate unlikely to introduce significant non-response bias (Lindner et al., 2001). Another 32 managers (11.7%) were rejected from the sample for lack of sufficient carnivore management data. The final sample considered for analyses included 221 respondents (523 interactions), 197 men and 24 women, who managed a total area of 26,090 km^2, or approximately 7.2% of the country's free-hold farming/tourism landscape (Mendelsohn, 2006). Except for one para-statally administered commercial cattle farm that was managed according to business principles of free-hold cattle ranches, all sampled units were privately owned. Management units in the Khomas, Hardap and Otjozondjupa regions compose 79.2% of the sample (Fig. 3.1). Therefore, this study is most representative of central Namibia's stock farming areas. Sampled management units were not randomly distributed across free-hold farmland (Ripley-K function: $z = -8.1$, $p < 0.001$). This bias is attributed to the location of three research sites in the Khomas and Hardap regions because sampled units were significantly closer to these sites than un-sampled properties (pooled Ripley-K: $t = -14.8$, $p < 0.001$).

3.3.2 Management unit characteristics

Managers controlled a median unit size of 76.2 km^2 corresponding with an average of 1.9 registered free-hold properties (median = 1, range = 1 - 14). Unit size varied between 0.5 km^2 - 1,905.5 km^2, including small, intensively managed farm plots as well as large livestock and tourism units.

Livestock production had the highest contribution to primary income as well as sample area, followed by wildlife farms, recreational tourism, business and land used for other purposes (Table 3.1). Most managers pursued dual (46.0%) or triple (26.2%) income generation strategies using 34 different combinations of land uses, most commonly involving coupled livestock-wildlife production systems ($n = 32$), livestock-recreation ($n = 30$), and livestock-business models ($n = 23$). Overall, 80.1% of respondents received some income from livestock production, 29.4% from recreational tourism, 14.0% from wildlife ranching, and 12.7% generated income from ungulate trophy hunting. Across all units with livestock ($n = 177$), 55.3% had cattle only, 13.6% produced only smallstock (goats and/or sheep), and 31.1% had a combination of cattle with smallstock. Main cattle breeds were Brahman, Afrikander, Bonsmara and Simmentaler, as well as cross-breeds between these races, and cattle stocking rates varied between 1:8.8 ha – 1:12.2 ha on units with primary income from cattle (*n* = 96). Smallstock farmers produced Dorper, Van Rooy, Damara and Blackhead Persian sheep as well as Boer goats with stocking rates of 1:0.8 ha – 1:3.4 ha on units with primary income from smallstock (*n* = 28).

Table 3.1 – Percent contribution of different land use forms to sample area and primary income on 221 free-hold management units in Namibia.

Income source/land use (sample size *n*)	**Primary income (*n* = 221)**	**Sample area (*n* = 26,090 km^2)**
Livestock (*n* = 124)	56.1	50.9
Wildlife (*n* = 33)	14.9	12.9
Recreation (*n* = 29)	13.1	25.8
Business (*n* = 29)	13.1	8.4
Others (*n* = 6)	2.8	2.0
Total	100	100

The proportions of units that used large carnivores for income generation varied significantly with primary income (χ^2 = 34.3; d.f. = 4; $p < 0.001$); 20.8% of managers received revenue from trophy hunting of large carnivores (mainly on livestock and wildlife units) whereas 15.8% promoted them as part of non-consumptive tourism (mainly on recreational units).

3.3.3 Conflict

Managers requested assistance for a broad range of topics such as carnivore tourism and reintroductions (combined *n* = 23). However, 82.8% of first contacts were conflict motivated (*n* = 183) with the major concerns being livestock predation (*n* = 152) and/or trapped individuals (*n* = 109). Managers perceived leopard and cheetah as the main conflict species (Table 3.2). These also had the highest occurrence of all large carnivores (Table 3.2). The proportion of sampled units that reported problems with any particular species was strongly associated with species occurrence (Spearman's r_s = 1.0, $p < 0.001$). Managers almost equally attributed problems to a single large carnivore (46.3%) or combinations of two or more species (53.6%), of which cheetah-leopard was the most common association (*n* = 51, 52.0%). Because of diversified revenue strategies, occurrence

Table 3.2 – Percent occurrence of indigenous large carnivore species, reported conflict and persecution on free-hold commercial properties in Namibia, 2008 - 2014. *Results represent data recorded during initial consultations.*

	Cheetah	**Leopard**	**Brown hyaena**	**Spotted hyaena**	**Lion**	**African wild dog**
Species occurrence						
All respondents (*n* = 221)	93.2	98.6	44.3	23.1	5.9	8.1
Conflict reported						
All respondents (*n* = 221)	60.2	79.0	16.0	12.2	01.1	1.7
All livestock producers (*n* = 177)	61.0	76.0	12.3	13.6	01.3	2.0
Primary income livestock (*n* = 124)	55.6	71.8	16.9	14.5	0.8	1.6
Primary income wildlife (*n* = 33)	72.7	72.7	15.2	3.0	3.0	3.0
Primary income recreation (*n* = 29)	17.2	31.0	---	3.4	---	---
Primary income business (*n* = 29)	27.6	58.6	10.3	6.9	---	3.4
Primary income others (*n* = 6)	33.3	50.0	---	---	---	---
Main conflict species (*n* = 183)	42.1	48.6	3.8	3.8	0.6	1.1
Persecution (trap and kill combined)						
All respondents (*n* = 221)	33.9	36.2	6.3	3.6	1.4	2.3
All livestock producers (*n* = 177)	31.6	46.9	7.9	4.5	1.1	1.7
Primary income livestock (*n* = 124)	33.1	47.6	7.3	5.6	1.6	2.4
Primary income wildlife (*n* = 33)	69.7	36.4	9.1	---	3.0	6.1
Primary income recreation (*n* = 29)	17.2	13.8	---	3.4	---	---
Primary income business (*n* = 29)	17.2	13.8	6.9	---	---	---
Primary income others (*n* = 6)	16.7	16.7	---	---	---	---

of conflict was not exclusive to specific land uses, or any combination thereof, but differed significantly across primary income sources ($\chi^2 = 51.56$, d.f. = 4, $p < 0.001$). It was lowest on recreational units (31.3%) and, due to anticipated or real predation impact, particularly high on livestock and wildlife production units (both >85.0% of respondents).

Livestock predation by large carnivores locally reached considerable dimensions. For example, in one year African wild dogs and leopards reportedly killed 128 un-weaned cattle calves with an estimated value of $51,106 on the largest cattle ranch. On another farm, a smallstock farmer lost 61 sheep with a projected value of $3,349 to cheetahs. Due to lower average values of sheep and goats (Meat Board of Namibia data, 2007-2014), smallstock farmers were willing to tolerate a significantly higher percentage loss of their total stock than cattle farmers before persecuting carnivores (t = 6.69, $p < 0.001$). The mean tolerance for annual smallstock losses (3.3% ± 2.4% SD, $n = 59$) also significantly exceeded reported predation (2.0% ± 2.7% SD, $n = 55$) (t = 2.87, $p = 0.002$). This was not the case for cattle loss (1.5% ± 1.4% SD, $n = 130$) and reported damage (1.2% ± 1.7% SD, $n = 132$) (t = 1.10, $p = 0.107$). Neither annual cattle ($r^2 < 0.001$, $n = 132$) nor smallstock losses ($r^2 = 0.019$, $n = 55$) were significantly associated with respective stocking rates (no. of stock/ha).

Livestock farmers with mixed cattle and smallstock production incurred highest annual financial losses from large carnivore predation ($2,727 ± $2,381 SD, *n* = 36), followed by cattle-only producers ($2,119 ± $4,790 SD, *n* = 96) and smallstock-only producers ($1,382 ± $903 SD, *n* = 19). Wildlife ranchers generally found it difficult to quantify large carnivore damage accurately but 20.4% of all respondents complained about breeding/hunting game losses. In one case cheetah predation on high-value roan (*Hippotragus equinus*) and sable antelope (*H. niger*) caused an estimated loss of $71,208 in one year.

Few livestock managers ($n = 3$) accepted annual losses as high as 10.0% of their total stock while 19.2% reported a zero-loss policy. Irrespective of stock numbers, 41.2% of cattle producers stated a fixed damage tolerance of 4 - 5 calves per year. These farmers mentioned that "*minimal losses from opportunistic predation were acceptable and budgeted for, but losses had to be halted before a habitual predation pattern manifested* (i.e. repeat predation incidents in the same area in short succession) *and became economically significant.*" Conversely, 83.9% of commercial wildlife ranchers would not tolerate any loss to carnivores. The remainder ($n = 3$) specifically stocked buffer prey species such as springbok, common impala, greater kudu, and oryx to divert predation pressure from more valuable species.

Prior to first interviews, only 40.7% of all livestock producers utilised non-lethal protection strategies. They mostly relied on simple measures such as night-kraaling of vulnerable juvenile stock (in the field or close to homes) and presence of herders during day-time hours (both 52.8%, *n* = 38). Few ranchers invested in special stock-guarding animals (domestic dogs (*C. l. familiaris*) and donkeys combined = 8.3%, *n* = 6). These mitigation efforts appeared ineffective at preventing predation. Mean annual stock losses attributed to large carnivores were not significantly different between units that employed protective measures and those that did not (smallstock: with protection = 2.0% ± 1.8% SD, *n* = 23 vs. without protection = 2.0% ± 3.1% SD, *n* = 32; t = 0.06, *p* = 0.948; cattle: with protection = 1.2% ± 1.8% SD, *n* = 61 vs. without protection = 1.2% ± 1.7% SD, *n* = 71; t = 0.20, *p* = 0.839).

A Decision-Tree interrogation of livestock losses (% annual loss, *n* = 152) that included stocking rate, unit size, farming area, year of consultation, primary income, primary conflict species, livestock protection and age class as explanatory variables showed that these factors, cumulatively, accounted for only 9.3% – 42.4% of the total variation in loss data (representing first and final split iterations (*n* = 22) respectively) (Appendix 9). They, therefore, insufficiently explained why livestock producers experienced the levels of losses they did.

3.3.4 Persecution

Mangers persecuted large carnivores on 74.7% of all sampled units, although not always successfully. Managers reported a cumulative persecution (pooled across the entire study) of 695 large carnivores through trapping and killing (Fig. 3.3) (summary statistics in Appendix 10). Managers had sought government permits for only 82 of these individuals, thus 88.2% of the total were not reported to the state wildlife department. The majority of managers had removed 0 – 3 large carnivores in the 24 months prior to first contact with the research team (Fig. 3.3), corresponding with an overall mean of 3.2 ± 7.4 SD per unit (95% CI = 2.2 – 4.2, range = 0 – 85, *n* = 219). Some, however, implemented continuous eradication campaigns with maximum annual carnivore killings of 82 cheetahs, 11 leopards, seven brown hyaenas, nine spotted hyaenas, nine lions, and 37 African wild dogs respectively. Thirteen managers contributed 41.9% to total carnivore persecution by removing ≥10 large carnivores each (Fig. 3.3) – 52.9% of all cheetahs, 12.0% of leopards, 30.2% of brown hyaenas, 42.9% of spotted hyaenas, 9.1% of lions, and 86.4% of African wild dogs respectively. Of these units, primary income included livestock (*n* = 6), wildlife (*n* = 5), and recreation (*n* = 1). The results demonstrate the potential impact of individual

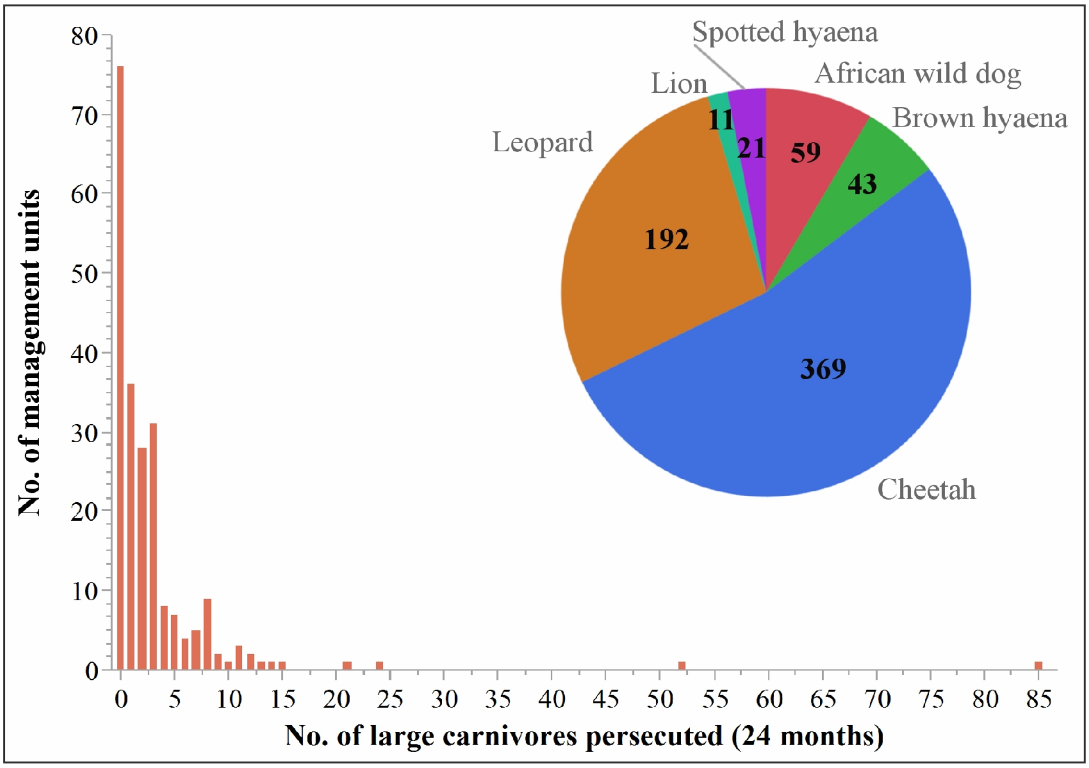


Figure 3.3 – Total large carnivore removal per free-hold management unit and species in Namibia. *Data represent persecution statistics for 24 months prior to first contact with land managers (n = 219 respondents), 2007 – 2014.*

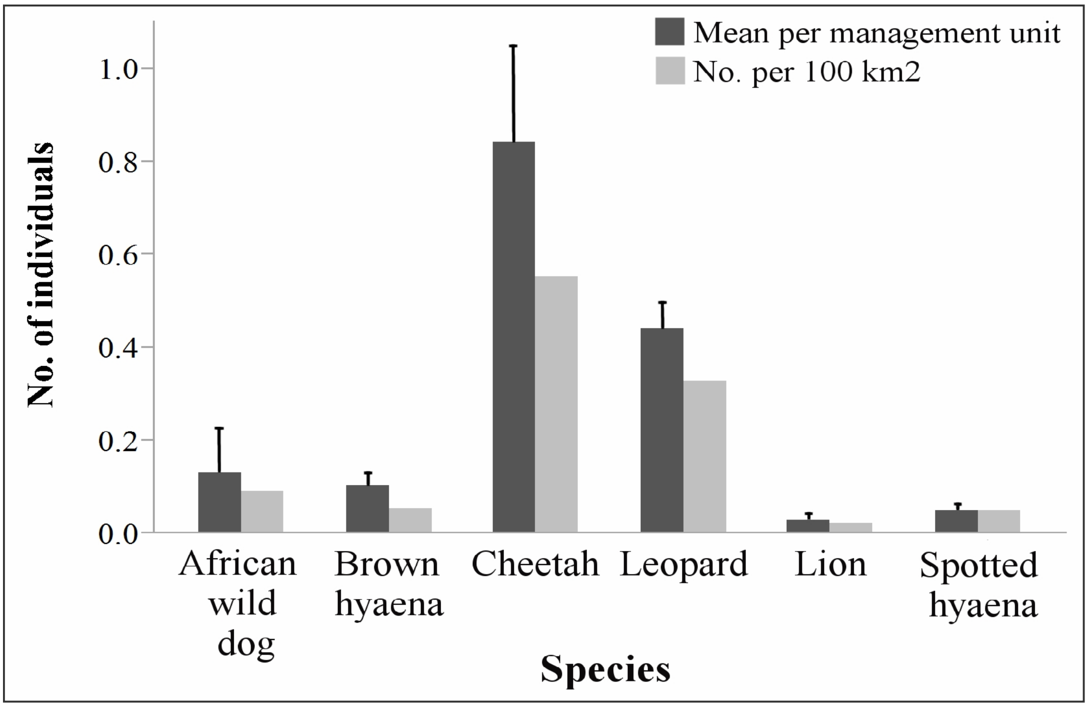


Figure 3.4 – Mean annual persecution of six indigenous large carnivores on free-hold management units in Namibia. *Bars represent mean number of individuals persecuted per property (dark grey) and per 100 km^2 (light grey). Data are pooled for the 2007 – 2014 sampling period (n = 221 respondents). Error bars show 1 SE of the mean.*

managers on free-ranging carnivore populations, resulting in local carnivore sinks, i.e. units where removal is potentially higher than natural recruitment.

Persecution of different large carnivores (proportion of units where it was attempted for each species) was strongly associated with species occurrence ($r^2 = 0.944$, $p = 0.001$, $n = 219$). Due to their current proliferation in stock farming areas, persecution particularly affected cheetahs and leopards (Fig. 3.4). Although managers perceived leopards as the biggest problem (Table 3.2), cheetahs were eliminated more fiercely because they were regarded as "*wasteful feeders*" and were "*easier to trap.*" For example, on 29 units captured cheetahs were used as live-bait to attract conspecifics and enable removal of social groups (i.e. mothers with cubs or male coalitions) (Appendix 11).

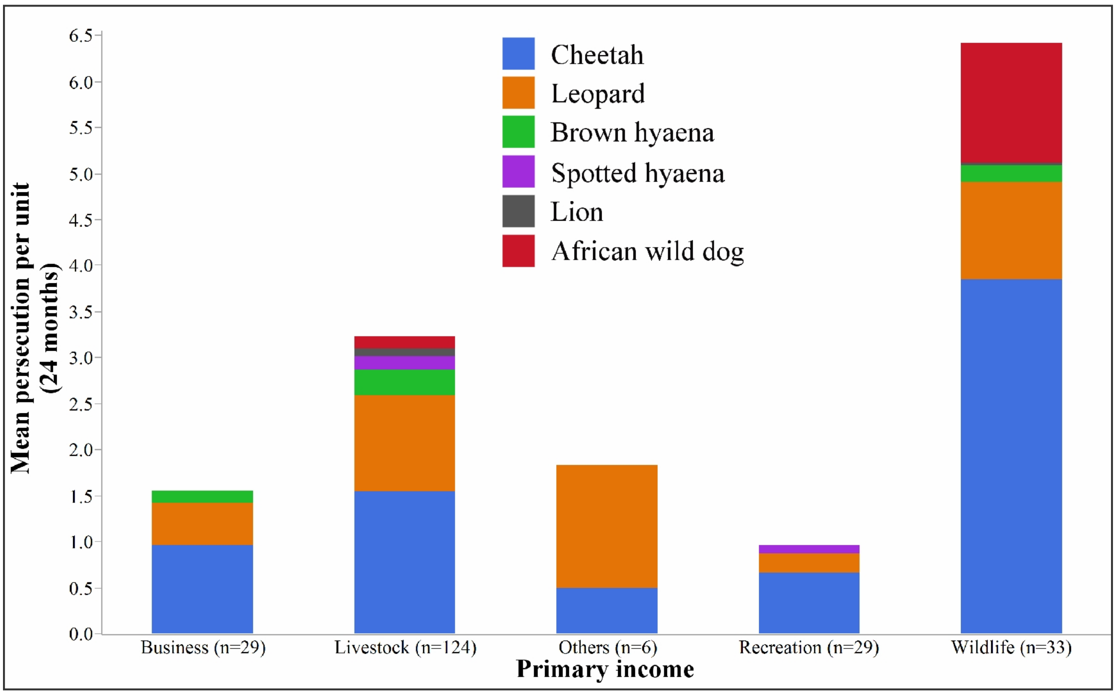


Figure 3.5 – Mean large carnivore persecution on free-hold management units in Namibia across primary income categories. *Means were calculated for 24 months prior to first contact with land managers.*

The proportion of units where persecution was practised varied significantly by primary income source ($\chi^2 = 55.99$; d.f. = 4; $p < 0.001$). There was a significant effect of primary income on average annual large carnivore removal per unit (One-way ANOVA for 24 months prior to first contact: $F_{4,219} = 2.73$, $p = 0.030$). Wildlife units had the highest removal rates (Fig. 3.5) (summary statistics in Appendix 12), specifically for cheetah and African wild dog, and disproportionately contributed to total large carnivore persecution ($\chi^2 = 409.47$; d.f. = 4; $p < 0.001$), whereas recreational units had disproportionately low impact (Fig. 3.6). Mean persecution was highest on units where large carnivores were also trophy hunted (excluding trophy animals: 3.1 ± 4.1 SD, $n = 46$). This was significantly different from units where large carnivores were promoted as part of non-consumptive tourism (0.6 ± 1.2 SD, $n = 35$) ($t = 3.40$, $p = 0.001$), and those where large carnivores did not contribute to income generation (1.4 ± 3.8 SD, $n = 140$) ($t = 2.57$, $p = 0.011$). There was no significant difference for mean persecution between the latter categories ($t = 1.12$, $p = 0.266$).

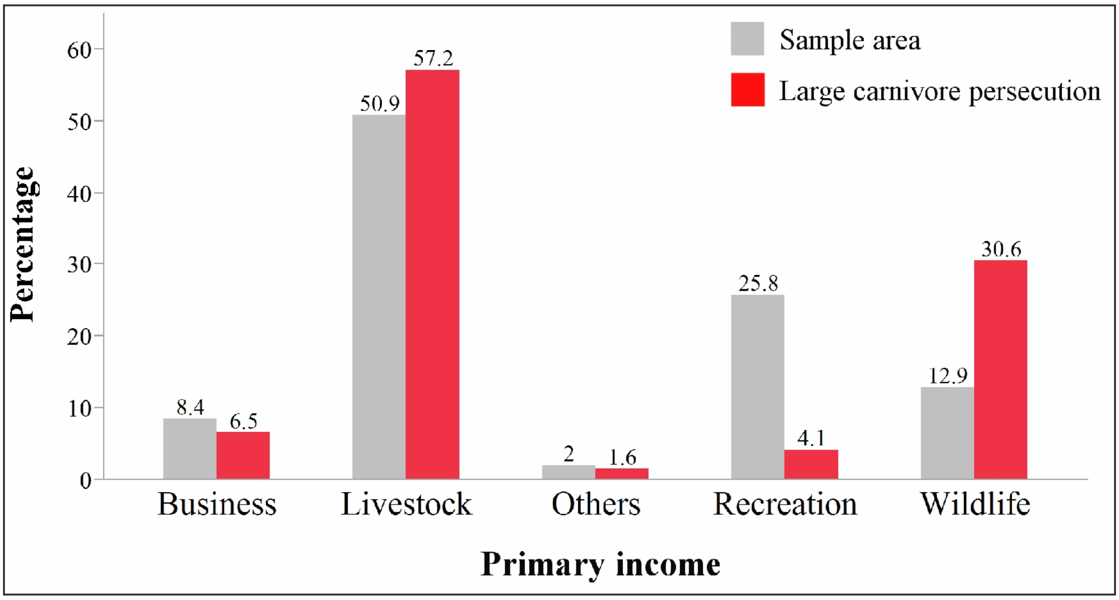


Figure 3.6 – Percent contribution of primary income categories to total sample area and large carnivore persecution in Namibia. *Graph demonstrates the disproportionately high contribution of wildlife land users to carnivore persecution and the disproportionately low impact by land users with priority income from non-consumptive recreational business. Sample area = 26,090 km*^{2}*; persecution = 695 large carnivores; persecution impact for two units with unverified removal was assumed as 0.*

Although annual losses to large carnivore predation amounted to less than the stated tolerance on most livestock units (55.2%), perceived conflict and fear of depredation often resulted in prophylactic persecution. Even semi-tolerant managers routinely persecuted large carnivores before their stated tolerance thresholds had been reached ($n = 59$ units) and reportedly when a "*distinct predation pattern*" emerged. Judged by the number of large carnivores trapped and killed on each unit, the degree of persecution was not associated with unit size ($r^2 < 0.001$, $n = 219$) or losses of cattle ($r^2 = 0.066$, $n = 132$) or smallstock ($r^2 = 0.061$, $n = 55$). The mean number of carnivores persecuted also did not significantly differ (t = 0.26, p = 0.396) between livestock units that protected stock (1.7 ± 5.1 SD, n = 75) and those that did not (1.5 ± 1.7 SD, n = 102).

Willingness to persecute, as expressed by methods used, did not significantly differ across respondent age classes (χ^2 = 0.76, d.f. = 3, p = 0.859) but the scale of carnivore removal per management unit appeared to increase with age (Fig. 3.7) (summary statistics in Appendix 13). It was nearly twice as high as in any other age class for managers between 51 – 60 years old. However, age class had no significant effect on mean carnivore removal (ANOVA: $F_{3,219}$ = 2.44, p = 0.065) but younger managers appeared to persecute large carnivores less intensively (Fig. 3.7).

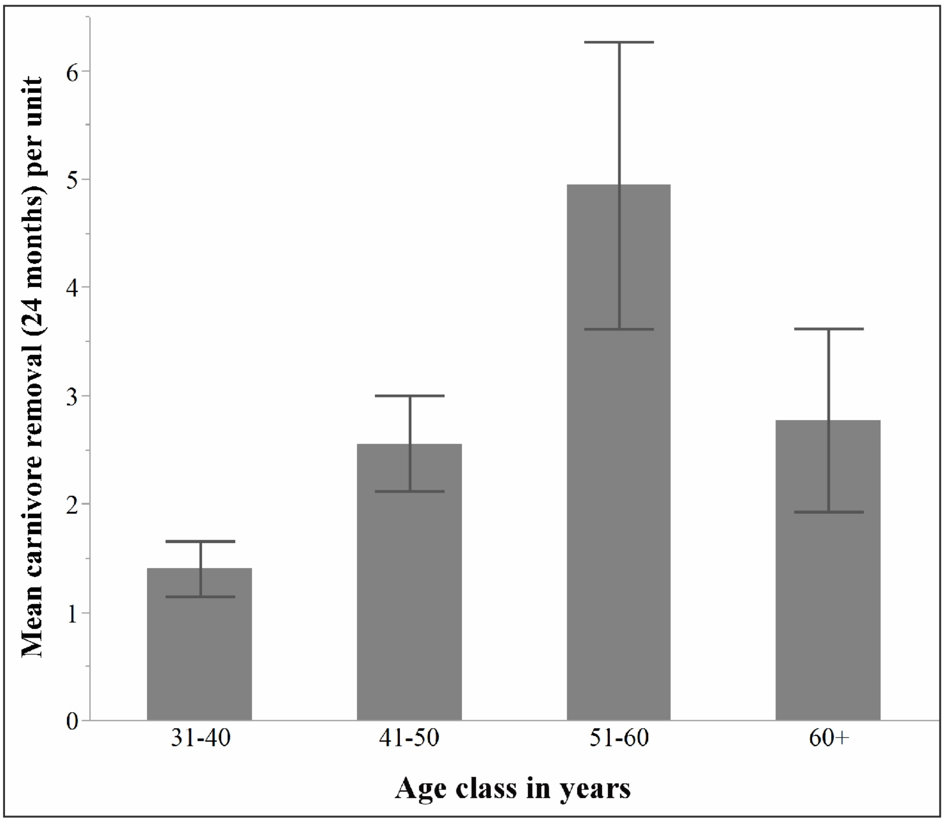


Figure 3.7 – Mean large carnivore removal per management unit by respondent age class. *Removal statistics represent total large carnivore persecution (n = 695 individuals) from 221 free-hold management units across six indigenous large carnivore species. Error bars show 1 SE of the mean.*

Mean carnivore persecution per unit (pooled for all six species and 24 months prior to first consultations) was highly variable during the study (Fig. 3.8). The general trend, which was strongly linked to the mean persecution of cheetah, decreased between 2008 and 2014 but was not significantly correlated with year of study (r_s = -0.25, p = 0.588, n = 7). These results probably were influenced by the non-random sampling design and varying sample sizes across years. In addition, they corresponded with the number of known carnivore sinks (persecution ≥10 large carnivores) recorded in different years (Fig. 3.8), demonstrating the disproportionate impact of these highly intolerant units relative to sample sizes.

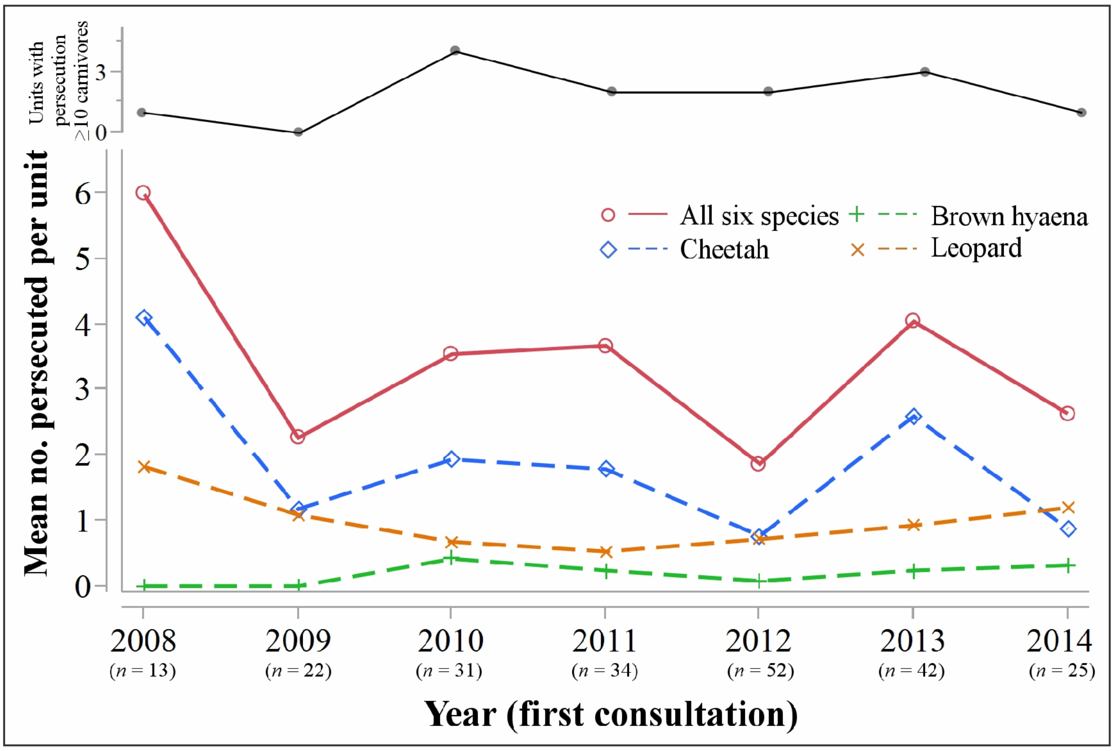


Figure 3.8 – Mean large carnivore persecution from free-hold management units in Namibia per year of study. *Curve at the top of the graph (grey line) displays the number of large carnivore sinks with a known removal of ≥ 10 large carnivores recorded in each year. Mean values represent removal for 24 months prior to first contact with managers. Two sample units were excluded due to unverified large carnivore persecution.*

Despite the strong influence of individual factors on large carnivore persecution results, a combined Decision-Tree assessment (*n* = 219) demonstrated that primary income, region, respondent age class, unit size, conflict experience, primary conflict species, asset protection, income from carnivores, and year of study accounted for only 8.0% – 35.0% (1 – 30 node split iterations, respectively) of the total variation of persecution data (Appendix 14). Approximately 57% of this variation was attributable to unit size and region (Appendix 14).

Managers acknowledged that persecution was mostly indiscriminate because it was often impractical to identify the responsible offender on extensive management units with multiple species and resident individuals. The selection of methods employed by managers reflects the indiscriminate nature of persecution (Fig. 3.9). On at least 41.4% of the units where trap cages were used (n = 128), these were not controlled on a daily basis. For 408 of the 695 persecuted carnivores, sex and age could be determined reliably (Table 3.3). Pooled for all species, the difference in proportions of sub-adults and adults was not significant (known sub-adult = 220, known adults = 188; $\chi^2 = 2.51$, d.f. = 1, $p = 0.113$) but it was skewed towards younger individuals. Similarly, the proportions of males and females affected by persecution were not significantly different (known males = 222, known females = 186; $\chi^2 = 3.17$, d.f. = 1, $p = 0.075$) but managers removed males more frequently. Twenty-nine managers maintained large carnivores in captivity (49 cheetahs, six leopards, one brown hyaena, one lion), and the same number of respondents had admitted an unknown total number of large carnivores into permanent captivity in the past.

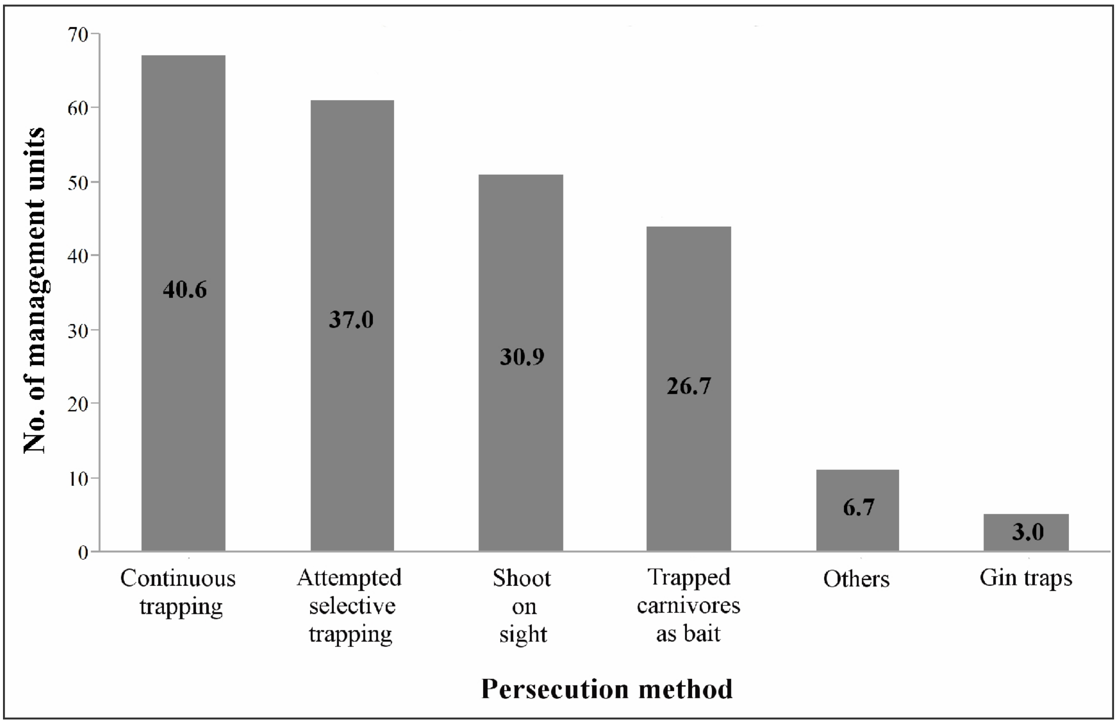


Figure 3.9 – Frequency distribution of carnivore persecution methods employed on 165 management units. *Graph demonstrates the common use of indiscriminate persecution methods. Value in bars shows percent use across all units where large carnivores were actively persecuted. Categories were not mutually exclusive. Category 'Others' includes poison, den disturbance, snares, and intentional killing with vehicles.*

Table 3.3 - Details of sex and age of persecuted large carnivores.

Species	**Male**		**Female**		**Total (total persecuted)**
	Sub-adult	Adult	Sub-adult	Adult	
Cheetah	73	62	53	41	229 (369)
Leopard	30	31	28	24	113 (192)
Brown hyaena	7	3	9	8	27 (43)
Spotted hyaena	---	---	---	---	0 (21)
Lion	1	2	1	3	7 (11)
African wild dog	7	6	11	8	32 (59)
Total	118	104	102	84	408 (695)

For those Namibian regions with a sample representation >8.0% of area (i.e. Khomas and Hardap in Fig. 3.1), the observed persecution rates per 100 km^2 (cheetah = 0.7084, leopard = 0.6694, brown hyaena = 0.117) suggest an annual removal of approximately 573 cheetahs, 502 leopards, and 43 brown hyaenas in these areas alone. These estimates are based on pooled removal statistics from 15,388 km^2 between 2008 and 2014 (127 sampled free-hold units in Khomas and Hardap with verified persecution data), corresponding with 13.0% of the entire free-hold farmland in the Khomas and Hardap regions (~118,924 km^2; Mendelsohn et al., 2002 farmland ownership data). For this particular analysis, PAs, un-sampled urban areas, communal areas, and areas without known species occurrence (according to Stein et al., 2012) were excluded. These figures, however, can only provide a crude estimation of anticipated removal because they are subject to the assumptions that:

1) There were no temporal trends in persecution;
2) Species-specific persecution reported by local managers was representative of the remainder of these regions; and
3) Land ownership did not change significantly since 2002 (Mendelsohn et al., 2002).

According to the most recent national population estimates for the species (Stein et al., 2012), the combined estimated removal in Khomas and Hardap regions represents between 4.2% – 7.5% of the country's cheetah population, 2.2% – 3.8% for leopards, and 1.8% - 7.6% for brown hyaenas.

3.3.5 Fragmentation and unit edges

There was a large degree of carnivore management heterogeneity across sampled units. Of the total, 135 units (61.1%) shared borders with at least one other sampled unit resulting in 24 conglomerates with a median number of 2.5 connected units (range = 2 – 36). Of these, seven conglomerates (29.2%) contained a known carnivore sink, i.e. a patch with a reported removal of ≥10 large carnivores (see Fig. 3.10). Only three conglomerates contained complete tolerance agreement (semi-tolerant) while 15 (62.6%) contained two tolerance types and six (25.0%) contained all three tolerance types.

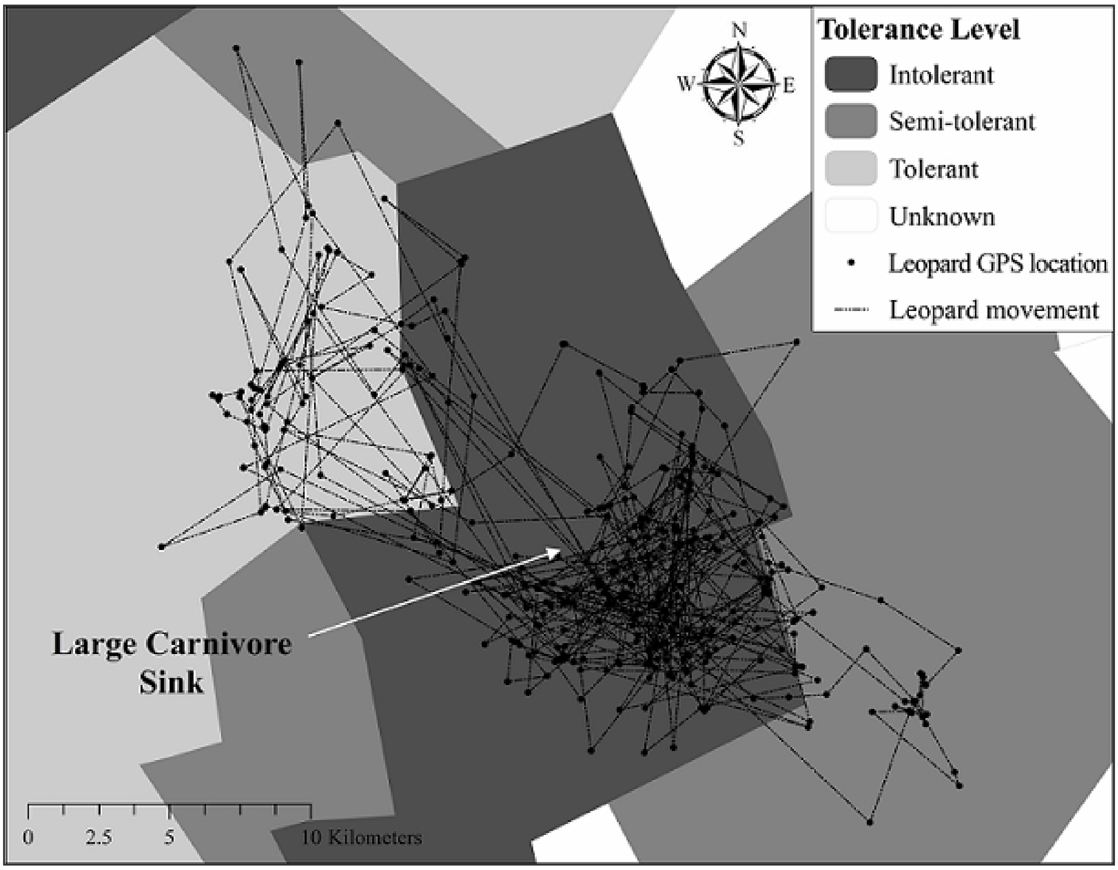


Figure 3.10 – Movements of an adult male leopard across free-hold units with different carnivore management attitudes. *Data represent 17 months of daily GPS-monitoring. Farm boundaries are obliterated to ensure anonymity of respondents. The sink indicates an intolerant patch with a known removal of two cheetahs, four leopards and nine spotted hyaenas in 18 months.*

Tolerant units were outnumbered by both semi-tolerant and intolerant units, but the large size (>300 km^2) of four ecotourism reserves (where large carnivores were not persecuted) contributed to a near equal representation of the three management types in terms of area (Table 3.4). Edge compositions between free-hold units, however, significantly deviated

Table 3.4 – Differences in large carnivore management activities across adjacent management units.

Based on 221 respondents between 2008 and 2014.

Assessment unit	**Sample size *n***				**χ^2 Goodness of Fit-Test**
		Trap with No-trap	Trap with Trap	No-trap with No-trap	
Borders between adjacent sample units	1,248.3 km	42.0%	14.4%	43.6%	$\chi^2 = 16.42$, d.f. = 2, $p < 0.001$
Borders between sample units and PAs[a]	444.1 km	30.1%	---	69.9%	$\chi^2 = 16.00$, d.f. = 1, $p < 0.001$
		Kill with No-kill	Kill with Kill	No-kill with No-kill	
Borders between adjacent sample units	1,248.3 km	51.4%	24.3%	24.3%	$\chi^2 = 14.29$, d.f. = 2, $p < 0.001$
Borders between sample units and PAs[a]	444.1 km	41.2%	---	58.8%	$\chi^2 = 3.24$, d.f. = 1, $p = 0.072$
		Tolerant	Semi-tolerant	Intolerant	
Management units	221	25.3%	36.7%	38.0%	$\chi^2 = 2.91$, d.f. = 2, $p = 0.234$
Area	26,090 km^2	34.9%	34.8%	30.3%	$\chi^2 = 0.42$, d.f. = 2, $p = 0.809$
Study carnivore locations	5,635	19.7%	55.9%	24.4%	$\chi^2 = 22.24$, d.f. = 2, $p < 0.001$

from equal distributions in terms of trapping and killing policies (Table 3.4); 56.4% of unit boundaries were unsafe in terms of trapping (i.e. risk of capture on one or both sides) and 75.5% were unsafe in terms of lethal control (i.e. risk of lethal control on one or both sides). This compares to 30.1% unsafe trapping boundaries ($z = 9.55$, $p < 0.001$) and 41.2% unsafe lethal control boundaries ($z = 12.13$, $p < 0.001$) along PAs respectively (Table 3.4). These statistics emphasise the degree of carnivore tolerance fragmentation and suggest strong local edge effects between commercial free-hold properties, significantly exceeding those observed along PAs (Table 3.4).

Using single daily locations of 23 resident large carnivores (i.e. 13 cheetahs, eight leopards and two brown hyaenas; subject details in Appendix 15) that were monitored with tracking collars on free-hold farms for 7,560 monitoring days between June 2008 and December 2014 (11.0 ± 12.6 SD monitoring months), the observed movements provide additional insight into the importance of intra-farmland edges. On average, resident study carnivores utilised 14.7 ± 15.9 SD different management units, with a minimum of three units for a leopard female and a maximum of 71 units for a dispersing cheetah male (Table 3.5). There was no significant difference in the mean number of units utilised by cheetahs and leopards ($t = 0.83$; $p = 0.412$) (Table 3.5). Pooled for the three species, the animals spent 2.0 days ± 2.9 days SD on any particular unit (range = 0 - 118 days, $n = 2{,}744$ inter-crossing time segments) and crossed a mean number of 23.0 ± 11.1 SD unit boundaries per month (range = 0 – 51, $n = 184$ months). Brown hyaenas spent least time on a specific unit before crossing into another and therefore had the highest crossing frequency per month (Table 3.5). Species had no significant effect on mean boundary crossings per months (ANOVA: $F_{2,184} = 2.11$, $p = 0.123$) and, despite some inter-specific differences in movements (Table 3.5), individuals of all three species crossed more than 20 unit boundaries per month, giving an indication of the high risk of entering unsafe areas.

Collared study animals also did not avoid unsafe farmland patches (e.g. Fig. 3.10). When areas with unknown carnivore management were excluded (1,925 monitoring days removed) there was a significant difference in the proportion of time that subjects spent on tolerant, semi-tolerant and intolerant units respectively, with >80% of all monitoring positions located on units where persecution was practised opportunistically or permanently (Table 3.4). The frequency of unit boundary transitions displayed by animals, in conjunction with management unit and edge characteristics in terms of persecution (Table 3.4), suggests a high exposure to persecution risk for carnivores that typically roam over multiple free-hold properties as part of their routine movements (Table 3.5; cheetahs in Marker, 2002; Wachter et al., 2006a; Marker et al., 2008a; leopards in Stein et al., 2011).

Table 3.5 – Utilisation of free-hold management units by different large carnivores. *Utilisation determined by tracking collar data, 2008 – 2014.*

Assessment	**Cheetah** **(*n* = 13)**	**Leopard** **(*n* = 8)**	**Brown hyaena** **(*n* = 2)**	**All** **(*n* = 23)**
No. of management units utilised				
Minimum	4	3	4	3
Mean ± SD	18.0 ± 18.3	11.6 ± 11.8	5.0 ± 1.0	14.7 ± 15.9
Maximum	71	41	6	71
No. of unit borders crossed (month)				
Minimum	0	6	21	0
Mean ± SD	24.3 ± 12.1	21.0 ± 9.3	25.8 ± 4.3	23.0 ± 11.1
Maximum	51	47	30	51
Sample (months)	73	107	4	184
No. of days per management unit				
Minimum	1	1	1	1
Mean ± SD	2.0 ± 3.5	2.0 ± 1.9	1.6 ± 1.3	2.0 ± 2.9
Maximum	118	16	7	118
Sample (inter-crossing segments)	1,618	1,056	70	2,744
Days per tolerance category (%)				
Tolerant	816 (24.6)	275 (21.7)	18 (12.0)	1,109 (19.7)
Semi-tolerant	1,828 (55.2)	1,283 (50.1)	42 (28.0)	3,153 (55.9)
Intolerant	670 (20.2)	613 (28.2)	90 (60.0)	1,373 (24.4)
Sample (days)	3,317	2,171	150	5,635

During the study, five monitored carnivores were killed (three cheetahs and two leopards), of which four were not known to cause any damage to livestock (established through regular data sharing with unit managers), and one cheetah was admitted into permanent captivity due to continued livestock predation.

3.3.6 Manager opinions on large carnivore management

As part of semi-structured interviews, managers were encouraged to present their personal view of contemporary large carnivore management in Namibia. Managers were usually aware of the endangerment and conservation status of Namibia's species. Nevertheless, the strongest opinion towards large carnivores was that they posed a danger to farming systems due to perceived local over-abundance, and thus required active management (Table 3.6). Many managers perceived carnivores as a "*controllable*" factor impacting their business operations whilst other cost parameters cannot be influenced (Table 3.6). Less than one third of private managers attached discrete economic, intrinsic or ecologic values to large carnivores, and wildlife ranchers and professional hunters, in particular, advocated the raising of trophy hunting quotas for cheetah (currently 150 per year) and leopard (currently 250 per year) to improve their consumptive value (Table 3.6). Furthermore, only a minority stated that they did not experience substantial problems from coexistence with large carnivores or considered selective removal only when absolutely necessary.

Table 3.6 – Common statements about large carnivores and their management.
Based on 221 interviews with free-hold land managers in Namibia, 2008 – 2014.

Statement (comments were not mutually exclusive)	**Percent of respondents (*n* = 217)**
Large carnivores are a considerable threat to livestock and require active control – there are too many carnivores	46.5
Large carnivore damage can be controlled whereas rising Living costs (materials, tax, vehicles, school fees etc) cannot	42.0
Selective management is necessary when conflict occurs	18.9
Carnivores generally are not a problem for farmers and coexistence is possible if sufficient efforts are made	17.5
Large carnivores have high non-consumptive tourism Value	15.2
Large carnivores have high intrinsic/ecologic value	11.1
Hunting regulations/quotas need to be improved/raised to increase the economic value of large carnivores	7.4
Large carnivores have detrimental impact on game farming but high trophy hunting value	6.9

Between 2008 and 2014, manager requests for assistance with carnivore issues (pooled initial and repeat events) progressively increased from 31 to 116 interactions (annual increase: $r_s = 0.991$, $p < 0.001$, $n = 7$). Most managers (57.9%) requested repeat help resulting in 2.4 ± 2.6 SD interactions per respondent, and a maximum of 28 events with one livestock farmer across seven years. Many managers demonstrated a motivation for non-lethal management and 108 (48.9%) suggested translocation as a potential management option. Of these, 52 managers specifically requested translocation (for 45 cheetahs, 19 leopards, nine brown hyaenas, one lion, and 17 African wild dogs), giving a mean of 7.4 ± 1.0 SD requests per year ($n = 7$). These managers opposed traditional trap-and-shoot removals and almost invariably stated that they "*did not agree with the current standard response of wildlife authorities to remove perceived conflict carnivores lethally,*" although at least 33 (63.5%) had done so in the past. Moreover, 17.6% of all managers reported that they had previously translocated large carnivores from their units, either through the state wildlife agency or a non-governmental carnivore programme.

Although this was not asked during interviews, managers also frequently offered opinions about conflict-related interactions with conservation agencies. Managers reported highly dissatisfactory interaction with the state wildlife department as well as different non-governmental carnivore conservation programmes (Table 3.7).

Table 3.7 – Manager feedback on carnivore conflict interaction with different conservation agencies in Namibia. *Based on 88 respondents, 2008 - 2014.*

Agency	**Frequency**		**Reason for response**	
	Positive	Negative	Positive	Negative
Government wildlife department	5	47	Translocation of offending animal	No response No action Lethal removal proposed
Carnivore conservation NGO	3	39	Assistance with livestock protection	No action Inadequate translocation (animals returned)
Independent research	8	1	Monitoring information shared	Wrong information

An independent research programme received more positive feedback, giving a polarised distribution of opinions about this topic ($\chi^2 = 38.39$; d.f. = 2; $p < 0.001$). Following dissatisfactory interactions with conservation agencies in the past (in a context of managing damage-causing carnivores) 13 managers reportedly "*increased lethal persecution of cheetahs and leopards,*" resulting in a significantly higher removal on these units (annual mean for 24 months before = 0.6 ± 0.4 SD vs. annual mean for 24 months after = 1.4 ± 0.5 SD) ($t = 4.05$, $p < 0.001$). Another seven respondents reportedly "*started to persecute large carnivores due to a lack of help.*" Thus, failure to address conflict appropriately in some cases compromised coexistence with large carnivores and conservation objectives.

3.4 Discussion

The results obtained from continued interaction with free-hold land managers show that conflict with large carnivores remains a widespread problem, resulting in indiscriminate persecution on many properties. Local carnivore management is highly heterogeneous, corresponding with a manager's own definition of conflict and tolerance. Differences in persecution levels result in significant edge effects and local carnivore sinks. Conflict and persecution appear strongly associated with species occurrence and were highest on commercial wildlife units. Few managers utilise protective measures and translocation of carnivores appears to be in high demand.

The role of private land managers as important carnivore custodians is receiving increasing attention worldwide (e.g. Maehr, 1990; Treves et al., 2006; Zajac et al., 2012; Athreya et al., 2013; Chapron et al., 2014; Winterbach et al., 2014). The long-term perseverance of viable, free-ranging carnivore populations primarily depends on how their coexistence with humans in extensive multi-use landscapes can be facilitated (Breitenmoser et al., 2012). Namibia provides a particularly compelling case study in that regard. Here, a small group of managers owns a large proportion of available carnivore habitat and exerts control over six species, five of which continue to decline across Africa (Table 3.8).

Table 3.8 – IUCN Red List of Threatened Species classification of Namibia's free-ranging large carnivores.

Species	Conservation status	Population trend	Source
Lion	Vulnerable	Declining	Bauer et al., 2012
Leopard	Near threatened	Declining	Henschel et al., 2008
Cheetah	Vulnerable	Declining	Durant et al., 2008
Spotted hyaena	Least concern	Declining	Honer et al., 2008
Brown hyaena	Near threatened	Stable	Wiesel, 2015
African wild dog	Endangered	Declining	Woodroffe and Sillero-Zubiri, 2012

Owing to a non-random sampling design, in conjunction with a high likelihood that conflict-affected managers will seek contact and be included in the sample, the present results may not be representative of Namibia's entire free-hold farmland in a strict statistical sense. However, the land use, income structure and carnivore occurrence characteristics recorded here are in agreement with Mendelsohn's (2006) detailed description of Namibia's commercial farming landscape, as well as those reported in another recent large-scale study involving 200 respondents and a similar proportion of free-hold units (412 farms = 28,038 km^2, Lindsey et al., 2013b). Considering that farming type and income source are key determinants of manager attitudes towards predators in private farming areas (e.g. Marker et al., 2003a; Lindsey et al., 2009a, 2013a; Thorn et al., 2015), the present study is of high relevance for assessing coexistence of land managers with large carnivores on free-hold lands. In addition, respondents provided verifiable information on carnivore management (e.g. unreported events of persecution) that typically go unrecorded (Morsbach 1986a; Marker-Kraus et al., 1996). Although this chapter can only give a summary overview of the manifold aspects pertaining to coexistence, it provides a clear picture of the frequency of conflict and the resulting persecution outside PAs.

Generally, there appeared to be no hard and fast rules as to how managers defined conflict and decided to persecute large carnivores. In Namibia, manager attitudes can be influenced by a broad range of factors such as ethnic group, misconceptions and prejudice, farming experience, and livestock management (e.g. Marker et al., 2003c, 2007; Schumann et al., 2008 ; Stein et al., 2010). In this study, which spanned several regions and a variety of production systems, conflict and persecution were mainly associated with large carnivore occurrence and land use, and to a lesser degree with age. Often, and predominantly where majority income accrued from the production of potential carnivore prey, the mere presence of a large carnivore constituted a problem and triggered indiscriminate and preventative persecution. According to Lindsey et al. (2013a), the main reasons why Namibia's private land managers do not want carnivores on their properties are livestock depredation, wildlife depredation, and the associated financial losses incurred. Here, nearly half of all managers explicitly stated that large carnivore predation was one of the few factors influencing their businesses that they could actively control through removal, even though predation impact was not (yet) economically significant. Similar findings were made in South Africa (Thorn et al., 2012). Reviewing the psychological components of human-wildlife conflicts, Treves et al. (2006) strongly emphasised that land manager attitudes are primarily defined by personal perceptions of wildlife presence and associated impacts, often irrespective of the actual financial implications.

Wildlife ranchers and hunters, those earning an income from natural carnivore prey, probably had the highest contribution to persecution because they attempt to prevent any predation of their assets (van Schalkwyk et al., 2010). Except for stocking of buffer prey, expensive electric fencing (estimated at \$782 construction cost per km and \$952 maintenance per km/year, Schumann et al., 2006) and use of swing gates as effective carnivore exclusion tactics (Schumann et al., 2006), game ranchers have few alternatives to guard valuable wildlife against free-ranging carnivores. Artificially high wildlife densities in commercial game farming areas (Erb, 2004) possibly also attract large carnivores and may lead to localised predation impact. Free-hold farmlands support an estimated 82% - 89% of Namibia's five most common ungulates (Barnes et al., 2009; Lindsey et al., 2013b) - i.e. warthog, springbok, greater kudu, red hartebeest, and oryx - all of which represent common prey species of cheetah and leopard (pers. obs.; Skinner and Smithers, 1990; Marker et al., 2003b; Bissett and Bernard, 2007; Stein and Hayssen, 2013; Voigt et al., 2014). They are also the most frequently hunted trophy species in Namibia (Humavindu and Barnes, 2003), and their live-sale ($337 median auction value in 2000) and trophy values ($398 median trophy price in 2001) (Erb, 2004) match cattle calf prices (~$391, Stein et al., 2010). Hence, and paradoxically, predation of natural prey by carnivores results in financial losses mirroring those from livestock losses. The motivation to persecute carnivores preventatively may be particularly pronounced where high-value species like sable and roan are stocked and losses result in disproportionate financial damage relative to predation events; e.g. median sales value 2000 – 2002: roan = $10,254, sable = $8,439; median trophy value (both) in 2001 = $7,630 (Erb, 2004). Managers often invest over $300,000 to establish exploitable game populations on their properties (C. Kruger, pers. comm. 2011; H. du Plessis, pers. comm. 2012). Therefore, the accelerating conversion of livestock farms into wildlife-only or mixed businesses provides suitable habitat conditions for large carnivores (Lindsey et al., 2013a) but also locally raises intolerance where high monetary value is attached to natural prey species (Marker et al., 2003a, 2007). Due to their disproportionately high impact on large carnivore safety and increasing emergence on free-hold lands, hunting and game ranches will require special conservation focus as key conflict areas in the future.

