Farming with Predators - An agroecological approach to human-wildlife conflict on Namibian farmlands

by

Eustatius du Chavoux Luyt

Dissertation presented for the degree of Doctor of Philosophy (Conservation Ecology) in the Faculty of Agriscience at Stellenbosch University

Department of Conservation Ecology & Entomology
University of Stellenbosch
Private Bag X1, 7602 Matieland, South Africa

Promoters:

Dr. A.J. Leslie Prof. C. Hui Dr. L.M. Marker

March 2020

Declaration

By submitting this dissertation electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof (save to the extent explicitly otherwise stated), that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

Date: March 2020.....
E. du C. Luyt

Abstract

Farming with Predators - An agroecological approach to human-wildlife conflict on Namibian farmlands

Human-wildlife conflict is a worldwide and increasingly important problem. In the past most "solutions" to this problem have been either from an agricultural perspective or from a conservation perspective. As the goals of these two perspectives differ, there has been frustratingly little progress in preventing human-wildlife conflict. The "Farming with Predators" project aimed to cross this divide by designing a Decision Support System (DSS) for farmers and land managers aimed at mitigating human-wildlife conflict by combining ecological and agricultural information in a single agroecological model. In Namibia, four predator species are widely considered to be responsible for most livestock losses, namely: leopards (*Panthera pardus*), cheetahs (*Acinonyx jubatus*), caracals (*Caracal caracal*) and black-backed jackals (*Canis mesomelas*). Two of these predator species (*A. jubatus* and *P. pardus*) are classified as vulnerable in the International Union for Conservation of Nature (IUCN) Red List of threatened species. Since about 90% of Namibia's cheetahs are found on farmlands, finding solutions to conflict with farmers is a priority, for both cheetah survival and farming in Namibia. Some relevant aspects of the predators' ecology are discussed using the literature and field data. The proposed DSS makes use of a cost-benefit analysis of current anti-predation measures used by Namibian farmers (using survey data) as well as ecological data on habitat preferences and the spatial behaviour of the predators, predator prey preferences and predator ecological interactions by using a Bayesian Network approach (Probabilistic Graphical Model). Important gaps in our knowledge that require further research in order to address farmer-predator conflict are highlighted in the process. If implemented and as farmers start using the program, the model can be updated with new data in order to enable greater accuracy. Some practical applications of agroecological principles in real-world farmer-predator conflict are also discussed.

Uittreksel

Farming with Predators - An agroecological approach to human-wildlife conflict on Namibian farmlands

Mens-wilde-dier-konflik is 'n wêreldwye en al groter probleem. Die meeste "oplossings" vir hierdie konflik in die verlede was óf uit 'n landbou óf uit 'n ekologiese oogpunt. Omdat die twee perspektiewe se doelwitte verskil, was daar frustrerend min vordering in pogings om die konflik te verhoed. Die "Boer saam met Roofdiere" projek wou hierdie skeiding oorbrug deur 'n besluitnemingsondersteuningstelsel te ontwerp vir boere en grondbestuurders wat mens-roofdier konflik verlig deur ekologiese en landboukundige inligting in een agroekologiese model te kombineer. In Namibië word 4 roofdierspesies wyd beskou as verantwoordelik vir die meeste veeverliese: luiperds (*Panthera pardus*), jagluiperds (*Acynonix jubatus*), rooikatte (*Caracal caracal*) en rooijakkalse (*Canis mesomelas*). Twee van hierdie spesies (*A. jubatus* en *P. pardus*) word as kwesbaar geklassifiseer in die IUCN se Rooi Lys van bedreigde spesies. Aangesien omtrent 90% van Namibiese jagluiperds op plase voorkom, is dit 'n prioriteit om oplossings vir konflik met boere te vind, beide vir jagluiperd oorlewing en volhoubare boerdery in Namibië. Relevante aspekte van die roofdiere se ekologie word bespreek vanuit die literatuur en veld data. Die voorgestelde besluitnemingsondersteuningstelsel gebruik bestaande teen-roofdier maatreëls van Namibiese boere se koste-doeltreffendheid (uit vraelys-data), ekologiese data oor habitatvoorkeure en ruimtelike gedrag van roofdiere, prooivoorkeure van roofdiere en roofdier ekologiese interaksies in 'n Baysiese Netwerk benadering (Grafiese Waarskynlikheidsmodel). As die program geskryf is en deur boere gebruik word, kan die model opgedateer word vir beter akkuraatheid. Die praktiese toepassing van agro-ekologiese beginsels in werklike boer-roofdier konflik situasies word ook bespreek.

Acknowledgements

This research project was funded by a NRF Innovation Doctoral Scholarship. I want to thank the Namibian Ministry for the Environment and Tourism (MET) for the permit to do the research. Many thanks to Rolf Ritter for the privilege of doing most of the field work on his farm and providing free lodging for me and all my co-workers. To the CCF for providing camera traps and N/a'an ku sê for making available their GPS collar data, many thanks. To my research assistant, Mariska, and trackers, Tjimbinae, Ismael and Eric, all my thanks for the hours of hard work in hot weather.

Dedications

To Mariska, my family for all their support and patience, and the One in whom, through whom and for whom all things were created, my heavenly Father.

Contents

List of Figures

List of Tables

List of Abbreviations

AM	Adaptive management
CCF	Cheetah Conservation Fund
CCT	Conservation conflict transformation
CPD	Conditional probability distribution
CTA	Conditioned taste aversion
DSS	Decision support system
EBC	Evidence-based conservation
EU	Expected utility
F.I.T.	Footprint Identification Technique
GIGO	Garbage in, garbage out
GPS	Global Positioning System
HWC	Human-wildlife conflict
KDE	Kernel Density Estimate
LCMAN	Large Carnivore Management Association of Namibia
LGD	Livestock Guarding Dog
LoCoH	Local Convex Hull
MCP	Minimum Convex Polygon (Convex Hull)
MET	Ministry of Environment and Tourism
MEU	Maximum Expected Utility
IUCN	International Union for Conservation of Nature
PGM	Probabilistic Graphical Model
SSI	Sexually selected infanticide
T-LoCoH	Time-inclusive Local Convex Hull
UD	Utilization distribution

List of Symbols

Operators:

$\bigvee$	(logical) OR (disjunction)
$\neg$	(logical) NOT (negation)
$\bigcap$	(logical) AND (conjuction)
$\mid$	(logical) given (conditional)

Chapter 1

Human-predator conflict: The need for a new approach

Abstract

Human-wildlife conflict is a growing global problem, and for many large carnivores even a greater threat than habitat destruction (Treves and Bruskotter, 2014). For both the economic survival of livestock farmers and the persistence of relatively species-rich ecosystems on livestock farms, it is important that solutions to farmer-predator conflict be found (Kinnaird and O'Brien, 2012). Here a brief overview of farmer-predator conflict globally and in Namibia is given. I contend that there are three reasons why this problem has not been solved yet: 1) The social aspect of breakdown in trust between farmers and environmentalists, 2) the risk to farmers of choosing an inappropriate predation management solution and the lack of adequate information for decision-making, 3) the gap between the agricultural and conservation ecological approaches to livestock depredation. An agroecological framework is proposed within which predator-livestock conflict can be mitigated using an adaptive management approach. A Decision Support System (DSS) is then introduced as a most useful tool to achieve this conflict mitigation aim within the adaptive management and agroecological frameworks. It is shown how a DSS can avoid some of the most troubling pitfalls of current approaches to human-wildlife conflict. The main information requirements for writing such a DSS is introduced and finally an overview is given of how each chapter in this thesis contributes to the final DSS.

Keywords: Adaptive management, agroecology, Decision Support System, farmer-predator conflict, human-wildlife conflict, Namibia, sustainability

1.1 Introduction – Human-wildlife conflict

Extensive livestock farming is ultimately dependent on sustainable ecosystem services for its survival as a viable economic activity (Bowe, 2000). For both the economic survival of livestock farmers and the persistence of relatively species-rich ecosystems on livestock farms (Kinnaird and O'Brien, 2012), it is important that solutions to farmer-predator conflict be found. The main aim of this "Farming with Predators" research project was to mitigate human predator conflict on Namibian farmlands by designing a Decision Support System (DSS) tool for farmers and other land managers, thus contributing to the continual survival of predators on Namibian farmlands and providing the basis for similar approaches elsewhere. Therefore it concentrated on those Namibian predator species that have a significant probability for conflict with humans as a result of livestock predation, i.e. cheetah (*Acinonyx jubatus*), leopard (*Panthera pardus*), caracal (*Caracal caracal*) and black-backed jackal (*Canis mesomelas*) (Marker-Kraus et al., 1996). The DSS is based on an agroecological view of the whole

ecosystem, including the farming system and ecological interactions between species, in order to benefit both wildlife and farmers.

Terminology – This project specifically addresses the problem of human-wildlife conflict (HWC) that results as a consequence of livestock depredation (Dickman et al., 2014) on Namibian commercial farms. Thus conflict that results from crop-damaging wildlife (Hill, 2004; Regmi et al., 2013) or from the killing of humans by wildlife (Wallace et al., 2011; Suthar et al., 2018) does not fall within the scope of this thesis. Similarly, human-carnivore conflict in the context of game farming, although important in Namibia, is not considered here since most of the possible solutions to predator-livestock conflict will not be applicable to game farming (Marker-Kraus et al., 1996; Lindsey et al., 2013b). "Commercial farming" as used here, refers to farms belonging to private individuals whose main or only source of income is farming, managed as a profit-making enterprise, rather than for subsistence only. The terms "human-carnivore conflict" (Khorozyan et al., 2015), "farmer-predator conflict" (Potgieter et al., 2016), "human-predator conflict" (Du Plessis et al., 2018), and "predator-livestock conflict" (Carruthers and Nattrass, 2018) have all been used to describe this conflict with slight differences in emphasis. In this text they are used interchangeably with "human-wildlife conflict" (HWC), with the understanding that as used here, it concerns conflict arising from livestock depredation — the continual cycle of livestock killed by predators and predators killed by humans in response. The term "predation" is used when referring to the action by predators and "depredation" when referring to the effect of the action as experienced by prey (including livestock) (Hanson, 2006).

1.1.1 Human-wildlife conflict – a global conservation problem

Human-wildlife conflict is a growing global problem (Messmer, 2000; Treves and Karanth, 2003b; Nyhus et al., 2005). The greatest threat to large carnivores, such as mammalian carnivores and sharks, is direct killing by humans (e.g. Treves and Bruskotter, 2014; Ripple et al., 2014), posing an even greater immediate threat than habitat destruction, especially in Africa (Ray et al., 2005). This includes killing of carnivores in retaliation for livestock depredation (Ogada et al., 2003 – See text box above for definitions).

Predators fulfil an important role in functioning ecosystems, and their extermination has many unintended consequences, including trophic cascades (Ripple et al., 2014) and extinctions (Estes et al., 2011; DeCesare et al., 2010), mesopredator release (Ritchie and Johnson, 2009), and savannah ecosystems becoming dominated by more thorny trees and shrubs as herbivore densities change (Ford et al., 2014). Ultimately, the persistence of ecosystem services (Reiss et al., 2009; CPW, 2014; Grace et al., 2016) and the stability of ecosystems as such, are dependent on their biodiversity (Cadotte et al., 2012; Hautier et al., 2015). There is good reason to believe that predators are a major driver of high biodiversity through top-down processes and the interactions of predation with competition, including apparent competition (Estes et al., 2011; Terborgh, 2015). Apparent competition, where a shared predator results in an indirect interaction between two prey species, has been shown to exist where apex predators are shared by lower-level predators (DeCesare et al., 2010; Kamler et al., 2012b). For example, because of their role in maintaining high biodiversity as apex predators, leopards (*Panthera pardus*), has been considered as a reliable indicator of a healthy ecosystem (Pitman, 2012).

Large carnivores, often apex predators, need large home ranges or territories to sustain themselves, making them more vulnerable to extinction (Cardillo et al., 2004; Schipper et al., 2008; Estes et al., 2011). The large home ranges and higher trophic levels of larger carnivores imply *inter alia* that they generally occur at lower densities. When in conflict with humans, the movements of individuals over larger areas also increase the probability of lethal encounters with humans (Ripple et al., 2014). Additionally, a high percentage of the habitat of many predators is outside conserved areas, including tigers (*Panthera tigris*), jaguars (*Panthera onca*) and snow leopards (*Panthera uncia*) (Miquelle et al., 1999; Nowell & Jackson, 1996; quoted in Dickman et al., 2011). Not only are they more vulnerable to direct killing by humans, but their reliance on larger prey for their survival, also makes them more vulnerable to extinction if their prey base is extirpated by hunting (Carbone et al., 2007;

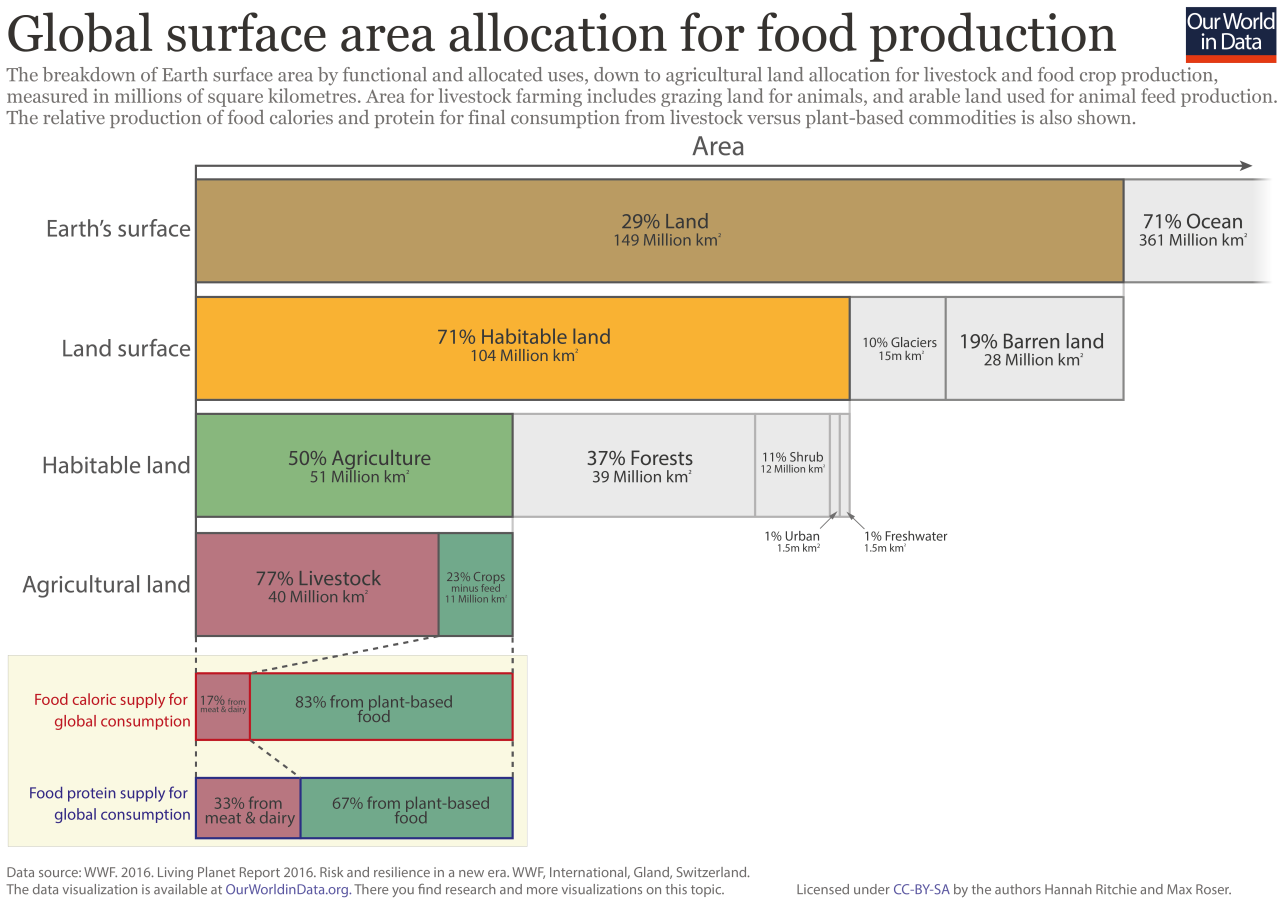


Figure 1.1: *Global human land use.* Agriculture represents the single largest land use, with livestock farming constituting 77% of that (with credit to H. Ritchie and M. Roser, CC-BY-SA).

Brook et al., 2008). This increased vulnerability is evidenced by the observation that historically, predator extermination efforts almost always resulted in the extermination of top predators first (e.g. free-roaming lions, *Panthera leo* and spotted hyaenas, *Crocuta crocuta* in Southern Africa – Shortridge, 1934; Stuart et al., 1985; Beinart, 1998; Carruthers and Nattrass, 2018).

Since habitat destruction is the major cause of extinctions worldwide (Brook et al., 2008; Schipper et al., 2008), and farmers are the custodians of the majority of land in most countries of the world (Figure 1.1; WWF, 2016), the survival of many species are directly in the hands of farmers.

> "...the farm has become the habitat of surviving forms of wildlife. Wildlife conservation can, therefore, only be effective with the support and good will of the farming community. The South African farmer, the descendent of pioneering stock, is a rugged individualist and the master on his own property. Conservation measures cannot be enforced, they can only be introduced on a basis of cooperation and mutual understanding. While the majority of farmers are prepared to accept wild animals as residents on their farms, they will not tolerate undue crop damage or losses of livestock." – Dr. Douglas Hey, former director of the Department of Nature Conservation in the Cape Province (Hey, 1964).

1.1.2 Human-wildlife conflict – a global agricultural problem

In addition to the conservation issues, conflict between predators and livestock farmers is a significant problem for agriculture as well. For example, in the USA there has been a decline of >85% in sheep (*Ovis aries*) numbers since 1942 (Berger, 2006), with a 25% decrease just in the years 1991-1996 (Knowlton et al., 1999). This has been ascribed mostly to the impact of predation (Knowlton et al., 1999; Shelton, 2004). Moreover, in spite of

killing more than 286 000 large carnivores (> 9 kg body mass) in 1989, this did not have any effect on sheep numbers (Berger, 2006). Similarly, in South Africa, sheep numbers have declined to 68% of their 1980 numbers by 2018 (Turpie and Akinyemi, 2018).

A telephone survey by Van Niekerk (2010) showed that livestock farmers in South Africa claimed a total annual direct loss of livestock to predators of R 1 390 453 062 (US$198 254 970 at 2010 exchange rates) (Bergman et al., 2013). These losses were unequally distributed with some districts and farmers having much higher losses than the average. In addition to direct losses, Howery and DeLiberto (2004) make the point that behavioural changes in livestock because of predation risk, can cause a significant fall in production and reproduction rates. Table 1.1 gives a summary of some of reported livestock losses in Southern Africa. From this data alone the large variance in reported livestock losses can be seen clearly, demonstrating the need for more reliable reporting and more intensive mitigation measures in some areas (also see Turpie and Akinyemi, 2018 for more on the economic impact in South Africa).

Predation management is sometimes centralised with specific government agencies responsible for (mostly lethal) predator management on a country-wide scale (e.g. USA and Australia – Du Plessis et al., 2018) and individual farmers responsible for implementing any additional non-lethal methods. This was also the situation in South Africa before 1990, with government subsidies for fencing and general support for predator management (Carruthers and Nattrass, 2018). In other parts of the world (e.g. Europe) most of the policy changed from extermination to policies aimed at the conservation of threatened predators, with hunting prohibited or severely restricted and farmers encouraged to use non-lethal predation management methods (Du Plessis et al., 2018).

1.1.3 Human-wildlife conflict – a Namibian agricultural problem

In Namibia, like South Africa, there has been a similar shift from government policy of either reducing predator numbers or eradicating them (Beinart, 1998; Pringle and Pringle, 1979) to farmers becoming responsible for managing livestock depredation themselves (Carruthers and Nattrass, 2018; Ruppel and Ruppel-Schlichting, 2013; Swanepoel, 2016). This situation, where individual landowners are responsible for managing the livestock depredation on their own land, is unlikely to change soon. Therefore, this project focused on the commercial livestock farmers in Namibia, keeping in mind that some of the same methods could work in other areas with similar circumstances and predators. Game farming and the conflict caused by the depredation of valuable game animals, are also important issues in Namibia, but differ in important aspects from livestock farming considered here. Similarly, in communal farming the implementation of most anti-predation management is dependent on community consensus (Turpie and Akinyemi, 2018; Hangara et al., 2011a), requiring a radically different approach.

Namibia is an arid country with one of the lowest human population densities on earth. It is therefore not well suited for crop farming except in the far North where mostly communal subsistence farming can be found (Mendelsohn et al., 2003; Chiriboga et al., 2008; MAWF, 2017). Livestock farming is thus the main source of livelihood in most of Namibia with the agricultural sector being the greatest provider of employment in the country (Chiriboga et al., 2008) and the second most important source of income for many households in Namibia (MAWF, 2017). The agricultural sector is the fourth highest contributor to the GDP, after mining, manufacturing and construction, with livestock farming providing 62% of this contribution (MAWF, 2017). In this context livestock depredation is seen as an increasingly important problem for agriculture and one reason for Namibian small livestock farmers switching to cattle (NAMMIC, 2011).

1.1.4 Human-wildlife conflict – a Namibian conservation problem

Green et al. (2018) mention that local costs – including increased wildlife damage – often exceed the costs of managing biodiversity for the benefit of the global community, while Nyhus et al. (2005) point out that

Table 1.1: *Some published Southern African livestock losses:* Small and large livestock have been combined into a single equivalent Livestock Unit (LSU) value with 10 small livestock equivalent to one LSU. All studies were based on losses reported by farmers, with ***n*** indicating the number of farmers included in the study. Where available, the minimum and maximum reported losses are also indicated. The price per kilogram of meat of small livestock can be up to twice that of cattle (*Bos taurus*), but the ratio is not constant. The total number of livestock per farm is also not taken into account, ignoring the fact that the *percentage* of livestock loss can be more significant than the average number of LSU lost per annum. In the Marker et al., 2003b study, losses were divided into farmers who considered cheetahs as a problem and those who did not (but not how many farmers were in each group).

Study	Where	Period	Average loss/farm/year (FAO LSU)	Notes (Predators)	Data collection method
Conradie and Piesse (2013)	Ceres South district, SA	1979-1987	0.15 (range: 0.14-11.4, n=152)	Small livestock only (Caracal and Leopard)	Hunting club trapper/hunter logbook
Bailey and Conradie (2013)	Mossel Bay district, SA	1976-1981	0.094 (range: 0.0047-0.23, n=43)	Small livestock only (Caracal and Jackal)	Hunting club hunter logbook
Stein et al. (2010)	Waterberg, Namibia	2005-2006	1.75 (n=19)	Average annual loss: US$1 370.00 (leopards)	Face to face interviews
Marker et al. (2003b)	North-Central Namibia	1991-1993	10.65 LSU | 4.55 LSU (n=241)	Large and small livestock combined (All)	Face to face interviews
Marker et al. (2003b)	North-Central Namibia	1993-1999	3.75 LSU | 1.81 LSU (n=241)	Large and small livestock combined (All)	Face to face interviews
Lindsey et al. (2013a)	Wider Namibia	2010-2011?	1.82 (n=250)	Period not specified; US$2 644.00 (leopards)	Face to face interviews
Van Niekerk (2010)	South Africa	2006-2007	11.45 LSU (n=1424)	Small livestock only (Caracal and Jackal)	Telephonic interviews

recognising this fact is the basic motivation for conservationists implementing compensation schemes. In the final instance, while everybody benefits from a functioning and biodiverse ecosystem providing various ecosystem services, the cost of conserving large and sometimes dangerous animals outside protected areas is often borne disproportionately by farmers with wildlife having a direct impact on their livelihood (Nyhus et al., 2003, 2005; Green et al., 2018).

Four carnivore species are generally considered as responsible for most conflict involving livestock in Namibia: black-backed jackals, caracals, cheetahs and leopards (Marker-Kraus et al., 1996; Marker et al., 2003b; Stein et al., 2010). Lions, spotted hyaenas and African wild dogs (*Lycaon pictus*) are also involved in HWC, but have been extirpated from most of the Namibia's commercial farmlands (Stander, 1990; Lindsey et al., 2013a). Brown hyaenas (*Hyaena brunnea*), while wide-spread and commonly persecuted, are not considered as major predators of livestock (Van der Merwe et al., 2009; Stein et al., 2010, 2013).

The global threat posed by HWC to the survival of large predators is exemplified by the cheetah (*Acinonyx jubatus*), classified as vulnerable in the IUCN Red List (Durant et al., 2015). In Namibia, the country with the largest surviving population of cheetahs in the world (Durant et al., 2015), an estimated 90% of cheetahs are found on commercial livestock farms (private ranches) outside formally protective reserves (Marker, 2000; Marker et al., 2003a). Being shot on livestock farms is the major cause of death of cheetahs (Marker, 2000; Marker et al., 2010). Similarly, in Southern Africa, most of the vulnerable leopard (*Panthera pardus*) habitat (Pitman, 2012; Swanepoel et al., 2013; Stein et al., 2015) is found outside protected areas and the major threat to leopard survival is direct killing by people (Swanepoel et al., 2015). In Namibia, even protected species may be killed legally to defend human life or livestock (Ruppel and Ruppel-Schlichting, 2013). In this context, solving farmer-carnivore conflict in Namibia can be seen as critical for the future survival of these large carnivore

species.

1.1.5 The three possible endpoints of farmer-predator conflict

In any conflict between farmers and predators on earth, there are only three logically possible endpoints:

1. The conflict continues, humans "win" and predators are extirpated on farmlands (this was the result in most of Europe and for top predators such as lions and spotted hyaenas on most Southern African farms). In the light of the importance of Namibian farmlands for the survival of more than one threatened carnivore species, this is not a realistic or desirable outcome.

2. The conflict continues, predators "win" and livestock farming becomes unsustainable on the land, forcing a switch to other agricultural activities such as tillage and crop production or non-agricultural land uses. In the light of the importance of livestock farming in the Namibian economy and the previously mentioned unsuitability of its climate for crop farming, this is also undesirable.

3. The conflict is managed and mitigated somehow so that farmers and carnivores learn to co-exist. This could be achieved through either lethal or non-lethal conflict mitigation methods. The aim is a win-win situation for both livestock and predators which is obviously the optimum outcome.

1.2 Why has human-wildlife conflict still not been resolved?

Despite the fact that various manuals with multiple possible solutions to human-wildlife conflict are widely available (e.g. Linnell et al., 1996; Shivik et al., 2003; Schumann, 2004; Shivik, 2004; Daly et al., 2006; Shivik, 2006; Smuts, 2008; Stone et al., 2008; Chardonnet et al., 2010; Van Bommel, 2010; Begg and Kushnir, 2013; Paige, 2015; Kerley et al., 2018), farmer-predator conflict is still a growing problem (Messmer, 2000; Hoogesteijn and Hoogesteijn, 2010). Three major explanations for why the conflict is continuing can be identified:

1.2.1 Social conflict and trust issues

A number of different surveys have shown that greater perceived depredation livestock losses generally correlates with more predators being killed and more negative attitudes towards predators, although this was not always the most important predictor of farmers' tolerance of predators (Ogada et al., 2003; Marker et al., 2003b; Shivik et al., 2003; Lindsey et al., 2013a; Rust et al., 2013; Dickman et al., 2014). Other studies have shown that in some cases the actual number of livestock losses to predators have little influence on farmer attitudes (Thorn et al., 2012; Babrgir et al., 2017; Thorn et al., 2015) and compensation for losses does little to change livestock owners' attitudes (Montag, 2003; Naughton-Treves et al., 2003; Madden and McQuinn, 2014). One possible explanation for this is that the risk of depredation losses can influence the attitude of a livestock owner just as much or even more than actual losses (Potgieter, 2011; Thorn et al., 2015; Conradie and Piesse, 2016; Miller et al., 2016).

Rather than livestock losses, Naughton-Treves et al. (2003) showed that a "deep-rooted social identity" was a better predictor of predator tolerance than compensation received (see Thorn et al., 2012, 2015). This led a number of researchers to conclude that much of the conflict is between different groups of people, rather than between people and wildlife (Montag, 2003; Treves and Karanth, 2003a; Conradie and Piesse, 2013; Nattrass and Conradie, 2015). Madden and McQuinn (2014) showed that material, visible manifestations of human-wildlife conflict are often rooted in less visible, more complex social conflicts between people and different stakeholder groups (e.g. farmers and environmentalists). This also explains why cultural and social aspects often have a greater influence on farmer attitudes towards predators than actual livestock losses (Thorn et al., 2012, 2015)

and require the use of "conflict transformation" tools to help solve these underlying, complex social conflicts (Madden and McQuinn, 2014). Conservation conflict transformation (CCT) is the adaptation of principles and processes from the field of peace-building to these kinds of conservation conflicts (Madden and McQuinn, 2014). This involves addressing conflict between stakeholders at three levels: 1) the most superficial level of *dispute* which addresses the *substance* of the disagreement and can be *settled* by finding a mutually agreed upon resolution to the cause of conflict (typically, conflict mitigation methods addresses this level of HWC); 2) the deeper level of *underlying conflict* which is the result of a history of unresolved conflict and mistrust and needs to be *resolved* by having an agreed upon quality *process* of decision-making which addresses issues of equity and authority among stakeholders; 3) the deepest level of *identity-based deep-rooted conflict* which addresses *relationships* requiring *reconciliation* and rebuilding respect and trust (Madden and McQuinn, 2014).

Where farmers feel excluded from the decision-making process about predators that will ultimately influence their livelihoods, the situation often deteriorates into one where farmers "shoot, shovel and shut up" (Hoogesteijn and Hoogesteijn, 2010; various Namibian farmers, personal communication). It is therefore not enough to know what method is most likely to succeed (CCT dispute level) – unless farmers are empowered to make the decision for themselves using well-researched, relevant knowledge (CCT process addressing underlying conflict), the uptake of more sustainable and cost-effective methods for preventing farmer-predator conflict will probably remain low (Snow, 2009; Pooley et al., 2016).

Montag (2003) mentions that "much of the conflict (about predator management) is around the control of landowner land, government intervention, and private land rights", also hinted at by Rust et al. (2016) and Rust (2016) (at the level of CCT identity-based, deep-rooted conflict). Madden and McQuinn (2014) mention that conservationists and governments often resist giving up decision-making control, because they already have the law on their side or they may fear what will happen when stakeholders who seem less committed, or even antagonistic to conservation objectives, are given a legitimate voice in decision-making. Similarly, farmers often resent environmentalists (or non-farmers in general) telling them "how to farm" (Montag, 2003; personal observation). However, in practice, working together is much more likely to result in win-win solutions (sustainable coexistence). For example, compensation schemes for livestock depredation from areas as far apart as Namibia, Turkmenistan and Mongolia which had some level of success, all had livestock owners participating actively in the decision-making process and sharing ownership of the final solutions (Stander et al., 1997; Lukarevsky, 2003; Mishra et al., 2003).

The final managers and implementers of any method used to reduce HWC are the farmers themselves (Yoder, 2000). The current increase in cheetah numbers in Namibia (Marker et al., 2007; Purchase et al., 2007) can be directly ascribed to an increase in tolerance of cheetahs by farmers (Marker et al., 2003b). However, conflicts with farmers also remains the main threat to the long-term survival of Namibian cheetahs (Durant et al., 2015). This ambiguity between farmers on the one hand being responsible for the Namibian cheetah population increase and simultaneously being the greatest threat to cheetah survival, demonstrates that farmers are not a homogeneous group and that different farmers approach livestock depredation differently (Rust, 2016). It also highlights the important point made by Pooley et al. (2016) that since both positive and negative interactions are involved, framing the relationship between humans and carnivores in terms of "conflict" alone can be unprofitable for reaching a win-win situation.

1.2.2 Silver bullet or scattergun

A number of researchers have mentioned that predation losses do not seem to be uniformly distributed, even within the same district (Conradie and Piesse, 2013), and some farmers might have extremely high depredation losses, while the majority have relatively few losses (Van Niekerk, 2010; Turpie and Akinyemi, 2018). This appears to be an international feature of farmer-predator conflict, with similar reports for example, from Slovakia (Rigg et al., 2011) and the USA (Shelton, 2004). These differences in levels of depredation imply that different

management decisions and different anti-predation methods will be appropriate on different farms. It highlights the need for a decision-making process that is appropriate for local circumstances.

Past approaches very seldom (if ever) took cognisance of the differences between individual farms and farmers with regards to both ecological and agricultural aspects of HWC. The approach has often been that of "one size fits all" (Schumann, 2004), where a predation management method that had been successful before, is presented as the proverbial *silver bullet* to solve human-carnivore conflict (Blackwell et al., 2016; Du Plessis et al., 2018). Blackwell et al. (2016) makes it clear that no such single solution exists; there is no silver bullet (Linnell et al., 1996). Farmers are quick to realise when a proposed HWC mitigation method will be impractical in their situation, and lose confidence in all alternative approaches to predation management. Krafte Holland et al. (2018) showed that even *in the scientific literature*, management techniques are often recommended without their effectiveness having been evaluated.

While often making the mistake of offering a "one size fits all" solution to farmers for their livestock depredation problems, the opposite mistake is made as well. This happens when scientists simply suggest a whole toolkit of possible (usually non-lethal) methods for farmers to use (Shivik, 2004; Daly et al., 2006; Shivik, 2006; Smuts, 2008; Stone et al., 2008; Chardonnet et al., 2010), without giving them any comparative guidance on the effectiveness, costs and limitations of the different methods. This approach is similar to using a *scattergun*, with the hope that one of the methods will hit the target. This makes it just as difficult, if not impossible, for a farmer to choose the most suitable method(s) for his particular farm.

Conservationists are often guilty of not spelling out the known drawbacks and limitations of proposed solutions to livestock depredation, thereby creating unrealistic expectations (see for example Smuts, 2008; Daly et al., 2006). Then, when farmers run into unexpected issues, they become discouraged and return to their known and trusted traditional management practices – even if ineffective and unsustainable (Nattrass and Conradie, 2015). Not only do they personally abandon any of the possibly more effective methods, but typically they also spread the word that a certain method "did not work for me", thus discouraging other farmers from attempting it (Shivik et al., 2003). Since farmers are naturally risk-averse, this can result in the permanent continuation of farmer-predator conflict (Conradie and Piesse, 2016). A decision-making process is needed that includes local conditions and makes explicit all disadvantages and disadvantages of the different mitigation methods in order to avoid both the "silver bullet" and the "scattergun" pitfalls. A Decision Support System (DSS) can facilitate this process (see Section 1.5 below on page 14).

1.2.3 The gulf between agriculture and ecology

Nattrass and Conradie (2015) showed that livestock farmers and conservationists approach the problem of farmer-jackal conflict from two opposite starting points (narratives). Livestock farmers generally want to maximize production or profits per hectare of land and minimize their risk (Conradie and Piesse, 2016). Environmentalists generally want to maximize biodiversity. However, both perspectives commonly share the following aims: 1) minimize conflict, 2) minimize cost, and 3) maximize sustainability. Since greater biodiversity also leads to greater sustainability (Cadotte et al., 2012), this means that farmers and conservationists share most aims, with the exception of maximizing production and minimizing livelihood risk that are unique to the agricultural viewpoint (Figure 1.2). One would therefore expect them to be natural allies, with ranchers (livestock farmers) considered as partners in conservation rather than opponents (Hoogesteijn and Hoogesteijn, 2010). As mentioned above, this partnership often does not materialize.

With a few exceptions (e.g. McManus et al., 2014), most published methods to mitigate HWC have been either from an agricultural perspective, with little regard for the sustainability or ecological consequences of the methods (Bergman et al., 2013; Shelton, 2004) or from a conservation ecological point of view, with little regard for or mention of the relative cost-effectiveness and actual practicality of the methods (CapeNature, Downloaded: 2016/09/15; Daly et al., 2006; Smuts, 2008). Even when the cost-effectiveness of the individual

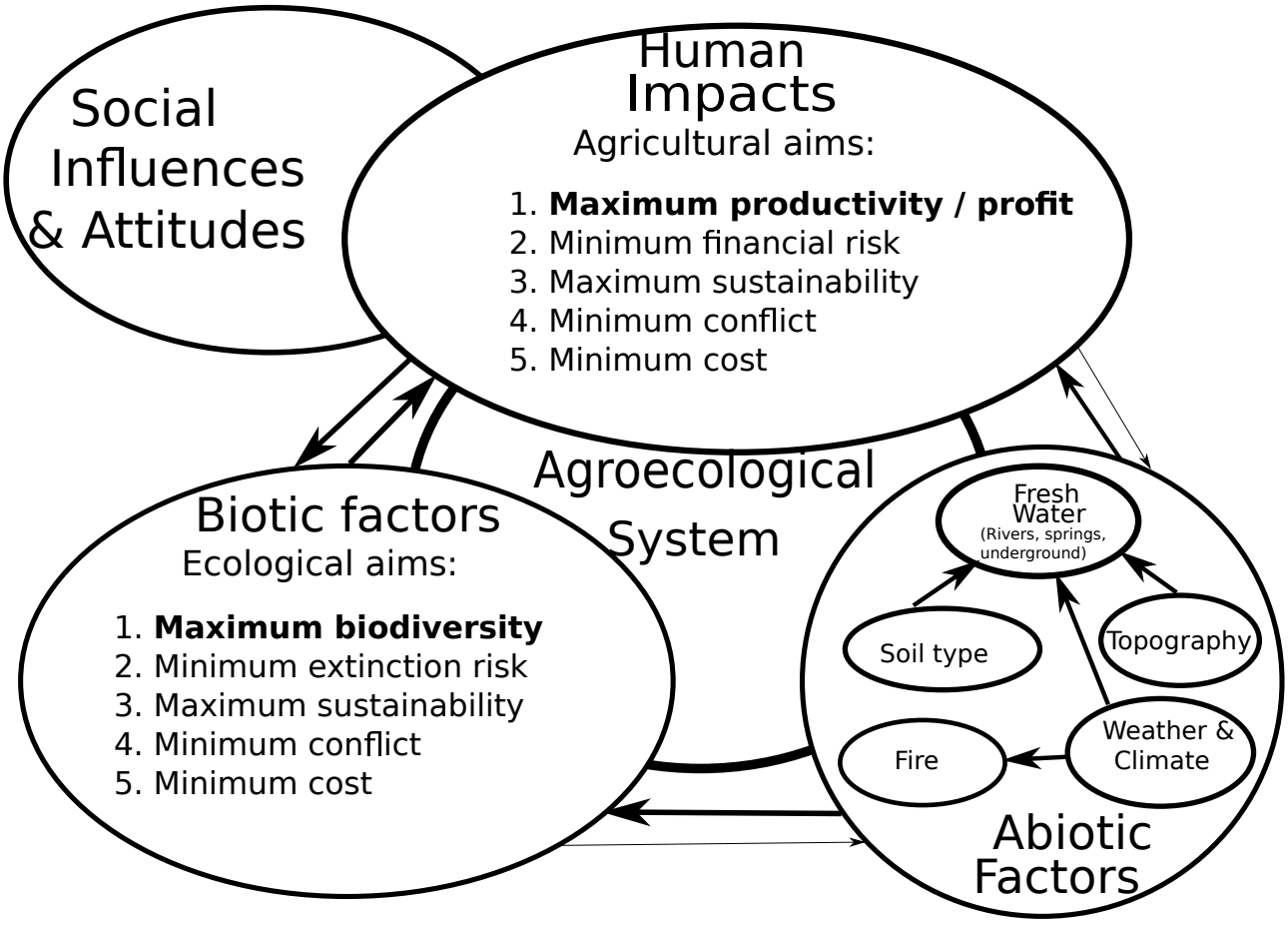


Figure 1.2: *Agricultural and ecological aims of predation management on a livestock farm.* Figure 1.4 further explains the holistic agroecological framework within which these aims are to be realised. The abiotic factors of the ecosystem are mostly a given. The biotic factors of the ecosystem should be managed to maximize biodiversity and minimize extinction risk (ecological perspective). The farming system should be managed to maximize production and minimize financial risk (agricultural perspective). The thickness of the arrows indicates the relative strength of the causal influence in a certain direction (see text box on page 12).

methods is considered, there is little consideration for the fact that methods that work well on some farms, will often fail in different agricultural circumstances. Shivik (2006) opined that future predation management methods "need to emerge from a mix of biology, sociology, and technology". Snow (2009) also made the point that a full system-based approach is more likely to find the links of cause and effect than when HWC is considered in isolation.

The common gulf between farmers' requirements of cost-effective, practical and farm-specific mitigation methods and conservationists' responsibility to preserve biodiversity all too often prevents such holistic system-based approaches. The result is that conservationists frequently advise farmers to use eco-friendly methods that are often impractical for their situation or not economically viable (Ellins, 1985; Shivik et al., 2003). Agriculturalists, on the other hand, often prefer methods that produce quick results, but are not sustainable in the long run and might actually aggravate the situation (Conradie and Piesse, 2013; Ogada, 2014; Nattrass and Conradie, 2018). One obvious solution would be a decision-making tool (e.g. DSS) that includes both the agricultural and the ecological perspectives in a single model.

1.3 An agroecological framework

As mentioned above, HWC has often been approached from either a purely conservation ecology point of view (Messmer, 2000; Marker, 2002) or from an agricultural production point of view(Shelton, 2004; Bergman et al.,

2013). Various researchers (Madden and McQuinn, 2014; Nattrass and Conradie, 2015) have shown that this often results in stakeholder groups being in opposition to each other and human-human conflict that can no longer be solved by means of purely ecological principles. Mason et al. (2018) thus consider HWC as one example of "wicked problems: intractable problems embedded in complex systems that are difficult to define and lack clear solutions". Differences in stakeholder values are central to wicked problems, and one of the roots of conservation conflict, a viewpoint that coincides with the "identity-based deep-rooted conflict" level of Madden and McQuinn (2014).

Sinclair et al. (2006) made the point that "all wildlife problems have to be addressed in the context of the whole ecosystem". The gulf between ecology and agriculture can be avoided by combining the two viewpoints into a single multidisciplinary agroecological approach. The term "agroecology" was first used by Bensin in 1928 and 1930, and while originally a scientific discipline, the term was later also used for an agricultural practice and a socio-political movement (Wezel et al., 2009). As a scientific discipline, it is probably best defined as the ecology of human food production systems (Francis et al., 2003). As explained by Conway (1985), "In agricultural development, ecosystems are transformed into hybrid agroecosystems for the purpose of food or fibre production" (Figure 1.3). In the past, agroecology mostly focused on crop production systems and only recently began to also consider the sustainability of livestock production systems (Wezel and Peeters, 2014; Cayre et al., 2018). While having a relatively long history in Europe and the Americas, virtually nothing has been published on African agricultural ecosystems using the term agroecology (Wezel et al., 2009) (e.g. ISI Web of Science found only a single article, from 2019, when using the search terms "((agroecology) AND (Africa))").

A conceptual model of the agroecological system on livestock farms is shown in Figure 1.4. The individual farm is considered the basic management unit on commercial farms in Namibia. The top sphere of human impacts corresponds mostly with the agricultural management viewpoint of the farming system with predation management as just one part of wildlife management on the farm. The two bottom spheres of biotic and abiotic factors represents the ecosystem within which the farming enterprise functions and on which it ultimately depends (Lindeman, 1942). On most Namibian livestock farming systems the mega-herbivores have been extirpated, the wild herbivore species have been reduced and partly replaced by livestock, and the apex predator species have been reduced or extirpated and partly replaced by humans (hunting and livestock off-take – see Figure 1.3) (Tambling et al., 2018). Both of these changes probably contributed to subsequent bush encroachment, changing the primary producer level (De Klerk, 2004). The causal interactions between the various factors are discussed in the text box under Figure 1.4 on page 1.4. As humans are strictly speaking part of the biotic aspect of an ecosystem, the interactions between the biotic and human spheres are strong in both directions.

Social, political and peer attitudes, while having an important influence, function at a larger scale than the individual farm; they are therefore not included in this model as part of the human management impact. For informing management and decision-making at the level of the individual farmer, they can be considered as a given external constant, which is not influenced significantly by management decisions of the individual farmer. Communal systems will require a different model with the socio-political aspects at least partly subsuming the agricultural human impacts identified here, since most decisions require buy-in from the whole community (Turpie and Akinyemi, 2018). Given the importance of individual commercial livestock farmers in Namibia for both the decision-making and implementation of anti-predation methods on their land, this project will focus on the ecosystem and agricultural systems.

The main contribution of this agroecological framework is to provide a holistic view of farmer-predator conflict at the level of the individual farm, within the context of the whole farming ecosystem, management aims (Figure 1.2) and decision-making. This should help to bridge the common gap between the agricultural approach and the ecological approach to solving HWC. It also helps to avoid simplistic solutions which do not adequately address all the different aspects that are important in successful predation management (Mason et al., 2018). This agroecological framework also addresses some of the major themes identified by Mason et al. (2018) for solving wicked problems:

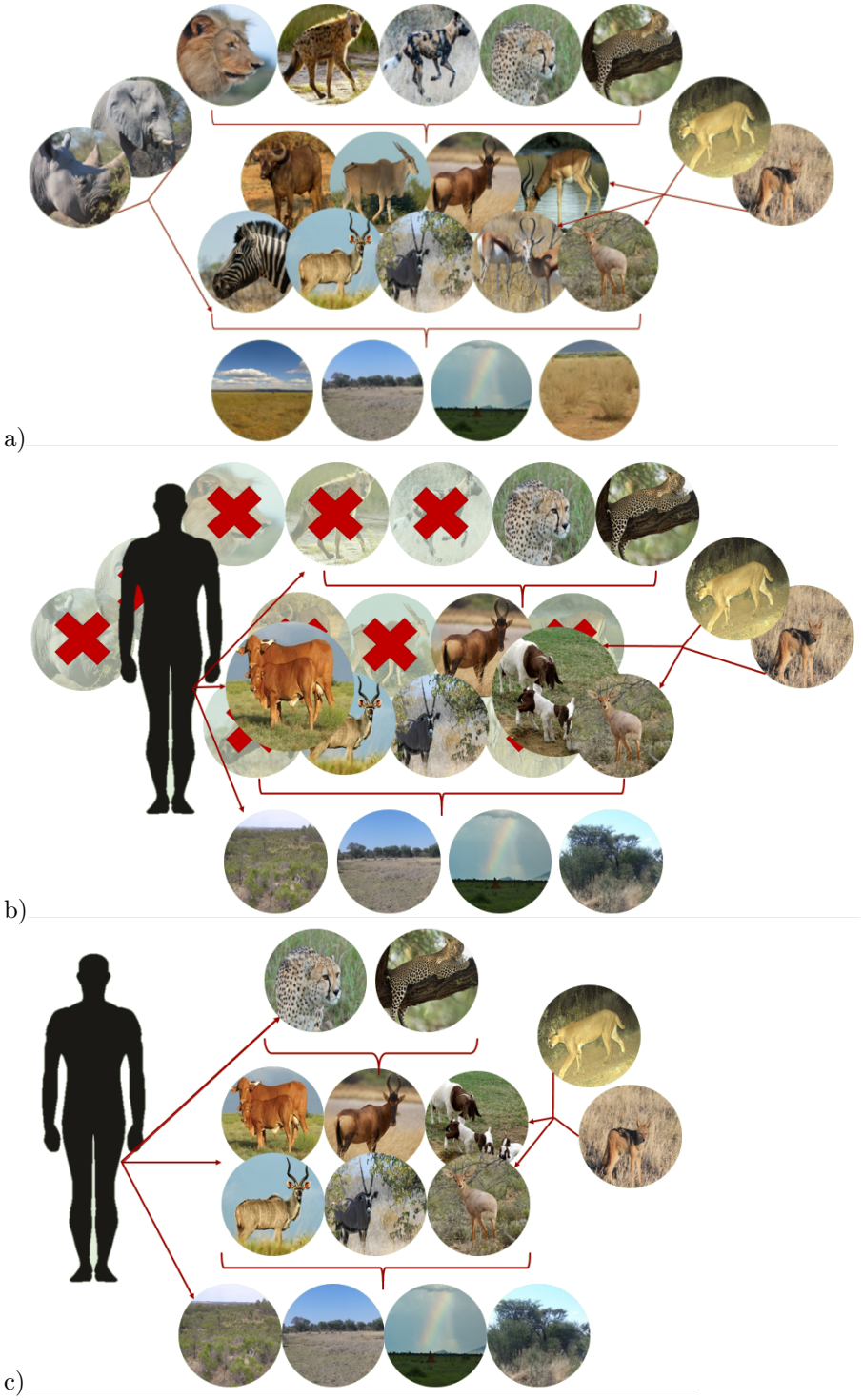


Figure 1.3: *Historical ecosystem change on Namibian farms.* Illustrating the change mentioned by Conway (1985) from an intact, biodiverse ecosystem (a), through the extermination of several large species and replacing some with livestock (b), to a typical hybrid argoecosystem (c).