On most free-hold units, active persecution contradicted stated damage thresholds, its scale did not appear to be associated with observed conflict or attempts to reduce predation, and it did not predominantly focus on leopard as the main conflict species. Instead, cheetahs were affected more significantly because they were easier to control, for example at marking trees where social groups can be trapped without difficulty (Marker-Kraus and Kraus, 1994; Marker-Kraus et al., 1996). Many managers appeared to control what they could control, rather than the perceived or true cause of conflict. This is supported by the fact that removal was largely indiscriminate. It involved all six species (where they occurred) and entailed mostly non-selective methods of persecution.

Persecution affected both sexes as well as different age classes, therefore also including younger individuals that did not yet hunt independently or cause damage. With a specific regard to cheetah, these findings strongly contradict others (Marker et al., 2003c) who reported increasing selectivity in persecution, and the species remains the least desired of the three common large carnivores on private Namibian farms (*cf.* Lindsey et al. 2013a).

There was very little consistency in terms of persecution in areas where sampled units bordered each other, even if land management was similar across unit boundaries. Despite increasing organisation of free-hold managers in commercial conservancies that promote standardised wildlife management protocols (Schumann et al., 2008; Lindsey et al., 2013b), carnivore control actions still reflect a patch-work of individual interests and practices. Although most managers only persecute opportunistically, areas with continuous eradication campaigns frequently were embedded within more tolerant units. Whilst carnivores are tolerated on one property they may face imminent persecution on adjacent land parcels. Therefore, this study provides evidence that significant edge effects and carnivore sinks occur beyond the boundaries of PAs (Woodroffe and Ginsberg, 1998; Balme et al., 2010; Bauer et al., 2014). These intra-farmland edges compromise carnivore safety and edge effects may negatively influence the long-term viability of free-ranging populations (e.g. Woodroffe and Ginsberg, 1998; Bauer et al., 2014). The high, and potentially unsustainable, impact of individual managers on carnivores observed here corroborates other Namibian accounts (e.g. >200 cheetahs reportedly killed by one rancher, Lindsey et al., 2013a) and reiterates the importance of private managers for large carnivore conservation. This work also confirms previous experience that the vast majority of carnivore removals are not officially reported (Morsbach, 1986a; Marker-Kraus et al., 1996), and hence do not influence national decisions towards sustainable trophy hunting quotas or permit regulations for the management of conflict carnivores.

Considering that this study represents less than 10% of Namibia's free-hold farmlands, the combined effect of opportunistic and continuous persecution suggest a cumulative annual removal of at least several hundred cheetahs and leopards. The results obtained here correspond with annual cheetah removals reported for 1978 - 2005 (e.g. 650 – 890, Morsbach, 1987; 134 – 1,032, Nowell, 1996). The estimated persecution of leopards (for Khomas and Hardap regions only) exceeds that reported in permit records between 1997 and 2003 approximately four-fold (median annual removal of 122 problem animals, Appendix 16). It broadly corresponds with a recent study in north-central Namibia, which showed that about 15% of farmers routinely remove leopards and another 60% persecute the species after damage has occurred, potentially resulting in annual removals equivalent to 14% of the adult population (Stein et al., 2010). The continued indiscriminate persecution of lions, African wild dogs and spotted hyaenas in areas where they enter free-hold farms suggests that these species currently have little opportunity to recover historic distributions on private lands.

It therefore appears that despite increasing non-governmental conservation efforts since the 1990s, coexistence of humans and large carnivores still presents a major challenge in the country. Researchers and NGOs have assessed different mitigation strategies, including the use of guard animals for improved livestock protection (donkeys, Marker, 2000; dogs, Marker et al., 2005a), intensified herd and farm management (Stein et al., 2010), education campaigns (Marker, 2002), and livestock farming training courses (Schumann, 2003; Marker and Boast, 2015). These measures were reported as effective to reduce predation impacts significantly at local levels (e.g. Marker et al., 2005b; Stein et al., 2010), and thus hold promising potential to increase tolerance of cheetahs (Marker and Boast, 2015). However, the widespread occurrence of conflict and persecution suggests that mitigation protocols are not yet implemented at the necessary landscape dimensions. In this study, the vast majority of livestock and game managers reported only minimal protective efforts, if any, to prevent depredation, and neither conflict nor persecution were significantly less on units that did. Managers also openly complained about a lack of response and action when they report carnivore problems to the state wildlife department or non-governmental programmes. As a consequence, they may turn to lethal persecution or increase levels of removal, thus alienating them from conservation objectives.

Conversely though, nearly half of the free-hold managers interviewed here expressed motivation to manage carnivores non-lethally, and mainly with a view on translocation. The prevailing rationale of these managers appears to be that large carnivores may generally be tolerated if problem-causing individuals can be removed when the need arises. The high demand for translocations likely stems from an increasing use of this strategy for conflict mitigation in Namibia since the 1990s. For example, one non-governmental carnivore programme relocated over 800 cheetahs and leopards between 1997 and 2010, and most individuals were subsequently released into semi-controlled conditions on a predator-proof fenced private sanctuary (Kelly, 2014). To date, there has not been a formal assessment of these efforts and the programme was stopped due to high logistic and financial demands, and the limited carrying capacity of the fenced sanctuary (T. Hoth, pers. comm. 2014). In addition, hundreds of cheetahs have been admitted into permanent captivity between 1993 and 2006 (Marker et al., 2007). However, this option cannot accommodate the high annual numbers of large carnivores trapped in Namibia and by default does not contribute to the conservation of free-ranging gene pools.

It is evident that many land managers seek and expect assistance with trapped carnivores. However, large carnivore management in Namibia currently appears haphazard, i.e. without a uniform approach or agreed strategy. Dissonances about whether, how, and where trapped individuals should be managed prevail. In the past, conservation activities that involved translocation of large carnivores produced different results and conclusions (Table 3.9) and only two studies empirically determined both biological and conflict outcomes of these experiments (lions, Stander, 1990a; leopards, Stander et al., 1997a). As regards wild-to-wild translocations (and those involving intermediate captivity), the only consensus that could be reached during a government-initiated workshop in 2013 (attended by representatives from the state wildlife department, non-governmental carnivore programmes, and independent researchers) was that more structured and reliable information is needed (with compulsory monitoring of all events) to define the future use of translocations in Namibia (O. Aschenborn, pers. comm. 2013). Similarly, Botswana's government wildlife department is currently reviewing the use of translocations as a management strategy to address human - carnivore conflicts.

In light of these developments the following chapters explore the efficacy of translocating leopards, cheetahs and brown hyaenas from conflict farms into private, free-hold tourism reserves, which represented the most tolerant category of management units in this study. Here, carnivores are provided with near ideal conditions due to recognition of their ecological and economic benefits, provision of vast habitats without commercial livestock or game production, and a general commitment towards restoration of historic carnivore guilds. In addition, many of the established reserves connect with large state PAs (Fig. 3.2) thus increasing patch connectivity and potentially allowing for dispersal after translocation. The reserves provide testing ground for assessing translocation of conflict carnivores into areas where they continue to contribute to free-ranging populations.

Table 3.9 – Summary of reported large carnivore translocations in Namibia.

Species (no. of individuals)	**Year(s) of study**	**Origin of animals**	**Recipient area**	**Objective**	**Conclusions**	**Source**
Cheetah (>150)	1960s	Private free-hold farms	Etosha NP	Population supplementation	No formal assessment but recommendations for improved introduction protocols provided; Known failure in 2 monitored cases – animals killed after leaving PA	Ebedes, 1970
Cheetah (30)	1970	Private free-hold farms	Etosha NP	Reintroduction	No formal assessment but recommendations for improved introduction protocols provided	du Preez, 1970
Cheetah (77)	1992 – 1999	Private free-hold livestock and game farms	Free-hold farms PAs South Africa Pas Zambia	Conflict mitigation Reintroduction Reintroduction	Suggested management strategy for cheetahs undesired on livestock and game farms; Successful re-introduction in 2 PAs in South Africa	Marker, 2002; Marker et al. 2003c
Leopard (2+4)	1968	Private free-hold farms	Etosha NP	Population supplementation	Failure – 2 leopards left PA and were killed; 4 leopards with unknown outcomes	Ebedes, 1970
Leopard (6)	1992 – 1995	Tsumkwe communal areas	Tsumkwe communal areas	Conflict mitigation	Temporary success but all leopards returned and eventually resumed depredation	Stander et al., 1997a
Lion (69)	1985 – 1988	Private free-hold farms surrounding Etosha NP	Etosha NP	Conflict mitigation	Partial success – translocation suggested management strategy for opportunistic livestock raiders, but not for habitual raiders (lethal control proposed instead)	Stander, 1990a
African wild dog (24)	1978, 1989, 1990	Captive rearing programmes	Etosha NP	Reintroduction	Failure – most subjects died, species not re-established through reintroduction	Scheepers and Venzke, 1995

3.5 Key points

1. Conflict with and persecution of large carnivores remains wide-spread across Namibia’s commercial farmlands.
2. Persecution was strongly associated with species occurrence – it affects all six large carnivores but particularly the two most common species, cheetah and leopard, resulting in annual removals of at least several hundred individuals.
3. Persecution is further influenced by land use, with highest levels observed on commercial wildlife units and lowest levels on recreational units.
4. Local large carnivore management is highly heterogeneous and indiscriminate in terms of carnivore security resulting in fragmented landscapes and significant edge effects. Individual managers disproportionately contribute to persecution, resulting in local carnivore sinks.
5. Management of conflict often appears haphazard and contradicts observed levels of damage and stated levels of tolerance. Despite availability of conflict mitigation options, few managers utilise preventative measures.
6. Although rarely evaluated, translocations have been used widely and continue to be in high demand for management of perceived conflict carnivores.

Chapter 4 Leopard (*Panthera pardus* – Linnaeus, 1758)

Disclaimer: Monitoring data for non-translocated leopards were generously provided by:
Individuals: PP51, PP54, PP56 – Dr Andrew Stein (Landmark College, USA)
Individuals: V10938, V11387 – Dr Ken Stratford (Ongava Research Centre, Namibia)
Individual: Pp48 – Johann Vaatz (Düsternbrook Guest Farm, Namibia)

4.1 Introduction

The leopard is the most widely distributed of all large felids (Nowell and Jackson, 1996) and occurs in nine recognised sub-species (Uphyrkina et al., 2001) across Africa, the Middle East, Asia and into the Russian Far East (Henschel et al., 2008; Stein and Hayssen, 2013). Due to significant range and population declines (Ceballos and Ehrlich, 2002; Ray et al., 2005), the conservation status of the species has recently been changed from "Least Concern" to "Near Threatened" by the IUCN (Table 3.8) and now is proposed as "Vulnerable" (A. Stein, pers. comm. 2015). Despite accelerating habitat fragmentation and anthropogenic threats (Ray et al., 2005; Henschel, 2009; Swanepoel et al., 2013, 2014; Toni and Lodé, 2013), the African leopard (*P. p. pardus*) still occurs throughout much of sub-Saharan Africa, whilst several other sub-species face increasing endangerment (Henschel et al., 2008). For example, in Russia's Far East, only few individuals of the Amur leopard (*P. p. orientalis*) remain (Miquelle and Murzin, 2003) and the population is at below viable capacity (Uphyrkina et al., 2001). Such a small population likely will depend on active management for persistence (Hebblewhite et al., 2011).

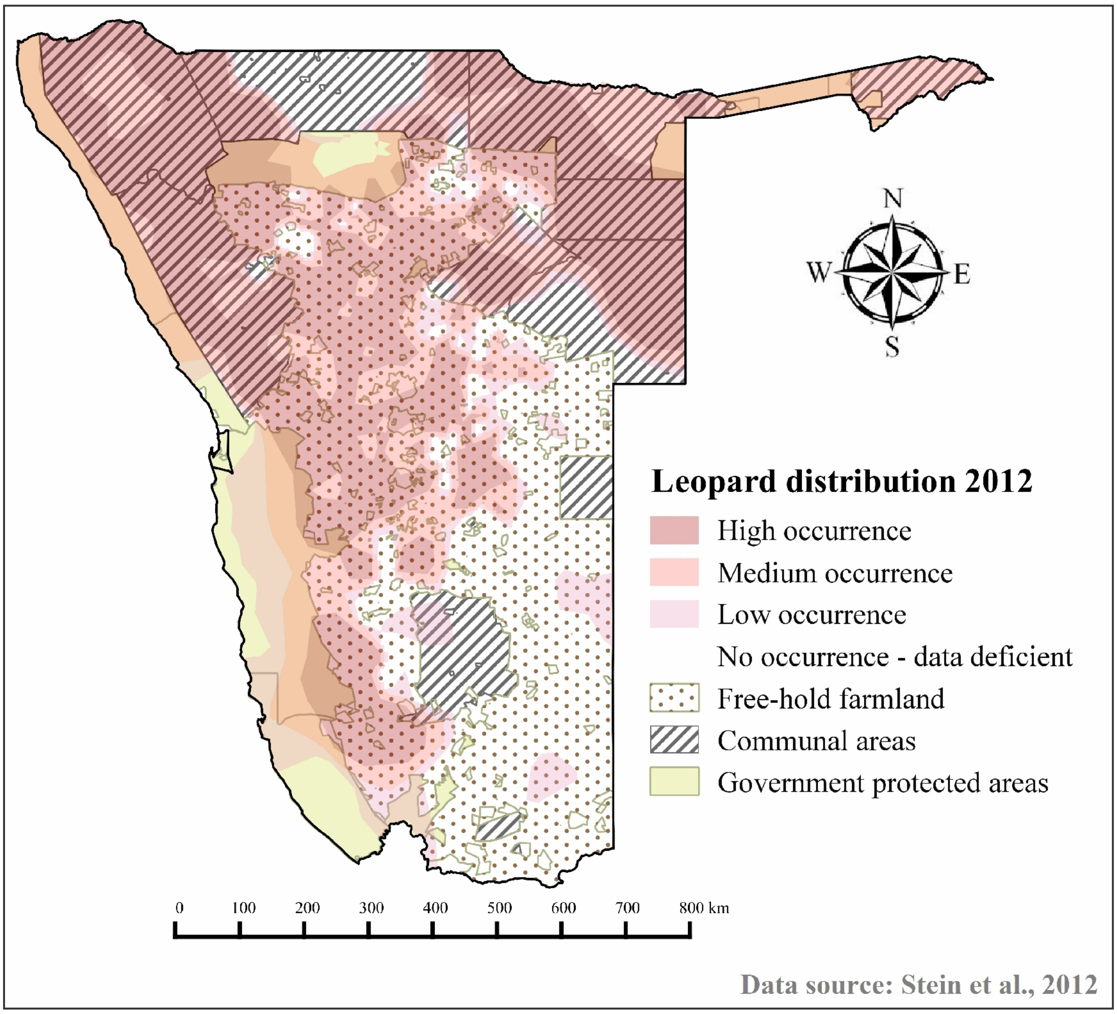
The leopard's wide distribution and perseverance are attributed to its remarkable flexibility in terms of habitat and food tolerance (Bothma and Le Riche, 1984; Stuart and Stuart, 1993; Sunquist and Sunquist, 2002; Stein and Hayssen, 2013). Leopards can persist in environments ranging from arid and semi-arid deserts to savannah and forest habitats as well as extreme mountain regions (summarised in Stein and Hayssen, 2013). Activity and range patterns vary with habitat conditions and are influenced by available prey biomass (Carbone and Gittleman, 2002; Marker and Dickman, 2005; Hayward et al., 2007c), human disturbance and presence of sympatric carnivores (Karanth and Sunquist, 2000; Ngoprasert et al., 2007; Wang and Macdonald, 2009; Odden et al., 2010; Henschel et al., 2011; Carter et al., 2015; du Preez et al., 2015).

In addition, leopards demonstrate an extraordinary degree of adaptability to human-dominated landscapes (Athreya, 2006; Athreya et al., 2007, 2013, 2014) where they coexist in close proximity with people (Odden et al., 2014). Here they may largely subsist on domestic prey (Goyal et al., 2000; Mukherjee and Misha, 2001; Athreya et al., 2007, 2013, 2014; Odden et al., 2014; Shehzad et al., 2015) whereas their natural food preferences include wild ungulates of 10-40 kg body weight (Hayward et al., 2006b; Hayward, 2009; Stein et al., 2015). Individuals specialise according to local prey availability and will consume any food they are able to kill, including insects, reptiles, fish, birds and small to large mammals (Ott et al., 2007; summarised in Stein and Hayssen, 2013) but also carrion and refuse (Hamilton, 1981). There are very few environmental conditions that leopards cannot tolerate.

In areas where the species coexists with humans, conflict is inevitable as leopards often pose a threat to people's livestock (e.g. Stander et al., 1997a; Schiess-Meier et al., 2007; Athreya et al., 2014) and may locally inflict considerable economic damage (chapter 3; Stuart, 1986; Wang and Macdonald, 2006; Stein et al., 2010; Jorge et al., 2013). Livestock depredation is a common problem throughout much of the leopard's extant range (e.g. Athreya et al., 2007; Ott et al., 2007; Romañach et al., 2007; Kissui, 2008; Sangay and Vernes, 2008; Stein et al., 2010; Yirga et al., 2011; Kala and Kothari, 2013; Farhadinia et al., 2014; Shehzad et al., 2015). The species is also potentially dangerous and sometimes compromises human safety (Corbett, 1947; Athreya et al., 2007; Hamilton, 1981; Treves and Naughton-Treves, 1999; Kumar and Chauhan, 2011; Farhadinia et al., 2015). Although leopards develop intricate spatio-temporal strategies to avoid dangerous encounters with people (Odden et al., 2014) and generally do not seek livestock as their preferred prey (Ott et al., 2007; Stein and Hayssen, 2013), their high tolerance of human activities renders them particularly prone to conflict involvement.

In many areas of its current distribution, the leopard is regarded as the primary conflict carnivore (chapter 3; Hamilton, 1981; Athreya et al., 2004; Wang and Macdonald, 2006; Schiess-Meier et al., 2007; Sangay and Vernes, 2008) leading to indiscriminate persecution in private or communal multi-use landscapes (Balme et al., 2010; Stein et al., 2010) where they are trapped and killed in potentially unsustainable numbers (chapter 3; Cobb, 1981; Hamilton, 1981; Esterhuizen and Norton 1985; Athreya et al., 2007; Sanei et al., 2012; Swanepoel et al., 2014; Pitman et al., 2015). In Namibia, most leopards live in communal or commercial farming areas (Fig. 4.1).

Due to persistent conflicts, some state wildlife departments initiated translocation campaigns of 'problem' leopards to remove offenders into PAs (Hamilton, 1981; Athreya, 2006; Weilenmann et al., 2010). Similar approaches have also been employed by NGOs in Namibia during the last decade (Kelly, 2014). In other situations, translocations were used to reintroduce leopards (Hayward et al., 2006a), to augment low-density populations to improve ecosystem integrity and tourism (Ebedes, 1970; Hayward et al., 2007a), and for releases of confiscated or rescued individuals (Houser et al., 2011; Mondal et al., 2013; Odden et al., 2014).



Data source: Stein et al., 2012

Figure 4.1 – Leopard distribution across Namibia’s main land tenures.

The scientific knowledge on leopards has increased substantially over the last three decades (Pitman, 2012). However, publications often do not address practical conservation management topics and are biased towards ecological descriptions of protected populations (Balme et al., 2014). Where conflict leopard translocations were evaluated, researchers reported very limited success due to poor release site fidelity (Weilenmann et al., 2010), homing behaviour and continued livestock depredation (Stander et al., 1997a). Many efforts resulted in delayed lethal control of individuals (Hamilton, 1981). In extreme cases, leopards caused increased conflict and human fatalities around release sites (Athreya et al., 2007, 2011). It appears that saturating conservation areas through continued releases into habitat with established conspecifics can give rise to undesirable outcomes and hence promote conservation failures (Hamilton, 1981; Weilenmann et al., 2010; Athreya et al., 2011). Although some case studies were more encouraging (Hayward et al., 2006a; Houser et al., 2011; Mondal et al., 2013), leopard translocations are widely regarded as an ineffective tool (Cobb, 1981; Hamilton, 1981; Weilenmann et al., 2010; Athreya et al., 2011). This has led wildlife authorities and researchers to question whether translocation should be a legal management option (Athreya, 2006; O. Aschenborn, pers. comm. 2013; M. Flyman, pers. comm. 2014).

In addition, small sample sizes and inconsistent monitoring complicate conclusions for this highly adaptable and opportunistic species. In Kenya and India, hundreds of leopards were translocated using methodologies that may have negatively influenced outcomes, e.g. by releasing stressed and injured individuals repeatedly into the same areas, and often without any structured monitoring of these events (Hamilton, 1981; Athreya, 2006). Prior to this study, the cumulative understanding of leopard post-translocation behaviour was based on 30 individuals for which some published information (not necessarily detailed) was available (combined from Ebedes, 1970; Hamilton, 1976, 1981; Stander et al., 1997a; Hayward et al., 2006a; Weilenmann et al., 2010; Houser et al., 2011, Mondal et al., 2013; Odden et al., 2014). These data (at the end of December 2014) included only six animals equipped with modern GPS transmitters and most individuals were followed for less than one year after release, in part due to failure of monitoring equipment (Hamilton, 1981; Weilenmann et al., 2010; Mondal et al., 2013). Frequently the outcomes of events were speculative or unknown (Hamilton, 1981).

Leopard translocations still happen across Africa (M. Flyman, pers. comm. 2014; B. Cain, pers. comm. 2015) and in Asia (e.g. Athreya et al., 2015). In several countries, translocation of ‘problem’ leopards remains the legally prescribed response to conflict (Ministry of Environment and Forests, 2011; M. Flyman, pers. comm. 2014) and the strategy may also be required (or become essential) for reintroduction of the critically endangered Amur leopard in order to promote reproduction, range expansion and population recovery (Uphyrkina and O’Brien, 2003; Hebblewhite et al., 2011). It is therefore imperative to define the circumstances that enhance translocation success.

In this chapter, the outcomes of six intensively monitored leopard translocations are evaluated in terms of leopard ecology and conflict mitigation. The results are assessed by comparing them with information from 12 resident conspecifics (including one short-distance translocation) and previous studies of the topic. Translocation protocols are investigated and a pragmatic, replicable modelling approach is used to determine suitable leopard recipient habitat across Namibia’s PAs. The aim of this chapter is to provide a more detailed understanding of leopard translocation outcomes, and their influencing factors, while assessing the potential role of this strategy with regard to leopard management in Namibia.

4.2 Methods (supplemental to chapter 2)

4.2.1 Resident (non-translocated) leopards

With permission from the Ministry of Environment and Tourism (permit numbers in chapter 2), resident leopards were trapped for research purposes (four independent research programmes) in different bio-geographic areas of Namibia (Fig. 4.2). In addition, resident subjects included indiscriminate captures and individuals believed to be opportunistic livestock raiders (Table 4.1) that were trapped by and subsequently monitored with private free-hold land managers.

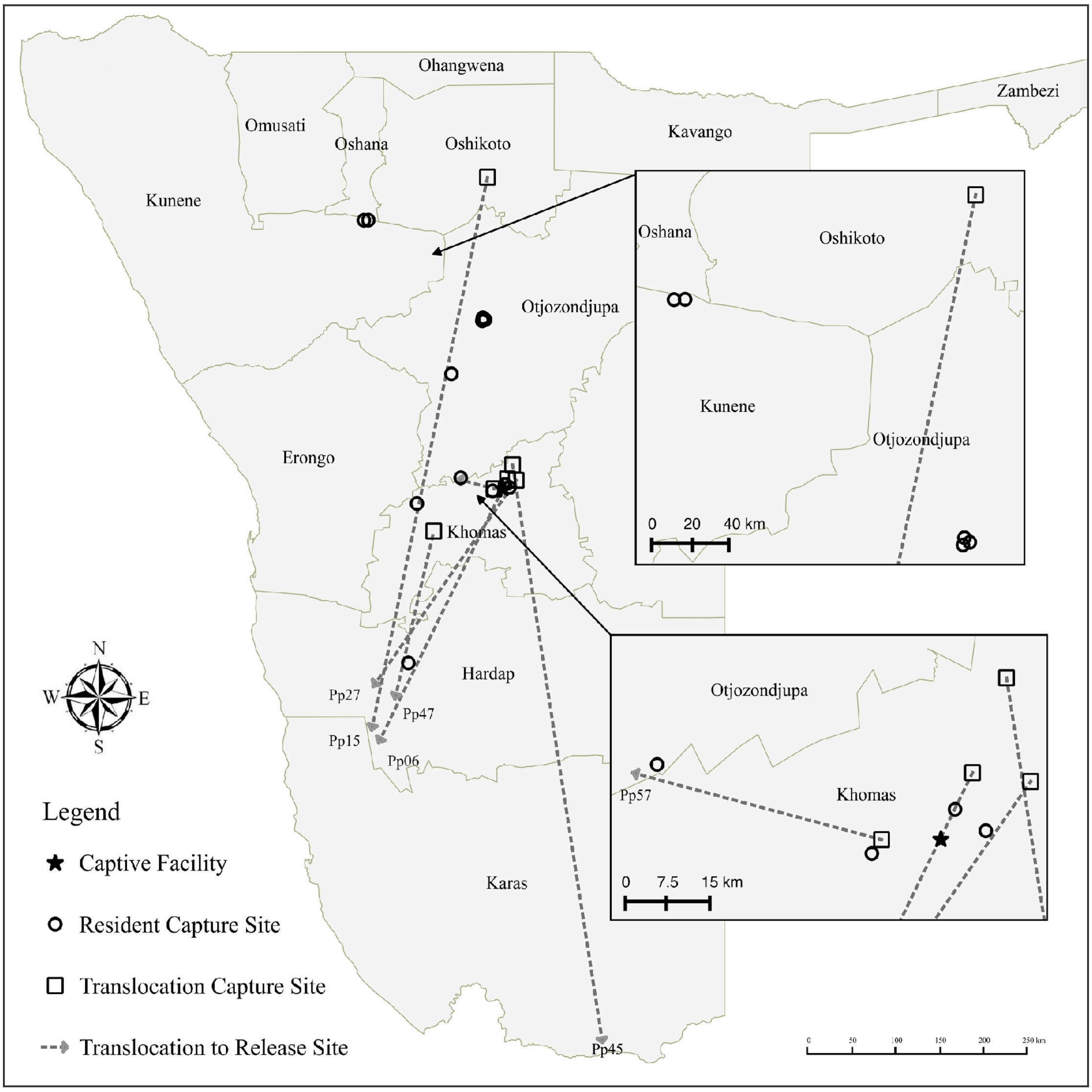


Figure 4.2 – Sample distribution of translocated and resident study leopards across Namibia, 2004 – 2014. *Inset boxes show magnified views of areas with clustered locations. ID numbers indicate translocated case studies.*

Trapped resident leopards were attended to within 24 h of capture. The animals were immobilised with suitable combinations of Ketamine with α2-agonists (Xylazine or Medetomidine) using the same protocols described in chapter 2 (individuals: Pp04/05/62/63/71/77). Other anaesthesia protocols included a standard dose of 350.0 mg Zoletil^{TM}, a combination of Tiletamine and Zolazepam hydrochloride (individuals: V10938/11387), or 4.0 – 6.0 mg/kg Telazol^{TM} (also Tiletamine/Zolazepam) (individuals: PPA51/54/56/Pp48) (Kreeger, 1996). Individuals V10938 and V11387 received a second dose of 50.0 mg Zoletil because full administration of the initial doses was not certain (K. Stratford, pers. comm. 2015). Tiletamine-Zolazepam combinations were not reversible and leopards immobilised with these drugs were placed into safety boxes to allow for undisturbed recovery from anaesthesia. Except for these variations in anaesthesia protocols, and site-specific differences in monitoring technology (Table 4.1), the handling protocol of resident leopards resembled that of translocated individuals (see section 2.2 in chapter 2). However, resident leopards were not moved into captivity and released at, or close to, their respective point of capture following complete recovery from anaesthesia. No adverse effects were observed as part of capture and immobilisation protocols.

4.2.2 Potential leopard recipient areas

To determine area suitability for future translocations throughout Namibia, the ArcGIS-based geospatial model *CaTSuiT* (Lemeris Jr., 2015) was modified specifically for leopards. The baseline model (Lemeris Jr., 2013) was transformed with the most recent data on national leopard density (Stein et al., 2012) and distribution (P. Gerngross, pers. comm. 2014), government PAs and private reserves (Mendelsohn et al., 2002; ConInfo 2010, 2011), and designated urban settlements (Namibia Statistics Agency, 2012) as input parameters.

Within the model, potential recipient areas were identified through step-wise discrimination of unsuitable habitat. First, suitability was restricted to public and private PAs to minimise the possibility of post-release livestock predation and consequently persecution. Secondly, the remaining area was constricted to include reported low-to-medium leopard densities only (according to Stein et al., 2012) as these were assumed to be able to accommodate an individual immigrant (Hayward et al., 2006a), rather than areas with a known high density approaching ecological carrying capacity. Areas with high or zero leopard density were excluded as they likely represented saturated (Weilenmann et al., 2010) or unsuitable/unsafe habitat, respectively. Considering the potential dangers from leopard overspill into human settlements surrounding recipient sites (Athreya et al., 2007,

Table 4.1 - Details of resident and translocated study leopards in Namibia, 2004 – 2014.

ID	Sex	Estimated age (years)	Weight (kg)	Capture region (release region)	Year[a]	Capture reason	Captivity (days)	Translocation distance (km)	Release Mode	Transmitter type
Residents										
PPA51	F	3 – 5	31	Otjozondjupa	2004	Research	n/a	n/a	Hard	VHF[c]
PPA54	M	5 – 7	54	Otjozondjupa	2004	Research	n/a	n/a	Hard	VHF[c]
PPA56	F	3 – 5	34	Otjozondjupa	2005	Research	n/a	n/a	Hard	GPS[d] satellite
Pp04	F	3 – 5	30	Khomas	2008	Research	n/a	n/a	Hard	GPS[e] cell
Pp05	M	9 – 12	63	Khomas	2008	Livestock raider	n/a	n/a	Hard	VHF[c]
Pp48	M	2 – 4	Unknown	Khomas	2010	Research	n/a	n/a	Hard	GPS[e] cell
Pp62	M	6 – 8	61	Khomas	2012	Livestock raider	n/a	n/a	Hard	GPS[e] satellite
Pp63	M	3 – 5	54	Khomas	2012	Indiscriminate	n/a	n/a	Hard	GPS[e] satellite
Pp71	F	4 – 6	38	Hardap	2013	Indiscriminate	n/a	13[b]	Hard	GPS[e] satellite
V10938	F	3 – 5	31	Kunene	2013	Research	n/a	n/a	Hard	GPS[d] satellite
V11387	M	5 – 6	48	Kunene	2013	Research	n/a	n/a	Hard	GPS[d] satellite
Pp77	F	4 – 6	36	Otjozondjupa	2014	Indiscriminate	n/a	n/a	Hard	GPS[e] satellite
Translocations										
Pp06	M	4 – 5	44	Khomas (Hardap)	2008	Livestock raider	16	369	Hard	VHF[c]
Pp15	F	4 – 6	35	Oshikoto (Hardap)	2009	Confiscation – rehab	168	715	Hard	GPS[f] ARGOS & VHF[e]
Pp27	F	2	31	Khomas (Hardap)	2009	Orphan – rehab	639	313	Hard	GPS[e] cell & GPS[e] satellite
Pp45	M	2 – 4	52	Khomas (Karas)	2011	Livestock raider	206	754	Hard	GPS[e] satellite
Pp47	M	2 – 4	42	Khomas (Hardap)	2011	Livestock raider	183	218	Soft	GPS[e] satellite
Pp57	F	4 – 6	39	Khomas (Khomas)	2012	Livestock raider	4	47	Hard	GPS[e] satellite

[a] Year of release
[b] Removed from a livestock enclosure but released within expected home range.
[c] Advanced Telemetry Systems, Insanti, USA
[d] Vectronics Aerospace, Berlin, GER
[e] Africa Wildlife Tracking, Pretoria, RSA
[f] Sirtrack, Hawkes Bay, NZ

2011), a circular 50 km safety buffer was designated around urban centres. Each of these steps was considered a minimum suitability criterion during the identification process and therefore had a 100% influence on recipient habitat suitability when applied. The order of application did not influence the final results.

In order to reduce the risk of displacement by conspecifics as a possible result of intra-specific aggression (Bailey, 1993; Hamilton, 1976, 1981; Balme et al., 2009), a conservative minimum inter-release interval of 18 months was designated for each suitable area. To increase the probability of site fidelity, any habitat patches that were smaller than the squared value of the furthest post-release distance moved by a leopard before settling (adopted from site fidelity formula in Spencer et al., 1990) were removed unless they connected with other suitable areas.

The potential number of leopard translocations in Namibia was determined by gauging the size of predicted suitable patches against the movements of the furthest-travelling leopard in this study (Pp57 in section 4.3.4). This resulted in a distinction of predicted areas into ‘stand-alone’ (those that were larger than Pp57’s squared movements) and ‘connected’ patches (those that were smaller but contributed to a joint area large enough to accommodate Pp57’s squared movements). The final estimate was derived by calculating the number of stand-alone and connected reserves that could simultaneously accommodate Pp57’s squared movements without overlap between hypothesised events.

Predicting leopard recipient areas would have benefited from including prey biomass as a key measure of habitat suitability (Hayward et al., 2007c; Hayward, 2009). However, reliable information was not available for the majority of prey species at national scale. Where it was available (e.g. Mendelsohn et al., 2002) data were significantly outdated by the time of this assessment and no longer reflected the true circumstances (pers. obs.). Prey sufficiency needs to be determined case-specifically with PA managers and should follow considerations presented by Hayward (2009). In addition, the leopard’s wide prey spectrum (Stein and Hayssen, 2013) permits the assumption that individuals adapt to available local resources at release sites.

Moreover, African leopards occurring in areas with high densities of dominant competitors (lion and spotted hyaena) avoid kleptoparasitism of kills by hoisting prey (Stein et al., 2015), they avoid direct competition by spatio-temporal separation (e.g. Vanak et al., 2013), and they suffer higher intra-specific mortality than inter-specific mortality (e.g. Balme et al., 2009). Therefore, the effects of kleptoparasitism and inter-specific competition were considered negligible and no adjustments were made in the model.

4.3 Results

4.3.1 Sample characteristics

Between 2008 and 2014, the outcomes of six leopard translocations (three males, three females) were investigated for 4,309 monitoring days. In addition, 12 resident leopards (six males, six females) were studied across four regions of Namibia for a total of 5,614 monitoring days between 2004 and 2014 (Fig. 4.2, Table 4.1). At the time the study was concluded (12/07/2014), one translocated individual and two residents were still being monitored. Of all translocated subjects, four were opportunistic livestock raiders and two were rehabilitation cases (one orphan and one confiscated long-term captive). The average captive time for translocated leopards was 202.6 days ± 230.8 days SD (range = 4 – 639 days). The animals were released at a mean distance of 402.7 km ± 279.6 km SD (range = 46.5 – 714.9 km) from their original capture sites. Translocated individuals were not habituated to human presence and retained wild characteristics during captivity. All resident leopards and five of the six translocated leopards were hard released (Table 4.1). Female Pp57 was released into an area with a higher estimated density of conspecifics (3.0 – 4.0 leopards / 100 km^2) and using the shortest distance (Table 4.1). Five of the translocations were into areas where those individuals experienced novel intra-guild competition with spotted hyaena. Although female Pp71 was moved 13.1 km from her capture site in a livestock enclosure, she was classed as a resident as she was not moved from her expected familiar terrain. Translocated leopards were monitored with combined GPS satellite - VHF transmitters (*n* = 5) or VHF transmitters (*n* = 1). When their original transmitters were depleted, the GPS unit of female Pp15 was replaced by a VHF transmitter, and female Pp27 was fitted with a second GPS unit.

4.3.2 Survival

None of the 18 study leopards was directly persecuted by landowners during the monitoring period but four individuals died from different causes. Resident male Pp05 succumbed to a severe gastro-intestinal inflammation (post-mortem results, I. Baines, pers. comm. 2008). Male Pp62 was shot by a farmer who investigated a livestock kill and was charged by the leopard. This incident occurred after the male’s collar was depleted. Female Pp71 was caught in a wire snare set for poaching of warthog and steenbok (*Raphicerus campestris*). Translocated female Pp57 died in a vehicle collision 110 days after release. An investigation of her carcass produced no evidence that the female had been shot or emaciated prior to this accident - an underlying medical condition or chronic stress effect, however, cannot be ruled out.

In this study, novel intra-guild competition with spotted hyaenas had no negative influence on the survivorship of translocated leopards. The mean annual Kaplan-Meier survivorship estimate (entire study) for translocated leopards (mean = 0.9, SD = 0.1, $n_y = 7$, one mortality) was not significantly different from that of residents (mean = 0.8, SD = 0.2, $n_y = 10$, three mortalities) (Mann-Whitney U-Test: U = 41.0, $p = 0.295$). For translocated leopards the probability of surviving two years after release was 0.83 (95% CI: 0.45 – 0.99) and 0.67 (95% CI: 0.35 – 0.89) for resident subjects and survivorship did not significantly differ between the groups ($z = 0.74$, $p = 0.459$). Long captive periods for females Pp15 and Pp27 did not appear to compromise their survival. Female Pp27 was still alive over four years after release and female Pp15 was 'censored' in year three after translocation.

Figure 4.3 – Camera trap record of translocated female leopard Pp27 (right) with 13-months old male cub (left) from her first litter of two.

4.3.3 Reproduction

Both surviving translocated females reproduced and successfully raised young ($n = 5$; two female cubs, one male cub, two unknown) in their novel environments (Fig. 4.3). Considering a gestation period of up to 106 days (Skinner and Smithers, 1990), Pp15 conceived about eight months post-release. Therefore, translocated leopards positively contributed to the free-ranging gene pool in year two after release when offspring recruitment exceeded the death of Pp57. In year three, successful recruitment exceeded known deaths by a ratio of 5:1. Furthermore, two of three translocated males (Pp06 and Pp45) were observed during courtship behaviour with wild females and Pp45 mated on two separate occasions. Courtship was first noted five months after release and confirmed

Table 4.2 – Movement and range values for resident and translocated study leopards in Namibia.

ID	Sex	Monitoring duration (days)	No. of daily locations used	MCP-km^2 100% (50%)	Kernel 95% (50%) in km^2	Distance of home range centroid to release site in km	Mean monthly movement in km (locations used)	Weeks until home range was established	Percentage overlap with release area	Comments
Residents										
PPA51	F	586	131	66.3 (16.6)	51.4 (9.8)	---	85.4 (51)	n/a	n/a	1 successful litter during monitoring
PPA54	M	797	94	127.9 (20.9)	81.4 (17.6)	---	137.7 (28)	n/a	n/a	---
PPA56	F	140	140	36.2 (12.6)	44.6 (12.6)	---	115.8 (133)	n/a	n/a	---
Pp04	F	1,923	497	178.1 (27.1)	138.9 (29.8)	---	74.5 (468)	n/a	n/a	2 successful litters during monitoring
Pp05	M	15	Insufficient data	insufficient data	insufficient data	---	insufficient data	n/a	n/a	Death from gastro-intestinal inflammation
Pp48	M	202	117	580.8 (57.9)	306.8 (31.5)	---	93.0 (107)	n/a	n/a	Dispersed from capture area
Pp62	M	600	411	184.4 (39.5)	111.9 (29.5)	---	132.3 (393)	n/a	n/a	Increased home range when territorial neighbours were removed
Pp63	M	442	398	205.9 (71.1)	198.4 (82.7)	---	143.8 (366)	n/a	n/a	Shot after monitoring due to cattle predation
Pp71	F	134	132	170.4 (26.5)	109.6 (20.8)	---	60.5 (130)	n/a	n/a	Reproduction suspected. Maintained resident range despite short distance translocation – death in wire snare
V10938	F	346	344	120.9 (19.2)	68.7 (10.5)	---	65.7 (341)	n/a	n/a	Reproduction suspected
V11387	M	238	42	132.1 (50.9)	149.2 (25.8)	---	116.1 (36)	n/a	n/a	Mating observed
Pp77	F	152	151	53.7 (14.4)	47.2 (15.0)	---	80.0 (366)	n/a	n/a	---
Translocations										
Pp06	M	377	27	300.9 (53.3)	161.1 (21.7)	6.7	insufficient data	insufficient data	100	Courtship behaviour observed
Pp15	F	825	174	54.3 (8.8)	38.4 (3.3)	25.7	57.2 (167)	25	43.3	1 successful litter during monitoring
Pp27 (first range)	F	1,675	205	147.3 (45.8)	127.5 (11.1)	23.7	61.1 (201)	4	1.7	Range shift observed due to road construction – 2 successful litters during monitoring
Pp45	M	815	355	481.1 (15.7)	245.6 (17.3)	13.1	66.3 (351)	3	82.4	Courtship behaviour observed twice
Pp47	M	505	385	453.7 (58.3)	255.6 (29.8)	13.3	114.3 (335)	5	6.7	---
Pp57	F	110	57	75.2 (16.4)	78.0 (13.1)	29.4	97.3 (56)	7	4.1	Death in road accident

with wildlife camera data. During monitoring, two of six resident females raised litters (four known cubs) and breeding was suspected for two other females (Table 4.2). One resident male was known to mate.

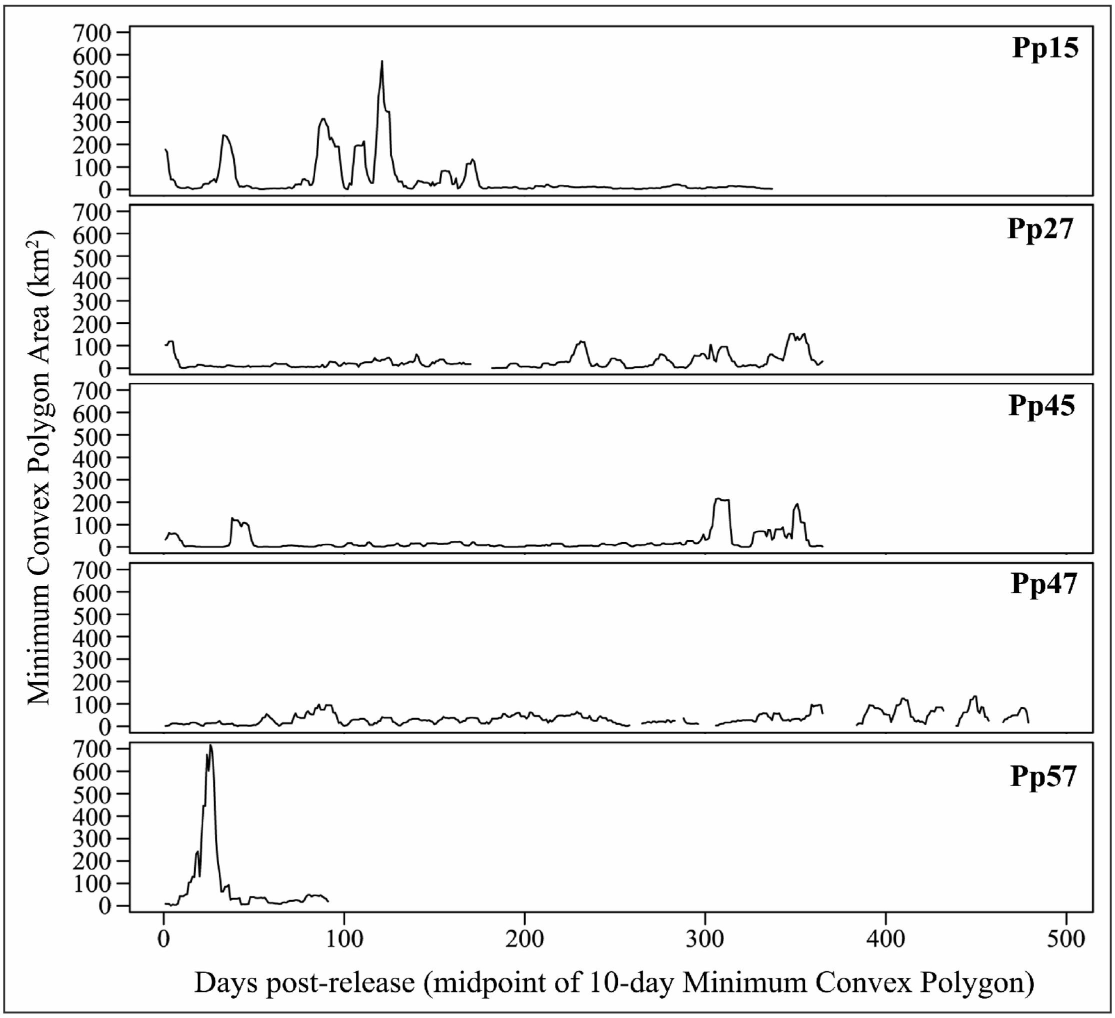


Figure 4.4 – Progressive 10-day post-release Minimum Convex Polygon area assessments for translocated leopards. *The graph illustrates settling behaviour as measured by progressive area use. Settling periods were highly variable lasting from approximately two weeks to 6.5 months post-release. MCP values were calculated with pruned daily locations. Male Pp06 is excluded for lack of sufficient data.*

4.3.4 Movements

All translocated leopards remained in the immediate vicinity of their release sites (<1 km) for between 24 - 72 h before starting exploratory movements of different scales (Fig. 4.4). During the exploration phase, all subjects at least temporarily left their recipient areas, but the three males showed some degree of release site fidelity (Fig. 4.5, Table 4.2). Female Pp57 (released into the highest density area) displayed the largest degree of exploration (Fig. 4.4), covering an area of approximately 875 km^2 during the first 35 days after release.

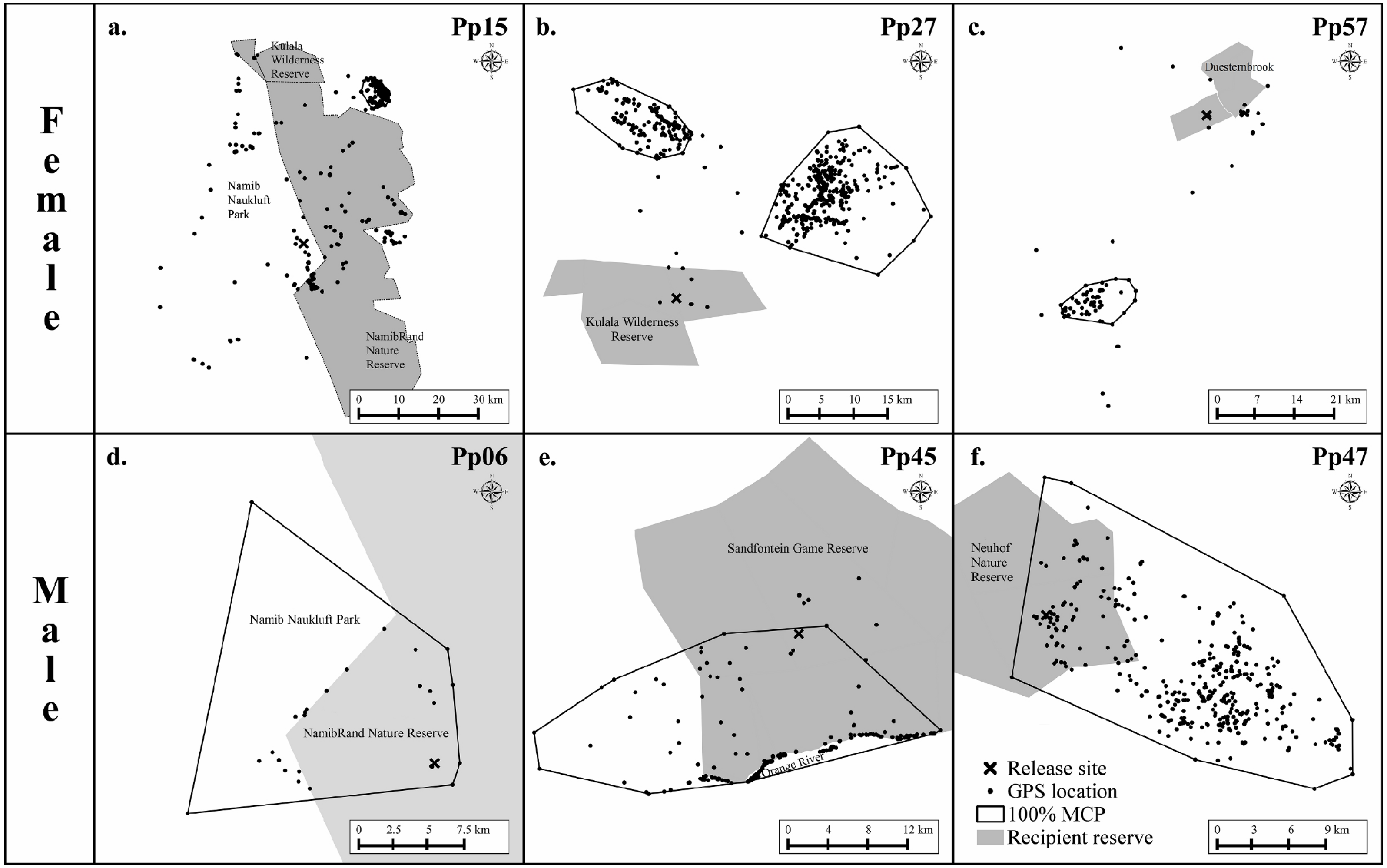


Figure 4.5 – Post-release movements of translocated leopards. *Movement plots illustrate variable site fidelity and ranging patterns of translocated subjects. Shaded areas represent recipient reserves. Range boundaries show 100% Minimum Convex Polygon calculations for post-exploration periods.*

Although Pp15 explored for much longer (Fig. 4.4), this animal largely remained on the recipient reserve and adjacent conservation areas (Fig. 4.5). Soft-released male Pp47 showed least roaming behaviour. Site fidelity results for Pp06 were constrained by limited positional data and site fidelity across all individuals varied between 1.7% - 100% in year one (median = 25.0%). The lowest values were recorded for individuals released onto reserves <400 km^2 (Table 4.2).

Progressive MCPs showed that all leopards successfully established new home ranges in areas with resident conspecifics between 0.5 - 6.5 months post-release (Fig. 4.4). Males appeared to settle quicker into their novel surroundings (<4 weeks). The ranges of Pp06 and Pp45 had the highest overlap with recipient reserves and almost entirely encompassed their movements during the exploration phase (Fig. 4.5). Although much of his initial home range was located outside of the recipient reserve (Fig. 4.5), male Pp47 shifted his core use area in year two, thus resulting in partial range overlap with the recipient reserve (23.5% of total 100% MCP) at the end of monitoring. The settled ranges of all three females were permanently located outside of the target recipient areas (Fig. 4.5). The mean distance between release site and first home range centroid (using 100% MCP range estimations) for all translocated individuals was 18.6 km ± 8.1 km SD. None of the centroids was further away from the release spot than 29.4 km (Table 4.2). Similarly, the mean distance between release and last known locations was 26.4 km ± 25.1 km SD (range = 2.4 – 60.1 km).

Habitat selection was beyond the scope of this study, but translocated leopards displayed a clear preference for mountainous terrain and all ranges incorporated escarpments with elevations rising 300 – 700 m above the surrounding landscape. It is also noteworthy that male Pp45 concentrated his activities (>92% of positional data) within a riverine forest patch along the permanent Orange River (Fig. 4.5). The range shift of Pp27 (Fig. 4.5) coincided with the beginning of road construction in the core area of her first range and thus probably was a function of anthropogenic disturbance. There were no interactions between the four leopards translocated into the same bio-geographic area.

After removal of exploratory movements, there was no significant difference between the home range estimates for residents and those for translocated leopards for any of the metrics used (Mann-Whitney U-Tests: 100% MCP (U = 91.0, p = 0.451), 50% MCP (U = 101.0, p = 0.880), 95% KDE (U = 92.0, p = 0.514) and 50% KDE (U = 108.0, p = 0.366)). Except for young adult male Pp48, who dispersed from the capture area, all resident leopards maintained stable home ranges throughout the study or until death was recorded (Fig. 4.6). In year two, male Pp62 increased his range by about 81.2% in response to removal of a neighbouring territorial male.

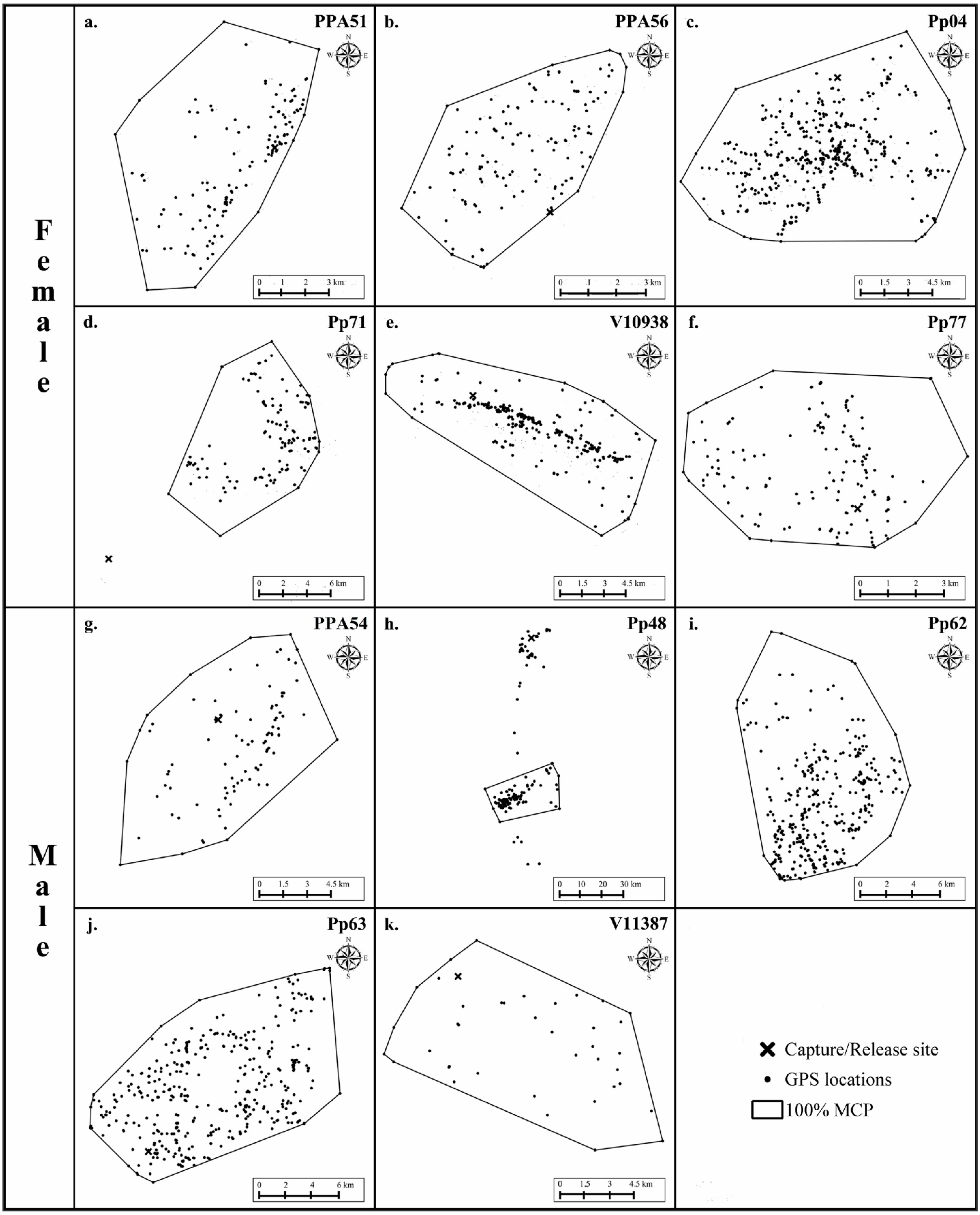


Figure 4.6 – Movements of resident study leopards in Namibia. *Movement plots demonstrate home range stability for all resident subjects except Pp48 that dispersed from the capture site and subsequently settled into a definable range pattern. GPS locations*

represent pruned daily locations. Range boundaries show 100% Minimum Convex Polygon (MCP in legend). Male Pp05 is excluded due to early mortality.

Despite explorations, none of the translocated leopards returned to their original capture sites or back to the captive facility (Fig. 4.7). Following long-distance translocations and variable periods of temporary captivity there was no evidence of oriented or directional movements towards home (Fig. 4.7). The average linear distance moved between release spots and last known locations was 21.8% ± 42.1% SD (range = 0.3 – 107.3%) of the average translocation distance. The mean direction moved by translocated leopards deviated 125.9° ± 108.8° SD from true home (range = 0.6 – 277.2°) (Fig. 4.7).