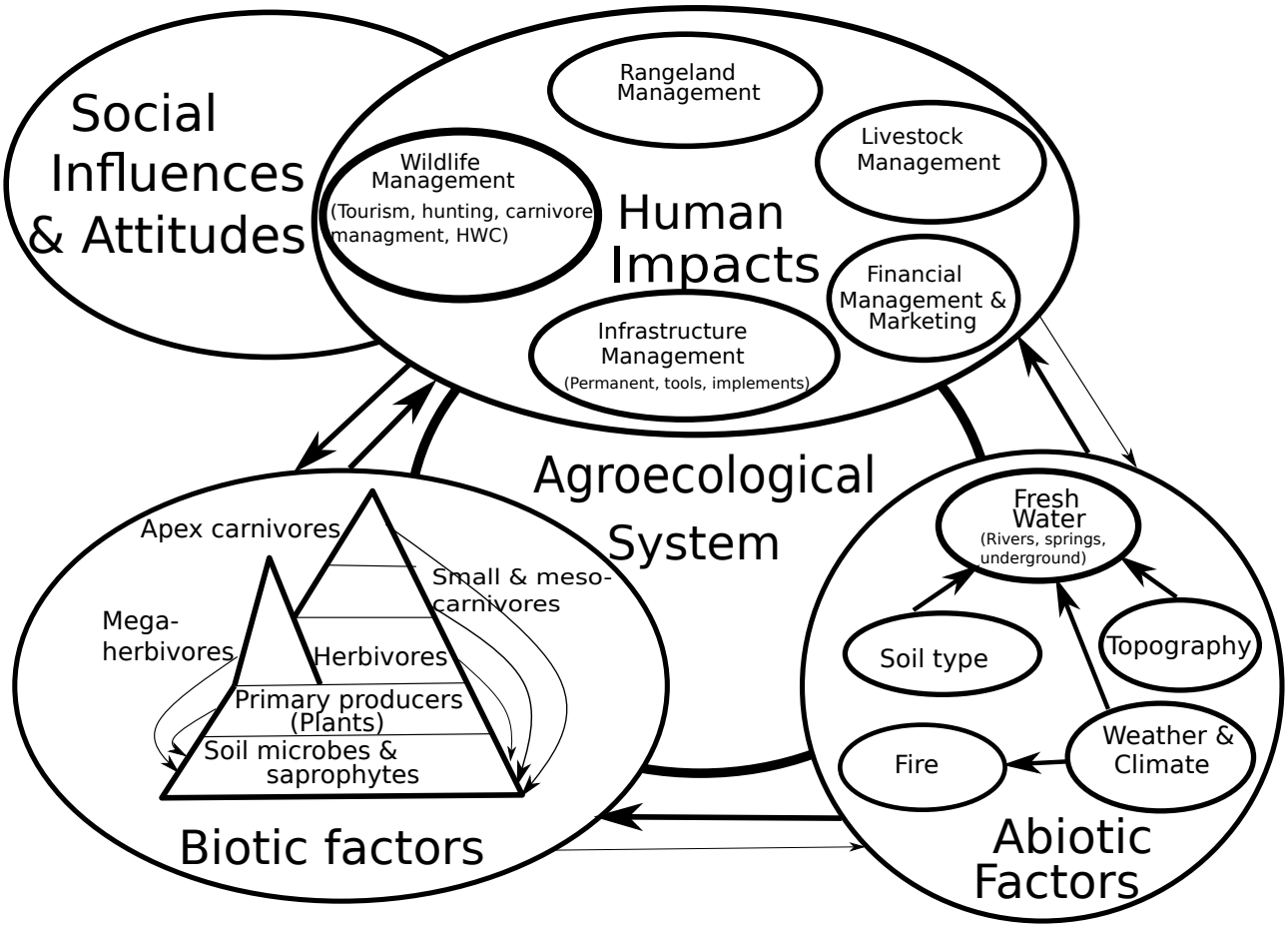


Figure 1.4: *An agroecological model for livestock farms in Namibia*, showing the interactions between variables, with thick arrows indicating strong interactions and thin arrows indicating weak interactions. Social influences and attitudes, while important, are outside the control of the individual farmer. HWC management or predation management is part of the larger farm management system (Du Plessis et al., 2018). Abiotic aspects of the ecosystem have a strong influence on both the farming system and the biotic factors (indicated by thick arrows), but are generally much less influenced by human management or biotic factors in an extensive farming system. On the other hand, biotic factors and farm management have a strong influence on each other. Biotic factors can be represented by an ecological pyramid (Lindeman, 1942). Since mega-herbivores (when occurring on farmland) are generally not subject to predation, they are represented by an extra apex to the pyramid.

Causal impacts in the model – This model entails that abiotic factors like the topography of the farm, rainfall, and available water will influence and limit the farmer's management options (e.g. drinking water availability will influence infrastructure like dams, pipes, pumps; climate and weather will affect the need for shade or shelters for livestock, rainfall and soil type will influence the possibility of ploughing and crop farming, etc.). The abiotic factors have a strong impact on human management aspects, while human management has relatively little effect on most abiotic factors (except fire, mostly by humans controlling wildfires). Similarly the abiotic factors strongly affects the primary producers (plants) of the biotic subsystem (e.g. soil type, topography and water table largely determine vegetation structure and species – Bell, 1982 – and are affected by rainfall in turn). The biotic factors have relatively little effect on the abiotic factors (e.g. some erosion caused by overgrazing – Hobbs, 1996). The biotic factors have a strong influence on almost every aspect of the human management system, however. Rangeland management directly has to do with the condition (quantity), species composition, diversity and nutritional quality of the vegetative component of the biotic subsystem (Tainton, 1999; Rohde and Hoffman, 2012). Livestock management, in turn is affected by some abiotic factors like drinking water and topography, but also by the biotic primary producers (plants) as food and carnivores as potential predators to be avoided (Wezel and Peeters, 2014). Livestock as the primary source of income, in turn influences the financial management of the farm, which is also influenced by the need for infrastructure and wildlife as additional income (McGranahan, 2008; Lindsey et al., 2013b; Turpie and Akinyemi, 2018). Infrastructure like fences and artificial water points in turn can strongly influence the spatial behaviour of wild herbivores and the predators that prey on them (McGranahan, 2011). Wildlife management more directly impacts on the biotic aspects of the agroecological system by hunting, (re)introducing game species, managing livestock depredation (HWC) and in turn is influenced by the movement of wildlife, their densities and species diversity (see Chapter 2).

1. Distributed decision-making – by having individual local farmers as the principle decision-makers,
2. Creative management practices, suited to the specific problem – by providing a framework within which farmers can share their experiences (since farmers are naturally risk-averse (Conradie and Piesse, 2016), they tend to learn new creative practices from each other),
3. Diverse expertise – by including both ecology and agricultural science in the decision-making,
4. Adaptive management and pattern-based evidence – by providing a framework within which different types of evidence can be placed and used to adapt management decisions,
5. Outcome-focused objectives – by explicitly stating the major shared and unique aims of both the agricultural and ecological management (Figure 1.2),
6. Trade-offs in objectives – by acknowledging that there may be trade-offs between the different aims,
7. Sharing failures – by providing a framework within which farmers can share the failures of various mitigation methods (whether it was a failure of the method itself, or a mismatch with the farmer's specific agricultural and/or ecological system).

1.4 Sure science or adaptive management – the knowledge gap

1.4.1 Evidence-based conservation

Sutherland et al. (2004) made the point that too much conservation practice is based on anecdotal evidence without solid scientific evidence and proposed the need for evidence-based conservation (EBC). This included the use of centralised, searchable, web-based databases with results from conservation projects and interventions. This data would include an indication of reliability and be collated using systematic reviews of the literature and meta-data analysis (Sutherland et al., 2004). They highlighted the lack of long-term monitoring for reaching meaningful conclusions on the effectiveness of various methods used in conservation, as a major short-coming of many research projects (Sutherland et al., 2004; O'Neill, 2007). The main thrust of evidence-based conservation is that management decisions should be based on "sure science" with interventions validated and demonstrated to be effective in the long run and by using "the accepted standard of scientific inference (random assignment or quasi- experimental case- control) without bias in sampling, treatment, measurement, or reporting" (Treves et al., 2016).

There has been appeals that carnivore conservation and predator control should be evidence-based (Treves et al., 2016; Krafte Holland et al., 2018; Van Eeden et al., 2018). It can be expected that farmers will be reluctant to use novel methods for which there is little hard evidence that it will work (Conradie and Piesse, 2016). Since first being proposed, an evidence-based conservation approach has been used a number of times to address various conservation issues (e.g. long-term dynamics of an ecosystem, impact of conservation interventions for leopards, evidence-informed conservation policy, predator-proof bomas to prevent lion predation – Sinclair et al., 2007; Balme et al., 2009; Dicks et al., 2014; Lichtenfeld et al., 2015; Lieury et al., 2015; Dicks et al., 2018). Balme et al. (2014) observed that there is still not enough relevant research to address the range-wide conservation needs of larger carnivores, considering HWC and the percentage of carnivores found outside protected areas. Du Plessis et al. (2015) found the same lack of data for addressing HWC involving black-backed jackals and caracals on farmlands.

Akçakaya et al. (2016) pointed out that it is not just a lack of data that can cripple the usefulness of management decisions, but that there is a need to relate different models with each other. Wimsatt (1994) showed that many scientific questions about cause and effect involved interrelated causes at different levels and perspectives (scientific disciplines) that demonstrate a "causal thicket" making it almost impossible to tease apart the strands

of individual cause-and-effect relationships, and resulting in uncertain, probabilistic inferences. It is these intertwined interrelationships between different aspects of HWC that caused Mason et al. (2018) to conclude that HWC is a "wicked problem" that has to be approached holistically.

1.4.2 Adaptive management

In contrast to the "sure science" approach of evidence-based conservation, adaptive management (AM) has emerged as the best available approach for managing biological systems in the presence of uncertainty (Westgate et al., 2013). Adaptive management has been defined as "a systematic approach for improving resource management by learning from management outcomes" (Westgate et al., 2013). An important part of the AM approach is creating several competing models of system function and then ascertaining which model best explains change in the system following management interventions (Westgate et al., 2013). Testing different models can be done either by actively using all alternatives, including those that are suspected of being ineffective (active AM) or by simply evaluating those management interventions that are already being used over time (passive AM). The primary focus of active AM is to increase knowledge, while passive AM focus on using solutions that are more likely to solve the problem and evaluating them over time (McDonald-Madden et al., 2010; Westgate et al., 2013).

A major problem experienced in the past (see the review by Westgate et al., 2013) is that for various reasons the whole AM process cycle has often not been followed through all the way to include long-term monitoring and feedback on the effectiveness of the management interventions. For instance, the proposed adaptive management of leopard trophy hunting by Balme et al. (2010) describes the start of an AM process, but leaves us in the dark about whether this approach was ever implemented and what the long-term outcome has been – these questions would need to be answered in follow-up studies. This highlights the same lack of information on intervention success that is also suffered by evidence-based conservation (Sutherland et al., 2004; Treves et al., 2016). Gillson et al. (2019) made the point that EBC and AM are typically more useful for different types of problems:

1. EBC is useful for "best practice", where there is enough scientific evidence and agreement between stakeholders and can be used with relative confidence by managers.

2. AM is more useful for situations of high uncertainty and/or high disagreement between different stakeholders, but where various interventions are evaluated with only a probabilistic expectation of success.

More importantly, EBC can be used as part of the initial planning of an AM approach to evaluate the known (published) interventions in terms of evidence for their effectiveness and in turn contribute to the evidence for EBC after every monitoring and evaluation stage of the adaptive management cycle. "The strength of AM is in embracing complexity and uncertainty, whereas EBC can contribute to the subset of problems where there is good-to-moderate understanding of the problem or subproblem and agreement over the desired outcomes" (Gillson et al., 2019). In the context of HWC, with large differences between stakeholders (section 1.2.1 above), uncertainty about the effectiveness of different predation management methods (Du Plessis et al., 2018) and significant gaps in our ecological knowledge of the predators involved in farmer-predator conflict (Balme et al., 2014; Du Plessis et al., 2015), an adaptive management approach fits the problem better.

1.5 A Decision Support System

The aim of adaptive management in farmer-predator conflict should be to find the most cost-effective and sustainable method or combination of methods that fit the situation on a specific farm (*i.e.* including both agricultural and ecological aims and impacts – Figure 1.2 & Figure 1.4). Schwartz et al. (2018) propose the use of decision support frameworks to increase planning rigour, project accountability, stakeholder participation,

transparency in decisions, and learning, during all three planning stages of scoping, planning and learning as standard practice for both practice and research.

One tool that potentially offers a way out of the current impasse, is the use of a decision support system (DSS) (Keen, 1980; Sprague, 1980) based on a Bayesian Probabilistic Graphical Model (PGM) (Charniak, 1991; Koller and Friedman, 2005; Jøsang, 2008). A similar approach would be to use an Expert System (Collopy and Armstrong, 1992). In both cases a PGM or similar probabilistic model could provide the "engine" of the software system. The most important differences between the two in the context of HWC are that, 1) a DSS provides the user with the relevant information to enable him to make an informed *choice* (in this case, the most appropriate predation management method) while the expert system simply provides a single *best solution* to the problem; 2) in a DSS the knowledge of the user contributes to the answer that is reached and *information is generated* through the process, while an expert system is meant for non-expert users in order to mimic a human expert and to provide them with the solution that a human expert is most likely to reach; all the *expert knowledge is pre-built* into the expert system; 3) a DSS is generally meant to support decisions where the boundaries and exact *requirements are unclear*, while an expert system is usually restricted to *a single, well-defined domain* (Shafer and Pearl, 1990). To a certain extent the differences between a Decision Support System and an expert system are similar to that between evidence-based conservation (expert system) and adaptive management (Decision Support System) and share many of the same advantages and disadvantages respectively (e.g. a lack of reliable information precludes a reliable expert system, while a DSS can still improve the decision-making process, albeit with some uncertainty).

PGMs are typically used for one of three purposes (Pearl and Russell, 2000; Koller and Friedman, 2005):

1. Causal reasoning (prediction). From the causal model, predictions are made of what will happen given a certain set of observed preconditions. An example of this kind of use, is the population viability analysis that was done for Namibian cheetahs using an object-oriented Bayesian network (Johnson et al., 2013). Decision Networks (also known as Influence Diagrams) adds decision nodes and value nodes to a basic Bayesian Network to predict the expected utility of decisions (Shafer and Pearl, 1990; Charniak, 1991; Koller and Friedman, 2005).

2. Evidential reasoning (inference). This involves adductive reasoning from observed effects to likely causes (Harman, 1965; Constantini, 1987; Aliseda, 2007).

3. Learning (machine learning). This involves using newly observed data to update prior probabilities of the models and even to the models themselves (Karkera, 2014; Barber, 2015; Bellot, 2016). Probabilistic Graphical Models have been used for learning models in biology, albeit not for solving human-wildlife conflict (Friedman, 2004).

1.5.1 Addressing short-comings of past approaches

Using a decision support system addresses most of the reasons discussed above for why past human-carnivore conflict mitigation approaches still fail to reduce the conflict, as follows:

1.5.1.1 Sidestepping trust issues

Most past approaches to HWC directly address the human-predator conflict itself and only recently started taking into account the social aspects of the conflict (section 1.2.1 above). But even these recent studies often focus on social aspects of the conflict that might at best be a contributing factor (e.g. labour relations, Rust et al., 2016), while ignoring the deeper levels of basic conflict between the ecological and the agricultural starting points (Nattrass and Conradie, 2015). This difference in priorities may be the most important reason why farmers are unwilling to risk or trust in "new" methods (Conradie and Piesse, 2016).

While the DSS itself does not resolve pre-existing underlying conflict nor addresses reconciliation, it enables the farmer or land manager to "sidestep" the breakdown in trust caused by past bad experiences with some environmentalists (see sections below – Impersonal, Open Source, Localised, Learning & Training). Because the DSS approach emphasizes participatory decision-making, rather than simply an expert opinion, it is also preferable to an expert system regarding social conflict between farmers and conservation experts. Farmers associations and agricultural unions using the DSS can share ownership of ecologically sound solutions to human-wildlife conflict (*cf.* Lukarevsky, 2003).

Impersonal: Because a DSS does not take the decision-making process out of the hands of farmers, and because it is not a person "telling the farmer what to do on his own farm", at least part of the common underlying distrust between conservation scientists and farmers is avoided (Axel Rothauge 2014, personal communication). If the DSS is written in order to be transparent with regards to the algorithms and data it uses, it makes it easier to be trusted. Because the farmer is not told what to do, but can make his own decision based on the information provided, with only some probability of success, there is less chance for all trust in the system to be lost or a return to previous unsustainable management techniques (as happened with some farmers in the study by McManus et al., 2014).

Open Source: Transparent, correctable and expandable If the DSS is written as open source software, it can be updated and improved as new knowledge and research becomes available. Because farming is inherently risky, farmers tend to be risk-averse and to keep doing what they know (Conradie and Piesse, 2016). However, circumstances have changed and keep on changing (e.g. climate change, market fluctuations, government support). Affordable and effective methods of the past, are no longer affordable or effective (Turpie and Akinyemi, 2018). By making these changes explicit, the PGM can help farmers realize why their familiar methods are no longer cost-effective and why other alternatives might be better. Moreover, by making the utilized probabilistic graphical model explicit, both farmers and conservationists can challenge or suggest improvements to any aspects of the model with which they disagree.

Localised decision making: Farmers themselves often feel left out and not well represented as partners in predator management decisions and advice (e.g. Bergman et al., 2013 when commenting on Daly et al., 2006). The importance of farmers as full partners in predator conservation on farmlands, and being the final implementers of any HWC mitigating methods, as well as having to personally bear the brunt of all costs and risks, can be at least partially recognized by providing them with a Decision Support System. This will enable them to make informed decisions with regards to which methods are more appropriate for their own level of livestock depredation losses.

Learning from data: Ultimately the DSS would use feedback from farmers themselves who are using the various methods, to re-evaluate or update the baseline data used (e.g. as costs change or if more limitations of a specific method are found). An automatic feedback loop can thus be built into the DSS, allowing it to adapt to changing circumstances or to refine the accuracy of its model as more farmers use it and expand the baseline data.

Training: Rust (2016) found that the single most popular method for mitigating HWC in Namibia was conservation education and husbandry training to reduce livestock depredation. Both farmers and behavioural ecologists still have great gaps in the knowledge of predator behaviour outside protected areas (Balme et al., 2014; Du Plessis et al., 2015). As farms have become larger in order to remain economically viable and farming methods are often more extensive than they used to be (Nattrass and Conradie, 2015), farmers often also know surprisingly little about the behaviour of their own livestock (especially their anti-predator behaviour). This

lack of knowledge can result in basic mistakes (like de-horning of all cattle), leading to unnecessary livestock losses.

According to Marker-Kraus et al. (1996), none of the predation management methods used on farms in Namibia made a significant difference to livestock depredation losses, except for the numbers of wildlife on the farm. One possible explanation for this result might be that the quality of execution of protection measures is more important than the choice of method. If true, this confirms the need for knowledge and training. Since implementation details of the various conflict mitigation methods would be included as part of a DSS, training in the use of the DSS will also include general training on predator behaviour and HWC management techniques.

1.5.1.2 Holistic bridging of the breach

Current methods often approach the conflict from either a conservation-ecological viewpoint or from a short-term agricultural economics viewpoint (section 1.2.3 above). Conservationists may advise farmers to use eco-friendly methods that are impractical or not economically viable (Linnell, 2000; Daly et al., 2006; Treves et al., 2016), while agriculturalists may prefer unsustainable methods that produce quick results and might aggravate the situation (Santangeli et al., 2016; Nattrass and Conradie, 2018). The DSS would address this by combining the two aspects, short-term cost-effectiveness and long-term sustainability, into a single agroecological PGM.

Record-keeping: One of the most important aspects of applying any predation-prevention method, is the good record keeping of the current situation on the farm (Stone et al., 2008). Knowing how many livestock are lost to predators, where most depredation happens on the farm and which predators are responsible, is basic knowledge that is required in order to make good management decisions. Hoogesteijn and Hoogesteijn (2010) actually list it as a preventative method by itself. As farmers become familiar with the DSS, the input it requires will highlight those important aspects of their farming system (the agricultural part of the agroecological model) that requires better bookkeeping. A smartphone application can be written (and later included as part of the DSS) that will help farmers to record depredation (and other) livestock losses, as well as for general farm record keeping (replacing the traditional pocket-book and pen – *cf.* Appendix C).

1.5.1.3 Situation-specific

Both the "scattergun approach" (Shivik, 2004; Daly et al., 2006; Shivik, 2006; Smuts, 2008; Stone et al., 2008; Chardonnet et al., 2010) and the "silver bullet approach" (Linnell et al., 1996; Schumann, 2004; Blackwell et al., 2016), leaves the farmer unable to choose the most appropriate method(s) for his particular farm and circumstances (section 1.2.2 above). Considering the individual farm as the management unit and explicitly listing the disadvantages and limitations of the various management options, will help individual farmers to choose a situation-specific approach to predation management.

The farm as management unit: The probabilistic model (PGM) can use the actual situation on an individual farm as input (given observations) to calculate the most cost-effective and sustainable solutions for that farm. Instead of giving a whole gamut of possible solutions the DSS will present only a small number (e.g. three) of the "best" conflict mitigating methods, based on the input of the specific farmer. When the reason for choosing these methods, as well as the advantages and disadvantages of each are explicitly stated, it becomes much easier for the farmer to make an informed choice.

Avoiding unrealistic expectations: To avoid needless discouragement, it is important that the advantages, disadvantages and limitations of the various HWC mitigation methods should be known up-front before a farmer commits himself to implementing it on his farm. Not only unrealistic expectations, but also the incorrect

implementation of methods can end in failure. This will have the same negative end result of farmers losing confidence and returning to familiar methods that are unsustainable or ineffective. By including the most important known disadvantages and limitations of the various predation management methods in all DSS results, this issue can be avoided (the different methods are described in Chapter 2). These data can also be updated from farmer inputs, as mentioned above.

1.5.2 Requirements for writing a decision support system

The most important requirement for writing a useful DSS is a reliable and robust agroecological model. The agroecological model should be based on data that is scientifically as reliable as possible, not only to produce reliable results, but also to inspire confidence in the potential users of the DSS. Given the fact that there are large knowledge gaps and uncertainty about many aspects of human-carnivore conflict in Namibia, the model should also be as robust as possible. Wimsatt (1994) defined the idea of robustness as things being "robust if they are accessible (detectable, measurable, derivable, definable, producible, or the like) in a variety of independent ways." In the context of an agroecological model this implies that we want a model that remains relatively reliable in the presence of uncertainty; so that even if a few of the input values are uncertain, it will not make a large difference to the result. One way to achieve this is by having the *relative* probabilities of success or failure of each method be reliable, so that even if the *actual* probability estimates are not accurate, the result will remain unchanged. Both agricultural and ecological data will inform the agroecological PGM (see Du Plessis, 2013).

1.6 Conclusions - The way forward

In order to achieve the aim of designing a DSS that can provide farmers with the relevant information to decide on the best way to manage livestock depredation on their own farm, a number of objectives were set at the start of the project. These objectives generally addressed either the agricultural or the ecological part of the agroecological model.

1.6.1 Agricultural objectives

The questions relevant to the agricultural aspect of the model are addressed in Chapters 2 and 3 of the thesis. Primarily, the advantages and disadvantages of different anti-predation methods were to be determined.

1. Determine which anti-predation methods are currently used by Namibian farmers.
2. Determine current livestock losses by Namibian farmers.
3. Determine any changes in predator numbers reported by Namibian farmers on their land.
4. Determine current stocking rates and/or farming intensity on Namibian farms.
5. Determine relative costs of each anti-predation method used in Namibia.
6. Determine the advantages and disadvantages of anti-predation methods used worldwide from an agricultural perspective (short-term cost-effectiveness).
7. Determine the relative effectiveness of each method from an agricultural viewpoint.

1.6.2 Ecological objectives

The questions that are relevant to the ecological part of the model are addressed in Chapters 2, and 4 - 5 of the thesis. All aspects of the behavioural ecology of each predator species can potentially be relevant to predation management.

The most obviously relevant ecological facet was diet and prey preference. Factors like habitat and prey availability will also influence individual predator diet.

1. Determine prey preferences of the four predator species.
2. Determine the individual degree of variance of prey preference within species.

The spatial ecology of the predator species has obvious implications for predation management.

1. Determine home range sizes for breeding/territorial males and females of each predator species.
2. Determine the relative proportion of resident and non-resident individuals in the population of each predator species.
3. Determine habitat use within their home ranges – determine if each species has areas within its home range or territory that are used more than the rest.
4. Determine if there are specific habitat areas within home ranges that are used for specific purposes.
5. Determine dispersal distances for each predator species.

Where a single predator species is responsible for the vast majority of livestock losses, predator intra-guild interactions can be relevant to livestock depredation management.

1. Determine if there is evidence of subordinate predator species showing spatial avoidance of dominant species or of spatial niche partitioning between the predator species (the effect of "fear and loathing" – Ritchie and Johnson, 2009).
2. Determine if there is any evidence of mesopredator release where larger predators have been extirpated (Palomares and Caro, 1999; Ritchie and Johnson, 2009).

1.6.3 Structure of the thesis

1.6.3.1 Chapter 2 - Literature review

This chapter consists of two parts:

1. A review of the different published methods used to mitigate human-predator conflict. A short overview is given of the different ways in which conflict mitigation methods have been classified in the past and a classification that is most appropriate for use in a DSS. For each method the advantages, disadvantages and limitations as applicable to Namibia is provided.
2. An important part of the adaptive management approach is the various theoretical underpinnings that is used by creating several competing models of system function and determining which model best explains change in the system following management interventions (Westgate et al., 2013). A review of different theoretical underpinnings that are applicable to the ecology of human-wildlife conflict on Namibian farms is given. It includes an overview of the conservation status and behavioural ecology of the four predators involved in most of the conflict in Namibia. The conservation status of the predator species directly impacts which management techniques will be appropriate. As much of the required ecological input for the DSS as possible will be acquired from the literature.

This chapter will provide the basic information given to farmers to support their decisions on the most appropriate predation management method for their situation.

1.6.3.2 Chapter 3 - Agricultural perspective on HWC in Namibia

This chapter discusses the results of a farmer survey on predation conflict in Namibia. Some of the missing agricultural data in the literature that is required as input to the DSS, are supplemented by these results. It also puts the survey results within the context of previously published research.

1.6.3.3 Chapter 4 - Ecological perspective on HWC in Namibia

This chapter focuses on the spatial ecology of the two larger predator species for which GPS collar data was available (leopards and cheetahs). It includes a short review of methods to determine home range, as well as the home range sizes, habitat preferences, spatial use of home ranges, spatial evidence of competition and differences between the two species as found from GPS data of predators that were collared in response to farmer conflict.

1.6.3.4 Chapter 5 - The PGM design for HWC

This chapter pulls together the agricultural and ecological data from the other chapters into a single probabilistic graphical model. The design of the DSS and the model that it would use to determine the most appropriate predation management method for a specific farm, is described. The current limitations of and future improvements to the model are discussed.

1.6.3.5 Chapter 6 - Conclusions

The overall results of the Farming with Predators project and future directions for further research, are discussed.

1.6.4 Standing on the shoulders of giants – What do we see?

Solving human-wildlife conflict involving predators and livestock farmers is a complicated process. Much has been done already, but frustratingly little progress has been made. Part of the reason for past frustrations has been the lack of a holistic and interdisciplinary conceptualisation of the problem. By using a holistic agroecological approach and combining the different relevant factors into a single probabilistic graphical model, this project could avoid some of the pitfalls from the past while building on existing knowledge to provide a baseline framework for future research in solving human-wildlife conflict.

References

Akçakaya, H., Pereira, H., Canziani, G., Mbow, C., Mori, A., Paloma, M., Soberón, J., Thuiler, W., Yachi, S., 2016. Improving the rigour and usefulness of scenarios and models through ongoing evaluation and refinement. In: Ferrier, S., Ninan, K., Leadly, P., Alkemade, R., Acosta, L., Akçakaya, H., Brotons, L., Cheung, W., Christensen, V., Harhash, K., Kabubo-Mariara, J., Lundquist, C., Obersteiner, M., Pereira, H., Peterson, G., Pichs-Madruga, R., Ravindranath, N., Rondinini, C., Wintle, B. (Eds.), IPBES (2016): The methodological assessment report on scenarios and models of biodiversity and ecosystem services. Secretariat of the Intergovernmental Science-Policy Platform for Biodiversity and Ecosystem Services, Ch. 8, pp. 257–290.

Aliseda, A., 2007. Abductive reasoning: challenges ahead. Theoria: An International Journal for Theory, History and Foundations of Science 22 (3(60)), 261–270.

Babrgir, S., Farhadinia, M., Moqanaki, E., 2017. Socio-economic consequences of cattle predation by the endangered Persian leopard *Panthera pardus saxicolor* in a Caucasian conflict hotspot, northern Iran. Oryx 51 (1), 124–130.

Bailey, A., Conradie, B., 2013. The effect of predator culling on livestock losses: Caracal control in Cooper hunting club, 1976 – 1981. Tech. rep., Centre for Social Science Research, Cape Town.

Balme, G., Hunter, L., Goodman, P., Ferguson, H., Craigie, J., Slotow, R., 2010. An adaptive management approach to trophy hunting of leopards (*Panthera pardus*): a case study from KwaZulu-Natal, South Africa. In: Macdonald and Loveridge (2010), Ch. 14, pp. 341–352.

Balme, G., Lindsey, P., Swanepoel, L., Hunter, L., 2014. Failure of research to address the rangewide conservation needs of large carnivores: leopards in South Africa as a case study. Conservation Letters 7 (1), 3–11.

Balme, G., Slotow, R., Hunter, L., 2009. Impact of conservation interventions on the dynamics and persistence of a persecuted leopard (*Panthera pardus*) population. Biological Conservation 142, 2681–2690.

Barber, D., 2015. Bayesian reasoning and machine learning. www.cs.ucl.ac.uk/staff/D.Barber/brml.

Begg, C., Kushnir, H., 2013. Human-lion conflict toolkit. Wildlife Conservation Network, www.appleseedesign.com.

Beinart, W., 1998. The night of the jackal: sheep, pastures and predators in the Cape. Past & Present 158, 172–206.

Bell, R., 1982. The effect of soil nutrient availability on community structure in African ecosystems. In: Huntley, B., Walker, B. (Eds.), Ecology of tropical savannas. Springer Verlag, pp. 193–216.

Bellot, D., 2016. Learning Probabilistic Graphical Models in R. Packt Publishing.

Berger, K., 2006. Carnivore-livestock conflicts: Effects of subsidized predator control and economic correlates on the sheep industry. Conservation Biology 20 (3), 751–761.

Bergman, D., De Waal, H., Avenant, N., Bodenchuk, M., Marlow, M., Nolte, D., 2013. The need to address black-backed jackal and caracal predation in South Africa. In: J.B., A., Gallagher, G. (Eds.), Proceedings of the 15th Wildlife Damage Management Conference. Wildlife Damage Management Conference, pp. 86–94.

Blackwell, B., Devault, T., Fernández-Juricic, E., Gese, E., Gilbert-Norton, L., Breck, S., 2016. No single solution: application of behavioural principles in mitigating human-wildlife conflict. Animal Behaviour 120, 245–254.

Bowe, R., Feb. 2000. Economy vs. ecology. Quivira Coalition Newsletter 3 (2), 8–9,17.

Brook, B., Sodhi, N., Bradshaw, J., 2008. Synergies among extinction drivers under global change. Trends in Ecology and Evolution 23 (8), 453–360.

Cadotte, M., Dinnage, R., Tilman, D., 2012. Phylogenetic diversity promotes ecosystem stability. Ecology 93 (8), S223–S233.

CapeNature, Downloaded: 2016/09/15. Landowner's guide: Human-wildlife conflict - Sensible solutions to living with wildlife. http://www.capenature.co.za/wp-content/uploads/2015/05/Landowners-Guide_Human-Wildlife-Conflict.pdf.

Carbone, C., Teacher, A., Rowcliffe, J., 2007. The costs of carnivory. PLoS Biology 5 (2), 363–368.

Cardillo, M., Purvis, A., Sechrest, W., Gittleman, J., Bielby, J., Mace, G., 2004. Human population density and extinction risk in the world's carnivores. PLoS Biology 2 (7), 909–914.

Carruthers, J., Nattrass, N., 2018. History of predator-stock conflict in South Africa. In: Kerley et al. (2018), Ch. 2, pp. 30–52.

Cayre, P., Michaud, A., Theau, J., Rigolot, C., 2018. The coexistence of multiple worldviews in livestock farming drives agroecological transition. A case study in French Protected Designation of Origin (PDO) cheese mountain areas. Sustainability 10 (4), 1–13.

Chardonnet, P., Soto, B., Fritz, H., Crosmary, W., Drouet-Hoguet, N., Mesochina, P., Pellerin, M., Mallon, D., Bakker, L., Boulet, H., Lamarque, F., 2010. Wildlife management working paper 13 - Managing the conflicts between people and lion - Review and insights from the literature and field experience. Tech. rep., Food and Agriculture Organization of the United Nations.

Charniak, E., 1991. Bayesian networks without tears. AI Magazine 12 (4), 50–63.

Chiriboga, L., Kilmer, C., Fan, R., Gawande, K., 2008. Does Namibia have a comparative advantage in beef production? Tech. rep., Texas A&M University, http://kishoregawande.net/wp-content/uploads/2008/12/finalreporti.pdf.

Collopy, F., Armstrong, J., 1992. Rule-based forecasting: Development and validation of an expert systems approach to combining time series extrapolations. Management Science 38 (10), 1394–1414.

Conradie, B., Piesse, J., 2013. The effect of predator culling on livestock losses: Ceres, South Africa, 1979 – 1987. Tech. Rep. 319, Centre for Social Science Research, Cape Town.

Conradie, B., Piesse, J., 2016. Ranking perceived risk to farmers: How important is the environment? African Journal of Agricultural and Resource Economics 11 (4), 263–276.

Constantini, D., 1987. Abductive inferences. Erkenntnis (1975-) 26 (3), 409–422.

Conway, G., 1985. Agroecosystem Analysis. Agricultural Administration 20, 31–55.

CPW, Oct. 2014. Sustainable wildlife management and biodiversity. www.fao.org/forestry/wildlife-partnership.

Daly, B., Davies-Mostert, H., Davies-Mostert, W., Evans, S., Friedmann, Y., King, N., Snow, T., Stadler, H. (Eds.), 2006. Prevention is the cure. Proceedings of a workshop on holistic management of human-wildlife conflict in the agricultural sector of South Africa. Endangered Wildlife Trust.

De Klerk, J., 2004. Bush encroachment in Namibia: Report on phase 1 of the bush encroachment research, monitoring and management project. Tech. rep., Ministry of Environment and Tourism.

DeCesare, N., Hebblewhite, M., Robinson, H., Musiani, M., 2010. Endangered, apparently: the role of apparent competition in endangered species conservation. Animal Conservation 13, 352–362.

Dickman, A., Hazzah, L., Carbone, C., Durant, S., 2014. Carnivores, culture and 'contagious conflict': Multiple factors influence perceived problems with carnivores in Tanzania's Ruaha landscape. Biological Conservation 178, 19–27.

Dickman, A., Macdonald, E., Macdonald, D., 2011. A review of financial instruments to pay for predator conservation and encourage human–carnivore coexistence. Proceedings of the National Academy of Sciences of the USA 108 (34), 13937–13944.

Dicks, L., Ashpole, J., Dänhardt, J., James, K., Jönsson, A., Randall, N., Showler, D., Smith, R., Turpie, S., Williams, D., Sutherland, W., 2018. Farmland conservation. In: Sutherland, W., Dicks, L., Ockendon, N., Petrovan, S., Smith, R. (Eds.), What Works in Conservation., third edition Edition. OpenBook Publishers, DOI: 10.11647/OBP.0131, Ch. 4, pp. 245–284.

Dicks, L., Hodge, I., Randall, N., Scharlemann, J., Siriwardena, G., Smith, H., Smith, R., Sutherland, W., 2014. A transparent process for "evidence-Informed" policy making. Conservation Letters 7 (2), 119–125.

Du Plessis, J., 2013. Towards the development of a sustainable management strategy for *Canis mesomelas* and *Caracal caracal* on rangeland. Ph.D. thesis, University of the Free State.

Du Plessis, J., Avenant, N., Botha, A., Mkhize, N., Müller, L., Mzileni, N., O'Riain, M., Parker, D., Potgieter, G., Richardson, P., Rode, S., Viljoen, N., Hawkins, H.-J., Tafani, M., 2018. Past and current management of predation on livestock. In: Kerley et al. (2018), Ch. 6, pp. 125–177.

Du Plessis, J., Avenant, N., De Waal, H., 2015. Quality and quantity of the scientific information available on black-backed jackals and caracals: contributing to human–predator conflict management? African Journal of Wildlife Research 45 (2), 138–157.

Durant, S., Mitchell, N., Ipavec, A., Groom, R., 2015. *Acinonyx jubatus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T219A50649567.en, p. e.T219A50649567.

Ellins, S., 1985. Coyote control and taste aversion: A predation problem or a people problem? Appetite 6, 272–275.

Estes, J., Terborgh, J., Brashares, J., Power, M., Berger, J., Bond, W., Carpenter, S., Essington, T., Holt, R., Jackson, J., Marquis, R., Oksanen, L., Oksanen, T., Paine, R., Pikitch, E., Ripple, W., Sandin, S., Scheffer, M., Schoener, T., Shurin, J., Sinclair, A., Soulé, M., Virtanen, R., Wardle, D., 2011. Trophic downgrading of planet Earth. Science 333 (6040), 301–306.

Ford, A., Goheen, J., Otieno, T., Bidner, L., Isbell, L., Palmer, T., Ward, D., Woodroffe, R., Pringle, R., 2014. Large carnivores make savanna tree communities less thorny. Science 346 (6207), 346–349.

Francis, C., Lieblein, G., Gliessman, S., Breland, T., Creamer, N., Harwoord, R., Salomonsson, L., Helenius, J., Rickerl, D., Salvador, R., Wiedenhoeft, M., Simmons, S., Allen, P., Altieri, M., Flora, C., Poincelot, R., 2003. Agroecology: The ecology of food systems. Journal of Sustainable Agriculture 22 (3), 99–118.

Friedman, N., 2004. Inferring cellular networks using probabilistic graphical models. Science 303 (5659), 799–805.

Gillson, L., Biggs, H., Smit, I., Virah-Sawmy, M., Rogers, K., 2019. Finding common ground between adaptive management and evidence-based approaches to biodiversity conservation. Trends in Ecology and Evolution 34 (1), 31–44.

Grace, J., Anderson, T., Seabloom, E., Borer, E., Adler, P., Harpole, W., Hautier, Y., Hillebrand, H., Lind, E., Pärtel, M., Bakker, J., Buckley, Y., Crawley, M., Damschen, E., Davies, K., Fay, P., Firn, J., Gruner, D., Hector, A., Knops, J., MacDougall, A., Melbourne, B., Morgan, J., Orrock, J., Prober, S., Smith, M., 2016. Integrative modelling reveals mechanisms linking productivity and plant species richness. Nature 529 (7586), 390–393.

Green, J., Fisher, B., Green, R., Makero, J., Platts, P., Robert, N., Schaafsma, M., Turner, R., Balmford, A., 2018. Local costs of conservation exceed those borne by the global majority. Global Ecology and Conservation 14, e00385.

Hangara, G., Teweldemedhin, M., Groenewald, I., 2011a. Major constraints for cattle productivity and managerial efficiency in communal areas of Omaheke Region, Namibia. International Journal of Agricultural Sustainability 9 (4), 495–507.

Hanson, T., 2006. To predate or depredate: What's the word?. Bulletin of the Ecological Society of America 87 (2), 128–131.

Harman, G., 1965. The inference to the best explanation. The Philosophical Review 74 (1), 88–95.

Hautier, Y., Tilman, D., Isbell, F., Seabloom, E., Borer, E., Reich, P., Apr. 2015. Anthropogenic environmental changes affect ecosystem stability via biodiversity. Science 348 (6232), 336–340.

Hey, D., 1964. The control of vertebrate problem animals in the province of the Cape of Good Hope, Republic of South Africa. In: Proceedings of the 2nd Vertebrate Pest Control Conference (1964). Vertebrate Pest Conference Proceedings. http://digitalcommons.unl.edu/vpc2/11, pp. 57–70.

Hill, C., 2004. Farmers' perspectives of conflict at the wildlife-agriculture boundary: Some lessons learned from African subsistence farmers. Human Dimensions of Wildlife: An International Journal 9 (4), 279–286.

Hobbs, N., 1996. Modification of ecosystems by ungulates. Journal of Wildlife Management 60 (4), 695–713.

Hoogesteijn, R., Hoogesteijn, A., 2010. Conserving wild felids in humanized landscapes - Strategies for reducing conflicts between jaguars and cattle. Wild Felid Monitor 3 (2), 1, 10–13.

Howery, L., DeLiberto, T., 2004. Indirect effects of carnivores on livestock foraging behavior and production. Sheep & Goat Research Journal 19, 53–57.

Johnson, S., Marker, L., Mengersen, K., Gordon, C., Melzheimer, J., Schmidt-Küntzel, A., Nghikembua, M., Fabiano, E., Henghali, J., Wachter, B., 2013. Modeling the viability of the free-ranging cheetah population in Namibia: an object-oriented Bayesian network approach. Ecosphere 4 (7), 90.

Jøsang, A., 2008. Conditional reasoning with subjective logic. Journal of Multiple-Valued Logic and Soft Computing 15 (1), 5–38.

Kamler, J., Stenkewitz, U., Klare, U., Jacobsen, N., Macdonald, D., 2012b. Resource partitioning among Cape foxes, bat-eared foxes, and black-backed jackals in South Africa. Journal of Wildlife Management 76 (6), 1241–1253.

Karkera, K., 2014. Solve machine learning problems using probabilistic graphical models implemented in Python with real-world applications. Packt Publishing.

Keen, P., 1980. Decision support systems: a research perspective. Tech. rep., Alfred P. Sloan School of Management. Center for Information Systems Research.

Kerley, G., Wilson, S., Balfour, D. (Eds.), 2018. Livestock predation and its management in South Africa: A scientific assessment. Centre for African Conservation Ecology.

Khorozyan, I., Ghoddousi, A., Soofi, M., Waltert, M., 2015. Big cats kill more livestock when wild prey reaches a minimum threshold. Biological Conservation 192, 268–275.

Kinnaird, M., O'Brien, T., 2012. Effects of private-land use, livestock management, and human tolerance on diversity, distribution, and abundance of large African mammals. Conservation Biology 26 (6), 1026–1039.

Knowlton, F., Gese, E., Jaeger, M., 1999. Coyote depredation control: An interface between biology and management. Journal of Range Management 52 (5), 398–412.

Koller, D., Friedman, N., 2005. Probabilistic Graphical Models: Principles and techniques. Massachusetts Institute of Technology.

Krafte Holland, K., Larson, L., Powell, R., 2018. Characterizing conflict between humans and big cats *Panthera spp*: A systematic review of research trends and management opportunities. PLoS ONE 13 (9), e0203877.

Lichtenfeld, L., Trout, C., Kisimir, E., 2015. Evidence-based conservation: predator-proof bomas protect livestock and lions. Biodiversity and Conservation 24 (3), 483–491.

Lieury, N., Ruette, S., Devillard, S., Alabaret, M., Drouyer, F., Baudoux, B., Millon, A., 2015. Compensatory immigration challenges predator control: An experimental evidence-based approach improves management. Journal of Wildlife Management 79 (3), 425–434.

Lindeman, R., 1942. The trophic-dynamic aspect of ecology. Ecology 23 (4), 399–417.

Lindsey, P., Havemann, C., Lines, R., Palazy, L., Price, A., Retief, T., Rhebergen, T., Van der Waal, C., 2013a. Determinants of persistence and tolerance of carnivores on Namibian ranches: Implications for conservation on Southern African private lands. PLoS ONE 8 (1), e52458.

Lindsey, P., Havemann, C., Lines, R., Price, A., Retief, E., Rhebergen, T., Van der Waal, C., Romañach, S., 2013b. Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. Oryx 47 (1), 41–53.

Linnell, J., 2000. Taste aversive conditioning: a comment. Carnivore Damage Prevention News 2, 4–5.

Linnell, J., Smith, M., Odden, J., Swenson, J., Kaczensky, P., 1996. Carnivores and sheep farming in Norway. 4. Strategies for the reduction of carnivore - livestock - conflicts: a review. Tech. rep., Norsk Institutt for Naturforskning.

Lukarevsky, V., 2003. Saving the Central Asian leopard in Turkmenistan. Carnivore Damage Prevention News 6, 13–15.

Macdonald, D., Loveridge, A. (Eds.), 2010. Biology and conservation of wild felids. Oxford University Press.

Madden, F., McQuinn, B., 2014. Conservation's blind spot: The case for conflict transformation in wildlife conservation. Biological Conservation 178, 97–106.

Marker, L., 2000. Aspects of the ecology of the cheetah (*Acinonyx jubatus*) on North Central Namibian farmlands. Journal of Namibia Scientific society 48, 40–48.

Marker, L., 2002. Aspects of cheetah (*Acinonyx jubatus*) biology, ecology and conservation strategies on Namibian farmlands. Ph.D. thesis, University of Oxford.

Marker, L., Dickman, A., Jeo, R., Mills, M., Macdonald, D., 2003a. Demography of the Namibian cheetah, *Acinonyx jubatus jubatus*. Biological Conservation 114, 413–425.

Marker, L., Dickman, A., Mills, M., Macdonald, D., 2010. Cheetahs and ranchers in Namibia: a case study. In: Macdonald, D., Loveridge, A. (Eds.), Biology and Conservation of Wild Felids. Oxford University Press, Ch. 15, pp. 353–372.

Marker, L., Dickman, A., Wilkinson, C., Schumann, B., Fabiano, E., 2007. The Namibian cheetah: Status report. Cat News 3, 4–13.

Marker, L., Mills, M., Macdonald, D., 2003b. Factors influencing perceptions of conflict and tolerance toward cheetahs on Namibian farmlands. Conservation Biology 17 (5), 1290–1298.

Marker-Kraus, L., Kraus, D., Barnett, D., Hurlbut, S., 1996. Cheetah survival on Namibian farmlands. Cheetah Conservation Fund.

Mason, T., Pollard, C., Chimalakonda, D., Guerrero, A., Kerr-Smith, C., Milheiras, S., Roberts, M., Ngafack, P., Bunnefeld, N., 2018. Wicked conflict: Using wicked problem thinking for holistic management of conservation conflict. Conservation Letters 82 (6), e12460.

MAWF, 2017. Agricultural statistics bulletin - 2010-2015. Tech. rep., Directorate of Planning and Business Development, Private Bag 13184, Government Office Park, Windhoek.

McDonald-Madden, E., Probert, W., Hauser, C., Runge, M., Possingham, H., Jones, M., Moore, J., Rout, T., Vesk, P., Wintle, B., 2010. Active adaptive conservation of threatened species in the face of uncertainty. Ecological Applications 20 (5), 1476–1489.

McGranahan, D., Jul. 2008. Managing private, commercial rangelands for agricultural production and wildlife diversity in Namibia and Zambia. Biodiversity and Conservation 17 (8), 1965–1977.

McGranahan, D., Jan. 2011. Identifying ecological sustainability assessment factors for ecotourism and trophy hunting operations on private rangeland in Namibia. Journal of Sustainable Tourism 19 (1), 115–131.

McManus, J., Dickman, A., Gaynor, D., Smuts, B., Macdonald, D., 2014. Dead or alive? Comparing costs and benefits of lethal and non-lethal human–wildlife conflict mitigation on livestock farms. Oryx 49 (4), 687–695.

Mendelsohn, J., Jarvis, A., Roberts, C., Robertson, T., 2003. Atlas of Namibia - A portrait of the land and its people. David Phillip Publishers, http://www.the-eis.com/.

Messmer, T., 2000. The emergence of human-wildlife conflict management: turning challenges into opportunities. International Biodeterioration & Biodegradation 45, 97–102.

Miller, J., Jhala, Y., Schmitz, O., 2016. Human perceptions mirror realities of carnivore attack risk for livestock: Implications for mitigating human-carnivore conflict. PLoS ONE 11 (9), e0162685.

Mishra, C., Allen, P., McCarthy, T., Madhusudan, M., Bayarjargal, A., Prins, H., Dec. 2003. The role of incentive programs in conserving the snow leopard. Conservation Biology 17 (6), 1512–1520.

Montag, J., Feb. 2003. Compensation and predator conservation: Limitations of compensation. Carnivore Damage Prevention News 6, 2–6.

NAMMIC, 2011. Namibian livestock sector strategy - Final report. Tech. rep., Namibian Meat Board.

Nattrass, N., Conradie, B., 2015. Jackal narratives and predator control in the Karoo, South Africa. Journal of Southern African Studies 41 (4), 1–19.

Nattrass, N., Conradie, B., 2018. Predators, livestock losses and poison in the South African Karoo. Journal of Cleaner Production 194, 777–785.

Naughton-Treves, L., Grossberg, R., Treves, A., 2003. Paying for tolerance: Rural citizens' attitudes toward wolf depredation and compensation. Conservation Biology 17 (6), 1500–1511.

Nyhus, P., Fischer, H., Madden, F., Osofsky, S., 2003. Taking the bite out of wildlife damage : The challenges of wildlife compensation schemes. Conservation in Practice 4, 37–40.

Nyhus, P., Osofsky, S., Ferraro, P., Madden, F., Fischer, H., 2005. Bearing the costs of human-wildlife conflict: The challenges of compensation schemes. In: Woodroffe, R., Thirgood, S., Rabinowitz, A. (Eds.), People and Wildlife: Conflict or Coexistence? Vol. 9. Cambridge University Press, Ch. 7, pp. 107–121.

Ogada, D., 2014. The power of poison: pesticide poisoning of Africa's wildlife. Annals of the New York Academy of Sciences 1322 (1), 1–20.

Ogada, M., Woodroffe, R., Oguge, N., Frank, L., 2003. Limiting depredation by African carnivores: the role of livestock husbandry. Conservation Biology 17 (6), 1521–1530.

O'Neill, 2007. Conservation audits: Auditing the conservation process - Lessons learned, 2003 - 2007. Tech. rep., Conservation Measures Partnership.

Paige, C., 2015. A Wyoming landowner's handbook to fences and wildlife: Practical tips for fencing with wildlife in mind. Wyoming Community Foundation, 2nd Edition.

Palomares, F., Caro, T., 1999. Interspecific killing among mammalian carnivores. The American Naturalist 153 (5), 492–508.

Pearl, J., Russell, S., 2000. Bayesian networks. Tech. rep., MIT.

Pitman, R., 2012. The conservation biology and ecology of the African leopard *Panthera pardus pardus*. The Plymouth Student Scientist 5 (2), 281–600.

Pooley, S., Barua, M., Beinart, W., Dickman, A., Holmes, G., Lorimer, J., Loveridge, A., Macdonald, D., Marvin, G., Redpath, S., Sillero-Zubiri, C., Zimmermann, A., Milner-Gulland, E., 2016. An interdisciplinary review of current and future approaches to improving human–predator relations. Conservation Biology 31 (3), 513–523.

Potgieter, G., 2011. The effectiveness of livestock guarding dogs for livestock production and conservation in Namibia. Master's thesis, Nelson Mandela Metropolitan University.

Potgieter, G., Kerley, G., Marker, L., 2016. More bark than bite? The role of livestock guarding dogs in predator control on Namibian farmlands. Oryx 50 (6), 514–522.

Pringle, J., Pringle, V., 1979. Observations on the lynx *Felis caracal* in the Bedford district. South African Journal of Zoology 14 (1), 1–4.

Purchase, G., Marker, L., Marnewick, K., Klein, R., Williams, S., 2007. Regional assessment of the status, distribution and conservation needs of cheetahs in Southern Africa. Cat News 3, 44–46.

Ray, J., Hunter, L., Zigouris, J., 2005. Setting conservation and research priorities for larger African carnivores. Tech. rep., Wildlife Conservation Society.

Regmi, G., Nekaris, K., Kandel, K., Nijman, V., 2013. Crop-raiding macaques: predictions, patterns and perceptions from Langtang National Park, Nepal. Endangered Species Research 20, 217–226.

Reiss, J., Bridle, J., Montoya, J., Woodward, G., 2009. Emerging horizons in biodiversity and ecosystem functioning research. Trends in Ecology and Evolution 24 (9), 505–514.

Rigg, R., Findo, S., Wechselberger, M., Gorman, M., Sillero-Zubiri, C., Macdonald, D., 2011. Mitigating carnivore–livestock conflict in Europe: lessons from Slovakia. Oryx 45 (2), 272–280.

Ripple, W., Estes, J., Beschta, R., Wilmers, C., Ritchie, E., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M., Schmitz, O., Smith, D., Wallach, A., Wirsing, A., 2014. Status and ecological effects of the world's largest carnivores. Science 343 (6167), 1241484.

Ritchie, E. G., Johnson, C. N., 2009. Predator interactions, mesopredator release and biodiversity conservation. Ecology Letters 12, 982–998.

Rohde, R., Hoffman, M., 2012. The historical ecology of Namibian rangelands: Vegetation change since 1876 in response to local and global drivers. Science of the Total Environment 416, 276–288.

Ruppel, O., Ruppel-Schlichting, K. (Eds.), 2013. Environmental law and policy in Namibia - Towards making Africa the tree of life., 2nd Edition. Hans Seidel Foundation.

Rust, N., 2016. Can stakeholders agree on how to reduce human-carnivore conflict on Namibian livestock farms? A novel Q-methodology and Delphi exercise. Oryx 51 (2), 1–8.

Rust, N., Tzanopoulos, J., Humle, T., MacMillan, D., 2016. Why has human-carnivore conflict not been resolved in Namibia? Society & Natural Resources 29 (9), 1079–1094.

Rust, N., Whitehouse-Tedd, K., MacMillan, D., 2013. Perceived efficacy of livestock-guarding dogs in South Africa: Implications for cheetah conservation. Wildlife Society Bulletin 37 (4), 690–697.

Santangeli, A., Arkumarev, V., Rust, N., Girardello, M., 2016. Understanding, quantifying and mapping the use of poison by commercial farmers in Namibia – Implications for scavengers' conservation and ecosystem health. Biological Conservation 204 (B), 205–211.

Schipper, J., Chanson, J., Chiozza, F., Cox, N., Hoffmann, M., Katariya, V., Lamoreux, J., Rodrigues, A., Stuart, S., Temple, H., Baillie, J., Boitani, L., Lacher, T. J., Mittermeier, R., Smith, A., Absolon, D., Aguiar, J., Amori, G., Bakkour, N., Baldi, R., Berridge, R., Bielby, J., Black, P., Blanc, J., Brooks, T., Burton, J., Butynski, T., Catullo, G., Chapman, R., Cokeliss, Z., Collen, B., Conroy, J., Cooke, J., Da Fonseca, G., Derocher, A., Dublin, H., Duckworth, J., Emmons, L., Emslie, R., Festa-Bianchet, M., Foster, M., Foster, S., Garshelis, D., Gates, C., Gimenez-Dixon, M., Gonzalez, S., Gonzalez-Maya, J., Good, T., Hammerson, G., Hammond, P., Happold, D., Happold, M., Hare, J., Harris, R., Hawkins, C., Haywood, M., Heaney, L., Hedges, S., Helgen, K., Hilton-Taylor, C., Hussain, S., Ishii, N., Jefferson, T., Jenkins, R., Johnston, C., Keith, M., Kingdon, J., Knox, D., Kovacs, K., Langhammer, P., Leus, K., Lewison, R., Lichtenstein, G., Lowry, L., Macavoy, Z., Mace, G., Mallon, D., Masi, M., McKnight, M., Medellin, R., Medici, P., Mills, G., Moehlman, P., Molur, S., Mora, A., Nowell, K., Oates, J., Olech, W., Oliver, W., Oprea, M., Patterson, B., Perrin, W., Polidoro, B., Pollock, C., Powel, A., Protas, Y., Racey, P., Ragle, J., Ramani, P., Rathbun, G., Reeves, R., Reilly, S., Reynolds, J. I., Rondinini, C., Rosell-Ambal, R., Rulli, M., Rylands, A., Savini, S., Schank, S., Sechrest, W., Self-Sullivan, C., Shoemaker, A., Sillero-Zubiri, C., De Silva, N., Smith, D., Srinivasulu, C., Stephenson, P., Van Strien, N., Talukdar, B., Taylor, B., Timmins, R., Tirira, D., Tognelli, M., Tsytsulina, K., Veiga, L., Vié, J., Williamson, E., Wyatt, S., Xie, Y., Young, B., 2008. The status of the world's land and marine mammals: Diversity, threat, and knowledge. Ecology 322 (5899), 225–230.

Schumann, M., 2004. Guide to integrated livestock and predator management. Cheetah Conservation Fund, P.O. Box 1755, Otjiwarongo, Namibia.

Schwartz, M., Cook, C., Pressey, R., Pullin, A., Runge, M., Salafsky, N., Sutherland, W., Williamson, M., 2018. Decision support frameworks and tools for conservation. Conservation Letters 11 (2), 1–12.

Shafer, G., Pearl, J. (Eds.), 1990. Readings in uncertain reasoning. Morgan Kaufmann Publishers Inc.

Shelton, M., 2004. Predation and livestock production: Perspective and overview. Sheep & Goat Research Journal 19, 2–5.

Shivik, J., 2004. Non-lethal alternatives for predation management. Sheep & Goat Research Journal 19, 64–71.

Shivik, J., 2006. Tools for the edge: What's new for conserving carnivores. BioScience 56 (3), 253–259.

Shivik, J., Tregenza, T., Callahan, P., 2003. Nonlethal techniques for managing predation: Primary and secondary repellents. Conservation Biology 17 (6), 1531–1537.

Shortridge, G., 1934. The mammals of South West Africa. Heinemann.

Sinclair, A., Fryxell, J., Caughley, G., 2006. Wildlife ecology, conservation, and management., 2nd Edition. Blackwell Publishing.

Sinclair, A., Mduma, S., Grant, J., Hopcraft, C., Fryxell, J., Hilborn, R., Thirgood, S., 2007. Long-term ecosystem dynamics in the Serengeti: Lessons for conservation. Conservation Biology 21 (3), 580–590.

Smuts, B., 2008. Predators on livestock farms - A practical farmers' manual for non-lethal, holistic, ecologically acceptable and ethical management. Landmark Foundation, www.landmarkfoundation.org.za.

Snow, T., 2009. A systems-thinking based evaluation of predator conflict management on selected South African farms. Master's thesis, University of KwaZulu-Natal.

Sprague, R. J., Dec. 1980. A framework for the development of decision support systems. MIS Quarterly 4 (4), 1–26.

Stander, P., 1990. A suggested management strategy for stock-raiding lions in Namibia. South African Journal of Wildlife Research 20 (2), 37–43.

Stander, P., //au, K., /ui, N., Dabe, T., Dabe, D., 1997. Non-consumptive utilisation of leopards: Community conservation and ecotourism in practice. In: Proceedings of a Symposium on Lions and Leopards as Game Ranch Animals. Desert Lion Conservation Project, Desert Lion Conservation Project, pp. 50–57.

Stein, A., Athreya, V., Gerngross, P., Balme, G., Henschel, P., Karanth, U., Miquelle, D., Rostro-García, S., Kamler, J., Laguardia, A., Khorozyan, I., Ghoddousi, A., 2015. *Panthera pardus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://www.iucnredlist.org/details/summary/15954/0, p. e.T15954A102421779.

Stein, A., Fuller, T., Damery, D., Sievert, L., Marker, L., 2010. Farm management and economic analyses of leopard conservation in north-central Namibia. Animal Conservation 13, 419–427.

Stein, A., Fuller, T., Marker, L., 2013. Brown hyaena feeding ecology on Namibian farmlands. South African Journal of Wildlife Research 43 (1), 27–32.

Stone, S., Fascione, N., Miller, C., Pissot, J., Schrader, G., Timberlake, J., 2008. Livestock and wolves - A guide to nonlethal tools and methods to reduce conflicts. Defenders of Wildlife.

Stuart, C., Macdonald, I., Mills, M., 1985. History, current status and conservation of large mammalian predators in Cape Province, Republic of South Africa. Biological Conservation 31, 7–19.

Suthar, A., Lakhmapurkar, J., Gavali, D., Nayyar, P., 2018. Assessment of human-leopard conflict in human dominated landscape and mitigation measures: A case study from Surat district, Gujarat. Tech. rep., Gujarat Ecological Society.

Sutherland, W., Pullin, A., Dolman, P., Knight, T., 2004. The need for evidence-based conservation. Trends in Ecology and Evolution 19 (6), 305–308.

Swanepoel, J., 2016. Habits of the hunters: the biopolitics of combatting predation amongst small stock farmers in southern Namibia. Journal of Contemporary African Studies 34 (1), 129–146.

Swanepoel, L., Lindsey, P., Somers, M., Van Hoven, W., Dalerum, F., 2013. Extent and fragmentation of suitable leopard habitat in South Africa. Animal Conservation 16 (1), 41–50.

Swanepoel, L., Somers, M., Van Hoven, W., Schiess-Meier, M., Owen, C., Snyman, A., Martins, Q., Senekal, C., Camacho, G., Boshoff, W., Dalerum, F., 2015. Survival rates and causes of mortality of leopards *Panthera pardus* in southern Africa. Oryx 49 (4), 595–603.

Tainton, N. (Ed.), 1999. Veld management in South Africa. University of Natal Press.

Tambling, C., Avenant, N., Drouilly, M., Melville, H., 2018. The role of mesopredators in ecosystems: Potential effects of managing their populations on ecosystem processes and biodiversity. In: Kerley et al. (2018), Ch. 8, pp. 205–227.

Terborgh, J., Sep. 2015. Toward a trophic theory of species diversity. PNAS 112 (37), 11415–11422.

Thorn, M., Green, M., Dalerum, F., Bateman, P., Scott, D., 2012. What drives human-carnivore conflict in the North West Province of South Africa? Biological Conservation 150, 23–32.

Thorn, M., Green, M., Marnewick, K., Scott, D., 2015. Determinants of attitudes to carnivores: implications for mitigating human–carnivore conflict on South African farmland. Oryx 49 (2), 270–277.

Treves, A., Bruskotter, J., 2014. Tolerance for predatory wildlife. Science 344, 476–477.

Treves, A., Karanth, K., 2003a. Human-carnivore conflict and perspectives on carnivore management worldwide. Conservation Biology 17 (6), 1491–1499.

Treves, A., Karanth, K., 2003b. Special section: Human-carnivore conflict: Local solutions with global applications. Conservation Biology 17 (6), 1489–1490.

Treves, A., Krofel, M., McManus, J., 2016. Predator control should not be a shot in the dark. Frontiers in Ecology and the Environment 14 (7), 380–388.

Turpie, J., Akinyemi, B., 2018. The socio-economic impacts of livestock predation and its prevention in South Africa. In: Kerley et al. (2018), pp. 53–81.

Van Bommel, L., 2010. Guardian dogs: Best practice manual for the use of livestock guardian dogs. University of Canberra, Kirinari Street, Bruce ACT 2617.

Van der Merwe, I., Tambling, C., Thorn, M., Scott, D., Yarnell, R., Green, M., Cameron, E., Bateman, P., 2009. An assessment of diet overlap of two mesocarnivores in the North West Province, South Africa. African Zoology 44 (2), 288–291.

Van Eeden, L., Ecklund, A., Miller, J., López-Bao, J., Chapron, G., Cejtin, M., Crowther, M., Dickman, C., Frank, J., Krofel, M., Macdonald, D., McManus, J., Meyer, T., Middleton, A., Newsome, T., Ripple, W., Ritchie, E., Schmitz, O., Stoner, K., Tourani, M., Treves, A., Sep. 2018. Carnivore conservation needs evidence-based livestock protection. PLoS Biology 16 (9), e2005577.

Van Niekerk, H., 2010. The cost of predation on small livestock in South Africa by medium-sized predators. Master's thesis, University of the Free State.

Wallace, K., Leslie, A., Coulson, T., 2011. Living with predators: a focus on the issues of human–crocodile conflict within the lower Zambezi valley. Wildlife Research 38, 747–755.

Westgate, M., Likens, G., Lindenmayer, D., 2013. Adaptive management of biological systems: A review. Biological Conservation 158, 128–139.

Wezel, A., Bellon, S., Doré, T., Francis, C., Vallod, D., David, C., 2009. Agroecology as a science, a movement and a practice. A review. Agronomy for Sustainable Development 29 (4), 503–515.

Wezel, A., Peeters, A., 2014. Agroecology and herbivore farming systems – principles and practices. Options Méditerranéennes A 109, 753–768.

Wimsatt, W., 1994. The ontology of complex systems: Levels of organization, perspectives, and causal thickets. Canadian Journal of Philosophy suppl 20, 207–274.

WWF, 2016. Living Planet Report 2016. Risk and resilience in a new era. Tech. rep., WWF International.

Yoder, J., 2000. Damage abatement and compensation programs as incentives for wildlife management on private land. In: Human Conflicts with Wildlife: Economic Considerations. USDA National Wildlife Research Center Symposia. USDA National Wildlife Research Center, USDA National Wildlife Research Center, pp. 17–28.

Chapter 2

Farmer-predator conflict in Namibia: What we know and what we need to know

Abstract

When attempting to solve farmer-predator conflict, it is important to be clear on what we know and what we do not know yet. This chapter focuses on the basic information available from the literature that landowners and managers in Namibia require to manage livestock depredation within the framework of an agroecological model. The published literature is discussed from the perspective of the individual Namibian commercial livestock farmer who needs to decide on how to manage livestock depredation on his land.

The first half of the literature review investigated 40 conflict mitigation methods found in the literature that could be applied in the Namibian context, highlighting their advantages, disadvantages and limitations. Most methods were preventative, five were compensatory, and only three were reactive. Of the 40 methods reviewed, 29 had been used in Namibia. Generally, reactive methods were cheaper than their preventative equivalent, but carried a higher predation loss risk. Compensatory methods had negligible effects on predation itself, but compensated for losses instead. Some preventative methods and all compensatory methods required cooperation with others for its success and are strictly not a management option for the *individual* farmer. Each method was classified as having a high, medium or low ecological impact, based on its expected impact on population numbers, existing habitat and habitat fragmentation. Ecological impact was used as a proxy for the ecological sustainability of each method. Where applicable, the known ecological principles on which the effectiveness of a specific method depends, were discussed briefly. A prior probability of success was assigned to each method that had been evaluated in the literature and an "unknown" prior success probability of 0.5 was used for the other methods following the principle of maximum entropy (Jaynes, 1968).

In the second half of the review, a larger ecological framework within which human-wildlife conflict management occurs, was provided by the "maturity" concept of Margalef (1963), extended by later developments by Holling (1973) and the state-and-transition model of Westoby et al. (1989). Within this framework, a theory-based defence was made for using minimum ecological impact as proxy for ecological sustainability as the best currently available hypothesis within an adaptive management approach. The importance of veld management and top-down vs. bottom-up control as additional aspects of ecological sustainability, were highlighted. Within the greater ecological perspective of sustainability, the ecology of the four main predator species on Namibian livestock was examined, with the specific aims of identifying gaps in our knowledge and further research requirements. For each of black-backed jackal (*Canis mesomelas*), caracal (*Caracal caracal*), cheetah (*Acinonyx*

jubatus) and leopard (*Panthera pardus*) their distribution, conservation status and threats were discussed, followed by their spatial ecology, breeding ecology, feeding ecology, and intra-guild interspecific interactions. For nine ecological questions further research requirements were highlighted, with possible application to human-wildlife conflict in Namibia and suggested research approaches to each question.

Keywords: Agroecology, black-backed jackal, caracal, cheetah, conflict mitigation methods, farmer-predator conflict, human-wildlife conflict, leopard, livestock depredation, Namibia

2.1 Introduction

When attempting to solve farmer-predator conflict, it is important to be clear on what we know and what we do not know yet. This chapter focuses on the basic information that landowners and managers in Namibia need before even starting to address livestock depredation. It provides an overview from the literature of what we already know – the settled science of evidence-based conservation ecology – and what we do not know yet – the uncertainties and needs for further research – as part of an adaptive management approach to mitigating human-wildlife conflict in Namibia.

The management unit on the commercial farmlands of Namibia is the single farm with the individual livestock farmer as the principle decision-making manager. The published literature is therefore discussed from the point of view of the individual Namibian commercial livestock farmer who needs to decide on how to manage livestock depredation on his land (*cf.* Miller et al., 2015). A livestock farmer who experiences livestock losses to carnivores, has to act while faced with uncertainty about which methods are going to be most cost-effective. The adaptive management framework, which operates with a certain level of uncertainty, thus provides an ideal framework for the management of farmer-predator conflict where current management actions are evaluated and adapted over time (McDonald-Madden et al., 2010; Westgate et al., 2013).

2.1.1 Adaptive management

The adaptive management approach includes decision-making when faced with uncertainty and probabilistic truth instead of a relative certainty of success (McDonald-Madden et al., 2010; Gillson et al., 2019). The probability of success of each conflict mitigation method can also change depending on circumstances, but an initial prior probability of success is required to estimate any final success probability. Jøsang (2008) mentions that where the actual prior probabilities, or base rates, cannot be known with absolute certainty, frequentist approximation through prior empirical observation can be used. For predation management methods, such frequentist approximations could be determined from three sources: 1) by doing a questionnaire survey and where the method had been used by enough Namibian farmers, the *percentage of individuals* from the survey data who had success with a specific conflict mitigation method, can be used (Chapter 3); 2) for methods not used in Namibia yet, the *percentage of published papers* where that specific method had been shown to be successful can be used (an evidence-based conservation approach, but with the danger of publication bias); 3) where a specific method had been evaluated for its effectiveness in Namibia before, the *percentage of individual farmers* who reported success with the specific method can be used (with data from a single or few published studies). In each case the percentage can be converted directly to a probability (between 0 and 1). Within an adaptive management approach, this is not a final probability of success, but rather the initial (evidence-based) estimate of success to be updated over time as the evidence increases.

2.1.2 Holistic approach

Livestock predation management is part of managing a larger agroecological system (Figure 1.4 on page 12) and should include both the agricultural (short-term) cost-effectiveness and the ecological (long-term) perspectives

in a single holistic model (see Figure 2.2). Conway (1985) proposed that the behaviour of agroecosystems can be described by four system properties: productivity, stability, sustainability and equitability at different hierarchical levels. Since equitability should be addressed on a larger scale than the individual farm (i.e. as part of the socio-political environment) it is not included in this analysis (but see Rust et al., 2016 and Potgieter et al., 2017 for different perspectives). Stability in the Namibian context, with its variable weather patterns and periodic droughts (Sweet, 1998; Chiriboga et al., 2008; Hangara et al., 2011a), is also largely beyond the control of the individual farmer (through the influence of the "Abiotic Factors" in Figure 1.4). This leaves (agricultural) productivity and (ecological) sustainability as the two main system properties of the Namibian agroecosystem that an individual farmer can manage.

To address both aspects of the agroecosystem, we need an overview of methods used to mitigate farmer-predator conflict worldwide, their advantages (pros) and disadvantages (cons), their limitations, their relative costs and their effectiveness. But we also need a good insight into the ecology of the four main predators of livestock in Namibia as it pertains to livestock depredation, *viz.* black-backed jackals (*Canis mesomelas*), caracals (*Caracal caracal*), cheetahs (*Acinonyx jubatus*) and leopards (*Panthera pardus*) (Marker-Kraus et al., 1996; Rust and Marker, 2013b). In the first part of this chapter, HWC mitigation methods that are used worldwide are discussed from the literature, with a focus on their relevance to Namibian commercial farms. This is followed by an overview of what we already know about the ecology of four common Namibian predator species as relevant to HWC and where our current knowledge is still lacking, in the second part.

2.2 Predation management methods – an overview

As shown in Figure 2.2a), managing livestock depredation can vary from doing nothing, costing nothing and with zero effectiveness (free-range, extensive farming) – to keeping livestock permanently in predator-proof barns or feeding lots, costing more than the potential income from livestock in Namibia (Spears and BFAP, 2013) and with a 100% effectiveness in preventing livestock depredation (fed livestock, intensive farming). In addition to the variation in agricultural cost-effectiveness, methods can also vary in ecological sustainability (Figure 2.2b).

"Practicality" as used here is defined simply as the possibility to implement. The practicality of a certain conflict mitigating method partly depends on whether the farming enterprise is intensive or extensive (Figure 2.2a). In general, a more intensive farming system attempts to maximize production or profit per hectare or per livestock unit while an extensive farming system attempts to minimize costs per hectare or livestock unit (Nemecek et al., 2011; Wezel et al., 2015). The economic and ecological carrying capacity (Figure 2.1, Caughley, 1976), determined by both grazing and drinking water availability, are usually the most important external factors determining the possible intensity of the farming system. However, distance to markets and fodder-producing areas may also play an important role (Chiriboga et al., 2008). For classifying the farming intensity needed to implement a certain predation conflict mitigation technique,

- intensive farming is defined as a system where the livestock owner sees or handles all his livestock at least once per day,
- a medium intensive system is where the livestock owner sees all his livestock at least once per week,
- a medium extensive system is where the farmer sees all his livestock at least once per month and
- extensive farming is where the farmer sees or handles all his livestock less than once per month.

Other factors that will typically impact the practicality of a certain conflict mitigation method include the livestock species, predator species, labour intensiveness, farm infrastructure, climate, topography and vegetation

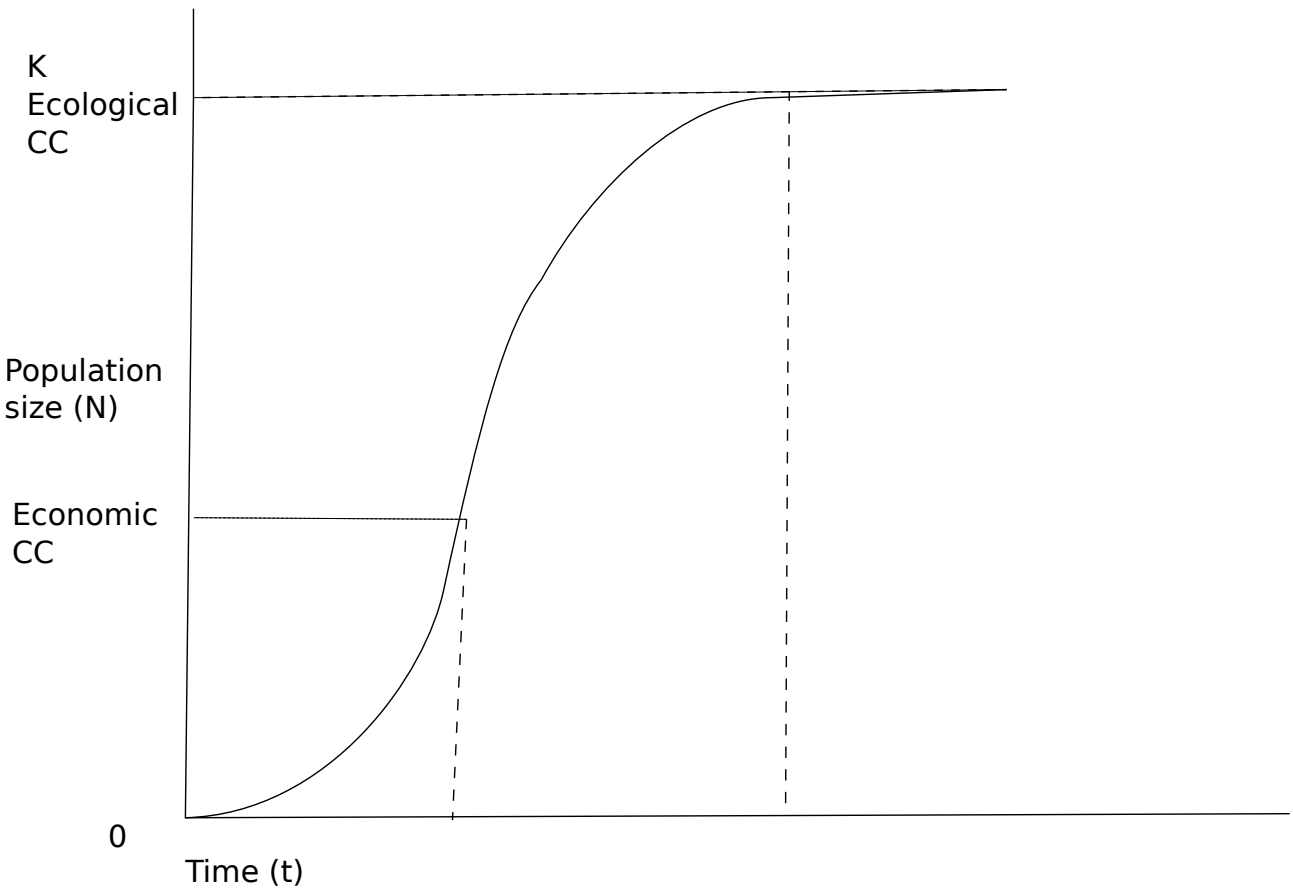


Figure 2.1: *Two types of "carrying capacity" as defined by Caughley (1976)* illustrated using the logistic growth model *(Dhondt, 1988; Begon et al., 1996; Hixon, 2008)* (CC = carrying capacity). Assuming an exponential and density-dependent growth rate of the population, an equilibrium density (K) will be reached at the highest population that the environment can support, where birth rate = mortality rate and further population growth is zero – this is known as the ecological carrying capacity (for simplicity a constant K is assumed – see Luyt, 2004 for a more nuanced perspective). However, the highest production (population growth rate, dN/dt, or biomass increase) will happen at about half the equilibrium density (Dieguez Cameroni and Fort, 2017), called the economic carrying capacity by Caughley (1976). For obvious reasons of production, farmers would prefer to keep their livestock stocking rates as close as possible to the economic carrying capacity (see also the first chapter of Luyt, 2004).

(habitat type), drinking water availability, existing management on neighbouring farms and the natural wildlife that occurs on the farm (e.g. Section 2.2.3 below, Du Plessis et al., 2018).

The two main aims of this part of the review were to 1) find as many predator conflict mitigation techniques as possible that might be of future use in Namibia and 2) evaluate each technique based on current knowledge of probable effectiveness, advantages, disadvantages and limitations for use by Namibian farmers.

2.2.1 Materials and methods

The Institute for Scientific Information (ISI) Web of Science database was searched for any articles containing the keywords "human wildlife conflict carnivore* mitigati*". From this search a list of 62 relevant articles were found containing actual conflict mitigation methods. A similar google search was done in order to find additional and non-published conflict mitigation methods. To avoid duplication of effort, articles (including reviews) that compared more than one method were used preferentially. Where other literature had been cited and appeared to provide more information on either a specific method or a comparison of methods, they were added (snowball sampling) until a total of 211 articles or book chapters were collected. A number of non-peer reviewed publications by Southern African conservation agencies which dealt with farmer-predator conflict (e.g.

the Cheetah Conservation Fund in Namibia, Cape Nature and the Landmark Foundation in South Africa) were also consulted and together with any unpublished theses, considered as "grey literature" (Wezel et al., 2015). Two popular South African agricultural magazines (Farmer's Weekly and Landbouweekblad) were followed over a year period (2013) and any new articles concerning predator conflict were read and noted whenever it mentioned a method that had not been found in any of the scientific publications – these, together with any personal communications, were considered as anecdotal sources. Whenever possible, preference was given to publications that included the Namibian context. Three PhD theses, five masters theses, seven best practice manuals, 10 book chapters and 11 technical reports were used as "grey literature", five unpublished publications from online sources, popular magazines or personal correspondence were included as anecdotal sources, and eight publications from conference proceedings and 162 articles in peer-reviewed journals were used as peer-reviewed sources.

The probable effectiveness of conflict mitigation methods (to be used in a DSS - Section 1.5 on page 14) was estimated from the literature. For methods not used in Namibia, the percentage of peer-reviewed papers where that specific method had been shown as successful, was used to assign a prior probability of success to them. This assumes that this percentage of peer-reviewed papers found is representative of all the papers published on that specific method and that the published papers are actually representative of the real success rate of the method worldwide (no publication bias). These methods were not quantified in terms of their effectiveness against specific predator species. For those methods where peer-reviewed papers had been published with an indication of their success in Namibia, the percentage of cases where the method had been judged successful, was used to assign the prior probability of success. Methods from anecdotal sources alone were considered as providing insufficient data to determine their probability of success and they were assigned a prior success probability of 0.5 (Jaynes, 1968). In the majority of cases, a method was mentioned, sometimes with guidelines to its use or suggesting it as a possible solution to conflict, but very few papers actually evaluated any mitigation method compared with either a control or another method (Miller et al., 2016b; Treves et al., 2016; Krafte Holland et al., 2018; Van Eeden et al., 2018). Thus, many methods mentioned in peer-reviewed papers were also assigned a 0.5 prior success probability since they had insufficient data.

2.2.2 Classification of management methods

Mitigating methods have generally been classified as either "lethal" or "non-lethal" (e.g. Daly et al., 2006; Shivik, 2004). From a conservation ecological viewpoint, non-lethal methods would then be considered as generally "better" than lethal methods (McManus et al., 2014). However, this classification has a number of problems:

1. A generally accepted "non-lethal" method may actually be lethal for some predators, and might even be approved by farmers because of this (e.g. Potgieter et al., 2016, 2013).

2. A bigger issue with the "lethal is bad, non-lethal is good" classification of HWC mitigation methods, is that often a lethal method (e.g. trophy hunting an old male that is past its reproductive prime, unable to hunt its natural prey and started to kill livestock) can be much less disruptive to the whole ecosystem than a non-lethal method (e.g. overgrazing and desertification due to kraaling, or translocation of a new individual into an existing stable population disrupting the territorial system, possible causing sexually selected infanticide (SSI), resulting in more than one individual killed or displaced).

Therefore, ecological disturbance or human impact (where less is better) would be a better measure of how appropriate a mitigation method is, rather than the common lethal / non-lethal classification (Figure 2.2b illustrates how methods can be classified along the two axes of effectiveness and sustainability). Instead of the binary classification of lethal / non-lethal, Treves and Karanth (2003) classified mitigation methods into three basic strategies with different combinations of lethal and non-lethal methods:

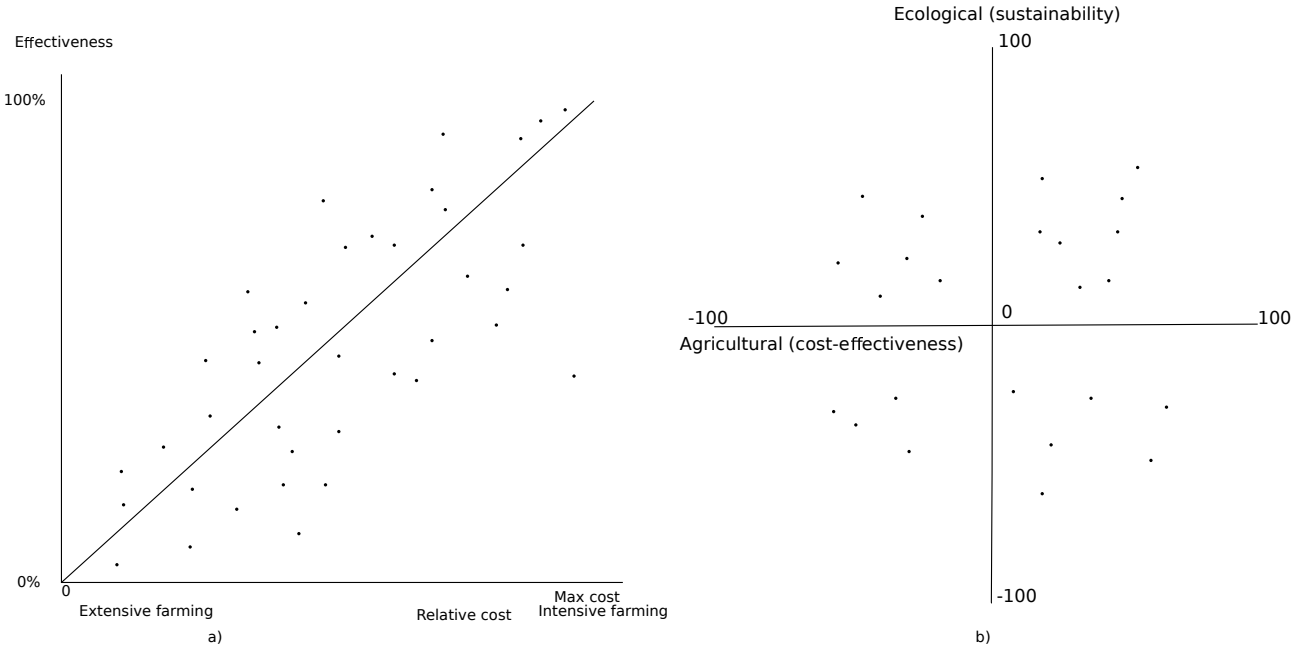


Figure 2.2: *Two views of the different conflict mitigation methods shown as scatter-plots.* a) The relative effectiveness per cost can vary from no cost and no effectiveness, to the maximum cost and 100% effectiveness - Methods that are relatively cost-effective will be above the line and those that are relatively less cost-effective would fall below the line. Both the cost and effectiveness of each method can also change depending on the current situation on the farm. b) Each method can also be plotted on an axis of the relative short-term agricultural cost-effectiveness (the same as its position above or below the line in Figure a) and its relative long-term ecological sustainability. The current management regime on the farm is considered as 0 on both axes. Agricultural cost-effectiveness is defined as the survival rate (reciprocal of depredation loss) of livestock offspring per cost unit and ecological sustainability is a measurement of the ecological impact (assuming that less disturbance leads to higher biodiversity and sustainability). In both cases a method can actually be less effective and/or less sustainable than the current situation, indicated by negative values.

1. Eradication, where the predators are extirpated on farmlands. This was the approach usually followed by governments in the past and are still being advocated by many agricultural organizations (see Bergman et al., 2013; Rust, 2016). This equates to an end-point of the conflict where the farmers "win" if successful. However, this strategy is likely to have many unintended consequences (e.g. the mesopredator release of jackals in some areas of Southern Africa, or the population explosion of rodents killing grass roots in areas where jackals were extirpated – Beinart, 1998; Ritchie and Johnson, 2009; Terborgh, 2015). If unsuccessful, this strategy will unintentionally end up identical to the second approach.

2. Regulated harvest, where predators are selectively hunted or killed, without the aim of eradication from farmlands. In effect, sustainable harvesting of predators is the management aim (Treves and Karanth, 2003). This approach has not been shown as effective for preventing livestock losses in the long term (De Wet, 2002, 2006; McManus et al., 2014; Treves et al., 2016) and could worsen the situation for both farmers and the ecosystem (De Wet, 2002; Bothma, 2012; Conradie and Piesse, 2013; Bailey and Conradie, 2013). The most likely effect of this strategy would be to depress the predator population size until it approaches the economic carrying capacity (Figure 2.1) *sensu* Caughley (1976), with the highest population and biomass growth rate (Minnie et al., 2016b; Dieguez Cameroni and Fort, 2017). If unsuccessful this strategy could result in either the predators "winning" (livestock farming becoming unsustainable) or else in the accidental eradication of all predators, while ideally it would enable co-existence if successfully implemented.

3. Preservation, where only non-lethal methods are used, leading to the coexistence of farmers and predators on the land. These methods are often expensive or difficult to implement (Treves and Karanth, 2003). If unsuccessful, this conservation strategy can result in farming becoming unsustainable, otherwise in

coexistence if successful.

Linnell et al. (1996), used the same classification, but suggested that it should be applied in different geographical zones, with an outside are zone only using lethal methods (eradication), a conservation zone only using non-lethal methods (preservation), and a buffer zone using both (regulated harvest). In effect this classification is still a variation of the lethal/non-lethal dichotomy. It has the advantage of not simply equating lethal with "bad" and non-lethal with "good" (*cf.* McManus et al., 2014). Linnell et al. (1996) made the important point that the choice of mitigation method also depends on the conservation status of the predators involved in the conflict. While their zoning classification of "conservation zone", "buffer zone", and "outside area" makes sense in the African context for those predator species (like lions, *Panthera leo,* or spotted hyaenas, *Crocuta crocuta*) which had already been extirpated from most farmlands, it is less useful for predators like leopards, cheetahs, caracals or black-backed jackals where the majority of the predator population is found outside formally protected areas on livestock farms with a relatively intact natural habitat. Should all this natural habitat be zoned as a conservation zone or be considered a buffer zone instead? This is very different from the European situation where most of the habitat used for livestock had been transformed, with a much smaller variety of natural prey, and consists of planted pastures which quite naturally fall in the "outside area" zone.

Treves and Karanth (2003) split non-lethal methods into either methods that change predator behaviour or methods that physically keep predators separated from livestock. Shivik (2004) subdivided non-lethal methods into 1) altering human behaviour, 2) altering husbandry or 3) altering predator behaviour. In practice, since any change in conflict mitigation methods will be implemented by humans, most likely by the land-owners or farmers, all of them include altering human behaviour to some extent. Madden and McQuinn (2014) classified methods used to address human-wildlife conflict into five groups: 1) physical/spatial (e.g. fences); 2) economic (e.g. incentive schemes); 3) technical (e.g. husbandry or farming methods); 4) legal (e.g. anti-poaching or quotas); 5) biological (e.g. using wild prey or predator behaviour – see box on *optimal foraging theory* on page 38). The only criticism of this classification would be that some husbandry techniques also include physical barriers, making them difficult to classify. Du Plessis et al. (2018) used an even more complicated classification system for predation management methods, including lethal methods: 1) disruptive deterrents, 2) husbandry practices, 3) aversive deterrents, 4) provisioning additional food to predators, 5) non-lethal population control, 6) producer management (compensation) and 7) lethal predator management.

Optimal foraging theory. Haswell et al. (2019) recently showed that optimal foraging theory provides an excellent ecological framework in which to consider most anti-predation methods. This is based on the formula $H = C + P + MOC$ where H is the quitting harvest rate (when the predator should switch from hunting livestock to other prey), C is the cost of foraging (energy spent searching, hunting and killing the prey), P is the predation risk (from e.g. humans or guarding animals) expressed in terms of energy, and MOC is the missed opportunity cost of not hunting other prey elsewhere (Brown, 1988). As long as the energy benefit (H = harvest rate) from hunting livestock is higher than the total costs (C + P + MOC) the predator will kill and eat livestock — conversely as soon as the total cost of killing livestock is higher than that of hunting other prey, the predator is likely to switch to other prey. However, for each predator species the variables in the equation will have different values. It should be noticed that this framework, while including lethal methods for instance, as causing higher P ("predation risk" by humans), or fences causing higher C (energy cost of hunting), inherently models changes in predator *behaviour* (the third group in the classification of Shivik, 2004) and is not directly applicable to those methods which do not depend on predator behaviour for their success (e.g. compensatory methods, local extermination, translocation, etc.). Ecologically, most predation management techniques discussed below in Section 2.2.3 are applying the basic formula from optimal foraging theory ($H = C + P + MOC$) to increase one of the factors on the right (for livestock) in order to make switching to other (natural) prey more attractive than killing livestock to the predator — in other words to lower the giving-up-densities (GUD) for livestock as prey (Brown, 1988; Haswell et al., 2019). Optimal foraging theory provides a simple ecological model for farmers who want to "farm with nature" instead of "fighting against nature" when applying anti-predation strategies.

A simple threefold classification of mitigation methods for decision-making is proposed here (*cf.* Weise et al., 2018 and box on *differentiating between ways of method use* below):

1. Preventative (pre-emptive) methods that attempt to prevent livestock depredation by carnivores. This could include both lethal (e.g. eradication of predators) and non-lethal methods (Shivik, 2004; Daly et al., 2006; Shivik, 2006).

2. Offset (incentive/compensatory) methods (e.g.Stander et al., 1997a; Mishra et al., 2003). These are methods that do not prevent or decrease livestock depredation, but in some way reward farmers for the presence of predators on their land, to the extent that they would conserve rather than exterminate predators on their land. It can include both non-lethal methods (e.g. non-consumptive use like tourism or dart gun hunting) and lethal methods (e.g. trophy hunting of predators).

3. Reactive methods that only react to livestock depredation, rather than trying to prevent it. They differ in costs and effectiveness from methods in the previous two groups, even when they appear similar (e.g. hunting of predators). They can be cheaper to implement (less often used) and cause less disturbance of the farming ecosystem, because they are only used in reaction to actual livestock losses, but also carry a higher risk of livestock losses.

Differentiating between ways of method use. It should be clear that a method (e.g. hunting which in other classification systems would simply be classified as "lethal") can be used in all three ways, with different costs and effectiveness. E.g. hunting can be done *pre-emptively*, to eradicate predators and thus prevent livestock losses, it can be used as an *off-set* to the risk of livestock losses by the farmer benefiting from having predators on his land through trophy hunting, or it can be used *reactively*, by only shooting individual animals after they killed livestock repeatedly. The "single method" of hunting should thus be considered as three different methods, differing in costs and effectiveness. The same is true of some other methods.

2.2.3 Forty management methods applied to Namibia

Marker-Kraus et al. (1996) already evaluated a range of management methods used by livestock farmers to prevent livestock losses from a survey in North-Central Namibia and found most to not show a significant decrease in livestock losses (compared to the average predation losses). Here, 40 methods that had been proposed or used worldwide, are evaluated in the Namibian context. The history and use of each method is briefly described, given a prior probability of success, and some of its pros, cons and limitations mentioned (see Table 2.1). Pros (advantages, as used here), are positive aspects of the specific method that could theoretically be quantified, while cons (disadvantages) are negative aspects that could also be quantified. Limitations measure the practicality of a method (as defined above in Section 2.2) with a binary value of either true (practical for specific situations) or false (impractical in other circumstances). Such limitations either follow logically from the method itself or else were explicitly mentioned in the literature.

Most of these methods have not even been evaluated in terms of cost-effectiveness for local situations (Krafte Holland et al., 2018) and evaluating their sustainability would require long-term studies, which have not been done. Methods are thus given a subjective sustainability probability related to its likely ecological impact (using historical impacts where possible). Local extermination of carnivore species will likely have large ecological impacts, including trophic downgrading, changes in vegetation, mesopredator release, increasing small mammal populations, niche expansion, cascading extinctions, lower ecosystem resilience, etc. (Palmer and Fairall, 1988; Ritchie and Johnson, 2009; Estes et al., 2011; Bartnick et al., 2013; Yarnell et al., 2013; Ford et al., 2014; Ripple et al., 2014; Terborgh, 2015; Flagel et al., 2017; Landi et al., 2018). All methods that aim for local eradication are therefore considered to have a low probability of being ecologically sustainable. Methods that only decrease predator population numbers are generally considered as more sustainable, but can have counter-intuitive effects on livestock depredation levels and may not be economically sustainable (Conradie and Piesse, 2016; Nattrass and Conradie, 2018). Where the ecosystem is relatively intact, methods that focus on livestock management and exploiting the natural behaviour of predators, prey and their interaction with the landscape, are considered as most likely to be ecologically sustainable.