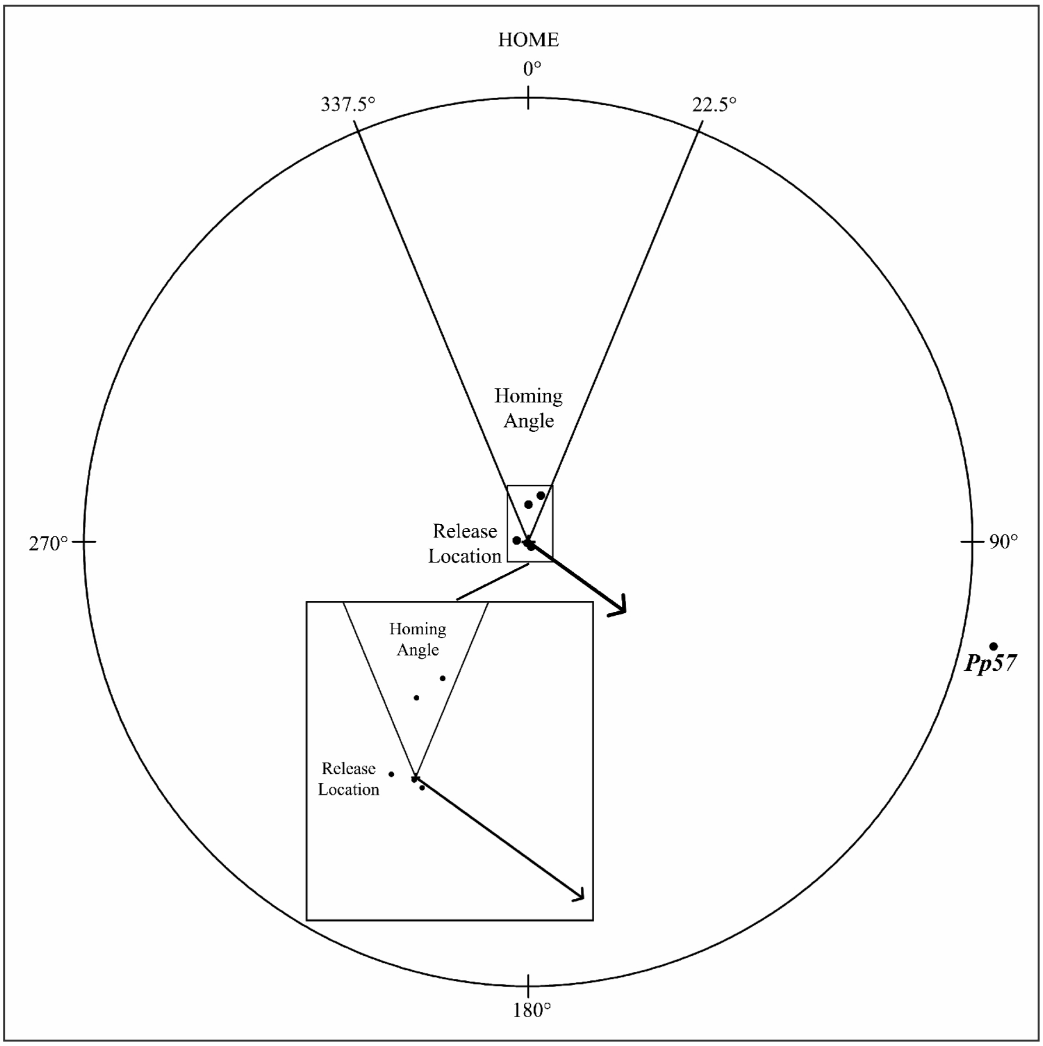


Figure 4.7 – Homing assessment for translocated leopards. *The distance from the circle's centre (release location) to outer edge represents an individual's distance travelled relative to its capture location. The arrow shows the mean orientation angle (n = 6). None of the individuals successfully homed to its respective capture site or the captive facility.*

There were no significant correlations between the direction moved and length of captivity (Spearman's $r_s = 0.08$, $p = 0.871$) or translocation distance ($r_s = 0.143$, $p = 0.787$). Although female Pp57 was moved only 47 km from her capture location, and subsequently explored further than her translocation distance, she did so at 102.7° from home (Fig. 4.7).

Conversely, the short-distance translocations of females Pp15 and Pp71 resulted in immediate returns to their established ranges. Female Pp15 was trapped accidentally 10 months post-release, after she had already settled into a stable range (Fig. 4.4). The leopard was then released a second time near her original release location and approximately 54 km from her new range, to which she returned within 48 h. Similarly, resident female Pp71 (translocated only 13 km) returned into the vicinity of her capture area, and thus into her range (Appendix 17), within 24 h. In both situations homing was an objective, and therefore the animals were released at sites within expected familiar terrain and with the purpose of facilitating unobstructed access to their previous ranges (Appendix 17).

Table 4.3 - Known prey of translocated and resident leopards.
Prey items represent kills located through GPS and VHF telemetry.

Prey	**Resident**	**Translocated**
Wildlife		
Greater kudu – *Tragelaphus strepsiceros*	17	6
Warthog – *Phacochoerus africanus*	17	3
Oryx – *Oryx gazelle*	8	9
Common duiker – *Sylvicapra grimmia*	4	0
Springbok – *Antidorcas marsupialis*	3	16
Plains and Mountain Zebra (combined) – *Equus quagga, E. zebra hartmannae*	1	2
Common eland – *Taurotragus oryx*	1	0
Steenbok – *Raphicerus campestris*	1	0
Red hartebeest – *Alcelaphus caama*	0	1
Blesbok – *Damaliscus pygargus phillipsi*	0	1
Porcupine – *Hystrix africaeaustralis*	0	1
Aardwolf – *Proteles cristata*	0	1
Livestock		
Cattle – *Bos spp.*	9	0
Sheep – *Ovis aries*	0	13 (all killed by male Pp45)
Total	61	53

4.3.5 Prey and conflict

A total of 40 and 52 wildlife kills were documented for translocated and resident leopards respectively (Table 4.3). Prey selection and preference were different between the two groups although >75% of all wildlife kills for each group were kudu, warthog, oryx and springbok (Table 4.3), reflecting the leopard's known preference range (Hayward et al., 2006b; Hayward, 2009). Soft-released male Pp47 hunted as early as 12 h after release and in the vicinity of his release pen (~711 m), suggesting that the leopard may have familiarised with local prey movements towards a near-by waterhole during the acclimatisation period. Long captive periods for females undergoing rehabilitation (Pp15, Pp27) did not compromise or deteriorate their hunting ability or lead to livestock predation.

Of the six translocated leopards, only male Pp45 was involved in post-release conflict by killing 13 smallstock that were illegally herded into his new home range on the recipient reserve. It remains speculative whether he (an opportunistic cattle raider) would have resumed conflict behaviour if stock had not been moved into the PA. The damage was fully compensated and Pp45 did not continue raiding. The author is confident that no other translocated leopard killed livestock because positional data were shared with landowners on a regular basis and no conflict reports were made. In comparison, three resident leopards killed livestock opportunistically (i.e. between one and three livestock per year).

To assess the effectiveness of moving conflict leopards, the four farmers who had experienced livestock depredation prior to translocations were interviewed. Pre-translocation livestock losses (an average of five cattle calves per property in the last 12 months) stopped for at least 16 months on all farms, suggesting that translocation involved the raiders responsible for damage. According to farmers, translocated individuals were replaced by "*new leopards*" within approximately six weeks after removal. On three properties repeat conflict with leopards occurred 16, 25 and 29 months after translocations, but it was tolerated because only 1 – 3 calves were killed per year. The fourth farmer did not experience repeat conflict although leopards still occurred on the property. Therefore, the non-lethal management of a few opportunistic livestock raiders also increased local tolerance of conspecifics that caused no conflict or acceptable levels of damage. However, three landowners considered repeat translocation of leopards from their properties and one livestock farmer continued to trap leopards indiscriminately. Following first translocations, two land managers improved livestock protection by night-kraaling of juvenile calves (see Appendix 18) while three did not adjust livestock husbandry practises.

4.3.6 Potential recipient areas in Namibia

Area suitability modelling with *CaTSuiT* initially eliminated 506,168 km^2 of Namibia (61.3% land cover) due to conflicting land uses. Of the remaining Namibian landscape, another 176,025 km^2 (21.2% land cover) were removed due to high or zero leopard density, 17,744 km^2 (2.1% land cover) for the safety buffer around urban sites, and 7,868 km^2 (1.0% land cover) that represented suitable recipient habitat but were composed of isolated patches smaller than the squared distance moved by Pp57 during explorations.

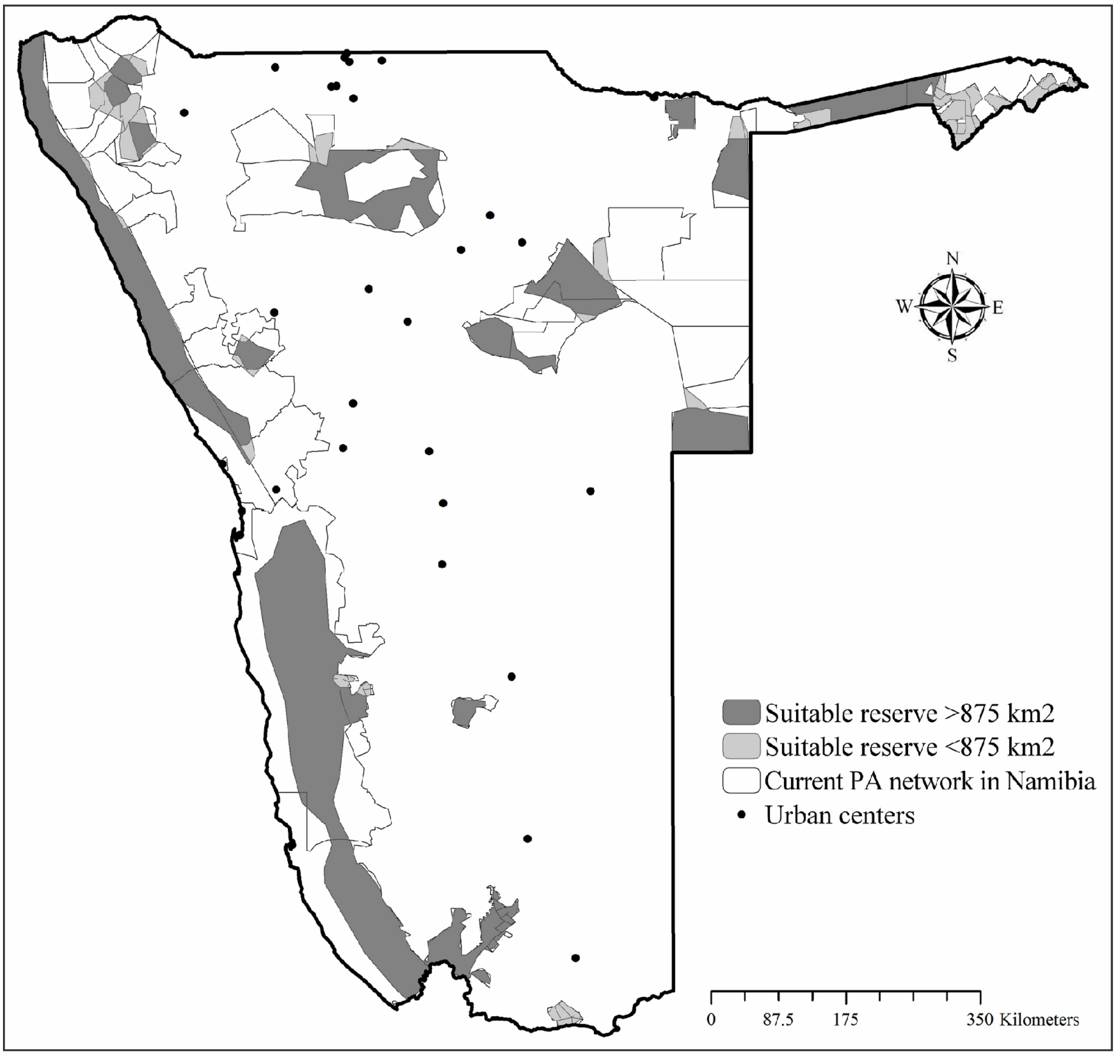


Figure 4.8 – Potential leopard recipient areas in Namibia. *The map shows suitable protected areas with low-medium density of conspecifics, >50 km away from urban centers, and with a high probability of being able to accommodate the largest post-release movements observed in this study (scaled to Pp57's explorations) either as a 'stand-alone' area (>875 km^2) or as a 'connected' area adjoining other suitable patches (<875 km^2). Output calculated with CaTSuiT (Lemeris Jr., 2015). Area-specific details are provided in Appendix 19.*

According to this selection process, *CaTSuiT* identified approximately 117,613 km^2 of available leopard translocation habitat (or 14.2% of Namibia's landscape) within 10 National Parks (81,016 km^2), 54 communal conservancies (33,147 km^2) and five private reserves (3,450 km^2) (Fig. 4.8). This estimate includes 23 stand-alone reserves as well as 49 connected patches (Appendix 19). Assuming that leopards can be released into a suitable patch and not come under threat if they move into adjacent suitable habitat, the model predicted that up to 87 translocations (approximately 0.004% - 0.007% of Namibia's estimated free-ranging population, Stein et al., 2012) can be supported initially, followed by an 18 months rest period at each recipient site. Total recipient habitat decreased to approximately 103,005 km^2 if smaller 'connected' units (<875 km^2; scaled to the movements of female Pp57) were removed from results, reducing the initial potential for translocations to ~71 individuals.

Across the total recipient area identified, there was a significant disparity in terms of broad habitat type (comparing open grass- and low shrub-lands against woodland vegetation types: χ^2 Goodness of Fit = 6.76; d.f. = 1; $p = 0.009$). Nearly two-thirds (63.3%) of the entire suitable area represent marginal desert and semi-desert habitats of low productivity whereas the remainder is composed of dry *Acacia* savannah and dense miombo woodlands.

4.3.7 Success

Using survival, post-release conflict and homing as key criteria, four of six translocations were successful, giving an overall success rate of 66.7%. These leopards established themselves in novel environments with resident conspecifics, refrained from livestock depredation and reproduced successfully. Success increased to 83.3% if compensated livestock damage (Pp45) was not considered in this assessment. Translocated individuals showed no significant difference in range behaviour, survivorship, reproduction or likelihood of conflict when compared with residents. This study also provides evidence that long-term rehabilitation of female leopards is feasible.

4.4 Discussion

Translocation of leopards has generally been criticised as an ineffective tool to achieve desirable levels of conflict reduction (Hamilton, 1981; Stander et al., 1997a; Weilenmann et al., 2010). Here, the selective live-removal of opportunistic livestock raiders stopped conflict for at least 16 months; only one translocated individual continued raiding and probably because livestock were illegally herded into the release area. However, new intruders (reportedly) soon filled the vacuum created by translocating offenders, an observation frequently made after removal of territorial felids (Rabinowitz, 1986; Bailey, 1993; Linnell et al., 1996; Athreya, 2006). In South Africa, vacant leopard territories were re-colonised by male dispersers within three months of the predecessor's death (Balme et al., 2009). Therefore, if the issue of livestock depredation is not addressed effectively at source sites, continued translocation of conflict leopards may potentially induce local source-sink dynamics (Balme et al., 2009). Where sustained removal (dead or alive) includes significant proportions of reproductive females, the viability of leopard populations may be compromised (Caro et al., 2009; Packer et al., 2009). Kerth et al. (2013) demonstrate the reality of this risk because stock-raiding leopards are often misidentified and both sexes appear to be equally involved in livestock killing, thus resulting in high levels of female persecution.

In addition, the social effects of translocation on source and recipient populations are poorly understood and require further attention through simultaneous monitoring of translocated and resident leopards. Particularly translocated males may cause serious social disruptions at recipient sites when challenging established territory holders (Corbett, 1947; Hamilton, 1976; Bailey, 1993; Balme et al., 2009) or through infanticide (Karanth and Madhusudan, 2002; Balme et al., 2009; Balme and Hunter, 2013).

From an ecological viewpoint, pre-prime young adults may be the best suited candidates for translocation. Results of mountain lion (*Puma concolor*) translocations in the US (Logan et al., 1996) showed that sub-adults settled into new environments faster than adults and were more successful, leading to the hypothesis that dispersal age subjects are naturally predisposed to encounter and deal with novel environmental conditions. Similarly, young leopard males typically disperse from maternal home ranges (Bailey, 1993) and sometimes do so over considerable distances (Fattebert et al., 2013), resulting in natural exposure to a broad range of environmental stimuli and stresses which resemble effects artificially created through translocation. Conversely, sub-adult females usually take over part of the mother's range (Bailey, 1993; Steyn and Funston, 2009) and, a priori, may not be suited for relocation. However, several published case studies (here Pp27; Hamilton, 1981; Houser et al., 2011) demonstrate that young adult females can successfully cope with translocation, become self-sustaining without causing conflict and contribute to free-range breeding. A leopard's age, social status, and physical condition should always be considered as important indicators of its suitability for translocation. Where there is strong evidence of serious ailments, translocation becomes unfeasible, even for individuals from endangered populations (Farhadinia et al., 2015).

Under any circumstances, rigorous candidate selection must be a priority element of translocation planning as there appears to be little space for potentially large numbers of translocations. In Namibia, and many other areas of its remaining distribution, the leopard is considered the primary 'problem' carnivore (chapter 3; Wang and Macdonald, 2006; Sangay and Vernes, 2008; Odden et al., 2014). Conflict animal records demonstrate the magnitude of the problem in different countries with hundreds of leopards reported as conflict individuals (Hamilton, 1981; Esterhuizen and Norton, 1985; Athreya, 2006; Schiess-Meier et al., 2007; Swanepoel et al., 2014) and often with an increasing trend (see Appendix 16 for Namibia). Considering that livestock depredation did recur on three of four source properties in this study, the demand for conflict-motivated translocations may soon outweigh the available capacity. Although area suitability modelling suggests a potential for approximately 87 initial leopard translocations, this estimate represents only a small fraction (0.004% – 0.007%) of Namibia's estimated free-ranging population (Stein et al., 2012), thus implying little biological relevance at the population level. It is also substantially lower than the annual number of reported 'problem' leopards in the country (median of 122 for 1997 - 2003, Appendix 16) and may therefore be exhausted in a single year. It should therefore not be utilised in a haphazard or reactive fashion, but when conflict prevention measures have failed and translocation of a few individuals significantly enhances tolerance of leopards amongst livestock producers (Treves and Karanth, 2003).

The identification of appropriate release areas presents one of the most crucial challenges of responsible leopard translocations. Strategic approaches to determine recipient habitat suitability, as demonstrated here, should be a priority in countries where translocation remains a legal management option. Such assessments should be repeated at least every two years to incorporate new information from translocation events as well as updated data of model input parameters.

In the present suitability assessment, communal conservancies need to be considered the least desirable recipient area category. Despite their focus on wildlife utilisation as a local driver for revenue generation (Paterson et al., 2008), communal conservancies also permit livestock grazing in restricted areas (Barnes et al., 2002) creating a potential risk for post-release conflict. If communal conservancies were removed from the suitability estimate altogether, and only areas with exclusive wildlife use were included, total recipient habitat further decreased to 84,466 km^2, equivalent of a potential for approximately 59 leopard translocations across 18 months.

The conditions affecting recipient area suitability (land use, prey, conspecifics, habitat etc.) significantly differ across Africa and therefore require case-specific modelling and assessment. Due to the country's low human density (Mendelsohn, 2006), its unusually large private and public PA network, and widespread distribution of wild prey (e.g. East and the IUCN/SSC Antelope Specialist Group, 1999; Lindsey et al., 2013b), Namibia provides a unique scenario and selective translocation into remote, marginal habitats may still be possible for some individuals.

Moreover, Namibia's leopards belong to the genetically diverse sub-species *P. p. pardus* (Uphyrkina et al., 2001) that form part of the near panmictic sub-Saharan population (Henschel et al., 2008; Stein and Hayssen, 2013). Southern African leopards may also naturally move across several countries (Fattebert et al., 2013). A priori, this implies that translocations might not result in deleterious effects from genetic drift. However, distinct genetic groups occur across the leopard's geographic range (Uphyrkina et al., 2001), including sub-populations within southern Africa (e.g. Tensen et al., 2013; McManus et al., 2015; Pitman et al., 2015). Here, variations in the species' natural genetic structure appear mainly driven by spatial isolation, with effective genetic dispersal being as low as approximately 82 km in some areas, suggesting that long-distance translocations (to prevent homing) could severely compromise local and regional population structures (Ropiquet et al., 2015).

This study provides preliminary evidence that opportunistic livestock raiders can be managed successfully through translocation, but the common dilemma of large numbers of conflict leopards needs to be addressed differently (Linnell et al., 1997; Weilenmann et al., 2010). Amongst many other avenues, non-lethal alternatives include exclusion tactics (Schumann et al., 2006), improved livestock protection and husbandry (Ogada et al., 2003; Woodroffe et al., 2007) and livestock insurance schemes (Kumar and Chauhan, 2011). Preventative measures that encourage coexistence with leopards should always precede symptomatic management (Weilenmann et al., 2010; Odden et al., 2014) to reduce the number of indiscriminately trapped leopards (Athreya, 2006). To these, translocation can contribute as a last resort tool to prevent lethal removal of individual leopards where necessary. Furthermore, the high economic and recreational potential of leopards (Stein et al., 2010; Maciejewski and Kerley, 2014; Braczkowski et al., 2015) can be promoted to increase their value beyond that of livestock losses. For example, Stander et al. (1997a) showed that monies earned from tracking offending leopards can exceed local livestock losses 12-fold and thus contribute to rural income generation. Selective trophy hunting may be an appealing option that is frequently recommended by conflict-affected land managers (e.g. Romañach et al., 2007; Stein et al., 2010). However, reliable field-identification of leopards is indeed very difficult (Balme et al., 2012; Kerth et al., 2013), thus potentially accelerating indiscriminate off-take and social disturbance of harvested populations (Caro et al., 2009; Packer et al., 2009, 2011; Braczkowski et al., 2015). Due to high logistic demands, the costs involved, and existence of controlled captive breeding programmes, a continued influx of conflict leopards into permanent captivity is also not a desirable option (Cobb, 1981).

There is sufficient evidence that mass relocation campaigns culminate in conservation failures (Hamilton, 1981) or, in fact, accelerate conflict significantly by provoking attacks on people (Athreya et al., 2007, 2011). Haphazard translocations should, therefore, be discontinued (Hamilton, 1981; Athreya, 2006). Instead, rather than focusing on conflict mitigation alone (Linnell et al., 1997; Weilenmann et al., 2010), leopard translocations should be designed to facilitate range expansion through reintroduction into suitable habitat, genetic augmentation of dwindling populations and preservation of valuable individuals in critically endangered populations (Hayward et al., 2006a, 2007b; Uphyrkina and O'Brien, 2003; Hebblewhite et al., 2011). This highlights the need to develop country-specific translocation criteria that reflect:

1) National leopard management objectives;
2) Population status and characteristics; and
3) Improved candidate selection.

Where translocation is necessary, conservative protocols need to be adopted. Based on the experiences from this study, and the published literature, protocol recommendations for leopard translocations are provided in Appendix 20. In agreement with other studies, all subjects displayed exploratory movements, but at smaller scales than previously reported (Hamilton, 1981; Weilenmann et al., 2010; Mondal et al., 2013). The quick settling of male Pp47 after soft release supports Hayward et al.'s (2006a) hypothesis that acclimatisation to the recipient area may reduce explorations. Males displayed partial site fidelity and females settled in the vicinity of recipient areas, suggesting that limited releases into habitats with low density of conspecifics may not compromise conservation objectives. Conversely, releases into areas with high leopard density frequently result in displacement (here Pp57; Athreya et al., 2007; Weilenmann et al., 2010). Intermediate captivity can provide sufficient time for managers to identify suitable recipient areas, as long as captive leopards retain wild characteristics (Houser et al., 2011) and are not unduly stressed or injured (see Hamilton, 1981; Athreya, 2006). Long captive periods may not necessarily compromise survival, hunting abilities, or reproduction (Pp15, Pp27; Houser et al., 2011, Mondal et al., 2013).

In stark contrast with previous accounts (Hamilton, 1981; Stander et al., 1997a; Weilenmann et al., 2010; Odden et al., 2014;), none of the subjects in this study homed or showed oriented movements towards home. It therefore appears that a combination of temporary captivity and removal of familiar environmental cues (here through long-distance translocation) notably reduces homing ability. Leopards translocated into familiar areas (here Pp15 and Pp71) returned to their established ranges and homing time may only be influenced by translocation distance (Stander et al., 1997a). Successful homing could be expected for female Pp57, but this leopard was moved into a valley where recognition of visual homing cues was no longer possible (similar to a male in Hayward et al., 2006a) (Appendix 17).

Despite its small sample size, the present study demonstrates that translocations can successfully conserve individual leopards and possibly other asocial carnivores with similar ecological characteristics under specific circumstances (Hedrick, 1995; Logan et al., 1996; Maehr et al., 2002; Kelly and Silver, 2009). The strategy should therefore be available as a management option, albeit with much curtailed use and in accordance with country-specific leopard management objectives that reflect available recipient habitat and the value of individuals to the free-ranging gene pool. The latter demands a better understanding of local and regional leopard populations and the reasons of their current declines (Henschel et al., 2008).

4.5 Key points

1. Under specific circumstances, the translocation of rehabilitated and livestock-raiding leopards can be successful – overall success was 66.7% and conflict was alleviated for a minimum of 16 months, although not permanently.
2. The survival, reproduction, movements (after explorations), and conflict involvement of translocated leopards did not significantly differ from that of resident conspecifics.
3. Long-distance translocations prevented homing and leopards settled into novel environments after explorations of 0.5 – 6.5 months, but site fidelity was highly variable.
4. Namibia provides suitable recipient habitat for approximately 87 initial leopard translocations. Despite favourable conditions in the country, the potential demand for translocations outweighs the available capacity.
5. Translocation of leopards can be employed as a selective, supplementary management tool, but not as a standard conflict-mitigation approach. It can be considered for endangered populations and after rigorous identification of suitable recipient areas and best-suited candidates.

Chapter 5 Cheetah (*Acinonyx jubatus* – Schreber, 1775)

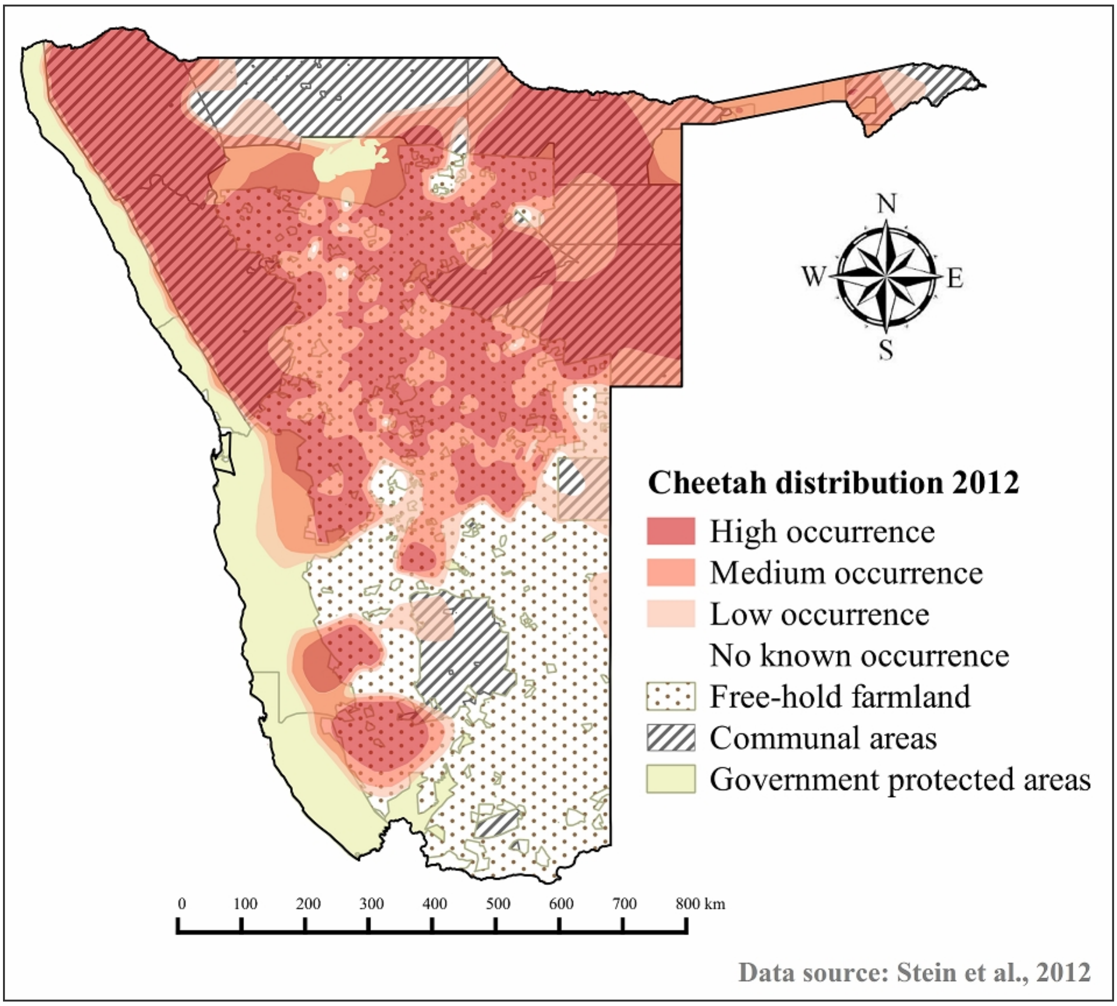
5.1 Introduction

Most large carnivore distributions continue to decrease at alarming rates (Ripple et al., 2014). The cheetah is no exception and its occurrence has decreased significantly (Marker, 2005; Ray et al., 2005). Primarily due to habitat loss, human persecution, and reduction of its natural prey base, the cheetah may have undergone a range-wide population decline of ~90% during the 20th century (Marker, 1998). The Asiatic sub-species (*A. j. venaticus*) has already disappeared from the Indian subcontinent and only a small remnant population survives in Iran (Farhadinia, 2004).

African populations are not far behind. Estimates suggest that wild cheetah numbers across the continent dropped from ~15,000 individuals in the 1970s (Myers, 1975) to possibly <10,000 adults in only three decades (Durant et al., 2008). This general trend corresponds with extinctions and near-extinctions in more than 10 African countries and population data are uncertain for most historic range states (Kraus and Marker-Kraus, 1991). At present, the cheetah is Africa's most endangered large felid and classed as "Vulnerable" by the IUCN (Durant et al., 2008), indicating a high risk of extinction in the wild.

Southern Africa currently supports the largest contiguous free-ranging population with ~6,500 animals, of which >75% occur outside PAs (IUCN/SCC, 2007). In Namibia – home to the single largest population in the world (Marker, 2005; Purchase et al., 2007) – the proportion of cheetahs living on unprotected commercial and communal farmlands is estimated to be as high as 90-95% (Morsbach 1986b, 1987). This peculiar distribution pattern (Fig. 5.1) has been attributed to highly favourable habitat conditions on managed farmlands, including accelerating bush encroachment, permanent access to water, abundant wildlife prey, and heavily reduced competition with larger-bodied competitors such as lion and spotted hyaena (McVittie, 1979; Lindsey et al., 2013a). The latter, in particular, may be a key advantage for cheetah survival. Where cheetahs occur sympatrically with high lion and spotted hyaena densities, the species is susceptible to high juvenile mortality (50% in Eaton, 1970a; 73% of all cub mortality in the Serengeti - Laurenson, 1994; Laurenson et al., 1995). Conversely, cub survival is approximately 3.5 times higher on Namibian farmlands (Wachter et al., 2011) where lions and spotted hyaenas have largely been extirpated (Lindsey et al., 2013a). Other competitive disadvantages of cheetahs in areas with intra-guild competition include considerable adult mortality (e.g. 37.5% of all adult deaths in Marnewick et al., 2009) and frequent kleptoparasitism of kills (rates of 9% - 14%

in Caro, 1994; Mills et al., 2004; Radloff and du Toit, 2004; Scantlebury et al., 2014), sometimes resulting in the loss of all kills across several days (Stander, 1990b).



Data source: Stein et al., 2012

Figure 5.1 – Cheetah distribution across Namibia's main land tenures.

Despite reduced competition in unprotected anthropogenic landscapes, the cheetah's occupancy of agro-pastoralist land-use systems results in conflict with livestock managers in many countries (e.g. chapter 3; Marker et al., 2003a; Purchase et al., 2006; Gusset et al., 2009b; Thorn et al., 2015). The increasing emergence of commercial wildlife ranching across southern Africa (e.g. Cousins et al., 2008; Lindsey et al. 2013b, 2013c) further complicates prospects for the species' conservation because game ranchers appear particularly intolerant of cheetahs and persecute disproportionately more individuals than do livestock farmers (chapter 3; Marker et al., 2003a).

Cheetahs rarely scavenge (Caro, 1982; Stander, 1990b) and the species' diet predominantly consists of wild ungulates with an average body mass of 23-56 kg (mode of 36 kg, Hayward et al., 2006c). Cheetahs show a preference for blesbok (*Damaliscus pygargus phillipsi*), impala, Thomson's (*Eudorcas thomsonii*) and Grant's (*Nanger granti*) gazelles as well as springbok (Hayward, 2009). Due to their slender build, cheetahs hunt prey that can easily be tackled and killed with least risk of injury and which can be consumed quick enough to reduce the probability of kleptoparasitism (Hayward et al., 2006c). The cheetah's prey spectrum extends to over 80 wildlife species, ranging from the size of mole rats (*Cryptomys* spp.) to giraffe (*Giraffa camelopardalis*) (pers. obs. and pooled from Mills, 1984a; FitzGibbon, 1990; Skinner and Smithers, 1990; Caro, 1994; Sunquist and Sunquist, 2002; Marker et al., 2003b; Hayward et al., 2006c). Prey selection is mainly influenced by local abundance of suitable prey (Hayward, 2009) whilst prey size is predicted by a cheetah's sex and sociality in that single females and lone males hunt smaller prey than male coalitions (e.g. Mills et al., 2004; Bissett and Bernard, 2007; Voigt et al., 2014; Rostro-García et al., 2015).

In Namibia, cheetahs exhibit no preference for killing livestock (Morsbach, 1986a; Marker et al., 2003b), but they do occasionally predate on domestic animals and valuable game on farmlands (chapter 3; Morsbach 1986b; Wachter et al., 2006b; Rust and Marker, 2013a; Voigt et al., 2014). Farmers regard the cheetah as a wasteful feeder (chapter 3; Lindsey et al., 2013a), further fuelling a motivation for persecution. Even though livestock depredation rates may be low in relation to stocking rates and other causes of mortality (e.g. 1.8% - 3.0% of all calf losses, Morsbach 1986a; ~0.76 calves and ~0.32 sheep per year per farm, Marker et al., 2003b), cheetahs are frequently, and erroneously, blamed for the majority of livestock losses (Morsbach, 1986a). Due to the large home range sizes of Namibian cheetahs that cover dozens of private properties, farmers also routinely overestimate the true number of cheetahs on their lands (Morsbach, 1986a; Wachter et al., 2006a; Marker et al., 2008a). This perceived local over-abundance, coupled with anecdotal and actual conflict, results in widespread persecution of the species (chapter 3; Nowell, 1996; Marker et al., 2007).

Annual cheetah removals from private farms ranged between 650 – 890 individuals in the 1980s (Morsbach, 1987) resulting in a total reported loss of at least 9,588 cheetahs between 1978 and 1994 (Nowell, 1996). More recently, continued indiscriminate persecution was reported to affect at least 100 – 340 cheetahs annually (Marker et al., 2007). These estimates are likely much lower than the true removal executed by farmers because official reports may only account for ~50% of real persecution (Morsbach, 1986a). Indiscriminate lethal removal can be as high as 82 cheetahs in one year on a single property (chapter 3), or >200 individuals across a farmer's lifetime (Lindsey et al., 2013a), resulting in local cheetah sinks that may suffice to compromise long-term population viability (Woodroffe and Ginsberg, 1998). However, due to a high degree of uncertainty as regards the number of cheetahs in Namibia (with median estimates ranging from 2,500 - 10,584 individuals (*cf.* Nowell, 1996; Hanssen and Stander, 2004; Stein et al., 2012)), it remains speculative how this off-take affects the free-ranging population. Nevertheless, it is agreed that direct persecution on free-hold lands constitutes a major threat to the species' long-term persistence in Namibia (Johnson et al., 2013), locally accounting for as much as ~80% of all wild cheetah mortality (Marker et al., 2003a).

The Namibian situation is representative of conflicts in other southern African cheetah range countries (e.g. Purchase et al., 2007; Gusset et al., 2009b, Thorn et al., 2013, 2015). Therefore, it is not surprising that translocation of perceived conflict cheetahs has been a common strategy in Namibia (du Preez, 1970; Marker et al., 2003a; T. Hoth, pers. comm. 2014) as well as other areas across southern Africa (Phiri, 1996; Purchase et al., 2006; Marnewick et al., 2009; Boast et al., 2015). In Botswana, translocation continues to be employed as a standard response to livestock predation by cheetahs (M. Flyman, pers. comm. 2014) and there appears to be a steady demand for live removals from private farmlands in several countries (chapter 3; South Africa, Marnewick et al., 2009; Botswana, Boast et al., 2015).

Long believed to be an open savannah specialist - a misperception mainly attributable to the strong focus of early research on the Serengeti ecosystem (e.g. Eaton, 1974; Durant et al., 1988; Caro, 1994) - studies from across the continent showed that cheetahs inhabit, and readily adapt to, a wide variety of different habitats including true deserts, open and mixed woodlands, as well as densely vegetated environments (e.g. Eaton, 1974; Skinner and Smithers, 1990; Caro, 1994; Gros and Rejmánek, 1999; Sunquist and Sunquist, 2002; Broomhall et al., 2003; Mills et al., 2004; Muntifering et al., 2006; Purchase et al., 2006; Bissett and Bernard, 2007; Marnewick et al., 2009; Rostro-García et al., 2015; Welch et al., 2015). In conjunction with this habitat flexibility, the dietary plasticity of cheetahs (see above) renders them potentially suitable for translocation in order to alleviate conflicts (Marker et al., 2003a; Purchase et al., 2006; Boast et al., 2015), to facilitate reintroductions of the species (Hayward et al., 2007a, 2007b; Marnewick et al., 2009) or to rehabilitate orphaned individuals (Pettifer, 1981; Houser et al., 2011). Similar to leopards (chapter 4), however, a comprehensive understanding of cheetah translocation efficacy (at least in Namibia) has been clouded by a lack of result reporting and partial assessments (du Preez, 1970; Marker et al., 2003a) that prevent conclusions.

This chapter aims to provide a detailed evaluation of cheetah translocations carried out into private free-range conservation areas in Namibia. Translocations are assessed in light of their biological results, including survivorship, movements, reproduction, and prey.

Furthermore, this evaluation considers pre- and post-release conflict, translocation distance, release mode, and the influence of temporary captivity as important variables. Finally, the results from this study, and those from other cheetah translocations in southern Africa, are synthesised to determine suitable cheetah recipient areas across Namibia's PA network.

5.2 Methods (supplemental to chapter 2)

5.2.1 Subjects

Translocated cheetahs (Table 5.1) were trapped on private free-hold properties on Namibia's commercial farmlands between May 2008 and December 2012 (Fig. 5.2). Cheetahs were housed in their original group compositions, e.g. mothers with dependent offspring, and sibling groups trapped together. Two artificial groups were established from long-term captive orphans trapped on different occasions (groups Aju29/30, and Aju42/43/44 in Table 5.1 respectively). The animals showed no signs of aggression and readily bonded when introduced to each other at approximately 12 months of age. Artificial group formation was undertaken as this was believed to benefit the animals' post-release chances of success (Hunter, 1998b) whilst preventing the risk of breeding among related males and females. To facilitate familiarity, male Aju19 and female Aju20 were introduced to each other during the last week prior to a joint release on the same reserve, but these individuals showed no tendency to bond. None of the attempts to introduce previously unfamiliar cheetahs resulted in hostile interactions or injuries.

Although provision of live prey to maintain or train hunting skills (*cf.* Houser et al., 2011) was not permitted (Ministry of Environment and Tourism in Namibia, 2012), all cheetahs were known to hunt successfully prior to release. Long-term captives, including orphans (Table 5.1), made kills when hares (*Lepus spp.*), guineafowl (*Numida meleagris*) or yellow mongoose (*Cynictis penicillata*) naturally entered enclosures.

5.2.2 Potential cheetah recipient areas

Considering cheetah ecology, the outcomes of this study as well as results from other published cheetah translocations, recipient area suitability was defined to include five key conditions (Table 5.2) at equal parameter weighting. Hence, if an area did not meet all of these conditions, it was considered unsuitable to receive translocated cheetahs. The rationale for this conservative approach was to maximise chances of post-release survival and reproduction whilst minimising the risk of conflict and persecution (Table 5.2).

Table 5.1 - Details of translocated cheetahs in Namibia, 2008 – 2012.

ID	Sex	Estimated age (years)	Weight (kg)	Capture region – release area	Year[a]	Background	Captivity (days)	Translocation distance (km)	Release mode (acclimatisation) – social composition	Degree of habituation	Transmitter type
Aju01	F	2-3	37	Otjozondjupa – NRNR	2008	Indiscriminate	10	526	Hard – sibling group	Wild	VHF[b]
Aju02	F	2-3	33	Otjozondjupa – NRNR	2008	Indiscriminate	10	526	Hard – sibling group	Wild	ID
Aju03	M	2-3	38	Otjozondjupa – NRNR	2008	Indiscriminate	10	526	Hard – sibling group	Wild	VHF[b]
Aju07	F	7-9	33	Khomas – NRNR	2008	Livestock raider	61	331	Hard – mother with 2 offspring	Wild	VHF[b]
Aju17	M	5-7	44	Otjozondjupa – NRNR	2009	Livestock raider	175	490	Hard – individual	Wild	GPS[c] ARGOS
Aju18	F	5-7	43	Otjozondjupa – NRNR	2009	Livestock raider	157	470	Hard – individual	Wild	GPS[c] ARGOS
Aju19	M	3-5	49	Khomas – KWR	2009	Indiscriminate	62	312	Hard – artificial group	Wild	VHF[d]
Aju20	F	6-8	37	Omaheke – KWR	2009	Indiscriminate	37	396	Hard – artificial group	Wild	GPS[c] ARGOS
Aju26	M	2-3	44	Otjozondjupa – FOC	2009	Indiscriminate	12	182	Hard – individual	Wild	GPS[d] cell
Aju29	F	3-4	38	Otjozondjupa – NRNR	2010	Orphan – rehab	596	482	Soft (10 weeks) – artificial group	Semi-habituated	VHF[d]
Aju30	M	3-4	41	Otjozondjupa – NRNR	2010	Orphan – rehab	446	431	Soft (10 weeks) – artificial group	Semi-habituated	GPS[c] ARGOS
Aju34	M	3-4	43	Omaheke – FOC	2010	Indiscriminate	47	137	Hard – individual	Wild	VHF[d]
Aju38	M	3-4	37	Khomas – SGF	2011	Indiscriminate	153	259	Soft (18 weeks) – individual	Wild	GPS[c] ARGOS
Aju40	F	3-4	37	Otjozondjupa – NDL	2012	Indiscriminate	1,184	289	Soft (38 weeks) – sibling group	Habituated	GPS[c] ARGOS
Aju41	F	3-4	38	Otjozondjupa – NDL	2012	Indiscriminate	1,184	289	Soft (38 weeks) – sibling group	Habituated	VHF[d]; GPS[c] ARGOS
Aju42	M	3-4	29	Otjozondjupa – SNGR	2011	Orphan – rehab	1,055	816	Soft (18.5 weeks) – artificial group	Habituated	VHF[b]
Aju43	M	3-4	32	Otjozondjupa – SNGR	2011	Orphan – rehab	1,106	806	Soft (18.5 weeks) – artificial group	Habituated	GPS[c] ARGOS
Aju44	M	3-4	32	Otjozondjupa – SNGR	2011	Orphan – rehab	1,055	816	Soft (18.5 weeks) – artificial group	Habituated	VHF[d]
Aju56	F	5-7	45	Otjozondjupa – FOC	2012	Indiscriminate	169	402	Hard – mother with 3 offspring	Wild	GPS[c] ARGOS
Aju58	F	5-7	40	Otjozondjupa – NRNR	2012	Indiscriminate	260	372	Soft (4 weeks) – mother with 2 offspring	Semi-habituated	GPS[d] satellite
Aju59	F	4-6	38	Khomas – SNGR	2012	Indiscriminate	272	659	Soft (1 week) – mother with 3 offspring	Semi-habituated	GPS[d] satellite
Aju65	M	6-7	51	Khomas – FOC	2012	Indiscriminate	1	71	Hard – male coalition	Wild	VHF[d]
Aju66	M	6-7	53	Khomas – FOC	2012	Indiscriminate	2	71	Hard – male coalition	Wild	GPS[d] satellite

[a] Year of release; [b] Advanced Telemetry Systems, Insanti, USA; [c] Sirtrack, Hawkes Bay, NZ ; [d] Africa Wildlife Tracking, Pretoria, RSA; NRNR = NamibRand Nature Reserve; KWR = Kulala Wilderness Reserve; FOC = Frauenstein – Ondekaremba Complex; NDL = Namib Desert Lodge; SGF = Solitaire Guest Farm; SNGR = Sandfontein Private Nature and Game Reserve (see Fig. 5.2)

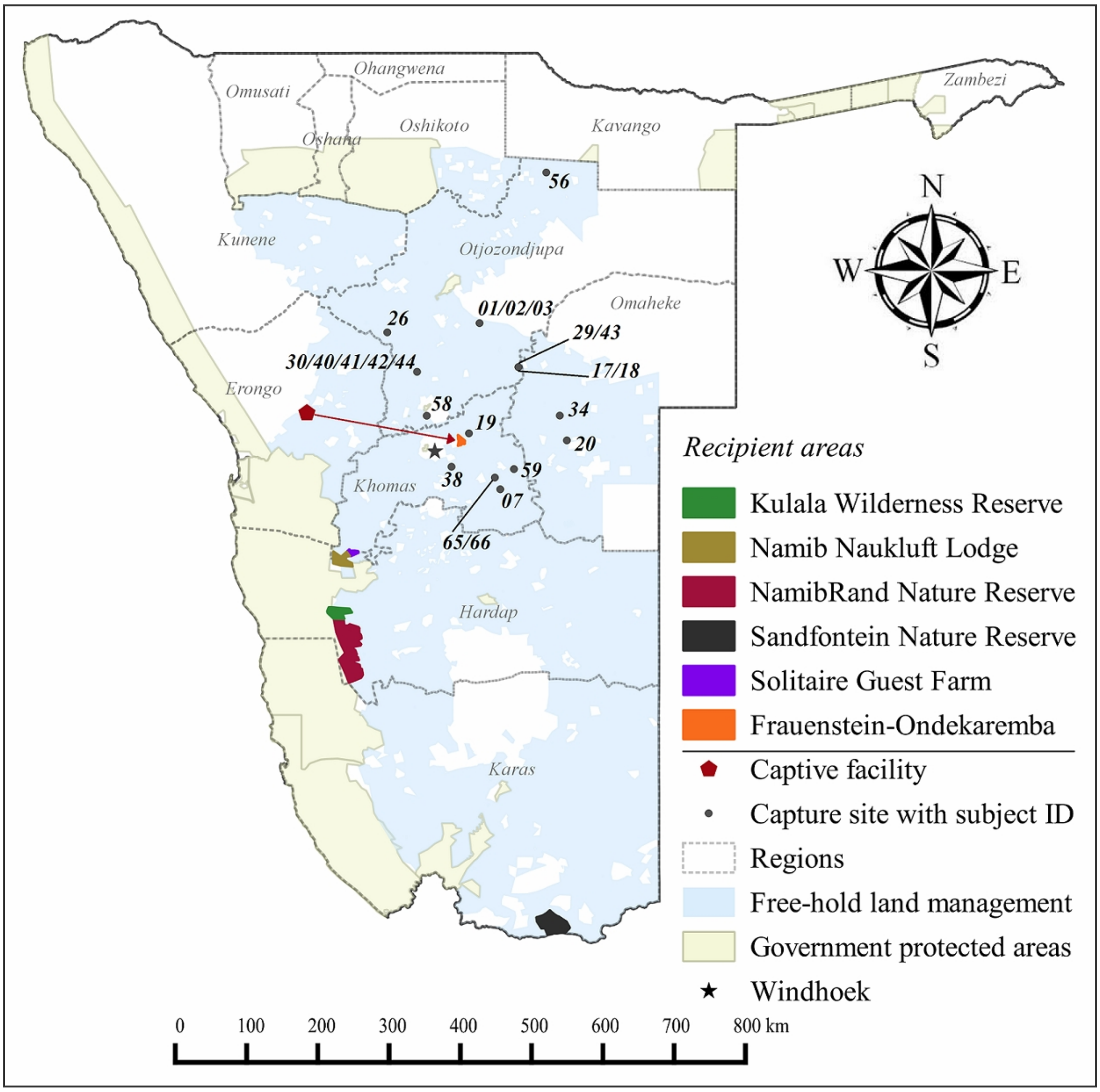


Figure 5.2 – Cheetah capture sites and recipient reserves across Namibia, 2008 - 2012. *Recipient reserve locations are shown in relation to government PAs and free-hold land tenure. The distribution of capture sites shows a bias towards Namibia's central stock-farming areas where cheetahs proliferate and do not experience competition from spotted hyaena.*

According to these stipulations, *CaTSuiT* (Lemeris Jr., 2015) was modified with the most recent available input data (Table 5.2) to determine potential cheetah recipient area suitability across all registered PAs in Namibia - including national parks (137,451.5 km^2), private conservation areas (4,883.9 km^2), and gazetted communal conservancies (160,245.5 km^2).

For any future translocations, the final assessment would also need to consider natural prey as a critical element of suitability (Hayward, 2007c, 2009; Lindsey et al., 2011; Winterbach et al., 2015), but it was impossible to source reliable, standardised wildlife density information for all designated PAs in the country. Moreover, PAs in Namibia are

Table 5.2 – Model input parameter characteristics to determine cheetah recipient area suitability.

Suitability condition	Exclusion criteria	Consideration [supporting studies]	Source of model input data
Land-use	Areas without designated protected status (e.g. commercial, communal farmlands)	1) Reduce risk of conflict or persecution outside of designated recipient area [du Preez, 1970; Phiri, 1996; Marker et al., 2003a; Houser et al., 201; Boast et al., 2015]	Mendelsohn et al., 2002 ConInfo, 2010, 2011
Intra-guild competition	Medium and high lion occurrence Medium and high spotted hyaena occurrence	1) Improve probability of post-release survival and successful reproduction 2) Reduce probability of kleptoparasitism [Caro, 1974; Eaton, 1974; Stander, 1990b; Laurenson, 1994; Durant, 1998a, 2000; Mills et al., 2004; Radloff and du Toit, 2004; Purchase et al., 2006; Hayward et al., 2007a; Marnewick et al., 2009; Wachter et al., 2011]	Stein et al., 2012
Intra-specific competition	High occurrence of conspecifics No cheetah occurrence	1) Provide connectivity and mating opportunities with free-ranging conspecifics at low – medium occurrence 2) Reduce risk of intra-specific competition/aggression (particularly for males) 3) Local carrying capacity [Eaton, 1970a; Caro and Collins, 1987; Caro and Laurenson, 1989; Caro, 1994; Hunter, 1998b; Hayward et al., 2007a, 2007c; Lindsey et al., 2011]	Stein et al., 2012
Site fidelity	Connected protected area patch size < max. observed exploratory movements	1) Reduce risk of conflict and persecution outside of targeted recipient area [du Preez, 1970; Pettifer et al., 1982; Phiri, 1996; Marker et al., 2003a; Houser et al., 2011; Boast et al., 2015]	This study
Urban areas	50 km buffer radius	1) Safety	Namibia Statistics Agency, 2012

rarely fenced in a fashion that obstructs wildlife movements effectively (Jones, 2014) and, therefore, local prey densities may fluctuate significantly and according to seasons. Sufficient prey availability should therefore be determined case-specifically with release area management prior to any translocation events. Except for Namibia's desert coastline, all habitats were considered as potentially suitable because historic species distribution records indicate that cheetahs were found widespread across the country during the 20th century (e.g. Gaerdes, 1974; Kraus and Marker-Kraus, 1991; Marker-Kraus et al., 1996). Furthermore, cheetahs have shown a remarkable tolerance of and adaptability to different biomes and habitat conditions (see Introduction).

Recipient area suitability was determined in a step-wise elimination process. Hence, the order of parameter inclusion had no influence on the final results. Two different model iterations were computed. First, all suitability criteria were included as outlined in Table 5.2. Second, site fidelity was excluded as a critical suitability parameter, reflecting the original design of the present study (chapter 2).

5.3 Results

5.3.1 Sample characteristics

The outcomes of 15 cheetah translocations, comprising 23 adults (12 males, 11 females) with 10 dependent offspring, were investigated for a total of 7,725 monitoring days (adults only) between 2008 and 2014. These translocations involved three cases with livestock depredation backgrounds (cheetahs observed on livestock kills), 15 indiscriminately captured animals (alleged livestock predation without evidence, or game predation), and another five individuals released after long-term captive rehabilitation (Table 5.1). Prior to release at an average distance of 419.6 km ± 216.1 km SD (range = 71 – 816 km) cheetahs were held captive for 1 – 1,184 days (350.6 days ± 439.0 days SD, $n = 23$), reflecting case-specific circumstances that were mainly influenced by availability of recipient areas, rearing of orphans, presence of offspring, as well as permit acquisition.

Both hard ($n = 13$) and soft releases ($n = 10$) were employed – the latter with a median acclimatisation period of 18.25 weeks (range = 1 – 38 weeks, $n = 10$). Cheetahs were released solitarily ($n = 5$), as mothers with dependent sub-adults ($n = 4$, + 10 offspring), as sibling groups ($n = 7$, three events) or as groups artificially bonded during captivity ($n = 7$, three events) (Table 5.1). Cheetahs were monitored with combined GPS satellite - VHF transmitters ($n = 12$), VHF-only transmitters ($n = 10$), and female Aju02 was tagged with an ID collar as part of a group release. The GPS unit of female Aju40 was fitted to female Aju41 at a later stage. According to the behavioural criteria outlined in

Table 2.1 (chapter 2, section 2.3), 15 cheetahs were classed as wild, four as semi-habituated, and five as fully habituated to human presence (not tame) at the time of release. All sub-adults released alongside their mothers retained wild characteristics. Except for those individuals released into central Namibia (five adults, three offspring), translocated cheetahs (18 adults, seven offspring) were moved into areas where they experienced novel intra-guild competition from spotted hyaenas.

5.3.2 Survival

Three GPS units (Aju20/26/30) failed during the first year. Of the remaining 20 collared adults, 10 died within 12 months of release, with the majority of deaths (n = 8) occurring during the first 71 days (Table 5.3). For all collared cheetahs (n = 23), the annual Kaplan-Meier survivorship estimate was 0.57 (95% CI: 0.35 – 0.76) (Fig. 5.3). The estimates for males (K-M = 0.56, 95% CI: 0.26 – 0.82, n = 12) and females (K-M = 0.53, 95% CI: 0.23 – 0.81, n = 11) did not significantly differ (z = -0.18, p = 0.857) (Fig. 5.3).

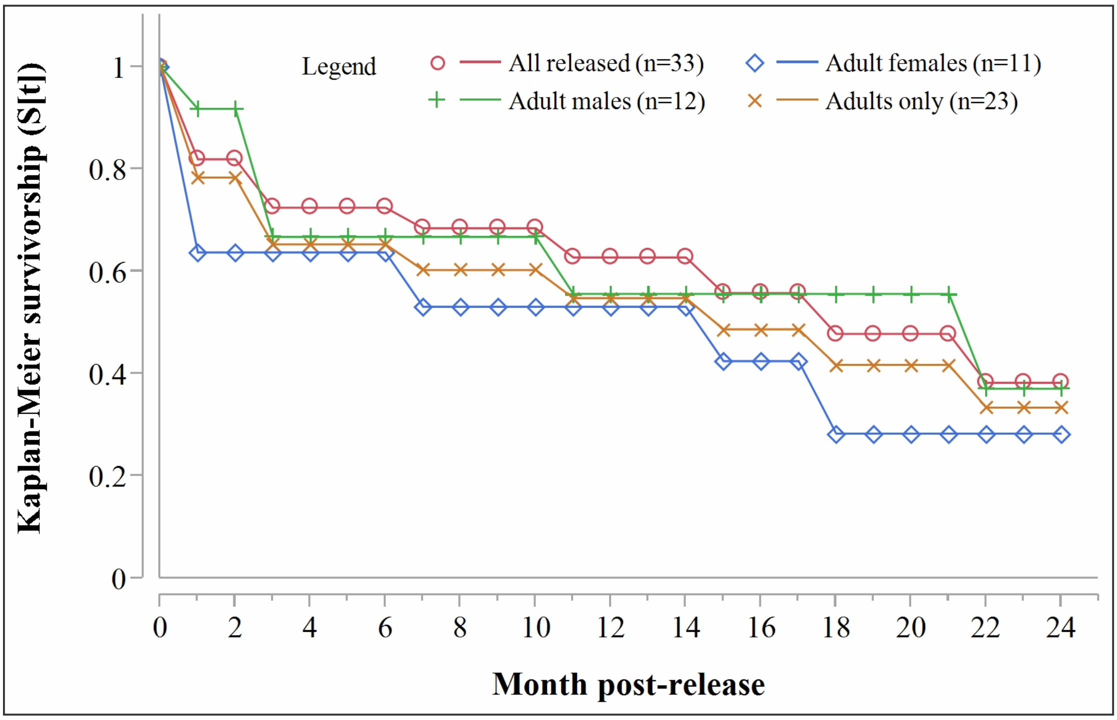


Figure 5.3 – Progressive Kaplan-Meier survivorship estimates for translocated cheetahs. *Data are standardised to month post-release.*

Despite acclimatisation to recipient areas, soft released cheetahs had lower survivorship in year one (K-M = 0.40, 95% CI: 0.14 – 0.73, n = 10) than hard released individuals (K-M = 0.67, 95% CI: 0.37 – 0.88, n = 13) (Fig. 5.3) but not significantly so (z = 1.31, p = 0.190). Fully habituated cheetahs (n = 5) had the lowest survivorship (Fig. 5.4). None of them survived for longer than 205 days. Private landowners shot four outside of recipient areas and a spotted hyaena killed the fifth – giving a median survival period of 2.4 months (range = 0.6 – 6.8 months, $n = 5$). According to land managers who killed fully habituated cheetahs, the animals were shot because they showed no natural flight response when approached, rather than for reasons of conflict – e.g. Aju42/43/44 were involved in livestock depredation but these incidents were compensated and resolved non-lethally. Although based on a small sample, the association between first year survival and degree of habituation was significant ($\chi^2 = 8.63$, d.f. = 2, $p = 0.013$, $n = 20$). There was no such association for cheetahs released in different group compositions ($\chi^2 = 1.38$, d.f. = 2, $p = 0.501$, $n = 20$).