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
1. Preventative methods																			
1.1. Major ecological impact - p(sustainable) = 0.01 - 0.05																			
1. Hunt to exterminate	PR, A	✔	0.57	- Better veld utilization and livestock productivity - Familiar and easy to implement - Easy to maintain once achieved - Most realistic in transformed habitats	- Low success rate historically unless combined in "fence and clean up" - Counter-productive when not effective - Unwanted side-effects when effective - Predators killed ≠ livestock saved	✗	✗	✗	✗	✔	✔	✔	✔	✔	✔	✗	✗	✔	✗
2. Dogs to exterminate	PR, G, A	✔	0.57	- Can be target-specific, if dogs well-trained - More success trailing smaller predators - Cheaper than expensive hunting methods - Dog scent can keep away some predators	- Kill non-target species unless well trained – training takes time - Same unwanted counter-productive side-effects as hunt - Predators killed ≠ livestock saved - Male dogs don't kill female jackals	✗	✗	✗	✗	✔	✔	✔	✔	✔	✔	✗	✗	✔	✗
3. Poison bait to eradicate	PR, G	✔	0.57	- Easy to use	- Ineffective, with far-reaching negative ecological effects (extinctions) - Counter-productive when not effective - Non-specific, more likely to target scavengers than predators - Illegal in Namibia	✔	✗	✗	✗	✔	✔	✔	✔	✔	✔	✗	✗	✗	✗
4. Carnivore-proof fencing	PR, G, A	✔	0.57	- One of the only methods that had some historic success to exterminate black-backed jackals	- Expensive to implement farm-wide - Stop free movement of most wildlife - Needs regular maintenance (double labour costs) - Once predator is inside, can cause surplus killing	✔	✗	✗	✔	✔	✗	✔	✔	✔	✔	✔	✗	✔	✗
1.2. Intermediate ecological impact - p(sustainable) = 0.06 – 0.6																			
5. Hunt to reduce density	PR, G, A	✔	0.57	- Cheap and - easy to implement	- Difficult to target guilty individuals - Often counter-productive	✔	✔	✔	✗	✔	✗	✔	✔	✔	✔	✔	✗	✔	✔

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
6. Trapping to kill	PR, A	✔	0.57	- Some traps can be selective - Fairly inexpensive - Some traps easy to use	- Most traps unselective - Some traps cruel, injuring non-target species - Cage traps kill more felids than jackals changing ecology - Cage traps injure leopards that can force them to kill livestock - Often counter-productive - Relatively labour-intensive - In fences catch first animal only, opening way for predator	✗	✔	✔	✗	✔	✔	✗	✔	✔	✔	✔	✗	✔	✔
1.3. Minor ecological impact - p(sustainable) = 0.61 – 0.9																			
7. Herding	PR, G, A	✔	0.87	- One of the most effective methods with dogs and kraals - Can be used together with most other methods - Works well with herding dogs and livestock guarding dogs	- Veld trampling around kraals - No grazing at night: lower production - Time needed for herders to know veld, predation hotspots and dogs	✔	✔	✔	✔	✗	✔	✔	✗	✔	✔	✔	✔	✔	✔
8. Patrolling (human presence)	PR, A	✗	0.5	- Well suited to extensive farming systems - Veld condition well observed - Predator hotspots identified - No need for kraaling: little	- Probably not very effective (livestock vulnerable at night) - Labour requirements	✔	✗	✔	✔	✔	✗	✔	✗	✔	✔	✔	✔	✔	✔
9. Kraaling at night	PR, G, A	✔	0.87	- Often only effective option (combined with herders/LGDs) at high predator densities - Can be used around water points with little additional management needed - Combined with LGD sleeping with livestock at night, very effective - Costs: labour only – US$ 2 000 / kraal	- Trampling around kraal - Livestock sleeping out (alternative water) - Higher risk of parasites & disease	✗	✔	✗	✔	✔	✔	✔	✗	✔	✗	✔	✔	✔	✔
10. Kraaling young permanently	PR, A	✔	0.87	- Shown effective in North-Central Namibia - Good option at high predation levels - Combines well with intensive management of young livestock - Slower growth rates temporary	- Relatively labour-intensive - Requires seasonal breeding - Possible trampling of veld - Weaker protective mothering instincts	✔	✔	✗	✔	✔	✗	✗	✗	✗	✗	✔	✗	✔	✔

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
11. Electric fencing	PR, A	✔	0.87	- Effective against many predator species - Can prevent digging underneath fences - Mobile: less labour intensive than herding for holistic grazing - Very effective for small areas	- More expensive than other fences to install - Needs regular maintenance - Not practical in all terrain types - Kill small innocent species & prevent free wildlife movement: high ecological impact	✔	✗	✗	✔	✔	✔	✔	✔	✔	✔	✔	✗	✔	✔
12. Calf / Lamb camps	PR, G, A	✔	0.87	- Easier to protect against predators - Enables more intensive livestock management - Higher survival compensates for slower growth - Higher production with irrigated pastures - Easy to combine with other	- Higher risk of parasites and disease - Possible trampling - Slower growth because of overgrazing	✔	✗	✗	✔	✔	✗	✗	✔	✗	✗	✔	✔	✔	✔
13. Static repellents	PR, A	✗	0.33	- Easy to set up with little labour - Cheap to maintain - 60-100% temporary reduction in predation - Turbo-fladry better than electric fence or fladry alone	- Always temporary (1 – 180 days) - Cheap fladry maintenance-intensive - Limited range - Danger of theft - Predator sanctuaries required	✔	✔	✗	✔	✔	✔	✗	✔	✔	✗	✔	✔	✔	✔
14. Projectile repellents	PR, A	✗	0.5	- Good fit for human-habituated predators killing livestock near homesteads - Combines well with patrolling land - Can be used in both intensive and extensive management	- Teach predators to avoid humans, rather than avoiding livestock or certain areas	✔	✔	✔	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔
15. Repellent collars	PR, G, A	✗	0.5	- High-tech version effective for longer - Combines well with other methods like LGDs and herders to warn off predators - Some systems can prevent livestock (LS) theft - Cellular (GSM) system good choice near towns	- Can become counter-productive if predators habituate - High-tech versions expensive (≈1 sheep: 1 collar / 10 sheep) (N$ 5000: 1 collar / flock) - Less effective for breeds where the herd disperses into small groups - Relatively labour-intensive - Strangulation of growing lambs possible	✔	✔	✔	✔	✗	✔	✗	✔	✔	✔	✔	✔	✔	✔

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
16. Protective collars	PR, A	✔	0.5	- Decreased cost and predation significantly in Eastern Cape (comparable to LGDs)	- Can kill growing livestock through strangulation - Relatively labour-intensive - Needs to collar all animals - Can be counter-productive when predators start attacking hind-quarters	✔	✔	✔	✔	✗	✔	✗	✔	✔	✔	✔	✔	✔	✔
17. Conditioned Taste Aversion	PR, G, A	✗	0.5	- Inexpensive - Safe for humans - Non-lethal for predators, scavengers and secondary consumers - Relatively long-lasting - Combines easily with most husbandry methods - Ecologically sustainable - Little maintenance labour needed	- Taste aversion to specific livestock species - Difficult logistics (finding & treating all carcasses) - Sensitive to methodological variation - Takes some time to work - Misapplication can be counter-productive - Removing carcasses can increase predation - Incompatible with predator removal	✗	✗	✔	✔	✔	✗	✔	✔	✔	✔	✗	✗	✗	✔
18. Chemical repellents	PR, A	✗	0.5	- If it works, works immediately (no learning behaviour required)	- Like protective collars, can cause changed attack method by predators - Effectivenes vary between predator species - Labour-intensive	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
19. Natural repellents	PR, G, A	✗	0.5	- Long-term effective (possibly permanently)	- Protect relatively small area - Difficult finding scat and/or urine - Relatively labour-intensive - Unknown effectiveness for different predator species - Ineffective if threat by predator species providing scat is unknown or non-existent in area	✔	✔	✔	✔	✗	✔	✗	✔	✔	✗	✔	✗	✔	✔
20. Increase natural prey	PR, A	✔	0.71	- Costs almost nothing if there are free-roaming wildlife species - Combines well with most other methods - Increases biodiversity and ecological sustainability	- Can be expensive to reintroduce wildlife if not there already - It can take time for wildlife numbers to increase enough to reduce livestock losses - If predators have learned to predate on livestock, may be insufficient to reduce losses	✔	✔	✔	✔	✔	✔	✔	✔	✔	✗	✗	✗	✔	✔

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
21. Livestock Guarding Dogs	PR, A	✔	0.91	- One of the most effective methods if well-trained - Relatively cheap to maintain (5-6 sheep / year) - After year of training, little effort or labour	- Takes about year to train & become confident enough to protect against larger predators - Needs herder during first year to train dog and protect livestock - Training year can be expensive - Average work life of 4-5 years only - Behavioural problems if not trained properly - High demand and small supply - More difficult to utilise remote areas - Management issues - Sometimes kill predators	✔	✔	✔	✔	✗	✔	✔	✗	✔	✔	✔	✗	✗	✔
22. Guard donkeys	PR, G, A	✔	0.59	- Faster to implement than LGDs (no training required) - Cheap to buy and maintain - Longer life expectance than LGDs (10-20 years) - Combines well with other methods	- Reportedly less effective than LGDs	✔	✔	✔	✔	✗	✗	✔	✔	✔	✔	✔	✔	✔	✔
23. Guard Llamas / Alpacas	PR, A	✗	0.5	- No need to train - Cheap to maintain - Stronger protective bonds with livestock than donkeys	- Scarcer & - More expensive than LGDs - Unknown adaptability to bush savannahs	✔	✔	✔	✔	✗	✗	✔	✔	✔	✔	✔	✔	✗	✔
24. Protection cattle (mixed species flerds)	PR, G, A	✔	0.5	- Reportedly effective for predators that can be chased by cattle - Combines well with other methods like holistic grazing - High cost of bonding can be recovered from other savings	- Expensive to bond sheep with cattle (45-90 days together in kraal)	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✔	✔
25. Avoid depredation hotspots	PR, A	✔	0.5	- Very easy to implement with camps - Almost zero costs - Minimum ecological impact and ecologically sustainable - Combine with most other methods	- Depends on heterogenous farm habitat - Impossible on farms consisting of only “difficult terrain”	✔	✗	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✔

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
26. GPS collar predators	PR, G, A	✔	0.5	- Almost no ecological impact - Bridges trust gap between ecologists and farmers - Can be quite effective - Combines well with translocation	- GPS collars expensive (≈ 6 cattle) - Possible difficulty capturing predator - Veterinarian needed for collaring	✔	✔	✔	✔	✔	✗	✔	✔	✔	✔	✗	✗	✔	✔
27. Seasonal breeding	PR, A	✔	0.5	- Synchronised with favoured predator prey: losses reduced - Avoiding predator breeding season: losses reduced - Manipulating natural livestock breeding season cheap and easy - Combines well and required for some other methods	- Changing breeding season to avoid predation can cause production losses - Requires more management and possibly higher labour costs - May result in lower birth rates	✔	✔	✔	✔	✔	✗	✔	✔	✔	✔	✗	✔	✔	✔
28. Change livestock type	PR, A	✔	0.5	- Simple to implement - Can be very effective in areas without larger predators - Sometimes the only option in very extensive production systems	- Switch can have negative economic implications - Replacing existing livestock with new species or breed takes time	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✗	✔	✔	✔
29. Change livestock age class: Weaner – ox	PR, G, A	✔	0.5	- Very effective when few larger predators - Fairly easy to implement - Faster to adapt stocking rates to rainfall	- No breeding selection possible - Weaners scarcer and more expensive in years of good rainfall (requires good planning)	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✗	✔
30. Horned cattle	PR, A	✔	0.5	- Negative costs and ease of implementation - With good mother selection, can be very cost-effective - With breed selection, one of few options in very extensive production systems	- Dangerous to handle in kraal - Can hurt each other when forced close together (in kraals or during transport) - Difficult to sell to feeding lots - Aesthetics?	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✗	✔	✔	✔
31. Large herds holistic grazing	PR, G, A	✔	0.5	- Improved veld - Reported to effectively reduce livestock losses in Namibia - Combines well with mixed species herds (flerds)	- Can (counter-productively) degrade grazing - Short-term reduced production - General livestock management more difficult	✔	✗	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔
32. Sterilize breeding alphas	PR, A	✗	0.5	- Where feeding of young drives predation on livestock, very effective	- Difficult & - Expensive (US$ 600/predator)	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✗	✔	✔

Method	Source PR, G, A	Used in Namibia	p(success)	Pros	Cons	Limitating factors													
						Farm size	Habitat	Infrastructure	Predator sp	Livestock sp	Current losses	Intensive/Extensive	Labour	Season	Veld	Game sp	Current methods	Material	Cooperation
2. Compensatory methods																			
2.1. Intermediate ecological impact - p(sustainable) = 0.06 – 0.6																			
33. Trophy hunting of predators	PR, G, A	✔	0.5	- Can make farmers more positive and motivate them to sustain predator numbers on their land	- Can disrupt predator population structure - Can cause SSI - Can endanger predator survival	✗	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✗	✔	✗
2.2. Minor ecological impact - p(sustainable) = 0.61 – 0.9																			
34. Direct loss compensation	PR, A	✔	0.64	- General farmer approval - Possibly only option in areas without any natural prey	- Moral hazard - Sometimes difficult to prove predation - Failed expectations counter-productive - Time and costs before payment - Low compensation rates - Sustainable funding difficult - Urban vs rural values	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✗	✔	✗
35. Insurance	PR, G, A	✗	0.5	- Likely affordable & sustainable - No outside funding required - Incentive for proactive methods	- Insurance payments still a nett loss - Usually very expensive - Needs verification of predation	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✗	✔	✗
36. Compensation for predators	PR, A	✗	0.5	- Decreases market risks - Directly addresses conservation - No moral hazard	- Wide farmer participation required - Participating shops required - Compliance verification difficult - Marketing costs - Consumer demand uncertain - Sustainable funding difficult	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✗	✔	✔
37. Ecotourism	PR, G, A	✔	0.25	- Can surpass farming income - Predators favourite wildlife to see - Option for single farmer	- Predators elusive (difficult to see) - Livestock farmers lack tourism skills - Time to become known by tourists - Could increase livestock losses	✔	✔	✔	✔	✔	✔	✔	✔	✔	✔	✗	✗	✔	✔
3. Reactive methods																			
3.1. Intermediate ecological impact - p(sustainable) = 0.06 – 0.6																			
38. Killing predator reactively	PR, A	✔	0.57	- Cheaper than pre-emptive hunting - Can be very selective - Can be very effective	- Only effective if habitual livestock killer - Can be counter-productive - Two losses required for confirmation	✔	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✔
39. Translocate predator	PR, A	✔	0.45	- Conservation of threatened specie - Removing single habitual livestock killer can stop all predation	- Only effect if habitial livestock killer - May simply relocate conflict - Danger of SSI and death of predators - Genetic adaptations diluted - Expensive (but not to farmer)	✔	✔	✔	✔	✔	✗	✔	✔	✔	✔	✔	✔	✔	✔
40. Poisonous livestock collar	PR, G, A	✔	0.57	- Selective in killing actual livestock killer - Farmer satisfaction in guilty killed	- One livestock animal still lost - Can be counter-productive - Too expensive to collar all sheep - Labour intensive, strangulation danger - Dangerous to humans, pets and livestock	✔	✔	✔	✔	✗	✔	✗	✔	✔	✔	✔	✔	✔	✔

Table 2.1: *Livestock depredation management methods.* Methods are classified as 1. pre-emptive, 2. compensatory or 3. reactive. Under each class, methods are classified according to their likely ecological impact with p(sustainable) giving the probability that the methods are sustainable in the long run. Since no long-term studies have been done to determine the actual ecological sustainability of these methods, a subjective prior probability is given here. As more data becomes available, these probabilities can be updated. Sources are classified as PR = peer reviewed, G = grey and A = anecdotal. The prior probability of success for the method is indicated by p(success). Pros are the advantages of the method and cons the disadvantages. LS = livestock. Limiting factors are those factors that might be a limitation on the practicality of the method, indicated by a ✗ (✔ indicates that a factor imposes no limitations on the use of the method). "Habitat" includes vegetation and topography. "Infrastructure" includes water points, fencing and kraals. The seasonality, predator responsible, livestock type and age of current losses can limit some methods. "Veld" is used for limitations based on current range-land management and veld condition. "Current methods" indicates if currently used methods could preclude use of the method. "Material" indicates if the availability of trained personnel or specific material could limit the practicality of the method. "Cooperation" denotes the limitations placed by the need for cooperation on a wider level than the individual farm for a method to succeed. See text for more details on individual methods.

2.2.3.1 Preventative methods

The different preventative methods include both lethal and non-lethal methods. The main focus is on preventing predators from killing livestock. One of the first methods used in the Cape of Good Hope by Dutch farmers was the active hunting of predators. It seems obvious that getting rid of the predators will prevent carnivores from killing livestock, which is one reason why this method is still favoured by some farmers (Rust, 2016). Killing predators can be done with a number of different objectives (Linnell et al., 1996):

- causing local extermination of a carnivore species (e.g. Nyhus et al., 2005),
- reducing the carnivore population to a lower density,
- preventing carnivores from (re)colonising areas with high conflict potential,
- selectively removing individual carnivores (see text box: *A note on selectivity of (lethal) methods* below).

Of these different aims, the selective removing of individual predators is not pre-emptive, since the predator must first kill livestock (repeatedly) before it can be identified as a "problem individual" (habitual livestock killer) (Stander, 1990a; Linnell et al., 1999).

A note on selectivity of (lethal) methods — In the literature some methods are often described as being "selective", but the term can be used with two very different meanings: 1) species selectiveness, in which the method is selective for a specific predator *species* (i.e. few non-target species killed), 2) individual selectiveness, in which the method is able to target a specific *individual* predator. Unfortunately, the literature does not always differentiate and it is often unclear to which meaning of "selective" an author is referring. It should be clear that most proactive methods can only be selective at the species level, while only reactive methods can be selective for "problem" individuals. Similarly, the term "problem animals" is used for both animal *species* and *individuals* (*cf* Brand et al., 1995 *vs.* Stander, 1990a). "Habitual livestock killer" is proposed as the more descriptive term for what is often called a "problem individual". Damage causing animal (DCA) has been commonly used as an alternative for "problem animal" when referring to the species (Gunter, 2008; Ogada, 2014; Anthony and Swemmer, 2015).

1. Hunting to exterminate predators: Active hunting of predators resulted in the extermination of most of the large top predators, like lions and spotted hyaenas, from farmlands (Beinart, 1998). However, active hunting *alone* had not been successful against jackals and caracals (De Wet, 2002, 2006; Bothma, 2012), illustrating that different predators are often differently impacted by hunting (Kissui, 2008). Instead, the numbers of jackals and caracals actually increased as the larger predators were removed from the land (Beinart, 1998), which can best be explained as the result of mesopredator release (Ritchie and Johnson, 2009).

Historical note on effectiveness of hunting predators — From the time of Van Riebeeck (1650's) until the 1950's when it was finally replaced by hunting clubs, a bounty system was commonly used as added incentive to kill predators in South Africa. However, hunting only became effective in South Africa when jackal-proof fencing was combined with organised hunt clubs in a "fence and clean-up" approach (Beinart, 1998; Nattrass and Conradie, 2015). By 1963 jackals were already exterminated from some farms in Southern Namibia using this approach (Swanepoel, 2016). Since independence, conditions changed so much through e.g. the increase in cattle (*Bos taurus*) farming, game farming and weekend farmers[a], as well as the subsidies for jackal-proof fencing being phased out, that this known and trusted method is no longer effective (Swanepoel, 2016). In recent decades, techniques have changed to include hunting by night using sound recordings (of prey or other predators) and spotlights to call predators – the "call and shoot" (CAS) method (Swanepoel, 2016) – or shooting from the air using helicopters in parts of South Africa and the USA (Harrington and Conover, 2007; Bothma, 2012; Du Plessis et al., 2018). Often semi-professional hunters are paid to kill the predators on a farm (Nattrass and Conradie, 2015; Swanepoel, 2016).

[a]So-called weekend farmers are often rich professionals who buy a farm as a "hobby", or farmers who because of drought (or other circumstances) had to find a job in town as additional income to survive (Nattrass and Conradie, 2015, 2018; Turpie and Akinyemi, 2018).

Ecologically, the explicit aim of this method is for humans to replace all extirpated carnivores as the only remaining predator. In this context, Ripple et al. (2014) made it clear that "human actions cannot fully replace the role of large carnivores." This would suggest that hunting for extermination is not sustainable in the long term (high ecological impact). Van Eeden et al. (2018) using an evidence-based approached, found all lethal methods to have an average success rate of 57% (17% - 86%) and a 20% (0% - 33%) chance of counter-productively increasing livestock losses (*cf.* Bailey and Conradie, 2013; Conradie and Piesse, 2013). This estimate of success is still relatively high, since it includes all types of lethal control – extermination has been shown to be a much more difficult goal in the Namibian situation (Swanepoel, 2016; see text box: *Historical note on effectiveness of hunting predators* above).

- Advantages: Once a predator species has been locally extirpated in a whole district or larger ecosystem, it is relatively cheap to maintain (Nattrass and Conradie, 2015), and boils down to preventing predators from re-colonising (Linnell et al., 1996). Among other things, it means that livestock can freely range on the land without the need for kraaling or herding, leading to longer grazing times, better veld utilization and higher productivity if used together with camps and rotational grazing (Beinart, 1998; Swanepoel, 2016). It is also relatively simple to implement, with little need for specialized knowledge or equipment (Du Plessis et al., 2018). In highly transformed landscapes, with little or no natural prey available to predators (and livestock being their only possible source of food), it is often the only realistic short-term option (Linnell et al., 1996). It can also be combined with other methods to eradicate and then exclude predators from specific parts of a farm (personal observation). According to Shwiff and Bodenchuk (2004), in addition to the livestock saved, it can have secondary tangible advantages (e.g. more game because of predator control) and intangible advantages (like farmers being more positive towards conservation because they are allowed to kill predators).

Shwiff and Bodenchuk (2004) also make the very questionable statement that "it is desirable but not necessary to achieve economic efficiency in predation management programs" — for commercial farmers the economic efficiency (cost-effectiveness) is the most important aspect of all farm management, including predation management (Macaskill, 2013; Dieguez Cameroni and Fort, 2017; Turpie and Akinyemi, 2018). Any program that costs more than the resulting decrease in livestock losses (high cost and low effectiveness in Figure 2.2a), is likely to result in bankruptcy (Nattrass and Conradie, 2018).

- Disadvantages: Since hunting is relatively labour intensive, it is also expensive in terms of time and money initially (Linnell et al., 1996). It can become very expensive, especially when professional hunters and

helicopters are used (M. Bijsterbosch 2018, Wildlife Vets Namibia, personal communication; Jaeger, 2004). It is very unlikely to succeed in exterminating all predators, except for conspicuous, large predators that are easy to find and kill (Beinart, 1998; Kissui, 2008). Exterminating top predators have other unwanted results, like an explosion in numbers of medium-sized predators (mesopredator release) and often do not solve the actual problem of livestock depredation (and as shown by Conradie and Piesse, 2013, might actually increase livestock losses). Using a 60-year data set, Berger (2006) showed that although United States of America (USA) government-subsidised predator control was very effective in killing predators, specifically coyotes (*Canis latrans*), it had no effect on sheep (*Ovis aries*) production.

- Limitations: Unless there is official government support, with farmers and land-owners in a district working together, it is very unlikely to succeed (Beinart, 1998; Nattrass and Conradie, 2015; Swanepoel, 2016). It is thus not practical for a single farmer to implement without the support of all his neighbours. Moreover, it is really only suitable to transformed landscapes (e.g. cultivated pastures) where no natural habitat or prey survive (e.g. parts of Europe where few predators would survive without livestock). It is not well suited for most of Africa with its abundant natural prey and habitat that would sustain predator populations even in the absence of livestock (Linnell et al., 1996). Factors like rainfall, topography and soil fertility limit the practicality of deliberately transforming the vegetation. Where the predator species are endangered (e.g. cheetahs or leopards) or otherwise important for a functioning and sustainable ecosystem, this method is not an option (Linnell et al., 1996; Estes et al., 2011).

2. Hunting with dogs for eradication: Dogs (*Canis lupus familiaris*) are often used together with hunting for exterminating predators. Therefore, it shares the same advantages and disadvantages as hunting for eradication and has a similar high ecological impact and low probability of being sustainable. It is based on the same defective ecological principle of humans totally replacing exterminated carnivores in the ecosystem. Already in the early 1800 Lord Charles Somerset introduced the English sport of fox hunting to the Cape colony for hunting jackals (Beinart, 1998). By this time most larger predators had already been extirpated and jackals were the major remaining cause of livestock depredation for sheep farmers. Since 1962 breeding centres for hunting hounds were established in a number of places in the old Cape Province, and even later Oranjejag, in the Free State Province, used dogs for hunting (Daly et al., 2006; Bergman et al., 2013). This method has also been used recently in Namibia with "great success", if success is measured as numbers of carnivores killed (Axel Rothauge 2014, personal communication; Swanepoel, 2016). However, in both the Natal and Free State Provinces, hunting with hounds did not decrease jackal or caracal numbers (Bothma, 2012) and can be considered to have a similar prior probability of success as hunting for extermination (0.57).

- Advantages: Using smell, dogs can be much more effective in finding predators than human trackers alone. This method is therefore more likely to succeed against smaller predators like jackals and caracals than hunting without dogs for the extermination of predators. Furthermore, a group of well-trained dogs are probably still cheaper than using helicopters or professional hunters (e.g. Swanepoel et al., 2016 reports its use by subsistence farmers). If used as a reactive method and with a well-trained pack, it can be very target-specific in following the tracks of the specific individual predator from a fresh kill site (Linnell et al., 1996; Snow, 2009).

- Disadvantages: The greatest disadvantage of using dogs for local predator extermination has been the indiscriminate nature of their hunting behaviour. Even well-trained dogs that can distinguish between prey animals (which should not be killed for various reasons) and carnivores, do not generally differentiate between damage-causing predator species and other carnivores, with the result that many non-target carnivores like bat-eared foxes (*Otocyon megalotis*), Cape foxes (*Vulpes chama*), aardwolves (*Proteles cristatus*), servals (*Leptailurus serval*), and African Wild Cats (*Felis lybica*) are killed as well (Bothma, 2012). Moreover, even among the target species the individuals are killed indiscriminately unless tracks are followed from a fresh kill site. Hunting with hound packs for eradication has failed, resulting in many

predators being killed, but with little long-term decrease in predator numbers or livestock depredation (Daly et al., 2006; Du Plessis, 2013). Additionally, dogs can have a devastating effect on the natural prey of predators and on innocent carnivores and if not well-trained, can even kill livestock. If outside dog packs are rented to kill predators, some dog handlers only use male dogs in their hunting pack, knowing full well that male dogs often will not kill female jackals (Henrico Engelbrecht 2016, owner of hunting dogs, personal communication). Thus, the jackal population is managed for sustainable harvesting instead of eradication (Treves and Karanth, 2003) and the dog owner ensures that he will have work again in the following year.

- Limitations: Good hunting dogs require good training not to kill non-target animals (Snow, 2009). This is both time-consuming and not always effective when considering the number of non-target animals killed (Snow, 2009; Bothma, 2012; Du Plessis, 2013). Historically, it has never been successful in completely eradicating predators without similar constraints as hunting; with the need for at least district-wide collaboration, government support to make it affordable, and transformed landscapes with fragmented ecosystems in a "fence and clean-up" approach (Beinart, 1998; Nattrass and Conradie, 2015; Swanepoel, 2016).

3. Poison bait to exterminate predators: Poison had been used in the Cape Province for eradicating problem predators from the 1880's (Beinart, 1998; Bergman et al., 2013). Different poisons have been used, including Strychnine, Compound 1080 (Sodium monofluoroacetate), Sodium Cyanide, and Thallium Sulphate (Linnell et al., 1996). The use of poisons were largely discontinued and replaced with jackal-proof fencing and hunting clubs in the early 1900's, mostly because of the ineffectiveness of poison, in addition to the negative ecological side-effects (Beinart, 1998). However, in recent years poisons have become popular again (perhaps out of desperation because hunting no longer works) in some parts of South Africa (Nattrass and Conradie, 2018). It is also the most popular method used in Australia against dingos (Wallach et al., 2017). Even though illegal in Namibia, about 20% of farmers, mostly small livestock farmers in the South, still use poison, of whom most use it reactively after livestock losses (Santangeli et al., 2016). Poison can be applied in different ways, including poisoning carcasses, bait, coyote getters (M44), and poisoned collars (Linnell et al., 1996; Du Plessis et al., 2018). Because poison collars normally include the death of the livestock animal that is killed by the predator, it is considered a reactive method and not discussed here. Ecologically, poisoning predators has the same unrealistic aim as the previous two methods (1 & 2 above) of humans taking over the ecological roles of extirpated carnivores. Additionally, it includes various unsustainable ecological knock-on effects and is therefore considered as having a very strong negative ecological impact (Santangeli et al., 2016). In light of the fact that historically poison use had been largely abandoned as ineffective, the prior success probability of 0.57 for lethal methods derived from Van Eeden et al. (2018), is probably over-optimistic for poison.

- Advantages: The only real advantage that poisons have, is that it is easy to use and some poisons are easy to acquire (Ogada, 2014).
- Disadvantages: It is almost impossible to use poison proactively to selectively kill predators and historically it has been more successful in killing scavengers than predators (Beinart, 1998; Bothma, 2012; Du Plessis et al., 2018). Some commonly used poisons often kill secondary scavengers (e.g. vultures that prey on predators that died from poison) and have other knock-on effects (Santangeli et al., 2016). Moreover, while killing many non-target carnivores, poison has not been able to exterminate jackals or decrease livestock losses, and shows a similar failure against dingos in Australia (Wallach et al., 2017; Nattrass and Conradie, 2018).

Marks and Wilson (2005) demonstrated that the M44 could be made more species-specific by using the weight (correlated with pulling force) of the animal to exclude smaller non-target species. However, this method would still kill larger species, unless they were so big that the dosages are too small to affect them, or naturally more resistant to the poison than the target species (e.g. the Tasmanian devil, *Sarcophilus harrisii,* was 32 times more resistant to 1080 than the red fox *Vulpes vulpes*).

- Limitations: Poison is an indiscriminate killer and not really effective for reducing livestock depredation (Bothma, 2012; Ogada, 2014; Santangeli et al., 2016; Wallach et al., 2017). It shares the limitations of the other methods used for extermination. Moreover, because of their similarity to target predator species, it precludes the use of hunting dogs or livestock guarding dogs. There is good reason for it being illegal in Namibia, with at least one vulture species nationally extinct largely because of poison use (Simmons et al., 2015; Santangeli et al., 2016).

4. Carnivore-proof fencing of predator-free areas: In Southern Africa, jackal-proof fencing was first introduced in the early 1900's and became compulsory in the sheep-producing areas (Bergman et al., 2013). With government subsidising jackal-proof fencing, combined with setting up various hunting clubs, the attempt at eradication of black-backed jackals finally succeeded in some districts (Beinart, 1998) to the extent that by 1967 the government felt that predators were relatively under control and no longer needed direct government involvement (Bergman et al., 2013). A totally predator-proof fence is possible, but very expensive, and only really economically viable for small breeding camps of high-value game species, like sable *(Hippotragus niger)* and roan *(Hippotragus equinus)* (W.P. Barnard 2015, personal communication). Predators are not killed by the fence itself, but by constricting the movement of predators within the camp and clearing the camps of predators using other methods, it can result in an almost total eradication of the predators in a whole district in a "fence and clean-up" method (Beinart, 1998; Nattrass and Conradie, 2015). Once this has been done, the fences serve to prevent recolonisation by predators (Linnell et al., 1996). Where predators had been eradicated from farmlands, Packer et al. (2013a) proposed the use of carnivore-proof fencing to keep large predators within protected areas, as the most cost-effective method of protecting predators, lions in particular (Packer et al., 2013b). This is basically an application of the zoning approach proposed by Linnell et al. (1996) (*cf.* Shivik, 2004). The basic ecological principle of this method is to prevent the dispersal and recolonisation by predators of areas from which they have been eradicated (as could happen in nature because of natural barriers – MacArthur and Wilson, 1963; Ferguson et al., 1983). Its ecological impact and sustainability will depend largely on the size of the area from which predators are excluded (small pastures/large camps/whole farms/whole districts) and how much it restricts the free movement of other wildlife (Skead, 1980; Beinart, 1998). For a new implementation, this method can be considered as having a similar success probability to other eradication methods (0.57) (Van Eeden et al., 2018). Once local extirpation has been achieved, it can be considered as having a similar prior probability of success as exclusion fencing in general (0.87) (Van Eeden et al., 2018). As exclusion fencing, within the optimal foraging framework of $H = C + P + MOC$ (see text box on page 38 above for the meaning of the formula variables), this method attempts to increase the cost of preying on livestock (digging underneath, breaking through or climbing over the fence) to higher than the greatest possible energy gain from killing livestock.

This method mostly differs from other methods where fences are used for excluding predators with regards to the *size* of the excluded area — instead of having a single, relatively small, camp from which predators are excluded, predator exclusion is done at the level of a whole farm and/or district. *This is one reason why it is considered as having a high ecological impact.*

- Advantages: For almost total eradication of predators, this is one of the only methods that have proved to be successful against black-backed jackal in some areas.

- Disadvantages: Putting up predator-proof fencing, especially without any government subsidies, is very expensive. Predator-proof fencing does not only stop predators, but also most other wildlife from freely moving around. This has a negative effect on the overall biodiversity of the veld and its long-term sustainability. Because digging animals like warthogs (*Phacochoerus africanus*) and aardvarks (*Orycteropus afer*) often dig underneath fences, it is important to regularly patrol and maintain all fences, if the fences are to be effective at all. This can be quite expensive and labour-intensive and with the typical increase in labour costs, no government subsidizing and low profit margins of livestock farming, make it more expensive and less effective than alternatives — Van Niekerk (2010) estimated that proper fence maintenance doubles the labour costs on a farm. Once the predator is inside the camp it can kill prey (including livestock) easier by cornering them against the fence, often leading to surplus killing. This is a potential disadvantage of all kinds of fences, including kraals (Ray et al., 2005).

- Limitations: Truly predator-proof fencing is not possible in certain terrain, like very mountainous habitats. Jackal-proof fencing, while relatively effective against jackals, are not effective against caracals and leopards, so it may only be effective when jackals are the only significant predator species. With the high costs of proper predator-proof fencing, the age and state of disrepair of current fences, the lack of government subsidies, and the lack of district-wide cooperation, carnivore-proof fencing to exclude predators from their land becomes an increasingly impractical option for a typical livestock farmer. The increase in "weekend" farmers with less time to spend on maintenance, makes any district-wide attempts at predator eradication impractical. The fall in profitability of farms per hectares, resulting in fewer farmers and larger farms to remain economically viable, also makes the maintenance of fences more difficult (Nattrass and Conradie, 2015). Linnell et al. (1996) concluded that fencing is only really practical to keep predators out of specific, relatively small, grazing pastures (discussed further below on page 59).

5. Hunting to reduce carnivore numbers: Many farmers realize that it is impractical to attempt eradication of all predators. And most farmers prefer not to kill predators at all, if it did not threaten their livelihood (Rust and Marker, 2013a; Rust, 2016). Instead of trying to eradicate all predators, some farmers shoot predators on sight just to "keep their numbers down". The farmer simply keeps his rifle with him when driving through his farm and shoots any potential livestock predators that cross his path during the working day (Francois Kok 2015, personal communication). In contrast to the attempts to eradicate predators, usually little extra effort or money is spend to actively hunt predators. Hunting that starts off with the aim of eradication, often ends up as an attempt to simply reduce predator population densities (Minnie et al., 2016b; Swanepoel, 2016). Where culling, rather than eradication is the aim, it has been shown that the season (just after the dispersal period, as the mating season starts) is very important for its short-term effectiveness in controlling red foxes (Lieury et al., 2015). Ecologically, in this method humans take the over the role of an apex predator killing smaller predators (Palomares and Caro, 1999; Ritchie and Johnson, 2009). Within the optimal foraging framework ($H = C + P + MOC$), the predation risk (P = risk of being killed by humans) is increased. Although theoretically a more realistic aim than humans replacing *all* carnivores, it is unlikely to have the same ecological effect as an actual apex predator: the "landscape of fear" created by the stronger competitor in intra-guild interference competition has been shown as having a greater effect on the density and distribution of mesopredators than the numbers being killed (Flagel et al., 2017). The predation risk is not so much raised for preying on livestock only, or only in some areas of high human presence (near livestock), but even when hunting natural prey (i.e. no heterogeneous "landscape of fear" with distinct areas of high risk is created). By reducing predator numbers to their "economic carrying capacity" (Figure 2.1) it can actually be counter-productive, resulting in an increase in predator densities instead (see text box below). However, it is still considered as having a medium ecological impact compared to the previous methods which have highly negative ecological impacts. Other methods have also been used to try and reduce predator numbers, like denning (killing predator pups with fumigating poisons within their dens) or contraception, but these methods are impractical or ineffective for most farmers (Jaeger, 2004). From the summary of articles by Van Eeden et al. (2018) a prior success probability of 0.57 can be

given to this lethal method. This is probably also the method responsible for most of the 20% chance of lethal methods *increasing* livestock losses (see text box *Counter-productive predator management* below).

- Advantages: This is one of the cheapest and easiest methods available to a farmer, on average the cost of a single bullet for every predator killed.
- Disadvantages: Jaeger (2004) showed that in coyotes it is mostly the adult territorial breeding alpha pair that are responsible for most livestock kills, explained by their higher energy requirements to feed their young (Sacks et al., 1999). However, they are also the most difficult to kill. It is possible that the same is true for at least some Namibian predators and it is usually young, sub-adult predators who have not yet learned that people can be dangerous, who are most likely to be seen and killed. Like other lethal methods, it can have the counter-intuitive result of increasing livestock predation (Bailey and Conradie, 2013; Conradie and Piesse, 2013).

Counter-productive predator management — For most territorial predators, their ecological carrying capacity (Figure 2.1) is mediated through territory size, resulting in self-regulation of their densities in an ideal pre-emptive distribution (Pulliam and Danielson, 1991; López-Bao and Rodrigues, 2014). The possible exception among the four Namibian carnivores, is the black-backed jackal (see below), where territorial pairs would allow other jackals into their territory to share large food sources or water, as long as they show submission to the territorial pair (Ferguson, 1978; Jenner et al., 2011). However, they would not allow any other jackals to *breed* in their territory, which would still regulate the long-term equilibrium density (Estes, 1991). In killing predators, the population density is decreased below the ecological carrying capacity, resulting in various compensatory immigration and reproduction responses to the disruption of social and territorial structure (Minnie et al., 2016b). Removing nine territorial female pumas (*Puma concolor*) resulted in them being replaced by 13 younger females and three males were likewise replaced by four in the same area (Linnell et al., 1996). Similarly, Stein et al. (2011) found higher leopard densities on the farms around the Waterberg National Park than in the park itself. Killing predators can create a source-sink system, in which sink populations often have higher densities than the source population (Pulliam, 1988), resulting in higher livestock depredation (Conradie and Piesse, 2013).

- Limitations: This method has very few constraints and can be applied on almost any farm. However, it can not be used together with any method that depends on predators learning to avoid livestock, nor with those predator species that respond with over-compensation to their removal.

6. Trapping: Whether the aim is the extermination of predators or just reducing predator numbers, the costs and effectiveness of trapping is more or less the same. Ecologically, it is based on the same principles as either extirpation (humans replace all carnivores – methods 1 - 3) or culling (humans attempt to emulate an apex predator – method 5 above) (Palomares and Caro, 1999). Where it differs significantly from the effect of an apex predator, is that predators usually make their presence known, either vocally or by scent-marking (Jackson et al., 2012; Melzheimer et al., 2018). Instead, trapping depends on the target predator not detecting the trap. It is therefore likely to be similarly unsustainable with a medium ecological impact and even less likely to create a "landscape of fear". Different kinds of traps are commonly used, including snares, leg-hold traps (gin traps), killer traps (quick kill traps / cony bear traps) and cage traps (Snow, 2009; Swanepoel, 2016). Traps are often used together with fences to catch animals as they move through holes in the fence (personal observation).

- Advantages: Some traps, including most cage traps and some kinds of leg-hold traps (soft traps) and even some kind of leg-hold snare traps, can be quite selective: while not selective in which animals they catch, when causing no permanent injury to the animal caught, a beneficial or non-target species can simply be released back into the wild (Kamler et al., 2008). If a specific individual problem predator is responsible

for all or most livestock depredation, such selection can even be at the individual level. The equipment is also fairly inexpensive and normally easy to use (Snow, 2009).

- Disadvantages: Most traps by themselves are fairly indiscriminate. They need to be checked at least once per day in order to prevent the death or injury of non-target species (Snow, 2009). Some kinds of traps, like most snares, all killer traps and most gin traps, often cause so much injury to the caught animals, that even non-target species have to be put down after having been caught (Kamler et al., 2008). These kinds of traps, even if checked regularly, are still non-selective. Cage traps are totally ineffective against jackals (Humphries et al., 2016), meaning that their use will result in more felid predators being killed. This will cause a change in any pre-existing ecological equilibrium (balance) with a resultant increase in both jackals and livestock depredation by jackals (Holmes and Holmes, 2006). Moreover, some species like leopards, react very aggressively to being caught in a cage trap and in the process often injure teeth or claws (Frank et al., 2003). This decreases their hunting ability, sometimes forcing them to start killing livestock (or even humans - Athreya et al., 2010) to survive if they are released again (see method 39 "Translocation", below on page 84). Like other lethal methods aimed at either the extermination of predators or just a decrease in their numbers, the evidence suggest that trapping has little long-term effect on livestock depredation levels (Berger, 2006), and might even be counter-productive (Conradie and Piesse, 2013). Setting and checking traps is also relatively labour intensive (*cf.* McManus et al., 2014) and many farms using this method need at least one labourer permanently employed for this purpose (Corrie van der Westhuizen 2016, personal communication). If traps are not checked regularly, it leads to the cruel deaths of many innocent or beneficial wild animals (up to 80% of the animals caught according to a Cape Leopard Trust survey in Namaqualand; Ben-Jon Dreyer 2012, personal communication; 50% of farmers in Snow, 2009). If, as is common, the trap is set at a hole underneath the fence, once an animal has been caught in the trap, predators can move freely through that hole (and even scavenge on the animals already caught in the trap), in effect making the fence useless for keeping predators away from livestock.

- Limitations: If the farm is very big, or the workforce on the farm very small, it might be impossible to check the traps regularly and their use becomes unethical (McManus et al., 2014). Setting traps to catch animals of the right size and without injury, requires some specialist knowledge (Snow, 2009). For trapping to be effective in either exterminating predators or decreasing livestock depredation, it really needs to be used district-wide (with other lethal methods). It is most likely to succeed in transformed landscapes with little natural habitat or prey for predators. As mentioned, some trapping methods work better against some predators while not working against other predator species at all. Black-backed jackals specifically, are notorious for being very difficult to catch, and avoiding both poison and cage traps (Brand et al., 1995). Even using leg-hold traps for catching jackals can be a challenge (e.g. 124 capture nights/jackal Humphries et al., 2016).

7. Herding: Even before hunting, most of the indigenous livestock-keeping people groups used herding and kraaling to protect their livestock from depredation (Ogada et al., 2003). Shortridge (1934) mentioned that in the early 20th century the black-backed jackal was not considered a problem species in Namibia (unlike South Africa), because most farmers used herders and kraaled their livestock at night. Ecologically, this method depends on the behaviour of predators avoiding direct interference competition with dominant species (Durant, 2000). The herder and guarding dogs are seen as dangerous competitors and are avoided (Cristescu et al., 2013). Typically, most predator species learn to hunt or are taught to hunt by their mothers (Moehlman, 1987; Estes, 1991). If predators seldom or never have the opportunity to predate on livestock, they are less likely to imprint on livestock as preferred prey (Hayward et al., 2006b,a). Within the optimal foraging framework ($H = C + P + MOC$), the method increases the cost (C) of hunting livestock, both by lowering encounter rates and by the greater total vigilance ("many eyes") of a large herd and the herder with dogs (King et al., 2012). It also increases the predation risk (P) of the predator being killed by either the herder or the livestock guarding dog (Potgieter et al., 2016). Being spatially and temporally restricted, this method has little ecological

impact and is thus highly likely to be ecologically sustainable. Worldwide, herding has traditionally and almost universally included the following aspects (Linnell et al., 1996): 1) By day the animals were allowed to graze as a herd, under the care of a herder. 2) Livestock were prevented from spreading out, either by the herder or a herding dog. 3) The herder directed their movement and selected the areas to be grazed during the daytime. 4) At night the flock was rounded up and either herded into a fenced area (kraal/corral/boma) or bedded under supervision in an open area where a shepherd slept near the flock (*cf.* Stone et al., 2017). 5) In most European and middle-Eastern regions, large guarding dogs were also kept with the flocks both by day and by night. Overgrazing was avoided by keeping the flocks on the move. In Van Eeden et al. (2018) guarding (both dogs and human herders) had a probability of success of 86.8% (0.87).

- Advantages: Herding with kraaling, and herding and/or guarding dogs, is still one of the most effective methods to prevent livestock depredation (Miller et al., 2016b; Van Eeden et al., 2018). Herding dogs can additionally make life easier for the herder by keeping the flock together and pushing them in the direction that the herder wants.
- Disadvantages: With settled farms instead of the nomadic lifestyle of the past, the greatest disadvantage of herding when it is combined with kraaling (as it almost always is), is the resulting trampling of the veld around the kraal. Another common objection is that, especially in hot climates, the livestock are not able to graze at night. Free-ranging livestock will take advantage of the cooler nights, especially on moonlit nights, to graze during the night and rest during the heat of the day in summer. This becomes impossible when herding and kraaling, resulting in an appreciably lower meat production, similar to that mentioned by Hansen (2005). Herding dogs need a lot of training for handling livestock. It also takes time for a good herder to get to know the veld, good grazing areas, predation danger spots and his livestock (and dogs) well.
- Limitations: The greatest limitation on using herders, is the difficulty of finding reliable herders. In Southern Africa, it has traditionally been the job of children, resulting in it being a low-paying and low-status job on most farms. An experiment by Conservation South Africa and the Cape Leopard Trust in Namaqualand evaluated "ecorangers", specially trained herders who used mobile GPS devices with a CyberTracker application to record data on livestock movements, depredation losses, predator tracks etc. From a group of 16 who went through a process of selection, interviews and training, at the end of the research period only eight were found to be reliable and doing a good job caring for the livestock, and only five said that they would consider a permanent career as ecoranger in future. It was found that one of the best incentives for a herder was owning some of the livestock (Corrie van der Westhuizen, Brenda Snyman, 2016 personal communication), something that few farmers might be willing to do (Rust et al., 2016). Alternatively, costs for herders can be sponsored from outside (e.g. the ecoranger project was sponsored in part by the government, or the Wildlife Guardian program of Defenders of Wildlife mentioned by Shivik, 2004), but this is unlikely to be sustainable.

8. Patrolling to scare away predators: "Range riders" is a method introduced by Defenders of Wildlife in the USA (Stone et al., 2008). Basically it involves using horses (*Equus ferus caballus*) to ride through areas where the livestock are grazing, visually assessing their health and condition, and using the human activity to scare predators away from the livestock. In the USA it is used for extensive farming on an open range. A similar approach has been used in extensive systems in Norway using a range inspector patrolling with a livestock guarding dog (LGD)(Hansen, 2005). Because of the resistance by some farmers against kraaling their livestock at night, the research project in Namaqualand by the Cape Leopard Trust (CLT) and Conservation South Africa (CSA) also used some of their ecorangers as range inspectors to patrol the camp where the livestock were with the dog. This was done to scare away predators through their presence (personal observation 2016). The ecological principle is basically one of creating a "landscape of fear" with humans or LGDs as the apex predator (Ritchie and Johnson, 2009; Nattrass and Conradie, 2015). Flagel et al. (2017) showed that the effect

of top predators on mesopredator densities through fear can be greater than actual intra-guild killing. It would appear obvious that the human presence should be concentrated in those areas with livestock (Stone et al., 2017). If correctly applied, this method increases the predation risk to predators when coming close to livestock within the optimal foraging framework ($H = C+P+MOC$). Because this method has not really been evaluated for its effectiveness in multiple studies, it is given the unknown prior probability of 0.5 (50/50 chance of success or failure).

- Advantages: This method is well suited to more extensive farming systems, where daily kraaling is not practical (Stone et al., 2008). In these kinds of systems the livestock is seen more frequently and any non-predator related problems or veld condition issues are also picked up earlier. Spoor tracking also gives an idea about where the predators spend most of their time and from where livestock should be moved for their own safety. Where the livestock breeds do not have a strong natural herding instinct or are forced to scatter to find enough grazing, this is one of the few methods that can still reduce livestock depredation (Linnell et al., 1996). If horses are not used, the costs of this method is about the same as that of a herder (and dog), but has the advantage that the dog does not have to first bond with the livestock and that the livestock can graze at night and thus grow and reproduce better.
- Disadvantages: Because most predators hunt at night when human presence is less likely to have an effect, this method is not very effective. Even Stone et al. (2008) had to admit that there was no direct proof "that range riders actually prevented livestock losses from predators". Labour is an ever-increasing agricultural cost and rangers that can ride horses in addition to the necessary horse-riding equipment (saddles, bridles etc.) can be quite expensive.
- Limitations: This method is more likely to work in flat open areas and may not be a good fit for thick bush or rough, mountainous farms. It is only likely to be worth the expense if livestock losses are relatively high, unless the farmer already has horses, riding gear, and labourers that can ride.

9. Kraaling at night: Kraaling involves the practice of having livestock sleep in an enclosure (called a kraal, corral, pen, or boma, depending on geographical part of the world) at night, when most predators are active. Ecologically, it mimics natural refuges from predation (Huffaker, 1958; Durant, 1998). In this sense, it has little ecological impact. However, it could result in trampling of the veld by forcing livestock to return and sleep in the same place every night, something that would not happen with most natural herbivore herds. Regardless, it can be considered as having relatively little ecological impact if the same kraal is not used continuously. Within the optimal foraging framework ($H = C + P + MOC$), it increases the cost (C) of hunting livestock (requiring the kraal fence to be breached) as well as increasing the predation risk (P) by forcing a nocturnal predator to hunt livestock during day-time, by having the kraal close to human habitation or having a livestock guarding dog sleeping within the kraal. These enclosures are usually made predator-proof in some way (Ogada et al., 2003; Woodroffe et al., 2007). Begg and Kushnir (2013) list a number of different kraal types that will protect against lions, including traditional *Acacia* thorn kraals, stone kraals, chain linked fence kraals, "living wall bomas" (Lichtenfeld et al., 2015), mobile kraals and small log pens for kids or lambs. Weise et al. (2018) makes the very important observation that the size, shape and type of kraal makes a real difference to its success rate. Miller et al. (2016b) found that kraaling at night reduced livestock losses by between 58% and 100%. Van Eeden et al. (2018) (including Miller et al.'s study) found an average success rate for all enclosures (including other methods like electric fences) of 86.6% (prior p(success) = 0.87).

- Advantages: Kraaling, combined with herding or guard animals, are often the only option where there are high predator densities. Kraaling can be used on its own with the kraal built around water troughs and kept closed while the livestock are out grazing and are only opened when they are let into the kraal at nightfall. When combined with guarding dogs sleeping inside the kraal, it is one of the most effective ways for keeping depredation low at night. Costs for building kraals can vary between only including

labour costs for traditional thorn bush (high maintenance) and stone kraals (almost no maintenance) to US$ 200.00 - US$ 900.00 for a chain link fence, an average of US$ 500.00 for the living wall bomas (almost no maintenance), US$ 1 300.00 for Non-Government Organization kraals built with local material (high maintenance), and about US$ 2 000.00 for a mobile kraal (Begg and Kushnir, 2013; Weise et al., 2018).

- Disadvantages: The greatest single objection to kraaling, is the trampling of the veld that usually happens around the kraal. If there is water available in pans or wetlands during the rainy seasons, some livestock might also not come to drink water at the kraal and thus sleep outside and remain vulnerable to depredation, when kraaling is not combined with herding (Weise et al., 2018). Another important draw-back is the accumulation of parasites and pathogen spores, and the resulting outbreaks of disease in livestock, which is much more likely to happen in the crowded situation and repeated use of the same area, typical of kraaling (Beinart, 1998).

- Limitations: Different levels of predator proofing are effective against different predators. Jackal-proof fences are easy for caracals and leopards to climb and might cause very high surplus killing if one of these species gets inside the kraal (personal observation; Ray et al., 2005; Weise et al., 2018). Moreover, the kraal obviously has to be strong enough to keep the livestock inside as well (e.g. fighting bulls or frightened cattle might break out of an enclosure that is not strong enough). While some of the veld trampling can be avoided by using mobile kraals that are moved occasionally, these mobile kraals are often insufficient to keep predators out (Stone et al., 2017; Weise et al., 2018). Additionally, with many different flocks in different camps, this method can be quite labour-intensive and impractical on large, extensive or mountainous farms where reaching the furthest camps can take many hours.

10. Kraaling young livestock full-time: In contrast to kraaling of all livestock at night, this involves keeping the young animals in a predator-proof enclosure permanently, with the adult females joining their young to suckle only at night while grazing during the day. Or alternatively (especially during hot summers), having the adults graze at night and suckling their young during the day in a kraal around a water trough. Ecologically, the maternal care behaviour of "hider" prey species or the "nursery groups" found in some herbivores, are mimicked in this method, with a general low impact (Estes, 1991; Howery and DeLiberto, 2004; Klare et al., 2010; Hayward et al., 2017; see text box on *Maternal care strategies* below). Within the optimal foraging framework ($H = C + P + MOC$), the cost (C) of hunting livestock is increased by excluding the most vulnerable individuals from the available free-roaming livestock population. This has been a traditional livestock management technique in Northern Namibia (Francois Kok, Johann Britz 2013, personal communication; Marker-Kraus et al., 1996). A variation on this method would be to kraal all livestock full-time in a feeding lot. However, feeding lots are not economically viable in Namibia (NAMMIC, 2011), mostly because of high prices for livestock feed and long distances to market. Following Van Eeden et al. (2018) a prior success probability of 0.87 can be given to this method.

- Advantages: For cattle this method has been shown to be quite effective in North-Central Namibia. In some situations, with high livestock depredation levels, it might be one of the few solutions available. For natural "hiders", like goats (*Capra aegagrus hircus*) (and cattle to some extent), this method combines well with intensive care of the young by the farmer. While calves staying in a kraal initially grow slower than free-ranging calves who are permanently with their mothers, by weaning age their weights are similar (Johann Britz 2013, personal communication).

Maternal care strategies — Ungulates with young typically have two maternal care strategies: 1) "Hiders", where the young are not strong enough to keep up with the adults and would normally remain hidden in one place until they become stronger, while their mothers graze alone or with the herd, and 2) "followers", where the young are strong enough to remain with their mothers after birth and join the herd soon after (see Estes, 1991). Some ungulates also use a "nursery" strategy, where the young ones that have already joined the herd is left hidden together as a group with one or more adult females keeping watch over them, while the other mothers go and graze or drink water in areas of higher predation risk (this strategy is commonly found in cattle – personal observation). See also Estes (1991).

- Disadvantages: This method is fairly labour-intensive (and thus costly). For each camp or cattle post with cows, a reliable worker is needed to let the adult cows out, while keeping the calves inside every day, seven days a week. To be at all practical, it has to be combined with birthing seasons. If a single central kraal is used, the same issues with trampling of the veld and weaker livestock growth and production starts to play a role. Some farmers also complain that this method results in cows having weaker protective and mothering instincts when their calves are finally allowed to go out and graze with them.

- Limitations: This is not a practical solution for extensive farming systems (e.g. where few water points, rough terrain and low grazing capacity result in very large camps that a grazing cow cannot cross in single day) or where livestock do not have a single breeding season.

11. Electric fencing: Electric fencing can be used either as permanent predator-proof fencing or as mobile, temporary fencing, primarily for livestock (usually as part of holistic short-duration high-intensity grazing systems). For mobile electric fences, a single strand is often used to keep the livestock together and while it might not keep the predators out, by keeping the livestock together, it effectively causes the livestock to use other anti-predator behaviours (like keeping livestock bunched together for protection and cattle protecting small livestock – method 24 on page 70). Stone et al. (2008) showed that combining electric fencing with fladry (a series of cloth flags hung at 45 cm intervals along a thin rope) can have better results than either on its own. It depends on the same ecological principle of creating predation refuges (see method 9 above), but because it can kill some beneficial small animals and stop the movement of other wildlife, it should be considered as having a higher ecological impact, depending on the size of the fenced area (Macaskill, 2013). Ecologically, it is simply a more effective carnivore-proof fence (see method 4 above), and similarly aims to increase the cost (C) of livestock predation within the optimal foraging framework ($H = C + P + MOC$). The effectiveness of electric fencing as such compared to other methods has not been done widely enough to give it a prior probability. Therefore, it is given the same p(success) = 0.87 as enclosures in general (Van Eeden et al., 2018), but probably has a higher effectiveness against some predators than other types of enclosures.

- Advantages: Electric fences can keep predators like caracals and leopards as well as jackals out, if high enough and installed properly. It can also prevent digging animals like aardvark or warthogs from making openings below the fences that predators can use to move through. Mobile electric fencing can be used as a management tool for short-duration, high-intensity grazing systems ("holistic grazing" – see method 31 below) without being nearly as labour-intensive as normal herding. It can be a very effective method to keep predators away from a relatively small area.

- Disadvantages: Electric fencing is quite a bit more expensive than other fencing and requires similar regular maintenance. Additionally, it does not work in all kinds of terrain, especially not in high or bushy vegetation. It also has a negative effect on ecology, biodiversity and sustainability by stopping the movement of all wildlife and often killing innocent or beneficial animals (>30 different species – Macaskill, 2013; Du Plessis et al., 2018).

- Limitations: Because it is so expensive, electric fencing is not an option for fencing whole districts. Even fencing a whole farm will be too expensive for a typical livestock farmer to afford, especially in areas with low economic or ecological carrying capacity (Caughley, 1976).

12. Calving or lambing camps: If possible, having safe areas for the vulnerable livestock, like young animals, is a good management practice. In higher rainfall areas, such camps can be irrigated or cultivated with high-quality grazing. Because of the low rainfall, this is not an option in most of Namibia, though. The main difference between small enclosures (kraals) and calving or lambing camps, is that the livestock (mother and young) can *graze and browse* inside the exclusion camp. Ecologically, this method creates an artificial predation refuge (Huffaker, 1958; Durant, 1998). Within the optimal foraging framework ($H = C+P+MOC$), it increases the cost (C) for the predator to hunt livestock (having to breach the fence in some way). Like the following preventative methods, it has a lower ecological impact and high sustainability from an ecological perspective since it is restricted to a relatively small area. This method is typically what much of the international literature from higher-rainfall areas refers to when discussing fencing as a livestock protection strategy (Paige, 2015; Stone et al., 2017). Like other enclosure measures, this method is given a prior probability of success of 0.87 (Van Eeden et al., 2018).

- Advantages: In addition to making it easier to protect livestock from predators, having small camps close to the homestead, makes more intensive livestock management possible as well. The increase in the survival of young may compensate for any decrease in growth rates of the young livestock. Where irrigated pastures are possible, it can actually lead to an increase in production compared to more extensive free-range systems. This method can also be easily combined with various other methods.
- Disadvantages: Similar to kraaling, an accumulation of parasites and disease epidemics are more likely than with free-ranging livestock. Grazing close to the homestead and trampling of the veld, might lead to lower production than free-ranging options and make this method less sustainable.
- Limitations: The availability of enough good grazing around the homestead for all female and their young (if calving/lambing seasons are used), is an important limitation on this method. If the young are born throughout the year, good grazing close to the homestead would need to be available the whole year round. It is thus not a well-suited method for extensive farming in areas with low sustainable stocking rates.

13. Static visual and acoustic predator repellents (deterrents): These kinds of repellents are normally used for relatively small areas on the farm, e.g. calving or lambing camps, with especially vulnerable livestock. Hunt (1984), distinguished between deterrents, meant to discourage the presence of a predator in a certain area before conflict occurs, repellents which are usually human-triggered and meant to discourage specific behaviour of a predator in a certain area, and aversive conditioning which is meant to modify previously undesirable predator behaviour. Shivik et al. (2003) differentiated between primary repellents that depend on novelty (neophobia), irritation or pain to stop a predator from displaying certain behaviours and secondary repellents that depend on previously learned predator behaviour to prevent other behaviour. Ecologically, these methods resemble the kind of acoustic and visual signals used by territorial animals to warn away other species or con-specifics (Moehlman, 1987; Holmes and Holmes, 2006; Melzheimer et al., 2018). Within the optimal foraging framework ($H = C + P + MOC$), these methods attempt to raise the predation risk (P) for the predator when hunting livestock (or coming close to any areas with livestock). The short duration of effectiveness of most of these methods, can therefore be explained by the fact that unlike actual territorial marking, indicating recent activity by an animal, they are seldom backed up by any real threat of aggression. As soon as an animal realises this, the threat (deterrent) no longer works. Most dominant carnivores also do not use all habitat types equally, meaning that there are usually "sanctuaries" for weaker competitors within the territorial system (Flagel et al., 2017). Therefore, some natural areas without livestock should be left

as "sanctuaries" without deterrents to which predators can flee — otherwise they will be forced to habituate to the deterrents if there is no escape. By combining this method with rotational grazing, these "predator sanctuaries" can be in the camps without vulnerable livestock. Linnell et al. (1996) discuss some of the visual and acoustic repellents as well as their effectiveness for keeping coyotes away from sheep. It included a gas "exploder" used at night and three different strobe light and siren devices. Known as RAG (radio activated guard) devices in the USA, some units only activate when collared predators approach (Stone et al., 2008). These units cost about US$ 3 000 per unit (Shivik, 2006). In Southern Africa the "Jakkalsjaer" is the most commonly available model (`http://www.pmfsa.co.za/home/ask-our-expert/itemlist/category/10-pmf`, last accessed 23 March 2017) which also includes the sound of a radio station in addition to sirens and lights. The cost (in 2008) was between N$ 1 550.00 and N$ 1 950.00 per unit. A variation of this method has been used around Cape Town recently to keep baboons away from residential areas and to prevent conflict with people. A simple timed propane cannon has been used in Namaqualand, in the Northern Cape, South Africa to scare baboons away from a kraal with kids (personal observation). Some of these electronic static "repellents" do not primarily aim to keep predators away from an area, but instead by using various combinations of light, sound and even smell at unpredictable times, it rather aims to disrupt the hunting behaviour of the predators (e.g. the "Skaapwagter" device – Ernst van Zyl 2019, personal communication). In the USA fladry, in which flags hang from ropes stretched a short distance above the ground and costing about US$ 781.00 per km (Shivik, 2006), is frequently used as a mobile repellent against wolves (Paige, 2015; Stone et al., 2017). When combined with an electrical line (turbofladry), fladry costs about US$ 1 328 per km (Shivik, 2006). Miller et al. (2016b) found that for deterrents there appears to be a trade-off, and the tools "with the greatest effectiveness often lasted for only a few hours, weeks, or months" and their effectiveness varied between 0% and 100%. It is not a permanent solution on its own and more research is needed to compare the effectiveness of the individual methods on various Namibian predators, but for big cats (*Panthera* spp.) it was found to have a success ratio of 0.33 (Krafte Holland et al., 2018) (used here as the prior success probability).

- Advantages: These repellents are usually easy to set up, not labour intensive and relatively cheap to maintain (Shivik, 2004). While seldom 100% effective for long, it has been reported to cause a 60% reduction in livestock losses over one season (Linnell et al., 1996). "Lion lights" in Kenya has had a 100% effectiveness in keeping lions from killing cattle in an enclosure at night (Howley, 2013) (so far) by mimicking people walking around with flash-lights at night. Stone et al. (2008) mentioned that when fladry is combined with (temporary) electric fencing it worked well, repelling wolves even when the fence was not electrified temporarily. This combination, known as turbofladry, might work well for other predators as well, by combining a very visual signal with a real painful stimulus, and work better than either the electric fence or the fladry on its own (Stone et al., 2008).

- Disadvantages: The greatest disadvantage is that like most repellents, predators can get used to them, making them ineffectual, so it is almost always a temporary solution. In the cases mentioned by Linnell et al. (1996), they succeeded in keeping coyotes away for between one and 180 days. They concluded that these methods are not really effective for longer periods of time. The explosive repellents have been found to be effective against coyotes for an average of 31 days (Linnell et al., 1996). Fladry also worked only for up to 60 days against wolves (Musiani et al., 2003). Fladry, while cheaper than the other options, is also maintenance-intensive. Even turbofladry has been shown to only protect against wolves for up to 90 days (Du Plessis et al., 2018). Similarly, the Jakkalsjaer repellents only have a limited range (about 150 ha per unit) and while livestock in a camp that is close to it might be safe, others that are further away, might still be vulnerable (or multiple devices per camp will be needed). Moreover, sometimes the lights and sounds attract thieves who steal not only the livestock, but also the expensive equipment (Macaskill, 2013). As most predators are territorial, repellents occurring in most of their home range may facilitate eventual habituation rendering the repellent ineffective. Therefore, it is important that the predators have "sanctuaries" without any livestock and where they can survive on wild prey, for this method to be effective.

- Limitations: These kinds of methods are really only useful for keeping predators away from small pasture camps or kraals. So far, fladry alone is not known to work against any other mammals except wolves (Musiani et al., 2003; Treves et al., 2016). This method appears to work for longer when the repellent mimics a real danger (e.g. people walking with flash-lights at night). Because predators get used to the unusual lights or sounds, Linnell et al. (1996) recommend that it only be used for relatively short periods when livestock are especially vulnerable and Stone et al. (2017) recommended that different types of repellents be randomly combined and interchanged regularly to improve long-term effectiveness.

14. Projectile predator repellents: This method involves shooting predators with projectiles (e.g. rubber bullets or paint balls) to chase them away from livestock (Linnell et al., 1996). It is usually used in combination with other methods like calving or lambing camps to keep predators away from a specific area. It has been shown that predators do not necessarily associate the repellents with livestock or the area, but rather with the person(s) shooting at them (Shivik, 2004, 2006). It can also be used by herders and patrolling range riders to scare away predators from the vicinity of livestock herds. This is especially useful when combined with visual and acoustic repellents alternating to prevent predator habituation (Stone et al., 2008, 2017). This method resembles the anti-predator behaviour found in some primates, throwing rocks or branches at predators to chase them away. Within the optimal foraging framework ($H = C + P + MOC$), it could be considered as increasing the cost (C) of preying on livestock (when close to a human). In the Namibian situation, where the presence of humans can be enough to scare away the four predator species, this method probably does not have a much greater success rate than human presence alone. Its actual probability of success is thus expected to be lower than the prior (unknown) success probability of 0.5 used here.

- Advantages: For predators that have become very used to people and attack livestock close to farm homesteads, this can be an easy and non-lethal method to chase them away. It also combines quite naturally with the patrolling of areas where livestock are grazing far from homesteads. Projectile repellents can thus be used in both intensive and extensive farming systems (Stone et al., 2017).
- Disadvantages: Because predators might not associate the repellent with livestock or the area around the homestead, they might simply become better at avoiding people who might shoot at them while continuing to kill livestock.
- Limitations: This method is very unlikely to work for predators that hunt at night (most Southern African predators). Unless the farmer catches the predators in the act of hunting livestock, this method is in effect useless.

15. Predator repellent collars on livestock: These kinds of repellents are usually affixed to the livestock with a collar. They vary in kind from age-old low-tech bell collars to modern high-tech E-shepherd collars (`http://eshepherd.biz/faq.html`, last accessed 15 February 2017). They vary greatly in price and effectiveness, but they all contribute to the ability of predators to distinguish between natural prey and livestock (Linnell et al., 1999). Within the optimal foraging framework ($H = C + P + MOC$), this method raises the cost (C) of preying on livestock, by disrupting the hunting of livestock using neophobia (Shivik et al., 2003; Shivik, 2006). The E-shepherd is a system where the collar gives an ultrasound alarm and shine LED lights as soon as the sheep starts running (being chased by a predator) or leave a certain area. Another high-tech type collar is the GSM based "Veldwagter" system (described by Lötter, 2006). These collars send an SMS to the farmer as soon as any unusual movement by the livestock is detected. Except for the bell collars, repellent collars have seen limited use in Namibia, but the E-shepherd collars are currently being tested by the CCF (`https://cheetah.org/ccf-blog/conservation/e-shepherd-collar-update-kavango-west/`). In the absence of more data, this method is given a prior success probability of 0.5.

- Advantages: For the high-tech versions which only gives an alarm when the livestock are being attacked, it takes longer for the predators to get used to them. Preliminary testing of the E-Shepherd collars has shown them to be effective for at least a year (`https://cheetah.org/ccf-blog/conservation/e-shepherd-collar-update-kavango-west/`). This implies that while predators might eventually get used to them, they are effective for longer periods than the static repellents. They can also be used in combination with other methods (e.g. livestock guarding dogs with bell collars) to warn predators away. The "Veldwagter" system has also helped to prevent stock theft (Smuts, 2008), another major source of livestock losses in parts of Southern Africa. In the best case so far, it has reduced livestock losses from 320 sheep per year down to 12 sheep per year (Smuts, 2008). It appears to be a good choice for farms close to towns where cell phone coverage is better and the more likely causes of livestock loss are theft and domestic feral dogs. Bell collars work well together with herders and LGDs, both in warning the predator that livestock are not natural prey, and in helping the herder to locate and keep the herd together (Begg and Kushnir, 2013).

- Disadvantage: Once predators have habituated to the repellent, it can have exactly the opposite effect of calling the predator to dinner, rather than warning the predator off. And since predators learn from each other, this behaviour might spread quite quickly within the predator population once a single predator has figured out that the collars are actually harmless. The high-tech versions of the collars are quite expensive, with e.g. the E-shepherd collars costing about the same as a fully-grown sheep (N$ 1 200.00 in the beginning of 2016 - E-Shepherd, personal communication). It is recommended that at least one collar per every 10 sheep be used, meaning that depredation losses must be very high before it begins to make economic sense. The "Veldwagter" system (Smuts, 2008) uses the cell phone network to warn the farmer of any unusual movement of sheep. It costs almost N$ 5 000.00 per collar (in 2008) and therefore only one collar per herd is recommended, meaning that if the herd breaks up into smaller groups (like Dorper sheep and many European breeds), it may no longer be effective. Like any other collaring system, this method is quite labour-intensive, especially when collars are used on growing lambs or kids (the most vulnerable individuals), since the collars have to be checked and adjusted every couple of weeks. In extensive farming systems where livestock are not seen regularly, the collar might start to strangle growing sub-adults (Charles Schreuder 2011, personal communication).

- Limitations: All these methods will only be effective in the long run if there is enough wild prey available to the predators as an alternative food source (or the neighbour's livestock when he is doing less to protect his livestock). If there is no other option, hunger could force the predator to attack livestock in spite of the repellent collars (Shivik et al., 2003). The high cost versions are only an option in cases of very high livestock depredation rates, while the low cost versions are only effective (if at all) in combination with other methods. It is also not a good option for extensive farming systems which aim to be less labour intensive. Some systems (e.g. the Veldwagter collars) require cell phone coverage to work, making it impractical for large parts of Namibia.

16. Protective livestock collars: In contrast to poison collars (a reactive measure) where the livestock animal with the collar is sacrificed in the process, these collars attempt to prevent the predator from killing livestock. It also differs from the repellent types of collars by not primarily trying to change the predator behaviour, but trying to make actual attacks less effective, instead. Most predators attack the neck or throat to kill livestock. These collars not only aim to reduce the predator's kill success, but also hope to change predator behaviour to avoid livestock and revert to natural prey instead. Within the optimal foraging framework ($H = C + P + MOC$), it increases the cost (C) of preying on livestock. The most commonly available versions in Southern Africa, are the KingCollars (broad, 1 mm thick, semi-rigid, high-density polyethylene collars that are attached to the entire flock of livestock) and the Dead Stop Collars (consisting of a broad metal mesh, which is epoxy-coated), both described in Smuts (2008). The only published study in Southern Africa that could be found where this method was evaluated (and found effective), was McManus et al. (2014) with a 0.71 success

rate. A single study with modest sample size is not considered as sufficient evidence for success in the different circumstances of Namibia and this method is thus given a (50/50) prior probability of success of 0.5.

- Advantages: In a cost-benefit study in the Eastern Cape Karoo, McManus et al. (2014) showed that Dead Stop Collars decreased costs and livestock losses significantly compared to trapping and hunting and did not differ significantly from the use of livestock guarding animals (dogs and alpacas – methods 21 & 23 below).
- Disadvantages: Like all collars, the use of the Dead Stop Collars (Smuts, 2008) (cost: just over N$ 20.00/collar in 2008) or KingCollars (King, 2006) (cost: N$ 5.00-6.00 depending on size in 2008) can kill livestock by strangulation if not checked and adjusted regularly on growing livestock (Charles Schreuder 2011, personal communication). It is thus fairly labour intensive. To be truly effective, every single vulnerable individual needs to be collared, since the aim of this method is not primarily to change predator behaviour. However, some predators (especially jackals) may adapt killing strategies and avoid throat or nape bites. This reduces the effectiveness against certain species (Shivik, 2006; Du Plessis et al., 2018).
- Limitations: The greatest limitation of these collars is that applying them can be fairly labour-intensive. In general, they are more effective against felid predators than against canids or hyaenas which are more likely to attack the hindquarters of their prey (Estes, 1991). There needs to be enough alternative prey for predators, otherwise they can change their method of attacking to avoid the neck and throat of livestock (Smuts, 2008). This might be the explanation for the fact that in the McManus et al. (2014) study, the only two farmers who had a slight (though not statistically significant) increase in depredation in the second year of using non-lethal methods, were those who only used protective livestock collars.

17. Food aversion conditioning of predators: Conditioned taste aversion (CTA) was already discovered as a unique kind of learning by Garcia et al. (1955) and first demonstrated as a possible method to decrease livestock depredation by Gustavson et al. (1974). Like the static or mobile repellent methods, the aim is to change predator behaviour (Linnell et al., 1996; Treves and Karanth, 2003; Shivik, 2004). Within the optimal foraging framework ($H = C + P + MOC$), this method depends less on increasing the energy costs of killing livestock (the right-hand side of the equation). Instead it lowers the harvest rate (H = energy gained by eating livestock) by forcing the predator to vomit up treated livestock meat (and thus losing any potential energy gains – see text box below on *The principles behind CTA*). The most common chemical used for this purpose is lithium chloride (LiCl), but other emetic compounds used include cupric sulphate (CuSO4), emetine hydrochloride (EHCl), alpha-naphthyl-thiourea (ANTU), anthelmintic thiabendazole (TBZ), and Ziram (Gustavson et al., 1974; Linnell et al., 1996; Dingfelder, 2010; Baker et al., 2007). It is more likely to work with predators that also scavenge, where it can be combined with treated baits. Thus, to have any chance of success, untreated carcasses have to be removed from the veld (sanitation – Du Plessis et al., 2018). Preventing carnivores from eating livestock carcasses decreases the probability that they will learn to consider livestock as a food source (Ellins, 1985). It can also help to prevent predator numbers from growing, by not providing them with extra free food (Linnell et al., 1996). Lamb tails when they are removed, or even the afterbirth of livestock, should be removed from the veld and burnt, for the same reasons (Daly et al., 2006). It is a good idea to have a predator-proof fenced carcass pit where any livestock carcasses or remains are burnt (Stone et al., 2008). This is thus the opposite approach to supplemental feeding of predators (also mentioned by Linnell et al., 1996; Du Plessis et al., 2018). The precise dosages and a good explanation of how it should be applied is given in Forthman (2000) and a comprehensive case for why it is unlikely to work in Europe, is given by Linnell (2000). Owing to the lack of field trials, this method is given a prior success probability of 0.5.

The principles behind CTA — CTA is based on the same principle exploited by organisms using Batesian mimicry, where even a single event of a predator eating some prey causing them to be sick, can cause a prolonged aversion to similar prey animals (Forthman, 2000; Dingfelder, 2010). This method has been less successful in field trials (Griffith et al., 1978; Linnell, 2000), mostly because it depends on the predators not tasting the chemical (LiCl) and thus not being able to differentiate it from the known taste of livestock meat (Forthman Quick et al., 1985; Forthman, 2000). The idea is that the dosage will be high enough that eating the meat will make the predator sick, but will not taste any different, so that the predator will associate the sickness with the taste of livestock species (and not the chemical). If the predator can taste the chemical, it will only avoid livestock carcasses with the same strange taste and may start killing living livestock instead (and the method becomes counter-productive). Because precise dosages, well distributed through the livestock carcasses are required (Forthman, 2000), farmers do not always implement it correctly, resulting in failure (Ellins, 1985). Too often the thinking is one of "rather too much than too little" when the basic principle behind the method is not understood correctly. One conservation application not considered here, is the creation of conditioned taste aversion to livestock in captive carnivores of threatened species, before releasing them into the wild (Dingfelder, 2010).

- Advantages: This method is inexpensive. In contrast to poisons, it is safe for humans and non-lethal to predators, scavengers and secondary consumers (little negative environmental impact). As long as the territorial predators with a developed taste aversion stay in their territories, its effect is relatively long-lasting. It can easily be combined with most husbandry methods. Because it encourages natural predator behaviour by teaching predators not to consider livestock as natural prey, it is ecologically sustainable. Trained territorial predators "protect" livestock from other predators, so once taste aversion has been established, it is not very labour-intensive to maintain (Forthman, 2000).

- Disadvantages: The taste is specific to a specific livestock type. In other words, if predators develop a taste aversion to sheep, they might still kill cattle and goats. The greatest disadvantage is probably the logistical issues (Ellins, 1985; Linnell, 2000). Finding all or most livestock carcasses in order to treat them or enough bait so that all residential predators develop an aversion to the taste of all livestock species, is difficult, and may be impossible. As a method, it is highly sensitive to methodological variation, depending on relatively precise chemical concentrations (enough to make the predator sick, but not enough to taste strange – Ellins, 1985). Moreover, it does not work overnight, so farmers might lose a number of livestock before predators start to avoid killing livestock. Misapplication is not neutral in the sense that it can actually increase depredation losses if done wrongly, e.g. if the predator learns to taste the chemical and start killing livestock (which will always be untreated) while avoiding treated carcasses. If the carcasses of livestock that died from natural causes are the only food available, getting rid of it might encourage predators to kill livestock instead for food. Finding and getting rid of all livestock carcasses on a large farm may be almost impossible and even on smaller farms with many areas impassable by vehicles, it may be very difficult to find all livestock carcasses (Shivik, 2004). Because it depends on resident predators learning to avoid killing livestock, it is incompatible with any method that involves predator removal (Forthman, 2000).

- Limitations: If the findings for coyotes as reported by Jaeger (2004) are also true for black-backed jackals, and alpha breeding pairs are the real killers of livestock while the betas and transients are mostly scavengers, this method will not work, since the alpha livestock killers will not eat treated carcasses in order to develop a taste aversion. It will obviously be useless against predators like cheetahs that do not normally scavenge and are thus unlikely to ever take any bait (Lindeque et al., 1998; Van der Merwe et al., 2016). A taste aversion to livestock carcasses does not necessarily result in predators not killing livestock (Linnell, 2000; Shivik et al., 2003). Baker et al. (2007) showed that the same chemical has different effects on different predator species. Since eating any untreated livestock meat will cause any previous conditioned

aversion to stop, the only situation where this method is an option is where it can be ensured that all livestock carcasses on the farm have been treated.

18. Chemical predator repellents: Various chemical repellents have been tested against predators. In contrast to conditioned taste aversion (CTA), it does not work by causing nausea and vomiting in predators as a means of teaching them to avoid livestock. Instead, strong-tasting chemicals like capsaicin (the active ingredient of red peppers), undecenovanillylamide (nor-capsaicin), cinnamaldehyde or various commercial products are put on foods to stop animals from eating it. It has worked with vegetative foods (Linnell et al., 1996), but little success has been demonstrated against predators except as a repellent when threatening humans (Hunt, 1984). However, the newest version of the static South African "Skaapwagter" system, also includes chemical repellents, in addition to lights and sounds (`http://www.skaapwagters.co.za/index.php/skaapwagter-3/`), adding smell to acoustic and visual "territorial markings". In Namibia it has been used by putting bad-smelling sulphurous compounds on sheep to repel jackals (Dawie Minnaar 2017, personal communication). From an ecological viewpoint, it can be considered as mimicking animals like zorillas (*Ictonyx striatus*) or bad-tasting insects to repel predators. Within the optimal foraging framework ($H = C + P + MOC$), it attempts to raise the cost (C) of predating on livestock. This differs from CTA (method 17 above), where it is poisonous prey that is being mimicked. There has been no formal evaluation and comparison of its success in preventing livestock depredation, so it is assigned a prior p(success) = 0.5.

- Advantages: Unlike CTA that takes a while to have any effect, if it works as a repellent, it would stop predators from killing livestock immediately when implemented.
- Disadvantages: While some success has been shown against some predators, other predators have changed their methods of attacking, similar to what happens against protective livestock collars (Linnell et al., 1996). This method does not work for all or even most predators. Putting repellents on all vulnerable livestock is labour-intensive. One of the greatest drawbacks of this method is that it is not predator-selective, but also affects livestock and humans (Shivik, 2004).
- Limitations: If used on the hindquarters of livestock together with livestock protective collars, this method has some potential, but it has not been demonstrated as effective in lowering livestock depredation, yet. When used in collars, it has been demonstrated as ineffective and teaching the predators to attack the hindquarters instead of the neck (Burns and Mason, 1996, quoted in Shivik, 2004). In addition, the fact that it is labour-intensive makes it unsuitable for extensive types of farming.

19. Natural repellents: Also known as "bioboundaries" (Jackson et al., 2012) or "bio-fences" (Du Plessis et al., 2018), this method depends on the ecological fact that most predators mark their territories to warn away other predators, both con-specifics and competing predator species. Within the optimal foraging framework ($H = C+P+MOC$), it increases the predation risk (P) to predators when hunting livestock (from cons-specifics or superior interference competitors). Jackals and caracals are known to kill each other's young (Bothma, 2012; Du Toit, 2013) and probably show habitat partitioning and niche contraction when sharing an area. Leopards kill other predators, including cheetahs, caracals and jackals and these predators might want to avoid leopards or experience a decrease in their numbers when there are leopards around (Ritchie and Johnson, 2009). Flagel et al. (2017) have shown that the avoidance behaviour of mesopredators have a stronger effect on their distribution (and resulting densities) than direct killing by top predators. The potential importance of such predator interactions is illustrated by Bothma (2012), "The average carnivore in Africa is prone to predation and ecological influence from 15 other types of carnivore at some stage of its life". Also, on an Eastern Cape farm where jackals used to be a major cause of livestock loss, the farm owners have claimed that caracals have been used successfully for years to decrease these losses (Holmes and Holmes, 2006, `http://www.farmersweekly.co.za/article.aspx?id=286`; Du Toit (2013), `http://karoospace.co.za/can-caracals-save-sheep` last

accessed 15 February 2017). In the latter case, after initially releasing some caracals on the land, they also started to put caracal scat around kraals and lambing camps to keep jackals away from the sheep. This method has been used successfully to limit the movement of African wild dogs (*Lycaon pictus*) (Jackson et al., 2012), but not for any of the other predator species in Namibia and is thus given a prior success probability of 0.5.

- Advantages: Unlike many human-made repellents that depend on the wariness of predators towards the unknown, because these kind of repellents are backed by the actual threat of inter-specific violent interactions (Ritchie and Johnson, 2009), they remain effective for periods of up to 10 years so far (Marion Holmes 2015, personal communication).
- Disadvantages: Because the method works on the principle of establishing a fake predator territory around the area where the livestock will be, it can only be used to protect a relatively small area. Finding enough faeces to implement this method might be impossible without captive predators to provide scat (e.g. wildlife rehabilitation centres). Because the scat is strewn twice weekly, it can be quite labour intensive. Too little research has been done on the interactions between all Southern African predators to know when and how effective this method will be in different situations (Caro, 1987; Cozzi et al., 2012; Durant, 1998; Hayward and Kerley, 2008; Jackson et al., 2012; Kamler et al., 2007, 2012b, 2013; Mills and Mills, 2014; Ritchie and Johnson, 2009). When there is no actual threat of inter-specific (or intra-specific) conflict between predators (e.g. when the dominant predator species is unknown in the area, or there are no sounds with the scat and urine in vocal species), bio-fences alone are sometimes ineffective.
- Limitations: This method is not compatible with lethal methods or other methods that disturb the territorial distribution of predators on the farm.

20. Increasing natural prey: A study in the 1990's by Marker-Kraus et al. (1996) in North-Central Namibia, found that the only factor to make a significant difference in the levels of livestock depredation, was the number of wild prey on the farm (see also Avenant and Du Plessis, 2008). For many of the other methods which depend on stable territories and predator populations reaching the equilibrium density to work, an alternative source of prey other than livestock, is required (Shivik et al., 2003; Soofi et al., 2019). Cheetahs (Marker et al., 2003d) and leopards (Mizuntani, 1999) take livestock in lower numbers than would be expected from livestock stocking rates if they simply took prey according to availability. Khorozyan et al. (2015) showed that specific densities of wildlife exist, below which predators will switch from wild prey to livestock. Similarly, Lindsey et al. (2013b) showed that livestock losses in Namibian conservancies without game fences and more wildlife were lower than where landowners farmed with livestock only. Ecologically it uses the fact that predators are more likely to hunt and kill familiar (natural) prey and also the concept of prey dilution (the more choice of prey there is, the smaller chance for any specific prey animal to be killed – King et al., 2012). Within the optimal foraging framework ($H = C + P + MOC$), it raises the missed opportunity costs (MOC) of hunting livestock instead of wild prey (Brown, 1988; Haswell et al., 2019). Using the number of studies in which it were successful (n = 7) from Miller et al. (2016b), it should have a prior success probability of 0.71.

- Advantages: Especially in Northern Namibia, but also in other parts of Southern Africa where relatively high densities of free-roaming prey species still occur, this method costs almost nothing except hunting less natural prey and limiting hunting dogs on farms (McGranahan, 2008). It combines well with most other methods and increases the biodiversity and thus ecological stability of the land.
- Disadvantages: Where there are no or few game species currently, it can be expensive to reintroduce wildlife on farms. Even where some original wildlife can still be found on a farm, it can take a while before game numbers reach high enough numbers to make a difference in livestock losses (Sinclair et al., 2003; Khorozyan et al., 2015). In many cases an increase in natural prey numbers may not be sufficient to decrease livestock losses, especially if some predators have learned to preferentially predate on livestock over the decades without wild prey (Drouilly et al., 2018).

- Limitations: This method is not really an option in transformed landscapes where little natural habitat remains for wildlife. In some areas where the total sustainable stocking rate for all herbivores is below the critical density found by Khorozyan et al. (2015), this method might also not be an option.

21. Livestock guarding dogs (LGDs): Livestock guarding dogs are already mentioned in one of the oldest books in the Bible (Job 30:1). They are bred to be trustworthy (will not harm the flock), attentive (stays with the flock), and protective (barks and defends the flock). Large enough to protect against most Namibian predator species, the Anatolian/Kangal is one of the best adapted breeds to the Namibian climate and long distances, although Rhodesian Ridgebacks have also been used with some success (Linnell et al., 1996; Andelt, 2004; Marker et al., 2005a,c; Urbigkit and Urbigkit, 2010; Van Bommel, 2010; Van Bommel and Johnson, 2012). While there are many other breeds of livestock guarding dogs that are protective, attentive and trustworthy, most of them were bred for different climates and are not a good fit for the heat and long daily livestock movements of Namibia (Marker et al., 2005c; Urbigkit and Urbigkit, 2010). LGDs can be used in different ways. The traditional way and the most effective, is the use of dogs together with herders and kraaling at night (Hansen, 2005). But they have also been used in some ways which are better suited to extensive farming conditions: the cheapest method (and the second most effective) is having LGDs alone in camps with livestock (Hansen, 2005). Two other options are having LGDs alone with livestock in open range (Van Bommel and Johnson, 2012) or patrolling the livestock together with a "range inspector" (Hansen, 2005). An evaluation of various livestock guarding dogs in Slovakia, showed that the success of dogs depended more on the attitude and diligence of the herders than on the breed of dog (Rigg et al., 2011). Ecologically, this method depends on the natural intra-guild, inter-specific aggression between predator species (similar to method 7 above), as well as the defensive pack behaviour of wolves (see text box *LGDs vs. herding dogs* below). Therefore, within the optimal foraging framework ($H = C + P + MOC$) it raises the predation risk (P) to predators attacking livestock. Where LGDs are used for patrolling (rather than guarding specific sheep flocks), the idea is to create a "landscape of fear" through the dogs scentmarking with predators avoiding areas of high dog activity (which should ideally correspond with areas of high livestock densities). In Namibia, LGDs were shown to reduce or eliminate livestock losses on 91% of the farms where they had been placed and 73% of farmers considered them as economically beneficial (Marker et al., 2005a; Potgieter et al., 2013). They are thus given a prior probability of success of 0.91.

LGDs vs. herding dogs — Livestock guarding dogs should be clearly distinguished from herding dogs (Schumann, 2004). Whereas herding dogs use behaviour similar to that of a predator to scare and bunch livestock together and move them in the direction they want, livestock guarding dogs behave as one of the flock. Livestock guarding dogs use their natural pack instinct and bond with the livestock herd in which they are raised, guarding them as fellow pack members, and in turn are accepted as part of the herd by the livestock. Herding dogs use their hunting instincts to dominate and control livestock through fear (livestock perceive them as predators) and they are usually taught and handled by a human, while LGDs (once trained) will work independently.

- Advantages: Well-trained livestock guarding dogs are one the best protectors of livestock against predators (Marker et al., 2005a). Compared to herders, they are also relatively cheap (the cost of the puppy in addition to about N$ 500.00/month for food and veterinary costs = cost of five to six small livestock units saved per year). Once the dog and livestock have bonded, self-feeders can be used to feed them and a minimum of effort and labour is required for maintaining the protection of livestock (Ivan van Niekerk 2016, personal communication).

- Disadvantages: It takes about a year for an Anatolian shepherd dog to be trained and develop enough confidence to chase away a large predator like a leopard or cheetah. Therefore, in the first year of its life, a livestock guarding dog needs to be accompanied by a herder in order to teach it not to chase game,

play with the young livestock or learn other problem behaviours (Rigg et al., 2011; Potgieter et al., 2013). Of course, the herd itself also needs to be protected until the LGD can take its place. For this reason, the first year of the dog's training can be quite expensive. On average an LGD has a working lifetime of about four to five years in Namibia and sometimes a dog can die from a snake bite within its first year of working (Marker et al., 2005b). However, since snakebites have been the major cause of death, snake aversion training might extend the average working life of LGDs (Rust et al., 2013). This means that a new dog needs to be trained on average every fifth year just to protect the same number of livestock. If a LGD is not trained properly, it can show all kinds of behavioural problems, including at worst, starting to kill livestock itself. Therefore the herder doing the training needs both to know how to train the dog and be trustworthy (Schumann, 2004; Potgieter, 2011). Another major drawback to using LGDs, is that the demand is much higher than the supply, so a farmer might not always be able get a dog when required (Schumann, 2004; L. Marker 2013, personal communication). After training, regular care of the dog is still required which makes the utilization of remote areas by livestock problematic, although self-feeders can ease the problem (Hansen, 2005). Sometimes the most vulnerable livestock is no longer in the herd to which the LGD has bonded and a whole process is involved to introduce the dog to its new herd, leading to more management headaches compared to other methods (personal observation; Ivan van Niekerk 2016, personal communication). An important potential disadvantage of this method is that LGDs often kill predators, which can cause the same issues with sustainability as other lethal methods – however, they often only attack predators which had come close enough to livestock herds to be a threat, making it still one of the most selective of the preventative lethal alternatives (Urbigkit and Urbigkit, 2010; Potgieter et al., 2016).

- Limitations: One LGD can protect no more than 200 small livestock effectively in the extensive farming systems used in Namibia (Hansen, 2005; Van Bommel and Johnson, 2012). This will also depend on the vegetation, terrain and the predator threat itself – where livestock are kept together in large herds (e.g. using electrified fladry) or are kept in small camps, a LGD could potentially protect more animals (Stone et al., 2017). Because the dogs bond with a specific herd, when two herds each with their own LGD are put into adjacent camps, the dogs might attack each other, attack the livestock of the neighbouring camp or even start to spend time with each other and ignore the livestock (personal observation; Ivan van Niekerk 2016, personal communication). LGDs have not been tested with cattle in Namibia, but it will probably be more difficult to bond the dog with cattle, since most cows will tend to be quite aggressive in protecting their calves and can easily kill the puppy (Marker et al., 2005c). However, they have been used successfully with cattle in South Africa (Rust et al., 2013). Not only does the dog need to bond with the livestock herd, but the herd also needs to bond with the dog. When this does not happen properly, some sheep will run away from the dog and can even break legs or kill themselves while running away from the dog, especially in mountainous terrain (personal observation). The dog is also limited to work only with the kind of livestock to which it is bonded, meaning that a dog that was raised with goats for example, cannot be used with sheep.

22. Guard donkeys: Donkeys (*Equus africanus asinus*) are one of the undervalued protection animals against predators. While they cannot protect against some of the same size predators that LGDs can, they do not require special feeding, as they graze with the other livestock and are much easier and cheaper to acquire (Linnell et al., 1996). The ecological principle behind this method is to use the protective instinct of a donkey for her foal to also protect other livestock. Within the optimal foraging framework ($H = C + P + MOC$), it increases the cost (C) of preying on livestock. Normally a female donkey that has recently foaled is put with her foal among livestock about to calf. Since she would not like to be alone, she tends to stay with the livestock and chase away any predators that threaten the young. To be successfully implemented, there are a number of guidelines to keep in mind: 1) use only a mare or gelding since donkey stallions can be aggressive to livestock; 2) have the donkey bond with the herd it is to protect for 4-6 weeks; 3) use only one donkey per herd, (or a

single jenny with a foal), otherwise a group of donkeys will rather graze on their own; 4) test a new donkey's response to predators (use a dog in a kraal with the donkey and do not use donkeys that react passively); and 5) use donkeys in small open pastures with a moderate-size herd (Marker, 2000b; Andelt, 2004). Van Eeden et al. (2018) pooled guarding animals in general (mostly LGDs) and found a 87% success rate. Linnell et al. (1996) reported one survey of 17 farmers showing an overall 59% effectiveness for donkeys (although another Texas survey had lower success for some predators) - thus a prior success probability of 0.59 is given to guardian donkeys.

- Advantages: Using donkeys is faster to implement than livestock guarding dogs (LGDs), with no need for training. They are cheap to buy and maintain, since they live on the same grazing as the other livestock. They have a longer life expectancy than LGDs (10-20 years), and can be used in combination with many other methods (Linnell et al., 1996).
- Disadvantages: Donkeys are reportedly less effective in preventing livestock losses than livestock guarding dogs (Linnell et al., 1996).
- Limitations: While donkeys are fairly effective against cheetahs and smaller predators, leopards and larger predators are not discouraged by donkeys and would sometimes go past calves in order to kill and eat a donkey foal (Francois Kok 2014, personal communication). For best results, the mare should have a foal a few weeks before the calving/lambing seasons of the livestock to be protected. This necessitates the use of seasonal breeding (see method 27 on page 72) to ensure that the young are not born throughout the year so that the donkey's breeding can be synchronised to that of the livestock.

23. Guard Llamas/Alpacas: Both llamas (*Lama glama*) and alpacas (*Vicugna pacos*) are natives of South America and can used as guarding animals, but reports from Israel imply that llamas are more aggressive towards predators than alpacas (Linnell et al., 1996). Ecologically, these animals are basically defending their territories when they attack predators. Within the optimal foraging framework ($H = C + P + MOC$), the cost (C) of preying on livestock is increased. While no publications could be found where this had been done, it is possible that closely related camels (*Camelus dromedarius*), which have been in Namibia for a long time, could be used for a similar purpose (`https://web.archive.org/web/20101119115428/http://abbott-infotech.co.za/kalahari-use-of-camels-by-south-african-police.html`, last accessed 1 December 2018; Du Plessis et al., 2018). While no record of alpacas being used in Namibia could be found, McManus et al. (2014) reported one case of successful use in the Eastern Cape (1 farm). In one USA study, 88% of producers using llamas or alpacas reported satisfaction (with average livestock losses dropping from 11% to 7%) (Linnell et al., 1996). Because of so few peer-reviewed reports, a prior success probability for Namibia of 0.5 is given to this method.

- Advantages: Like donkeys, they do not need to be trained or fed specifically, but graze with other livestock. Unlike donkeys, they appear to form strong bonds with the livestock they protect, especially being protective of small lambs.
- Disadvantages: Llamas are even more scarce and expensive to buy than livestock guarding dogs in Southern Africa (NAMMIC, 2011; McManus et al., 2014; MAWF, 2017; Du Plessis et al., 2018). How well they are adapted to savannah bush vegetation is not known yet.
- Limitations: Even though they are aggressive towards smaller predators, in South America they are known to be killed by pumas and flee from them, so their use is probably restricted to protecting against smaller predators like caracals and jackals (Linnell et al., 1996; Bonacic et al., 2016). Because this method depends on the territorial behaviour of the guarding animals, it is likely to work better in smaller camps where livestock will not leave these territories (Linnell et al., 1996; Andelt, 2004).

24. Protection cattle: Cattle can be used to protect small livestock from predators by the different livestock species bonding with each other (mixed herds/flerds – Du Plessis et al., 2018). Sheep are bonded with cattle by putting lambs (45-90 days of age), with the cattle in a kraal for 60 days (Linnell et al., 1996). Goats appear not to bond directly with the cattle, but bond with sheep who in turn are bonded with cattle (Linnell et al., 1996; Andelt, 2004). This method does not depend on the cattle actively protecting the small livestock, but rather on the sheep and goats remaining close to the bonded cattle and benefiting from the cattle protecting themselves and their young (similar to donkeys, mentioned above as method 22 – Linnell et al., 1996). This method also corresponds to the natural ecological behavioural adaptation of large herds consisting of mixed species (see method 20 above about prey dilution). In nature this happens typically to increase the total vigilance of the herd, without the individual animal having to spend so much time being on the lookout for predators (Howery and DeLiberto, 2004; King et al., 2012). Within the optimal foraging framework ($H = C + P + MOC$), the aim is to increase the cost (C) of attacking livestock instead of other prey options. Additionally, it depends on the natural protective behaviour of many cattle breeds and herds of wild ungulates like buffalo (*Syncerus caffer*). This method has not been sufficiently evaluated and compared in the literature and is therefore given a prior success probability of 0.5.