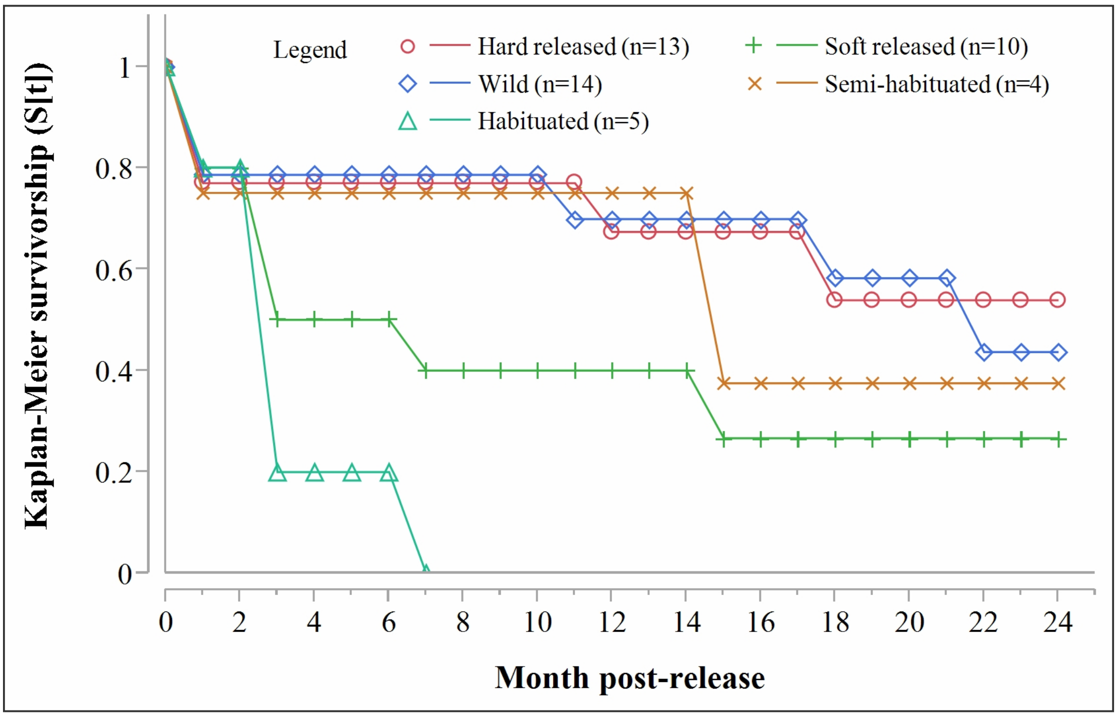


Figure 5.4 – Progressive Kaplan-Meier survivorship estimates for translocated cheetahs by degree of habituation and release mode. *The graph demonstrates the influence of degree of habituation to human presence on likelihood of survival, with none of the fully habituated cheetahs surviving for longer than 205 days post-release. Data are standardised to month post-release.*

In addition to the early deaths of fully habituated individuals, two spotted hyaenas killed male Aju19 13 days after release. Female Aju07 was re-captured 11 days after release due to poor physical condition and lack of hunting. She had become emaciated and dehydrated and succumbed to acute renal failure (I. Baines, pers. comm. 2010) >12 months later. After leaving the recipient area, female Aju59 died 19 days post-release due to severe injuries sustained during accidental capture in a gin trap set for control of black-backed jackals (*Canis mesomelas*). Female Aju18 appeared to have died from shock/exhaustion 14 days post-release – a post-mortem delivered no definitive results. She hunted successfully on the day of her release and found at least two permanent water holes on the recipient reserve. Male Aju03 was found with a broken pelvis/spine near a rocky outcrop after almost one year, but the carcass showed no obvious signs of a hyaena attack.

Following three additional mortalities (Aju01/38/58 in Table 5.3) and 'censoring' of four cases with depleted tracking collars in year two after release, the progressive Kaplan-Meier survivorship estimate for adults decreased to 0.40 (95% CI: 0.21 – 0.62) (Fig. 5.3). Assessed independently across years post-release, adult survivorship was higher in year two (K-M = 0.70, 95% CI: 0.35 – 0.92, *n* = 10) than year one (see above) but not significantly so (z = -0.70, *p* = 0.483). After highest mortality during the first three months cheetah survivorship improved and remained ≥0.80 (Fig. 5.5; Appendix 21), suggesting that the initial post-release orientation/exploration period is particularly important in terms of survival. Excluding cases with unknown outcomes, cheetahs surviving the first 90 days (*n* = 12) had an 83.3% chance of surviving one year and a 37.5% chance of surviving for two.

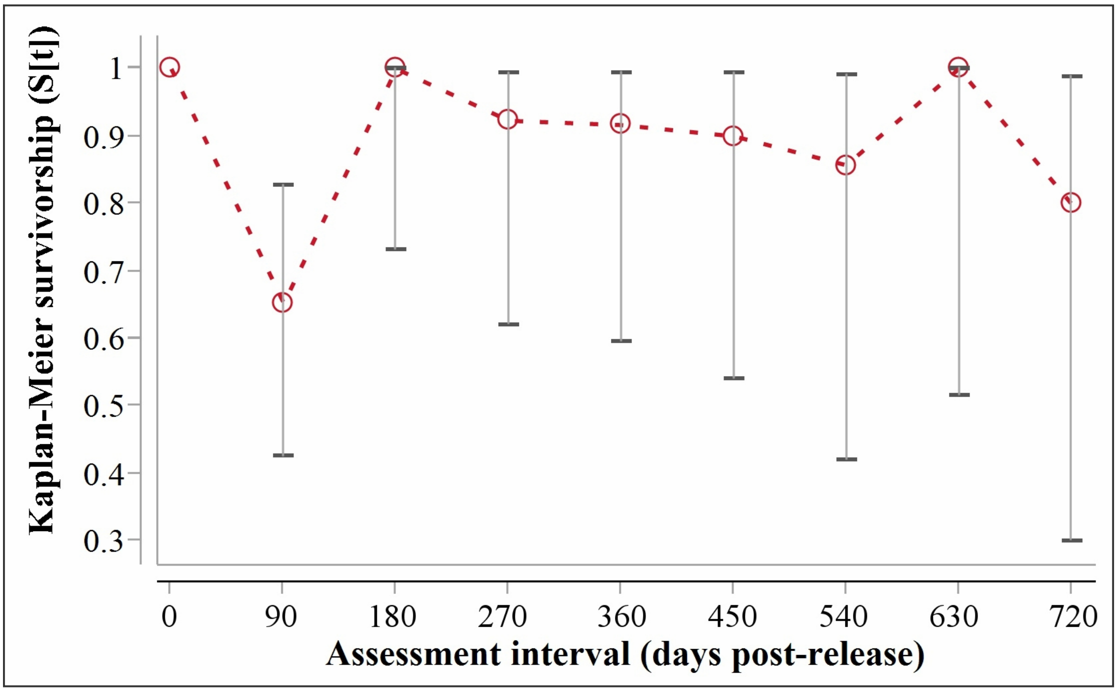


Figure 5.5 – Kaplan-Meier survivorship analysis for adult translocated cheetahs (*n* = 23) assessed in 90-days intervals post-release. *The graph demonstrates that cheetah survival was lowest during the first three months post-release, thereafter consistently remaining*

≥0.80. Error bars show the 95% CI of the K-M survivorship estimate. Confidence intervals increased as sample size decreased.

Overall, human persecution (*n* = 7) and spotted hyaena attacks (*n* = 4) had the highest impact on cheetah survival and accounted for 50.0% and 28.6% of confirmed adult cheetah mortalities (*n* = 14) respectively (Table 5.3). Eight translocated adults died outside PAs whereas six animals died in conservation areas. The mean known survival period of translocated adults was 12.1 ± 11.1 months SD (range = 0.4 – 33.0 months, *n* = 20). The survival periods recorded for females (10.5 months ± 10.2 months SD, range = 0.4 – 28.0 months, *n* = 10) and males (13.7 months ± 12.3 months SD, range = 0.4 – 33.0 months, *n* = 10) did not significantly differ (Wilcoxon-Mann-Whitney U-Test: U = 58.5, *p* = 0.544). Hard released cheetahs, on average, survived for six months longer (14.8 months ± 11.7 months SD, range = 0.4 – 33.0 months, *n* = 11) than soft released ones (8.7 months ± 10.1 months SD, range = 0.6 – 28.0 months, *n* = 9), but there was no significant difference between release modes (U = 85.5, *p* = 0.517) (Fig. 5.4).

The monitoring of sub-adults released with their mothers was restricted to opportunistic observations. Despite the early death of Aju59, her three offspring survived in the same area for at least another nine months. There was only one confirmed mortality for released sub-adults – one cub of Aju07 was found dead one month after release. The median known survival period for released offspring was 6.5 months (range = 0.5 – 13.0 months, *n* = 10).

5.3.3 Reproduction

Of the six females that survived >90 days, four produced litters after settling into permanent ranges. Two females (Aju56/58) successfully reared offspring released with them (*n* = 5) and reproduced again (eight cubs). Females Aju02/29 produced first offspring (*n* = 6) (Table 5.3). Median litter size after emergence from lairs was 3.5 (range = 2 – 5, *n* = 4). Considering a gestation period of ~94 days for African cheetahs (Brown et al., 1996) and emergence of cubs from the lair at eight weeks of age (Laurenson, 1994), Aju02 conceived as soon as three months post-release. At that time, a resident coalition of three male cheetahs was reported crossing through the release area of Aju02 (A. Keding, pers. comm. 2008). New litters of females Aju29/56/58 were first observed between 13 - 14 months post-release when cubs appeared to be between 2 - 4 months old (according to phenotypic characteristics described in Eaton, 1969). The litters of females Aju02/56/58 were found along rocky mountain outcrops and hill-sides whereas Aju29 hid her cubs in

Table 5.3 – Outcomes of cheetah translocations.

ID	Known survival (monitoring days)	Homing?	Cause of death	Conflict?	Reproduction	Success?	Comments
Aju01	519	No	Spotted hyaena	No	---	Yes	Stable sibling group
Aju02	426	No	---	No	4 cubs	Yes	Stable sibling group
Aju03	322	No	Natural accident (suspected)	No	Unknown	No	Stable sibling group
Aju07	11	No	Recaptured	No	---	No	Death from acute renal failure (captivity)
Aju17	960	No	---	No	Mating suspected	Yes	---
Aju18	14	No	Heat shock/Exhaustion	No	---	No	---
Aju19	13	No	Spotted hyaena	No	---	No	Split upon release
Aju20	67	No	---	Unknown	Unknown	Unknown	Split upon release – collar failure
Aju26	112	No	---	Unknown	---	Unknown	Collar failure
Aju29	840	No	---	No	2 cubs	Yes	Split from Aju30 upon release
Aju30	290	No	---	No	---	Unknown	Male accidentally trapped and re-released
Aju34	991	No	Injury – Shot	No	Unknown	Yes	Shot due to severe front leg injury
Aju38	636	No	Spotted hyaena	No	Mating suspected	Yes	---
Aju40	19	No	Spotted hyaena	No	---	No	---
Aju41	205	No	Shot	No	---	No	Killed due to lack of fear of humans
Aju42	71	No	Shot	Yes	---	No	Killed due to lack of fear of humans
Aju43	71	No	Shot	Yes	---	No	Killed due to lack of fear of humans
Aju44	71	No	Shot	Yes	---	No	Killed due to lack of fear of humans
Aju56	680	No	---	No	3 cubs	Yes	---
Aju58	427	No	Shot – livestock carcass	Yes	5 cubs	Yes	Killed due to livestock depredation
Aju59	19	No	Gin trap	No	---	No	Sub-adults survived after female's death
Aju65	371	Yes	---	No	Unknown	No	Stable coalition
Aju66	590	Yes	---	No	Unknown	No	Stable coalition

the sand dunes of the Namib Desert. The known median survival period for new-born cubs was 8.0 months (range = 1 – 14 months, *n* = 14) and although cubs could not be followed reliably, at least 64.3% reached an age of nine months.

In addition, two translocated males (Aju17/38) were observed during courtship behaviour with wild un-collared females, but mating events could not be confirmed.

5.3.4 Movements

5.3.4.1 Explorations and settling behaviour

All translocated cheetahs displayed exploratory movements that extended beyond the boundaries of target recipient reserves and connected PAs (Fig. 5.6). According to 10-day 100% MCP area progressions, the durations and spatial scales of exploratory movements were highly variable but distinct peaks occurred during the first 6.5 months (Fig. 5.7). Soft release did not prevent large scale explorations (Fig. 5.7) which were also not site-specific (Fig. 5.6).

Measured as the total 100% MCP area utilised, at least five individuals roamed over areas >2,000 km^2 during the first three months after release, and covered >4,000 km^2 within six months (Table 5.4). Hard released male Aju26 roamed across 19,743 km^2 within 111 days (until collar failure), whilst soft released male Aju30's movements encompassed 9,049 km^2 in 180 days. Extensive movements were also not sex-specific. Together with her three sub-adult offspring, Aju56 roamed across 13,596 km^2 within six months (Table 5.4). For those cheetahs that established/resumed ranges, settling occurred 13 – 190 days post-release (median = 93 days) (Table 5.4), whereas Aju26/30/41/42-44 showed no tendency to settle before collars failed or subjects died (Fig. 5.7).

Neither mountain escarpments with 500 – 600m elevations (often at near vertical inclines) nor true desert environments limited exploratory movements of cheetahs. Seven adults traversed large areas of the hyper-arid Namib Desert (see Aju30 in Fig. 5.6). Male Aju17 established a permanent range encompassing sand dunes and female Aju29 raised cubs in this environment.

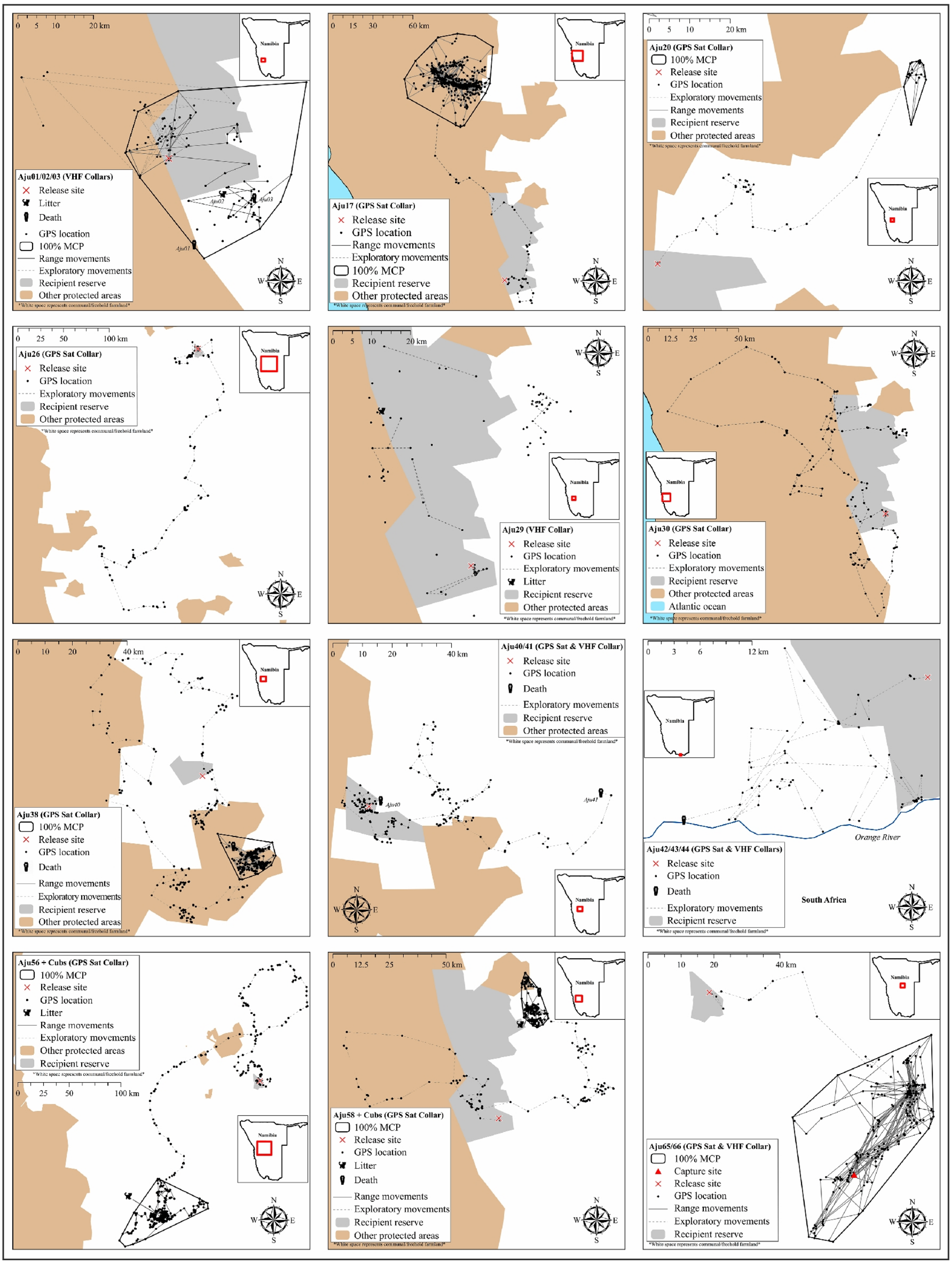


Figure 5.6 – Movements of cheetahs translocated into free-range environments in Namibia. *Movement plots illustrate variable site fidelity and ranging patterns of translocated subjects. Grey-shaded areas represent recipient reserves; red-shaded areas represent connected private and government PAs. GPS locations represent pruned daily location data. Range boundaries show 100% Minimum Convex Polygon calculations for post-exploration periods. Early deaths and case studies with <60 monitoring locations are excluded.*

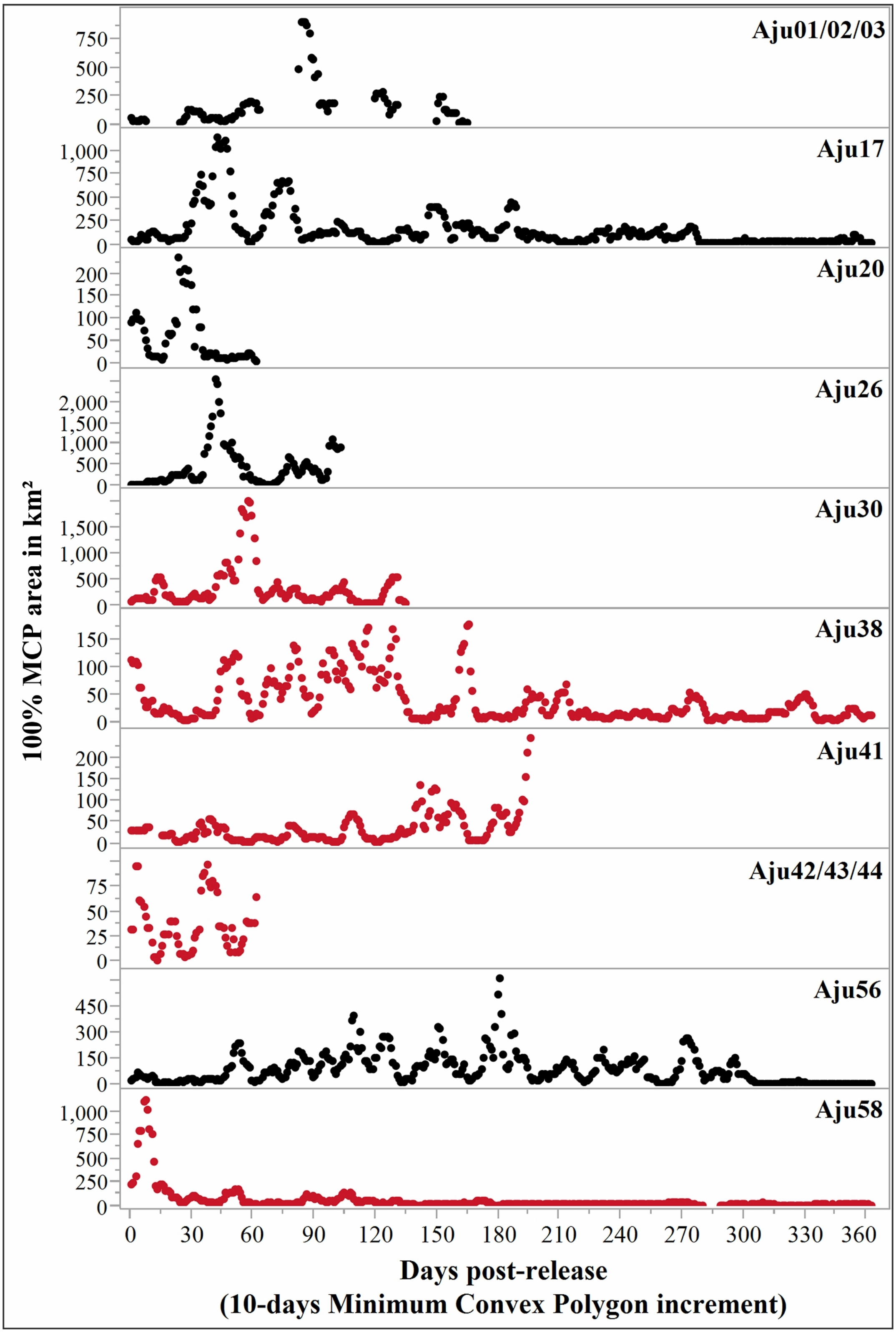


Figure 5.7 – Progressive 10-day 100% Minimum Convex Polygon analysis for hard released (black curves) and soft released (red curves) cheetahs. *The graph illustrates post-release settling behaviour as measured by progressive area use. Settling periods were highly variable and could not be confirmed for all case studies. Exploratory movements showed distinct peaks during the first 6.5 months after release. MCP values were calculated with pruned daily locations. Early deaths (<3 weeks) and case studies with*

insufficient movement records are excluded. Scales along y-axis differ between case studies. Area estimates were only calculated if at least five data points were available for 10-day segments.

5.3.4.2 Homing

Despite extensive post-release explorations, only the shortest translocation (males Aju65/66 at 71 km) resulted in successful homing (Fig. 5.8). This male coalition returned to its capture site (and assumed previous home range) within two weeks after release (panel Aju65/66 - Fig 5.6). All cheetahs translocated >137 km did not return to their capture sites or the captive facilities where they were held and there was no evidence of oriented, directional movements towards true home (Fig. 5.8).

For example, all GPS-collared cheetahs (except Aju66) monitored >90 days ($n = 7$) moved farther than their respective translocation distances within 3 - 5 months. However, 100 days post-release, these animals moved in directions with an average deviation of 71.5° ± 61.3° SD from true home. At that interval, only female Aju56 appeared home-bound (1.0° of true home), but she was located 400.8 km from her capture site and subsequently explored in the opposite direction (panel Aju56 - Fig. 5.6).

Pooled for all translocated cheetahs, the mean distance of last known locations to release sites (Table 5.4) was 57.1 km ± 63.2 km SD ($n = 23$) and, therefore, was <14% of the average translocation distance. For non-homing cheetahs, the mean distance of last known locations to capture sites was 454.1 km ± 197.8 km SD (range = 153.2 – 826.7 km, $n = 21$). The mean direction of an individual's last location in relation to release site deviated 173.5° ± 106.2° SD ($n = 23$) from true home and movements generally appeared to point away from home rather than toward it (Fig. 5.8). When analysed in 90° segments around the release centroid, the distribution of directions in relation to true home was random ($\chi^2 = 0.13$, d.f. = 3, $p = 0.988$, $n = 23$), suggesting that cheetahs did not orient toward any particular geographic direction. The distribution of angles located within and outside of the homing sector (337.5° - 22.5°), i.e. four = in, 19 = out, also significantly differed from the distribution that would be expected if all cheetahs had homed ($\chi^2 < 0.01$, d.f. = 1, $p < 0.001$, $n = 23$). Neither translocation distance ($r^2 = 0.059$, $p = 0.269$, $n = 23$) nor time in captivity ($r^2 = 0.012$, $p = 0.612$, $n = 23$) were strongly correlated with an individual's post-release orientation. These associations further weakened (translocation distance: $r^2 = 0.035$, $p = 0.418$, $n = 21$; captive time: $r^2 = 0.006$, $p = 0.748$, $n = 21$) when successful homers Aju65/66 were excluded.

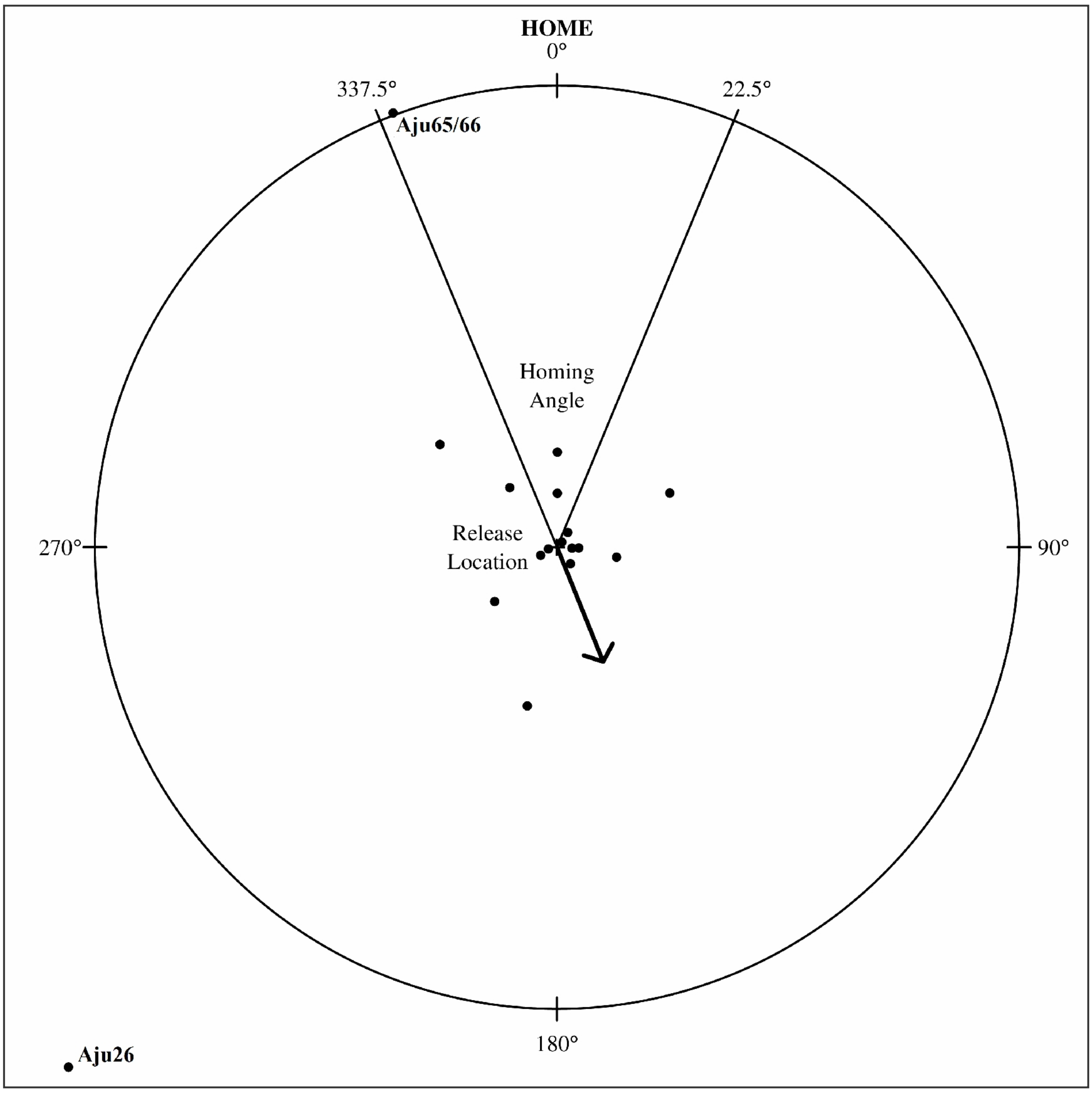


Figure 5.8 – Homing assessment for adult translocated cheetahs. *The distance from the circle's centre (individual's release location) to outer edge represents an individual's distance travelled relative to its capture location. The arrow shows mean angle (158.1° ± 27.8° SE) for all individuals and stable groups (n = 17). This analysis confirms ground monitoring information in that only the shortest translocation event (Aju65/66) resulted in successful homing to the capture site. Overall, directions travelled in relation to original capture sites and the captive facility appeared random.*

5.3.4.3 Ranges

Of the 14 individuals that were monitored for >90 days, 10 settled into recognisable range patterns (Fig. 5.6), including the return to a previous home range by homing males Aju65/66. Hence, only 34.8% of released individuals successfully established new ranges. Although VHF-tagged cheetahs Aju29 and Aju34 appeared to have settled successfully (e.g. Aju29 produced cubs), location data were too sporadic to calculate reliable range values. The GPS units of cheetahs Aju26 and Aju30 failed before these males showed

Table 5.4 – Summary movement statistics for translocated cheetahs.

ID	**Linear distance (km) moved in first 3 months (no. of fixes)**	**100% MCP[a] area (km^2) covered**		**Duration of post-release exploratory movements (days)**	**Estimated home range size in km^2**			**Percent overlap with recipient reserve (all PAs)**	**Distance (km) of centroid to release site**	**Distance (km) of last known location to release site**
		3 months (no. of fixes)	6 months (no. of fixes)		No. of fixes	100% MCP[a] (50%)	95% KDE[b] (50%)			
Aju01	Insufficient data	820.2 (54)	1,316.0 (103)	93	92	1,509.0 (284.6)	1,196.3 (108.6)	39 (54)	13.5	24.2
Aju02	Insufficient data	820.2 (54)	1,316.0 (103)	93	92	1,509.0 (284.6)	1,196.3 (108.6)	39 (54)	13.5	17.0
Aju03	Insufficient data	820.2 (54)	1,316.0 (103)	93	92	1,509.0 (284.6)	1,196.3 (108.6)	39 (54)	13.5	24.7
Aju07	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	3.9
Aju17	524.1 (89)	7,905.2 (89)	8,373.6 (181)	85	883	3,209.1 (253.9)	931.7 (203.1)	0 (89)	147.1	159.4
Aju18	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	38.3
Aju19	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	10.9
Aju20	n/a	989.8 (64)	n/a	36	31	49.0 (4.4)	51.8 (6.3)	0 (0)	51.7	81.5
Aju26	590.5 (91)	11,474.1 (91)	n/a	>112	---	n/a	n/a	n/a	n/a	281.0
Aju29	Insufficient data	1,068.5 (46)	1,068.5 (56)	Unknown	---	n/a	n/a	n/a	n/a	9.2
Aju30	577.6 (92)	7,907.3 (92)	9,048.9 (141)	>290	---	n/a	n/a	n/a	n/a	71.0
Aju34	Insufficient data	Insufficient data	Insufficient data	Unknown	---	n/a	n/a	n/a	n/a	24.4
Aju38	265.8 (92)	1,179.9 (92)	3,292.6 (181)	171	418	217.0 (26.9)	121.6 (18.3)	0 (74)	26.1	28.2
Aju40	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	4.4
Aju41	147.1 (90)	138.9 (90)	1,099.5 (160)	n/a	---	n/a	n/a	n/a	n/a	78.0
Aju42	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	31.9
Aju43	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	31.9
Aju44	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	31.9
Aju56	250.1 (87)	2,465.0 (87)	13,596.0 (179)	190	497	2,624.8 (94.6)	929.5 (81.5)	0 (0)	118.1	140.6
Aju58	363.0 (91)	2,520.1 (91)	4,694.1 (181)	114	318	176.8 (18.6)	138.7 (13.9)	0 (14)	41.7	59.2
Aju59	n/a	n/a	n/a	n/a	---	n/a	n/a	n/a	n/a	26.0
Aju65	582.6 (91)	1,877 (91)	2,015.0 (165)	13	275	1,614.4 (370.1)	1,182.2 (193.2)	0 (0)	64.7	71.1
Aju66	582.6 (91)	1,877 (91)	2,015.0 (165)	13	275	1,614.4 (370.1)	1,182.2 (193.2)	0 (0)	64.7	65.2

[a] MCP = Minimum Convex Polygon

[b] KDE = Kernel Density Estimation

settling behaviour (Fig. 5.7).

Range sizes (excluding Aju65/66) in semi-arid to hyper-arid areas of south-western Namibia varied between 49.0 – 3,209.1 km^2 (Table 5.4). The average distance of geographic home range centroids to release sites was 66.4 km ± 53.7 km SD (range = 13.5 – 147.1 km, n = 6). Only the range of Aju01-03 overlapped with the recipient reserve whereas two ranges did not overlap with any PAs and four had partial overlap with other conservation areas (Fig. 5.6; Table 5.4). Average percent overlap of ranges with all PAs was 38.5% ± 39.0% SD (range = 0 - 89.0%) and ranges appeared stable until monitoring ceased. Also, the movements of cheetahs Aju29 and Aju30 had high overlap with the recipient reserve and connected PAs (Fig. 5.6).

5.3.4.4 Site fidelity

Due to extensive explorations, site fidelity was highly variable and overall appeared low (Table 5.5). Mean overlap of daily locations with recipient reserves in year one was 39.8% ± 7.2% SE (n = 23). All cheetahs left recipient reserves and moved into free-hold farmland areas, or into other private and public conservation areas (Fig. 5.6). Site fidelity increased significantly to 54.9% ± 7.5% SE when defined to include occupancy of other conservation areas (Table 5.5). This is important as it demonstrates the value of conservation area connectivity to assure site fidelity, and hence safety, for cheetahs released into free-ranging environments.

Site fidelity of males and females (Table 5.5) did not differ significantly when assessed for recipient reserves only (U = 9.5, p = 0.187) and for all PAs (U = 2.5, p = 0.410). Likewise, site fidelity did not differ between hard and soft released animals (for recipient reserves and all PAs: U = -9.5, p = 0.855). There was also no difference in the mean number of days that hard released (19.0 days ± 15.4 days SD, n = 12) and soft released (19.7 days ± 23.1 days SD, n = 10) cheetahs spent within a 10 km radius around release sites (U = 79.0, p = 0.706). For soft released individuals, time spent in acclimatisation pens did not significantly influence site fidelity on recipient reserves (r_s = 0.331, p = 0.349, n = 10).

Cheetah site fidelity results were also not significantly influenced by recipient area size (r^2 = 0.073, p = 0.209, n = 23). Although mean percent overlap of positional data with recipient areas was higher for cheetahs released onto reserves measuring >700 km^2 (45.7% ± 29.5% SD, n = 13) than for those released onto reserves <350 km^2 (32.1% ± 40.1% SD, n =10), the difference was not significant (U = 117.0, p = 0.099). This shows that even the largest private PAs (e.g. NamibRand Nature Reserve with 1,722 km^2 in Fig. 5.2) cannot

Table 5.5 – Percent site fidelity of translocated cheetahs.

Category	**No. of locations (*n*)**	**A - All conservation area**				**B - Recipient reserves only**				**Comparison A vs. B**
	(first 12 months)	Min	Mean (SE)	Max	Total	Min	Mean (SE)	Max	Total	Wilcoxon signed rank test
All adults (*n* = 23)	3,199	1.0	54.9 (7.5)	100	44.5	0.8	39.8 (7.2)	100	16.9	S = 27.5, *p* = 0.001[*]
Males (*n* = 12)	1,916	1.0	50.3 (10.6)	100	50.3	0.8	29.3 (8.5)	100	12.2	S = 5.0, *p* = 0.062
Females (*n* = 11)	1,283	4.7	59.9 (10.9)	100	35.8	3.6	51.3 (11.2)	100	23.9	S = 10.5, *p* = 0.015[*]
Soft released (*n* = 10)	1,340	26.7	54.3 (8.2)	100	53.3	0.8	37.0 (8.8)	100	19.3	S = 7.5, *p* = 0.031[*]
Hard released (*n* = 13)	1,859	1.0	55.4 (12.0)	100	38.0	1.0	42.0 (11.0)	100	15.2	S = 7.5, *p* = 0.031[*]
GPS collars (*n* = 12)	2,336	1.0	46.2 (10.7)	100	44.7	0.8	24.8 (8.7)	100	10.8	S = 10.5, *p* = 0.015[*]
VHF collars (*n* = 11)[a]	863	1.0	64.4 (10.2)	100	47.8	1.0	56.2 (9.7)	100	33.4	S = 5.0, *p* = 0.062

[a] including female Aju02 with ID collar

[*] denotes statistical significance at α = 0.05

reliably contain extensive post-release movements. Of all cheetahs monitored >90 days (*n* = 14), eight left recipient reserves permanently whilst six returned sporadically. Neither time in captivity ($r^2 = 0.012$, $p = 0.612$, $n = 23$) nor translocation distance ($r^2 = 0.012$, $p = 0.612$, $n = 23$) were significantly associated with site fidelity results.

Although deployed at near-equal rates (GPS units = 12, VHF/ID units = 11), VHF units provided <27.0% of positional data in year one. However, the site fidelity estimate for this category (recipient reserves only) was more than twice as high as that for GPS-tracked cheetahs (Table 5.5). VHF-tagged cheetahs could not be detected for prolonged periods (range = 1 – 253 days), resulting in 124 data gaps (at least one day missing between subsequent locations) with a mean gap length of 11.0 days ± 34.8 days SD. Conversely, GPS collars produced only 37 data gaps and the mean gap length (3.4 days ± 4.6 days SD, range = 1 – 22 days) was significantly shorter (one-sided t = 2.36, $p = 0.009$). Considering this impact of tracking technology on detection probability, true cheetah site fidelity probably was <24.8% on recipient reserves, and <46.2% on all PAs (category GPS in Table 5.5).

5.3.5 Group cohesion

True sibling groups remained stable after release, or until death occurred. Female Aju01 remained associated with Aju02 after this female produced a litter and after the associated male Aju03 died. During the monitoring of this group, it appeared that male Aju03 led exploratory movements, and the two females followed his path after 6 – 12 h on several occasions. The adult male coalition Aju65/66 was still observed together >12 months post-release and the artificial male coalition Aju42-44, consisting of two brothers and one unrelated male, also remained stable until death. Conversely, artificial group Aju29/30, consisting of an unrelated male and female of the same age, immediately split upon release, as did unrelated adults Aju19 (male) and Aju20 (female) that were introduced to each shortly before translocation.

5.3.6 Prey and conflict

Medium to large ungulates comprised the bulk of documented wildlife prey (*n* = 76) (Fig. 5.9), with springbok being the preferred prey species (65.8% of known kills). Cheetahs killed approximately similar proportions of sub-adult and adult ungulates ($\chi^2 = 1.35$, d.f. = 1, $p = 0.245$, $n = 74$) and showed no preference for different springbok age classes ($\chi^2 = 0.32$, d.f. = 1, $p = 0.572$, $n = 50$). Female Aju18 hunted as soon as ~8 h after release, whereas Aju07 did not begin to hunt for 11 days, resulting in her re-capture.

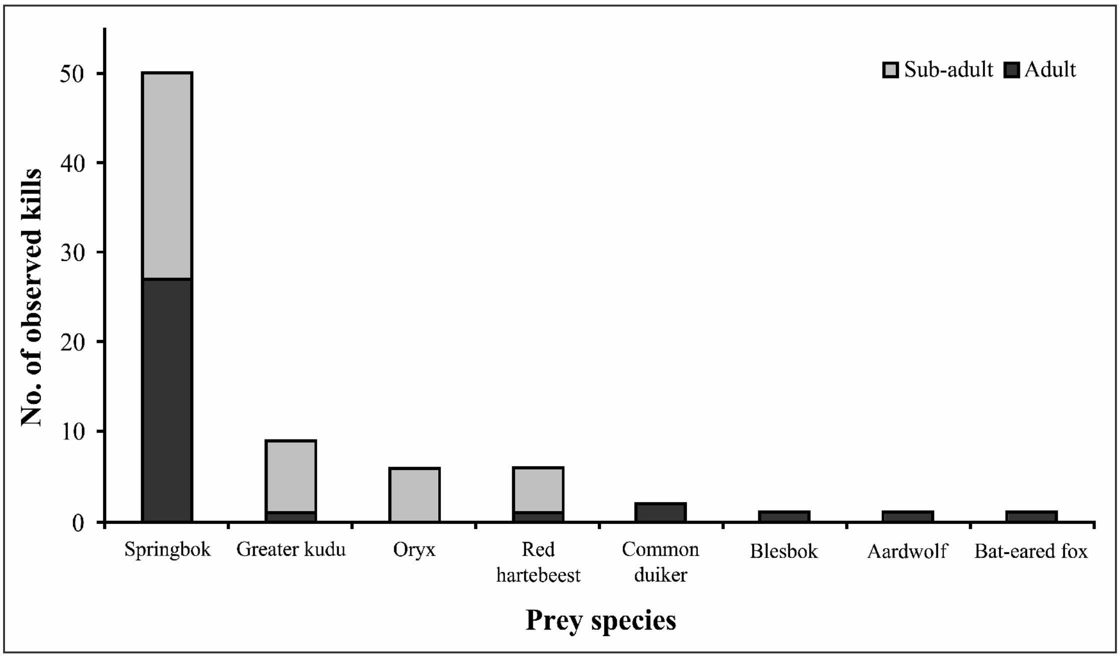


Figure 5.9 – Documented wildlife prey (*n* = 76) of translocated cheetahs. *Prey items were recorded following investigation of GPS clusters or direct observation of hunts. The graph shows a disproportionately high contribution of springbok to overall prey.*

With the exceptions of female Aju58 that was shot at a cattle calf carcass >14 months post-release and the male coalition Aju42-44 that killed 19 smallstock across 11 days after leaving their recipient reserve (the damage was compensated), data sharing with land managers showed that translocated cheetahs caused little conflict (Table 5.3). Therefore, none of the known or suspected livestock raiders (Table 5.1) was involved in livestock depredation even though these individuals moved across free-hold farmland areas outside designated PAs temporarily or permanently (Fig. 5.6). During explorations, however, cheetahs frequently roamed across multiple farmland properties within 24 h periods, making effective data sharing impossible. It remains unknown whether they killed livestock during these periods.

Farmers from whose properties known livestock raiders had been moved (*n* = 2) reported new conflict with cheetahs 12 – 24 months later, showing that translocation was not a permanent solution. Due to continued perceived threats to valuable game (*n* = 5) and livestock (*n* = 4), the majority of managers (64.3%) from whose properties cheetahs were translocated (*n* = 14), requested repeat removals within two years of first events, and 50% continued to trap cheetahs indiscriminately. Consequently, first translocations did not result in once-off problem mitigation and there was considerable demand for continued removals. These managers also did not improve protective measures for game and livestock whereas three livestock farmers adjusted husbandry by night-kraaling of juvenile cattle (Appendix 18), permanent herder presence during day-time hours, and shifting of herds from areas with known cheetah marking trees.

5.3.7 Success

Success evaluations were possible for 20 translocated adults (Table 5.3), giving an overall success rate of 40.0%. The GPS units of three individuals failed before 12 months had elapsed, although there was no indication that these animals were unsuccessful – e.g. male Aju30 was still observed on his recipient area 2.5 months prior to qualifying as successful. Fully habituated, long-term captive cheetahs were particularly unsuccessful and the success rate increased to 53.3% (*n* = 15) if these individuals were not considered.

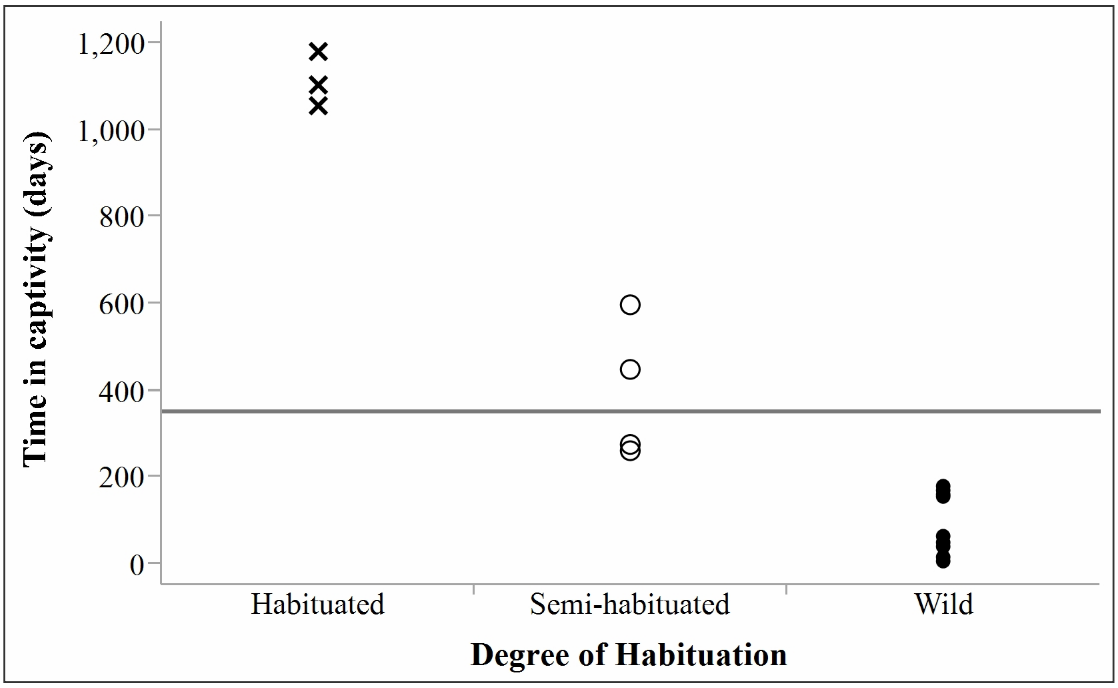


Figure 5.10 – Degree of habituation to humans exhibited by cheetahs as influenced by time spent in temporary captivity. *The graph demonstrates that prolonged captivity progressively influences a cheetah's response to human proximity. Horizontal line shows mean captive time (350.6 days ± 439.0 days SD) for all adult individuals (n = 23). Note: individual data points may represent multiple individuals held captive for the same period.*

Translocation success was not significantly associated with sex (χ^2 = 0.84, d.f. = 1, p = 0.359, n = 20) or release mode (χ^2 = 1.29, d.f. = 1, p = 0.252, n = 20) but with degree of habituation to humans (χ^2 = 6.47, d.f. = 2, p = 0.039, n = 20). The latter was strongly associated with the amount of time that cheetahs spent in captivity (LogLikelihood = 21.58, χ^2 = 43.15, d.f. = 2, $p < 0.001$, n = 23), suggesting that animals intended for free-range release should not be raised in captivity and ideally be held <250 days (Fig. 5.10) to increase their chances of survival in anthropogenic landscapes. Spotted hyaena related mortality ($n = 4$), as another important factor influencing survival, accounted for 33.3% of confirmed deaths ($n = 12$) of adults translocated into areas with new competition ($n = 18$).

5.3.8 Potential cheetah recipient areas in Namibia

Including all parameters to determine recipient area suitability across Namibia's PA network (Table 5.2; Appendix 22), *CaTSuiT* did not identify any suitable areas for cheetah releases. Due to extensive exploratory movements (Fig. 5.6; Table 5.4), site fidelity had the highest individual parameter impact on area suitability (Table 5.6), eliminating 95% of available public and private PAs. Currently few unfenced PAs exist that are large enough to prevent cheetahs from re-entering commercial farmlands where persecution presents a serious concern (chapter 3; Table 5.3).

Table 5.6 – Impact of individual parameters on potential cheetah recipient area suitability.

Exclusion criterion (Table 5.2)	**Suitable area (km^2) after exclusion from available PAs (302,580.9 km^2)**	**Percent of available PAs eliminated**	**Comment**
High and zero cheetah occurrence	81,157.0	73.2	Appendix 24
Medium – high lion occurrence	176,739.5	41.6	Appendix 25
Medium – high spotted hyaena occurrence	86,883.0	71.3	Appendix 26
Urban safety buffer	269,902.8	10.2	Appendix 27
Site fidelity (all translocated cheetahs)	15,071.3	95.0	Fig. 5.11

Excluding site fidelity as a prerequisite for cheetah translocation success, *CaTSuiT* identified 10 isolated patches (cumulative size = 15,071.3 km^2) in nine PAs (Fig. 5.11; Appendix 23) that fulfilled suitability criteria (see individual modelling steps in Appendix 24 – 27). However, nine of these patches represent suitable areas <1,800 km^2, six patches were <400 km^2, and three patches were <100 km^2 (Appendix 23), suggesting that most translocated cheetahs will move beyond their boundaries within three months as part of their post-release explorations (Table 5.4). Vast movements away from suitable patches would likely result in cheetahs entering areas of communal and/or commercial livestock production (Fig. 5.1), areas with high occurrence of resident conspecifics (Appendix 24), or areas with medium-high occurrence of lion and spotted hyaena (Appendices 25 and 26), thus significantly compromising chances of post-release survival (e.g. Purchase et al., 2006; Marnewick et al., 2009; Houser et al., 2011; Boast et al., 2015).

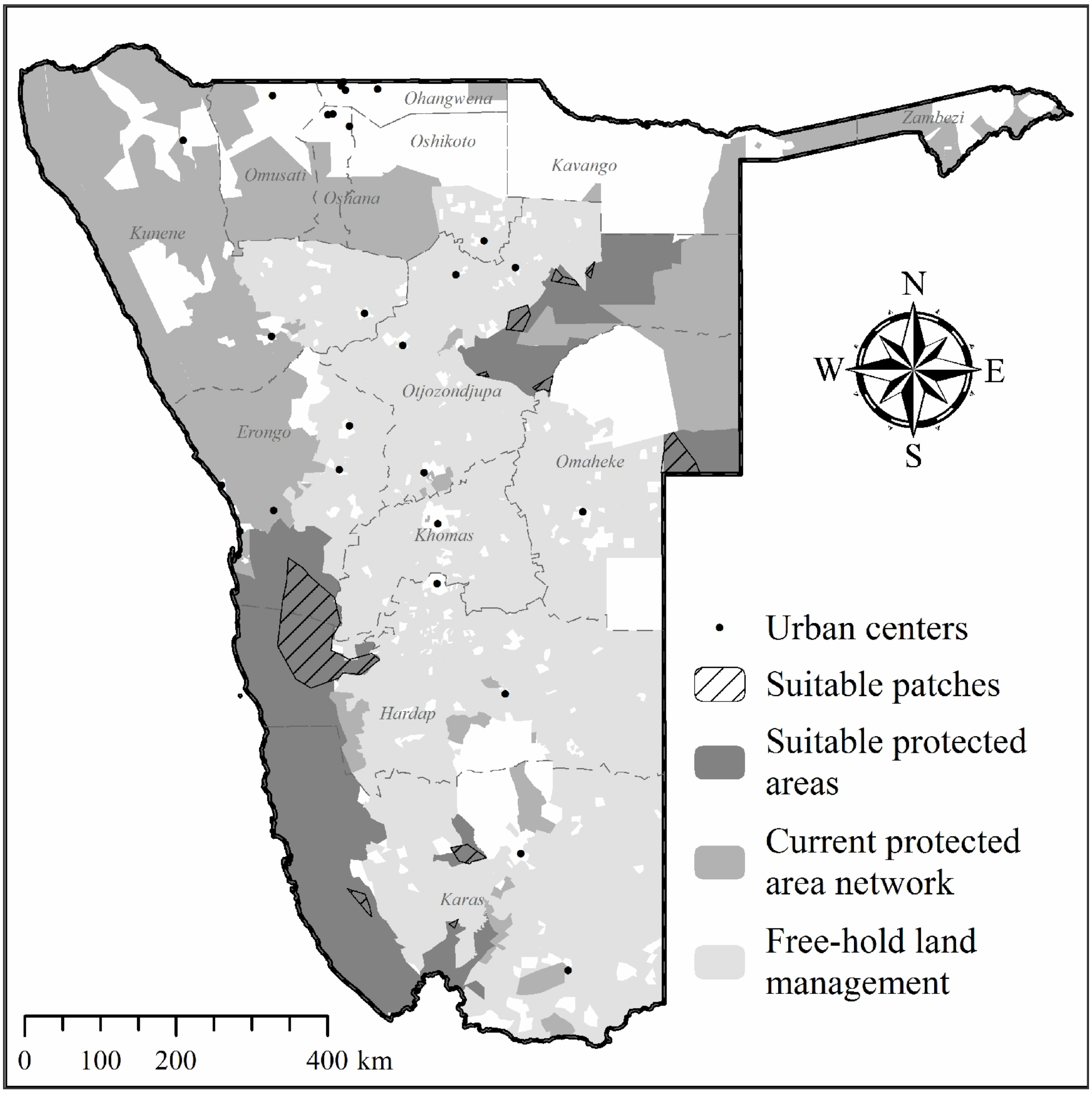


Figure 5.11 – Potential cheetah recipient areas in Namibia without site fidelity considerations. *The map shows suitable protected area patches with low-medium density of conspecifics, >50 km away from urban centers, and no or low occurrence of lion and spotted hyaena (following Stein et al., 2012). Suitable patches are shown in relation to free-hold land tenure where likelihood of conflict and persecution is high. Output calculated with CaTSuiT (Lemeris Jr., 2015). Site-specific detail provided in Appendix 23.*

Using the squared distance from release site to home range centroid as a crude estimate of a cheetah's spatial requirement to settle down (for those individuals that did – Table 5.4), the median area requirement was 2,672.9 km^2 (range = 182.3 – 21,638.4 km^2, $n = 7$). This circular metric accounts for unpredictable post-release directions (Fig. 5.8), but the median estimate exceeds the size of nine potentially suitable patches (Appendix 23). Even the largest available patch (10,785.3 km^2 in western Namibia – Fig. 5.11) was <50% of the area required for Aju17 (21,638.4 km^2). In addition, several translocated cheetahs have already moved into the largest patch, and following successful reproduction on nearby reserves (females Aju02/29/58 – Table 5.3), translocations into the pro-Namib ecosystem were stopped in 2012 because an increasing frequency of cheetah observations suggests recovery of the local population (Q. Hartung, N. Odendaal, S. Bachran, pers. comm. 2014).

5.4 Discussion

Contrary to the results for leopards (chapter 4), the outcomes of cheetah translocations were strongly polarised. Cheetahs either died soon after release or the animals survived and integrated into novel environments with good chances of contributing to the free-ranging gene pool. The documented success rate of 40.0% (possibly ~50% considering three unknown cases) was much higher than for translocations carried out under very similar circumstances in Botswana (~18%, Boast et al., 2015). However, survivorship estimates were significantly lower than for strategic reintroductions in South Africa where annual survival of adults was 84.6% across five years with a median survival of 28 months (Marnewick et al., 2009). The survivorship results obtained here reflect those of free-ranging farmland cheetahs in north-central Namibia (cumulative annual survival of 9.4 months for males and 10.2 months for females, Marker et al., 2003d) suggesting that cheetahs translocated into uncontrolled environments experience similar pre- and post-translocation mortality, although for different reasons. In support of this, Marker et al. (2003d) showed that there was no significant difference in survivorship of cheetahs released at the capture site, those released within 100 km of the capture location, and those moved between 100 km and 600 km.

In keeping with other cheetah translocations (e.g. Hayward et al., 2007a; Marnewick et al., 2009), competition with larger predators (here spotted hyaena) had a marked effect on survivorship and local lion and spotted hyaena densities probably are key predictors of cheetah translocation success (Purchase et al., 2006). Nevertheless, human-induced persecution (directly or indirectly) had the highest impact on cheetah survival after animals left designated recipient areas (also in Phiri, 1996; Houser et al., 2011; Boast et al., 2015). Fully habituated cheetahs were particularly unsuccessful, with four subjects shot within 6.5 months, rendering re-wildling of captive reared cheetahs an unsuccessful strategy (also observed by Pettifer, 1981) and one that incurs particularly high individual costs (~$6,000 per individual per year, Houser et al., 2011). Because degree of habituation was directly attributable to time spent in captivity, these findings strongly contradict others

(Marker et al., 2003d) who hypothesised that captive time does not influence chances of survival. Cheetahs easily become accustomed to human presence when exposed to regular contact (Marnewick et al., 2009). Specific least interaction protocols (Houser et al., 2011) may need to be adopted to prevent detrimental impact from habituation during prolonged captivity. Furthermore, long captive periods can adversely influence an individual's chances of survival through compromised avoidance of competing predators as well as impoverished skills to source food, seek appropriate shelter, interact with wild conspecifics or move and orient in a complex novel environment (Kleiman, 1989; Jule et al., 2008; Hunter and Rabinowitz, 2009). In the case of cheetahs, the significant influence of humans and competing large carnivores on post-release survival may only be mitigated effectively if individuals can be translocated into situations that provide control over movements (hence also human persecution) and intra-guild competition (by artificial control of species overlap), thus improving prospects of survival considerably (Marnewick et al., 2009).

Other problems commonly associated with large carnivore translocations such as homing and post-release conflict (Linnell et al., 1997) played less important roles in determining success. Long-distance translocations (>137 km), coupled with variable intermediate captivity, proved effective in preventing homing of cheetahs (*cf.* Marker, 2002), irrespective of the release mode employed. Contrary to studies elsewhere (e.g. Pettifer et al., 1982; Hunter, 1998a, 1998b), soft release had no beneficial effect on reducing post-release explorations, even after acclimatisation of up to 9.5 months. All individuals exhibited extensive movements beyond the boundaries of recipient areas, resulting in low site fidelity and putting cheetahs at risk of persecution, whether involved in conflict or not (consistent with Boast et al., 2015). Extensive explorations suggest that cheetah translocations into unconfined areas, or inadequately fenced reserves (du Preez, 1970; Houser et al., 2011), incur a considerable risk of individuals re-entering commercial farmlands. Again, this may only be prevented effectively by predator-proof fencing (Hayward et al., 2007a, 2007b; Marnewick et al., 2009) or through pre-release fence training such as exposure of cheetahs to electrified wires during acclimatisation (Hunter, 1998b).