- Advantages: For predators that can be chased away by cattle, this method is apparently very effective (Linnell et al., 1996; Du Plessis et al., 2018). It can also be used very effectively together with other livestock management techniques like holistic (high-intensity, short duration) grazing and result in better veld utilization by encouraging grazing succession (Vesey-FitzGerald, 1960; Gwynne and Bell, 1968). The reduction in losses and time spent searching for sheep, probably make up for the relatively high costs of the bonding period (Linnell et al., 1996).
- Disadvantages: The greatest disadvantage to this method is the time and money spent during the period of bonding (US$ 0.51/lamb/day according to Linnell et al., 1996). In Namibia, where feed is relatively expensive, this cost might be even higher.
- Limitations: This method is unlikely to work against some cattle-killing predators, although not de-horning cattle is likely to have an impact even on losses due to larger predators (see method 30 on page 75). Some veld types are not suitable for cattle (Du Plessis et al., 2018; also see method 28, "Change breed or species of livestock", below on page 73).

25. Avoiding predation hotspots: Predators often prefer to hunt in certain types of habitat within their home ranges (Adabe et al., 2014). This terrain in which predators prefers to hunt is determined to a large extent by their way of hunting, e.g. an ambush predator would prefer areas with enough cover for hiding and stalking while both coursing and speed hunters can be expected to prefer relatively flat areas rather than rocky or mountainous terrain for hunting. It is known that leopards avoid open areas (Martins, 2010), and this has been used to decrease leopard predation on calves by keeping cows with calves away from bushy, mountainous parts of a farm (while utilizing those areas by larger oxen and cows with horns; e.g. see Landbouweekblad, 24 June 2011; Minnie et al., 2015). Balme et al. (2007) showed that leopards actually preferred to hunt in areas of intermediate cover, but also that the habitat type was more important than the prey species abundance. Eaton (1970) showed that for cheetahs in a heterogeneous environment, the hunting technique and preferred hunting areas and prey differed between different cheetah groups. However, in general, it does appear as if bush edges next to open areas are the preferred cheetah hunting habitat (Mills et al., 2004). Spatial models to identify livestock depredation hotspots have been used for tigers (Miller et al., 2015, 2016a), leopards (Miller et al., 2016a) and jaguars (Carvalho et al., 2015). Females normally don't move very far once they have small young, thus creating a temporary predation hotspot around their denning sites, to be avoided by vulnerable livestock. Within the optimal foraging framework ($H = C + P + MOC$), a predation "hotspot" (preferred hunting habitat of a certain predator species) can be considered as patches where either high prey harvest rates (H) is possible, or the energy costs (C) of hunting is low (Brown, 1988). By excluding livestock from these areas, the harvest

rates (H) of hunting livestock is decreased and/or the energy costs (C) of hunting livestock is increased. For the smaller mesopredators, like jackals and caracals, little is known about their habitat preferences in general and their preferred hunting habitat in particular. Identifying conflict hotspots was one of the most common aspects in human-predator conflict research (22 out of the 62 articles found in ISI Web of Science database focused on it), but it was not generally applied by comparing livestock losses when hotspots were avoided to a control, with little indication of how much difference applying it would actually make to livestock losses (Miller, 2015). It was thus too seldom applied as a conflict mitigation method to evaluate it (e.g. it is mentioned by Linnell et al., 1996 and Miller et al., 2016b, but not evaluated in terms of decreasing livestock losses, while Treves et al., 2016; Du Plessis et al., 2018 and Van Eeden et al., 2018 do not even mention it). For this reason it is given a prior probability of 0.5 for success in Namibia.

- Advantages: Where livestock are kept in camps, like almost all commercial farms in Namibia, it is a very easy method to implement, costing almost nothing except planning. Because of a minimum interference with the ecology and behaviour of predators, this method "works with nature" and is one of the more sustainable methods. It can also be combined with most other methods.
- Disadvantages: This method is dependent on the veld of a farm providing heterogeneous habitat with different hunting preferences by predators. If the whole farm consists of "difficult terrain", as sometimes happens, it cannot be implemented. This might be one reason for the often reported differences in livestock depredation rates, being very high on some farms while many of the other farms in a district have negligible losses (e.g. Marker et al., 2003c; Van Niekerk, 2010; Rigg et al., 2011; Minnie et al., 2015).
- Limitations: This method is only likely to work if there are disproportionate livestock losses in some areas of the farm. In general, more research about the spatial behaviour of economically significant predators is needed for this option to work well. Miller et al. (2015) has shown that for tigers in India, the habitat scale needs to be known (and mapped) at a 20m scale or finer for accurately identifying hunting hotspots (modelling livestock depredation risk) in the habitat.

26. GPS collar of predators: This is actually similar to method 25 above (avoiding depredation hotspots), but instead of spatial avoidance only, it involves a dynamic, adaptive approach. This method has been used with great success in Namibia by some conservation NGO's (e.g. N/a'an ku sê). The predator is collared and released (with the farmer's permission) back onto his land. Farmers are e-mailed with a daily update on the movement of the collared predator. Farmers are also updated with the coordinates whenever there is cluster of GPS points indicating a possible kill, in order to check and confirm which prey species was killed. For territorial predators with large home ranges, like leopards, the farmer can then move their vulnerable livestock into protective kraals when it approaches their farm and let them out again when the predator leaves. Ecologically, this method largely depends on the territoriality and large territory sizes of many Namibian predators for its effectiveness. Within the optimal foraging framework ($H = C + P + MOC$), it functions by increasing the costs (C) of livestock predation temporarily. This is a fairly new method, and no mention of GPS collars applied as a conflict mitigation method could be found in the literature (but see text box below: *GPS shock collars*). Although this method looks promising, the lack of published studies currently requires a prior success probability of 0.5 for GPS collars.

GPS shock collars — A variation on this method is collars that give wolves an electric shock when coming close to a homestead (e.g. Stone et al., 2008). Unlike conditioned taste aversion (CTA, see method 17 on page 63) the avoidance behaviour change did not last long. A number of factors make this method impractical: 1) Unless a single animal is causing all the damage and it is being used as reactive method, this method cannot be used. 2) It can help to keep the predator away from a certain area, but will not prevent the predator from killing livestock elsewhere. All vulnerable livestock therefore need to be kept inside the "protected" area that will cause the collar to shock the predator. 3) The period of effectiveness will be restricted pretty much to the shock collar's battery life. 4) The difficulty in capturing the right individual causing the livestock depredation plus the collaring cost, limits it usefulness further. Shock collars will thus not be considered further here (but see Du Plessis et al., 2018 for more). It is important to note that creating trust between farmers and conservationists (who do the collaring and daily updates) is the primary contribution of GPS collaring of predators. Identifying predation hotspots, prey preferences, home range sizes and densities of predators and enabling farmers to use this information in their livestock management, are all secondary advantages of GPS collars.

- Advantages: Ecologically, this method has almost no impact. It bridges the trust gap between farmers and ecologists by showing farmers the movement of predators instead of telling them. Farmers can see that the predator actually prefer natural prey (if it is not a habitual livestock killer) and that the home range of the predator is larger than only his own farm. And even if it is a habitual livestock killer, the farmer can "teach" it that livestock are not available as prey species, by removing vulnerable livestock. However, this method can be quite effective with some farmers reporting a drop from 30 calves lost per year to only one (A. Weise 2017, personal communication). Where a predator has been confirmed as a habitual livestock killer with this method, it can be captured again and translocated (see method 39 below on page 84).

- Disadvantages: Current GPS collars are expensive, too expensive for most farmers to afford. Unless collars are provided by some conservation organisation, a farmer will need to lose at least six cattle a year to make it a worthwhile alternative. The actual collaring itself will also require a wildlife veterinarian and the predator needs to be captured first.

- Limitations: This method is unlikely to work well for predators that are not strictly territorial. It will probably also be prohibitively expensive for smaller predators with more than one territorial individuals on the same farm.

27. Seasonal avoidance of predation (seasonal breeding): While the previous two methods depend on spatial avoidance of predators, this method depends on temporal avoidance. Within the optimal foraging framework ($H = C + P + MOC$), it decreased the availability of livestock as prey (and thus harvest rate, H) during the periods of highest energy requirements of predators. By having specific calving/lambing seasons, the same strategy used by many natural prey species is mimicked. By not providing the predators with a ready source of food throughout the year, an increase in predator densities is avoided. Any of the other preventative methods could also be used for a short period in those seasons when livestock are most vulnerable (e.g. when there are young animals). Such seasonal changes will need to take into account both the seasonal behaviour of the predators (e.g. breeding seasons etc. Shivik, 2004) and the seasonal grazing requirements of the livestock. Linnell et al. (1996) summarize this method as using 1) seasonal changes in livestock husbandry practices, 2) seasonal changes in the life cycle of livestock (birth seasons), 3) seasonal changes in predator life cycles, 4) seasonal changes in the availability of alternative prey and 5) seasonal changes in predation levels. While mentioned relatively frequently in the literature (e.g. Marker-Kraus et al., 1996; Du Plessis et al., 2018), this method has not been evaluated and compared to other methods for effectiveness, and is assigned a prior success probability of 0.5.

- Advantages: If the birth season is synchronized with that of a preferred wild prey species, it can drastically reduce livestock losses compared to livestock giving birth throughout the year. If there is an abundance of young vulnerable natural prey, the lower comparative attractiveness of livestock (and higher risk from a predator's viewpoint), makes it less likely that livestock will be attacked. Additionally, by providing predators with an overabundance of prey all at once, the predation risk of each individual is lowered (dilution effect / "selfish herd theory" – King et al., 2012). Alternatively, if the predator species on a farm have a distinct breeding season (e.g. black-backed jackals in springtime) during which they would have utilized the birthing season of their natural prey in the wild to feed their young, avoiding this season for the birth of livestock can make a big difference on livestock depredation losses (Kamler et al., 2012a). Many livestock species have a natural breeding season in any case and manipulating this a bit is easy and cheap. This method can also be combined with many other methods and is almost required for some other methods to work.

- Disadvantages: There is a reason why natural prey species (those with seasonal breeding) have their young ones in a specific season. It is usually the season with the best veld condition (quantity and nutritional quality) for raising their young (Tainton, 1999). This is also true for livestock. So changing the birth season to avoid predators might result in productivity losses due to unsuitable veld conditions. Having distinct breeding seasons requires more direct management of the livestock and possibly higher labour costs. One of the greatest drawbacks is that it might result in lower birth percentages for livestock, especially in extensive farming systems where the males might simply not get to all the receptive females in time for those livestock species who are not naturally seasonal breeders, while year-round breeding increases the chance of higher birth percentages in extensive farming systems.

- Limitations: Where livestock depredation do not have seasonal peaks or does not happen primarily to feed young predators, this method is impractical. When aiming for a birth season to coincide with an increased availability of natural prey, this method is only practical where enough wild prey is available on the farm.

28. Changing the breed or species of livestock: Where predation levels are very high, the best option for the farmer might be to change to other breeds or species of livestock. Cattle are in general less susceptible to depredation than small livestock (Sinclair et al., 2003). In mountainous areas, goats can experience less depredation than sheep. Goats tend to scatter uphill when attacked by predators, leading to less surplus killing, unlike sheep that tend to mill around in one place (but sometimes defend their lambs better in the herd against smaller predators). Behavioural attributes of different breeds can influence depredation in two ways (Linnell et al., 1996): 1) through behavioural traits making other management methods easier (e.g. making herding easier) or 2) through specific anti-predator behaviour (e.g. aggressive attacks of predators). Simple size might put livestock out of the preferred prey size range of some predators (e.g. cattle are not generally vulnerable to caracals). Damara and Karakul sheep breeds are known to bunch together more in larger herds than breeds like Dorpers. The natural behaviour of sheep actually varies all the way from large grazing-while-moving herds, adapted better for lower grass cover, to more selective, more sedentary grazers in smaller groups, to sheep breeds that form territorial breeding pairs. Some cattle breeds of European origin (*Bos taurus taurus*) in South America developed anti-predator behaviour (Hoogesteijn and Hoogesteijn, 2010), while in North America the Brahman (*Bos taurus indicus)* and longhorn cattle are known for their protective behaviour (Stone et al., 2008). Cattle that know the kind of terrain on a farm (e.g. mountain-raised cattle) also tend to suffer less depredation losses than cattle from other areas. Generally, horned cattle breeds are likely to be able to better protect their calves than polled or short-horn breeds, but individual cows also differ in their protectiveness. In South America, water buffalo (*Bubalis bubalis*) has been shown as less vulnerable to predation than cattle (Hoogesteijn and Hoogesteijn, 2008). Angora goats are also more vulnerable to predation than Boer goats, possibly because of being smaller in size. There are unconfirmed claims by some farmers that by not weaning their female Damara lambs, it allows the mothers to teach their daughters how to protect their lambs and also help to establish

a herd hierarchy and anti-predator behaviour, leading to less depredations losses to jackals. Ecologically, this method depends on both the natural anti-predation and crowding instincts of livestock and the preferred prey size class and habitat of predators. Within the optimal foraging framework ($H = C + P + MOC$), it depends on the idea of raising the cost (C) of predating on livestock. While many articles mention livestock species or breeds as a method to prevent depredation, and anecdotal evidence would suggest that it can be very effective, no article comparing it to other methods in terms of effectiveness could be found, leaving it with a prior success probability of 0.5.

- Advantages: This method is simple and fairly easy to implement. If changing from small livestock to cattle-sized livestock in areas with small mesopredator species only, this method can be close to 100% effective (Linnell et al., 1996). However, in some cases it can have limited success in reducing livestock depredation unless combined with other methods. In very extensive farming systems, it might be the only option, combined with letting cattle grow horns (Hoogesteijn and Hoogesteijn, 2010).
- Disadvantages: The switch to other livestock species or breed can have negative economic consequences (e.g. beef is generally significantly cheaper than mutton and cattle have a much lower rate of increase than either sheep or goats) (Du Plessis et al., 2018). Because few farmers can afford to make such a switch quickly, any change has to be done slowly, leading to continued losses until the switch has been completed.
- Limitations: Dairy breeds are generally not well-adapted to the hot climate, thorny vegetation and long distances required to walk between grazing and drinking water typical of many parts of Namibia. Cattle are tall grass grazers, using their tongues to ingest grass, making a switch to cattle not an option in areas dominated by short grasses where tall grass grazers do not occur naturally. Additionally, for a stud farm where the stud might have built up over generations, switching to a different breed is both financially and emotionally not a realistic option (Harry Schneider-Waterberg 2013, personal communication).

29. Changing production livestock age: Some farmers in Namibia use a steer production farming system whereby they buy weaner calves and then graze the steers and heifers on natural grazing to gain weight in order to sell them at profit (NAMMIC, 2011). By weaning age, most calves are no longer vulnerable to many predators and depredation losses can be minimal in this kind of farming system (Hoogesteijn and Hoogesteijn, 2010). This method depends totally on the preferred prey size classes of the predator species commonly found on Namibian farmlands. Within the optimal foraging framework ($H = C + P + MOC$), it raises the cost (C) of preying on livestock. This method has not been widely published or evaluated and compared to other methods in terms of effectiveness, so while anecdotal evidence would suggest it is quite effective, it is given a prior success probability of 0.5.

- Advantages: Especially when there are few predators larger than cheetahs on the farm, this method can be very effective. It is fairly easy to implement, although it needs careful record keeping and management to succeed economically. One major advantage is that with this method it is easier to quickly adapt stocking rates according to the rainfall (i.e. buy fewer new cattle when the veld condition is becoming bad).
- Disadvantages: No breeding selection is possible and the farmer is largely stuck with whatever is available for sale at the time of buying, with little control over livestock quality. In years of good rainfall (especially after droughts), many farmers would rather keep their calves to increase their own livestock numbers and weaners become scarce and expensive.
- Limitations: Steer production farming is impossible without enough weaner calves being produced (and offered for sale) in other cow-and-calf production systems. It is thus not an option for all farmers. Hoogesteijn and Hoogesteijn (2010) proposed that it preferably be used in areas of high livestock losses.

30. Horned cattle: Many livestock breeds that are native to or adapted well to Africa, have horns that help them in protecting their young ones from predators (e.g. Afrikaner and Nguni cattle). Putting experienced bulls, horned oxen and old cows with young animals in a single herds in order to teach younger animals how to protect against predators is a simple way to allow livestock to use their natural instincts and behaviour for reducing depredation losses (Hoogesteijn and Hoogesteijn, 2010). In Namibia horned oxen and cows have traditionally been kept with calving herds for protection against predators (Marker-Kraus et al., 1996; Marker, 2002). Predators generally prefer to hunt prey with a low risk to benefit ratio (Brown, 1988; Carbone et al., 2007; Christiansen and Wroe, 2007; Haswell et al., 2019). Generally, in the same size class prey without horns would be less risky to kill than those with horns. This forms the basic ecological rationale behind this method. Within the optimal foraging framework ($H = C + P + MOC$), it increases the cost (C) of preying on livestock (more energy spent to avoid horns). However, since this method has not been compared and evaluated against other livestock protection methods, it is given a prior success probability of 0.5.

- Advantages: This is one of the easiest methods when farming with horned livestock breeds, since it actually involves less labour and thus labour costs than the alternative of de-horning. It is also very easy to implement in extensive farming systems. When combined with strong behavioural breeding selection for those cows who are able to protect their young from predators, this can be a very cost-effective method to reduce livestock depredation, with higher wean percentages compensating for all of the potential drawbacks. Together with choosing the right kind and breed of livestock, this might be one of the only methods available to very extensive farming systems (Hoogesteijn and Hoogesteijn, 2010).
- Disadvantages: Many farmers report that cattle with horns are more dangerous when handling them in the kraal or hurt each other when forced close together, e.g. in feeding lots or while being transported. Some feeding lots appear to discriminate against cattle with horns for this reason. Apparently, some farmers also prefer de-horned cattle for aesthetic reasons (Johann Britz 2014, personal communication).
- Limitations: This method may not be a good fit for those farmers whose primary market is feeding lots or who are farming with polled breeds for other reasons.

31. Grouping of livestock (holistic grazing with large herds): While holistic grazing (short duration, high intensity grazing) was primarily developed for veld (grazing) improvement (Bingham, 1997; Jones, 2000; Savory, 2013), it mimics another natural strategy used by many prey species to minimize depredation by crowding together for protection (Smuts, 1978). Instead of having many small herds of livestock distributed through the landscape, having a single large herd (or only a few large herds) means both the chances of a predator encounter is smaller and that the herd is better able to protect its vulnerable members (Linnell et al., 1996). A larger herd is also more likely to see a predator and warn each other (many eyes hypothesis – Howery and DeLiberto, 2004). Within the optimal foraging framework ($H = C + P + MOC$), this method raises the cost (C) of searching for livestock as prey as well as avoiding the anti-predator behaviour of large herds. While anecdotal evidence suggests that this method is effective in preventing livestock depredation, too little has been published on its use for this purpose and it is given a prior success probability of 0.5.

- Advantages: Holistic grazing systems can result in the long-term improvement of the range-land (mostly by encouraging nutrient recycling – Hobbs, 1996), and because it mimics natural herd-forming behaviour of wild herbivores, it also leads to a significant decrease in livestock depredation (Uwe Gressmann 2013, personal communication). This method can also be very easily combined with mixed herds for cattle to protect small livestock (see method 24 above on page 70).
- Disadvantages: Unless intensively managed adaptively, it can lead to veld degradation instead of improvement (Carter et al., 2014). In general, because livestock are forced not to graze selectively for more nutritious grasses, this method, while improving the veld in the long run, can cause lower production

by livestock in the short term. General livestock management (e.g. vaccinations or early detection of any health issues) can be more difficult with large herds compared to smaller herds in separate camps.

Rangeland impact of holistic grazing management in arid regions — Negative trampling and piosphere effects have been demonstrated in arid and semi-arid regions of the world (e.g. Von Wehrden et al., 2012). The benefit of holistic grazing over other grazing management systems like normal rotational grazing or adaptive grazing regimes has not been demonstrated (Carter et al., 2014). E.g. selective grazing early in the growing season for a short period ($<$ 2 weeks) to stimulate palatable grasses, followed by long periods of grazing all grasses down after the grasses had seeded, could give better results in arid areas. Here high-intensity grazing and trampling will not result in a vegetative ground litter cover of the veld (one claimed advantage of holistic grazing systems), simply due to too little available vegetation – breaking up the upper soil crust might result in soil erosion instead. It should be kept in mind that the large migrations of springbok *(Antidorcas marsupialis)* ("trekbokke") did not happen every year, unlike migrations in the higher rainfall areas of Africa (Skead, 1980; Beinart, 1998). The ecosystem being emulated is thus not the same as those ecosystems which first inspired the holistic grazing management paradigm (Savory, 2013).

- Limitations: This method might not be a good option for the more arid parts of Namibia and Southern Africa (Carter et al., 2014; see also text box above: *Rangeland impact of holistic grazing management in arid regions*). Unless combined with other methods like herding dogs, herders or mobile electric fences, it is not a good fit for livestock breeds in which the natural behaviour is to break up into small groups rather than flocking together in large herds. Large livestock herds staying in the same area for a short period only, depend on the territorial behaviour of predators to reduce the encounter rate of the livestock by predators (Linnell et al., 1996). This method is therefore less likely to work together with methods that disturb the natural territoriality of predators by removing individuals (Linnell et al., 1996).

32. Sterilizing breeding alphas: This method is not primarily to keep predator numbers down as often supposed, but because some predators (specifically canids like black-backed jackals and coyotes) tend to predate on livestock to feed their pups (Shivik, 2004). These species, where you have territorial alpha pairs who prevent others from breeding in their territories, can survive on other sources of food, similar to non-breeding individuals, if there are no pups to feed (see Jaeger, 2004). Denning (the killing of young and/or breeding alphas in their dens with poison fumes) or injection of predators with contraceptives works on the same principle (Linnell et al., 1996; Du Plessis et al., 2018). Denning is often used in some parts of Namibia to kill or capture African wild dog pups, thus encouraging the wide-roaming packs to move on (personal observation). Within the optimal foraging framework ($H = C + P + MOC$), this method works on the principle of decreased energy requirements (lowering giving-up densities) and possibly the slightly higher predation risk (P) by humans that is already associated with livestock, making smaller natural prey items a more attractive (or at least sufficiently productive) alternative. While mentioned a few times in the literature, this method has not been evaluated or compared for its effectiveness. However, a number of studies mentioned that it is impractical for various reasons, so its true success probability is likely to be lower than the assigned prior p(success) = 0.5 (Jaeger, 2004; Shivik, 2004; Du Plessis et al., 2018).

- Advantages: Where the primary reason for livestock depredation is the feeding of their young, this method can be very effective (Jaeger, 2004).

- Disadvantages: This method is both difficult (finding the breeding animals in the first place) and expensive to implement (about US$ 600 per animal - Shivik, 2006).

- Limitations: Like lethal methods, this method is not really an option against threatened or scarce predator species (Linnell et al., 1996). Only where most livestock depredation coincides with the breeding season of the predators on the farm, would this method have any chance of success. Because of the expertise and

costs required this is not really an option for most farmers to implement by themselves (Jaeger, 2004; Du Plessis et al., 2018).

2.2.3.2 Compensatory methods

Fundamentally, compensatory methods (also called "payments to encourage coexistence" – Dickman et al., 2011) are ways to address the fact that there is a global benefit to the continued persistence of biodiversity (including carnivores), but that most of the costs of coexistence are borne by local communities (including livestock farmers) (Nyhus et al., 2003, 2005). These methods attempt to either increase the benefit of persistent high biodiversity (coexistence with predators) to local denizens or to spread this cost of coexistence globally (Dickman et al., 2011). These methods generally depend on economic, rather than ecological principles and are thus considered to have little ecological impact, except for trophy hunting, which is considered to have medium ecological impact since it disturbs the territorial stability of the predator population. In terms of the ecological framework of optimal foraging theory ($H = C + P + MOC$), they have little effect, except for trophy hunting possibly raising the predation risk (P) slightly for predators attacking livestock. An important observation is that these methods do not aim to reduce actual livestock losses, but help farmers to absorb the costs of farming with predators on their land. They should be combined with other methods that can actually decrease livestock losses.

33. Trophy hunting of predators: Because of the effectiveness of utilization of other game animals leading to an increase in their value and numbers on farms, many Namibian farmers are in favour of a similar use of predators via trophy hunting (various, personal communication). A similar approach has worked for leopards in Uganda (Treves and Karanth, 2003) and for lions in Africa (Lindsey et al., 2012; Bouché et al., 2016). However, Treves (2009) makes the point that hunting for protecting livestock and sustainable hunting for conservation of predators, are really different aims and might be ultimately incompatible. Too little is known about both predator behaviour and hunter behaviour to be sure that the aim of coexistence will be reached and therefore careful monitoring and adaptive management is required. Balme et al. (2010) drew up a range of requirements to increase the sustainability of leopard trophy hunting in Kwazulu-Natal province, South Africa. They made the important point that trophy animals are taken *in addition to* illegally killed leopards. The N/a'an ku sê conservation and research NGO in Namibia has been lobbying that trophy hunting Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) permits in Namibia should be given preferably to farms that have experienced high livestock losses (R. van Vuuren 2017, personal communication). The restriction of trophy hunting to males that are past their reproductive age (and coincidentally often also responsible for most livestock depredation – Linnell et al., 1999) is considered as one of the better ways to ensure sustainable hunting levels when there is a lack of reliable population monitoring (Hunter et al., 2013). However, as a conflict mitigation method Treves (2009) showed that it does not necessarily decrease conflict. This method is thus assigned the value of 0.5 as its prior probability of success, indicating this current uncertainty.

- Advantages: Putting an economic value on predators, can potentially change the attitude of many farmers to become more positive towards carnivores and if the value is high enough, to try and sustain carnivore numbers on their farms. Currently far more predators are killed in Namibia to protect livestock, than are trophy hunted; with many farmers taking a "shoot, shovel and shut up" attitude (Marker et al., 2003b; Hoogesteijn and Hoogesteijn, 2010; various Namibian farmers, personal communication).

- Disadvantages: Trophy hunting is normally selective for large males and can change the population structure of predators negatively if not managed properly (Balme et al., 2010; Keehner et al., 2015). Additionally, it can cause other negative ecological effects like sexually selected infanticide (SSI) (Balme et al., 2009; Packer et al., 2009). Trophy hunting can ultimately have a negative impact on predator survival, unless it is very well managed (Hunter et al., 2013). Lindsey et al. (2013a) mention some current management

practices that undermine the sustainability of lion trophy hunting and a number of changes to make it more sustainable - some of which might also be applicable to ensuring sustainable trophy hunting of other predators.

- Limitations: The greatest constraint on the use of trophy hunting of apex predators, is that typically being on the top of the trophic pyramid (see Figure 1.4 on page 12) means that their total biomass will be less than that of their prey (which are lower on the food pyramid and include the usual trophy hunted game species in Namibia). Top predators are therefore likely to occur at lower densities and typically have larger home ranges than their prey (Estes et al., 2011; Ripple et al., 2014). This means that a single predator often use more than one farm as part of its home range and that a single farm with its available natural prey and habitat, can often support only relatively few predators (compared to herbivores). Unless the income from a single trophy predator shot every few years surpasses the livestock losses due to those predators living on the land, it will not act as an incentive for farmers to sustain predator populations on their land. In this case trophy hunting of a predator will at best offset some of the costs spend by the farmer to prevent livestock losses (see also McGranahan, 2011). One obvious exception to this constraint would be if mesopredators, such as jackals which are found in much higher densities and with smaller home ranges, were trophy-hunted for their pelts.

34. Compensation for livestock depredation losses: Compensation schemes have had mixed results (Anthony and Swemmer, 2015). The idea behind this method is to enable and encourage farmers to coexist with predators by paying them in part or fully for livestock losses due to predators. There are typically three sources of funding for the payments, 1) government, 2) hunters' associations or 3) private conservation organizations (Linnell et al., 1996). Bauer et al. (2017) mentions using tourism income to pay compensation for livestock losses due to lions (also Stander, 2008). To function at all, such compensation schemes need to be 1) simple, 2) rapid and 3) safeguard against false claims (Linnell et al., 1996). Nyhus et al. (2003) found the most effective compensation projects to be 1) fair, 2) transparent and 3) above all, fast. This included: 1) quick, accurate verification of damage; 2) prompt and fair payment. 3) sufficient and sustainable funds; 4) being site specific; 5) clear rules and guidelines (simple and transparent); 6) measures success appropriately as having an effect on numbers of predators killed (see text box *Case study* below). Dickman et al. (2011) found that in practice compensation rarely covers all the costs of coexistence with predators. Paying below market value of losses are often done to prevent a perverse incentive ("moral hazard") when the compensation pays better than taking livestock to the market, resulting in farmers not expending any effort to prevent livestock losses and sometimes even claiming for non-predation livestock losses (Nyhus et al., 2003; Bauer et al., 2017). Mishra et al. (2003) opined that most initiatives to offset the costs of living with carnivores and to make conservation beneficial to affected people, have thus far been small, isolated, and heavily subsidized. Montag (2003) concludes, "Compensation may be viewed as a useful tool, but one with limitations and possible unanticipated adverse consequences." An overview by Krafte Holland et al. (2018) found compensation being considered effective in 64% of human-*Panthera* conflict cases, giving it a prior success probability of 0.64.

Case study — One of the few reported cases where compensation appears to be sustainable and effective, involved a World Wildlife Fund (WWF) funded project for Central Asian leopards in Turkmenistan (Lukarevsky, 2003). However, it had a number of unique features: 1) The team included members of the local community, including the leader himself. It was thus not a case of "outsiders" running the project. 2) Local residents actively participated in planning a strategy for leopard conservation. 3) Instead of money or other means of compensation, it was decided to compensate farmers in kind (i.e. with other livestock to replace any livestock killed by a leopard). 4) Using the funding money from WWF a herd of sheep was bought, from which the offspring would be used to give compensation for depredation losses. 5) The herd became the property of the local-based Catena Ecoclub, enabling them to make decisions with regards to how the flock would be organized, how cases of leopard attacks would be analysed, etc. 6) Because the livestock herd can reproduce and was not used for any other purpose, this ensured the sustainability of the original WWF funding. A flock of 650 – 700 sheep would grow on its own and cover the cost of paying shepherds and veterinarians. 7) Next, 40 of the most influential and respected ranchers were invited to participate in a seminar where the project was explained to them. A council was elected to look after the herd on an annual basis and it was decided that the livestock would eventually become the property of the local farmers association once it was established. 8) This local council elected two experts to investigate cases of supposed leopard kills. 9) A simple procedure was established by which a farmer would register a case of livestock losses, and the experts would not only decide whether it truly was a leopard kill, but also whether the farmer's livestock were properly managed to prevent predation. 10) The local communities were kept up to date about the progress of the project using poster display stands and education. The long-term plan was to combine education and compensation to ensure the survival of Central Asian leopards. Perhaps the most important lesson from this case study was the role of local leadership in all phases of the project, ensuring that the long-term beneficiaries and implementers of the project were also involved in the decision-making process and the resulting project was well-adapted to local conditions (see also Anthony and Swemmer, 2015). The separation of the two arms respectively verifying predation kills and the other deciding on how much compensation needs to be paid, appears to be a general requirement for successful compensation schemes (Nyhus et al., 2003).

- Advantages: Most farmers agree that compensation is a nice idea if implemented properly (Rust, 2016). In areas without any natural prey and where predators are dependent on livestock for their survival, this might be the only way to achieve coexistence of livestock and predators (Linnell et al., 1996; Mishra et al., 2003).

- Disadvantages: One obvious drawback to this method is that on its own it gives no incentive for farmers to protect their livestock from predators. Additionally, it is not always easy for farmers to prove that losses were due to predators, especially when the whole prey animal is consumed. Failed expectations of farmers about compensation often result in unanticipated negative results, when farmers become even more negative towards predators than before (Montag, 2003). Mishra et al. (2003) mentioned that some reasons why current compensation schemes did not work, included 1) time and costs involved in getting compensation and 2) low compensation rates (only 3% of total loss). The greatest issue with this method is the question of who should be responsible for funding the compensation and how to keep funding sustainable. The difference between urban and rural values (Montag, 2003; Rust, 2016), where sometimes farmers have the perception that people who do not have to deal with the consequences, are trying to manipulate them into keeping predators on their land, can also result in the situation where compensation payouts have no significant effect on tolerance for predators (Naughton-Treves et al., 2003).

- Limitations: This is not really a method that an individual farmer can implement, but it needs the cooperation of various role-players, with policy, responsibilities and roles well defined to address issues about the legality and liability of wild-life damage, Endangered Species Act legislation, and private property rights. In situations with protected species, the government is the only body that can assume certain

responsibilities for human-wildlife conflicts in areas where wildlife is under the stewardship of the people (Montag, 2003).

35. Insurance schemes for livestock depredation losses: Few instances of successful insurances schemes for livestock depredation losses have been reported (Mishra et al., 2003 discuss one notable exception). This method basically shares the same strengths and weaknesses of direct compensation for livestock losses. However, it differs in that the farmers themselves carry the costs for the compensation. Insurance is commonly used in Europe, but in general an unsubsidised, purely private insurance scheme is not viable (Nyhus et al., 2005). While thus likely having a lower actual success probability, it is here assigned the probability p(success) = 0.5 indicating the current lack of evidence.

- Advantages: Especially given the high reported rates of livestock losses and its uneven distribution (e.g. Van Niekerk, 2010), insurance might be an affordable and sustainable way to compensate for livestock losses. It can make the compensation independent and provide something that farmers can mostly do for themselves, instead of having to rely on outside funding organizations or government like other compensation schemes. This method can provide farmers with some peace of mind and works well with almost any other method. Where insurance is working well (i.e. sustainable), payment of rewards for those farmers with the least number of livestock losses to predators (least number of claims), can be an added incentive for good livestock husbandry practices, combining insurance with preventative methods (e.g. Mishra et al., 2003).
- Disadvantages: Paying for insurance against depredation losses still equate to a nett loss for most farmers (Montag, 2003). Most insurance companies do not have the expertise for a realistic assessment of livestock depredation risks and setting realistic premiums, therefore it is usually very expensive and only worthwhile in cases of extremely high livestock depredation rates. And like with direct compensation, there is the need for verification of predator kills when livestock are lost (Shivik, 2004).
- Limitations: Insurance against depredation livestock losses is not offered by most insurance companies in Southern Africa, so unless the agricultural unions can set up their own insurance scheme, it is unlikely to be available. However, it appears likely that it would not be economically viable according to Nyhus et al. (2005).

36. Compensation for predators on farmlands: This is one of the few compensation methods where successes have been recorded (e.g. Mishra et al., 2003). The best-known example is the payment by the Swedish government to Sami reindeer (*Rangifer tarantus*) herders for every certified carnivore reproduction event on their land (Dickman et al., 2011). Instead of directly paying farmers, paying price premiums for their produce (e.g. the "Wildlife Friendly Meat" project by Conservation South Africa in Namaqualand, South Africa) is an alternative method to compensate farmers for coexisting with predators on their land. Instead of paying for losses of livestock to predators, the meat from farms who followed certain management practices were to be branded as "Wildlife Friendly" and sold in Woolworths stores for higher than normal prices with the price premium being passed on to the farmers. Such niche marketing basically depends on changing the level of predation losses that is economically (and socially) acceptable to farmers (Shivik, 2004). It helps to shift the cost of having predators on their land to the consumers who prefer "green" non-lethal methods of predation control. While this method probably has a higher chance of success than direct compensation or insurance, it is also given a prior success probability of 0.5 due to a current lack of data.

- Advantages: Unstable markets and fluctuations in the prices of livestock products is a major issue for all commercial farmers. A price premium is one important way to help farmers manage the risk of coexisting with predators on their land instead of aiming for extirpation of all predators. Through rewarding livestock

farmers directly for conserving predators on their land, the real aim of these kind of compensation schemes is directly addressed (instead of a roundabout way of partial compensation for economic losses only). The farmers protecting their livestock well by using preventative methods, still profit by having greater livestock survival rates — i.e. there is no danger of a perverse incentive to allow livestock depredation in order to be compensated (*cf.* the snow leopard case study mentioned by Mishra et al., 2003).

- Disadvantages: Unless such a compensation scheme is adapted well to the local situation (e.g. by including local agricultural leaders in the planning and decision-making process), lack of participation by farmers might make it useless. Because many of the threatened predators have home ranges that overlap with more than one farm, a single farmer or even a few dispersed farmers, might not contribute much to the long-term survival of predator species. Finding shops that are willing to sell predator-friendly livestock products as a separate brand and are moreover willing to pass the premium price on to the farmers, has been difficult in practice. Similar efforts in Namibia to market and sell "cheetah-friendly meat" in the past fell apart because 1) it was difficult to ensure farmers' compliance with the required management practices and 2) separate branding and marketing (and passing higher prices on to the producers) was an issue for the supply chain (L.L. Marker 2013, personal communication). There was and still is an uncertainty on whether the demand for such products is high enough to sustain a higher price. Consumers might support the idea of "predator-friendly" meat production theoretically, but are they willing to pay more for such meat in practice? Woolworths, as a franchise that targets the upmarket section of consumers, might have been considered as the ideal partner in such a project, but up to the present, higher producer prices failed to materialise. Like in the case of compensation for livestock losses, confirmation of certain husbandry practices or numbers of carnivores is still required before payment can be done (Nyhus et al., 2005). Direct payment for predators suffers from the same issues with sustainable funding as compensation for livestock depredation.

- Limitations: This is not really an option for a single farmer without the support of at least the local farmers association or even better, the national agricultural union.

37. Ecotourism: Like trophy hunting, ecotourism is another way to benefit directly from predators on farmlands. In communal areas, where regular income from livestock sales are relatively low (Hangara et al., 2011b, 2012), ecotourism involving predators can potentially generate greater income than livestock farming (Stander et al., 1997a). In Namibia, it has been shown by Lindsey et al. (2013b) that income from ecotourism was an important determining factor in commercial farmers being more positive towards predators on their land and also positively correlated with membership of a conservancy (Lindsey et al., 2013c). Big cats were the preferred species that interviewed tourists in Namibia wanted to see in in the wild (Stein et al., 2010). Krafte Holland et al. (2018) reported a 0.25 success ratio for tourism resolving human-*Panthera* conflict (used as prior probability).

- Advantages: Ecotourism can increase farm income a lot and in some cases even replace livestock farming (McGranahan, 2011). The possibility to see predators in the wild is a significant drawing card for many tourists and hunters (Hoogesteijn and Hoogesteijn, 2010; Stein et al., 2010). Unlike many of the other compensation methods, it can be implemented by a single farmer on his own land.

- Disadvantages: Because predators are often so elusive, tourists might be disappointed and avoid returning to a tourist enterprise if they had unrealistic expectations of seeing a predator in the wild. It is difficult to combine tourism with hunting. Most livestock farmers do not have the skills or inclination for the kind of work required for a well-managed tourism enterprise (McGranahan, 2011). Becoming known well enough to attract enough tourists to compensate for all livestock losses due to predators, will probably also take some time. Some "ecotourism" activities, like using baits to attract predators for tourists, can habituate predators to humans and teach them to associate humans with food, leading to more livestock

depredation and even endangering people's lives (Orams, 2002; Hoogesteijn and Hoogesteijn, 2010; Athreya et al., 2010)!

- Limitations: Finding wild predators for tourist viewing require excellent tracking skills, a soil substrate in which tracking is relatively easy, and enough time, unless some of the predators have been radio/GPS collared. Tourism is not a realistic option for remote farms in a monotonous landscape that are unlikely to attract many, if any, tourists.

2.2.3.3 Reactive methods

Many Namibian farmers prefer not to do anything about predators or possible livestock depredation, until sustaining actual livestock losses (Stein et al., 2010). Where livestock losses are relatively low, this can be a cost-effective approach. Because in all three cases the territorial system of the predator species is disturbed, these methods are considered as having a medium ecological impact. In terms of the optimal foraging framework ($H = C + P + MOC$), these methods have little effect. They may cause a slight increase in the predation risk (P) to predators hunting livestock, but because they are retroactive instead of proactive, it is unlikely to result in the creation of a "landscape of fear" that would be required to keep predators away from livestock.

38. Hunting and trapping reactively (killing a single "problem predator"): In contrast to the preventative use of hunting to eradicate predators or decrease predator numbers, hunting (with or without dogs) can also be used reactively. It has long been known that removal of a single predator can sometimes stop any further livestock losses. The main objective is to kill the specific individual predator responsible. Even if the method has no effect on livestock depredation, it does give the farmer some intrinsic satisfaction to kill the guilty party (Forthman Quick et al., 1985). However, Chapron and Treves (2016) showed that allowing killing of predators or government culling of predators did not make livestock owners more positive towards predators, but actually increased illegal killing of predators. An important assumption of this method, and one that has been challenged (e.g. Linnell et al., 1999) is that there are specific individuals that become habitual livestock killers or "problem animals" (see box below: *Problem animals?*). For some species (e.g. coyotes - Sacks et al., 1999; Jaeger, 2004) it has been shown that a few individual animals are responsible for most of the livestock depredation (*cf.* Stander, 1990a). However, this same species simply takes sheep according to availability and doesn't show any preference for sheep (Sacks and Neale, 2002). Other species of predators also seem to utilize livestock within their preferred size range according to their availability, making little differentiation between livestock and other prey species (e.g. brown bears *Ursus arctos* and wolves *Canis lupus* in Norway – Linnell et al., 1996, 1999, and possibly caracals on Karoo farmlands Drouilly et al., 2018). In this case, removing any individual animal will simply result in its place been taken by another individual who will continue to opportunistically prey on livestock, at best giving a short reprieve in livestock losses. Balme et al. (2009) suggested the following procedure to ensure that only habitual livestock killing leopards were removed: 1) inspect depredation events within 24 hours of being reported; 2) if a leopard is verified as being responsible for the damage, an attempt is made to identify the individual by deploying camera-traps at the kill site; 3) a destruction permit is granted only when the same leopard is known to be responsible for at least *three depredation events* within a two-month period. A similar method has been used occasionally in Namibia, but using a GPS collar on the suspect leopards, instead of camera traps (Stuart Munro 2015, personal communication; see method 26 on page 71). Ecologically, predators are generally considered as being on a scale varying between specialist (preying on one of only a few prey species) and generalists (having a wide dietary niche and preying on many species – *cf.* Avenant and Nel, 2002 and Peers et al., 2012). However, it has been shown that within some generalist *species*, individuals can still show narrow prey preferences and be *individual* specialists (e.g. Matich et al., 2011). This method thus depends on the fact that some *individuals* of a certain predator species (with different individual prey preferences), could come to prefer livestock species as prey. Because the literature did not differentiate

between proactive and reactive lethal methods, this method has the same prior probability of success as hunting in general: p(success) = 0.57.

Problem animals? — Jaeger (2004) showed that in coyotes (which have similar behaviour to black-backed jackals, see also Estes, 1991; Ferguson et al., 1983) it is principally the resident alpha breeding pair that are killing livestock, and removing them will stop livestock killing until a new alpha pair is established three to four months later (Sacks et al., 1999). Indiscriminate killing of coyotes had no effect on livestock losses at all, mostly because the betas and transient individuals were much more vulnerable to capture than the alphas. However, they did not investigate what the long-term effect will be, and the Conradie and Piesse (2013) study showed that it could increase livestock losses in the next year. Three possible ecological explanations for such an increase could be that the neighbouring alpha pairs expand their territories so that the territory which used to house one territorial breeding pair is now shared by six or more breeding pairs. Alternatively, the helpers (betas) might all start breeding in the next year since there is no alpha pair to suppress their breeding. Or dispersing young animals from outside may settle in the territory, but with two or more pairs settling there instead of only one pair. The season of the killing will probably play an important role in what happens (see Lieury et al., 2015). For felids a similar process probably occurs, as demonstrated by pumas (recounted on page 53 from Linnell et al., 1996 – see also López-Bao and Rodrigues, 2014 and Fattebert et al., 2016).

Linnell et al. (1999) grant that there are examples from the literature where individual predators start to preferentially prey on livestock. However, the majority of livestock losses is simply a case of "wrong time, wrong place". These predators they called "Type 1 problem animals". Some predator sex or age classes might simply be *more likely to encounter livestock* (Linnell et al., 1996) or have higher energy requirements that would be met by any prey in the same size class as livestock (Kamler et al., 2012a; Klare et al., 2010). Because no attempt is made to protect livestock or differentiate them from wild prey, these predators perceive them as just another prey species[a] (e.g. coyotes – Sacks and Neale, 2002). The other kind of "problem animals", called "Type 2", are those who kill more than one livestock animal in a single hunting event (*surplus killing*). This is simply natural hunting behaviour in an unnatural situation (Linnell et al., 1999). Neither of these cases should therefore be considered as "problem animals". Both Type 1 and Type 2 "problem animals" can be better described as "opportunistic livestock killers" and removing them will have little effect on livestock losses (or even be counter-productive). Reactive killing of a predator however, is aimed at *habitual* livestock killers; i.e. where the same individual starts to prefer and hunt livestock as prey and kill *multiple livestock* in *different* hunting events.

[a] At least for leopards and cheetahs in Namibia this is not generally the case, since they have been shown to take less livestock than what is available, demonstrating some avoidance of livestock as prey (Mizuntani, 1999; Marker et al., 2003d; Stein, 2008).

- Advantages: Since it is only used after livestock had been killed, it can be substantially less expensive than pre-emptive hunting. Especially if dogs are used to take the trail from where livestock have been killed, this method can be very selective in order to kill the specific individual that was responsible for killing livestock (Snow, 2009). If the losses were simply because of an opportunistic predator, this method requires no further action (or costs). If a single predator individual is responsible for all or most livestock depredations, killing it can be very effective.

- Disadvantages: If the guilty predator was not a habitual livestock killer, removing it might have no effect at all on the livestock losses[1]. In the case study from the Ceres Karoo hunting club (Conradie and Piesse, 2013), predators were mostly killed in response to livestock losses. And yet, the result was that those farms where most predators were killed in a certain year, had an increase in livestock depredation in the following year (however, there is little indication that this killing was very selective). Killing predator

[1] At least one case is known where a collared male leopard killed more than 20 sheep in a single night on two neighbouring farms, but never killed another sheep again over a period of years that it was collared, even though passing within 50 m of the same sheep-folds (Quinton Martins 2010, personal communication). This leopard could have been classified as a "Type 2 problem animal", but was clearly not a habitual livestock killer (Linnell et al., 1999).

reactively might therefore be counter-productive (see method 5 above). It is actually quite difficult to be certain that the specific individual who had killed livestock, is being targeted. This method (even if effective), requires at least two livestock depredation events (and resulting losses) before a habitual livestock killer can be identified.

- Limitations: Being selective and finding the guilty predator depends critically on any livestock depredation events being found shortly after the kill. If the livestock are spread out widely over a big farm or the labour force on the farm is small, this might not be practical. It might really only be effective in those situations where an individual predator has become a specialist killer of livestock. As shown by Linnell et al. (1999), even a single "over-kill" (surplus killing) event does not mean that the individual predator has become a habitual livestock killer.

39. Translocation of problem animals: On the farm where the predator is captured, this method has pretty much the same effect as trying to kill a "problem" individual and reactively targeting individuals that had killed livestock. However, not only the effect on the farm from where it was removed, but also on the area to which it is relocated, would determine the method's effectiveness (Weilenmann et al., 2010). Griffith et al. (1989) gave an overview of factors that will determine relocation success from a conservation point of view (*cf.* IUCN/SSC, 2013). For its success on the farm from which a predator is removed, this method depends on the same ecological principles as method 38 (individual prey specialisation). Moreover, it is additionally influenced by the territorial systems of the various predator species, their homing instincts, their usual dispersal distances and the sizes of their home ranges for its success at the receiving end of the translocation. Strictly speaking, translocation can be used pre-emptively if an individual predator is removed without any evidence of being a habitual livestock killer — in which case it will resemble method 5 above (hunting to reduce carnivore numbers) in effectiveness and disadvantages, but it will cost much more. It is thus not considered further here. Various authors indicated that translocations had a low probability of success overall, especially for mitigating human-wildlife conflict (Linnell et al., 1997; Fischer and Lindenmayer, 2000), with Krafte Holland et al. (2018) reporting a success ratio of only 0.14 for the big cats (*Panthera*). When pooled, ignoring species, overall 44% of translocations in Namibia were successful, with 10% unknown outcomes (Weise et al., 2014; see text box *Case study* below), giving a prior success probability of 0.45 for this method.

Case study — In Namibia Weise et al. (2015a) found a 40% success rate for cheetahs and a 67% success for leopard translocations (Weise et al., 2015b). For leopards, all farms (n=4) from where they were removed had no further livestock losses for 16 months after previously losing on average five calves per farm. After 16 months they lost ≤ 3 calves per year to predation, which were below the tolerance level previously reported by Namibian farmers (Stein et al., 2010). It is thus likely that in these cases the translocated leopards were actual habitual livestock killers (although most of them did not resume livestock killing after translocation – Weise et al., 2015b). For cheetahs, however, landowners reported renewed conflict with cheetahs within 12-24 months after translocation (Weise et al., 2015a).

- Advantages: For threatened or scarce species, relocating an individual that has killed livestock to somewhere else, can mitigate conflict, save the life of that specific animal, decrease livestock losses, and advance the overall survival of the predator species (Griffith et al., 1989; Linnell et al., 1996; Fischer and Lindenmayer, 2000). Similar to killing reactively, removing a single habitual livestock killer (or pride of predators) which has been responsible for most livestock kills, can stop most or all livestock depredation on a farm.

- Disadvantages: This method will make no difference if there is no single "problem" individual. Linnell et al. (1999) makes the point that there are two kinds of "problem predators": those who are simply in the wrong place (i.e. hunting in a habitat where there are lots of livestock) or those who actually start to specialize in killing livestock. In the former case, the territorial animal who was responsible for livestock losses will simply be replaced by one or more others who will continue killing livestock. Various studies have

shown that predators, especially felids, often move great distances after having been relocated, and either resume killing livestock somewhere else or return to their original home range (and sometimes continue killing livestock there – Stander et al., 1997a; Weilenmann et al., 2010; Weise et al., 2015a). Additionally, if it is moved into an area with high densities of con-specifics, a territorial predator will either have to replace the current inhabitants of the territory (followed by SSI in males), be killed by a current territorial individual, or have to move out and find an empty space for itself, often on farmlands again (Hayward et al., 2007a; Balme et al., 2009). In effect, a so-called "non-lethal" method still results in the death of one or more predators (Treves and Karanth, 2003). The effect would be no different than that of trophy hunting (Balme et al., 2010; Keehner et al., 2015), leading some conservationists to conclude that it is sometimes better to simply destroy problem individuals, rather than trying to relocate them (Stander, 1990a; Ropiquet et al., 2015). Relocating some species over long distances can compromise their genetic integrity, thus diluting their adaptability to local conditions (Ropiquet et al., 2015). Compared to just shooting the problem animal, it is quite an expensive method, with an estimate for leopard relocation in Namibia of US$ 3 140.00 per animal (Weise et al., 2015b). However, it is one of the few available methods where the farmer does not have to carry all the costs (except for the initial capture of the predator), which might partially explain its popularity in Namibia. Because predators often travel long distances, it can simply relocate the problem of livestock depredation to another area when the relocated predator leaves the conserved area to which it had been moved (Weilenmann et al., 2010; Weise et al., 2015a,b).

- Limitations: For stock-killing lions, Stander (1990a) found that relocation of habitual livestock killers did not work, with the lions simply moving back to areas where they would continue to prey on livestock. In this case, relocation of lions was only successful for occasional opportunistic livestock killers. The second constraint was that lion prides had to be either relocated to their original territories within protected areas, or moved to areas where there were no other lions yet. The same principles hold true for some other predator species. However, this necessitates that enough is known about both the animals that are to be relocated and the area to which they will be relocated. Weise et al. (2015b) found that there are some situations in which leopards can be successfully translocated in Namibia. However, the model will differ between countries and species. For leopards in Namibia, it was proposed that translocation should happen over distances larger than 200 km, to a conservation area (or land-use without conflict potential) of at least 875 km^2, with current low resident leopard densities (<2 leopards/100 km^2). Releases should also happen at least 18 months apart to mimic natural dispersion events of sub-adult leopards. Although having a data point of one female only, a soft release method appeared to result in less movement post-release. The model had a 67% to 83% success rate (depending on how success was defined), but they still did not recommend it as a standard response to carnivore conflict, since there are only a limited number of places in Namibia left to which leopards (or other predators) can be relocated using the attributes required by the model of Weise et al. (2015b). For cheetahs, the greatest limitation was finding conservation areas that were large enough for their release without renewed conflict in the release area Weise et al. (2015a). Cheetahs roamed far outside even the largest conserved areas into which they were released, leading the authors to conclude that this method is not recommended for HWC involving cheetahs in Namibia (Weise et al., 2015a). Some researchers have questioned the very existence of "problem animals" and the results of cheetah translocation in Namibia would support this idea that most cheetahs simply take prey in their preferred size class according to availability and most recent experience (Linnell et al., 1999; Zhang and Hui, 2014; Weise et al., 2015a).

40. Poisonous Livestock Protection Collars (LPC): One of the few uses of poison that can be defended from an ecological point of view (because of its selectiveness), is the use of collars on livestock with poison pockets around the throat of the sheep (Linnell et al., 1996; Bothma, 2012; Du Plessis et al., 2018). Ecologically, this method depends on the hunting and killing technique of most predators for its effectiveness. From the literature this method is given the same 0.57 prior success probability as other lethal methods.

- Advantages: This is the only use of poison that are selective in killing only those predators that had attacked livestock. This could lead to surviving predators avoiding sheep in future. However, since the poison is usually very effective, there are very few if any survivors. The farmer does have the personal satisfaction of the guilty predator being killed.

- Disadvantages: The greatest disadvantage is that the attacked sheep is still killed in order to kill the predator! Moreover, if any predators survive biting into the collar, they might learn to avoid the collared sheep only, while still attacking other sheep. Moreover, Brand et al. (1995) showed that jackals could learn to avoid coyote getters from observing other jackals getting killed — the same may be true for poison collars. Another great disadvantage of poison collars is the high costs of the collars. This means that not all sheep can be collared, and thus that sheep without LPCs remain vulnerable to predators. Additionally, it has all the same disadvantages of other kinds of collars (labour intensive, collars needing to be replaced regularly in growing sheep, predators changing their attack method to avoid the neck or throat, etc. – see methods 15 and 16 above). Because the poison is also dangerous to humans, pets and even livestock, care is needed when using the collars.

- Limitations: Scavengers that eat from the carcasses of predators killed by poison, may in turn die from the poison (depending on the specific poison used) (Bothma, 2012). It can thus have the same ecological ripple effect on non-target animals as other poison delivery systems (see method 3 on page 50). In this case, all the advantages of the collars being more selective than other applications of poison are lost.

2.3 The ecology of farmer-predator conflict in Namibia

This part of the chapter focuses on the ecological aspects of farmer-predator conflict in Namibia (see Figure 1.4 on page 12). Du Plessis et al. (2015) highlighted that the quality and quantity of scientific information in South Africa on black-backed jackals and caracals were insufficient to make any real contribution to mitigating human-wildlife conflict. In Namibia, even less relevant information has been published about these two species on farmlands, with almost all publications on jackals located on the arid Namib coast and a single publication on caracal ecology restricted to the North-Central part of Namibia (Marker and Dickman, 2005b). The latter area is primarily a cattle and game farming area (Mendelsohn et al., 2003; MAWF, 2017). Caracals are generally not considered an important livestock predator of cattle in Namibia and most conflict with caracals occur in the small livestock farming areas of Southern Namibia (Shortridge, 1934; Schumann, 2004; Potgieter, 2011). Similarly, although more has been published on their ecology, Balme et al. (2014) showed that past research on large predators and leopards in particular did not sufficiently address the issues of human-wildlife conflict. This current lack of evidence-based scientific knowledge emphasises the importance of an adaptive management approach in mitigating farmer-predator conflict.

It is important to know what we don't know. The main aims of the ecological part this literature review were to identify and describe:

1. those aspects of the ecology of the four Namibian "problem predator" species that we already know and which can be applied with greater confidence within an ecological theoretical framework,

2. those aspects where current knowledge is lacking and solutions are based on intelligent guesswork (sometimes called "expert knowledge – Johnson et al., 2013; Child et al., 2016) and

3. priorities for further ecological research in order to improve our decision-making model for mitigating conflict.

In chapter one (Section 1.6.2 on page 19) three basic aspects of predator ecology which can be relevant to mitigating farmer-predator conflict were identified:

1. spatial ecology[2 3 4 5],
2. diet and prey preferences (including hunting behaviour)[6 7] and
3. intra-guild interspecific interactions[8 9].
4. As seen above, Linnell et al. (1996) made the observation that the conservation status of a predator species can also limit the suitability of some conflict mitigation techniques.

These four aspects of the ecology of the four predator species are thus the focus of this ecological review with the three main aims mentioned above. Du Plessis et al. (2015) mentioned a number of possible behavioural changes in the ecology of black-backed jackals and caracals that can influence the effectiveness of predation management on farmlands:

1. how do territorial behaviour, density and ranging behaviour (habitat preferences, activity periods, daily distances moved) on farmlands compare to populations in "natural" protected areas (spatial behaviour);
2. how do prey preferences and seasonal diet selection compare on farmlands to more "natural" populations;
3. have there been any changes in breeding seasons; and
4. does compensatory breeding happen on farmlands and how could this influence hunting behaviour?

2.3.1 Materials and methods

The quantity and quality of published ecological information differed markedly between the four predator species. Using the search term "Namibia *Canis mesomelas*", the ISI Web of Science database returned only

[2]Home range size can be used to get an indication of the resident population density which in turn will give an indication of the likely success of HWC mitigation methods that depend on population size control or extermination. For predators where the home ranges are larger than the individual farm, when combined with the average daily movement of the predator it will give an indication of how often and how long they will be on each farm and of how intensive the required management interventions will need to be. Threatened carnivores with large home ranges are also more likely to be exposed to farmers who use lethal predation management and because they are shared by more individual farms, this can give an exaggerated impression of the number of carnivores (Marker et al., 2008). Conservancies consisting of more than one farm managing the ecosystem together, can have a beneficial effect on the stability of large carnivore spatial structure and indirectly on livestock losses (Lindsey et al., 2013b).

[3]In some predator species (e.g. coyotes), it is mostly the resident breeding individuals that are responsible for most livestock kills (Jaeger, 2004). In other predator species it might be the opposite, with inexperienced dispersing youngsters and old, displaced individuals more willing to risk killing unfamiliar prey out of desperation (Melzheimer et al., 2018). Similarly, for some species, capturing or killing the residents are more difficult (Jaeger, 2004), since they are more likely to avoid unfamiliar objects in their home range, while for others it is the opposite, since they regularly visit specific marking sites in their territories (Melzheimer et al., 2018).

[4]Habitat preferences can influence predation management by identifying those areas which is preferred by the predator species causing most damage and keeping vulnerable livestock away from there. It has been shown for some predators that hunting habitat is more important than the prey species (Balme et al., 2007). Similarly, females with young often show a contraction in home range, remaining closer to denning sites (Loveridge and Macdonald, 2001). Thus not only the generally preferred habitat, but also the predator utilization of different habitats, can be used to manage livestock losses.

[5]Dispersal differences will influence both the minimum distances required for translocation to prevent the problem animal simply returning to its previous home range, and the minimum distance from a source population of predators for local extirpation of predators to have any significant effect.

[6]There are three alternative hypotheses: 1) predators actually prefer livestock within their preferred prey sizes to natural prey (livestock is commonly thought of by farmers as "easy prey"), 2) they only take livestock opportunistically according to their availability (Linnell et al., 1999), or 3) they actually prefer their natural prey to livestock. The more likely predators are to prefer livestock, the more intensive management will be required.

[7]It should be noted that the predator *species* could be a generalist, but *individuals* within the species can still specialize on specific prey species (Matich et al., 2011). Conversely, the predator species can have a relatively small number of prey species (specialist), but individuals show little difference in their prey selection among those species. The more individual prey specialisation a predator species shows, the more likely it becomes that removing certain problem individuals will mitigate livestock depredation.

[8]If interspecific competition between predators causes spatial partitioning, vulnerable livestock can be kept away from those areas preferred by a more damage-causing predator species. Because predator species have different prey preferences, the most vulnerable livestock in a certain season may fall in the preferred prey size for one species, but not in the preferred prey size for the other. Exploiting such spatial partitioning can thus lead to a significant decrease in livestock depredation with little additional costs.

[9]If mesopredator numbers have exploded to unsustainable levels due to the extirpation of larger predators, the restoration of apex predators may mitigate livestock losses (Ritchie and Johnson, 2009). Jackals, for example, also change their prey preferences in the presence of apex predators, not only as a result of scavenging (Hayward et al., 2017). While larger predators can still cause significant livestock losses, managing predation losses can be easier because of the lower densities at which they generally occur.

38 articles, of which only 15 were relevant to jackal ecology (13 in the Namib/coastal area and two in Etosha National Park). Searching the ISI Web of Science database with the search term "Namibia Caracal", resulted in only 12 articles, of which only one was relevant to caracal ecology (the others were mostly about parasites and diseases or about human-wildlife conflict, often with only incidental reference to caracals). Searching the JSTOR database added no more articles for either species. The results were a little better for "Namibia *Panthera pardus*" (Web of Science 36; JSTOR 111 results) and "Namibia *Acynonix jubatus"* (Web of Science 93; JSTOR 114 results), although the majority of the articles did not focus on the *ecology* of either species. These literature search results were therefore supplemented by information and citations from the LCMAN Red Data List for Namibian Carnivores (in prep) – to which the author also contributed. This publication included not only extensive additional literature citations (including as yet unpublished new findings), but also included the most up-to-date photographic and geo-referenced records of carnivore species in Namibia.

2.3.2 An ecological framework

Within the ecological viewpoint, human-wildlife conflict has often been approached from a single species (conservation) paradigm (e.g. Sacks et al., 1999; Marker, 2002; Wallace et al., 2011; Pitman, 2012; Packer et al., 2013a; Carvalho et al., 2015; Swanepoel et al., 2015). Here, a holistic whole ecosystem paradigm in human-wildlife conflict management is suggested (see Figure 1.4). Whole system approaches help avoid situations where one "solves" one problem, but the solution becomes unsustainable because of not taking into account the wider ecosystem sufficiently (Snow, 2009). It also helps to view solutions from a proper theoretical framework, so that we can understand *why* a certain management option works and not only *if* it works. Such a theoretical framework is a typical core principle of adaptive management (McDonald-Madden et al., 2010). This enables the application of results to the wider human-wildlife conflict issues and to know when a certain management option is applicable and likely to work in other conflict situations (i.e. with other wildlife species and other types of agroecosystems).

2.3.2.1 Ecosystem sustainability

In Section 2.2 above it was assumed that those management techniques with less human impact on the existing ecosystem will be more sustainable in the long run. This section provides a theoretical basis for that assumption. Margalef (1963) already made a case for unifying principles to be established in ecology. He provided a basic framework within which ecological results can be interpreted in terms of "maturity" and the primary productivity:biomass ratio in an ecosystem. It was proposed that ecosystems can be graded in terms of "maturity", with more mature ecosystems driven mostly by biotic interactions, being more biodiverse, having a higher biomass to primary production rate, able to sustain itself on lower energy levels, and being generally more stable (Margalef, 1963). Less mature ecosystems on the other hand, are mostly driven by less predictable abiotic factors with higher variability, have typically higher primary production per unit biomass, have more young and dispersing individuals of a single population, have larger variation in population numbers of individual species, and lower biodiversity. Also, typically the mature ecosystem would export more young, dispersing individuals when compared to the less mature ecosystem, while free energy will tend to move from the less mature to the more mature ecosystems. This largely corresponds with the later idea of ecological sources (mature) and sinks (immature) (Pulliam, 1988; Pulliam and Danielson, 1991; text box *New developments on the Margalef framework* below).

New developments on the Margalef framework — Margalef (1963) still considered ecosystems from a strict Clementian point of view with a single most mature climax community as the end result of succession, and the trend of undisturbed ecosystems always moving towards maturity (Rothauge, 2011a). Some rangeland ecologists however, have controversially argued that most arid and semi-arid ecosystems are inherently non-equilibrium systems and should be managed as such (Ellis and Swift, 1988; Westoby et al., 1989; Rothauge, 2011a). Such a system would be driven mostly by variable abiotic factors (e.g. rainfall and fire), so that biotic equilibrium and a "climax community" at carrying capacity will never be actualised and should therefore not be a management aim (McLeod, 1997). In the maturity framework of Margalef (1963), this would resemble a typical "immature" ecosystem with abiotic rather than biotic drivers. Westoby et al. (1989) proposed the state-and-transition model as a more useful model for rangeland management in such disequilibrium systems. They argued that ecosystems will still tend towards equilibrium, but that succession can result in more than one stable endpoint, depending on various transition factors (Westoby et al., 1989). Instead of a single mature climax equilibrium community, there are multiple possible equilibrium states for an ecosystem, some more mature *sensu* Margalef (1963) and some less mature or "immature". Only certain transitions would be possible between certain states, and not all transitions are reversible (see Rothauge, 2011a,b for an application of this to the management of Namibian veld). The theoretical foundation for this applied veld management model was actually already laid by Holling (1973) — see Ludwig et al. (1997) for an expanded exposition of sustainability, stability and resilience within this framework.

The correlation between biodiversity and ecosystem resilience and stability has long been known (Holling, 1973; Reiss et al., 2009, but see Landi et al., 2018). This correlation was proven mathematically by MacArthur (1955) who showed that food webs as energy converters were basically Markov chains. More recently it has been shown that ecosystem stability is caused by biodiversity, rather than the other way around (Cadotte et al., 2012; Hautier et al., 2015). In addition to biodiversity, other aspects of the ecosystem (e.g. connectivity, trophic levels, symmetry of interactions, node degree distribution, modularity and adaptive foraging[10]) also influence the stability of ecosystems (Landi et al., 2018). Grace et al. (2016) claimed that higher plant species richness also increases ecosystem productivity[11]. Therefore, our choice of conflict mitigation methods should promote ecological sustainability by causing the least disturbance to a healthy (biodiverse and productive) ecosystem. It cannot be assumed that any farm is currently in "pristine ecological condition" or in a state of maximum biodiversity (Figure 1.3 in Chapter 1) — however, minimizing disturbance is more likely to avoid an unintended transition to an undesirable state, while it is not known if transition to a more mature state is possible (without additional information on the current management practices and ecological state of the farm).

2.3.2.2 Rangeland management

While this principle of least disturbance and minimizing the ecological impact of a mitigation method works at the level of the whole ecosystem, there are other farm management aspects that influence ecological sustainability at only some of the trophic levels of the ecological pyramid (Lindeman, 1942). One ecological effect that was mentioned repeatedly as a possible disadvantage of some conflict mitigation methods above, is overgrazing or trampling of the veld, influencing the "primary producer" and "soil" levels of the pyramid (see Figure 1.4 in Chapter 1). Various ecological studies (MacArthur, 1955; Margalef, 1963; Caughley, 1976) have already shown that any natural grazing system will tend to one or more relatively persistent equilibrium states, since past

[10] E.g. MacArthur (1955) showed that a given stability can be achieved either by a large number of species, each with a fairly restricted diet (specialists), or by a smaller number of species, each eating a wide variety of other species (generalists).

[11] This would appear to contradict the maturity framework of Margalef (1963). However, the cause and effect relationship works in opposite directions. In the maturity framework, the "high productivity" of immature (sink) ecosystems is mostly caused by the frequent influx of young (immature) growing individuals, not by the primary productivity of the system itself. Additionally, it is the *ratio* between biomass and (free) energy that is biased towards energy in a more immature system, not the total productivity. Another way to view it is that the immature ecosystem *tend* to have individuals with an r strategy (high reproduction with few survivors) and the mature ecosystems to a K strategy (density-dependent equilibrium with lower reproduction and better survival) (but see Reznick et al., 2002 for a critique of the r vs K paradigm).

natural selection at the ecosystem level would get rid of non-persistent ecosystems. However, the productivity of this equilibrium endpoint will depend on the management of the herbivores and rangeland (Westoby et al., 1989; Tainton, 1999). In Namibia, two such negative end-points of grazing systems are desertification and bush encroachment, both leading to a long-term decline in sustainable stocking rates, with desertification probably irreversible (Rothauge, 2011a,b). De Klerk (2004) considered bush encroachment as another form of desertification. For savannah systems Rothauge (2011a) suggested a state-and-transition model with six basic states: 1) pioneer grassy state (immature) which can transition to 2) climax grassy state (mature), 3) to desertification (irreversible in human lifetimes) or 4) to bush seedling establishment, which can in turn transition back to either of the grassy states or 5) to a vigorous bushy (bush-encroached) state (immature), which can only slowly transition to 6) a senescent bushy state (mature), which can in turn transition back to either of the grassy states (see text box below: *Veld management within the state-and-transition rangeland model*). Once the immature bush-encroached state has been reached it can take decades for it to reach the senescent bushy state which can be managed again to return to a grassy state. The two critical states where the wrong management choice can have disastrous effects are the pioneer grassy state and the bush seedling establishment state (Rothauge, 2011b). Thus any conflict mitigation method that could impact the veld negatively, should be avoided if the system is in either of these two states. Hot, late winter, controlled fires or heavy goat or sheep browsing (with very light or no grazing) is required to prevent bush seedlings from transitioning to the bush-encroached state (see Rothauge, 2011b). Bush encroachment has been linked to an increase in leopard numbers and a decrease in cheetah numbers on Namibian farmlands (LCMAN Red Data List for Namibian Carnivores, in prep).

Veld management within the state-and-transition rangeland model — Typically veld can be classified as 1) sweet-veld, where the limiting factor for herbivore production is the *quantity* of grass available and is typical for the arid and semi-arid Southern and Western parts of Namibia, 2) sour-veld where the limiting factor is typically the *nutritional quality* and palatability of the veld (typical for higher rainfall grasslands) or a combination of the two in 3) mixed veld, common in Namibia's savannah vegetation types (Tainton, 1999; Rutherford et al., 2006).

Because mismanaged sweet veld tends towards desertification, overgrazing or trampling should be avoided. Using a mythical equilibrium "carrying capacity" as the basis for veld management is problematic, especially in a semi-arid region like Namibia with its variable rainfall (Luyt, 2004; Lubbe, 2005; Hixon, 2008). Caughley (1976) already mentioned that when the coefficient of variation of annual rainfall exceeds 30% (typical for semi-arid and arid environments), the concept of an ecological carrying capacity becomes illusionary (see also Casaway et al., 1996; Terborgh, 2015: "*Carrying capacity has thus evolved into a scientific ghost — something everyone thinks is important but that no one has ever seen.*"). Chapter 1 of Luyt (2004) explains the dangers of unnoticed veld degradation when using "carrying capacity" for veld management, especially in non-equilibrium systems. Variable abiotic factors like rainfall can mask the biotic interactions between vegetation, herbivores and carnivores and their long-term effects (see Margalef, 1963; Ellis and Swift, 1988; McLeod, 1997; Lubbe and Espach, 2006). An adaptive management approach, where livestock movements and numbers are continuously adapted to rainfall and current available grazing, is therefore advised (Westoby et al., 1989). Rothauge (2011a) gives more detail on how to manage the savannah (mixed veld) to avoid desertification and grassland degradation. The grass in sweet veld areas typically retain most of their above-ground nutritional value when dry, even for years (Tainton, 1999). It is thus a good practice to rest enough ungrazed camps to provide livestock with grazing through drought years in sweet veld areas.

Mismanaged sour veld tends to transition to a change in the species composition of the vegetation, like bush encroachment or less palatable grasses becoming dominant (Rothauge, 2011a,b). Mega-herbivores, fire and migrations (both seasonal and periodic), sustained the productivity of the veld in the past (Rohde and Hoffman, 2012). The migrations have mostly been halted by fences and the increase in permanent water points in Namibia (McGranahan, 2011; Rohde and Hoffman, 2012). Fire suppression in higher rainfall areas (>350 mm per year) is currently the greatest driver of bush encroachment (McGranahan, 2008; Rothauge, 2011b). For the central savannah vegetation, Rothauge (2011b) gave a detailed explanation and management system to prevent bush encroachment.

Champions of "holistic management" have claimed that herding mimics the natural anti-predator and migratory behaviour of wild herbivores and should thus be equally sustainable in avoiding veld degradation (Smuts, 1978; Bingham, 1997; Savory, 2013; Carter et al., 2014; see method 31 above on page 75). Veblen et al. (2016) showed that this assumption does not necessarily hold, since cattle can at best be considered as a substitute for maybe one species of tall grass grazer (e.g. buffalo) and not as a true replacement for the whole herbivore guild. "Holistic management", when defined as high-intensity short-duration grazing, has not been shown to be more beneficial to the veld compared to regular rotational grazing or similar management (Carter et al., 2014; Dicks et al., 2018) and it has the drawback of not allowing the livestock to select the most palatable and nutritious grazing (Cayre et al., 2018).

In practice, short duration, light grazing during the growing season just after the first rains, allowing the livestock to prune and stimulate the most palatable grasses, followed by heavy or longer duration grazing once the palatable grasses have seeded, has been shown to improve the relative abundance of more nutritious grasses, possibly more so than most alternative veld management practices (Hobbs, 1996; Tainton, 1999).

Any farmer-predator conflict mitigation method needs to be seen within this framework of rangeland management — the primary productivity of any livestock or game farm is based on its vegetation (Figure 1.3)

2.3.2.3 Top-down vs bottom-up control

Another ecosystem-wide property that may influence predation conflict, is whether it is mostly top-down or bottom-up controlled. This principally affects the two predator trophic levels of the ecological pyramid (Figure 1.4 on page 12). It has been shown that in a biodiverse ecosystem, both bottom-up and top-down processes are at work and that different predators have different effects on the ecosystem depending on the size of their prey (Sinclair et al., 2003, see also Bosc et al., 2018; Tambling et al., 2018). In an ecosystem where bottom-up processes dominate, the top predator densities are food controlled. High, year-round livestock densities may then result in abnormally high predator densities and a greater predation risk for livestock. On the other hand, if the ecosystem is dominated by top-down processes, apex predator densities are primarily determined by inter-specific and/or intra-specific competition (Balme et al., 2012a; López-Bao and Rodrigues, 2014). High prey numbers (including livestock) will then have little effect on densities of predators. The various mitigation methods discussed above may thus have different effects and success rates depending on whether the specific agroecosystem is dominated by bottom-up or top-down processes (e.g. in a top-down controlled ecosystem, removing territorial animals could have a much greater effect because of a breakdown in the territorial system, while in a bottom-up, food controlled ecosystem the densities are less likely to change because the food availability remains the same – Casaway et al., 1996; Ritchie and Johnson, 2009; Terborgh, 2015). Some predation control methods might even change the ecosystem from top-down controlled to bottom-up controlled (Estes et al., 2011; Terborgh, 2015). Moreover, as Sinclair et al. (2003) showed, in the same ecosystem different predator species can experience either top-down or bottom-up density control and the same species can be subject to either in different circumstances.

Predators above 14-20 kg are dependent on prey animals >45% of their own body size to meet their energetic costs (Carbone et al., 2007). While competition and intra-guild predation can complicate this, it would imply that predator densities for species above this transition weight are dependent on prey densities in these size classes and bottom-up processes, rather than top-down processes. This is an observation that seems to hold true for leopards (Stander et al., 1997c; Marker and Dickman, 2005a; Hayward et al., 2007b), but less clearly so for cheetahs (Durant, 1998). Jackals and caracals fall right in the critical weight class that can switch between larger (>45 %) and smaller prey and are thus much less likely to be prey density dependent (Melville et al., 2004; Carbone et al., 2007; Hayward et al., 2017). Sinclair et al. (2003) showed that in a biodiverse ecosystem, ungulates smaller than ∼150 kg are subject to top-down control by predators, while larger ungulates are controlled by bottom-up processes of food limitations. This was explained by the dietary niche overlap of top predators (lions and spotted hyaenas) with smaller predators, so that smaller prey were subject to predation by almost all predator species, while larger ungulates were only subject to lion predation. In Etosha National Park, Casaway et al. (1996) showed that in plains ungulates (blue wildebeest *Connochaetes taurinus*, plains zebra *Equus quagga*, springbok *Antidorcas marsupialis*, and oryx *Oryx gazella*) densities were constrained by predation and disease, rather than bottom-up food limitations. This can be explained at least partly by a breakdown in the historical annual migration of game around the pan that has been stopped by the northern park fence, forcing the ungulates to remain in the same areas and provide year-round food to resident predators.

Terborgh (2015) made a strong case that density-dependent predation and prey switching favours coexistence by reducing intra-guild competition at lower trophic levels (*cf.* also Estes et al., 2011). This, in turn, favours less common prey species compared to common species and thus higher α-diversity (at lower predation levels, competition causes niche compression and β-diversity becomes more important for overall biodiversity) — through this interaction between predation and competition, top-down forcing promotes higher biodiversity. In terms of the framework provided by Margalef (1963), more mature ecosystems are more likely to be dominated by top-down processes and less mature ecosystems by bottom-up processes and competition.

It needs to be determined which processes, bottom-up or top-down (or a combination of the two) are dominant in the different ecosystems on Namibian livestock farms. It is hypothesised that on the more biodiverse farmlands to the north of Windhoek, a combination of effects will be at play, with leopards being food and habitat

restricted (bottom-up), and in turn exerting top-down control on cheetahs, jackals and caracals. In the south which is less biodiverse, with few or no leopards and cheetahs, the ecosystems are probably mostly subject to bottom-up control and also more unstable (Cadotte et al., 2012; Hautier et al., 2015; Terborgh, 2015).

2.3.3 Predator ecology

In a Decision Support System, the user is part of the decision-making process. It is therefore important that farmers will be as well informed about the behavioural ecology of the four predator species as possible, since sometimes a seemingly unimportant behavioural adaptation can make the difference between success and failure (e.g. the importance of culling red foxes in the right season – Lieury et al., 2015, the fact that non-selective removal of coyotes had no effect on livestock depredation – Sacks et al., 1999; Sacks and Neale, 2002; Jaeger, 2004, the difference in translocation success between cheetahs and leopards – Weise et al., 2015a,b). The ecology of the four predator species are particularly relevant to those methods which can be limited by the predator species (Table 2.1).

2.3.3.1 *Canis mesomelas*

Distribution, conservation status and threats: Globally the black-backed jackal occurs in two discrete areas, separated by the Miombo woodland zone of approximately 900 km. *Canis mesomelas schmidti* occurs in East Africa and *Canis mesomelas mesomelas* in southern Africa. The southern African subspecies is found in parts of South Africa, Botswana, Angola, Zimbabwe and Mozambique and throughout Namibia (Estes, 1991; Hoffman, 2014). Black-backed jackals are found in a wide variety of habitats including arid coastal desert (Dreyer and Nel, 1990), montane grassland (Rowe-Rowe, 1982; Humphries et al., 2016), open savannah (Estes, 1991), woodland savannah mosaics (Loveridge and Macdonald, 2002) and are also common on farmland (Humphries et al., 2016; Swanepoel, 2016; Ray et al., 2005). Joubert and Mostert (1975) reported jackals as occurring on most commercial farmlands in Namibia. Where they occur together with side-striped jackals (*Canis adustus*), there is some niche partitioning, with the black-back jackal preferring the more open habitat and aggressively displacing the side-striped jackal from grassland (Loveridge and Macdonald, 2002).

In spite of heavy persecution on many livestock farms, black-backed jackals are still found throughout Namibia and are classified as "least concern" by the IUCN (Hoffman, 2014). In the past Shortridge (1934) and Hoffman (2014) had alleged that black-backed jackals were replaced by the side-striped jackal in the Northeast of Namibia, but recent remote camera evidence have placed them in the far Northeast (Zambezi region) of Namibia (Lisa Hanssen 2018, personal communication). This means that the current black-backed jackal range includes all of Namibia (LCMAN Red Data List for Namibian Carnivores, in prep).

Black-backed jackals have a long history of conflict with farmers of sheep and goats in the neighbouring South Africa (Beinart, 1998; Swanepoel, 2016). By contrast, in Namibia jackals were not considered a major livestock predator up to at least the 1930's, since most small-stock farmers kept their sheep in kraals at night, used herders and farmed mostly with Karakul sheep which did not need to raise their vulnerable lambs (Shortridge, 1934; Swanepoel, 2016). This has since changed, with the black-backed jackal now considered the main predator on small livestock in Namibia according to farmers (Joubert and Mostert, 1975; Potgieter, 2011).

Spatial ecology: Typically, jackals form territorial, monogamous breeding pairs (Ferguson et al., 1983). The territorial breeding pair often keeps their offspring from previous years around to act as helpers in raising the new litter of puppies (Estes, 1991). A resident pack (excluding the new puppies) can thus vary between two to eight per territory (one to six helpers) (Jenner et al., 2011). The breeding pair is territorial and aggressively defends their territory (Moehlman, 1987). In South Africa they have been found to occupy home ranges between 1.3 to 575 km^2 when including non-territorial, unmated individuals (Ferguson et al., 1983), with home ranges being

11.4 ± 4.3 km² per territorial male on farmlands in the Natal Highlands (Humphries et al., 2016), 14.7 km² per territorial pair in the Kalahari (Ferguson et al., 1983), 17.8 km² in the savannah near Kimberley (Kamler et al., 2012b), 18.2 km² in a reserve in the Drakensberg mountains and up to 27.7 km² in Gauteng and the North West Province (the latter two areas included farmland) (Ferguson et al., 1983). In south-eastern Botswana the average territory size was 15.9 km² (Kaunda, 2001). In Namibia, territoriality has only been studied along the Namib coast. Territory sizes around Cape Cross, a mainland seal colony, varied from 0.20–11.11 km² during the breeding season and were positively correlated with the distance to the food source (Jenner et al., 2011). Along the coast away from mainland seal colonies, where food patches are mostly small and widely separated, jackal group sizes are small, and territories are narrow and extremely elongated (Nel et al., 2013). Where food patches are rich, fairly clumped and also heterogeneous, group sizes are large and territory sizes small (Nel et al., 2013). Larger group sizes can improve the black-backed jackals' efficiency as hunters (*cf.* Estes, 1991; Jenner et al., 2011; Merrifield, 2012; Murray et al., 2012) by allowing them to take larger prey such as adult springboks (Krofel, 2007).

Although territorial breeding pairs will not allow other jackals to breed in their territories, they have been known to allow other non-related jackals to enter their territories when there is an abundant food source or a drinking water point (Ferguson et al., 1983). This has sometimes been interpreted as a breakdown of territoriality (Hiscocks and Perrin, 1988), but Jenner et al. (2011) showed that even when allowing outsiders in their territories, the visitors would attempt to avoid the territorial pair and the owners of the territory were aggressive until the visitors showed the appropriate submissive behaviour.

Breeding ecology: The breeding season is often synchronised with the main lambing season of their prey in spring (varying from August - December across southern Africa), enabling the jackals to provide high-quality food for the new pups (Estes, 1991; Klare et al., 2010; Kamler et al., 2012a; Tambling et al., 2018). An average of 4.6 pups (range one to eight) are born in dens after a gestation period of 60 days (Minnie et al., 2018, LCMAN Red Data List for Namibian Carnivores, in prep). Food is regurgitated by the parents and helpers for the puppies, and carried back to the den when they are older (Estes, 1991). Helpers also guard the pups and their presence increases overall pup survival (Moehlman, 1987). Pups are fully weaned by eight to nine weeks of age, when they start foraging together with their parents (Ferguson et al., 1983). Subadults reach sexual maturity at 11 months, but usually only start breeding at two years of age (Ferguson et al., 1983). They normally disperse at one year of age when not staying as helpers (Moehlman, 1987) and dispersal distances of up to 150 km have been recorded in South Africa (Ferguson et al., 1983; Humphries et al., 2016). Some dispersed offspring that have already set up a territory of their own, may occasionally return to their natal range to help their parents raise the next litter (Loveridge and Macdonald, 2001). In agreement with the theoretical framework of Margalef (1963), it has been found that jackals on farmlands are significantly younger (median of two to three years) than the jackals in conserved areas (median of five to six years) (Minnie et al., 2016b).

Feeding ecology Black-backed jackals are normally crepuscular in activity, but become predominately nocturnal on farmland where they are persecuted, and may be active during both the day and the night in undisturbed areas (Stuart, 1981; Hiscocks and Perrin, 1988; Kamler et al., 2012b; Minnie et al., 2018). They are opportunistic and eat whatever is seasonally available from plants, invertebrates and reptiles to birds and mammals (Estes, 1991; Stuart, 1976; Stuart and Shaughnessy, 1984; Kamler et al., 2012a). Klare et al. (2010) showed that on game farms in the Northern Cape near Kimberley, ungulates formed the major proportion of the black-backed jackal diet. They readily scavenge, but when hunting they show a preference for "hider" ungulate species, especially springbok, *Antidorcas marsupialis* (Klare et al., 2010; Kamler et al., 2012a). In years of good rainfall, they may feed extensively on insects (Hall-Martin and Botha, 1980), with a resulting reduction in predation on livestock in Namaqualand (personal observation, Corrie van der Westhuizen 2015, personal communication). When foraging, black-backed jackals respond to prey distress calls and will also often follow larger carnivores in order to scavenge (Bothma and Le Riche, 1984; Estes, 1991). They drink fresh water when

available, and will even leave their territory in order to find water (Kaunda, 2001). However, the widespread occurrence of black-backed jackals along the Namib Desert coast may indicate that they can get essential water from their food to survive. In this regard, Kolar (2005) hypothesised that the reason why jackals kill seal pups rather than eating available old carcasses, is because fresh carcasses contain more essential liquids. Stuart (1976) suggested that the relatively high vegetation content in their diet (7.3 – 14.3% in the central Namib desert) probably contributes a significant proportion of their need for liquids.

Black-backed jackals in Southern Africa overall have shown a preference for some specific "hider" ungulate species, like common duiker (*Sylvicapra grimmia*), bushbuck (*Tragelaphus scriptus*) and springbok (*Antidorcas marsupialis*), while taking livestock (mostly sheep) and other common prey species, like steenbok (*Raphicerus campestris*), rodents, oryx (*Oryx gazella*) and impala (*Aepyceros melampus*), according to availability (Hayward et al., 2017; Minnie et al., 2018). In taking sheep according to availability, they resemble coyotes in America, which generally also took sheep according to availability (Sacks and Neale, 2002). What is not known, is if black-backed jackals also show similar behaviour to coyotes wherein mostly alpha pairs kill livestock (Jaeger, 2004) and if it is specific individuals that are responsible for most livestock kills (Sacks et al., 1999). Drouilly et al. (2018) showed that in some sheep farming areas, black-backed jackals on farms actually started to prefer sheep as prey. Kamler et al. (2012a) previously showed that where natural "hider" prey ungulates are available on farms, they will be preferred over small livestock as prey.

Interspecific interactions: Black-backed jackals have a negative impact on Cape fox (*Vulpes chama*) and bat-eared fox (*Otocyon megalotis*) numbers, both through killing them and by excluding them from some areas (Kamler et al., 2012b, 2013). In turn, black-backed jackals may be preyed upon by caracals (*Caracal caracal*) (Melville et al., 2004), leopards (*Panthera pardus*) and brown hyaenas (*Parahyaena brunnea*) (Bothma and Le Riche, 1994b; Stein et al., 2013). However, black-backed jackals may also kill caracal cubs and the removal of black-backed jackals may result in an increase of caracal numbers (Pringle and Pringle, 1979).

Application to HWC: *Conservation status* – Black-backed jackals are wide-spread and common throughout Namibia with currently little danger to their continued survival as a species. There are thus no restrictions on using lethal or other predator control methods which might be incompatible with a threatened species.

Spatial ecology – Spatially black-backed jackals have fairly small territories on average and probably the highest densities of the four predator species examined. Moreover they can disperse over very long distances (Ferguson et al., 1983). This, together with their dietary flexibility, makes them less vulnerable to eradication attempts than most of the other predator species considered here (Stuart, 1982; Beinart, 1998; De Wet, 2002). The presence of sexually mature helpers in the territory of their dominant monogamous parents in some instances, who are prevented from breeding solely by the dominance of the breeding pair, makes it possible that these helpers might start breeding in the same area when either of the parents are killed — even in the absence of immigration from elsewhere, or of the neighbouring adults taking over the territory by expanding their own. This would support the observation that killing jackals is usually not successful in reducing livestock losses in the long term and in some areas actually increases the livestock losses in the following year (e.g. Conradie and Piesse, 2013 in the Ceres Karoo; Nattrass and Conradie, 2018).

Historically, the only circumstances where the hunting of black-backed jackals appeared to be at least partially successful in decreasing their numbers, had been when jackal-proof fencing were put up (with government subsidies), and all jackals were exterminated using district-wide "fence and clean-up" operations (Beinart, 1998; Nattrass and Conradie, 2015; Swanepoel, 2016). By effectively preventing the dispersal and recolonisation of those areas where jackals had been extirpated, farmers could change their farming methods to dispense with herding and kraaling, and to use rotational grazing in camps instead (see method 1 on page 47 above).

Breeding ecology – The findings of Minnie et al. (2016b) showed the average age of black-backed jackals on farms to be much lower than those of black-backed jackals in conserved areas and that they had litters at a

younger age. Minnie et al. (2016b) also found that the populations on the farms compensated for persecution by having larger litters than in the conserved areas. These are the kind of demographic changes we would expect as the population density is lowered sufficiently below the density-dependent K (ecological carrying capacity – see Figure 2.1 above) and starts to approach the economic carrying capacity of highest growth and reproduction rates. The data from Minnie et al. (2016b) suggests that jackals in a more mature ecosystem (without human persecution) largely self-regulate their numbers by having smaller litters and starting to breed later in life. The higher reproductive output of black-backed jackals on farmlands suggests that they are probably not dependent on immigration from the more mature conservation areas for their survival. This suggests that it is not primarily a permanent source-sink situation and that jackals could persist and perhaps increase in the absence of conserved areas close by because of compensatory breeding (Margalef, 1963; Pulliam, 1988; Pulliam and Danielson, 1991; Drouilly et al., 2018). This seems to be borne out by the widespread persistence of jackal numbers all over southern Namibia, where they have been heavily persecuted, far from any conserved areas (Swanepoel, 2016).

Feeding ecology – It is possible that providing jackals with alternative food sources can significantly reduce livestock predation (Avenant and Du Plessis, 2008). Knowing that jackals in their natural environment prefer springbok, for example, increasing the numbers of such wild prey could be an effective way to keep them from killing livestock for food (Minnie et al., 2018). Since sheep shows a "follower" type behaviour in contrast to springbok having a typical "hider" behaviour, it could explain why jackals prefer springbok lambs to sheep lambs when both are available (Klare et al., 2010; Kamler et al., 2012a). It is also in agreement with the findings of Hayward et al. (2017) that sheep is generally not a preferred prey species, while springbok is (*cf.* Kamler et al., 2012a). However, this will probably depend to some extent on the behaviour of the specific sheep breed and the size of the herds as well. If the preference for hider lambs is simply the result of higher encounter rates (rather than anti-predation herding behaviour), more springbok lambs might not decrease jackal predation of livestock, especially where sheep lambs are available throughout the year (e.g. Drouilly et al., 2018).

Research on the related American coyote (*Canis latrans*) has shown that it is mostly the alpha breeding pair that kill livestock (Sacks et al., 1999; Jaeger, 2004). This makes sense if the main reason why they kill livestock, is to feed their pups. If black-backed jackals show a similar pattern, changing lambing seasons to be outside the jackal breeding season could significantly reduce livestock losses. Furthermore, the research by Sacks et al. (1999) showed that it was specific individuals that were responsible for almost all livestock losses. If jackals display the same behaviour, it would suggest that selective removal of specific, individual jackals could indeed reduce livestock losses.

Although mostly taking rodents according to availability, explosions in rodent numbers and resulting damage to grazing have also been reported where jackals had been extirpated on the edges of the Kalahari (Beinart, 1998, *cf.* Minnie et al., 2016a).

Interspecific interactions – Competition and mutual exclusion between jackals and caracals have allegedly been used with success to keep jackals away from livestock through the use of biological repellents (caracal scat and release of some caracals on the land) to keep jackals away from kraals or camps with young vulnerable livestock (Holmes and Holmes, 2006). Caracals generally show a lower preference for livestock than jackals, but the scat would probably also keep away actual caracals (Jansen, 2016; Drouilly et al., 2018).

There is anecdotal evidence that like wolves and pumas (Bartnick et al., 2013), some niche partitioning happens between jackals and caracals with jackals preferring open, flatter habitat and caracals preferring more mountainous and denser habitat, although further research is needed to confirm this (Minnie et al., 2018). Such difference in habitat preferences between caracals and jackals can be exploited by farmers to keep their livestock away from those areas where the livestock are most vulnerable to one species or the other – if most losses are because of a single species (e.g. if only one species has enough alternative prey). Even without niche partitioning, the competition and effect of "fear and loathing" (Ritchie and Johnson, 2009) between caracals and jackals are likely to keep their densities (and potential for conflict with livestock owners) lower than if only one of the two species were there. Other predators that prey on jackals, like leopards, are also likely to keep jackal densities

lower (Bothma and Le Riche, 1994b). This might be part of the reason for the demographic differences of jackal populations between conserved areas and farmlands noticed by Minnie et al. (2016b). Livestock guarding dogs urinating and scent-marking, could have a similar effect in reducing jackal densities or restricting their spatial behaviour (Hansen, 2005; Smuts, 2008).

The unwanted side-effect of the historical extermination of black-backed jackals from some districts in South Africa, was an increase in caracals and consequent livestock losses, since jackal-proof fencing is no barrier to caracal movement and the lack of competition enabled caracals to increase or colonise areas from which they had been excluded by jackals before (Pringle and Pringle, 1979).

Research required: Given the importance of black-backed jackals as livestock predators in Namibia, it was surprising that no research about their ecology on Namibian farms has been published yet (Potgieter, 2011; Swanepoel, 2016).

Spatial ecology – One important question raised by Du Plessis et al. (2015) that has not been addressed at all on Namibian farmlands is how the spatial behaviour of jackals in areas where they are not persecuted and with other predators present in a more biodiverse ecosystem, compares to farmlands where they are heavily persecuted. It is not known if the higher reproductive output on farmlands with a frequent removal of territorial animals and mesopredator release because of the absence of larger predators, result in a higher density of jackals on farmlands than in conserved areas. Even the sizes of black-backed jackal territories in Namibia are unknown, except for the highly unusual ecotone on the Namib Desert coast (Du Plessis et al., 2015). No comparison is possible if both quantities are unknown. All the relevant aspects of black-backed jackal spatial behaviour on Namibian farmlands are currently unknown:

1. Home range sizes of breeding territorial pairs are currently unknown (although the range will probably be similar to that of South African findings, since the range in home range sizes is not very large for most of South Africa – it is unlikely to be anything like the home range sizes found in the Namib and the coast).
2. The ratio between non-resident dispersing young and residential groups, as well as group sizes on farms are unknown. Once known, the reciprocal of territory size multiplied by group size could simultaneously provide some indication of resident jackal density and expected prey requirements (potential human conflict), taking into account uncertainty about group sizes because of their flexible territorial behaviour.
3. We have no idea if there are hotspots of jackal activity within their home ranges and in what kind of habitat such hotspots would be found.
4. We have no idea if jackals prefer different habitat types for different purposes.
5. While we have some idea of jackal dispersal distances in South Africa, it is unknown how far jackals in Namibia typically disperse in farming areas (although it is probably similar).

Breeding ecology – While it could be expected that a similar difference in breeding age and litter size as in South Africa will be found between black-backed jackals on Namibian farms and conservation areas (Minnie et al., 2016b), the differences between the two countries are large enough that it is unlikely. Cattle farming is more common in much of central and northern Namibia, with many farmers not actively persecuting jackals. Differences in breeding ecology (if any) are therefore more likely to be between southern and northern Namibia. In either case, the question of Du Plessis et al. (2015) on possible compensatory breeding remains unanswered.

Feeding ecology – It is unknown whether alpha breeding pairs are responsible for most jackal livestock predation events or perhaps food-stressed non-territorial dispersers. Similarly, we don't know if, like coyotes, certain individual black-backed jackals are responsible for most livestock losses. In general, the prey preferences of jackals on Namibian farms and in conserved areas are both unknown.

Interspecific interactions – Jackals have been seen to follow leopard prey drag marks (personal observation). It is known that jackals are killed by leopards (Bothma and Le Riche, 1984). It is therefore not even certain if jackals benefit by leopards or avoid them. Nothing definite is known of the effect of leopards on jackal numbers or distribution on Namibian farms. Similarly, while there are plenty of anecdotal evidence of black-backed jackals killing caracal kittens and caracals killing jackal pups, it is still unknown if this actually results in spatial niche partitioning between the two species.

2.3.3.2 *Caracal caracal*

Distribution, conservation status and threats: Caracals occur in semi-arid savannahs and bushveld throughout Africa and into the Near East all the way to India and the Karakum desert in Russia (Estes, 1991). They are found over most of Africa with the exception of much of the central Sahara and the equatorial forest belt (Avgan et al., 2016). Caracals have a broad range of habitats including dry woodlands, acacia scrub, savannah, arid mountain areas to 2500 m high and hilly steppe. They are often associated with ecotones where forests and grasslands meet, and although they may use open grasslands at night, they require access to rocks and bushes for daytime rest spots (Ray et al., 2005). They may be replaced by servals (*Leptailurus serval*) in wetter areas with dominant grass cover through the year (Estes, 1991).

In Namibia, their distribution does not seem to have changed much. Shortridge (1934) recorded them as occurring throughout Namibia. The IUCN distribution map (Avgan et al., 2016) does not include the Namib Coast as part of their distribution range, but they have been recorded all the way to the coast (LCMAN Red Data List for Namibian Carnivores, in prep).