As regards extreme dispersals, there may be any number of explanatory or contributing factors. For example, large-scale explorations during the initial orientation phase may represent an individual's attempt to home (Hunter, 1998b) or to familiarise with resources in novel environments. In Namibia's arid South and South-West, important resources such as permanent water and vegetation cover are more widely distributed and less frequent than on bush-encroached central or north-central farmlands (Mendelsohn et al., 2002) where study cheetahs were trapped. In southern Africa, cheetahs generally utilise and thrive in areas with mixed vegetation cover that provides edges between different sub-habitats, rather than in open environments (e.g. Purchase and du Toit, 2000; Mills et al., 2004; Muntifering et al., 2006; Marker et al., 2008a; Welch et al., 2015). These eco-tones and mixed habitat structures have been linked with reduced kleptoparasitism, flexible hunting strategies, and security (e.g. Hunter, 1998a; Broomhall et al., 2003; Mills et al., 2004; Bissett and Bernard, 2007; Rostro-García et al., 2015; Welch et al., 2015), thus providing effective competition refuges that enable coexistence with other large carnivores (Durant, 1998a; Purchase et al., 2006). Vegetation cover, e.g. in the form of thickets, is particularly important for females to increase cub survival (Durant, 1998a, 1998b) and females typically show an increased use of denser habitats (Durant, 2000; Broomhall et al., 2003; Mills et al., 2004; Bissett and Bernard, 2007; Rostro-García et al., 2015; Welch et al., 2015). Hence, after release into open desert and semi-desert areas, cheetahs may search for and seek out areas with resource compositions providing maximum security, or those reflecting familiar conditions at original capture sites or provided during intermediate captivity, i.e. mixed and dense woodland habitats.

Vast erratic post-release movements may also be a result of reactive avoidance behaviour following encounters with spotted hyaenas (Broekhuis et al., 2013). Moreover, in desert environments like the Namib, preferred cheetah prey such as springbok and oryx migrate according to rainfall patterns (Logan, 1960; Skinner and Smithers, 1990) and the cheetah's strong dependence on migratory prey has been associated with large ranging areas (Durant et al., 1988; Caro, 1994; Broomhall, 2001). In some cases, post-release explorations may simply reflect the ecological status of individuals. Long-term research has demonstrated that cheetahs are not strictly territorial throughout their lives (Eaton, 1974; Caro 1994; J. Melzheimer, pers. comm. 2015). In Namibia, young adult males frequently disperse for >100 km from maternal ranges when reaching independence (Morsbach, 1986a). As prime adults, males may not establish permanent ranges with fixed territories (especially lone males) and move across large, flexible home range areas >500 km^2 (Morsbach, 1986a; Marker et al., 2008a), occasionally occupying up to ~2,800 km^2 within one year (Wachter et al., 2006a). Male coalitions, on the other hand, fiercely defend small territories within their ranges to monopolise access to reproductive females (Serengeti cheetahs, Caro and Laurenson, 1989). Females in Namibia show little tendency of exclusive territoriality and tolerate high overlap between large home ranges that can measure up to ~1,700 km^2 (Morsbach, 1986a; Marker et al., 2008a). Therefore, most cheetahs naturally lead non-territorial lifestyles and move nomadically or seminomadically for much of the time (Caro, 1994; J. Melzheimer, pers. comm. 2015), using scent markings to enable an intricate system of spatio-temporal separation that reduces intra-specific competition and aggression (Eaton, 1970a). Consequently, at least for certain individuals (e.g. male floaters, Caro, 1994), post-translocation settling behaviour is unlikely. The exploratory movements of females Aju02/29/56/58 only appeared to stop when they settled to deliver litters. In this study, extensive explorations were not site-, habitat-, or sex-specific and may therefore also indicate an inherent response to the process of translocation. Under free-range conditions, the magnitudes and complexity of factors influencing cheetah spatial ecology would be very difficult to predict and/or control, resulting in a high degree of uncertainty as regards the outcomes of translocations.

Additional complications of cheetah translocations arise from concerns over fitness-related disadvantages of translocated individuals due to local variations in pathogen exposure and associated immune-responses and adaptations in Namibian cheetahs (Thalwitzer et al., 2010; Castro-Prieto et al., 2012). Furthermore, translocation involves considerable stress that may exacerbate existing pathologies or induce chronic stress (Teixeira et al., 2007; Dickens et al., 2010). The post-release apathy and sudden death of females Aju07 and Aju18 respectively provide reason for concern over potential stress impacts. For example, stress monitoring in captive cheetahs showed that most subjects exhibited significantly elevated stress levels, and sometimes for prolonged periods, following intensive handling, transfers between captive facilities, or when introduced to unfamiliar conspecifics (Terio et al., 1999; Wells et al., 2004). Hence, it appears possible that translocation-induced stress intensified a previously undiagnosed kidney condition in female Aju07 that resulted in her subsequent death from acute renal failure after re-capture. Despite a detailed post-mortem that was carried out on female Aju18 and which ruled out injury, snake bite, poisoning and other potential causes of mortality, there were no obvious signs explaining her death, suggesting a combination of stress and exhaustion as a possible cause. The female had found water, hunted successfully and moved until the morning of her death. Although this was not an evident problem in this study, the translocation of males into extant range areas could potentially result in increased mortality and competitive exclusion resulting from intra-specific aggression with established resident males (Eaton, 1974; Caro and Collins, 1987; J. Melzheimer, pers. comm. 2015). Contrary to other large felids like lion and leopard (Bertram, 1975; Packer and Pusey, 1983; Balme and Hunter, 2013), however, artificially induced infanticide from translocation events does not constitute a likely problem in cheetahs (Hunter and Skinner, 2003).

Despite apparent failures in translocating cheetahs, several females successfully raised cubs released with them, attesting to their vigilance and adaptation to potential threats (Caro, 1987; Durant, 2000). Moreover, surviving females produced new litters and, although it is unknown how many cubs reached independence, these events compensated for initial losses. A resident cheetah population established in the pro-Namib desert transition zone (Q. Hartung, N. Odendaal, pers. comm. 2014) where most animals were released. Increased carcass availability from cheetah kills probably also benefited the local recovery of lappet-faced (*Torgos tracheliotos*) and white-backed vultures (*Gyps africanus*) at two different sites (pers. obs.). From the limited data available, it appears that translocated cheetahs mainly hunted medium-sized prey, including sub-adults of large ungulates. Therefore, detected prey items were in strong agreement with the food selection of Namibian conspecifics (e.g. Morsbach, 1986a; Marker et al., 2003b), that of cheetahs released into similar environmental conditions (i.e. arid ecosystem with sparse vegetation cover) in South Africa (Welch et al., 2015), and prey preferences of the species in general (Hayward et al., 2006c; Hayward, 2009). The high contribution of springbok to overall prey appears to be a result of the high abundance of this desert-adapted species in southern Namibia (East and the IUCN/SSC Antelope Specialist Group, 1999) in relation to other ungulates. Within their preferred prey range, cheetahs hunt according to the relative local availability of different species, preferentially selecting for the most common (Hayward, 2009). The fact that only few cheetahs were involved in livestock predation shows that cheetahs do not preferentially select for domestic animals where wild ungulate prey is available (Morsbach, 1986a; Marker et al., 2003b; Voigt et al., 2014). It further confirms that conflict is not a default outcome of carnivore translocations (chapter 4). Released cheetahs also supported local tourism efforts on recipient reserves as well as on farms where they settled. Managers received regular position updates that enabled direct observations by guests, thus attaching tangible value to cheetahs that are highly sought-after tourism attractions (Lindsey et al., 2007; Maciejewski and Kerley, 2014).

Similar to experiences in Botswana (Boast et al., 2015), however, the results from this study corroborate that translocations rarely resolve conflict on source properties permanently (also see chapter 4 for leopards) resulting in repeat requests for live removals. Hence, continued translocations may support removal of cheetahs from their indigenous areas, contradicting the general objective to maintain a viable free-ranging population (Nowell, 1996). In South Africa, the potential population sink effect of sustained translocations from conflict farms into protected reserves was one of the main reasons to cease a successful relocation programme (K. Marnewick, pers. comm. 2015). There are currently only a few fenced reserves in Namibia (Jones, 2014) that could potentially accommodate cheetahs and, if so, only at limited carrying capacities (Hayward et al., 2007b, 2007c; Lindsey et al., 2011). Furthermore, live removals from conflict farms have already resulted in a substantial increase in Namibia's captive cheetah population (Fig. 5.12).

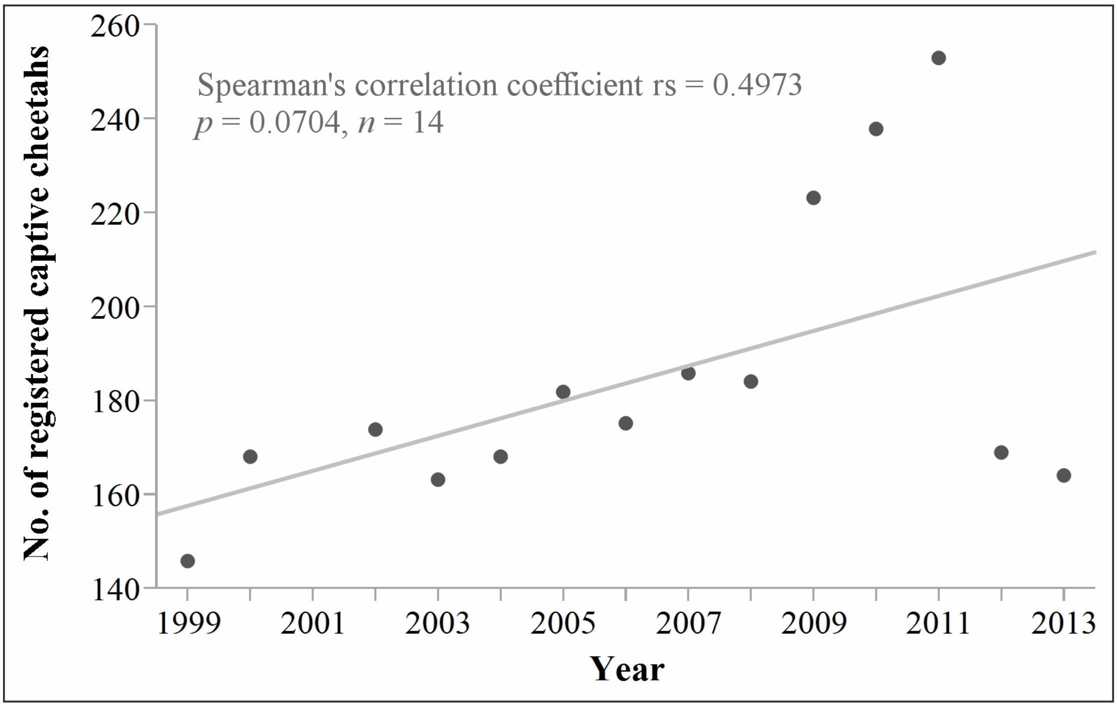


Figure 5.12 - Namibia's registered captive cheetah population, 1999 - 2013. *Trend line shows robust fit. The sudden drop in 2012/2013 likely is a function of reduced reporting rather than a decrease in actual captive numbers. Data source: International Cheetah Studbook (1999 – 2013); Cheetah Conservation Fund, Otjiwarongo, Namibia.*

Some carnivore programmes stopped rescuing cheetahs due to the limited capacity of fenced reserves to accommodate a persistent influx of perceived problem animals (T. Hoth, pers. comm. 2014). Namibia's wildlife authorities also stopped the live export of perceived conflict cheetahs, a policy used to supplement reintroduction efforts elsewhere in southern Africa in the past (e.g. Phiri, 1996; Hunter, 1998b; Marker et al., 2003a). At first glance, Namibia provides seemingly favourable conditions for free-range cheetah translocations. These include a PA network covering >35% of the country, suitable prey populations within and outside of PAs (e.g. East and the IUCN/SSC Antelope Specialist Group, 1999; Erb, 2004; Lindsey et al., 2013b), and a very low human population density of <1.0 people per km^2 in most areas (Mendelsohn, 2006). However, the modelling of recipient areas shows that the capacity for continued translocations into unconfined environments is very limited. In light of trapping rates of several hundred cheetahs per year in the country (chapter 3; Morsbach, 1986a; Marker et al., 2007) the potential demand for translocations clearly outweighs the available recipient habitat. Therefore, the results from this study support other assessments that translocation should not be a standard strategy to manage perceived conflict cheetahs (Marnewick et al., 2009; Fontúrbel and Simonetti, 2011; Boast et al. 2015).

Cheetah translocations probably are most cost- and conservation-effective where they can contribute to structured reintroductions of the species (Hayward et al., 2007a, 2007b; Purchase et al., 2006; Marnewick et al., 2009) and where there is long-term commitment from a variety of stakeholders to ensure their success (Lindsey et al., 2009b). However, such efforts require large fenced areas and local prey supplementation (Lindsey et al., 2011). The successful establishment of isolated populations on fenced reserves also necessitates management through a meta-population approach (Lindsey et al., 2009b; Marnewick et al., 2009). Namibia supports the largest free-ranging population of cheetahs in any country (Marker, 2005; Purchase et al., 2007) and one that appears to be expanding into historic range areas (*cf.* Hanssen and Stander, 2004; Stein et al., 2012). Given the diverse constraints identified in this study, translocations can successfully conserve a small number of cheetahs, with add-on effects of boosting low-density populations where land-use has changed and supports the species' presence. The actual conservation task that remains, however, is to ensure the perseverance of the entire free-ranging population on free-hold farms. There is great potential to increase cooperation with Namibian livestock and game farmers through concepts such as rapid conflict response coupled with participatory monitoring to increase tolerance of resident cheetahs (see Bangs et al., 2006 for wolves; Wachter et al., 2006a).

Translocation of cheetahs should be reserved as a last-resort, supplementary management tool when other conservation options have been exhausted. It should predominantly be used for those populations facing imminent local extinction, such as the critically endangered Asiatic cheetah (Durant et al., 2008). Where translocation is necessary, rigorous candidate and recipient area selection can improve its efficacy. Finally, there is increasing evidence that the strategy does not curb the motivation of land managers to remove cheetahs repeatedly (also see Boast et al., 2015), thus fuelling a psychology that is counter-productive to improving coexistence and maintenance of free-ranging cheetah gene pools across their current distribution.

5.5 Key points

1. The outcomes of cheetah translocation were highly variable, corresponding with eight early deaths (<90 days) and animals successfully settling into novel environments and reproducing.
2. The overall success rate of 40.0% was mostly influenced by an individual's ability to survive year one, and much less by post-release conflict (*n* = 2) or successful homing which was prevented by long distance translocations.
3. Human persecution and spotted hyaena-induced mortality had marked impact on cheetah survival. Time in captivity directly influenced the degree of habituation to humans and fully habituated subjects were least successful (five early deaths).
4. All cheetahs displayed extensive post-release explorations, resulting in low site fidelity and leading individuals beyond protected area boundaries - soft release and recipient area size did not improve site fidelity.
5. Due to consistent post-release dispersal, the outcomes of cheetah translocations into unconfined environments are highly unpredictable and most individuals are exposed to pre-translocation threats after leaving PAs.
6. When considering cheetah occurrence, intra-guild competition, urban areas, site fidelity, and land-use as key determinants for translocation success, there are currently no PAs in Namibia that could ascertain post-release safety.
7. Cheetah translocations can locally boost low-density populations but have little potential to address widespread conflicts with private land managers.

Chapter 6 Brown hyaena (*Parahyaena brunnea* – Thunberg, 1820)

Disclaimer: Except for individuals Hbr55 and Hbr75, brown hyaena monitoring data from Namibia were collected and generously provided by Dr Ingrid Wiesel of the Brown Hyena Research Project, Namibia.

6.1 Introduction

The brown hyaena is endemic to southern Africa (Wiesel, 2015a) where much of its current distribution falls outside PAs (Fig. 6.1; Thorn et al., 2011; Kent and Hill, 2013; Lindsey et al., 2013a), resulting in perceived and actual conflict with livestock producers (chapter 3; Skinner, 1976; Stein et al., 2010; Thorn et al., 2012, van As, 2012). Although predominantly a scavenger (Mills and Mills, 1978; Owens and Owens, 1978; Mills, 1984b), and with little supporting evidence from feeding ecology studies on commercial farmlands (e.g. Maude and Mills, 2005; Stein et al., 2013), brown hyaenas are often accused of livestock predation. For example, in South Africa's North West Province (Thorn et al., 2012), and in Botswana (Schiess-Meier et al., 2007), the species was blamed for about 12% of all livestock losses. Similarly, in Namibia, different studies recorded low conflict levels in comparison with other carnivores (16% in chapter 3; Stein et al., 2010) but approximately 72% of livestock ranchers believe that the species poses a threat to their animals (Lindsey et al., 2013a).

It is widely agreed that extensive unprotected pastoralist systems are an important mainstay habitat for the brown hyaena (Thorn et al., 2011; Kent and Hill, 2013; Lindsey et al., 2013a; Stein et al., 2013; Winterbach et al., 2014). In Namibia, the species is still found in many PAs but its widespread historic distribution throughout the entire country (see Mills, 1982a) has been reduced to include only a partial occupancy of the commercial livestock farming areas (Fig. 6.1; Wiesel, 2015b). The cause of this decline is poorly understood, but direct persecution probably contributed to local extirpation on Namibia's south-eastern smallstock farms (Wiesel, 2015a). The species is also accidentally affected by control methods aimed at other carnivores and continued persecution (e.g. through non-selective removal, poisoning, trapping, shooting) presents a serious concern throughout the brown hyaena's whole range (Smithers, 1983; Owens and Owens, 1984; Wiesel, 2015a).

Contrary to cheetahs and leopards, the brown hyaena is a generalist feeder, consuming a wide variety of available food items that mainly include carrion, invertebrates, fruits, and eggs (Mills and Mills, 1978; Owens and Owens, 1978; Mills, 1982a) which are collected during long nocturnal foraging bouts (average of 31 km travelled, Mills, 1984b).

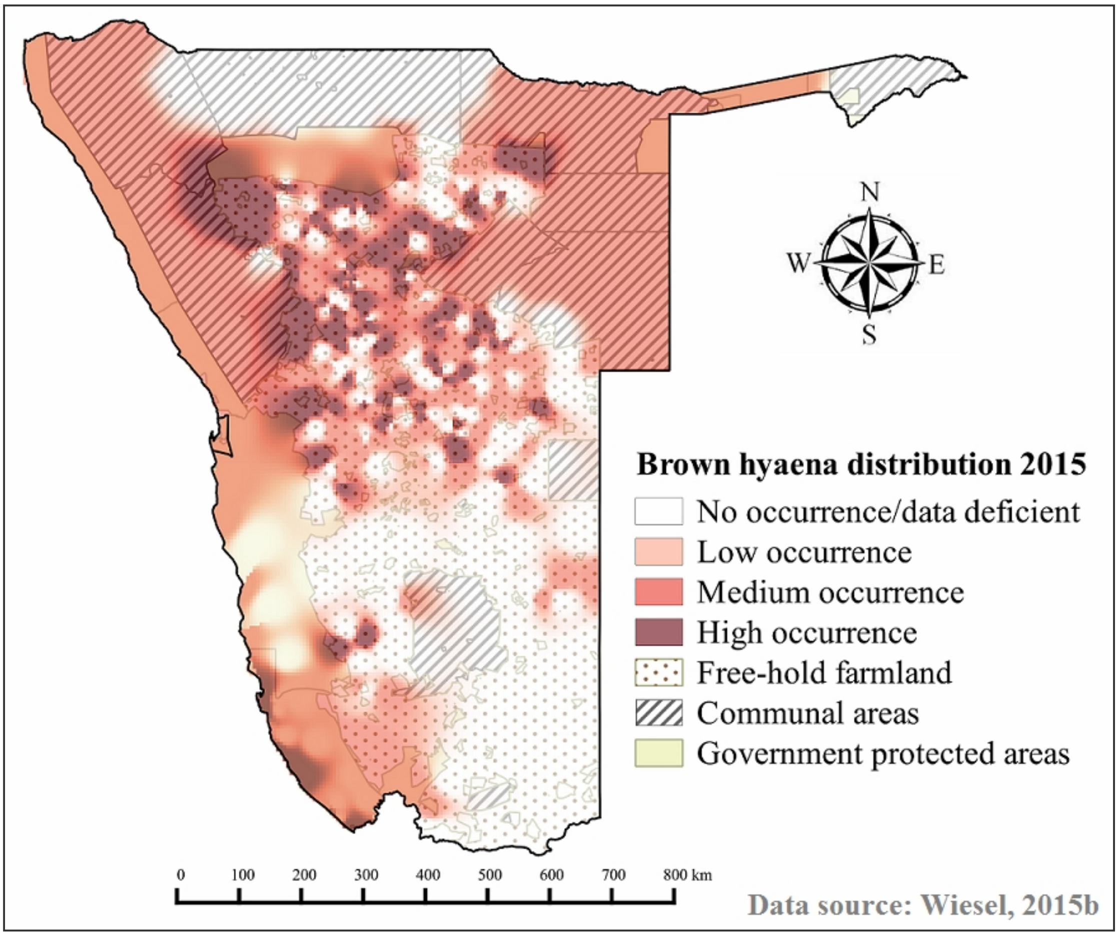


Figure 6.1 – Brown hyaena distribution across Namibia's main land tenures.

Brown hyaenas occasionally hunt small mammals and ground-living birds, but inefficiently so (Mills, 1978, Owens and Owens, 1978). Attempts to kill prey typically involve only short chases and appear unsophisticated (Mills and Mills, 1978; Owens and Owens, 1978; Mills, 1984b). In the southern Kalahari, merely 4.7% of 128 observed hunts were successful and vertebrate prey comprised only 4.2% of the brown hyaena's overall diet (Mills, 1990).

Brown hyaenas are not well suited to hunting livestock and are often wrongfully accused of raiding (Skinner, 1976) when feeding on carcasses killed by sympatric carnivores (Maude, 2005; Stein et al., 2013) or on livestock that died from other causes (Maude and Mills, 2005). Nevertheless, in areas with largely immobile prey such as Cape fur seal (*Arctocephalus pusillus*) pups, the species' predatory capacity has been documented in detail (e.g. Kuhn et al., 2008; Wiesel, 2010). Here, live prey constitutes a higher proportion of food intake (9.6% of consumed seal pups were killed) because it is easy to catch and available throughout the year (Wiesel, 2010). There also is some evidence of attacks on confined smallstock (Skinner, 1976) and cattle calves (Smithers, 1966; Skinner, 1976; Mills, 1982b). A recent study in South Africa reported similar conflict involvement of brown hyaenas and leopards with evidence of at least 10 lethal attacks on cattle calves by hyaenas over a 5-year period (van As, 2012). Such incidents are attributed to individual brown hyaenas who may develop a skill for and habit of raiding, and which can cause substantial losses that, in extreme cases, may exceed 100 livestock in only four months (Skinner, 1976).

According to the land managers interviewed as part of the present study (chapter 3) attacks on livestock appear pronounced during peak calving/lambing seasons and invariably involve juvenile prey. In addition to kills, farmers report severe brown hyaena inflicted bite injuries to new-born lambs, calves and foals during the first weeks after births (also see Mills, 1982b) when offspring are occasionally left unattended by mother animals and have not yet developed the ability to flee from opportunistic hunting attempts (Mills, 1984b). Non-lethal bite injuries frequently become infected and require lengthy and costly wound treatments that often are unsuccessful (pers. obs.). In conjunction with the brown hyaena's low hunting success (Mills, 1978, 1990) this has brought about the reputation of a "*clumsy and messy killer*" (Skinner, 1976:266).

Consequently, brown hyaenas are trapped purposefully or indiscriminately for lethal control or live translocation (chapter 3). The removal of habitual raiders can mitigate localised conflict effectively (Skinner, 1976). With regard to live translocation, however, more detailed information is needed to formulate suitable protocols because the controversy around conflict-translocations (Linnell et al., 1997; Massei et al., 2010; Fontúrbel and Simonetti, 2011) is further complicated when individual hyaenas are moved solitarily without matrilineal kin support (Mills and Hofer, 1998). For instance, brown hyaenas forage solitarily (Mills, 1978, 1982a, Owens and Owens, 1978) but females otherwise live in well-integrated clans which maintain strong social ties that are essential for cooperative litter rearing and protection (Owens and Owens, 1978, 1979a, 1979b, 1996; Mills, 1982a). Moreover, cubs may wean as late as 14-15 months old and thereafter may continue to be partially dependent on communal food provisions at the clan's den site (Skinner, 1976; Owens and Owens, 1978, 1979a, 1996). Referring to the larger spotted hyaena, Mills and Hofer (1998) further express concern over intra-specific competition and aggression resulting from hyaena translocations into extant range areas with established conspecifics. Following translocation into a new enclosure, a captive male spotted hyaena showed significantly elevated stress levels (Goymann et al., 1999). Social tensions and physical combat in a captive group produced similar results (Goymann et al., 1999). In South Africa's Madikwe Game Reserve, resident spotted hyaenas killed an introduced clan (Hofmeyr et al., 2003, cited in Hayward et al., 2007b).

Whilst detailed translocation evaluations are increasingly becoming available for other social African predators such as lion (e.g. Stander, 1990a; Trinkel et al., 2008) and African wild dog (Somers et al., 2008; Gusset et al., 2010), very little is understood about hyaena translocations (Mills and Hofer, 1998). In South Africa, range use data were recorded for only three of 30 translocated brown hyaenas (Skinner and van Aarde, 1987). Summarised results have been published for reintroductions into fenced tourism reserves for a total of 36 animals that were not intensively monitored as individuals (Hayward et al., 2007a, 2007b), followed by a preliminary assessment of feeding ecology (Slater and Muller, 2014). These South African experiences suggest that brown hyaenas are capable of adjusting to unfamiliar environments devoid of conspecifics. The animals established themselves for prolonged periods without causing conflict (despite visiting human habitation, Skinner and van Aarde, 1987) and reproduced successfully (Hayward et al., 2007a, 2007b), thus fulfilling some of the criteria regarded as important to achieve translocation success (Linnell et al., 1997; Massei et al., 2010).

Prior to the present case study, however, there have been no attempts to assess brown hyaena translocations in a context of conflict mitigation. With an estimated worldwide population of 4,365 – 10,111 mature individuals and considering a declining population trend for adults (Wiesel, 2015a) informed management protocols will be useful in the future conservation of brown hyaenas, regardless of the circumstances under which individuals are trapped.

This chapter aims to evaluate the outcomes of a conflict-related translocation involving a single sub-adult female in central Namibia. It represents the first GPS-monitored event for the species. The results are assessed with regard to species ecology, sociality in particular, and the data available from previous brown hyaena translocations. Furthermore, the results are compared with information from resident conspecifics near the release site and other locations in Namibia.

6.2 Methods (supplemental to chapter 2)

6.2.1 Translocated female Hbr55

The translocated female hyaena (Hbr55) was captured in a baited live trap on a commercial livestock farm in central Namibia in January 2012, following four incidents of bite injuries inflicted on new-born cattle calves that were less than seven days old at the time. In this case, the examination of bite wounds provided evidence (i.e. sub-cutaneous haemorrhaging on hind quarters and hind legs coupled with only mild skin perforations and signs of tearing, dragging and pulling) that these were indeed atypical of the neck bites usually inflicted by other indigenous large predators such as cheetah and leopard (Kruuk and Turner, 1967; Eaton, 1970b), and hence most likely inflicted by a brown hyaena. Bite marks and injuries were consistent across all attacked calves and exceeded the dimensions that would potentially be caused by smaller predators such as black-backed jackal or caracal (*Caracal caracal*). Confusion with spotted hyaena predation was ruled out because the species is no longer present in central Namibia (Stein et al., 2012; Lindsey et al., 2013a). Attacks on calves were restricted to a small, fenced cattle night enclosure, and such confinement potentially encourages livestock attacks by brown hyaenas (Skinner, 2006; Kuhn et al., 2008). In the corral fresh hyaena spoor and fence burrowing activities had been observed, without obvious signs of other predators. The hyaena was trapped a few hours after the last bite incident. The affected landowner requested translocation of the animal and explicitly refrained from lethal control. To determine whether Hbr55 was responsible for previous calf attacks, the conflict area was monitored intensively post-translocation.

Hbr55 was translocated a linear distance of 63 km and released onto a private wildlife reserve (with game-proof but not predator-proof fencing) within 48 hrs of capture. This area supported resident brown hyaenas at estimated low to medium density (Hanssen and Stander, 2004; Stein et al., 2012). Before release, the hyaena was immobilised with a combination of Ketamine (75.0 mg) and Medetomidine (1.5 mg). The animal was measured, examined, fitted with a GPS iridium satellite transponder (Africa Wildlife Tracking, Pretoria, RSA) and biological samples were collected. The animal was approximately 12 months old (estimated by I. Wiesel assessing tooth eruption and wear), weighed 27 kg and measured 48 cm around the neck, 68 cm at the shoulder and 112 cm in length (naso-anal). The GPS unit was programmed to record and transmit three locations every 24 h, i.e. 19:00, 00:00 and 05:00 (local time) reflecting expected times of emerging, peak and declining activity (Owens and Owens, 1978). After recovery from anaesthesia (following an injection of 7.5 mg Atipamezole), Hbr55 was released without acclimatisation to the recipient area. Habitat conditions at source and recipient sites were generally consistent – both localities form part of the mountainous, semi-arid, bush-encroached highland savannah biome in Namibia’s Khomas Highlands. Hbr55’s location data were shared with private landowners in the release area on a daily basis (section 2.8 in chapter 2) in order to determine details of post-release conflict and ecology.

6.2.2 Resident male Hbr75

In October 2013, a sub-adult male brown hyaena (Hbr75) was captured as a non-target species in a live trap only 9.0 km from Hbr55's release location (Fig. 6.2). The male was also fitted with a GPS iridium satellite transponder (Africa Wildlife Tracking, Pretoria, RSA) and released at the capture site immediately. Hbr75 was approximately 24 months old, weighed 33 kg and measured 52 cm around the neck, 69 cm at the shoulder and 109 cm in length (naso-anal). Similar to Hbr55, the male's collar was programmed to record and relay at least three locations every night (19:00-20:00; 00:00-01:00 and 05:00-06:00). When his GPS unit failed less than two weeks into the monitoring, his movements were recorded opportunistically through standard VHF ground telemetry (section 2.7 in chapter 2). Hbr75's anaesthesia and data sharing protocols were consistent with those of Hbr55.

6.2.3 Resident brown hyaenas (coastal population)

To interpret Hbr55's post-release movements in more detail, results were also evaluated against data from resident Namib Desert brown hyaenas, including one adult male, two adult breeding females and two sub-adult females (Table 6.1, Fig. 6.3). These hyaenas belonged to different but adjoining coastal clans and were GPS-monitored (Tellus 2D collars, Followit, SE) between 2005 and 2011 as part of a long term ecological study (I. Wiesel, Brown Hyena Research Project). For these individuals, positional data were standardised with reference to Hbr55's recording schedule. Additional details on desert study animals are available from Wiesel (2006, 2010).

6.3 Results

Upon release, translocated female Hbr55 displayed large scale exploratory movements for 16 days (Fig. 6.2), roaming over an area of approximately 582.2 km^2 (100% MCP). During this time, the hyaena travelled a minimum total distance of 133 km (median = 10.1 km/day ± 2.0 km SE, range = 1.5 - 21.9 km/day, n = 11) and moved through 15 private farms managed for livestock and/or game production, trophy hunting as well as non-consumptive wildlife tourism. Hbr55 did not show any release site fidelity and left the reserve during the night following translocation, but did not return to her capture site. In comparison, the resident hyaena Hbr75 displayed no exploratory movements of such scale after its release (Fig. 6.2).

Following initial explorations, Hbr55 established a stable range which she maintained until her death four months later (see below). The range's geographic centroid was located 19 km North-West of the release site and 69 km from the original capture

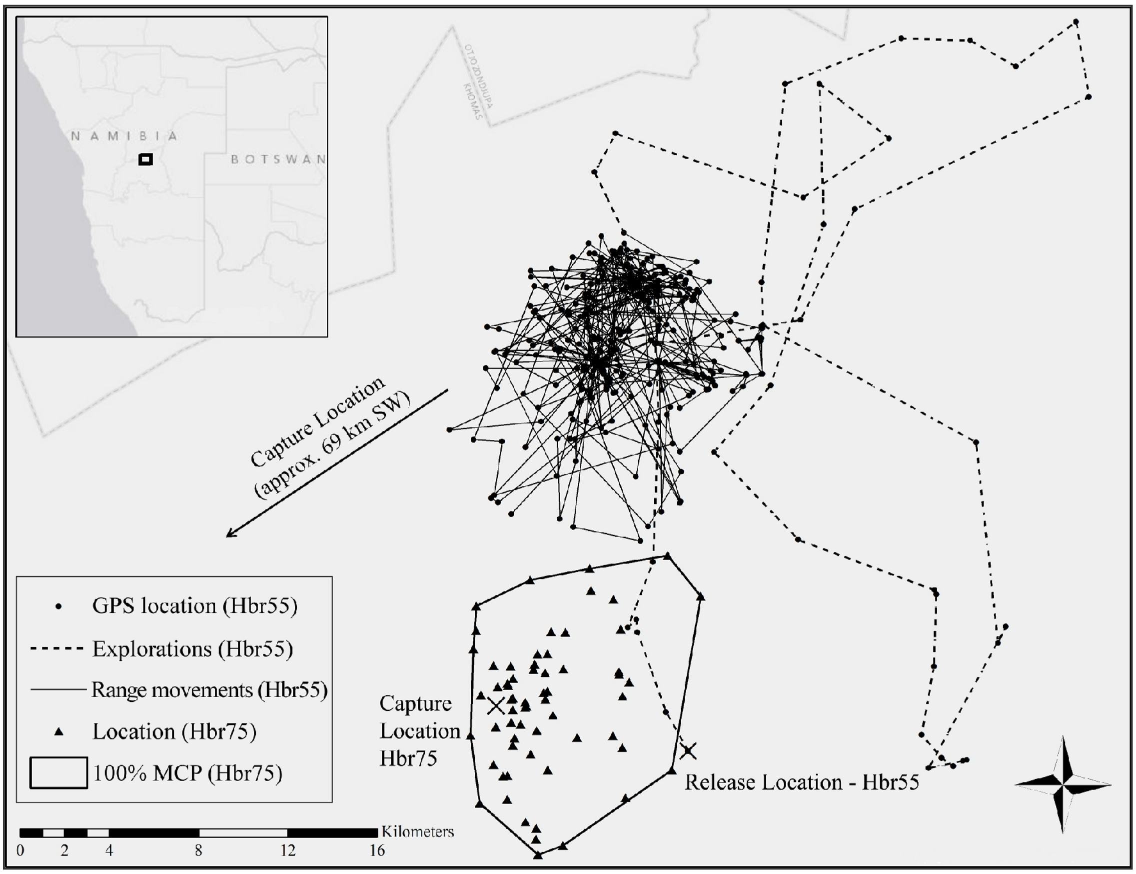


Figure 6.2 – Movements of two sub-adult brown hyaenas (translocated female Hbr55 and resident male Hbr75) in the same bio-geographic region of central Namibia. *The map shows similarities in the settled ranges of these subadults while Hbr55 displayed wide post-release exploratory movements that were not evident in Hbr75's movements.*

location (Fig. 6.2). Local habitat was mainly characterised by undulating mountain terrain, large areas of black-thorn (*Acacia mellifera*) encroached bush thickets, semi-open grassland glades, seasonal river courses and swamps, as well as large permanent water dams. Road density was higher in Hbr55's release area with 1.4 km/km^2 when compared with 0.9 km/km^2 in her capture area (recorded in 100 km^2 quadrates around trap and release sites respectively). Hbr55's range measured 133.5 km^2 (100% MCP) and daily movements decreased to a median of 8.3 km/day ± 0.4 km SE (range = 1.6 - 19.4 km/day *n* = 94) which were significantly different from those during the exploration phase (median = 10.1 km/day ± 2.0 km SE, range = 1.5 - 21.9 km/day, *n* = 11) (Mann-Whitney U Test: U = 688.0, *p* = 0.037). In comparison with resident sub-adults in Namibia, her mean nocturnal movements were similar to those of inland male Hbr75 (U = 380.0, *p* = 0.067) but significantly differed from the coastal sub-adult female LHb32f (U = 2,360.0, *p* = 0.007) (Table 6.1).

Table 6.1 - Comparison of home ranges and movement details for translocated and resident brown hyaenas in South Africa and Namibia.

Individual – area	**Sex**	**Age class**	**Treatment**	**Locations Recorded**	**Monitoring duration (days)**	**100% MCP range size in km^2 (50% core area)**	**Mean movement per night ± SD in km^2**	**Source**
A – Transvaal, South Africa	M	Adult	Translocated	43	91	8.8	12.5 ± 4.9 (*n* = 81)	Skinner and van Aarde (1987)
B – Transvaal, South Africa	F	Adult	Translocated	52	107	5.5	pooled value for hyaenas: A, B and C	Skinner and van Aarde (1987)
C – Transvaal, South Africa	M	Adult	Translocated	187	461	48.9		Skinner and van Aarde (1987)
Hbr55 – inland, Namibia	F	Sub-adult	Translocated	364	137	133.5 (21.4)	8.6 ± 4.0 (*n* = 94)*	This study
Hbr75 – inland, Namibia	M	Sub-adult	Resident	66	191	103.2 (12.9)	10.4 ± 2.5 (*n* = 12)*	This study
LHb32f – coastal, Namibia	F	Sub-adult	Resident	225	78	217.9 (46.7)	11.1 ± 6.0 (*n* = 67)*	Brown Hyena Research Project

[a] Note: Night movements were calculated with three locations as opposed to pruned daily locations used for range calculations.

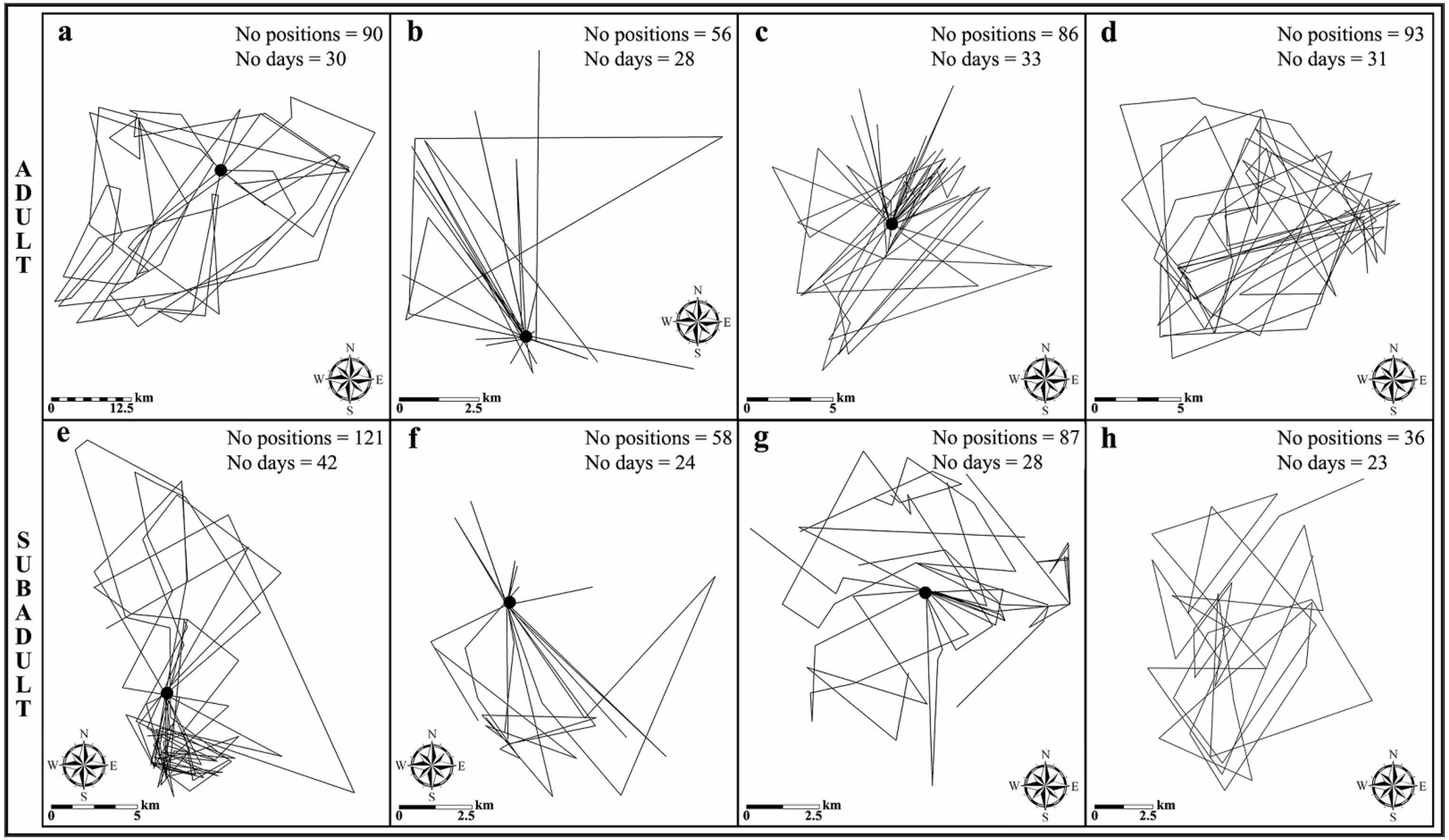


Figure 6.3 – Brown hyaena movements as influenced by denning activity. *Panels show oriented movements of resident individuals in the direction of their clan's known active den site for cub provisioning and social interaction (male Alfie (a), breeding females Obelixa (b) and Tosca (c), young females Helene (e) and Alaika (f)), non-oriented movements of Tosca outside the breeding period (d), oriented movements of translocated female Hbr55 to a suspected active den site (g) and non-oriented movements of male Hbr75 suggesting his clan was not denning (h). Den locations are indicated by black dots. Panels display selected movements standardised for consistent data and time periods. Repeat site visits apparent in panel d) represent confirmed resting sites as opposed to active dens (I. Wiesel, pers. comm. 2014).*

Hbr55 settled into an area in which landowners regularly observed resident brown hyaenas on wildlife cameras and where the species was known to reproduce successfully (C. Kruger, A. Wiese, pers. comm. 2012). Within the range, Hbr55 displayed distinct oriented movements towards one specific location for 28 days, closely resembling those of resident hyaenas during cub-provisioning and social interaction at active dens (Fig. 6.3). During the same period, Hbr55 was once observed in company of an adult brown hyaena and both animals fled in the same direction when approached (C. Kruger, pers. comm. 2012). Hbr55's movements hence suggest that she found and visited an active den site of the resident hyaenas and possibly became assimilated into the local clan. Den-oriented activity was not detected in Hbr75's location data (Fig. 6.3) but may have been obscured by the inconsistent sampling regime.

Hbr55's post-release range encompassed livestock and game production properties, trophy hunting and other tourism enterprises as well as a sand mining concession. Through regular consultation with these landowners and sharing of positional data it was established that the female caused no further conflict. In addition, no further bite incidents occurred at the original conflict site for at least 10 weeks after the hyaena was removed and calves were mobile enough to escape hyaena attacks (W. Teubner, pers. comm. 2012). No conflict was reported for sub-adult male Hbr75.

The translocated female died in a vehicle accident in her new range 137 days after release, whereas resident male Hbr75 was still alive when the study was concluded.

6.4 Discussion

Albeit limited to a single animal and in part anecdotal, the outcomes of Hbr55's case study provide first evidence that translocated brown hyaenas can, under certain circumstances, successfully adapt to novel free-range areas with established conspecifics. Despite crossing through several cattle ranches and living close to human habitation, no conflict occurred as a consequence of this translocation (also in Skinner and van Aarde, 1987), and there were no further reports of livestock attacks at the capture site in the 2.5 months following translocation. Therefore, this effort was successful in alleviating livestock attacks at the source location without moving conflict elsewhere, contradicting Skinner's (1976) conclusion that livestock-raiding brown hyaenas will not cease their habit.

Hbr55's extensive and immediate exploratory movements are consistent with those of relocated adult females (Skinner and van Aarde, 1987). It remains unclear whether these explorations reflect attempts to return to a familiar den or home range but the fact that there was no indication of oriented movements toward the original capture location, suggests not. Similar to observations from free-range brown hyaena translocations in South Africa (Skinner and van Aarde, 1987), Hbr55 left the release reserve and settled into a definable range in its vicinity. Hence, true site fidelity for the species may only be achieved through prolonged acclimatisation in pre-release pens and electric fencing of recipient areas (Hayward et al., 2007b).

The new range incorporated large tracts of mountainous savannah habitat, which appears to be a preferred environment by providing opportunities to shelter with little disturbance from humans (Skinner, 1976) and corroborates earlier observations that settling may be encouraged by this type of environment (Skinner and van Aarde, 1987). In comparison with translocated individuals in South Africa, range parameter differences probably were effects of non-standardised sampling regimes, i.e. consistent GPS telemetry data for Hbr55 as opposed to opportunistic VHF telemetry recordings by Skinner and van Aarde (1987). Hbr55's movements closely resembled those of sub-adult male Hbr75 (monitored in the same environment) suggesting that translocation-induced movement differences only manifested during the initial exploratory phase, but not thereafter. However, central Namibian home ranges were notably smaller than that of coastal sub-adult LHb32f. This difference could be a result of habitat-specific resource availability, food dispersion in particular, which locally influences the spatial ecology of brown hyaenas (Mills, 1981, Skinner et al., 1995). Namibia's coastal desert hyaenas feed on Cape fur seals throughout the year and increasingly so when new-born pups become available (Wiesel, 2010). To access spatially concentrated seal colonies as a reliable energy-rich food source, coastal hyaenas frequently travel variable distances (range = 0.7 - 14.6 km) depending on their den locations (Kuhn et al., 2008). Along the desert coast, local seal availability affects territory size (Skinner et al., 1995) and LHb32f's movements probably were influenced by this mechanism. Her range is broadly consistent with those previously reported for coastal hyaenas (range = 32 – 220 km^2, Skinner et al., 1995). Conversely, no such concentration of easily-obtainable, abundant food (that may strongly affect movements) occurs in central Namibia.

The movements of Hbr55 also strongly suggest successful assimilation into a resident brown hyaena clan. The female exhibited oriented movements towards a specific location in her new range, closely resembling those of known breeding females and clan members provisioning cubs at communal dens. Moreover, during this phase, Hbr55 was observed in the presence of an adult hyaena. Hbr55's premature age probably contributed to this success. Active dens are the social meeting spot for resident clans and group cohesion is facilitated through a greeting ceremony, physical contact and other interactions such as food provisioning for cubs (Owens and Owens, 1979a, 1979b; Mills, 1982a, 1982b, 1983). Sub-adult brown hyaenas display submissive behaviours during intra-specific encounters (Owens and Owens, 1978; Mills, 1981, 1983) and, with few exceptions, are tolerated by resident groups when they disperse from natal clans (Owens and Owens, 1978; Mills, 1983). Young males emigrate more frequently than do females (Owens and Owens, 1978). They become nomadic or integrate into another clan, and only non-related males will mate with reproductive females (Mills, 1984b, 1990). Sub-adult females more often remain with their natal clan (Owens and Owens, 1978; Mills, 1981) and there assist with communal cub rearing (Owens and Owens, 1979a, 1979b). However, some individuals leave the maternal range (Mills 1984b) and join other established groups (Owens and Owens, 1978). Aggression between unrelated brown hyaenas generally involves members of adjoining clans and much less often, dispersers or nomads (Mills, 1983). Aggression may result in neck-biting (Owens and Owens, 1978; Mills 1983) and the chasing of intruders has been observed (Owens and Owens, 1979b). Nevertheless, physical combat is uncommon (Mills, 1983) and clans are not isolated units and experience frequent emigration and immigration. Sub-adult females appear better suited for solitary translocation than adults who sometimes, though not always (Mills, 1981, 1983), maintain intricate social hierarchies (Owens and Owens, 1978, 1996), thus bearing the risk of compromising clan integrity, and in the case of lactating mothers the welfare of dependent cubs. In addition, translocation of adult females could potentially result in fierce intra-sexual aggression with resident females (Mills, 1983).

The importance of considering sociality during translocation of group-living large carnivores cannot be over-emphasised (Somers and Gusset, 2009). For example, to increase translocation success, the need for effective group integration has already been demonstrated for several social species (African wild dog, Gusset et al., 2006, 2010; lion, Trinkel et al., 2008). Due to the ad hoc nature of conflict translocation scenarios, however, managers cannot anticipate the age, sex or social status (all of which influence an individual’s translocation suitability) of candidate hyaenas. Because brown hyaenas typically forage solitarily (Owens and Owens, 1978; Mills, 1983) it appears almost impossible to trap cohesive family groups (in an ethical manner) for translocation purposes. Specifically for translocations into extant range areas, intensive studies of the recipient population’s social structure would be required to minimise risks of intra-specific aggression and disturbance.

Instead of haphazard solitary releases, state wildlife agencies may therefore consider admitting individual conflict hyaenas into appropriate captive facilities for artificial bonding and identification of adequate clan recipient sites. With regard to the latter, and contrary to other large carnivores (chapter 5; Hayward et al., 2007a), brown hyaenas potentially benefit from the presence of intra-guild competitors in release areas through regular access to carcasses (Owens and Owens, 1978; Mills, 1984b) and opportunities to steal kills (Skinner, 1976; Owens and Owens, 1978). In conjunction with the species' broad habitat and dietary tolerance (Mills, 1982a), this affords managers with high flexibility in terms of recipient area choice. Juvenile brown hyaenas can be raised successfully in captivity (Skinner, 1976). In South Africa, wild-caught adults and sub-adults of both sexes were successfully released into several reserves with high intra-guild competition following artificial group formation in pre-release pens (Hayward et al., 2007b).

The accelerating conversion of livestock farms into large wildlife-based enterprises in southern African rangelands (e.g. Lindsey et al., 2013a, 2013c) potentially offers suitable recipient habitat for the conservation of conflict hyaenas in areas of their former distribution. Thus, translocations should predominantly be designed to facilitate strategic population recovery in historic habitats. Based on the results of the present case study, supplementation of existing hyaena populations with individual sub-adults appears possible. However, the issue requires further detailed study before clear recommendations can be made. Under all circumstances, translocated hyaenas (except juveniles) should be equipped with GPS satellite transmitters to enable intensive monitoring and critical evaluation of translocations.

Translocation of perceived conflict hyaenas should not be a standard management strategy. It should be utilised as a last-resort intervention to prevent lethal control and be limited to the most suitable candidates. The brown hyaena's remarkable olfactory sense, which is used to detect food items over long distances (Owens and Owens, 1978; Mills, 1978, 1990), in conjunction with its dietary preference for carrion (Mills and Mills, 1978; Owens and Owens, 1978), predispose the species towards accidental and indiscriminate capture in baited trap cages. Baited traps are widely utilised by Namibian farmers to remove carnivores non-selectively (chapter 3; Marker-Kraus et al., 1996), thus potentially resulting in high annual numbers of non-conflict hyaenas to be moved. Farmland hyaenas also regularly feed on carcasses killed by other carnivores (Maude, 2005; Stein et al., 2013). In Botswana, brown hyaenas selected livestock carcasses as their primary food source but were not responsible for their death (Maude and Mills, 2005), further exacerbating the possibility of mistaken conflict involvement. Where conflict is doubtful or can be managed on-site, the immediate release of trapped hyaenas (especially adult females) should be promoted, for example through participatory research. In the present study, involving land managers in monitoring activities was beneficial and improved local perception of the species. Neither of the study hyaenas was persecuted and collaborative efforts to monitor hyaena ecology and conflict involvement followed.

Hbr55 died less than six months post-release, thus failing to achieve true translocation success. Although vehicle accidents are a common mortality cause for Namibian brown hyaenas (I. Wiesel, pers. comm. 2014; pers. obs.), and possibly a result of hyaenas feeding on road kill along motorways (IUCN Hyaena Specialist Group, 2015), it is possible that Hbr55's death was an effect of translocation because the female was moved into an area where road density was higher than in her capture area.

6.5 Key points

1. Despite failure to achieve true success in terms of survival and reproduction, the translocation of a female sub-adult brown hyaena mitigated livestock attacks at the source site and did not result in new conflict in the recipient area.
2. The hyaena did not return to its capture site and post-exploration movements closely resembled those of a resident sub-adult male in the same area. GPS data also suggest successful assimilation into a resident clan of conspecifics.
3. Whilst translocation should not be a preferred hyaena conflict management option, it appears to provide a feasible supplementary tool. Case-specific decisions need to consider the animal's age, sex, and social status.

Chapter 7 Translocation Costs

7.1 Introduction

Economic considerations play an increasingly important role in conservation management, and prioritization of limited financial resources for species protection is paramount (Shogren et al., 1999; Marsh et al., 2007; Carwadine et al., 2008; Martin-López et al., 2008; Wilson et al., 2011). Decision-makers depend on accurate costing of conservation activities to enable comparative evaluations of cost-efficiency (Ferraro and Pattanayak, 2006) as this can determine the most feasible approaches and areas for large carnivore conservation (e.g. Main et al., 1999; Lindsey et al., 2005a; Rondinini and Boitani, 2007; Barlow et al., 2010; Dickman et al., 2011; Packer et al., 2013; McManus et al., 2014). Cost-efficiency considerations during conservation planning also increase the prospects of receiving funds (Moran et al., 1997). Therefore, beyond ecological and conflict evaluations, one of the key questions pertaining to large carnivore translocations is whether the strategy provides an efficient use of conservation funding (Linnell et al., 1997).

The answer to this question has been confounded by a lack of cost reporting (Fontúrbel and Simonetti, 2011). In a broader animal relocation context, a review of published studies found that only 3% contained any cost data (Fischer and Lindenmayer, 2000). This is surprising considering that such interventions usually require substantial funding. For example, total costs for structured large carnivore reintroductions, involving translocation as a central mechanism, can be as high as hundreds of thousands (Lindsey et al., 2005a) or millions of USD (Bangs and Fritts, 1996).

Although only few attempts at reporting of translocation costs have been made for those studies involving large carnivores in free-range systems (Riley et al., 1994; Purchase, 1998; Boast et al., 2015), the strategy is generally regarded as an "*expensive*" tool (e.g. Miller et al., 1999:59; Massei et al., 2010:434; Fontúrbel and Simonetti, 2011:221). Without a formal assessment of costs, or cost-efficiency, one non-governmental organisation in Namibia terminated a rescue programme for conflict cheetahs, leopards, and hyaenas, in part because of the perception that financial demands were high (T. Hoth, pers. comm. 2014). From the little evidence that is available, it has been estimated that large felids are amongst the most expensive predators to translocate, with figures ranging up to $4,000 per animal (Fontúrbel and Simonetti, 2011). This is highly relevant considering that 71% of translocated conflict carnivores are large felids (Fontúrbel and Simonetti, 2011), all of the extant large felids are frequently involved in conflict situations (Inskip and Zimmermann, 2009; Linnell et al., 2012), and wildlife departments in different countries still allocate considerable time and resources to continued cheetah, leopard, and lion translocations from conflict areas (Athreya et al., 2007; M. Flyman, pers. comm. 2014; M. Reuben, pers. comm. 2014; B. Cain, pers. comm. 2015).

It also appears unclear to whom translocation costs should accrue but a variety of different public and private stakeholders are typically involved in the process and have an interest in its successful outcome (Massei et al., 2010). Public interest in, and support of, non-lethal carnivore management indisputably is growing, and the charisma of large carnivores in particular attracts substantial support from public funds. In Asia, for example, international donor expenditures towards wild tiger conservation were estimated in excess of $40 million in less than one decade (Linkie and Christie, 2007). At the same time, however, a steep increase in the number of non-governmental conservation initiatives around the world (Princen and Finger, 1994) intensifies competition for restricted conservation funds. Thus, funding agencies need to rely on transparent accounting and success evaluations to allocate thin budgets most effectively (Ferraro and Pattanayak, 2006).

The aim of this chapter is to help address the significant lack of empiric translocation cost information by providing detailed costings of the events described in chapters 4 – 6. Translocation costs are then linked to observed success rates to determine the Individual Conservation Cost (ICC) for the different species and translocation scenarios. This chapter also provides an analysis of the variables influencing cost results and translocation success. Finally, it compares translocation costs with those of other common non-lethal carnivore management strategies and determines which translocation costs can successfully be recuperated from public interest groups.

7.2 Methods

7.2.1 Cost recording

Translocation costs were recorded as true costs at the time when expenses occurred. All original costs were converted from ZAR into USD to enable international comparisons. Conversions were made on the 15th of every month during which costs occurred, with rates ranging from $1.00 - 9.86 ZAR to $1.00 - 14.67 ZAR during the study. All values in this chapter are reported as USD unless otherwise indicated. Any expenses contributing to the total cost were classified into one of the following distinct categories:

1) government permits;
2) tracking (monitoring technology such as VHF and GPS transmitters as well as GPS data retrieval for the first 12 months);

3) veterinary expenses (salary, immobilization, identification tags, disease screening, health assessments, biological samples);
4) transport (travel from source site to temporary captivity site and subsequently to recipient site including fuel and standard vehicle wear-and-tear rates);
5) captive holding (enclosure facilities);
6) captive feeding; and
7) staff salary.

Although substantial, the cost of follow-up field monitoring at recipient sites is excluded from this analysis because the intensity of this monitoring, and consequently its cost, is highly biased towards the specific scientific objectives of the translocating agency and thus not representative of other translocation operations. Moreover, most carnivores in this study were fitted with GPS transmitters (Table 7.1) and could therefore be monitored remotely (at least partly) without significant extra costs other than the purchase of tracking units.

Cost elements that were successfully reclaimed from external funding bodies were recorded and calculated as the overall proportions of total costs that could be recuperated from public support.

7.2.2 Individual Conservation Cost (ICC)

In order to provide a realistic estimate of translocation cost-efficiency, i.e. the successful translocation of a large carnivore according to criteria in chapter 2 (section 2.6), Individual Conservation Cost is defined as the cost of successfully releasing one individual in any cost category adjusted by the costs for unsuccessful translocations in the same category. ICCs were calculated by dividing the median cost for different translocation categories by their respective success rates, thereby ensuring that both successful and unsuccessful cases contributed to the results.

7.2.3 Literature review

To enable cost comparisons with the results from this study, an extensive literature survey was conducted to identify published materials that report the cost of large carnivore translocations as well as costs of other non-lethal conflict mitigation strategies. For this purpose, initial searches were conducted in *Google Scholar*, *ISI Web of Science/Knowledge*, *Science Direct*, *JStor* and *Wiley Online Library* databases. The complete list of search terms and their respective combinations is shown in Appendix 28. Subsequently, more focused searches were carried out in the online databases of the following publications which had delivered the majority of articles with useful results during initial searches: *Carnivore Damage Prevention News*, *Wildlife Society Bulletin*, *Journal of Wildlife Management*, *Oryx*, *Journal of Wildlife Research*, *Human Dimensions of Wildlife*, *Journal of Range Management*, *Vertebrate Pest Conference*, and *Wildlife Damage Management*.

Searches were neither constrained by year of publication nor by large carnivore species, but only English publications and those dealing with terrestrial mammalian predators were considered. In every search, the first 100 items were screened for discrete cost information. Additional articles were sourced opportunistically from colleagues and carnivore researchers.

For multiple studies dealing with the same, or geographically overlapping, cost samples (*cf.* wolf compensation, Mech, 1999; Treves et al., 2002), only the most recent estimate with the largest sample size was considered. Where values were reported in ranges due to variable cost modelling scenarios (e.g. impact cost of African wild dog, Woodroffe et al., 2005b), the median value of published costs was considered for analyses. Due to the high variability in conservation cost reporting, only studies with detailed costs at the individual level (e.g. per individual carnivore, or per individual guard animal) were considered. For livestock guarding dogs, independence of cost values was accepted for different breeds (e.g. Great Pyrenee, Akbash, and Maremma, Bergman et al., 1998).

If values were published in currencies other than USD, these were converted to USD by using the mean annual conversion rate of that currency to USD for the respective study period or, if study periods we not clearly stated, for the year of publication. Historic currency conversion rates were sourced from online databases of the The World Bank (2015) and The European Central Bank (2015).

7.3 Results

7.3.1 Sample characteristics

Cost and outcome data were recorded from 22 translocation events for a total of 30 animals - six leopards (chapter 4), 23 cheetahs (chapter 5), and one brown hyaena (chapter 6). Of these, nine cheetahs, all leopards and the hyaena were translocated as individuals while the remaining 14 cheetahs were captured and released in groups (two groups of three, four groups of two). The individual costs for 10 dependent sub-adult cheetahs that were translocated together with their mothers were excluded because the presence of offspring did not significantly increase case-specific costs for these females (cheetahs 07, 56, 58 and 59 in Table 7.1 respectively), nor could costs be attributed to individual sub-adults reliably.

Carnivores were held for periods between 1 - 1,138 days corresponding respectively to immediate release and the rearing of orphaned cubs. Transportation distances (collection - holding - release, or collection - collaring - release) were dependent on source site location and recipient site selection, but ranged from 63 km - 842 km. Both hard (*n* = 19) and soft (*n* = 11) releases were employed. Where possible GPS transmitters were deployed (*n* = 18) but for group releases of animals expected to remain together (only for cheetahs), a combination of one GPS transmitter and VHF transmitters (*n* = 7) was deployed. If GPS transmitters were not immediately available, VHF units were also deployed for releases of individuals (*n* = 3), but only in situations when an on-site post-release monitoring team was present at the recipient site. One identification collar was deployed on a cheetah female as part of a group release (Aju02 in Table 7.1).

7.3.2 Cost summary

There was a wide range of total cost between animals, with the most expensive ($7,559) relating to an orphaned cheetah cub (held until old enough to release at four years, translocated 348 km and soft released with a GPS transmitter) and the cheapest ($269) associated with the immediate hard release of an adult cheetah translocated 71 km with a VHF transmitter (Aju40 and Aju65 respectively in Table 7.1). Total translocation cost for all 30 individuals was $80,681. Median translocation cost per individual was $2,393, and $2,669 per event (Table 7.2). Median cost per cheetah was $2,760 (*n* = 23), and $2,108 per leopard (*n* = 6). The hyaena was translocated at a cost of $1,672 (Table 7.2).

The distribution of these costs into different cost categories is shown in Figure 7.1. Since the cost of tracking equipment and expenses for GPS data retrievals comprised more than half of the total cost ($44,906.05, 56%), the distribution of costs without tracking is also shown in Figure 7.1 and Table 7.2. Each cost category showed a wide range of associated costs, although their magnitudes were very different. Permit costs were the cheapest ($1.27 - $4.12), followed by staff ($13.77 - $203.84), veterinary ($23.76 - $756.22) and feeding ($0 - $889.43) costs. Tracking technology ($27.00 - $3,649.00) and holding ($0 - $3,656.83) costs were the biggest factors in determining the total cost for an individual animal (Fig. 7.1; Appendix 29).

Table 7.1 – Details and translocation cost for 30 large carnivores, 2008-2012.

Individual	Age (years)	Sex	Year	Capture reason	Captivity (days)	Transportation distance (km)	Collar Type	Degree of habituation	Release mode	Success	Total cost ($US)
Aju01	2-3	F	2008	Indiscriminate	10	530	VHF	Wild	Hard – group	Yes	636.52
Aju02	2-3	F	2008	Indiscriminate	10	530	ID	Wild	Hard – group	Yes	443.52
Aju03	2-3	M	2008	Indiscriminate	10	530	VHF	Wild	Hard – group	No	636.52
Pp06	4-5	M	2008	Livestock raider	16	372	VHF	Wild	Hard – single	Yes	867.97
Aju07	7-9	F	2008	Livestock raider	61	461	VHF	Wild	Hard – single[a]	No	1,117.44
Pp15	4-6	F	2009	Confiscation	168	741	GPS	Wild	Hard – single	Yes	4,145.27
Aju17	5-7	M	2009	Livestock raider	175	493	GPS	Wild	Hard – single	Yes	3,980.34
Aju18	5-7	F	2009	Livestock raider	157	473	GPS	Wild	Hard – single	No	3,994.94
Aju19	3-5	M	2009	Indiscriminate	62	314	VHF	Wild	Hard – group	No	829.78
Aju20	6-8	F	2009	Indiscriminate	37	444	GPS	Wild	Hard – group	Unknown	4,228.91
Aju26	3-5	M	2009	Indiscriminate	12	184	GPS	Wild	Hard – single	Unknown	1,748.82
Pp27	2	F	2009	Orphan	639	316	GPS	Wild	Hard – single	Yes	3,027.05
Aju29	2-3	F	2010	Orphan	596	485	VHF	Semi-habituated	Soft – group	Yes	1,478.02
Aju30	2-3	M	2010	Orphan	446	467	GPS	Semi-habituated	Soft – group	Unknown	2,965.06
Aju34	3-4	M	2010	Indiscriminate	47	141	VHF	Wild	Hard – single	Yes	512.92
Aju38	3-4	M	2011	Indiscriminate	153	265	GPS	Wild	Soft – single	Yes	7,433.32
Aju40	3-4	F	2012	Indiscriminate	1,184	348	GPS	Habituated	Soft – group	No	7,558.86
Aju41	3-4	F	2012	Indiscriminate	1,184	348	VHF	Habituated	Soft – group	No	4,466.08
Aju42	3-4	M	2011	Orphan	1,055	824	VHF	Habituated	Soft – group	No	2,828.41
Aju43	3-4	M	2011	Orphan	1,106	842	GPS	Habituated	Soft – group	No	6,180.96
Aju44	3-4	M	2011	Orphan	1,055	824	VHF	Habituated	Soft – group	No	2,858.16
Pp55	2-4	M	2011	Livestock raider	206	764	GPS	Wild	Hard – single	No	2,005.67
Pp47	2-4	M	2012	Livestock raider	183	400	GPS	Wild	Soft – single	Yes	2,208.40

Hbr55	1-2	F	2012	Livestock raider	3	63	GPS	Wild	Hard – single	No	1,671.82
Aju56	5-7	F	2012	Indiscriminate	169	403	GPS	Wild	Hard – single[b]	Yes	3,848.38
Pp57	3-4	F	2012	Livestock raider	4	71	GPS	Wild	Hard – single	No	1,744.09
Aju58	5-7	F	2012	Indiscriminate	260	411	GPS	Semi-habituated	Soft – single[a]	Yes	2,759.10
Aju59	4-6	F	2012	Indiscriminate	272	801	GPS	Semi-habituated	Soft – single[b]	No	2,577.36
Aju65	6-7	M	2012	Indiscriminate	1	71	VHF	Wild	Hard – group	No	268.81
Aju66	6-7	M	2012	Indiscriminate	2	71	GPS	Wild	Hard – group	No	1,658.41

Aju indicates cheetah; Pp indicates leopard; Hbr indicates brown hyaena. Year is year of release. Group releases represent coalitions and not presence of offspring; groups are indicated by light grey shading.

[a] translocated with two dependant offspring;

[b] translocated with three dependant offspring

Table 7.2 – Success rates, costs and estimated Individual Conservation Cost (ICC) for large carnivore translocations.

All values are in USD. A – Total cost for translocation study by category. B – Total cost without post-release tracking technology and expenses.

		A–Total Cost				**B–Total Cost without Tracking**			
Total		80,680.91				35,774.86			
Category	Success rate[a]	Min.	Median	Max.	Estimated ICC	Min.	Median	Max.	Estimated ICC
All individuals (*n* = 30)	0.44	268.81	2,392.88	7,558.86	5,982.20	75.03	627.31	4,275.83	1,568.28
All events (*n* = 22)	---	512.92	2,668.23	12,024.94	---	88.44	799.89	8,534.69	---
Cheetah (*n* = 23)	0.40	268.81	2,759.10	7,558.86	6,897.75	75.03	606.65	4,275.83	1,516.63
Leopard (*n* = 6)	0.67	867.97	2,107.03	4,145.27	3,139.47	160.71	675.16	1,641.81	1,005.99
Brown hyaena (*n* = 1)	0.00	1,671.82	---	1,671.82	---	88.44	---	88.44	---
Hard release (*n* = 19)	0.47	268.81	1,671.82	4,228.91	3,557.06	75.03	496.27	1,641.81	1,055.89
Soft release (*n* = 11)	0.40	1,478.02	2,858.16	7,558.86	7,145.40	905.07	2,638.16	4,275.83	6,595.40

[a] Individuals with unknown translocation outcome (*n* = 3) were removed from success rate calculations. Cost per ICC was estimated as the median cost / translocation success in each category, except for category *Events* because cheetah group releases resulted in both successful and unsuccessful translocations.

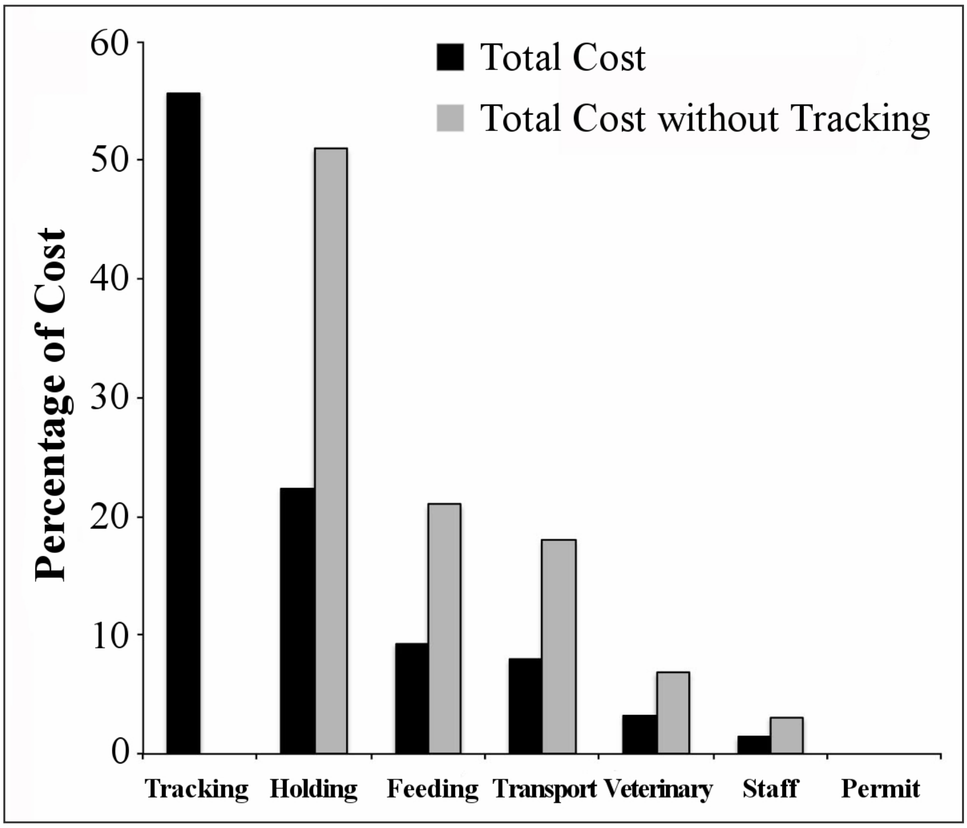


Figure 7.1 – Percent contribution of different cost categories to overall translocation cost of 30 large carnivores. *Graph shows the contribution of different cost categories with tracking technology included (black bars) and excluded (grey bars).*

7.3.3 Variables influencing costs

An individual's total translocation cost and time spent in captivity were strongly correlated (Spearman's correlation: $r_s = 0.654$, $p < 0.001$), but there was no significant correlation between an individual's total cost and translocation distance ($r_s = 0.194$, $p = 0.303$). When tracking costs were excluded, translocation cost and captive time remained strongly correlated ($r_s = 0.897$, $p < 0.005$) and translocation distance was also significantly correlated with total cost ($r_s = 0.367$, $p < 0.05$). There were no significant correlations of total cost with year of capture ($r_s = 0.203$, $p = 0.281$) or year of release ($r_s = 0.280$, $p = 0.135$) suggesting that inflation and currency conversion rates had no significant impact on cost results.

There was also no significant difference in costs for cheetahs and leopards (Table 7.3) although maximum individual costs were considerably higher for cheetah ($7,558.86; $4,275.83 without tracking costs) than for leopard ($4,145.27; $1,641.81 without tracking costs), or the hyaena ($1,671.82) (Table 7.2). There were no significant differences in costs when analysed by sex, background/capture reason, degree of habituation, or group composition (Table 7.3). There was, however, a significant difference between mean

Table 7.3 – Test statistics for comparison between different translocation cost categories.

Category	**Assessment**	**Test**	**Test scores**	**Probability**
Species	Cheetah (*n* = 23) vs. Leopard (*n* = 6)	Wilcoxon-Mann-Whitney U	U = 346.0	*p* = 0.979
Sex	Male (*n* = 15) vs. Female (*n* = 15)	Wilcoxon-Mann-Whitney U	U = 90.5	*p* = 0.373
Release mode	Hard (*n* = 19) vs. Soft (*n* = 11)	Wilcoxon-Mann-Whitney U	U = 237.0	$p < 0.014^*$
Background	Indiscriminate (*n* = 15) vs. Livestock raider (*n* = 8)	Kruskal-Wallis	H = 0.67; z = 0.22	*p* = 0.821
	Indiscriminate (*n* = 15) vs. Rehabilitation (*n* = 7)	(pair-wise)	H = 1.88; z = 0.63	*p* = 0.525
	Livestock raider (*n* = 8) vs. Rehabilitation (*n* = 7)		H = 1.47; z = 0.63	*p* = 0.524
Collar type	VHF/ID (*n* = 12) vs. GPS (*n* = 18)	Wilcoxon-Mann-Whitney U	U = 97.0	$p < 0.001^*$
Group composition	Individual (*n* = 12) vs. Group (*n* = 14)	Kruskal-Wallis	H = 3.17; z = 1.05	*p* = 0.291
	Mother with cubs (*n* = 4) vs. Group (*n* = 14)	(pair-wise)	H = 1.44; z = 0.47	*p* = 0.632
	Individual (*n* = 12) vs. Mother with cubs (*n* = 4)		H = 0.00; z = 0.00	*p* = 1.00
Degree of habituation	Wild (*n* = 19) vs. Semi-habituated (*n* = 4)	Kruskal-Wallis	H = -2.23; z = -0.55	*p* = 0.578
	Wild (*n* = 19) vs. Habituated (*n* = 5)	(pair-wise)	H = -5.94; z = -1.56	*p* = 0.118
	Semi-habituated (*n* = 4) vs. Habituated (*n* = 5)		H = -1.57; z = -0.85	*p* = 0.391

* denotes significance at $\alpha = 0.05$

translocation costs when analysed by release mode (Table 7.3) with soft releases (\$3,937.61 ± \$648.53 SE, n = 11) being significantly more expensive than hard releases (\$1966.69 ± 329.19 SE, n = 19) (Table 7.3). This was also the case when comparing cost by release mode for cheetahs only (U = 26.0, $p < 0.02$). However, in this study, soft release did not have the desired effect of increasing site fidelity and survival of cheetahs (chapter 5).

Due to the high unit costs for GPS tracking collars and 12-months data fees (range = \$1,303.33 - \$3,649.00, n = 18), translocation costs were also significantly higher for individuals released with GPS units when compared with individuals with VHF tracking collars (Table 7.3). When tracking costs were removed, orphaned individuals (n = 6) cost significantly more than non-orphans (n = 24) (U = 147.0, $p = 0.001$) corresponding with long captive times for these individuals (Table 7.1). Feeding costs were highly correlated with time spent in captivity at the holding site ($r_s = 0.994$, $p < 0.001$) and there was also a strong correlation between transport costs and transportation distance ($r_s = 0.475$, $p = 0.008$). The summary statistics for different cost variables are shown in Appendix 30.

Including species, sex, collar type and release mode as categorical variables, as well as captive time and translocation distance as continuous variables, Decision-Tree partitioning (Fig. 7.2) showed that collar type was predicted to have the highest overall influence on differences in translocation costs, followed by captive time and release mode (Table 7.4). This partitioning interrogation accounted for nearly 60% of variation in translocation cost data ($r^2 = 0.595$, n = 30), to which collar type, captive time and release mode contributed 60.3%, 26.4% and 13.3% respectively. However, the limited sample size restrained partitioning to four node splits.

Table 7.4 – Association of different cost variables with translocation cost.

Results determined with Decision Tree partitioning (Fig. 2).

Candidate	Candidate SS	LogWorth	Cut point
Collar type	377735.24.22*	4.040657850	VHF
Captive time (days)	36543887.24	2.945667719	153
Release mode	16466365.66	1.587087917	Hard
Sex	4055517.75	0.502900636	Male
Translocation distance	11784943.21	0.356294278	217.84
Species	728117.32	0.061238403	Brown hyaena, Leopard

[*] denotes optimum split for available data

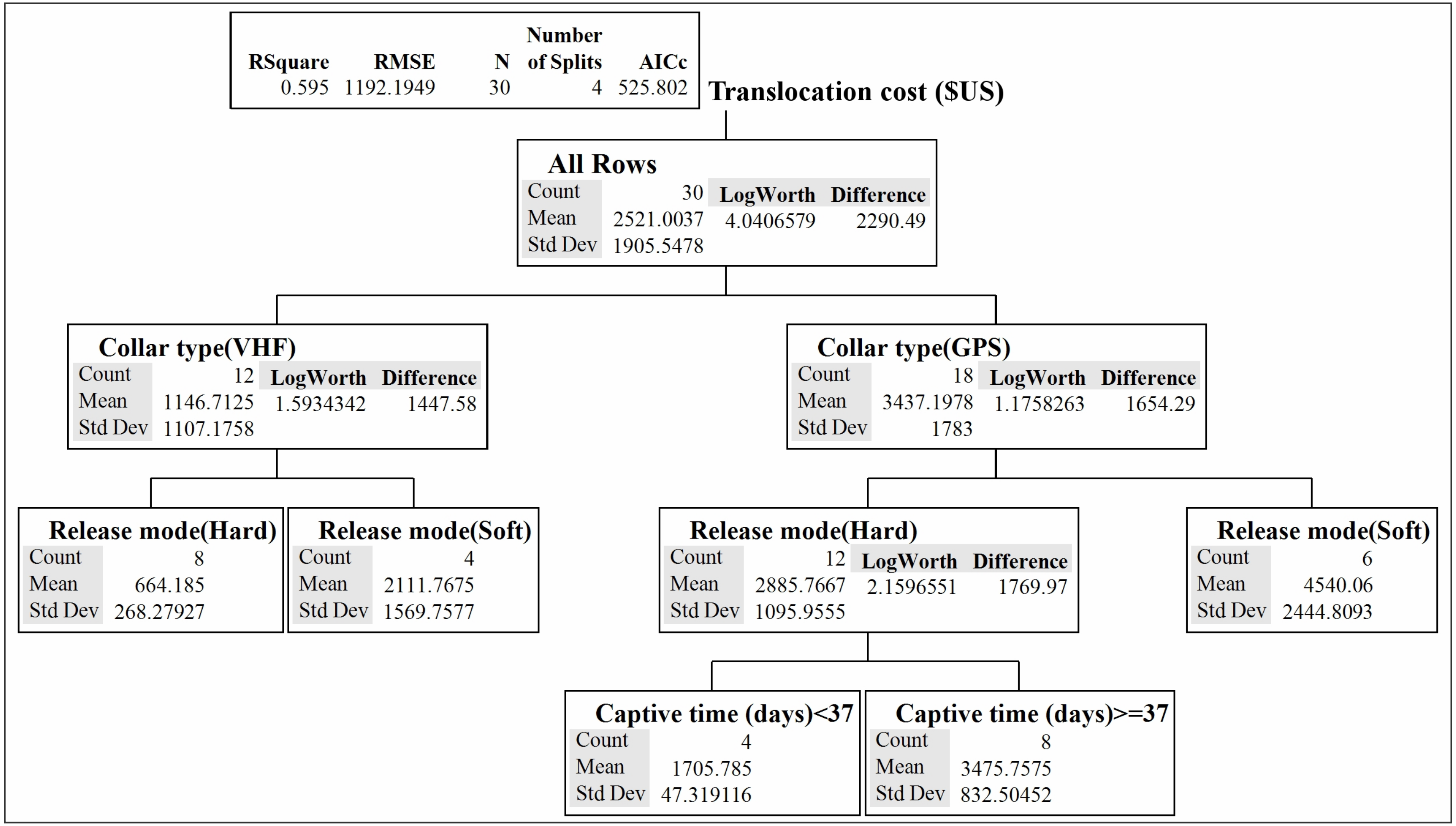


Figure 7.2 – Results of Decision Tree partitioning interrogation of translocation cost variables. *Graph shows the results for three node splits. Explanatory variables included species, sex, collar type and release mode as categorical factors, and captive time and translocation distance as continuous factors.*

7.3.4 Variables influencing translocation success

Due to the high variability in observed translocation success (*cf.* chapters 4 and 5) and its direct impact on associated ICCs (Table 7.2), it is imperative to assess the factors that may increase translocation success rates and maximise cost-efficiency.

Leopards had the highest translocation success rate in this study (Table 7.2) although overall success was not significantly associated with species ($\chi^2 = 2.53$, d.f. = 2, $p = 0.281$, $n = 27$) and this did not change when the single hyaena was removed from this analysis ($\chi^2 = 1.33$, d.f. = 1, $p = 0.248$, $n = 26$). Success was also not significantly associated with sex ($\chi^2 = 0.35$, d.f. = 1, $p = 0.546$, $n = 27$), capture background of individuals ($\chi^2 = 0.24$, d.f. = 2, $p = 0.883$, $n = 27$), or with group composition at the time of release ($\chi^2 = 3.63$, d.f. = 2, $p = 0.162$, $n = 27$).

Although all translocations carried out at <100 km failed (cheetah males Aju65/Aju66 successfully homed; leopard Pp57 and brown hyaena Hbr55 were killed in road accidents), translocation distance was only weakly associated with translocation success (Log-Likelihood = 18.55, $r^2 = 0.001$, $\chi^2 = 0.05$, d.f. = 1, $p = 0.821$, $n = 27$). In addition, time in captivity did not explain translocation success satisfactorily (Log-Likelihood = 1.05, $r^2 = 0.057$, $\chi^2 = 2.10$, d.f. = 1, $p = 0.147$, $n = 27$), but cheetahs held >1,000 days were particularly unsuccessful (chapter 5). The failure of these events was strongly associated with the animals' fully habituated behavioural state (chapter 5) that was influenced by long captivity (cheetahs only: LogLikelihood = 21.58, $\chi^2 = 43.15$, d.f. = 2, $p < 0.001$, $n = 23$) contributing to a lack of fear from human approach. Four of five fully-habituated cheetahs were shot within seven months of their release (chapter 5). Conversely, leopards did not habituate to human presence, regardless of the length of temporary captivity (Table 7.1). Hence, degree of habituation to human presence was one of the strongest predictors of translocation success ($\chi^2 = 6.99$, d.f. = 2, $p = 0.030$, $n = 27$). The association between degree of habituation and translocation success was even stronger when the outcomes of all individuals with wild and semi-habituated behavioural states were pooled (these individuals retained an active flight response from human presence, see definition in chapter 2) and assessed against the fully habituated behavioural state ($\chi^2 = 6.78$, d.f. = 1, $p = 0.009$, $n = 27$). The least successful individuals in this study (cheetahs Aju40-44) were also among the most expensive (Table 7.1).

The poor survival of fully habituated cheetahs also influenced the assessment of translocation success by release mode. Overall, there was no significant association between release mode and translocation success ($\chi^2 = 0.68$, d.f. = 1, $p = 0.407$, $n = 27$), but all five fully habituated cheetahs were released from acclimatisation pens and soft-release success increased to 80.0% (n = 5) when these individuals were not considered. Nevertheless, this did not result in a significant association between release mode and translocation success ($\chi^2 = 0.86$, d.f. = 1, $p = 0.352$, $n = 22$).

7.3.5 Linking cost with success – the Individual Conservation Cost concept

Based on the monitoring outcomes described in chapters 4 – 6, success evaluations were possible for 27 of the 30 translocated animals. The GPS collars failed on three cheetahs (Aju20, Aju26 and Aju30 in Table 7.1) before 12 months had elapsed. Using both biological and conflict parameters (chapter 2), four of the six leopards (66.7%) were classed as successful (chapter 4), as were eight of the 20 cheetahs with known outcome (40.0%, chapter 5) (Table 7.2). The single brown hyaena was classed as unsuccessful (chapter 6). This gives an overall success rate of 44.4% (Table 7.2). Of the 15 confirmed unsuccessful translocations, 11 individuals died in their first year post-release (one leopard, nine cheetahs and the brown hyaena), two cheetahs homed back to the capture site (Aju65/66), one cheetah was returned into the captive facility (Aju07), and the remaining leopard (Pp45) was reported to raid livestock.

Due to the manifold explanatory variables (e.g. species, sex, age, capture reason, group composition, captive time, transportation distance, release mode, and degree of habituation), analysing the inter-relations of costs with translocation success is very difficult. Such a nine-dimensional problem requires approximately 1,024 samples with known outcomes for modelling paradigms to achieve predictive power. Indeed, extensive explorations of the available data with various analysis paradigms such as Bayesian Networks (Linklater et al., 2011), K-means cluster analysis (Sparks, 1973), Generalised Linear Models, and Random Forests (Breiman, 2001) did not deliver definitive results of the associations among costs, translocations success and the influencing factors. Therefore, an adjustment of costs by observed success rate (the ICC) was employed to produce a realistic estimate of the cost that managers have to anticipate for successful translocations.

Overall, the mean costs of successfully ($2,581.13 ± $597.41 SE, n = 12) and unsuccessfully ($2,393.15 ± $508.60 SE, n = 15) translocated carnivores did not differ significantly (Wilcoxon-Mann-Whitney U test: U = 120.5, $p = 0.755$). Pooled for all species, the estimated ICC was $5,983 (Table 7.2) giving an indication of the true expense incurred for one successfully translocated large carnivore. However, due to the lower success rate of cheetahs, the ICC for this species ($6,897.75) was more than double that for leopards ($3,139.47). When the five fully habituated, rehabilitated cheetahs were removed from this analysis, the ICC for cheetahs decreased to $2,427.04 (Table 7.4) and was less than that of leopards. The low success of these five costly cheetahs also influenced the high ICC for soft release mode ($7,145.40), being more than double that of hard releases ($3,557.06), although this was also attributable to high costs associated with the building of large acclimatisation pens in remote areas of Namibia.

7.3.6 Cost recuperation

In this study, translocation costs were successfully recuperated from external funding sources, including zoos, non-governmental carnivore conservation groups, private individuals and businesses for expenses in four categories – tracking, veterinary, transport and holding (Fig. 7.3; Appendix 29). There were no opportunities for recuperation of permit costs or staff overheads. In the case of tracking costs, the most costly category at 56% of total costs (Fig. 7.1), external funding fully covered the expenses of 20 of 29 transmitters deployed (15 of 18 GPS transmitters, five of 11 VHF transmitters) giving a total recuperation of $34,517.62 (77%). Similarly, 76% of veterinary costs, 91% of transport costs, and 58% of holding costs were successfully recuperated (Fig. 7.3).

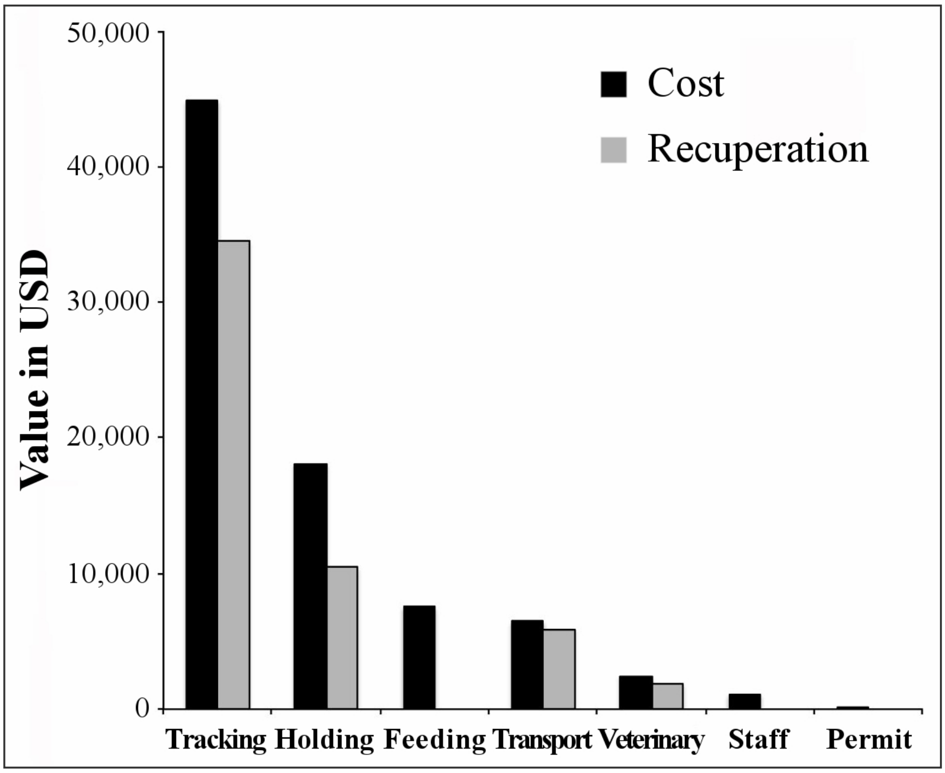


Figure 7.3 – Total translocation cost recuperation per cost category. *Graph shows the amount of translocation cost recuperated from external funding bodies (grey bars) in different cost categories in relation to total expenditure (black bars). Data represent 30 large carnivore translocations in Namibia, 2008 -2012.*

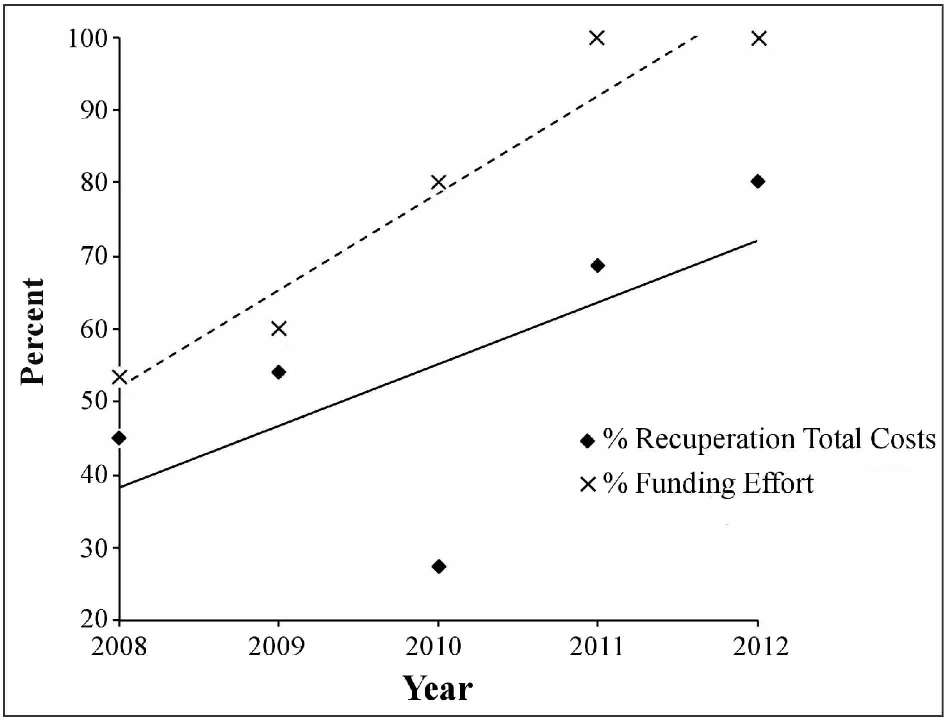


Figure 7.4 – Percent recuperation of total large carnivore translocation costs and funding effort across the study period. *Graph shows the proportion of translocation cost that could successfully be recuperated from external funding bodies – the increasing trend in recuperation corresponds with an increasing effort to source external funding. Year indicates year of release. Lines show best linear regression fits. The low recuperated value in 2010 was associated with a small sample of released individuals (n = 3) for which cost recuperation was less successful (see Appendix 29).*

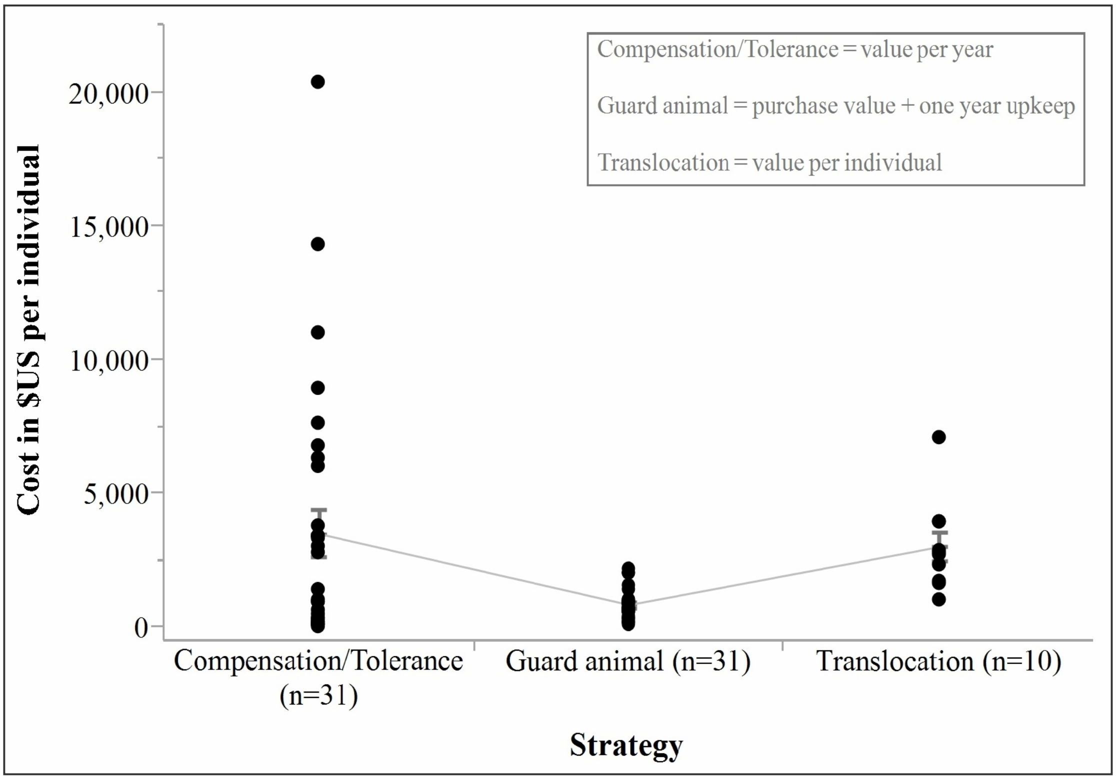
The total amount recuperated from external sources was $52,791.58, equivalent to 65.4% of the total cost of all translocations. This figure decreases to 51.1% if tracking costs are removed. Total recuperation equates to a mean of $1,762.45 ± $304.58 SE per animal ($0 - $6,270.42) or 65.5% of the average translocation cost ($2,521.00 ± $347.00 SE, $n = 30$). Mean percentage recuperation was not significantly different between cheetah (61.6% ± 6.0% SE, $n = 23$) and leopard (62.1% ± 13.5% SE, $n = 6$) (U = 92.0, $p = 0.9356$) and 98.6% of the hyaena translocation cost was recuperated (Appendix 29).

The percentage of total translocation costs recuperated per year increased during the study - it almost doubled from 2008 (45.2%) to 2012 (80.4%). This trend was linked to an increasing effort to source funding – moving from eight proposals submitted to external funding agencies in 2008 to 15 proposals in 2012 (Fig. 7.4).

7.3.7 Translocation cost in relation to other non-lethal strategies

From the extensive literature search on cost estimates for various non-lethal large carnivore conflict mitigation options, the most common strategies evaluated were compensation of livestock damage and livestock guarding animals. These categories appeared as key concepts in >50% of reviewed publications, although rarely with comparable cost estimates. In total, compensation and tolerance payment costs (based on observed livestock/game farming damage) were sourced from 19 publications (for a range of 11 large carnivore species) and guard animal costs from another 25 studies (for a range of four guard animal species) (reference details in Fig. 7.5). These articles reported sufficiently independent cost estimates that were scaled or convertible to expenses relating to an individual/year (not considering life-time estimates, Zabel and Holm-Müller, 2008). Conversely, only five studies reported translocation costs for conflict large carnivores (Riley et al., 1994; Purchase, 1998; Marnewick et al., 2009; Fontúrbel and Simonetti, 2011; Boast et al., 2015) and with a strong bias towards cheetah as the key focal species.

Based on published cost information and the median translocation cost per species from this study (Table 7.2), the average cost of a large carnivore translocation (\$3,020.8 ± \$543.04 SE, $n = 10$) was less than, but not significantly different from (Wilcoxon-Mann-Whitney U test: U = 246.0, $p = 0.281$) that of annual compensation/damage tolerance payments (\$3,506.57 ± \$862.63 SE, $n = 31$). In comparison with translocation costs, compensation/tolerance payments showed a high degree of variability (Fig. 7.5) reflecting the strong influence of local damage circumstances on compensation/tolerance costs. Conversely, the average purchase price of different livestock guard animals (including one year maintenance cost) was less variable and reported at \$857.75 ± \$93.63 SE ($n = 31$) (Fig. 7.5). It was therefore significantly different from translocation and compensation costs (Welch's ANOVA: $F_{2,19} = 11.74$, $p < 0.001$).



7.5 – Published cost estimates for different non-lethal large carnivore conflict management strategies. *Graph shows published costs for resolving human – predator conflict by means of post-hoc damage compensation and tolerance payments, through translocation of offending carnivores or by purchase of a suitable guarding animal including one year maintenance cost. All published cost data were converted to USD. Lines connect mean cost estimate per category. Error bars show 1 SE of the mean. Data only include published cost values from studies that reported expenses on the basis of a single individual (including average estimates). Note: published costs are not adjusted by inflation.*

Category *Compensation/Tolerance* includes values for wolf (*n*=11), lion (*n*=4), African wild dog (*n*=3), leopard (*n*=3), cheetah (*n*=1), spotted hyaena (*n*=1), brown bear (*n*=2), wolverine (*n*=2), Eurasian lynx (*n*=2), puma (*n*=1), and snow leopard (*n*=1). Category *Guard animal* includes values for domestic dog (*n*=24), llama (*n*=3), donkey (*n*=3), and alpaca (*n*=1). Category *Translocation* includes values for cheetah (*n*=4), leopard (*n*=1), brown hyaena (*n*=1), grizzly bear (*n*=1), canids (*n*=1), ursids (*n*=1) and felids (*n*=1). Data sources (ordered alphabetically): Andelt, 1985; Bergman et al., 1998; Blanco et al., 1992; Boast et al., 2015; Boitani et al., 2010; Corff, 2005; Cotterill, 1997; Echegaray and Vilà, 2010; Fontúrbel and Simonetti, 2011; Franklin and Powell, 1993; Gehring et al., 2010; Gilady, 2000 (cited in Haney et al., 2007); González et al., 2012; Green, 1989; Green and Woodruff, 1999; Green et al., 1980, 1984; Haney et al., 2007; Hansen and Smith, 1999; Hötte and Bereznuk, 2001; Lindsey et al., 2005a; Maclennan et al., 2009; Marnewick et al., 2009; McManus et al., 2014; Meadows and Knowlton, 2000; Mech, 1999; Mertens and Promberger, 2001; Mizutani, 1999; Otstavel et al., 2009; Patterson et al., 2004; Pfeifer and Goos, 1982; Potgieter, 2011; Purchase, 1998; Ribeiro and Petrucci-Fonseca, 2005; Riley et al., 1994; Śmietana, 2005; Spearing, 2002; Swarner, 2004; Swenson and Andrén, 2005; Treves et al., 2002; van Bommel, 2010; VerCauteren et al., 2008; Verdade and Campos, 2004; Walton and Feild, 1989; Wilbanks, 1995; Woodroffe et al., 2005b; Woodruff and Green, 1995.

7.4 Discussion

The conservation of large carnivores continues to present a major challenge around the world (Ripple et al., 2014), and, to a large extent, depends on the availability and most efficient allocation of financial resources (Dickman et al., 2011). The results from this study show that free-range translocations (for conflict reduction and rehabilitation purposes) are costly. Although the median translocation cost per individual ($2,393) was less than previously estimated for large terrestrial carnivores (*cf.* ~$3,800, Fontúrbel and Simonetti, 2011), the adjusted cost per successful animal ($5,983) was considerably higher.

Translocation costs – across species, individuals and translocation scenarios – were highly variable, reflecting the case-specific circumstances of events. For example, the translocation of leopards was nearly twice as successful as that of cheetahs whilst less than half as expensive - the ICC for leopard was $3,140 and that for cheetah $6,898. In agreement with Boast et al. (2015), tracking technology was the most significant cost factor. Because releases were carried out into unconfined environments, the deployment of tracking technology was considered important in this study and it was employed to enable intensive post-release monitoring for several reasons. First, very little translocation-related information was available for the focal species and the need for intensive monitoring has been strongly emphasized in previous reviews of the strategy (Linnell et al., 1997; Craven et al., 1998; Miller et al., 1999; Massei et al., 2010). Second, the dissemination of monitoring information improved accountability of these translocations for local land managers. However, since tracking is not a biological prerequisite for translocation success, translocation costs could potentially be reduced by over 50% if animals were released without collars. Under semi-controlled conditions (i.e. reserves with predator-proof fencing), VHF units are sufficient to achieve effective monitoring (e.g. Hunter, 1998a), reducing the average translocation cost by approximately $2,300 per animal. Correspondingly, translocation studies that utilised VHF units for monitoring reported an individual's cost at $1,000 - $2,000 (Riley et al., 1994; Purchase, 1998). However, where large carnivores are released into free-range environments, the use of GPS technology is recommended because VHF-tagged subjects may easily be lost (e.g. Hamilton, 1981; Purchase, 1998). Moreover, long-term VHF-based radio telemetry tracking of far-ranging carnivores requires regular use of aircraft (e.g. Hamilton, 1981; Belden and Hagedorn, 1993; Stander et al., 1997a; Ruth et al., 1998; Bradley et al., 2005), resulting in considerable additional monitoring costs and high operational risks for personnel (Bangs et al., 2006). Although at a higher initial cost, the use of GPS transmitters also allows for more detailed evaluation of translocations (Weilenmann et al., 2010; Houser et al., 2011;

Boast et al., 2015) and, due to strong public interest in the fate of individual carnivores, a substantial proportion (~77%) of tracking costs can successfully be recuperated from external funding bodies.

Another significant cost factor in this study was release mode. For example, the use of soft releases for cheetahs resulted in case-specific costs exceeding $5,000 - here resulting from the necessity to build large enclosures in remote semi-desert areas. Despite higher costs, soft release is reported to improve translocation outcomes and reintroduction success for different large carnivores (e.g. Belden and Hagedorn, 1993; Bradley et al., 2005; Hayward et al., 2007b; Gusset et al., 2008). However, in terms of translocation success, the results from this study show little beneficial effect of the technique in comparison with hard release. Soft release success increased to 80% when the five habituated cheetahs were excluded from this assessment, but the significant extra labour effort and cost (~$2,000 per animal) associated with this strategy did not effect the key objective of improving site fidelity for these individuals (chapter 5; also observed for leopards in Kenya, Hamilton, 1981). Therefore, the approach may be more cost-efficient during targeted reintroductions rather than free-range translocations. To contain carnivores with predator-proof fencing, reserves need to account for high initial infrastructure investments of up to ZAR 2,000,000 and additional annual maintenance costs of approximately ZAR 150,000 – 400,000 (Hayward et al., 2007b).

Translocation costs were also influenced by the amount of time that individuals were maintained in temporary captivity. Holding and feeding accounted for about 67% of the total cost of all translocations when tracking technology was not considered. Captive rehabilitation of orphaned and confiscated individuals in particular increases case-specific expenses - the annual rehabilitation cost for individual cheetahs and leopards can be as high as ~$6,000 (Houser et al., 2011). In the present study, cheetah translocation success was strongly influenced by the failed releases of five habituated individuals (Aju40-44), all of which were held captive for over 1,000 days. This failure was primarily attributed to the cheetahs' naivety resulting from habituation to humans during prolonged captivity (Jule et al., 2008). Therefore, the least successful translocations also involved some of the most expensive specimens, inflating the ICC for cheetahs. Similarly, three orphaned, rehabilitated cheetahs released in Botswana were shot within seven months, even after pre-release training was employed to mitigate habituation to humans (Houser et al., 2011). Excluding rehabilitations from the results of this study, the ICC for cheetahs decreased to $2,428 ($n$ = 15). In comparison, rehabilitation was a feasible strategy for leopards, and the species' ICC was \$2,577 when another successful case study with known costs (Houser et al., 2011) was included in the results.

In general, the unsuitability of most carnivores for release after prolonged captivity (Jule et al., 2008; Hunter and Rabinowitz, 2009; Houser et al., 2011), coupled with high costs for these subjects, suggests that their use in free-range translocations should generally be avoided, or limited to the most suitable candidate animals (e.g. leopards). Particularly for human-imprinted cheetahs, the cost-efficiency of rehabilitation and translocation efforts can be improved significantly if subjects are released onto fenced reserves that minimise the risk of persecution, facilitate range expansion of the species, and contribute to tourism efforts that are enhanced by habituation to humans (Marnewick et al., 2009). Due to higher survival rates the estimated ICC for cheetahs relocated to private reserves in South Africa was \$3,192 (Marnewick et al., 2009) and therefore <50% of the cost of semi-successful free-range events reported here (\$6,898), and <10% of the cost of failed free-range translocations reported in Botswana (estimated ICC = \$39,066, Boast et al., 2015). Thus, depending on local translocation circumstances, the ICC for cheetahs may vary by a magnitude of >12. This emphasises the need to identify the most successful translocation protocols, the most suitable recipient sites, as well as the most suitable candidate cheetahs. However, successful establishment of conflict or rehabilitated cheetahs on isolated reserves will eventually require artificial population management through a meta-population approach (Marnewick et al., 2009) that necessitates considerable additional funding for maintenance and logistics (e.g. for African wild dogs in South Africa, Lindsey et al., 2005a).

Due to the significant costs associated with translocations, agencies that undertake them benefit from distributing costs across a wider stakeholder group with an interest in non-lethal management of conflict carnivores. The charisma and high public profile of large carnivores can generate substantial non-governmental funding over extended periods of time (e.g. >\$550,000 for wolf damage compensation between 1987 – 2005, Bangs et al., 2006), thus alleviating financial strain on limited government funds. Public support was an important component in this study too, and nearly two-thirds of the total translocation costs were successfully recuperated from national and international non-governmental organizations, institutions and/or individuals. For example, almost 80% of the main cost (tracking technology) was recuperated. This was possible because >75% of all funding proposals (n = 59) specifically focussed on sourcing collar/data funds, some manufacturers provided units at discounted rates, and collars were the favoured funding choice by individual donors and sponsors. Significant support could also be sourced through in-kind donations of veterinary services and research vehicles. In addition, recipient areas provided important logistical assistance (e.g. soft release enclosures and animal care during acclimatisation) and supported monitoring efforts. Similarly, in return for receiving tourism attractions (relocated conflict cheetahs), South African recipient areas covered all costs pertaining to translocations, including a ZAR 10,000 payment to landowners trapping perceived conflict cheetahs and ZAR 5,000 for operational costs (Buk and Marnewick, 2010). Here, costs that were not recoverable (i.e. salaries, permits, feeding, and partial holding) were carried by a private wildlife sanctuary that generated matching funds through an international volunteering programme and tourism lodge. Income from non-consumptive tourism has great potential to support carnivore conservation activities (Lindsey et al., 2005b), and if exploited effectively, the tourism value of large carnivores far outweighs the cost of their maintenance (including provisions for damage compensation) on private and public PAs (e.g. Stuart-Hill and Grossman, 1993; Cotterill, 1997; Bangs et al., 2006; Hayward et al., 2007b).

Therefore, where expensive carnivore translocations form a necessary part of endangered species management (e.g. Goodrich and Miquelle, 2005; Gussett et al., 2008; Marnewick et al., 2009), conservation agencies can utilise different funding avenues to distribute costs across a wide audience. Public-private partnerships could be a particularly effective way of distributing high carnivore conservation costs across various stakeholders (Bangs et al., 2006). Nevertheless, involving a variety of interest groups in conservation funding schemes can introduce different perceptions and expectations. Although this was not a problem in the present study, diverging opinions about species management may result in controversies over payment rules and ultimately in political, and even legal, complications that could erode funding support (Treves et al., 2009). Here, the proportion of costs that was recuperated increased on an annual basis but this trend was directly linked to increasing efforts to source funding and thus resulted in considerable administrative efforts that need to be borne in mind by managers.

Conservation cost accounting has improved in recent years, but direct comparison of this study with other non-lethal conflict mitigation alternatives remains difficult since there is no standardised methodology for reporting costs. When considering the cost by individual, translocation did not appear more expensive than damage compensation, which provides an alternative symptomatic conflict mitigation approach. According to Fontúrbel and Simonetti (2011), the cost of a single cheetah/leopard translocation is equivalent to compensation of eight livestock. For those individuals with a confirmed livestock depredation background (Table 7.1), the reported pre-translocation damage amounted to an average of 13.3 livestock/cheetah (*n* = 3) and five livestock/leopard (*n* = 4), making translocation a comparatively costly option for conflict leopards, but not for endangered cheetahs. At Namibian costs (~$391/calf, Stein et al., 2010), the hyaena's translocation cost matched the livestock losses (*n* = 4) attributed to this individual (chapter 6). In relation to the pooled international compensation/tolerance cost estimate for a single large carnivore (~$3,507/year), however, the median translocation cost incurred here was approximately 32% cheaper. Implemented at large scales, carnivore conflict compensation schemes require substantial annual payments, i.e. hundreds of thousands or millions of USD (e.g. Nyhus et al., 2003; Swenson and Andrén, 2005; Boitani et al., 2010; Yilmaz et al., 2015), to the extent that rising costs may compromise conservation funding for other endangered wildlife (Treves et al., 2009). Moreover, compensation payments do not necessarily increase local tolerance of large carnivores (Naughton-Treves et al., 2003; Nyhus et al., 2005) and the administration of funds is often flawed by mismanagement, thus undermining the strategy's conservation value (reviewed by Nyhus et al., 2005; Treves et al., 2009; Boitani et al., 2010). Any financial conservation instruments – be they compensation, communal livestock insurance schemes (Hussain, 2000; Johnson et al., 2006) or coexistence incentives (e.g. Mishra et al., 2003; Nistler, 2007; Zabel and Holm-Müller, 2008) - also need to ensure that revenues are distributed equitably and reach land managers mostly affected by conflict (reviewed by Nyhus et al., 2003; Dickman et al., 2011; Pettigrew et al., 2012).

It has also been argued that lethal carnivore control should be an available conflict mitigation option (e.g. Stander, 1990a; Linnell et al., 1997; Craven et al., 1998). From an economic viewpoint, and only where livestock losses can reliably be attributed to a particular carnivore, the targeted trophy hunting of such animals could realise substantial income (e.g. average leopard trophy value across Africa of $4,759, Braczkowski et al., 2015) to help offset local costs of coexistence. In support of this notion, nearly 50% of safari hunters expressed consent to target-hunt specific 'problem animals' at standard or increased trophy fees, even if this meant foregoing a more attractive trophy (Lindsey et al., 2006). The lethal removal of a particular livestock raider may indeed stop depredation (Skinner, 1976), but as with translocations, an important prerequisite for such focussed operations would be to ensure selectivity which, in terms of identifying the responsible offender, can be very difficult (Kerth et al., 2013), if not impossible (Linnell et al., 1999).

7.5 Key points

7. The average translocation cost for one large carnivore was $2,393 – when adjusted to include outcomes, the Individual Conservation Cost per individual was $5,983 (cheetah = $6,898; leopard = $3,140).
8. Large carnivore translocation costs are highly variable depending on the case-specific circumstances of events.
9. Tracking technology (GPS collars), release mode (soft releases) and long captive times were the most significant factors increasing translocation costs.
10. High translocation costs prohibit use of the strategy as a standard response to large carnivore conflicts.
11. Translocation costs were similar to compensation/tolerance costs – livestock guard animals were the cheapest non-lethal conflict mitigation strategy.
12. Translocation cost-efficiency can be improved by rigorous selection of suitable candidates and recipient areas.
13. Where translocation is necessary, a large proportion (65.4%) of costs can successfully be recuperated from a wide public conservation audience.

Chapter 8 – General Discussion

8.1 Outlook

The successes of large carnivore recovery and reintroduction that have recently been achieved in North America (Musiani and Paquet, 2004), Europe (Chapron et al., 2014) and Africa (Hayward and Somers, 2009) may contribute directly to restoration of ecosystem integrity (e.g. Licht et al., 2010; Ritchie et al., 2012), support income generation activities (Bangs et al., 2001; Hayward et al., 2007b), and establish important sub-populations of threatened species (Davies-Mostert et al., 2009, 2015). However, these achievements often go hand-in-hand with the re-emergence of human-carnivore conflict (e.g. Blanco et al., 1992; Cozza et al., 1996; Zimmermann et al., 2010), for example when reintroduced predators leave protected areas (Oakleaf et al., 2003; Smith and Bangs, 2009). Even sophisticated fencing cannot prevent outbreaks from conservation areas (e.g. Hunter, 1998b; Davies-Mostert et al., 2009; Marnewick et al., 2009). Such incidents and, more importantly, the accelerating overlap of human and large carnivore habitats throughout the world, suggest that conflict will perpetuate, and probably increase in frequency.

In Africa, human populations are growing fast, approximately by 4.8% annually (International Planned Parenthood Federation, cited in Aribisala, 2013), and indeed faster than anywhere else (United Nations, 2015). If estimates hold true and current trends persist, by 2050 every third child will be born in Africa (UNICEF, cited in Aribisala, 2013). Rising human densities directly threaten carnivore persistence (Woodroffe, 2000) mainly because associated land management transformations (e.g. through subsistence agriculture) compromise large carnivore habitats (Riggio et al., 2013) while continually increasing the interface of human-carnivore interactions. These typically manifest in livestock predation and/or jeopardised human safety (Inskip and Zimmermann, 2009). Even in sparsely populated countries like Namibia, conflict between people and carnivores is rife and frequently results in lethal control of so-called ‘problem animals’ (see chapter 3).

Translocation of large carnivores away from conflict areas has been, and continues to be, an often used response to mitigate conflict non-lethally. The strategy has been employed for a range of African species such as leopard (e.g. Hamilton, 1981; Stander et al., 1997a; Weilenmann et al., 2010), cheetah (e.g. Marnewick et al., 2009; Boast et al., 2015), and lion (e.g. Van der Meulen, 1977; Stander, 1990a). Beyond Africa’s boundaries, translocations are practised to manage offending tigers (Goodrich and Miquelle, 2005; Miller et al., 2011), leopards (e.g. Athreya et al., 2011; Mondal et al., 2013; Odden et al., 2014), bears (Shivik et al., 2011; Alldredge et al., 2015; Mukesh et al., 2015), jaguars

(Rabinowitz, 1986; Author unknown, 2015), wolves (Bradley et al., 2005), and mountain lions (Logan et al., 1996; Ruth et al., 1998). These examples by no means constitute a comprehensive list of large carnivore translocations around the globe. Until recently, however, research into large carnivores frequently overlooked applied conservation management topics (Balme et al., 2014) and rarely have managers evaluated translocations in necessary detail (Linnell et al., 1997; Fischer and Lindenmayer, 2000; Massei et al., 2010).

In an effort to improve our understanding of the strengths and weaknesses of translocation as a conservation strategy, the purpose of this work was to determine detailed biological and financial outcomes from intensive monitoring of 22 events carried out into free-range environments in Namibia between 2008 and 2012. Translocated subjects were monitored for as long as possible in order to enable a more comprehensive evaluation of success than previously available for free-range, conflict-mitigation and rehabilitation efforts (see partial assessments of Hamilton 1981; Stander, 1990a; Stander et al., 1997a; Weilenmann et al., 2010; Boast et al., 2015). This included a combined assessment of post-release biology, conflict, and homing, all of which are considered to be important measures of success (Linnell et al., 1997; Miller et al., 1999; Massei et al., 2010; Fontúrbel and Simonetti, 2011). Furthermore, observed success was calibrated against the cost of events to provide a replicable, pragmatic evaluation of cost-effectiveness, the Individual Conservation Cost, in relation to other available strategies (Chapter 7). The focus of this chapter is to provide a synopsis of this study's key findings, and critically assess the overall efficacy of large carnivore translocation by pinpointing specific problems associated with its use. This chapter also considers alternative approaches, suggests recommendations to improve translocation success and monitoring, highlights limitations, and outlines opportunities for additional research.

8.2 Variability: an important element of carnivore translocation outcomes

Carnivores respond differently to translocation. Hence, the results of conflict translocations generally show a high degree of intra- and inter-specific variability (Linnell et al., 1997; Miller et al., 1999; Massei et al., 2010; Fontúrbel and Simonetti, 2011). In previous studies variable outcomes have been attributed to a wide range of factors such as release mode, recipient area quality, climatic season, translocation distance, and a subject's sex and age as well as social status (pooled from Rogers, 1986a; Belden and Hagedorn, 1993; Riley et al., 1994; Bradley et al., 2005; Mukesh et al., 2015). Carnivore translocation outcomes may further be influenced by a species' sociality (Bradley et al., 2005; Gusset et al., 2006, 2010;

Trinkel et al., 2008; Somers and Gusset, 2009), an individual's familiarity with competing predators (Marnewick et al., 2009), pre-release management in captivity (Houser et al., 2011), and the unknown effects of stress and injury (e.g. Hamilton, 1981). Cumulatively, these parameters, and likely many more, define a complicated matrix that determines how an animal will be able to cope with a novel situation. While small sample sizes prevent fine-scale interrogation of observed results and the inter-play of influencing factors (further discussed in section 8.4), there seem to be few trends that enable managers to predict outcomes.

The present study is no exception to this rule. While soft release, an often-recommended approach to acclimatise individuals to recipient areas (Belden and Hagedorn, 1993; Hunter, 1998a; Purchase, 1998; Hayward et al., 2007b), did not improve cheetah site fidelity or survival (also see leopards in Hamilton, 1981), it did so during free-range translocations of wolves in the US (Bradley et al., 2005). Long-term captive rehabilitation proved an effective strategy for female leopards in Namibia (see chapter 4) and in Botswana (Houser et al., 2011) whereas cheetahs habituated to human management, the degree of which significantly influenced chances of survival (chapter 5). Therefore, what may work for one species may not be a suitable approach for another (also see Houser et al., 2011). The degree of post-release explorations also strongly varied across the focal species. With one exception (Pp15), all leopards (chapter 4) and the hyaena (chapter 6) settled within eight weeks of release. Conversely, cheetahs displayed roaming behaviour for much longer and several individuals did not establish recognisable ranges (chapter 5).

These observations demonstrate that there cannot be a standard recipe for successful large carnivore translocations. Possibly the only consistent outcome is that, given the choice, large carnivores typically leave the areas that were considered appropriate for their release (chapters 4-6; Hamilton, 1981; Rogers, 1986a; Ruth et al., 1998; Weilenmann et al., 2010), thereby exposing them to pre-translocation threats (chapter 5; Miller et al., 2011; Boast et al., 2015) and frequently resulting in repeat translocation of the same individuals (Riley et al., 1994; Stander et al., 1997a; Bradley et al., 2005). Low site fidelity, in particular, makes free-range translocations an inherently unreliable conservation tool as it affords managers little confidence in predicting outcomes, both in light of the animal's welfare and the potentially disastrous effect on land users outside PAs (Athreya et al., 2007, 2011).

8.3 Contributions to translocation knowledge

In previous free-range translocation studies involving large African carnivores, case-specific outcomes often remained unknown (e.g. Purchase, 1998) in part due to failure of tracking technology (Hamilton, 1981; Weilenmann et al., 2010) or a lack of direct follow-up (Boast et al., 2015) and reporting (Marker et al., 2003a). Some studies mentioned translocation events without any further reporting of their outcomes (e.g. leopards, Balme et al., 2009) or provided partial assessments. For example, in Botswana, causes of mortality could only be established for 27% (*n* = 3) of GPS-monitored cheetahs (*n* = 11) and translocation outcomes were assessed for only 28% (*n* = 11) of the 39 released subjects (Boast et al., 2015). In the past, few studies have attempted to link biological outcomes with conflict considerations to provide a combined assessment of translocation efficacy (Stander, 1990a; Weilenmann et al., 2010). In Zimbabwe, limited access to the areas from which study cheetahs were sourced prevented an evaluation of the conflict-reduction potential of translocations (Purchase et al., 2006). So, what can we learn from the monitoring of merely 30 study animals across three species in Namibia?

First, this work enabled a robust evaluation of translocation success for 90% (*n* = 27) of released adults (*n* = 30). This required on-going concerted efforts among the researchers, recipient area management and personnel, land managers outside PAs, other research organisations in Namibia, and the state wildlife department across seven years. Based on the available data, this study empirically confirms that, in general, large carnivore translocation success is low, i.e. 44% (*cf.* 42%, Fontúrbel and Simonetti, 2011). Success rates of free-range cheetah translocations (as measured by annual survival only: 50%, this study; 36%, Purchase and Vhurumuku, 2005; 18%, Boast et al., 2015) also appear considerably lower than for semi-controlled reintroductions (86%, Marnewick, 2009). Conversely, leopard survival was 83% and there was no difference in survival between translocated and resident study animals (chapter 4). This emphasises that translocation success can be highly situation- and species-specific (also see chapter 7 and section 8.2) and suggests that translocation policies/guidelines need to be formulated independently for different carnivore species and management approaches.

Second, the present study provides the first detailed cost accounting for large carnivore translocations based on true cost recording rather than partial and/or broad estimates of operational costs (*cf.* Riley et al., 1994; Purchase, 1998; Marnewick et al., 2009; Houser et al., 2011; Boast et al., 2015). This enabled a replicable measure of cost-effectiveness for different species and translocation scenarios (the ICC – see chapter 7), giving an overall cost of $5,983 per successful adult. As was the case for survival, costs significantly differed across species and scenarios (chapter 7). Documenting these financial implications in detail is important because different species require different management approaches (Hayward and Somers, 2009) and the projected costs of conservation actions influence the choice of management strategies for endangered carnivores (e.g. section 8.8; Wemmer and Sunquist, 1988; Derocher et al., 2013). The results, however, indicate that agencies tasked with the management of ‘problem carnivores’ may successfully recuperate almost 66% of all translocation costs from public support groups. Particularly for the most significant cost factor, i.e. tracking technology (56% of total costs), recuperation was high (77%).

This study also reports first evidence of breeding success in translocated leopards (chapter 4; *cf.* previously hypothesised by Hamilton (1981) and Houser et al. (2011)) and cheetahs (chapter 5) by following females for up to four years after release. These observations have strong implications because confirmed breeding events eventually compensated for initial mortalities, demonstrating that translocations can locally supplement and support free-ranging gene pools. This also challenges the rationale of assessing translocation success after 12 months (a frequently used time period) because the majority of cubs were born in years two and three after releases (see chapters 4 and 5). In addition, the risk of under-recording reproductive success emphasises the importance of linking telemetry-based monitoring with intensive field investigations for prolonged periods, not only to determine the fate of tagged subjects and the feasibility of management protocols (e.g. Houser et al., 2011) but also to ascertain reproduction, which is a key measure of success and efficacy (e.g. Marnewick et al., 2009). In support of this, the present study found that sub-adult cheetah cubs released alongside their mothers can successfully withstand the considerable risks and stresses associated with translocation and endure large-scale explorations in novel environments (see chapter 5).

In addition, this work describes the first detailed GPS-monitored account of a brown hyaena translocation (chapter 6), thereby providing a preliminary baseline against which to gauge future decisions. Similarly, chapters 4 (leopards) and 5 (cheetahs) contribute no less than 50% (n = 6; including short-distance event of Pp71) and 48% (n = 13) to all published GPS-monitored free-range translocations for these species respectively - leopard GPS sample data pooled from (Weilenmann et al., 2010; Houser et al., 2011, Mondal et al., 2013; Odden et al., 2014); cheetah pooled from (Houser et al., 2011; Boast et al., 2015). The leopards reported here constitute 20% of all detailed translocations assessed for the species (n = 35) and they were the first individuals to be monitored intensively for at least 12 months.

Only through intensive monitoring of released individuals will we be able to gain a better understanding of the manifold factors that influence translocation success (Linnell et al., 1997; Wolf et al., 1998; Miller et al., 1999; Massei et al., 2010). This is particularly important in light of the large degree of intra- and inter-specific variability observed in translocation outcomes (section 8.2). For example, this work shows that long-distance translocations coupled with temporary captivity can prevent successful homing of cheetahs, leopards and hyaenas (chapters 4-6). Of the 28 individuals that did not home, 11 were kept for periods shorter than 90 days, suggesting that translocation distance (and associated familiarity with environmental cues) is a better predictor of homing than captive time. This is supported by observations on black bears in North America where increasing translocation distance reduced homing incidents significantly (Rogers, 1986a). The low homing rate reported here strongly contradicts previous translocation studies of the same species (e.g. Hamilton, 1981; Stander et al., 1997a; Weilenmann et al., 2010; Boast et al., 2015) as well as reviews of the strategy (Linnell et al., 1997; Massei et al., 2010), which caution that homing is one of the key risks associated with translocations. Moreover, the results clearly demonstrate that conflict need not be a routine outcome of translocations into free-range systems, as relocated leopards did not cause higher levels of depredation when compared with resident conspecifics (chapter 4), only two cheetah case studies resulted in confirmed conflict (chapter 5), and the hyaena ceased raiding post-translocation (chapter 6).

Most importantly, however, the present work provides the first structured attempt to determine recipient area suitability for free-range large carnivore translocations (chapters 4 and 5). To date, release areas for ad hoc translocations have been selected haphazardly when the need arose, and typically without any formal feasibility studies and/or Environmental Impact Assessments (Purchase et al., 2006; Boast et al., 2015; Ropiquet et al., 2015). Conversely, recipient area evaluations have become an integral element of large carnivore reintroductions (Larsen and Ripple, 2006; Hayward et al., 2007c; Lindsey et al., 2011) because habitat suitability - with its various components such as prey availability, vegetation structure, human threats, intra-guild competition, etc. - is considered to be a strong predictor of translocation/reintroduction success (Griffith et al., 1989; Wolf et al., 1996, 1998; Macdonald, 2009).

At first glance, Namibia appears to provide excellent conditions for the management of large carnivores by way of free-range translocations. The country has a large PA network (>35% of surface area) and most unprotected parts have low human densities of less than 1.0 people per km^2 (Mendelsohn, 2006, Namibia Statistics Agency, 2012). Wild prey are abundant in both protected and farming areas (East and the IUCN/SSC Antelope Specialist Group, 1999; Barnes et al., 2009; Lindsey et al., 2013b) and the focal carnivore species appear to occur as panmictic populations (Marker et al., 2008b; Stein et al., 2012). The results obtained during suitability modelling, however, suggest that private and public PAs can support relatively few leopard translocations (approximately 87 initial events across 18 months – chapter 4) and none for cheetahs (see chapter 5). Therefore, the capacity for translocations seems far less than the potential demand arising from captures on free-hold lands (see chapter 3). This has strong implications for the future scale of use of translocations. Moreover, these considerations apply not only to Namibia and the focal study species, but to all large carnivores frequently involved in conflict (see Linnell et al., 2012) and to any country where translocation remains a legal management option.

8.4 Limitations and recommendations for additional research

A large amount of work still lies ahead for carnivore researchers and managers to evaluate free-range translocations in a truly comprehensive manner, as many questions remain to be answered.

For example, how do translocation events impact the source and recipient populations in terms of direct interactions between conspecifics (e.g. through aggression, competitive displacement and/or the risk of inducing infanticide, Rogers, 1986a; Balme and Hunter, 2013), local social dynamics (Balme et al., 2009), disease transmission (Woodford and Rossiter, 1993; Murray et al., 1999; Chipman et al., 2008), intra-guild competition (Marnewick et al., 2009), and of course at the genetic level (see Frankham, 2009; Ropiquet et al., 2015)? Any deleterious effects, albeit unintentional, could seriously compromise the well-intended objectives of conserving individual predators.

In the case of free-range translocations, rigorous evaluation of risks requires intensive pre-translocation surveys and simultaneous monitoring of conspecifics (and potential competitors) at both source and recipient locations. However, this appears logistically impossible, at least for source populations in Namibia. Conflicts are widespread across Namibia's free-hold commercial farming area (see chapter 3) and, therefore, managers cannot anticipate the geographic origin of translocation candidates. Additional translocation requests may arise for large carnivores causing damage on communal lands (Rust and Marker, 2013a). Namibia is estimated to support approximately 10,584 cheetahs, 18,031 leopards, and 1,503 brown hyaenas (median values from Stein et al., 2012) across about 559,715 km^2, 571,917 km^2, and 496,839 km^2, respectively. The size of potential source populations and the extent of their current distributions render invasive monitoring impractical, unfeasible, and unethical. A variety of non-invasive sampling methods are available for conservation genetics, disease profiling, and social population parameters (e.g. Gompper et al., 2006; Long et al., 2008; Busby et al., 2009; Kelly et al., 2012; Wengert et al., 2012). These, however, would provide little insight into the direct interactions between translocated and resident individuals, or between competitors, and country-wide surveys would have to be conducted prior to any further translocations to reliably rule out adverse effects.

Another important, and presently unaddressed, issue in translocation evaluations concerns the stress induced by such interventions. It is clear that anaesthesia, handling, transport, temporary captivity, and exposure to novel environments/conspecifics/competitors can cause significantly elevated levels of stress in large carnivores (e.g. Goymann et al., 1999; Terio et al., 1999; Wells et al., 2004; Young et al., 2004). Assessments of such stress (e.g. as indicated by suppressed reproduction, Jurke et al., 1997), have, however, been limited to captive scenarios. Due to the many potentially detrimental impacts of stress during translocations (reviewed by Teixeira et al., 2007; Dickens et al., 2010), future studies need to incorporate this topic into monitoring protocols.

Stress can be determined non-invasively from faecal samples by analysis of glucocorticoids and their metabolites (Monfort et al., 1997; Schatz and Palme, 2001; Young et al., 2004; Keay et al., 2006). This requires both detection of samples in vast, remote, unobstructed landscapes and reliable attribution of samples to specific individuals. Wasser et al. (2004) describe the usefulness of trained scat detection dogs under similar circumstances (sympatric occurrence of different bear species in free-range environments in Canada) and molecular DNA profiling can be employed to ascertain identity (Farrell et al., 2000; Busby et al., 2009; Laguardia et al., 2015). Comparison of pre-translocation baseline stress values with post-translocation essays could possibly help explain the outcomes of cases with speculative mortality reasons (see cheetah females Aju07/18 in chapter 5). During soft-releases, it may also provide a measurable approach to determine when settling and acclimatisation with the recipient area have occurred.

The sensory mechanisms that enable large carnivores to home, sometimes over distances exceeding 220 km (black bears, Rogers, 1986a), require further study and consideration during planning of events. The importance of familiar topographic features for navigation towards a home site has been described for other animal taxa (e.g. reptiles, Emlen, 1969; birds, Braithwaite and Newman, 1994; insects, Capaldi and Dyer, 1999). It has been proposed (Linnell et al., 1997) and refuted (Rogers, 1986b) for large carnivores, but to date without any rigorous assessment. Caldwell and Nams (2006) hypothesise that preventing line of sight detection of familiar marks reduces an animal's ability to orient its movements (also see Appendix 17 for leopards Pp15/57/71). In another Namibian study, leopards released into familiar terrain invariably homed and homing time was predicted by translocation distance (Stander et al., 1997a).

Other limitations of the current study stem from the restricted sample sizes (chapters 4-7). These frequently prevented testing for the effects of multiple explanatory variables and their interactions (see chapter 7) or differences between treatment groups (see chapter 6). Therefore, the results presented here are preliminary, although they do provide much needed baseline data for the focal species (section 8.3).

Our knowledge of translocation outcomes and their influencing factors certainly would benefit from meta-analysis of published events. This is currently impossible due to significant data deficiencies. We can use the leopard as an example. For the 35 translocated individuals with some available information (pooled from this study; Hamilton, 1976, 1981; Stander et al., 1997a; Hayward et al., 2006a; Weilenmann et al., 2010; Houser et al., 2011, Mondal et al., 2013; Odden et al., 2014) relevant data to evaluate success frequently are not reported. For example: survival at 12 months (20% deficient); reproduction (80% deficient); prey (63% deficient); post-release conflict (49% deficient); settling behaviour (34% deficient). In addition, important protocol details such as captive time, translocation distance and/or body measurements often are not stated (or provided inaccurately), thus preventing more detailed assessments and analyses of pooled samples.

To address these shortcomings, Open-Access publication of any translocation events (success and failures alike) involving large carnivores in free-range systems is strongly recommended. Such feedback-loops were implemented for large carnivore reintroductions in South Africa (see Hayward et al., 2007a, 2007b) and over time can provide sufficient cumulative information to review the efficacy of conservation interventions (Breitenmoser et al., 2001; Hayward and Somers, 2009). Furthermore, to enable integrated analyses of all monitored events in the future, translocation meta-databases need to be established for each species under the auspices of the IUCN Reintroduction Specialist Group, or as part of the proposed Global Large Carnivore Initiative (Ripple et al., 2014). Pooled samples with detailed data would enable analysis with stronger multi-variable paradigms (see Gusset et al., 2008, 2009a for African wild dog reintroductions) for individual species or taxonomic groups.

The present work also largely relies on retrospective observations that were significantly influenced by the type of tracking technology deployed on different subjects.

Taking into account the quality of data that were available to evaluate outcomes (e.g. see difference in site fidelity estimates for cheetahs in Table 5.5) future monitoring of free-range translocations needs to be based on GPS-technology rather than VHF-only radio transmitters. Primarily due to the extensive post-release movements displayed by nearly all subjects in the present study (chapters 4-6), and observed elsewhere for large carnivores (e.g. Rogers, 1986a, 1986b; Ruth et al., 1998; Bradley et al., 2005; Goodrich and Miquelle, 2005; Weilenmann et al., 2010), VHF-based monitoring appears to be insufficiently accurate for the tracking of far-ranging species in unconfined environments. The limited detection range of VHF-telemetry diminishes the chances of reliable follow-up (also see Hamilton, 1981).

Finally, the estimates provided for potential cheetah and leopard releases in Namibia (chapters 4 and 5) are limited by the input data available for modelling. Reliable country-wide assessments of carnivore prey are needed to supplement the predictions presented here (*cf.* Winterbach et al., 2015). Modelling should be repeated every two years to incorporate updated information of the various model input parameters as well as any data from translocation events. In addition, the Namibian population estimates available for the focal species were subject to large CIs (Stein et al., 2012) and, therefore, it remains uncertain how persecution on commercial farmlands (chapter 3) and predictions of translocation suitability (chapters 4 and 5) relate to the free-ranging gene pools of these species.

8.5 Challenging the rationale of conflict translocations

There can be no doubt that large carnivores predate on livestock in many areas of the world (Kaczensky, 1999; Sillero-Zubiri and Laurenson, 2001; Graham et al., 2005; Baker et al., 2008; Inskip and Zimmermann, 2009). A recent review of the topic found that no less than 24 species (including felids, canids, ursids, mustelids, and hyaenas) are regularly involved in livestock depredation (Linnell et al., 2012). Therefore, the decision of whether to manage conflict by way of translocation applies to all habitats where large carnivore distributions overlap with commercial production of potential prey (livestock and/or game).

The rationale for conflict-mitigation translocation directly hinges upon the concept of persistent ‘problem animals’, i.e. those that preferentially (and habitually) hunt commercial stock, thereby causing unacceptable levels of local damage (Linnell et al., 1999). This may happen when physical ailments prevent carnivores from pursuing alternative (wild) prey (Rabinowitz, 1986; Farhadinia et al., 2015) or domestic animals remain the only available choice in human-modified landscapes (e.g. Athreya et al., 2014;

Shehzad et al., 2015). Carnivores may occasionally also exhibit surplus killing, i.e. the killing of large numbers of livestock or wild prey in a single event (Kruuk, 1972; Andelt et al., 1980; Horstman and Gunson, 1982; Miller et al., 1985; Stuart, 1986; Kossak, 1989; DelGiudice, 1998; Short et al., 2002; Wiesel, 2010) and in amounts larger than immediately consumable and necessary to meet energy demands. Although surplus killing probably reflects adaptive predatory behaviour rather than proof for a habitual preference (see Linnell et al., 1999), such events, understandably, can cause intense frustration among livestock and game producers, as they appear excessive and unnecessary.

For translocation to be a feasible management option, the removal of one specific raider (or a small group) needs to have a reasonable probability of stopping conflict despite the continued presence of non-offending conspecifics (Linnell et al., 1997). For example, in South Africa, depredation ceased following lethal control (resulting in the same effect as successful translocation, i.e. permanent removal from the site of conflict) of a single brown hyaena that had attacked over 130 smallstock in only four months (Skinner, 1976). The key element and core assumption of a successful conflict-mitigation translocation is that it must target the raider responsible (Hamilton, 1981). Therefore, it is imperative to identify these individuals reliably, if indeed they exist (Linnell et al., 1999). This is particularly important when considering the limited availability of suitable recipient areas for large carnivore translocations (e.g. chapters 4 and 5; Lindsey et al., 2009b). In Namibia, Stander (1990a) offered a behavioural differentiation between habitual and opportunistic stock-raiding lions based on their detailed pre-translocation histories that were established through intensive monitoring and continued correspondence with the local farming community. He suggested that opportunistic raiders could be managed successfully by moving them back to their original home ranges in Etosha NP while proposing lethal control of habitual stock marauders.

The question arises whether managers always have sufficient data on conflict involvement of specific carnivores to make an informed decision, or whether many events involve indiscriminately trapped individuals without confirmed conflict backgrounds, thereby resulting in erroneous translocation. In the present study, low post-release conflict levels suggest that most subjects were only opportunistic livestock predators, or had no conflict background. Conflicts also resumed on the majority of source properties after the removal of suspected raiders (also see Weilenmann et al., 2010; Boast et al., 2015). A review of livestock predation accounts found no strong evidence for existence of individual ‘problem animals’ (Linnell et al., 1999), thereby challenging the rationale for conflict-mitigation translocation. This is supported by the fact that detailed conflict studies demonstrate direct links between livestock predation levels and stock protection efforts (e.g. Ogada et al., 2003; Woodroffe et al., 2007; Stein et al., 2010; Rust et al., 2013; McManus et al., 2014). Logically, some carnivores (although usually only a small fraction of a local population, Claar, 1986; Stander, 1990a; Mizutani, 1993) will continue to predate on stock animals after removal of a conspecific if they are not sufficiently deterred from doing so (Linnell et al., 1999). Hence the occurrence of conflict is a function of opportunity that is mainly predicted by the availability of sufficient wild prey (Khorozyan et al., 2015) rather than by specialised behaviour of certain predators (Linnell et al., 1999, 2012).

In addition to the uncertainty around specific ‘problem animals’, translocation also imposes the risk of exacerbating existing conflict through perturbation of local carnivore ecology. First, disturbance of social structures at conflict sites, as is effected by the sudden removal of a conflict carnivore, leads to rapid re-colonisation because conspecifics from neighbouring territories quickly settle into vacant habitat patches. For example, badger (*Meles meles*) culling operations in the UK lead to localised increases in subordinate individuals (e.g. Tuyttens et al., 2000; Carter et al., 2007). In territorial large felids, rapid re-colonisation of vacant ranges (usually within weeks of the death/removal of a territorial individual) by young adults (typically dispersing males) has been documented in leopards (e.g. Athreya, 2006; Balme et al., 2009), jaguars (Rabinowitz, 1986), and mountain lions (Linnell et al., 1996). Here, resident male leopard Pp62, a known opportunistic livestock raider, responded to the killing of a territorial neighbour by expanding his home range by over 80% immediately (see chapter 4). Therefore, the indiscriminate removal of a non-offending predator or opportunistic livestock raider through translocation could in fact encourage immigration of several other conspecifics with unknown conflict histories, thus increasing the probability of conflict. Furthermore, local carnivore densities at removal sites may rise when persecuted carnivore species offset adult mortality through compensatory recruitment that could lead to expanding populations (e.g. Minnie et al., 2016). In combination, the potentially significant disturbance of social structures and spatial dynamics at translocation source sites may increase the likelihood of conflict rather than alleviating it effectively and/or permanently.

Correspondingly, the effect of removing a predator (dead or alive) on stock safety generally appears short-lived. To date, only few studies have empirically confirmed significantly reduced livestock conflicts following translocation of raiders (e.g. Stander, 1990a), one of which employed preventative translocation of bears before depredation could occur (Armistead et al., 1994). There is increasing evidence that translocations only temporarily resolve depredation issues if no further actions are taken (this study; Stander et al., 1997a; Weilenmann, 2010; Boast et al., 2015). Consequently, translocations would need to be carried out indefinitely and thus could locally induce source-sink dynamics. The effect of sustained removal of large carnivores from resident populations can seriously compromise population viability at local and regional levels (e.g. Knight et al., 1988; Woodroffe and Ginsberg, 1998; Novaro et al., 2005; Woodroffe and Frank, 2006; Balme et al., 2010; Andreasen et al., 2012; Bauer et al., 2014). Similar to the results from this study (see chapters 4 and 5), the median interval between repeat large carnivore translocations in Botswana was only two years and 85% of land managers considered repeat translocations despite there being little indication of their effectiveness (Boast et al., 2015). In South Africa, continued requests for translocations of endangered cheetahs from farmlands resulted in cessation of a successful relocation programme because it was estimated that removals might affect as much as 25% of the remaining free-ranging population (K. Marnewick, pers. comm. 2015).

8.6 Improving identification of conflict-causing carnivores

Considering these risks and the general lack of evidence for existence of specific ‘problem animals’ (Linnell et al., 1999), it should become a mandatory step to ascertain that depredation can reliably (and consistently) be attributed to a particular individual (or group) before deciding whether translocation presents a feasible mitigation approach. This would ensure selectivity in the local management of large carnivores, prevent unjustified removals purely based on anecdotal information, and considerably increase chances of success while acknowledging the limited capacity for translocation (chapters 4 and 5). The risk of accidental trapping is particularly high for scavenging carnivores that are attracted to carcasses and baited live traps. The diagnostic methods available to carnivore researchers and land managers have improved significantly and should be utilised to their best potential, particularly in countries like Namibia where multiple taxonomically similar carnivores occur sympatrically and leave potentially ambiguous predation signs at carcasses (Linnell et al., 2012).

Traditionally, skilled examiners had to rely on assessment of spoor, bite marks/locations, claw patterns, and signs of specific killing techniques found on carcasses to identify the species responsible for depredation (Linnell et al., 2012). These indicators, however, usually allowed little insight into the discrete identity of specific conflict-causing animals. Recent development of molecular DNA profiling from saliva found in bite wounds can supplement and improve assessments by enabling objective identification of raiders at the sex and individual level (Ernest and Boyce, 2000; Williams et al., 2003; Williams and Johnston, 2004; Blejwas et al., 2006; Mumma et al., 2014; Harms et al., 2015). In addition, some carnivores mark large prey items by defecation around kill sites (e.g. leopard, Eisenberg and Lockhart, 1972; mountain lion, Lang, 2001). Hence, DNA extracted from faeces could help elucidate identity (e.g. Farrell et al., 2000; Laguardia et al., 2015). In areas where large carnivores are tagged for monitoring purposes, regular data sharing with land managers enables evaluation of kill sites (present study) by investigation of GPS clusters that indicate prey locations (e.g. Knopff et al., 2009; Bacon et al., 2011; Krofel et al., 2013; Svoboda et al., 2013; Blecha and Alldredge, 2015).

The feasibility of identifying individuals from species with unique markings by means of coat pattern differentiation from wildlife camera footage has been demonstrated for many large carnivores (e.g. jaguar, Silver et al., 2004; cheetah, Marnewick et al., 2008; brown hyaena, Thorn et al., 2009; tiger, Karanth et al., 2011; leopard, Carter et al., 2015). Therefore, if livestock carcasses can be found quickly, offending individuals can potentially be recorded during repeat feeding bouts. Motion triggered wildlife cameras are now widely available and provide a cost-effective method of monitoring. Similarly, robust footprint identification methods have been developed for discrete identification of individuals from some carnivore species (e.g. tiger, Riordan, 1998; Sharma et al., 2005; Gu et al., 2014; mountain lion, Jewell et al., 2014; cheetah, Jewell et al., 2015; brown bear (*U. arctos*), Petridou et al., 2008). Where substrate permits, these novel technologies as well as traditional tracking skills (see Stander et al., 1997b) can be employed to aid rigorous identification of livestock raiders.

Importantly, and regardless of the methods used, results need to be assessed in a way that acknowledges the risk of confusion between actual predation and scavenging events (Linnell et al., 2012) to avoid erroneous over-representation of scavengers such as the hyaenas. Rapid carcass detection is paramount to minimise the possibility of obliteration or contamination of important indicators. This requires frequent herd controls and can be improved by use of trained carcass detection dogs. In addition, managers can deploy 'mortality signal emitting telemetry gear' on those livestock considered most vulnerable to depredation, e.g. calves and smallstock ranging on extensive pastures (e.g. Bjärvall and Franzen, 1981; Warren et al., 2001; Oakleaf et al., 2003; Knarrum et al., 2006). Detecting predation events quickly will also allow managers to implement additional anti-predatory measures (see section 8.7 below) before losses become substantial. However, managers should generally leave carcasses at kill sites as their removal can stimulate repeat depredation events in the same area when predators compensate for lost food (Krofel et al., 2012; Linnell et al., 2012).

8.7 Alternatives to translocation

Today, conservationists in the public and private sectors can chose from a dynamic, and continually diversifying, repertoire of large carnivore conflict management techniques (see synopsis of non-lethal methods in Table 8.1). These non-lethal predator damage mitigation methods either aim at changing human behaviour in terms of asset and carnivore management, reducing the likelihood of damage occurrence, or inducing some form of interference with natural predatory behaviour (Shivik, 2004). None of the available options comes without limitations and previous studies have emphasised that there cannot be a single, stand-alone panacea to conflict resolution, both in terms of effectiveness and cost-efficiency (e.g. Franklin and Powell, 1993; Nyhus et al., 2003; Treves and Karanth, 2003; Shivik, 2004, 2006; Bangs et al., 2006; Reinhardt et al., 2012). Depending on the nature and scale of local conflicts, the carnivores involved, and the legal provisions governing carnivore conservation, managers will need to adopt a flexible, situation-specific, and multi-facetted approach to deal with the challenges of human-carnivore coexistence (Treves and Karanth, 2003; Inskip and Zimmermann, 2009). Comprehensive conservation plans need to make provisions for selective translocation (e.g. Stander, 1990a; Goodrich, 2010), but only as a last-resort option.

The importance of utilising pro-active conflict prevention measures cannot be over-emphasised. Instead of relying on symptomatic, reactive treatments (i.e. translocation and compensation), land managers can significantly reduce losses by adjusting their stock husbandry (Ogada et al., 2003; Woodroffe et al., 2007; Stein et al., 2010; McManus et al., 2014).

In Namibia, several non-lethal methods have been tested with encouraging results (Marker and Boast, 2015). Livestock farmers that employed at least one protective measure on average reported 85% less losses when compared with farmers who did not protect their stock (Stein et al., 2010). Suitable methods included kraaling of livestock (see Appendix 18), herders, keeping livestock near human habitation, maintenance of a synchronised calving season, separation of vulnerable stock during calving periods, as well as guard animals (Stein et al., 2010). The latter can reduce livestock predation immediately and with prolonged effect (e.g. donkeys, Marker, 2000; guard dogs, Marker et al., 2005b; Potgieter et al., 2013, 2015). For example, of 44 commercial livestock ranchers, 37 (84%) experienced losses to predators prior to use of livestock guarding dogs (Potgieter et al.,

Table 8.1 - Synopsis of non-lethal large carnivore conflict management approaches. *Adopted from Shivik (2004).*

Purpose	Method (example)[a]
Altering human behaviour	
	Niche marketing (predator-friendly beef at adjusted production Cost)
	Compensation (ex-post damage payment)
	Insurance (annual livestock head premium toward communal Compensation fund)
	Incentive schemes (revenue sharing from carnivore-related income such as photographic or hunting tourism)
	Zoning (selection of farming areas with least predation risk; buffer zones between carnivore habitat and farming)
	Tolerance payment (financial reward to encourage predator presence and tolerance of expected damage)
	Research and public outreach (dissemination of monitoring data and depredation facts from tracking conflict carnivores)
Altering asset husbandry	
	Animal armour (neck protection collars)
	Herding/vigilance (human supervision; additional herders)
	Fencing (exclusion through electrified barriers or swing gates)
	Night and seasonal enclosures (thorn-bush kraals/bomas, portable electric pens)
	Time of breeding (risk reduction through synchronised calving and lambing)
	Selective pasturing, lambing, and calving (avoiding high risk Areas)
	Altering herd composition (inclusion of aggressive breeds)
	Sanitation (reducing attraction to farming areas)
Altering predator behaviour	
Primary repellents	Disruptive stimuli
	Visual stimuli (motion-triggered flashing lights)
	Noise (motion-triggered sirens and propane gas exploders)
	Electronic guards (electric fencing)
	Fladry (flag-fencing barriers)
	Chemical repellents (toxic neck collars)
	Biological odour repellents (scent mark bio-boundary)
	Disruptive harassment (human presence, disturbance of active Dens, chasing from livestock areas)
	Guard animals (dogs, donkeys, llamas, alpacas)
Secondary repellents	Aversive stimuli
	Aversive harassment (sub-lethal ammunition)
	Conditioned taste aversion (laced asset bait inducing sickness)
	Electronic training collars (Geo-boundaries programmed into collars that elicit shock upon boundary breaches)
	Diversionary feeding, altering prey populations (provision of buffer prey or alternative feeds to divert predation pressure)
	Reproductive inhibition (neutering, castration, hormonal inhibition)

[a] Supplemented with information from: Smith et al., 2000; Nyhus et al., 2003; Ogada et al., 2003; Asheim and Mysterud, 2004; Bangs et al., 2006; Schumann et al., 2006; Shivik, 2006; Zabel and Holm-Müller, 2008; Stein et al., 2010; Jackson et al., 2012.

2015). Once integrated, 92% of dogs either reduced or eliminated livestock predation on the sampled ranches (Potgieter et al., 2013), thereby significantly decreasing the proportion of affected ranchers to 34% (Potgieter et al., 2015).

The applicability of specific measures will strongly depend on the size of management units. Where landscapes rather than individual properties are of concern, carnivore managers can promote tolerance through financial incentives that compensate for coexistence risks rather than de facto damage (e.g. Zabel and Holm-Müller, 2008). In addition, zoning within coexistence landscapes can help delineate specific livestock and/or carnivore management areas according to conflict risk as predicted by carnivore occurrence (Breitenmoser et al., 2012; Linnell et al., 2012). Such efforts can be supplemented with livestock insurance schemes (Hussain, 2000; Aryal et al., 2014) for highest risk areas. With global trends toward eco-friendly products and sustainable production (Kasriel-Alexander, 2015), consumers are also increasingly willing to pay premiums for certified carnivore-friendly meat to reimburse local farmers for predation-induced losses. In turn, farmers agree to refrain from carnivore persecution and employ best practice stock husbandry (Table 8.1) (Predator Friendly, 2015). Commercial brands such as *Predator Friendly* have been registered as early as the 1990s (Predator Friendly, 2015) and are actively marketed through environmental NGOs (e.g. Keystone Conservation, 2015).

As regards conflict-related translocations, and considering their high cost factor at ICC rates, the strategy should probably only be used when other non-lethal mitigation strategies have already been exhausted. Moreover, it should be reserved for highly endangered carnivore populations (e.g. Goodrich and Miquelle, 2005) for which the value of an individual to the remaining gene pool justifies significant expenses and efforts. As local large carnivore populations continue to decline, individuals become more important to species persistence (Christie, 2009). Therefore, the cost-benefit ratio of translocations correlates with the conservation status of each species/population in question (Fig. 8.1 from Shivik, 2004). The decision of whether or not to employ translocation should predominantly be guided by this rationale.

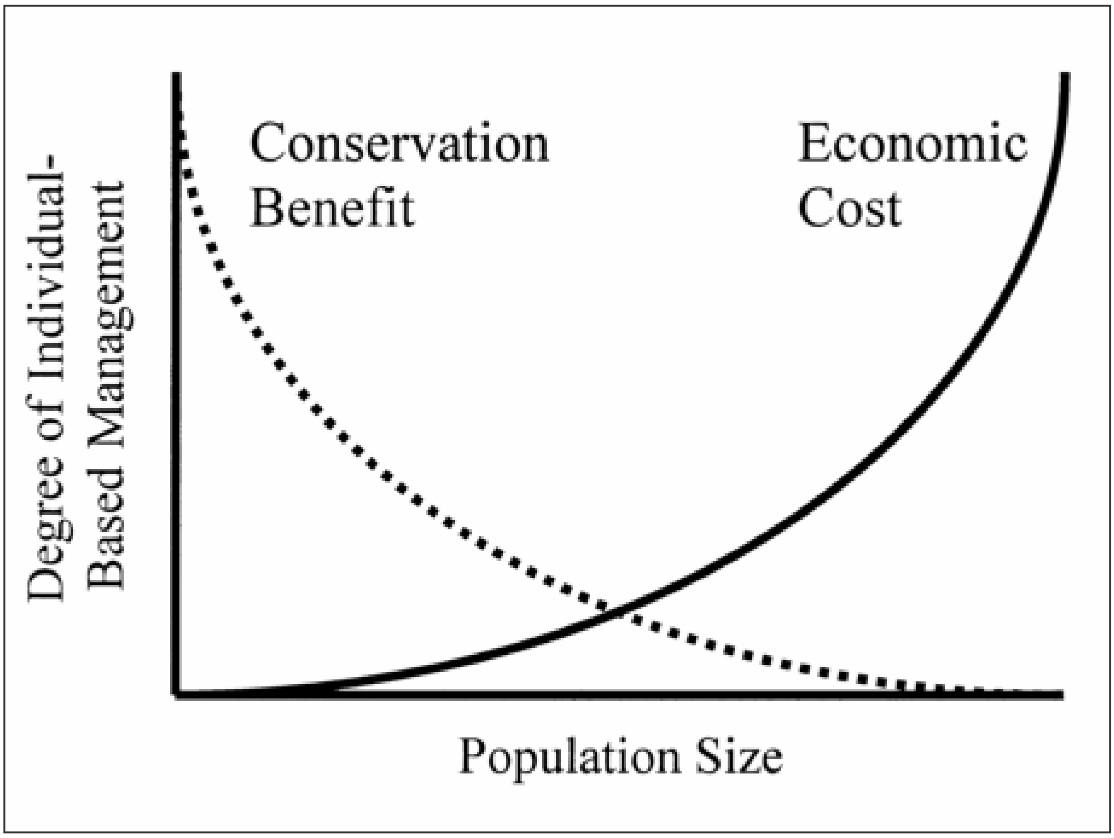


Figure 8.1 – Conceptual cost-benefit model of individual-based predator management by conservation need. *Source: Shivik (2004). When predators are rare and highly valuable (in terms of genetics and conservation), the cost-benefit ratio favours the use of individual-based, non-lethal methods. When predators are abundant and not as valuable individually, the costs of individual-based management approaches rises dramatically, altering the cost:benefit ratio away from efficiency of individual-based management techniques.*

8.8 Costs implications

From a conservation perspective, investment into conflict prevention should take priority over symptomatic mitigation of damage. At Namibian cost rates, for example, the median translocation cost per carnivore ($2,393) would pay for the placement of over three livestock guarding dogs (Potgieter, 2011) or enable employment of nearly two additional livestock herders for one year (Rust and Marker, 2013a). Alternatively, one translocation would cover the purchase of 7 – 15 head of preferred buffer prey of cheetah and leopard - at median auction values of oryx, greater kudu, common impala, red hartebeest, springbok (Erb, 2004) – to replenish depleted wildlife stocks locally or divert predation pressure on conflict farms. The cost of one translocation would also suffice to erect 1.06 km of electrified game proof fencing with swing gates for non-lethal exclusion of cheetahs and leopards from commercial game farms (Schumann et al., 2006), which represented the most antagonistic land use category in this study (see section 3.3.4 in chapter 3). Recent research in South Africa has shown that stock protection can be cheaper than continued carnivore persecution (McManus et al., 2014). In addition to ethical considerations, this provides conflict-affected farmers with a financial incentive to adopt preventative action in the future.

Based on the available literature (for any non-lethal mitigation strategy with ≥10 published cost estimates, see chapter 7), the purchase of a suitable livestock guard animal (dog, donkey, llama (*Lama glama*), or alpaca (*Vicugna pacos*)) appeared to be much cheaper than either translocation or compensation (chapter 7). Where introduced successfully, guard animals can decrease livestock losses to predators by over $1,000 per property per year (Green et al., 1984; Franklin and Powell, 1993; Coppinger et al., 1998; Andelt and Hopper, 2000; Gese, 2003; Rust et al., 2013), thus quickly recapturing their purchase and annual maintenance costs. However, the cost-effectiveness of livestock protectors varies significantly across users (Smith et al., 2000; Andelt, 2001, 2004; Gese et al., 2005; Potgieter et al., 2013). Guard dogs also require special training and food (Smith et al., 2000; Rigg, 2001). In addition, guard animals sometimes ignore, harass, or injure the livestock they are supposed to protect (reviewed by Andelt, 2001, 2004; Rigg, 2001) and may occasionally kill the predators they are expected to deter (Marker, 2000; Potgieter et al., 2015). They stand little chance of defending domestic animals against group-hunting large carnivores such as the wolf, African wild dog, lion or spotted hyaena. Where social carnivores are the primary concern, livestock producers can improve stock protection with low-cost thorn-bush kraals (Ogada et al., 2003; Woodroffe et al., 2005b) or innovative deterrence tactics, including a variety of aversive and disruptive alarm systems (Pfeifer and Goos, 1982; Breck et al., 2002; Gegner, 2002; Bangs et al., 2006) (Table 8.1). Non-consumptive local tourism ventures have the potential to outweigh financial losses from livestock depredation significantly – in Namibia, the monies earned from tracking known conflict leopards with international guests exceeded losses 12-fold and considerably contributed to rural income generation (Stander et al., 1997a).

Reducing the costs incurred from stock losses is one of the most important elements of assuring the long-term persistence of large carnivore populations on free-hold, commercial farmlands in Namibia (Marker et al., 2007; Lindsey et al., 2013a). Because translocations do not solve conflicts permanently (chapters 4 and 5; Stander et al., 1997a) cumulative costs will arise from the need of repeat events (also see Riley et al., 1994; Bradley et al., 2005; Boast et al., 2015), from losses occurring after first translocations or the need to employ alternative conflict management measures. This provides a strong financial argument against the widespread and sustained use of translocation for conflict mitigation. Where they need to be carried out, the number of events should be limited in accordance with available recipient habitat. Selective translocations should also be linked to the condition that stock protection is improved at the source site to minimise the likelihood of recurring conflict.

8.9 Improving planning and outcomes

There are many ways in which carnivore translocation planning and efficacy can be improved. Frameworks that outline critical pre-translocation considerations have been published (see Appendix 31 from Massei et al., 2010; also see Miller et al., 1999; Rout et al., 2005) and provide a baseline for managers/agencies to evaluate decisions prior to any actions. Even before such case-sensitive decision instruments come into play, government wildlife departments need to determine whether translocation is considered a feasible option and, if so, under which circumstances. Bayesian networks provide powerful assessment tools to synthesise available stakeholder expertise while defining associated risks and challenges objectively (see Johnson et al. (2010) for modelling of cheetah relocation success in southern Africa). For conflict-related carnivore translocations, it will be imperative to move away from the typical trial-and-error approach that restricts analyses to post-hoc observations and instead base decisions on best predictions.

In this study, suitability modelling of potential recipient areas showed that the capacity for future cheetah and leopard translocations across Namibia differs (*cf.* chapters 4 and 5). *CaTSuiT* (Lemeris Jr., 2013, 2015) provides a flexible, integrative tool to increase the probability of success. This simple Open-Access modelling system (leopard iteration, Lemeris Jr., 2015) allows managers to incorporate past translocation experiences while defining any parameters considered important to improve translocation efficacy (e.g. prey, habitat structure, density of conspecifics and competitors, translocation distance etc) for a specific locality, event and/or species.

There are other tools and mechanisms that can strengthen predictions and should be used in combination with analysis instruments such as *CaTSuiT*. Strategic recipient area planning (also see Lindsey et al., 2009b for cheetahs in South Africa) needs to acknowledge genetic variants at regional scales (Ropiquet et al., 2015) while reliably forecasting prey suitability (see Hayward, 2009) in conjunction with local habitat conditions (Lindsey et al., 2011) to determine appropriate carrying capacities (Hayward et al., 2007c). There also is increasing knowledge of the importance of local habitat structures for large carnivore dispersal movements (Fattebert et al., 2015), post-release home range establishment and habitat selection (e.g. Skinner and van Aarde, 1987; Purchase and du

Toit, 2000; Rostro-García et al., 2015; Welch et al., 2015), reproduction (Durant, 1998a, 1998b), hunting strategies (Belden and Hagedorn, 1993; Muntifering et al., 2006; Bissett and Bernard, 2007) and, therefore, also likelihood of conflict (Minnie et al., 2015). Integrating habitat components into modelling can improve release site selection for future translocations and thereby successful settling. Matching source and recipient site habitats closely may decrease post-release explorations that typically lead large carnivores beyond the boundaries of targeted recipient areas (see chapters 4-6; Hamilton, 1981; Weilenmann et al., 2010; Boast et al., 2015).

The question, however, is not only where to release, but also whom? In terms of candidate individuals, the hypothesis is that young adults may be better suited to the challenges of translocations than older individuals because they show reduced tendency to home while having similar or higher survival rates (e.g. Ruth et al., 1998; Bradley et al., 2005; Alldredge et al., 2015). Natural dispersal exposes pre-prime individuals to challenges induced by translocation, i.e. novel environments with associated risks of competition and safety and unknown distribution of important resources such as food, shelter, and water. Sub-adult males in particular typically disperse over long distances from maternal home ranges prior to assuming territorial tenure (e.g. cheetah, Caro, 1994; leopard, Fattebert et al., 2013; lion, Schaller, 1972; Pusey and Packer, 1987; Funston, 2011; mountain lion, Sweanor et al., 2000; Thompson and Jenks, 2005; Elbroch et al., 2009; tiger, Smith, 1993; Gour et al., 2013; Wang et al., 2015). However, long-distance dispersal is not exclusive to males (e.g. Weaver et al., 1996; Stoner et al., 2008). A priori, we can expect young adults to be better suited to the challenges of translocation rather than individuals that have already established territoriality. Once translocation meta-databases have been created for the various species (see section 8.4), this assumption should be tested rigorously.

In a nutshell, the ecology of large carnivores differs significantly across species and life stages. Therefore, it is essential that assessments and evaluations are made species- and case-specific (also see Hayward and Somers, 2009), and should consider the diversity of life history traits, variable social systems, and animal behaviour. The key consideration during all large carnivore relocation attempts (be they free-range translocations or reintroductions) should be that apex predators require large areas in which threats to their occurrence need to be mitigated successfully prior to any releases (Macdonald, 2009; Stoskopf, 2012).

Where free-range translocations proceed, strategic data sharing with land managers outside PAs should be considered to evaluate outcomes jointly and to improve transparency of events. In this study, land managers voluntarily reported details of conflict involvement, reproduction, and prey selection, while some managers used tracking information for their tourism enterprises. When study carnivores were persecuted purposefully, this happened during explorations and invariably on properties that had not (yet) received monitoring information, or when livestock predation became an issue (chapter 5). Farmers with regular access to collar data did not persecute carnivores and appreciated regular updates.

8.10 Conclusions

Large carnivores are amongst the most difficult animal taxa to conserve because the cost of coexistence usually outweighs the benefits accrued from their presence. Despite the availability of cost-effective damage prevention measures, conflict with and persecution of large carnivores remains a widespread problem across Namibia's commercial free-hold farmland.

Except for lethal control, the translocation of unwanted or conflict-causing carnivores is probably the most invasive form of management intervention that conservationists can attempt. In the case of large carnivores, it places highly complex organisms into an unfamiliar and challenging environment. Consequently, translocation involves manifold risks (for animals and people alike) and caveats that may compromise its efficacy (Linnell et al., 1997; Wolf et al., 1998; Miller et al., 1999; Massei et al., 2010).

This study demonstrates that translocation can successfully conserve individual carnivores (from different species and under specific circumstances) without causing significant post-release conflicts. Homing can be prevented by moving carnivores over long distances. Following an initial period of orientation and exploration, the ecology of translocated large carnivores reflects that of resident conspecifics. Successful breeding contributes to supplementation and maintenance of free-ranging gene pools. There also is strong public support to finance costly translocations.

The results, however, show a high degree of inter- and intra-specific variability and there are no predictable translocation outcomes, making the strategy an unreliable tool. Extensive movements away from target recipient PAs present a serious concern as they expose carnivores to pre-translocation threats. Hence, rigorous candidate and recipient area selection are imperative to maximise the strategy's efficacy. Due to its symptomatic nature, considerable initial mortality, and the limited availability of suitable recipient areas, translocation of unwanted or damage-causing carnivores cannot serve as a standard response to human-predator conflicts. It can delay conflict locally but does not solve problems permanently, thus resulting in repeat requests for carnivore removals.

With species extinctions reaching unprecedented scales (Pimm et al., 1995), however, translocation should be available to carnivore managers as a last-resort - a supplementary tool, albeit with curtailed use, with clearly defined objectives, and with rigorous rules in place to limit the number of events in accordance with local capacities. Translocation is best suited to conserve individuals from endangered species for which animals rather than populations have become the focus of conservation management, and to facilitate range recovery and expansion into sufficiently large, suitable landscapes.

Chapter 9 References

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Appendix 1 – Exterior (a) and interior (b) views of the leopard holding facility.

Photo: Florian J Weise

Appendix 2 – Cheetah holding facility with electric fencing.

Photo: Courtesy of Stuart Munro

Appendix 3 – Cheetah female Aju59 released from transport crate into a holding pen at the recipient area.

Photo: Courtesy of Femke Spoor

Appendix 4 – Carcass feed for cheetah Aju30 awaiting release.

Photo: Courtesy of Lars Baum

Appendix 5 – Rope-pulley system used to facilitate safe release of leopards.

Photo: Courtesy of Shaun Callow

Appendix 6 – Key characteristics of large carnivore recipient areas in Namibia.

ID (Region) Location	Size in km^2	Key climate characteristics[a]	Key habitat and vegetation characteristics[a]	Key prey species (>5 kg)	Documented large (density)[b] and meso-predators	Connection with other conservation areas	Animals received
NamibRand Nature Reserve (Hardap) GPS: S -24.97068 E 16.02452	1,722	Arid Annual rain: 100-150mm (50-70% variation) Erratic rains Regular droughts	Habitat: Namib plains with prominent inselbergs Biome: Nama-Karoo and Desert Vegetation structure: mixed southern desert and desert – dwarf shrub transition Features: Sand dunes Open grass plains with rock and gravel substrate Dwarf shrub savannah Mountain escarpment Artificial water holes No fencing	steenbok, klipspringer, springbok, red hartebeest, oryx, greater kudu, mountain zebra, plains zebra, blesbok, ostrich	leopard (low-medium), cheetah (low), spotted hyaena (medium-high), brown hyaena (low-medium), black-backed jackal, caracal	Yes	Aju01,02,03 Aju07 + 2 cubs Aju17 Aju18 Aju29,30 Aju58 + 2 cubs Pp06 Pp15
Sandfontein Nature and Game Reserve (Karas) GPS: S -28.6752 E 18.59589	760	Arid Annual rain: 50-100mm (50-60% variation) Erratic rains Regular droughts	Habitat: Gomkab Basin with dissecting rolling hills Biome: Nama Karoo Vegetation structure: Karas dwarf shrubland Features: Open grass plains with rock and gravel substrate Dwarf shrub savannah Inselbergs Mountain escarpment Artificial water holes Access to permanent Orange River Riverine woodland Game fencing	steenbok, common duiker, klipspringer, springbok, greater kudu, oryx, red hartebeest, plains zebra, common eland, common impala, ostrich	leopard (low), cheetah (n/a), spotted hyaena (n/a), brown hyaena (low), black-backed jackal, caracal	No	Aju42,43,44 Aju59 + 3 cubs Pp45
Kulala Wilderness Reserve (Hardap) GPS: S -24.70415 E 15.82466	339	Arid Annual rain: 50-150mm (50-60% variation) Erratic rains Regular droughts	Habitat: Namib plains with prominent inselbergs Biome: Nama-Karoo Vegetation type: desert – dwarf shrub transition Features: Sand dunes Open grass plains with rock and gravel substrate Dwarf shrub savannah	steenbok, klipspringer, springbok, oryx, greater kudu, ostrich	leopard (low), cheetah (low), spotted hyaena (medium-high), brown hyaena (low-medium), black-backed	Yes	Aju19,20 Pp27

			Artificial water holes No fencing		jackal, caracal		
Namib Desert Lodge (Khomas and Hardap) GPS: S -23.92416 E 15.81008	250	Arid Annual rain: 100-150mm (50-70% variation) Erratic rains Regular droughts	Habitat: Namib plains with prominent inselbergs Biome: Nama-Karoo Vegetation type: desert – dwarf shrub transition Features: Sand dunes Open grass plains with rock and gravel substrate Dwarf shrub savannah Mountain escarpment Artificial water holes No fencing	steenbok, klipspringer, springbok, oryx, greater kudu, ostrich	leopard (low-medium), cheetah (low), spotted hyaena (medium), brown hyaena (low-medium), black-backed jackal, caracal	Yes	Aju40,41
Neuhof Nature Reserve (Hardap) GPS: S -24.73009 E 16.11709	274	Arid Annual rain: 100-150mm (60-70% variation) Erratic rains Regular droughts	Habitat: Flat lying plateau with boulders Biome: Nama-Karoo Vegetation type: desert - dwarf shrub transition Features: Open grass plains with rock and gravel substrate Dwarf shrub savannah Mountain escarpment Artificial water holes No fencing	steenbok, klipspringer, springbok, oryx, mountain zebra, greater kudu, ostrich	leopard (medium), cheetah (low), spotted hyaena (medium-high), brown hyaena (low-medium), black-backed jackal, caracal	Yes	Pp47
Düsternbrook Guest Farm (Otjozondjupa) GPS: S -22.22135 E 16.91315	120	Semi-arid Annual rain: 300-350mm (30-40% variation) Variable rains Irregular droughts	Habitat: Khomas Hochland Plateau Biome: Savannah Vegetation type: Highland shrubland Features: Undulating mountainous terrain *Acacia*-encroached thickets with open glades Permanent swamps and water dams Artificial water holes Riverine woodland Game fencing	steenbok, common duiker, springbok, oryx, greater kudu, red hartebeest, plains zebra, black and blue wildebeest, common waterbuck, common impala, ostrich, warthog	leopard (medium-high), cheetah (medium-high), brown hyaena (medium), black-backed jackal, caracal	No	Pp57
Frauenstein - Ondekaremba Complex (Khomas)	112	Semi-arid Annual rain: 350-400mm (30-40% variation) Variable rains	Habitat: Khomas Hochland Plateau Biome: Savannah Vegetation type: Highland shrubland Features:	steenbok, common duiker, springbok, oryx, greater kudu, red hartebeest, ostrich,	leopard (medium), cheetah (medium), brown hyaena (low), black-	No	Aju26 Aju34 Aju56 + 3 cubs Aju65,66

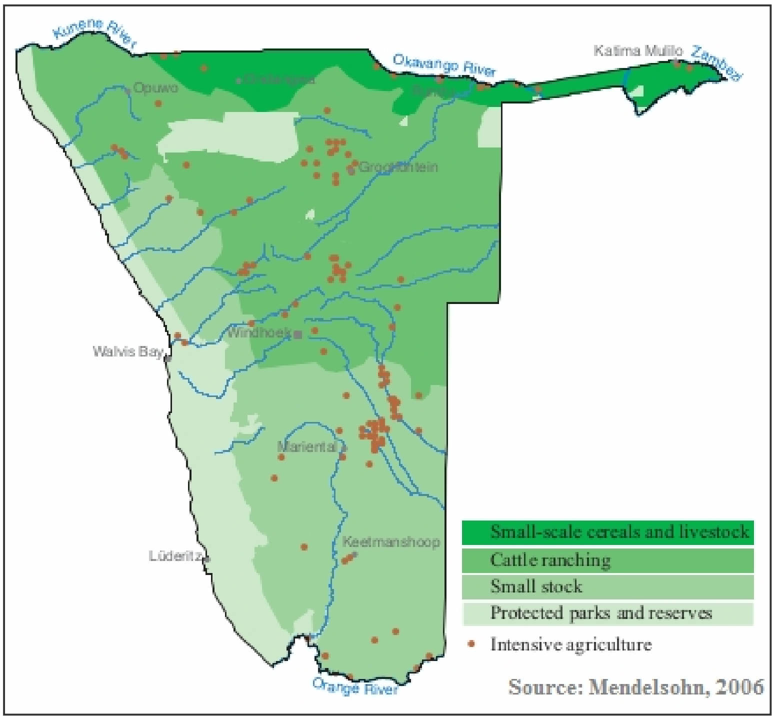
GPS: S -22.40366 E 17.42077		Irregular droughts	Undulating terrain *Acacia*-encroached thickets with open glades Artificial water holes Cattle and game fencing	warthog	backed jackal, caracal		Hbr55
Solitaire Guest Farm (Khomas) GPS: S -23.88794 E 16.01535	95	Arid - semi-arid Annual rain: 100-200mm (50-60% variation) Erratic rains Regular droughts	Habitat: Namib plains with prominent inselbergs Biome: Nama-Karoo Vegetation type: desert – dwarf shrub transition Features: Open grass plains with rock and gravel substrate Dwarf shrub savannah Mountain escarpment Artificial water holes Cattle and game fencing	steenbok, klipspringer, springbok, oryx, greater kudu, ostrich	leopard (low-medium), cheetah (low), spotted hyaena (medium-high), brown hyaena (low-medium), black-backed jackal, caracal	Yes	Aju38

[a] according to Mendelsohn et al., 2002

[b] according to Hanssen and Stander, 2004

Appendix 7 – Distribution of farming systems in Namibia.

Source: Mendelsohn, 2006.



Source: Mendelsohn, 2006

Appendix 8 – Minimum information catalogue – Manager consultations (2008-2014). *Data recorded for 24 months prior to first consultation.*

Interviewer
Date
Time
Reason of consultation request

Manager characteristics:

1) Name of respondent
2) Age and gender of respondent
3) Management mandate (owner/lessee/other)
4) Management tenure (in years)
5) Size of management unit(s) in hectares
6) Contact details

Property characteristics:

1) Property name(s) and FID registration no.(s)
2) Region
3) Income structure by priority contribution (cattle, smallstock, trophy hunting, non-consumptive/recreational tourism, wildlife farming, agriculture, other business etc)
4) If livestock are stocked, record numbers and races
5) Which large carnivores and meso-carnivores are present on the property?

Conflict characteristics:

1) Do large carnivores cause conflict on the property?
2) Type of conflict (security threat, livestock/game predation, other)
2) Which species cause conflict, by priority impact?
4) Annual loss to large carnivores (number of stock, % of total stock)
5) Estimated value of loss per annum (record in USD or ZAR)
6) Loss prevention strategies implemented on property

Large carnivore management:

1) What is annual tolerance of loss (stock numbers or % of total stock) before offending individuals are persecuted?

2) Which species are persecuted? (Include meso-carnivores)
3) Numbers removed (live vs. lethal) per species (available evidence)
4) Methods of persecution (capture, shoot-on-sight, gin traps, poison, night patrols, etc.)
5) Are large carnivores utilised for income generation (consumptive, non-consumptive)?
6) How can large carnivore management be improved in Namibia?

Consultation outcomes:

1) Conflict reduction measures suggested/implemented
2) Carnivores released/collared/destroyed/translocated
3) Contact facilitation with other respondents?
4) Notes for follow-up feedback necessary?
5) Follow-up date (6-12 months) for impact assessment_____________________

Respondent comments:

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Interviewer comments:

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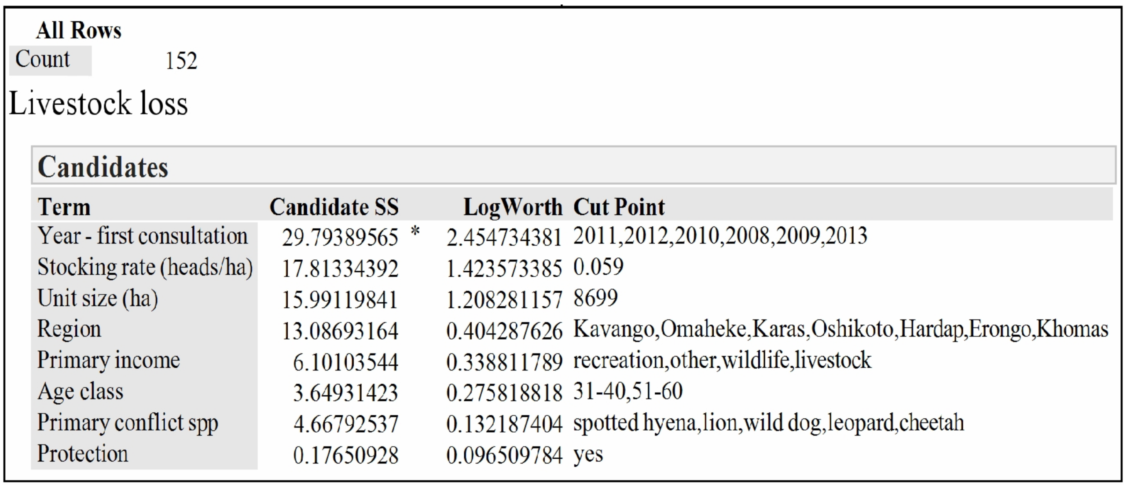
Appendix 9 – Model results of Decision Tree interrogation of different factors influencing annual percentage stock loss on free-hold management units.

Factors included unit size, stocking rate, primary income category, primary carnivore conflict species, farming area, year of consultation, respondent age class and protective efforts.

No. of node splits	R^2-Value	RMSE	AICc
1	0.093	1.3807997	535.609
2	0.240	1.2644369	510.956
3	0.266	1.242113	507.68
4	0.279	1.2314237	507.221
5	0.292	1.2201514	506.623
6	0.299	1.2142841	507.387
7	0.312	1.203173	506.853
8	0.313	1.2018959	508.823
9	0.317	1.1979899	510.159
10	0.355	1.1824392	508.546
11	0.341	1.1773937	509.639
12	0.350	1.1689064	509.868
13	0.360	1.1598864	509.977
14	0.364	1.1568185	511.672
15	0.383	1.1387355	509.42
16	0.401	1.1225887	507.654
17	0.404	1.1189808	509.29
18	0.406	1.1173066	511.49
19	0.409	1.1148131	513.506
20	0.411	1.112789	515.691
21	0.414	1.1100708	517.727
22	0.424	1.1007001	517.974



All Rows

Count 152

Livestock loss

Candidates

Term	Candidate SS		LogWorth	Cut Point
Year - first consultation	29.79389565	*	2.454734381	2011,2012,2010,2008,2009,2013
Stocking rate (heads/ha)	17.81334392		1.423573385	0.059
Unit size (ha)	15.99119841		1.208281157	8699
Region	13.08693164		0.404287626	Kavango,Omaheke,Karas,Oshikoto,Hardap,Erongo,Khomas
Primary income	6.10103544		0.338811789	recreation,other,wildlife,livestock
Age class	3.64931423		0.275818818	31-40,51-60
Primary conflict spp	4.66792537		0.132187404	spotted hyena,lion,wild dog,leopard,cheetah
Protection	0.17650928		0.096509784	yes

Appendix 10 – Mean annual large carnivore persecution on free-hold management units. *Data based on 219 respondents, 2008 - 2014.*

Species	**No. per management unit**				**No. per 100 / km^2**
	Min	Mean (SE)	Mode	Max	
African wild dog (*n* = 59)	0	0.1 (0.1)	0	37	0.0882
Brown hyaena (*n* = 43)	0	0.1 (0.0)	0	7	0.0537
Cheetah (*n* = 369)	0	0.8 (0.2)	0	82	0.5481
Leopard (*n* = 192)	0	0.4 (0.1)	0	11	0.3258
Lion (*n* = 11)	0	0.0 (0.0)	0	9	0.0211
Spotted hyaena (*n* = 21)	0	0.1 (0.0)	0	9	0.0498

Appendix 11 – Trapped cheetah cubs used as live bait.

Photo: Florian J Weise

Appendix 12 – Large carnivore persecution (24 months) per management unit by primary income.

Species	**Livestock** (*n* = 124)				**Wildlife** (*n* = 33)				**Business** (*n* = 29)				**Recreation** (*n* = 29)				**Other** (*n* = 6)			
	Min	Mean	SE	Max	Min	Mean	SE	Max	Min	Mean	SE	Max	Min	Mean	SE	Max	Min	Mean	SE	Max
African wild dog	0	0.1	0.1	14	0	1.3	1.1	37	---	---	---	---	---	---	---	---	---	---	---	---
Brown hyaena	0	0.3	0.1	13	0	0.2	0.1	3	0	0.1	0.1	2	---	---	---	---	---	---	---	---
Cheetah	0	1.6	0.7	82	0	3.9	0.9	24	0	1.0	0.6	14	0	0.7	0.4	3	0	0.5	0.5	3
Leopard	0	1.1	0.2	11	0	1.1	0.3	9	0	0.5	0.2	3	0	0.2	0.1	2	0	1.3	1.3	5
Lion	0	0.1	0.1	9	0	0.0	0.0	1	---	---	---	---	---	---	---	---	---	---	---	---
Spotted hyaena	0	0.2	0.1	9	---	---	---	---	---	---	---	---	0	0.1	0.1	3	---	---	---	---

Appendix 13 – Large carnivore persecution by respondent age class.

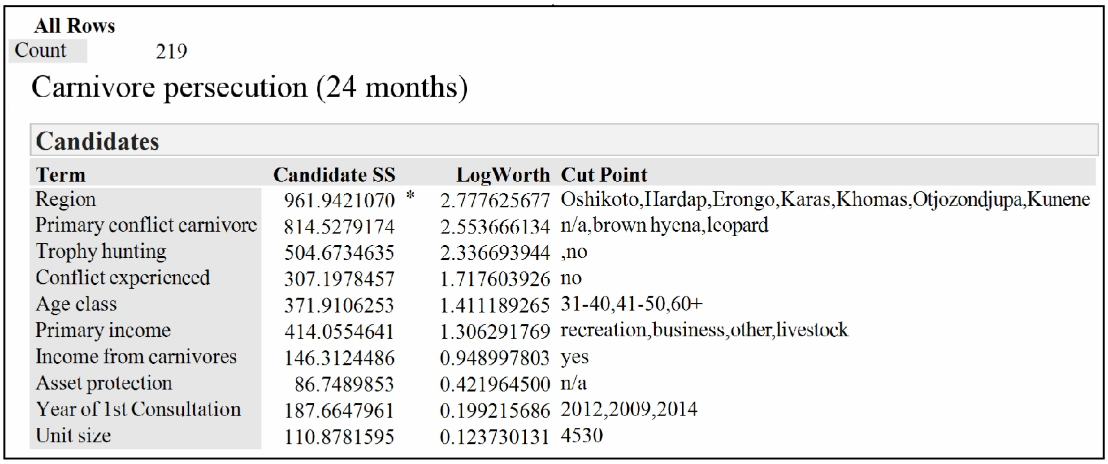
Data represent combined trapping and killing.

Age class	**No. per management unit (24 months)**			
	Min	Mean (SE)	Mode	Max
31 – 40 years (*n* = 47)	0	1.4 (0.3)	0	6
41 – 50 years (*n* = 78)	0	2.6 (0.4)	0	24
51 – 60 years (*n* = 78)	0	4.8 (1.2)	0	85
>60 years (*n* = 18)	0	2.8 (0.9)	0	13

Appendix 14 – Model results of Decision Tree interrogation of different factors influencing carnivore persecution on free-hold management units.

Factors included unit size, primary income category, primary carnivore conflict species, region, year of study, respondent age class, asset protection, conflict experience and income derivation from carnivores.

No. of node splits	R^2-Value	RMSE	AICc
1	0.080	7.0857321	1485.25
2	0.124	6.9169936	1576.77
3	0.134	6.874695	1476.17
4	0.135	6.8711909	1478.07
5	0.137	6.8654414	1479.83
6	0.143	6.8397416	1480.35
7	0.150	6.8132033	1480.82
8	0.150	6.8122604	1482.95
9	0.151	6.8070608	1484.84
10	0.152	6.8031152	1486.82
11	0.170	6.7308244	1484.4
12	0.217	6.5373134	1473.91
13	0.218	6.5328473	1475.92
14	0.220	6.5263458	1477.81
15	0.223	6.5126059	1479.26
16	0.226	6.5027768	1480.95
17	0.226	6.5025067	1483.33
18	0.228	6.4943388	1485.21
19	0.230	6.4828095	1486.87
20	0.232	6.4751203	1488.83
21	0.233	6.4726816	1491.16
22	0.233	6.4709601	1493.57
23	0.234	6.4681363	1495.93
24	0.234	6.4674295	1498.46
25	0.234	6.466767	1501.02
26	0.343	5.9898524	1470.09
27	0.343	5.9897707	1472.74
28	0.345	5.9787448	1474.62
29	0.349	5.9602337	1475.98
30	0.350	5.9587291	1478.62



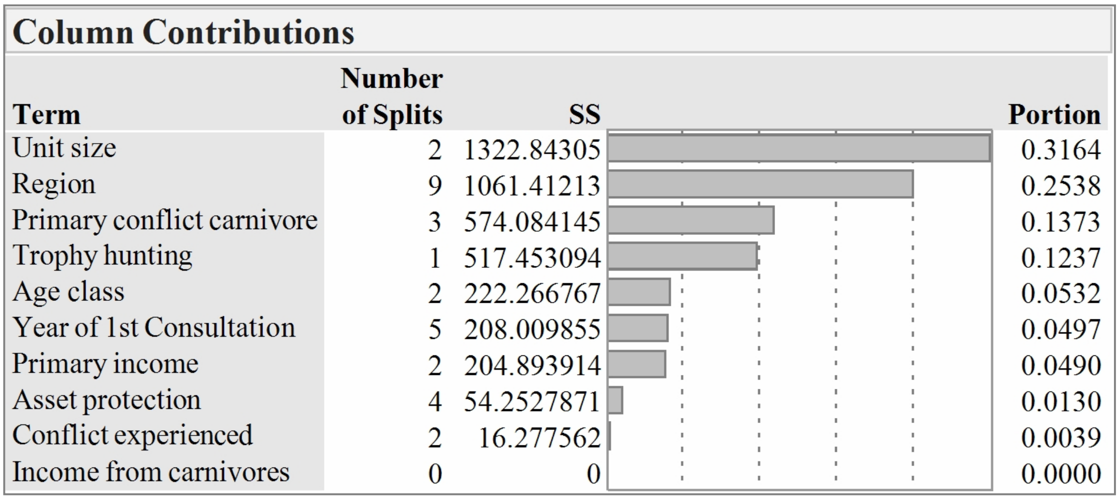
All Rows

Count 219

Carnivore persecution (24 months)

Candidates

Term	Candidate SS		LogWorth	Cut Point
Region	961.9421070	*	2.777625677	Oshikoto,Hardap,Erongo,Karas,Khomas,Otjozondjupa,Kunene
Primary conflict carnivore	814.5279174		2.553666134	n/a,brown hyena,leopard
Trophy hunting	504.6734635		2.336693944	,no
Conflict experienced	307.1978457		1.717603926	no
Age class	371.9106253		1.411189265	31-40,41-50,60+
Primary income	414.0554641		1.306291769	recreation,business,other,livestock
Income from carnivores	146.3124486		0.948997803	yes
Asset protection	86.7489853		0.421964500	n/a
Year of 1st Consultation	187.6647961		0.199215686	2012,2009,2014
Unit size	110.8781595		0.123730131	4530



Column Contributions

Term	Number of Splits	SS	Portion
Unit size	2	1322.84305	0.3164
Region	9	1061.41213	0.2538
Primary conflict carnivore	3	574.084145	0.1373
Trophy hunting	1	517.453094	0.1237
Age class	2	222.266767	0.0532
Year of 1st Consultation	5	208.009855	0.0497
Primary income	2	204.893914	0.0490
Asset protection	4	54.2527871	0.0130
Conflict experienced	2	16.277562	0.0039
Income from carnivores	0	0	0.0000

Appendix 15 – Details of resident large carnivores monitored on free-hold management units, 2008 – 2014.

ID	Sex	Age class[a]	Body weight (kg)	Year	Capture region	Transmitter type	Monitoring days[b]	Comments
Leopard								
Pp04	F	Adult	30	2008	Khomas	GPS/GSM satellite[c]	1,923	Addition detail in chapter 4
Pp48	M	Adult	Unknown	2010	Otjozondjupa	GPS/GSM satellite[c]	202	Addition detail in chapter 4
Pp62	M	Adult	61	2012	Khomas	GPS iridium satellite[c]	600	Addition detail in chapter 4
Pp63	M	Adult	54	2012	Khomas	GPS iridium satellite[c]	442	Addition detail in chapter 4
Pp71	F	Adult	38	2013	Hardap	GPS iridium satellite[c]	134	Addition detail in chapter 4
Pp77	F	Adult	36	2014	Otjozondjupa	GPS iridium satellite[c]	322	Addition detail in chapter 4
Pp82	M	Adult	~50	2014	Khomas	GPS iridium satellite[c]	155	Trapped indiscriminately
Pp83	M	Adult	>75	2014	Otjozondjupa	GPS iridium satellite[c]	85	Suspected livestock raider
Cheetah								
Aju10	F	Adult	39	2008	Khomas	GPS/GSM satellite[c]	164	Trapped for research purposes
Aju11	M	Adult	50	2008	Khomas	GPS/GSM satellite[c]	412	Trapped for research purposes
Aju16	M	Adult	46	2009	Khomas	VHF[d]	16	Trapped for research purposes
Aju22	M	Adult	53	2009	Khomas	GPS/GSM satellite[c]	173	Trapped for research purposes
Aju23	M	Adult	51	2009	Khomas	GPS/GSM satellite[c]	288	Trapped for research purposes
Aju24	M	Adult	47	2009	Khomas	GPS/GSM satellite[c]	276	Trapped for research purposes
Aju25	M	Adult	49	2009	Khomas	GPS/GSM satellite[c]	79	Trapped for research purposes
Aju70	F	Adult	38	2013	Hardap	GPS satellite[e]	573	Trapped for research purposes
Aju74	M	Adult	47	2013	Khomas	GPS iridium satellite[c]	485	Trapped for research purposes
Aju76	M	Adult	41	2013	Khomas	GPS iridium satellite[c]	419	Trapped for research purposes
Aju78	M	Adult	48	2014	Hardap	GPS iridium satellite[c]	27	Suspected livestock raider
Aju79	M	Adult	37	2014	Khomas	GPS iridium satellite[c]	272	Trapped for research purposes
Aju80	M	Adult	~47	2014	Khomas	GPS iridium satellite[c]	217	Trapped indiscriminately
Brown hyaena								
Hbr75	M	Sub-adult	33	2013	Khomas	GPS iridium satellite[c]	105	Additional detail in chapter 6
Hbr81	M	Sub-adult	30	2014	Khomas	GPS iridium satellite[c]	191	Trapped indiscriminately

[a] Animals estimated <24 months = sub-adult, >24 months = adults. [b] Data collection concluded on 31 December 2014. [c] Africa Wildlife Tracking, Pretoria, RSA

[d] Advanced Telemetry Systems, Insanti, USA. [e] Sirtrack, Hawkes Bay, NZ

Appendix 16 – Lethal leopard removal in Namibia 1997-2003.

Source: Permit Office records, Ministry of Environment and Tourism, Windhoek, Namibia.

Year	Problem animal control[a,b]	Trophy hunting[c]	Cumulative removal[d]
1997	52	57	109
1998	93	33	126
1999	89	56	145
2000	138	59	197
2001	131	66	197
2002	122	98	220
2003	145	106	251

[a] Reported values are likely significantly less than true removal because farmers often do not report persecuted leopards.

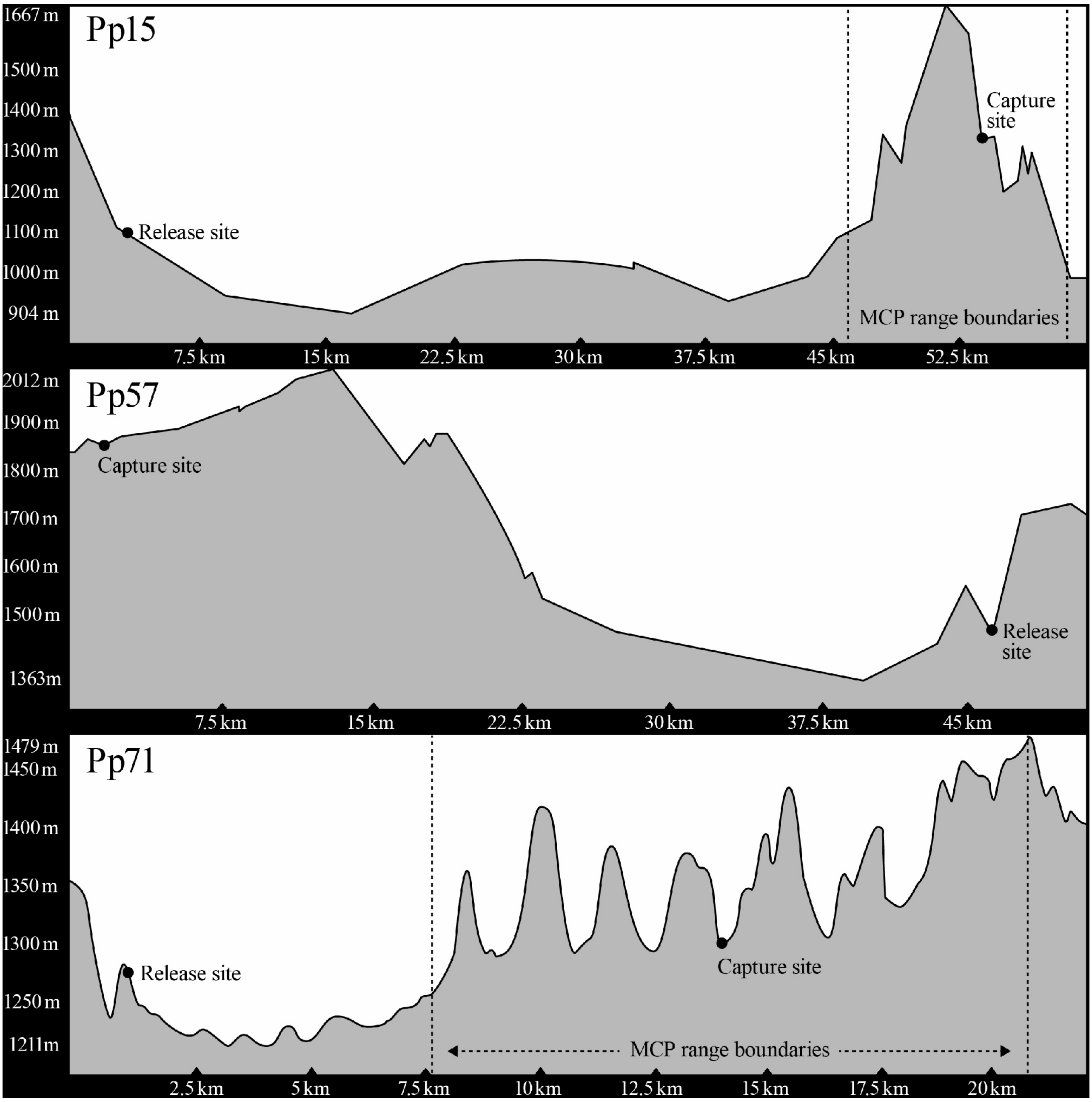
[b] significant correlation Problem leopard control with Year: $r_s = 0.821$, $p = 0.023$

[c] significant correlation Trophy leopards hunted with Year: $r_s = 0.892$, $p = 0.006$

[d] significant correlation Cumulative removal with Year: $r_s = 0.991$, $p < 0.001$

Appendix 17 – Capture and release sites of three short-distance leopard translocations in relation to local topography.

Displays include one successful case of homing to a known pre-capture range (Pp15), one case without homing (Pp57) and one successful case of homing to an expected range later confirmed by GPS monitoring (Pp71). Females Pp15 and Pp71 were released with unobstructed view of their ranges to enable homing.



Appendix 18 – Thorn-bush enforced night kraal for protection of cattle calves against carnivore predation.

Photo: Florian J Weise

Appendix 19 – List of suitable leopard recipient reserves in Namibia.

Results were determined with CaTSuiT for Leopard (Lemeris Jr., 2015), with reference to whether each area can support a leopard translocation on its own (stand-alone), or if additional connected habitat is required across its borders (connected).

Recipient area	*Suitable area (km^2)*	*Stand-alone*	*Connected*
National Parks			
Ai-Ais Hot Springs	3,995.8	✓	
Bwabwata National Park-Buffalo core area	616.1		✓
Bwabwata National Park-Kwando core area	1,203.4	✓	
Bwabwata National Park-Mahango	185.0		✓
Bwabwata National Park-Multiple use area	3,821.0	✓	
Dorob National Park	2,551.9	✓	
Etosha National Park	9,766.6	✓	
Khaudum National Park	2,767.5	✓	
Mamili National Park	271.8		✓
Mudumu National Park	669.0		✓
Namib-Naukluft Park	28,100.6	✓	
Skeleton Coast Park	15,742.3	✓	
Sperrgebiet National Park	11,325.1	✓	
Communal Conservancies			
=/=Gaingu	279.7		✓
African Wild Dog (North)	169.2		✓
African Wild Dog (South)	998.8	✓	
Balyerwa	222.0		✓
Bamunu	390.0		✓
Doro !nawas	176.6		✓
Dzoti	285.1		✓
Epupa	54.6		✓
Etanga	899.7	✓	
George Mukoya	483.8		✓
Huibes	954.8	✓	
Iipumbu ya Tshilongo	731.7		✓
Impalila	32.5		✓
Joseph Mbambangandu	10.3		✓
Kabulabula	75.6		✓
Kasika	138.6		✓
King Nehale	478.3		✓
Kwandu	159.2		✓
Mashi	249.0		✓
Mayuni	150.6		✓
Muduva Nyangana	75.2		✓
N=/=a Jaqna	802.6		✓
Okamatapati	1,498.5	✓	
Okanguati	365.5		✓
Okondjombo	284.0		✓
Ombujokanguindi	90.0		✓
Omuramba Ua Mbinda	398.6		✓
Ondjou	25.3		✓
Ongongo	274.8		✓
Orupembe	558.3		✓
Otjitanda	218.4		✓
Otjituuo	4,456.7	✓	
Otjiu-West	955.3	✓	
Otjombinde	5,169.5	✓	
Ozonahi	2,163.3	✓	
Puros	121.5		✓
Salambala	610.4		✓
Sanitatas	281.5		✓
Sesfontein	105.4		✓
Sheya Shuushona	194.5		✓
Sobbe	390.7		✓
Sorri-Sorris	1,200.8	✓	
Torra	62.6		✓
Tsiseb	1,181.7	✓	
Tsiseb	103.9		✓
Wuparo	147.6		✓
Emerging Conservancies			
Ehama	904.7	✓	
Ekoto	707.4		✓
Lusese	239.0		✓
Mahachani	364.2		✓
Mbara	128.0		✓
Nakobolelwa	104.6		✓
Otjikongo	733.3		✓
Kapinga Kamwalye	1,290.3	✓	
Private Reserves			
Kulala Wilderness	332.9		✓
Gondwana	974.3	✓	
NamibRand	1,083.2	✓	
Neuhof	221.0		✓
Sandfontein	838.3		✓

Appendix 20 – Recommendations and protocol considerations for leopard translocations.

1) Conservation agencies (public and private) should focus on identification of suitable release areas prior to translocations;
2) Translocate only when other conservation strategies have already been explored and exhausted, or in combination with alternative strategies (e.g. improving stock husbandry at source sites);
3) Thoroughly define conservation objectives before translocation, and critically assess outcomes post-implementation;
4) Release animals into low to medium density extant range only (Hayward et al., 2006a) - no translocations into high density (saturated) areas;
5) No translocation of known habitual livestock raiders (Hamilton, 1981) - only opportunistic livestock raiders should be moved if release areas are available (see raider assessment provided in Stander, 1990a);
6) No continuous translocations into one release area (see Hamilton, 1981; Athreya et al., 2011);
7) Provide a minimum inter-release period of 18 months to allow assimilation of individuals into the local population;
8) Consider area-specific sex ratios when multiple leopards are released into the same area (approximately F 2 : 1 M) (Hamilton, 1981; Bailey, 1993);
9) Leopards should not be released under the age of 18 months (cubs only alongside mothers) or if habituated to human presence;
10) Where possible, and genetically justifiable (Ropiquet et al., 2015) translocate leopards over long distances (>200 km) to prevent familiarity with the release site and homing (Linnell et al., 1997);
11) Rather than releasing haphazardly, maintain leopards in appropriate temporary captivity with minimal human contact until an adequate release site has been identified;
12) Evaluate further soft-releases (from secure, roofed acclimatisation pens) for the species;
13) Monitor translocated leopards with GPS satellite transmitters[*] to assess outcomes according to pre-defined objectives;
14) Share available monitoring information from tracking units with landowners to increase accountability and tolerance;
15) Formulate relevant conflict reduction or compensation agreements with immediate release area neighbours; and
16) Where feasible, improve value of translocated individuals through tourism incentives such as tracking operations (see Stander et al., 1997a).

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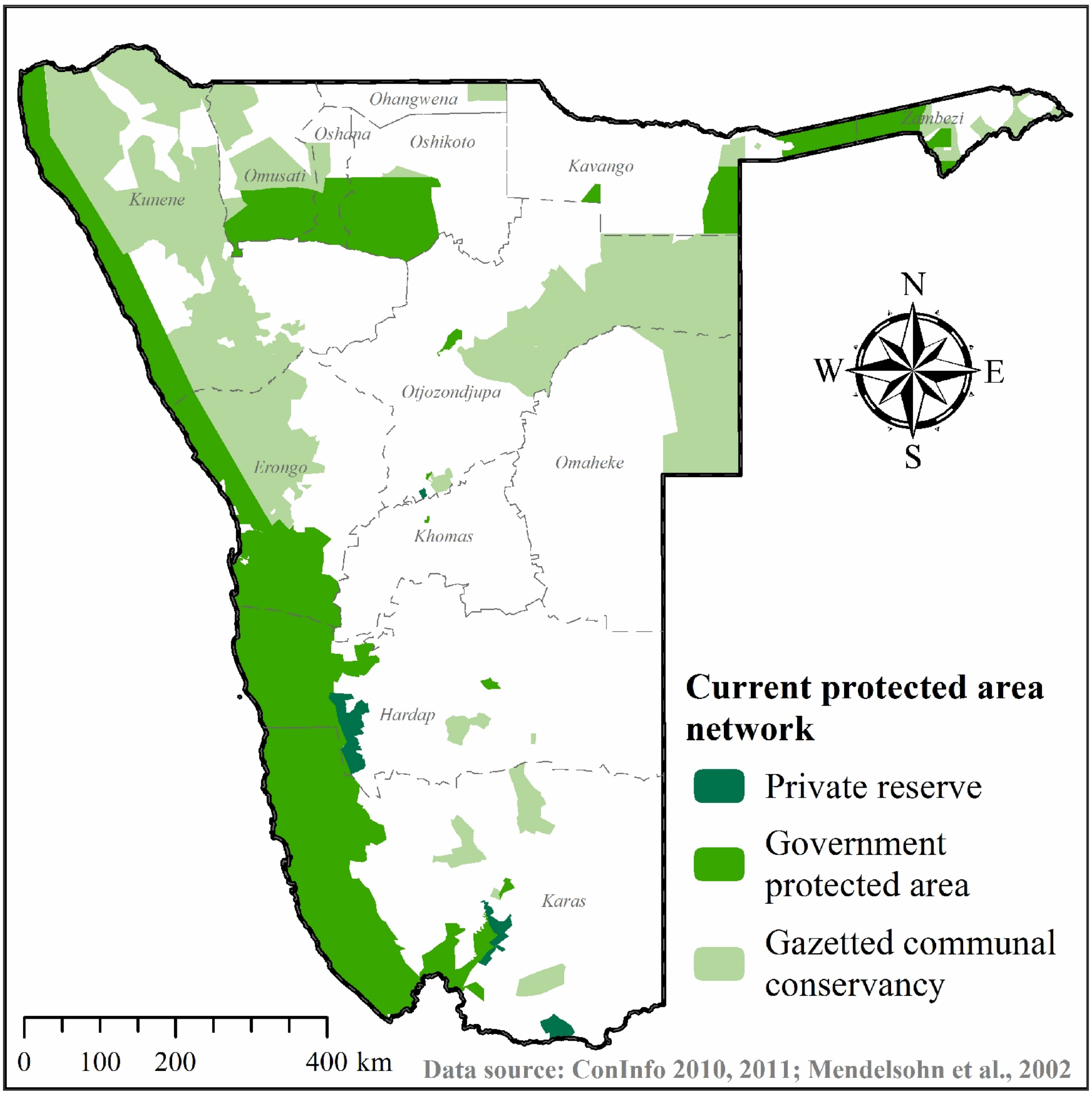
[*] VHF-only transmitters are often not adequate to document translocation outcomes in sufficient detail

Appendix 21 – Kaplan-Meier survivorship of adult translocated cheetahs.

Assessment interval (days post-release)	**At risk (*n*)**	**Censored**	**Mortalities**	**Kaplan-Meier survivorship estimate**	**95% CI low**	**95% CI high**
1-90	23	1	8	0.65	0.43	0.83
91-180	14	1	0	1.00	0.73	1.00
181-270	13	0	1	0.92	0.62	0.99
271-360	12	1	1	0.92	0.60	0.99
361-450	10	2	1	0.90	0.54	0.99
451-540	7	0	1	0.86	0.42	0.99
541-630	6	0	1	1.00	0.52	1.00
631-720	5	1	1	0.80	0.30	0.99

Appendix 22 – Namibia’s public and private protected area network.

CaTSuiT input area (302,580.9 km²) potentially available for cheetah translocations.



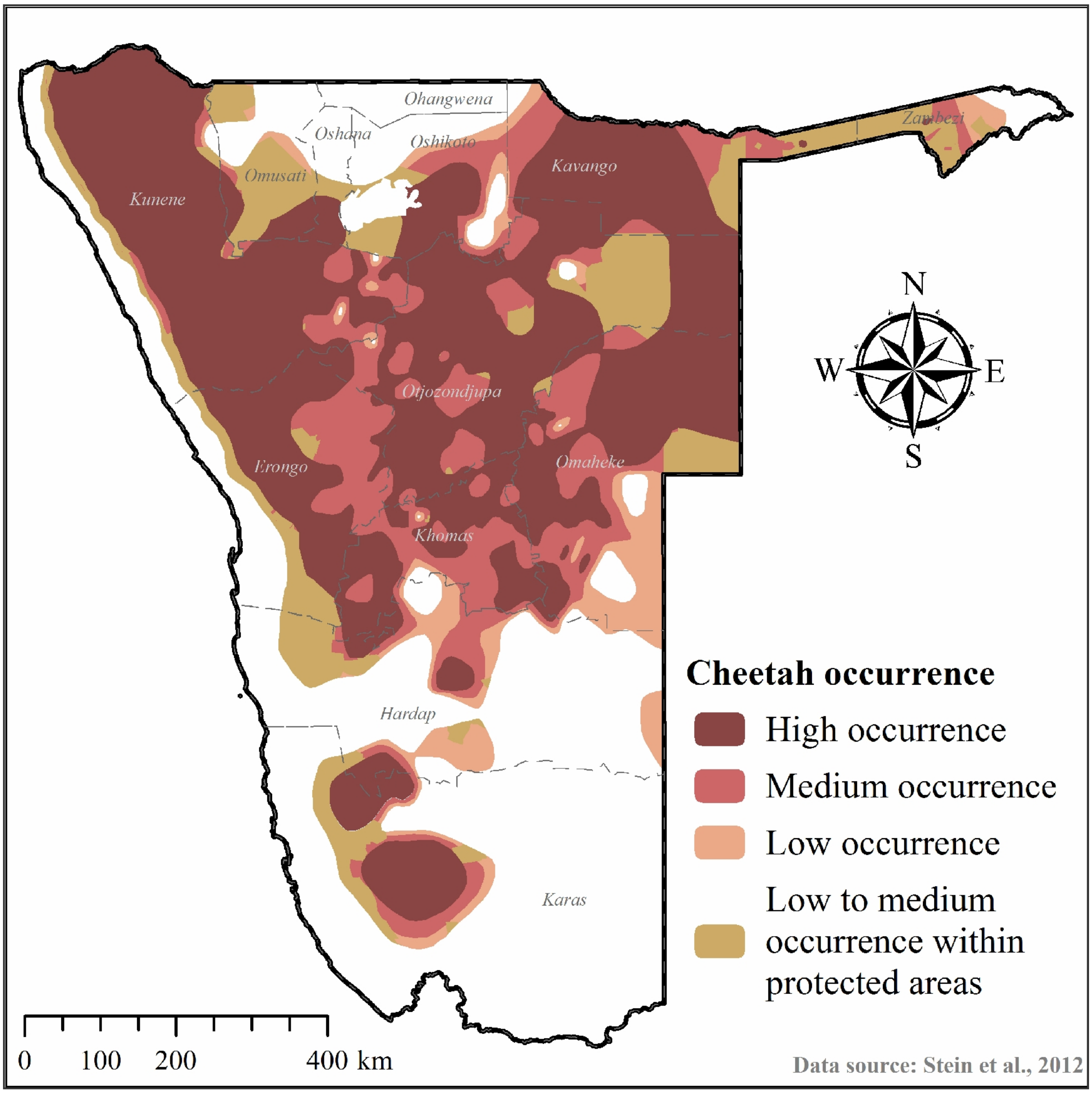
Data source: ConInfo 2010, 2011; Mendelsohn et al., 2002

Appendix 23 – Potential cheetah recipient areas in Namibia without site fidelity considerations.

ID (region)	Suitable patch size (km^2)	Protected area category
Namib Naukluft Park (Hardap/Erongo)	10,785.3	National park
Otjombinde (Omaheke)	1,738.0	Communal conservancy
Otjituuo 1 (Otjozondjupa)	765.6	Communal conservancy
!Han /Awab (Karas)	672.7	Communal conservancy
Sperrgebiet (Karas)	351.5	National park
Otjituuo 2 (Otjozondjupa)	294.4	Communal conservancy
African Wild Dog (Otjozondjupa)	262.0	Communal conservancy
N=/=a Jaqna (Otjozondjupa)	96.9	Communal conservancy
Ai-Ais Hot Springs (Karas)	54.8	National park
Ozonahi (Otjozondjupa)	50.1	Communal conservancy

Appendix 24 – Cheetah occurrence in Namibia.

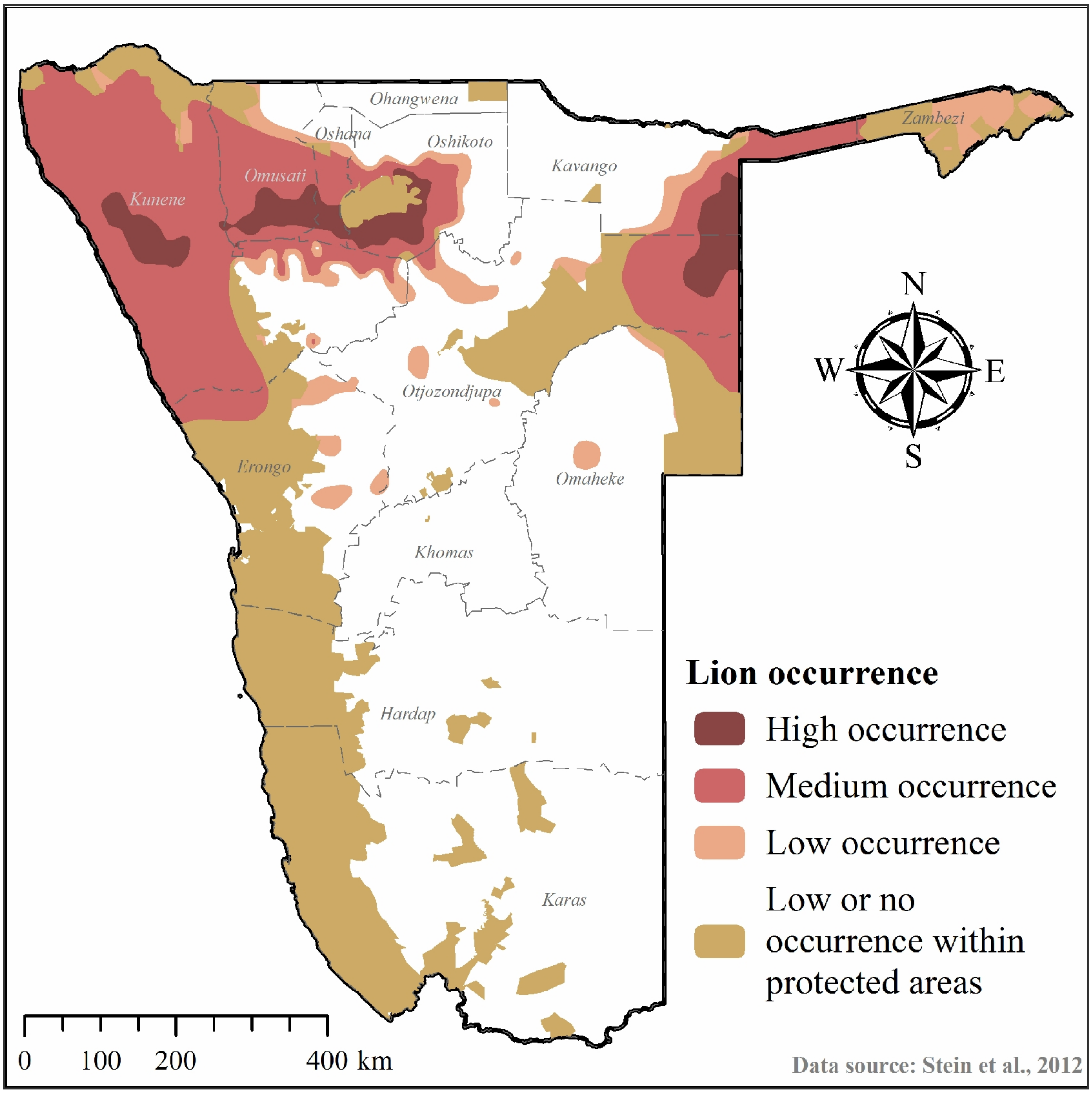
Data according to Stein et al. (2012). Map displays suitable protected area patches with low – medium cheetah occurrence.



Data source: Stein et al., 2012

Appendix 25 – Lion occurrence in Namibia.

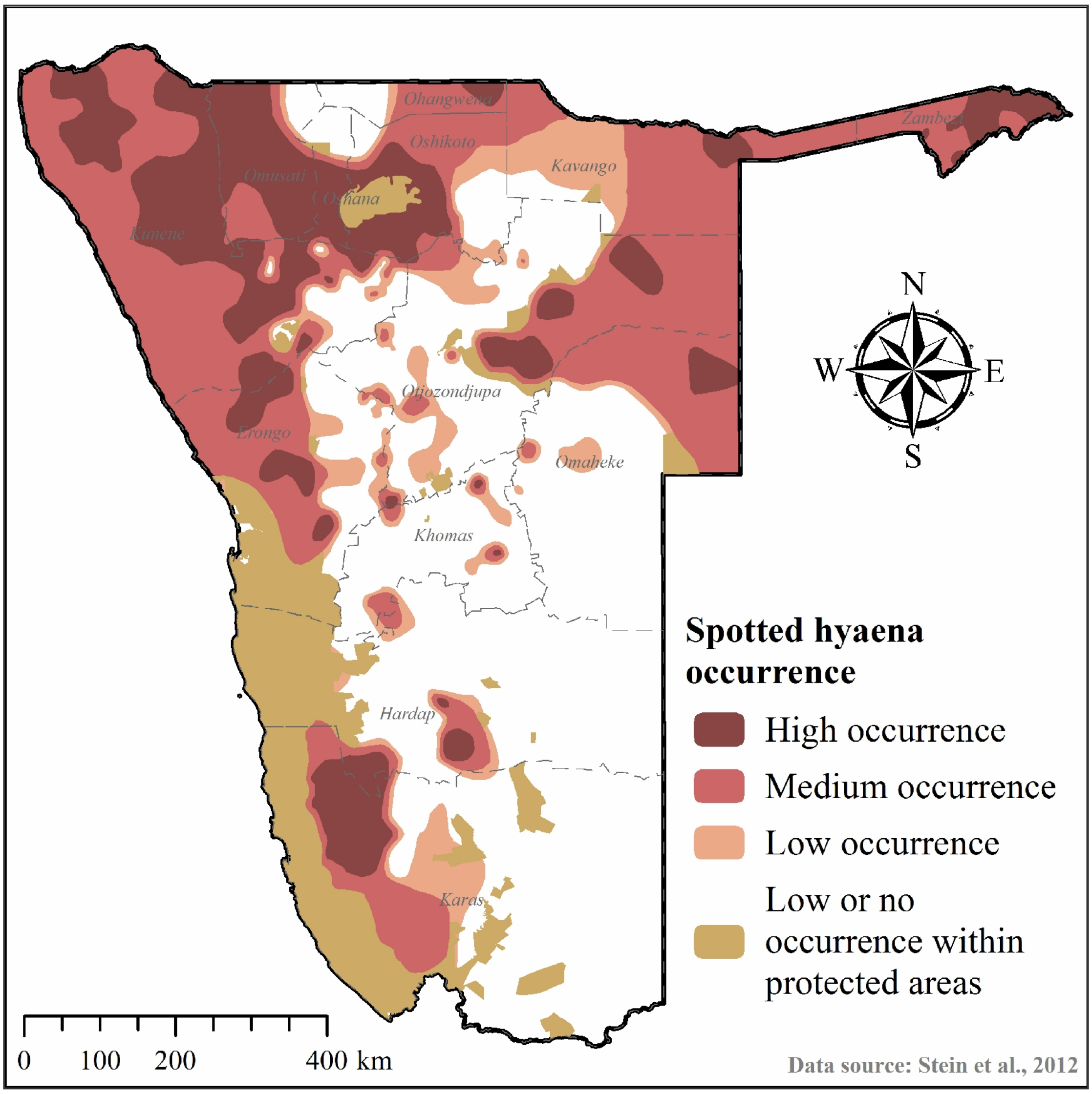
Data according to Stein et al. (2012). Map displays suitable PA patches excluding medium – high lion occurrence.



Data source: Stein et al., 2012

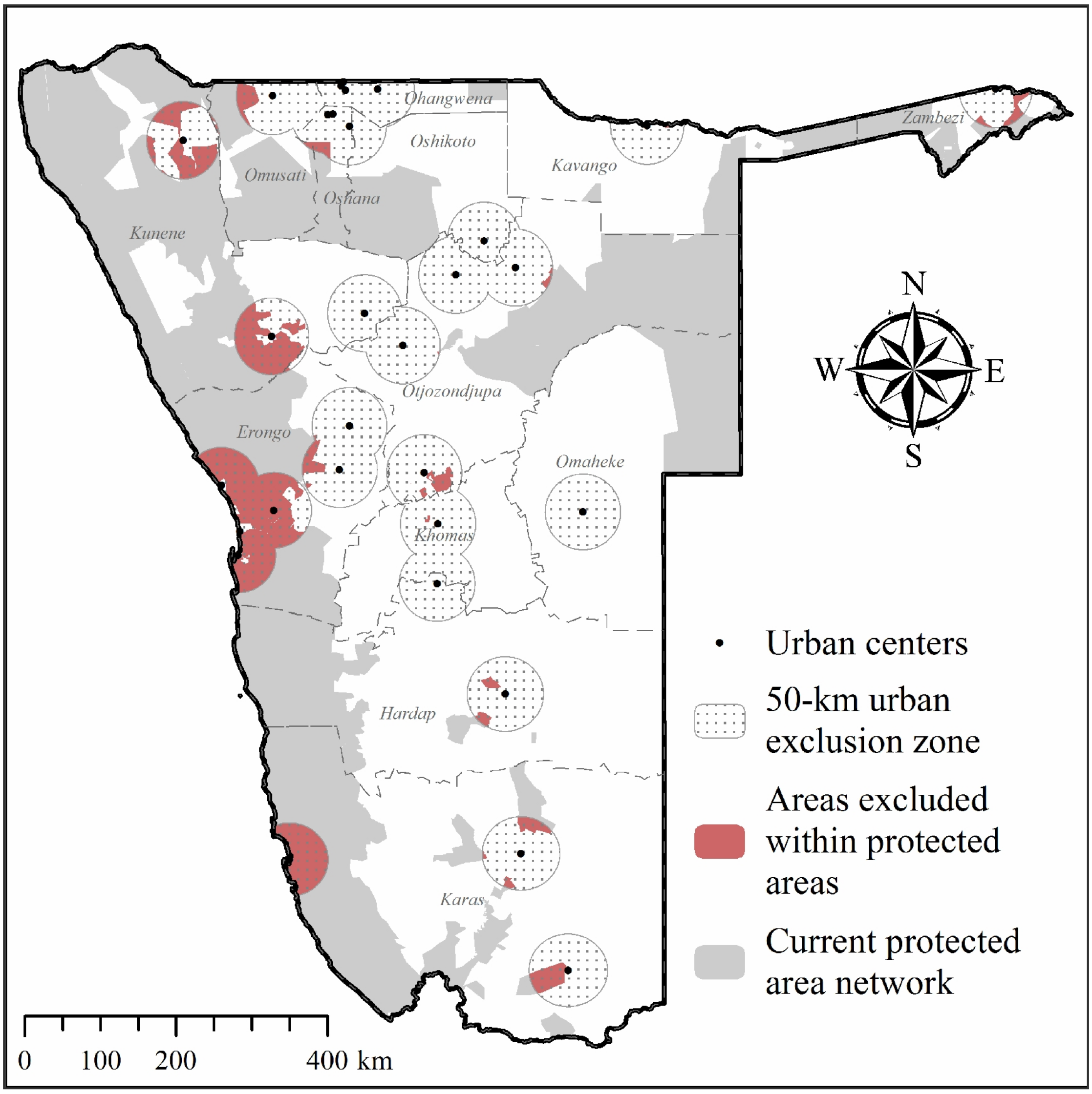
Appendix 26 – Spotted hyaena occurrence in Namibia.

Data according to Stein et al. (2012). Map displays suitable PA patches excluding medium – high spotted hyaena occurrence.



Appendix 27 – Safety buffer exclusion around registered urban areas.

Data according to Namibia Statistics Agency (2012). Map displays safety exclusion patches.



Appendix 28 – Cost literature search terminology and combinations used.

Search catalogue used for identification of publications containing costs of carnivore translocation and alternative non-lethal carnivore conflict management strategies.

First term[a]	**Second term**	**Third term**	**Term definition[a] (synonyms)**
Translocation costs			
Relocat*	Carnivor*		Carnivor* (carnivore(s), carnivorous)
Relocat*	Carnivor*	Cost	Financ* (financial, finance)
Relocat*	Carnivor*	Financ*	Predat* (predation, predator)
Relocat*	Predat*		Reloc* (relocate, relocation)
Relocat*	Predat*	Cost	Translocat* (translocate, translocation)
Relocat*	Predat*	Financ*	
Translocat*	Carnivor*		
Translocat*	Carnivor*	Cost	
Translocat*	Carnivor*	Financ*	
Translocat*	Predat*		
Translocat*	Predat*	Cost	
Translocat*	Predat*	Financ*	
Cost of alternative non-lethal carnivore conflict management strategies			
Allev*	Carnivor*	Cost	Allev* (alleviate, alleviation)
Allev*	Predat*	Cost	Control* (controlling, control)
Compensat*	Carnivor*		Compensat* (compensate, compensation)
Compensat*	Predat*		Damag* (damaging, damage)
Conflict	Carnivor*	Cost	Deter* (deter(ing), deterrence, deterrent)
Conflict	Predat*	Cost	Guard* (guardian, guarding, guard)
Control	Carnivor*	Cost	Mitigat* (mitigate, mitigation)
Control	Predat*	Cost	Reduc* (reduce, reduction)
Damag*	Carnivor*	Cost	Repel* (repel, repellent)
Damag*	Predat*	Cost	
Deter*	Carnivor*	Cost	
Deter*	Predat*	Cost	
Guard*	Carnivor*	Cost	
Guard*	Predat*	Cost	
Mitigat*	Carnivor*	Cost	
Mitigat*	Predat*	Cost	
Non-lethal	Carnivor*	Cost	
Non-lethal	Predat*	Cost	
Reduc*	Carnivor*	Cost	
Reduc*	Predat*	Cost	
Repel*	Carnivor*	Cost	
Repel*	Predat*	Cost	

[a] Ordered alphabetically

Appendix 29 – Translocation cost (USD) and percentage recuperation per cost category.

ID	Permit	Tracking	% Rec[a]	Veterinary	% Rec[a]	Transport	% Rec[a]	Staff	Holding	% Rec[a]	Feeding	Total cost	Total % Rec[a]
Aju01	1.27	220.00	0	89.00	70	252.50	100	20.34	44.87	0	8.54	636.52	49.5
Aju02	1.27	27.00	0	89.00	70	252.50	100	20.34	44.87	0	8.54	443.52	71.0
Aju03	1.27	220.00	0	89.00	70	252.50	100	20.34	44.87	0	8.54	636.52	49.5
Pp06	3.81	220.00	0	89.00	70	301.25	100	30.51	209.73	0	13.67	867.97	41.9
Aju07	3.81	220.00	0	89.00	70	575.43	52	38.13	139.82	0	51.25	1,117.44	32.5
Pp15	3.81	3,649.00	0	89.00	70	158.67	100	22.88	80.97	0	140.94	4,145.27	5.3
Aju17	3.81	3,399.00	100	26.69	0	285.61	100	30.51	80.97	0	153.75	3,980.34	92.6
Aju18	3.98	3,399.00	100	27.86	0	298.10	100	47.76	84.51	0	133.73	3,994.94	92.5
Aju19	1.99	223.13	0	139.30	80	294.15	100	27.86	84.51	0	58.84	829.78	48.9
Aju20	1.99	3,649.00	0	139.30	80	294.15	100	27.86	84.51	0	32.10	4,228.91	9.6
Aju26	3.98	1,385.24	100	27.86	0	146.79	0	29.18	145.07	37	10.70	1,748.82	82.3
Pp27	3.98	1,385.24	100	756.22	89	169.10	100	66.32	84.51	0	561.68	3,027.05	73.6
Aju29	1.88	211.30	100	175.87	85	238.80	100	22.61	320.98	57	506.58	1,478.02	52.9
Aju30	1.88	1,749.00	0	175.87	85	238.80	100	22.61	320.98	57	455.92	2,965.06	19.2
Aju34	3.77	211.30	0	26.38	0	111.45	0	27.64	80.03	0	52.35	512.92	0.0
Aju38	3.77	3,300.00	100	26.38	0	278.99	100	40.71	3,656.83	74	126.64	7,433.32	84.4
Aju40	1.70	3,300.00	100	23.76	100	37.47	100	13.91	3,292.59	74	889.43	7,558.86	76.5
Aju41	1.70	190.25	100	23.76	100	37.47	100	30.88	3,292.59	74	889.43	4,466.08	59.9
Aju42	1.13	190.25	100	23.76	0	174.05	100	17.31	1,623.70	46	798.21	2,828.41	39.2
Aju43	1.13	3,520.00	100	23.76	0	174.05	100	17.31	1,623.70	46	821.01	6,180.96	71.8

Aju44	1.13	220.00	100	23.76	0	174.05	100	17.31	1,623.70	46	798.21	2,858.16	39.9
Pp45	3.40	1,303.33	100	47.52	0	384.97	100	46.15	72.06	0	148.24	2,005.67	84.2
Pp47	3.40	1,303.33	100	47.52	100	226.21	100	65.16	425.94	0	136.84	2,208.40	71.4
Hbr55	4.12	1,583.38	100	28.86	100	35.67	100	19.79	0.00	0	0.00	1,671.82	98.6
Aju56	4.12	3,300.00	100	28.86	100	173.16	100	39.58	150.28	37	152.38	3,848.38	92.4
Pp57	4.12	1,583.38	100	28.86	100	64.67	100	59.37	0.00	0	3.69	1,744.09	96.1
Aju58	4.12	1,583.38	100	28.86	100	340.89	100	203.84	333.88	56	264.13	2,759.10	77.6
Aju59	4.12	1,583.38	100	28.86	100	451.25	100	54.70	150.28	37	304.77	2,577.36	82.2
Aju65	2.06	193.78	100	28.86	100	29.42	100	13.77	0.00	0	0.92	268.81	93.8
Aju66	2.06	1,583.38	100	28.86	100	29.42	100	13.77	0.00	0	0.92	1,658.41	99.0

[a] % Rec indicates percentage of cost recuperated from external funding sources.

Aju indicates cheetah.

Pp indicates leopard.

Hbr indicates brown hyaena.

Appendix 30 - Translocation cost summary statistics by category.

Category	Group	Sample size n	Mean	SD	SE	Median	1st Quartile	3rd Quartile	Min.	Max.
Sex	Male	15	2465.57	2069.64	534.38	2005.67	829.76	2965.06	268.81	7433.32
	Female	15	2913.16	1869.15	482.61	2759.1	1478.02	4145.27	443.52	7558.86
Release mode	Hard	19	1966.69	1434.77	329.19	1671.82	636.52	3848.38	268.81	4228.91
	Soft	11	3937.61	2150.95	648.53	2858.16	2577.36	6180.96	1478.02	7558.86
Background	Indiscriminate	15	2640.49	2428.15	626.95	1748.82	636.52	4228.91	268.81	7558.86
	Livestock raider	8	2198.83	1186.97	419.66	1874.88	1256.04	3537.36	867.97	3994.94
	Rehabilitation	7	3354.70	1467.28	554.58	2965.06	2828.41	4145.27	1478.02	6180.96
Collar type	VHF (+ID)	12	1146.71	1107.18	319.61	848.88	543.82	1319.91	268.81	4466.08
	GPS	18	3437.20	1783.00	420.26	2904.26	1941.46	4166.18	1658.41	7558.86
Group structure	Individual	12	2778.38	1896.93	547.60	2107.04	1689.89	3991.29	512.92	7433.32
	Group	14	2645.57	2267.61	606.04	2243.41	636.52	4288.20	268.81	7558.86
	Mother with cubs	4	2575.57	1122.48	561.24	2668.23	1482.42	3576.06	1117.44	3848.38
Habituation	Wild	21	2238.52	1808.93	394.74	1744.09	733.15	3914.36	268.81	7433.32
	Semi-habituated	4	2307.19	682.06	341.03	2576.43	1614.69	2730.44	1294.42	2781.46
	Habituated	5	3878.49	2616.74	1170.24	4466.08	1343.29	6119.91	1328.41	7558.86

Appendix 31 – Conceptual flow chart for evaluating the use of translocation to resolve human–wildlife conflicts.

Source: Massei et al. (2010).