Like jackals, caracals are heavily persecuted by farmers in small livestock farming areas. This persecution does not currently appear to be a serious threat to their survival, and the IUCN still classify caracals as "Least Concern" (Avgan et al., 2016).

Spatial ecology: Caracals exhibit typical felid spatial behaviour with solitary territorial adults. Males have territories about three to four times larger than and overlapping several smaller female territories (Estes, 1991; Avenant and Nel, 1998). They will defend their territories against any trespassers of the same sex, except for subadult young that are with their territorial mothers (Estes, 1991). Throughout their range, the territory size of caracals (and thus density of residents) varies considerably, from 5.5 km^2 for females in South Africa (Moolman, 1986 in Bothma, 2012) to 1116 km^2 for a male in Arabia (Van Heezik and Seddon, 1998). Bothma and Le Riche (1994a) mentioned that the size of caracal home ranges are determined mostly by hunting success, which is in turn determined by prey density and cover in the habitat (Avenant and Nel, 1998). More productive, high-rainfall habitats, like the south and south west coastal areas of South Africa can thus be expected to have smaller home ranges than more arid areas with lower prey densities and less cover (Stuart, 1982; Bothma and Le Riche, 1994a; Avenant and Nel, 1998; Van Heezik and Seddon, 1998; Marker and Dickman, 2005b; Bothma, 2012). This pattern can be seen from the average caracal home ranges in the literature: 5.5 km^2 – 19.1 km^2 in the Eastern Cape, 7.39 km^2 – 48 km^2 in the South-Western Cape, 308.4 km^2 in the southern Kalahari (single male), 79.3 km^2 – 439.8 km^2 in North-Central Namibia (males only) to 1 116 km^2 for a single male in the arid Arabia (Bothma, 2012; Avenant and Nel, 1998; Stuart, 1982; Bothma and Le Riche, 1994a; Marker and Dickman, 2005b; Van Heezik and Seddon, 1998). Caracal territory size therefore shows greater variation between different habitat types (5.5 – 1116 km^2) than black-backed jackals (1.3 – 27.7 km^2). Dispersing non-territorial individuals show similar ranges for both species: 126 km (575 km^2) for dispersing jackals (Ferguson et al., 1983) and 138 km moved by a dispersing caracal male (Stuart, 1982).

Like for leopards, it was long assumed that males were more mobile due to their greater territory sizes (Stuart, 1982; Avenant and Nel, 1998). However, it is more likely that males tend to move in a relatively straight line along the outside of their territories patrolling, while females criss-cross, using more of the inside of their

territories, and both sexes actually move the same daily distance, like leopards (Martins, 2010). It makes sense that the females, having to feed their young, also have a better knowledge of their smaller territories in terms of hunting sites and prey availability than the males, but there has been no research to show this for caracals. The research by Moolman (1986, in Bothma, 2012) showed larger home ranges for caracals on farms than in the conserved area. This can be explained either by lower prey availability (or less cover) on farmlands or as a result of hunting pressure by farmers.

Breeding ecology: Caracals are polyestrous and may have kittens at any time of the year, although there is a birth peak during October to February in South Africa (Stuart, 1982; Bernard and Stuart, 1987). The average number of kittens is 2.2 and the female caracal raises the family alone (Estes, 1991). Females with small kittens must utilize smaller areas than males because of their restricted mobility in this time. Females must bring food to kittens which are left behind while the female hunts. At between 12 to 14 months the kittens are sexually mature and typically disperse to find their own new territory (Estes, 1991).

Feeding ecology: Caracals are naturally nocturnal, independent of persecution by humans. Unlike the serval which specializes on rodents, the caracal has a wide dietary range and can kill prey more than twice their own size using a throat bite. Smaller prey are often killed by a bite to the nape of the neck (Stuart, 1982; Roberts, 1986). Caracals generally feed on prey that weighs less than 5 kg, such as hares (*Lepus* spp.), hyrax (*Procavia capensis*), rodents, and birds (Palmer and Fairall, 1988; Melville and Bothma, 2006). However, they will take prey well over 15 kg, including adult impalas, springbok, a sitting ostrich (*Struthio camelus*), and even young kudus (*Tragelaphus strepsiceros*), if the opportunity presents itself (Shortridge, 1934; Grobler, 1981; Moolman, 1984; Estes, 1991). In terms of absolute numbers, rodents typically form the greatest part of their diet, but in terms of biomass, ungulates (livestock and antelopes) make the most important contribution to their diet (Stuart, 1982). In stressful situations like droughts or cold winters, when smaller prey like rodents decrease, large mammalian prey becomes relatively more important in their diet (Palmer and Fairall, 1988; Melville et al., 2004). Unlike black-backed jackals, insects, plants (berries) and carrion are insignificant in the caracal diet, and in this sense they are specialist hunters (Drouilly et al., 2018). However, for those species falling within their (wide) preferred size class, they generally take prey according to seasonal availability and can thus be classified as generalist predators (Moolman, 1984; Avenant and Nel, 2002; Minnie et al., 2018).

Interspecific interactions: Caracals are in direct competition with black-backed jackals and both species kill each other's young ones (Melville et al., 2004; Ray et al., 2005; Bothma, 2012; Tambling et al., 2018). It is possible that there is some level of niche partition with caracals preferring thicker bush and mountainous areas and black-backed jackals preferring the more open and flat plains, but more research is needed on the interactions between the two species (Minnie et al., 2018). Caracals will readily hunt other small carnivores such as African wildcat (*Felis sylvestris*), Cape grey mongoose (*Galerella pulverulenta*), slender mongoose (*Galerella sanguinea*), yellow mongoose (*Cynictis penicillata*), genets (*Genetta* spp.), otters (*Aonyx capensis*/*Hydrictis maculicollis*), polecat (*Ictonyx striatus*), suricate (*Suricata suricatta*), black-backed jackal (*Canis mesomelas*), Cape fox (*Vulpes chama*) and bat-eared fox (*Otocyon megalotis*) (Melville and Bothma, 2006; Avenant et al., 2016).

Application to HWC: *Conservation status* – Because caracals are widespread throughout Namibia with no evidence that their numbers are decreasing, there are no restrictions on using lethal or other predator control methods which would be incompatible with a threatened species, to mitigate human-caracal conflict.

Spatial ecology – The long dispersal distances of caracals would suggest that methods that aim at local extermination are less likely to succeed in the long run. However, the size of home their home ranges imply that the same caracal could often be found on more than one farm and that they would naturally be found at lower

densities than black-backed jackals. Therefore, most lethal methods are likely to have a bigger effect on the caracal population and territorial structure than on jackals if both occur in the same area. Like pumas, it is possible that when territorial individuals are removed, they will be replaced by younger individuals with smaller home ranges and greater densities (Linnell et al., 1996, see box under method 5 on page 53).

The differences between caracals and black-backed jackals in habitat preferences and the possibility of niche contraction (Bartnick et al., 2013; Tambling et al., 2018), as well as the (slight) difference in preferred livestock prey age class (caracals normally preferring adult small livestock and jackals preferring lambs – Roberts, 1986) could be used by farmers to decrease livestock losses (see description under "Canis mesomelas" on page 95 above).

Breeding ecology – Unlike black-backed jackals that have specific breeding seasons which can be used for temporal avoidance of predation (see above), the fact that caracals don't have specific breeding seasons precludes this use of seasonality. Irrespective of this, it is doubtful that any caracal predation on livestock is caused by the need to feed their young, since they generally take prey throughout the year according to availability (Avenant and Nel, 2002; Drouilly et al., 2018).

It is known that female caracal territories decrease when they have young (Avenant and Nel, 1998; Minnie et al., 2018). If the denning area of a female caracal with young can be identified, excluding livestock from that area could reduce predation. However, it is unknown how surrounding caracals react to such a range reduction.

Feeding ecology – As a species, caracals are generalists (*cf. Panthera pardus* below), taking prey according to availability. It is unknown if individual caracals might differ in their prey preferences. If particular caracals have particular prey preferences, it would indicate that some individuals might learn to preferentially prey on livestock. The lack of any such evidence, means that any method that depends on the removal of "problem individuals" is unlikely to succeed with caracals.

A number of studies appear to support the idea that caracals switch to larger prey like livestock when smaller prey like rodents, hyraxes and lagomorphs become scarcer. This would imply that keeping high numbers of these smaller prey animals on farmlands, might be an effective method to reduce livestock depredation by caracals (Avenant and Du Plessis, 2008). The hunting of small mammals by people could therefore lead to an increase in livestock losses.

Caracals, like many felids, are known for surplus killing where they kill many more livestock than they can eat, in a single hunt (Stuart, 1981; Ray et al., 2005). This happens most often where prey are cornered in a camp or kraal and is thus the result of an unnatural situation for the hunting cat (Ray et al., 2005). Such surplus killing are probably responsible for a large part of the conflict involving caracals and can be avoided by either keeping livestock in predator-proof kraals, not having water points (where livestock often rests) next to fences, or keeping livestock in larger herds with livestock guarding animals and sleeping in the open, away from any fences.

Interspecific interactions – Removal of caracals could potentially cause an increase in black-backed jackal numbers and a resulting increase in livestock depredation by jackals (Du Plessis, 2013; Tambling et al., 2018). The mechanism for the year-lagged increase in livestock losses following removal of predators reported by Conradie and Piesse (2013) is still unknown, but this increase in jackal numbers could be responsible (jackals typically occur at higher densities than caracals – see spatial behaviour above).

Alternative hypotheses to explain lagged increase in livestock depredation following removal of predators (Conradie and Piesse, 2013) – It should be mentioned that this counter-productive effect of predator removal is not always seen (e.g. Gunter, 2008; Bailey and Conradie, 2013). See also text box under method 5 on page 53. The reasons for this effect could be either direct by changing the behaviour of the specific predator species, or indirect, by causing more instability in the ecosystem and a decrease in the maturity of the ecosystem (Margalef, 1963):

- The indirect effect of disturbing the ecosystem by pushing it towards a state of lower maturity, with the accompanied effects including less stability, more young (non-territorial) animals and thus greater livestock depredation, since it is often young individuals that cannot hunt their natural prey effectively or old individuals that can no longer hunt their natural prey effectively, that turn to livestock (personal observation) (Minnie et al., 2016b).
- The direct effect of killing dominant territorial jackals or caracals, allowing younger individuals to set up territories which are smaller, since they have less confidence in defending larger territories, resulting in a higher predator density (*cf.* Linnell et al., 1996).
- Killing one territorial animal whose territory might have covered most of the farm, can allow neighbouring individuals to expand their territories so that instead of a single territory on most of the farm, the farm becomes the meeting point of various neighbouring territories and thus higher predator density (Baker et al., 2000; Fattebert et al., 2016).
- It is possible that caracals are easier to catch and kill than jackals (or the opposite in some habitats) and that the change in the natural balance of the two species result in an explosion in the population of the other species because of competitive release (Connell, 1961; Tambling et al., 2018).
- Compensatory breeding by caracals and jackals (larger litters at a younger age) is a possible reaction to persecution (Minnie et al., 2016b). This could be part of the general shift from a more mature ecosystem to a less mature system (Margalef, 1963).

Research required: *Conservation status* – While caracals are still widespread over all of Namibia, there has been some concern that their numbers are decreasing (Ruben Portas 2018, personal communication). No study has ever been done in Namibia to estimate caracal population size and they are heavily persecuted in some areas.

Spatial ecology – Since only a single study on caracal has been published in a single location in Namibia and using males only, all basic aspects of their spatial ecology on Namibian farmlands are still unknown (see research required for jackals above).

Breeding ecology – It is unknown if caracals demonstrate any compensatory breeding due to persecution on Namibian farms – the questions of Du Plessis et al. (2015).

Feeding ecology – The most important currently unknown fact about the feeding ecology of caracals is if there exists any prey specialisation by individuals.

2.3.3.3 *Acinonyx jubatus*

Distribution, conservation status and threats: Cheetahs occur globally in 33 populations found in 20 countries of Africa and the Near East (Durant et al., 2017). The discontinuous populations are found throughout southern and eastern Africa, along a belt on the southern margin of the Sahara, and in isolated, small and fragmented populations in North Africa. The last of the Asian cheetahs are found in Iran (Durant et al., 2015, 2017) with cheetahs now confined to 9% of their historical global distribution range (Durant et al.,

2017). The largest remaining viable populations of free-ranging cheetahs are in southern Africa, estimated to be approximately 4 000 individuals, of which less than 25% are in protected areas (Durant et al., 2017; Weise et al., 2017). The Namibian cheetah is part of this southern African subspecies, *Acinonyx jubatus jubatus*. Their original range included all kinds of habitats except equatorial forests and true deserts (Estes, 1991). In southern Africa, cheetahs are found primarily in the savannah biome composed of grass, bush, and wooded areas (Rutherford et al., 2006; Bissett and Bernard, 2007; Mills et al., 2004). Habitat selection of cheetahs has been explained by visibility (ability of cheetah to spot prey), prey availability and large predator avoidance (Hayward et al., 2006b; Marker et al., 2008; Muntifering et al., 2006) with habitat possibly being more important than prey density (Nghikembua et al., 2016). In Namibia, the preferred cheetah habitat has likely decreased through bush encroachment (Muntifering et al., 2006; De Klerk, 2004; Nghikembua et al., 2016), but whether this has consequences for the cheetahs in terms of survival has not been shown.

Historically, cheetahs were distributed widely throughout Namibia (Shortridge, 1934; Joubert and Mostert, 1975; Marker-Kraus et al., 1996), with population densities varying based on rainfall, habitat cover, prey availability and density of competitors. Cheetahs were extirpated from the southern part of the country, but they are still found widely throughout the central and north-western parts of the country (Weise et al., 2017). In 2015, the western and south-western Namib was designated as transient range for cheetahs (RWCP and IUCN/SSC, 2015). During 2016, intensive research in these areas confirmed a resident, reproducing population of desert adapted cheetahs (LCMAN Red Data List for Namibian Carnivores, in prep). Little is known about the distribution of cheetahs in the southern part of the country, in the north-western corner and in the communal land located in the north-east of Namibia (LCMAN Red Data List for Namibian Carnivores, in prep). Cheetah densities in Namibia have been estimated between 0.1 cheetahs/100 km^2 in the Namib to 3.0 cheetahs/100 km^2 (LCMAN Red Data List for Namibian Carnivores, in prep).

Globally, the cheetah is classified as vulnerable by the IUCN (Durant et al., 2015). Durant et al. (2017) showed that 77% of cheetahs in their global range occur outside protected areas, where conflict with farmers of game and livestock are their major threat to survival (Durant et al., 2015; RWCP and IUCN/SSC, 2015). Conservation actions based on protected areas only, are therefore unlikely to ensure the survival of many species like cheetahs, where the majority of populations are found outside protected areas. In the light of this, cheetahs in Namibia, where 90% occur on farmlands outside formally protected areas (Marker, 2000a; Marker et al., 2003a), have recently been re-classified as endangered in Namibia (LCMAN Red Data List for Namibian Carnivores, in prep). In view of this, calls have been made that the global threat status of cheetahs be updated to endangered too (Weise et al., 2017; Durant et al., 2017). In addition to human-wildlife conflict as the major threat, the decline of habitat or prey availability (due to bush encroachment), fragmentation of habitat due to fencing, intra-guild competition with other predator species, trophy hunting skewing the population sex ratio, legal and illegal trade and the keeping of cheetahs as pets, road mortalities and possibly vulnerability to disease due to high genetic homogeneity are additional threats to cheetah survival (Ray et al., 2005; LCMAN Red Data List for Namibian Carnivores, in prep).

Spatial ecology: Cheetah females are solitary, with large home ranges, while males can be either solitary of form coalitions of typically two to four animals (Eaton, 1974; Durant et al., 2015; Melzheimer et al., 2018). Female cheetahs have been seen with two litters of different ages, so they may occasionally be joined by one or more males when in oestrus, while still having their cubs with them (McVittie, 1979). Excluding lions, cheetahs are the most social of the felids (Estes, 1991; Mills, 1998). The dispersing young of the same litter often stay together in a groups after leaving their mother, and will sometimes meet up with her again for a few days or even temporarily with adult males (Stander, 1990b). As they reach sexual maturity, the females in the litter will one by one leave their brothers, who will often stick together, sometimes joined by unrelated males to form a male coalition (Eaton, 1974; McVittie, 1979; Estes, 1991). Reversing the pattern seen in most carnivores, cheetah females typically have larger home ranges with extensive overlap between them, than the male territories (561.3 - 2 323.9 km^2 reported for females; 514.5 - 862.5 km^2 reported for coalition males in

Marker et al., 2008). Large female home ranges have been explained either by a low prey density (e.g. in the southern Kalahari) or by migratory prey being followed by the cheetahs (e.g. in the Serengeti) (Mills, 1998), neither of which appears to be true for Namibian cheetahs on farmlands – prey densities on farmlands where cheetahs are found, are fairly high, and because of the provisioning of permanent water points by farmers, most prey species are no longer migratory (Marker-Kraus et al., 1996; Marker, 2002; Lindsey et al., 2013c). Males typically set up their territories in areas of high preferred prey density and good hunting conditions that will be favoured by females (Caro and Collins, 1987). Two strategies are followed by male cheetahs: they either become territorial, defending their territory and even killing trespassing males (Hunter and Skinner, 1995; Marker et al., 2008), or they persist as "floaters" with home ranges even larger than that of some females (up to 2 205.4 km^2 reported by Marker et al., 2008). Typically larger and more mature male cheetahs in coalitions are much more likely to set up territories than smaller and single cheetahs (Caro and Collins, 1986, 1987; Melzheimer et al., 2018). The "floating strategy" can thus be interpreted as the result of competition for territory, forcing these males to keep on searching for a territory until an opportunity presents itself (Melzheimer et al., 2018). If most territories in the area where they were released, had already been taken by established larger coalitions, this would explain the large and overlapping male home ranges found by Marker et al. (2008), since it is very unlikely that a smaller coalition or single male will ever take over the territory of a larger coalition (Caro and Collins, 1987). In times of drought when there are too few prey animals around, territorial males may leave their territories temporarily (Caro and Collins, 1987; Estes, 1991; Marker et al., 2008), which could also contribute to the large male home ranges observed in Namibia. On the other hand, the more even distribution of prey (and hunting habitat) in the marula-knobthorn savannah of the Kruger National Park led to male territories and female home ranges being of similar size, which provides an alternative explanation for the similarity in home range sizes between male and female cheetahs in Namibia (Mills, 1998).

Territorial males will scent-mark prominent features of the environment, such as rocks, bushes or trees (Caro and Collins, 1986). In Namibia they use trees with good visibility known as marking trees (or commonly referred to as "speelbome", which directly translate into "play trees") (Walker et al., 2016), which are typically near the junction of several territories (McVittie, 1979). The trees act as general communication stations, allowing cheetahs with their overlapping home ranges to establish a time-share approach to territorial spacing (Walker et al., 2016). Females also increasingly scent mark at these trees as they go into oestrus (Estes, 1991). In light of the social and spatial structure described above, females may mate with more than one male (Caro and Collins, 1987), and unlike most big cats infanticide by males is also almost non-existent among cheetahs (Hunter, 1998).

Breeding ecology: Female cheetahs first conceive at 21-22 months and give birth after a gestation period of three months (Estes, 1991; Durant et al., 2015). They give birth throughout the year with litters of one to six cubs and a mean litter size of 3.4 (Mills and Mills, 2014). On the farmlands of central Namibia they have several birth peaks distributed throughout the year: in the rainy season in February and March, in the cold dry season of June and July, and in the hot dry season in October and November, but the reasons for these peaks are unknown (Marker-Kraus et al., 1996; Marker et al., 2003a). At about 17-23 months the young cheetahs become independent and leave their mother who can mate again 18 months after their birth (Estes, 1991). Young independent males typically disperse over large distances and have been reported to move up to 200 km from their natal range before settling down (Marker et al., 2008).

Feeding ecology: Cheetahs are known to be diurnal (Estes, 1991) with most hunting and kills happening in the morning between 07h00 and 10h00, and late afternoon between 16h00 and 18h00 in East African protected areas (Cooper et al., 2007). They probably hunt at these times partly to avoid kleptoparasitism (i.e. the stealing of their prey by other carnivores) (Durant, 1998; Hayward and Slotow, 2009; Scantlebury et al., 2014; Dröge et al., 2017). However, in the Okavango Delta, Cozzi et al. (2012) have shown that cheetahs spend on average 25% of their daily energy budget at night. They (Cozzi et al., 2012) found that on moonlit nights, up to 40% of their daily activity happened at night in the presence of lions and hyaenas. This strongly suggests that

their hunting method of using speed and sight, was the main reason for their diurnal activity in the Okavango Delta, rather than predator avoidance. On Namibian farmlands cheetahs are also known to be more nocturnal and crepuscular in their activity, probably due to persecution by humans (Marker, 2002; Wachter et al., 2006; Fabiano, 2013), although they may also show peak activity in the morning and late afternoon (Nghikembua et al., 2016).

Unlike leopards that are quintessential ambush predators, cheetahs hunt using their exceptional speed. The cheetah is the fastest land animal with a maximum recorded speed of 104.4 km/h (29 m s^{-1}) (Sharp, 1997). However, contrary to what could be expected, their top speed while hunting has been recorded as 93 km/h (25.9 m s^{-1}) and on average they showed a top speed of only 14.9 m s^{-1} (53,6 km/h) during a hunt (Wilson et al., 2013).

The preferred prey of cheetahs is often the most abundant medium-sized antelope with a body mass range of 23–56 kg and a peak (mode) at 36 kg (Mills, 1998; Hayward et al., 2006b). This agrees with the daily energy expenditure (DEE) model by Carbone et al. (2007) which predicts that larger carnivores need to consume prey close to their own size (> 45% of their body mass) in order to survive. This contrasts with jackals and caracals which are right at that body size range that can switch between much smaller prey and prey >45% of their own body weight (Carbone et al., 2007; *cf.* the feeding ecology descriptions for both species above). The preferred cheetah prey species include blesbok (*Damaliscus pygargus phillipsi*), impala (*Aepyceros melampus*), Thomson's gazelle (*Eudorcas thomsonii*), Grant's gazelle (*Nanger granti*), and springbok (*Antidorcas marsupialis*) (Hayward et al., 2006b), but preliminary analysis showed that on Namibian farms, greater kudu (*Tragelaphus strepsiceros*) are the prey species most often taken (Marker et al., 2003d). This confirms the hypothesis that cheetahs show diet niche expansion in areas like Namibian farmlands without other large predators – here their diet includes prey species that are larger than what they normally take where larger predators are present (McVittie, 1979). It is not known if the higher number of kudu in the diet of Namibia's free-roaming cheetahs is simply because of high availability, or because of actual preference. This question cannot be answered without better data on prey species availability (Hayward et al., 2006b). However, it *is* known that livestock comprises about two thirds of the available prey on farmlands, but is found in only 6.4% of cheetah scat (Marker-Kraus et al., 1996). This shows a strong preference for natural prey by cheetahs, possibly because the human presence associated with livestock poses a risk. Thus the relative avoidance of livestock as prey, can be explained as typical cheetah behaviour of avoiding competing predators (humans in this case – see e.g. Hunter et al., 2007).

Eaton (1970) clarified that the number of prey per availability indices do not necessarily show a preference (choice of prey to hunt), but sometimes just show hunting success rate. More importantly for human-wildlife conflict, different cheetah groups have different apparent prey preferences with coalition males typically killing significantly larger prey than single females (Eaton, 1970; Mills et al., 2004; Bissett and Bernard, 2007). This means that different cheetahs (or social groups of cheetahs that hunt together) can differ in their prey preferences and that problem individuals (or coalitions) that specialise on livestock can indeed exist (*cf.* Linnell et al., 1999).

In the *Acacia* savannah of Botswana, cheetah groups generally preferred to hunt in open areas (48%, 176 of 367 runs, 28% of runs in open shrub / around large trees, and 24% in dense vegetation), but had more success in heavy bush areas adjacent to open plains or savannah (20% success in open grasslands vs 31% in dense cover) (Wilson et al., 2013) a pattern already observed by Eaton (1970). This preference for the edges between bush and open grassland (Durant, 1998; Muntifering et al., 2006; Bissett and Bernard, 2007; Nghikembua et al., 2016) is partly explained by them being able to get closer to their prey during the stalking phase of their hunt before charging out in the open where they can use their superior speed to catch their prey (Mills et al., 2004). But it is also explained by the findings of Scantlebury et al. (2014) that cheetahs spend most of their daily energy expenditure searching for prey rather than on the actual hunt. Even if their hunting success is greater in bushy habitat (and not by much), it is generally easier to find prey on the open plains than in thick bush. Bush next to open grassland can also decrease kleptoparasitism and help them hide their cubs from intra-guild predation by competing carnivores (Durant, 2000; Mills and Mills, 2014).

Interspecific interactions: Cheetahs, particularly cubs, are subject to intra-guild predation by other large carnivores such as lions, spotted hyaenas, leopards and even occasionally wild dogs and many aspects of their ecology is explained by this (Durant, 1998; Hayward and Slotow, 2009; Durant et al., 2015). Their physiological and anatomical adaptations for speed also make them lighter, weaker and with smaller teeth than their competitors. Combined with their mostly solitary live-style and need for speed to hunt successfully, it means that they cannot afford injuries that will curtail their hunting ability. For this reason, they will usually abandon any prey when threatened by large predators and seldom, if ever, return to old kills. Scavenging also becomes unattractive in the light of possible conflict with other carnivores (Estes, 1991). Cheetahs, especially cubs, are also vulnerable to predation by other large carnivores such as lions, spotted hyenas, leopards and possibly jackals and honey badgers (Caro, 1987; Hayward et al., 2006a; Mills and Mills, 2014). In Namibia, several cases of leopards killing adult cheetahs have been reported in the central farmlands (personal observation).

Application to HWC: *Conservation status* – Because of the huge home ranges of cheetahs (i.e. the same cheetah or group of cheetahs seen on multiple farms), Namibian farmers often over-estimate the real number of cheetahs in Namibia and there is the perception among many Namibian farmers that cheetahs are common. It is therefore important to highlight the global dire conservation picture for this species (Marker-Kraus and Kraus, 1994; Marker, 1998; Marker-Kraus et al., 1996; Durant et al., 2017). Moreover, even in southern Africa cheetah numbers are probably lower than previous estimates and are declining (Weise et al., 2017). It is for these reasons that cheetahs in Namibia, the country with the largest remaining population of free-roaming cheetahs, are now considered as endangered (Durant et al., 2015; RWCP and IUCN/SSC, 2015; LCMAN Red Data List for Namibian Carnivores, in prep). In the light of this new research on cheetah numbers (Weise et al., 2017), farmers should be well informed of their own role in the future survival of cheetahs. Probably the most important aspect of managing human conflict with cheetahs, is the fact that 90% of Namibia's cheetahs are found on farmlands outside protected areas (Marker et al., 2003a). This implies that any conflict management option which relies on the local extirpation of cheetahs, is not an option. Moreover, even other methods that negatively impact the ability of cheetah populations to sustain themselves on farmlands, should be avoided (Weise et al., 2015a), including management actions that could increase bush encroachment (De Klerk, 2004; Rothauge, 2011a,b; Nghikembua et al., 2016).

Spatial ecology – The long distances that cheetahs move, means that translocation of cheetahs as a management technique for solving human-wildlife conflict is unlikely to succeed (Weise et al., 2015a). For example, of 14 cheetahs donated by a farmer to Etosha National Park in the 1970's, two returned to the original capture site, a distance of about *450 km* (McVittie, 1979) – the fate of the other 12 remained unknown. Because of the large home ranges and home range overlaps and relatively few individuals that show any territoriality, there is little in the spatial behaviour of cheetahs that can be used by farmers to reduce livestock losses.

Breeding ecology – There is little about the breeding ecology of cheetahs that can be exploited to reduce livestock depredation. However, it is important for farmers to understand why larger groups of cheetahs are sometimes seen together and that it is not an indication of high cheetah densities (McVittie, 1979).

Feeding ecology – The known prey preferences of cheetahs can be exploited by first of all keeping calves in the preferred size class (younger than six months old) out of the reach of cheetahs, using any of the preventative methods mentioned above that will enable this. It should be kept in mind that male coalitions will kill larger prey than individual males or females (Eaton, 1970; Melzheimer et al., 2018).

Since individual cheetahs show definite prey preferences, confirmed repeat livestock killers should be removed (Eaton, 1970; Hayward et al., 2006b). If they are released in a protected area, they need to be collared and surrounding farmers should be prepared for the fact that they are likely to leave the protected area (Weise et al., 2015a). As mentioned above, a definite distinction should be made between a habitual livestock killer (one with a greater preference for livestock than most or for whom livestock became their preferred prey) and "surplus killing" (Linnell et al., 1999).

In a full agroecological perspective, having more free-roaming game on the farm can be considered as diversification of the farming enterprise (Stein et al., 2010; Lindsey et al., 2013c,b; Rust and Marker, 2013a). Ecologically, it also uses an aspect of prey dilution to reduce the encounter rates of cheetahs with livestock individuals as prey (Haas et al., 2011; King et al., 2012).

Interspecific interactions – Cheetahs' avoidance of other large predators can be exploited in different ways. The easiest is probably the use of livestock guarding dogs to protect livestock (Marker, 2000b; Marker et al., 2005a; Potgieter, 2011; Potgieter et al., 2013). Another option to be investigated is the use of bio-boundaries (Jackson et al., 2012).

Research required: Of the four Namibian species considered here, cheetahs were the best studied (see the section on methods, 2.3.1 above on page 87).

Conservation status – There have been reports by Namibian farmers that cheetah numbers have decreased and leopards have increased (various, e.g. Francois Kok, Johann Britz, Rolf Ritter 2014 - 2017, personal communication). Continuous monitoring of cheetah numbers, and determining the reasons for such decline, once confirmed, is probably the greatest research need for cheetahs at present (Weise et al., 2017). An active adaptive management approach is important (McDonald-Madden et al., 2010).

Spatial ecology – Recent research has highlighted the spatial strategies of cheetahs (Melzheimer et al., 2018). Still missing in Namibia, is determining how cheetahs use their home ranges (i.e. general habitat preferences), how it differs between males and females (*cf.* Caro and Collins, 1987; Eaton, 1974; Broomhall et al., 2003, but see Nghikembua et al., 2016 for a start) and if different habitat types are preferred for different activities.

Breeding ecology – It is unknown if or by how much female cheetahs contract their home ranges when having young cubs. Connected to the spatial ecology, is determining which factors determine their selection of denning sites when having young. Both of these pieces of the behavioural puzzle could help farmers avoid possible predation hotspots.

Feeding ecology – While it is known that cheetahs generally take livestock at lower numbers than their availability (avoidance), and that kudu are the most common prey item in Namibia, it is not known what their actual prey preferences look like on Namibian farms (*cf.* Hayward et al., 2006b).

Interspecific interactions – How (or if) cheetah spatial use change in the presence of leopards, still needs to be determined.

2.3.3.4 *Panthera pardus*

Distribution, conservation status and threats: Leopards are the most widely distributed wild felids, spread across Africa and Asia in nine subspecies. Although widespread, leopard populations have become reduced and isolated from each other, and they are now extirpated from large portions of their global historic range (Stein et al., 2015). They experienced marked range loss from Africa north of the Sahara, West Africa, Northeast Africa and South Africa (Stein et al., 2015). Leopards are found in a wide variety of habitats, and the leopard is the only felid species found in both equatorial forests and forests in Africa. They are most abundant in woodland, grassland savannah and all forest types but also occur widely in montane habitats, coastal scrub, shrubland, semi-desert and desert (Ray et al., 2005), as long as there is enough cover for concealment (Estes, 1991; Swanepoel et al., 2013). Their density in Africa south of the Sahara shows a positive correlation to the biomass of prey weighing between 16 and 50 kg (Stander et al., 1997c; Marker and Dickman, 2005a; Hayward et al., 2007b).

Due to the worldwide decline in leopard numbers, the leopard is classified as vulnerable in the IUCN red data list (Stein et al., 2015). In Namibia they are found over most of the country except for the central far North and the south-eastern small livestock farming area (Stein et al., 2015). In North-Central Namibia there has been

reports of an increase in leopard range, from their preferred rugged mountainous areas (Swanepoel et al., 2013) into the plains, a movement that has been attributed to bush encroachment creating a more favourable leopard habitat (various farmers, personal communication), but there has been no long-term research to confirm this. The majority of leopards are found outside protected areas (Pitman, 2012; Stein et al., 2015) and conflict with humans remains the major threat to their survival (Swanepoel et al., 2015). Thus leopards are still considered as vulnerable in Namibia, especially in the light of their global decline (LCMAN Red Data List for Namibian Carnivores, in prep).

Spatial ecology: Similar to caracals, leopards are solitary, territorial, with males having larger territories overlapping with those of females which are smaller (Estes, 1991; Hayward et al., 2006a). For both sexes home range sizes are the smallest in forested and rocky areas (Stein and Hayssen, 2013). Home ranges vary between 8 km^2 in Sri Lanka to 14 km^2 in Kenya for females to 1 803 km^2 (as MCP, 2 750.05 km^2 as 95% KDE) for males in the southern Kalahari (Bothma et al., 1997; Stein and Hayssen, 2013). In Namibia home ranges have been estimated at between 40 – 393 km^2 for females and 108 – 1 164 km^2 for males (Stander et al., 1997c; Marker and Dickman, 2005a; Stein et al., 2011). A number of studies have shown that low leopard density and large home ranges lead to large range overlaps (up to 59.4% in some seasons), but with temporal avoidance (Stander et al., 1997c; Marker and Dickman, 2005a; Martins, 2010) and usually still exclusive core territories (Stein and Hayssen, 2013).

Breeding ecology: In Namibia, leopards breed throughout the year, typically after females have reached 23-32 months and males 18 months of age. Litter size is one to six with two being the most common. Most commonly inter-birth intervals are 8-12 months, with cubs reaching independence at 13 months (Estes, 1991; Stein and Hayssen, 2013). Infanticide by the new male is common when a territorial male is removed before cubs had reached independence.

Feeding ecology: Typically leopards are nocturnal with peak activity during the hours of dawn and dusk (Hayward and Slotow, 2009; Fröhlich, 2011; Stein and Hayssen, 2013). In forest habitats however, they are commonly diurnal because of the reduced visibility (Stein and Hayssen, 2013).

Leopards are stealth hunters. The most preferred mass of leopard prey among all species is 25 kg, with the significantly preferred prey species having a mean body mass of 23 kg (Hayward et al., 2006a). They prefer prey within a body mass range of 10 to 40 kg, which occur in small herds, in dense habitat and afford the hunter minimal risk of injury during capture (Hayward et al., 2006a). Overall, bushbuck, impala, and common duiker are significantly preferred (Hayward et al., 2006a). The main prey species of leopards in North-Central Namibia are kudu, warthog and oryx, with livestock being killed much less than their availability (Stein, 2008). In the north-eastern communal area, duiker formed the most important element in their diet, steenbok the second highest contributor, followed by a combination of different carnivores (aardwolf, cheetah, genet, bat-eared fox, African wild cat and python (*Python sebae*)(Stander et al., 1997c). This contribution of carnivores to the leopard diet was also seen in Kalahari leopards, with black-backed jackals in 15% of scat samples (Bothma and Le Riche, 1994b), and elsewhere (Estes, 1991).

Balme et al. (2007) showed that leopards made most of their kills in areas of intermediate bush cover because of a greater ability to see and find prey there compared to denser bush, while also providing enough cover for stealth hunting. The preference shown by leopards for rugged areas (Swanepoel et al., 2013), can also be explained by more cover resulting in greater hunting success as well as high numbers of preferred prey like klipspringer and rock hyrax (Hayward et al., 2006a; Martins et al., 2011). While leopards are known as generalists with one of the widest ranges in number of prey species taken (92 prey species recorded in sub-Saharan Africa – Hayward et al., 2006a), including small to medium-sized ungulates, fish, reptiles, birds, and small mammals (Stein and Hayssen, 2013), there is some evidence for individual specialisation (Balme et al., 2007; Natasha de Woronin

Britz 2014, personal communication). Voigt et al. (2018) showed that this individual specialisation was more noticeable in males than in female leopards. This is similar to what has been found for pumas in North America (Knopff and Boyce, 2007).

In the absence of other large predators, leopards, like cheetahs, tend to increase their dietary niche by taking larger prey (Hayward et al., 2006a; Stein, 2008; Stein and Hayssen, 2013). They are also less likely to drag prey up into trees in the absence of other large predators, but will rather drag carcasses into thick bushes to hide (Balme et al., 2007; Stander et al., 1997c). The fact that leopard densities on farmlands in Namibia are generally higher than in some protected areas (Waterberg and Etosha National Park – Stander et al., 1997c), can be ascribed to lower prey densities in some protected areas (Stein et al., 2011), conflict with other predators (Cristescu et al., 2013), and/or a source-sink system leading to higher densities in the sink populations (Pulliam, 1988; Pulliam and Danielson, 1991).

Interspecific interactions: Leopards are subject to intra-guild competition through both interference and exploitive competition (Hayward and Kerley, 2008). Unlike cheetahs and wild dogs, leopards are seldom the victims of kleptoparasitism, because they drag their prey into trees when larger predators are around or they hide their prey in bushes (Balme et al., 2007). Leopard number appears to be limited by prey density, habitat quality and intra-specific exclusion competition, rather than competition with other predators (Hayward et al., 2007b). Their solitary asociality allows them to use stealth for avoiding conflict with larger predators (Stander et al., 1997c). Both cheetahs and black-backed jackals can be a significant portion of leopard diet (Hayward et al., 2006a; Bothma and Le Riche, 1984, 1994b; Stander et al., 1997c). If it is taken into account that predators (higher trophic levels) will likely have a lower biomass (and densities) than lower trophic levels (Hutchinson, 1959), the influence of this intra-guild predation can have significant top-down effects on the whole ecosystem (Estes et al., 2011; Ford et al., 2014; Terborgh, 2015; Flagel et al., 2017). On Namibian farmlands without lions and hyaenas, the leopard is the apex predator (Voigt et al., 2018).

Application to HWC: *Conservation status* –Although leopards appear to be increasing their range in Namibia, they remain vulnerable to extinction (Stein et al., 2015; LCMAN Red Data List for Namibian Carnivores, in prep). The fact that they occur mostly outside protected areas, excludes any conflict management option that depends on the extirpation of leopards from farmlands (Ray et al., 2005).

Spatial ecology – Even when reducing numbers rather than local extirpation is the aim, indiscriminate removal of leopards is unlikely to be effective due to the extensive roaming by subadult non-resident leopards, and large home ranges. It will simply create a sink in which new leopards will establish themselves in the vacated territory, possibly at higher densities than a stable territorial structure (Linnell et al., 1996). These large home ranges, strong homing instinct and mobility of leopards also means that translocations need to be further than 200 km from their capture home range and only used as a last resort (Weise et al., 2015b). Large home ranges also imply that a single habitual livestock killer can cause damage on more than one farm – this can easily result in farmers over-estimating leopard numbers and their impact on livestock, thinking that multiple leopards are involved, because the losses are so widespread.

It is commonly claimed that having territorial leopards on your land that are not killing livestock, is an effective way to keep out young, inexperienced leopards that are more likely to kill livestock because livestock is easy prey, they have not learned to avoid people yet, and are often stressed for food after leaving their mothers (see e.g. Stander et al., 1997c; Fattebert et al., 2016). On the other hand, if territoriality of leopards actually breaks down on Namibian farms, this approach will be ineffective.

Breeding ecology – Leopards don't have breeding seasons, so seasonal management alone is unlikely to influence livestock losses. While no definite evidence for it could be found in the literature for Namibia, it is likely that females restrict their home ranges when they have cubs – thus there could be specific areas to be avoided by livestock when a female leopard has young cubs.

Feeding ecology – Because leopards in Southern Africa are mostly nocturnal (Hayward and Slotow, 2009), keeping vulnerable livestock in predator-proof kraals at night while grazing during the day, might be enough to prevent livestock losses (Marker-Kraus et al., 1996). See Section 2.2.3 on page 39 above, for more on how kraaling can be done in different ways and combined with various other methods to prevent livestock depredation.

Like cheetahs, leopards prefer prey other than livestock, and a buffer of their preferred prey species could reduce livestock predation by leopards (Mizuntani, 1999). If it is true that leopards as a species are generalists, while individual leopards tend to specialise on specific prey species (and mothers teach their offspring to hunt the same prey), it means that removal of "problem individuals" can be an effective conflict mitigating method. The observation that male leopards show greater individual prey specialization than females, partly explains why male leopards are more likely to run afoul of farmers than females (Voigt et al., 2018; Weise et al., 2015b). One way in which this can be exploited, is by trophy hunting of identified habitual livestock killers (Jacobs, 2009), instead of the current system where prime males are often selected (Treves, 2009; Packer et al., 2009; Lindsey et al., 2013a; Bouché et al., 2016). Since females typically teach their young to hunt, removal of females that habitually kill livestock, could also help to prevent this behaviour from spreading in the population.

The habitat type in which leopards prefer to hunt, can be exploited by avoiding rugged, dense or mountainous areas and by using open grassland camps for vulnerable livestock (Hayward et al., 2006a). Rather than direct indiscriminate killing of leopards, any management options that reverses the process of bush encroachment on Namibian savannah would also restrict leopard hunting habitat and reduce the potential for livestock depredation. If successful, this could also cause a transition to a state of higher rangeland productivity, and possibly higher long-term resilience if managed sustainably (Rothauge, 2011a,b).

Interspecific interactions –It is unclear to what extent the density or behaviour of other carnivores are changed by leopards killing and predating on them. However, it is possible that the current reported high small livestock losses due to jackals and caracals are largely the result of mesopredator release (Ritchie and Johnson, 2009). If the mesopredator release transition is reversible to the previous stable state within a reasonable time span (Holling, 1973; Westoby et al., 1989), reintroduction of cheetahs and leopards could return the ecosystem to a stable state with lower mesopredator densities and lower livestock losses. This also presupposes enough natural prey for the larger predators to survive without needing to kill livestock.

Research required: *Conservation status* –There are contradictory claims about what is happening with the leopard population in Namibia (LCMAN Red Data List for Namibian Carnivores, in prep). More research on the actual trends in the Namibian population, the level of persecution by farmers, and the direct and indirect effects of bush encroachment on leopard numbers are required.

Spatial ecology – While it has been claimed that leopards have enlarged their range in Namibia because of bush encroachment (LCMAN Red Data List for Namibian Carnivores, in prep), no actual research has been done to prove this. While generally, it has been found that leopards prefer rugged habitats with dense woody cover, it has not been determined how this translates into the Namibian vegetation types and topography (Swanepoel et al., 2013). Most spatial research on leopards has been restricted to one area with little comparison on how different habitat factors influence home range sizes and population densities across the country (Stander et al., 1997c; Stander, 1998; Marker and Dickman, 2005a; Stein, 2008; Stein et al., 2011; Edwards et al., 2016).

Like for all three other species, fine-scale habitat studies, including how leopards use the habitat within their home ranges and which habitat types are preferred for certain activities, have not been done in Namibia (*cf.* Balme et al., 2007).

Breeding ecology – It is unknown if female leopards in Namibia actually reduce their home ranges when they have cubs. They might still use the whole home range with frequent return trips to the den site and frequent movements of the cubs between different den sites. The effects of breeding on the spatial behaviour of female leopards (if any) still needs to be studied.

Feeding ecology – While recent research has shown that male and female leopards differ in their diet and the breadth of their dietary niche, it is not known what this looks like in terms of prey species (Voigt et al., 2018). The few Namibian studies that investigated prey species preferences of leopards, were geographically restricted and the results cannot be extrapolated to all of Namibia's farmlands (Stander et al., 1997c; Stein, 2008).

Interspecific interactions –Leopards are known to kill all three of the other predator species on occasion (Bothma and Le Riche, 1994b; Stander et al., 1997c; Bothma, 2012). However, it still needs to be established whether this is enough to create a "landscape of fear", cause niche contraction or decrease the densities of any of the other species (*cf.* Flagel et al., 2017).

2.4 Conclusions

The published literature was reviewed from the viewpoint of the individual Namibian commercial livestock farmer faced with managing livestock depredation on his land. Although interconnected as parts of a single agroecological system, this review considered the relevant agricultural management methods and its larger ecological framework separately.

The agricultural management part of this review had two main aims: 1) find as many predator conflict mitigation techniques as possible that might be of future use in Namibia and 2) give them an initial evaluation from what is currently known, in terms of probable effectiveness, advantages, disadvantages and limitations for use by Namibian farmers. Forty different methods were identified from the literature, of which most were preventative, five compensatory, and only three reactive. Of these 40 methods, 29 had been used in Namibia. In general, reactive methods were cheaper than their preventative equivalent, but carried a higher risk of predation loss. Compensatory methods had a negligible effect on predation, but instead compensated for livestock losses (to some extent). Both compensatory carnivore conflict resolution methods and some preventative methods (like extermination), depended on working together with others for its success (shown as one of the possible limitations in Table 2.1). Strictly speaking, such a method is therefore not a management option for the *individual* farmer who needs to decide how to manage livestock depredation on his farm. Each method was classified as having high, medium or low ecological impact, used as a proxy for its ecological sustainability. Where applicable, the known ecological principles on which the effectiveness of a specific method depends, was briefly discussed. A prior probability of success was assigned to each method that had been evaluated in the literature. For those methods of which their effectiveness was not evaluated or compared in the literature, a prior success probability of p(success) = 0.5 was assigned. For each method, its pros, cons and limitations were made explicit. Of the seven agricultural objectives relevant to developing a Decision Support System for Namibian farmers (Section 1.6.1 in Chapter 1), three were addressed: 1) anti-predation methods known to be used by Namibian farmers, 2) advantages and disadvantages of anti-predation methods used worldwide and 3) the relative short-term effectiveness of each method from an agricultural perspective (see Table 2.1 on page 47 for a summary). To get a better estimate of the effectiveness of the methods used in Namibia, a farmer survey was done (described in Chapter 3).

The ecological sustainability of the agroecosytem was addressed within the framework of ecosystem maturity, extended to include multiple stable states and possible transitions between them (Margalef, 1963; Holling, 1973; Westoby et al., 1989; Ludwig et al., 1997). Long-term studies that focus on Namibian farmland ecosystems and the factors determining their productivity and sustainability are seriously lacking (see Ward, 2005; Sinclair et al., 2007; Caro, 2011). From a theoretical ecological perspective however, a case was made that "ecological disturbance" or impact of a management technique is likely to directly influence its long-term sustainability (Section 2.3.2.1 above). The importance of veld management as managing the primary producer level of Namibian agroecosystems was highlighted briefly — the possible effects of predation management methods on the veld should be included when evaluating them for long-term sustainability. It was shown from the literature that ecosystems dominated by top-down control generally show higher biodiversity, resilience and stability than

bottom-up controlled ecosystems (Estes et al., 2011; Terborgh, 2015). Allowing top predators to fulfil this role of top-down control in the agroecological system is thus more likely to be sustainable in the long run — the challenge is to exclude livestock species from this control by predators and have human managers fulfil this role for livestock instead.

Within this greater ecological framework of sustainability, the ecology of the four predator species on Namibian farmlands was examined, with the specific aims of identifying gaps in our knowledge and further research requirements. One factor limiting the choice of available management techniques, is the conservation status of the predator species (Linnell et al., 1996). The conservation status of leopards and especially cheetahs was found to preclude the use of methods aimed at local extirpation as well as judicious use of most pre-emptive lethal methods and methods with higher ecological impact, for these two species. No such species conservation-based limitations were found for jackals or caracals.

In Section 1.6.2 of Chapter 1, nine relevant ecological aspects of the agroecological model were identified:

1. Two important facets of predator diet were: 1) the prey preferences of the species and 2) the dietary specialization of individuals.

   Of the four species, black-backed jackals were the most likely to scavenge, with their well-developed sense of smell contributing to finding carcasses for food (Estes, 1991; Bothma, 2012). While caracals and leopards would also occasionally scavenge, cheetahs almost exclusively lived from what they killed for themselves (Stander, 1990b; Lindeque et al., 1998). Jackals were also the only one of the four species where vegetation (fruit) and invertebrates sometimes comprised the majority of their diet, illustrating their adaptability and dietary width (Stuart, 1976; Goldenberg et al., 2010). While the three felid species were more exclusively carnivorous, caracals for example, had a larger average ungulate prey size range than jackals (Drouilly et al., 2018). In practice, caracals are more likely to take both adult and young small livestock, while black-backed jackals would prefer lambs (Roberts, 1986). Leopards had a similarly large prey size range, while cheetahs took prey within a more restricted range. Both larger predator species are capable of taking calves as well as adult small livestock, with leopards occasionally able to kill calves older than nine months (Figure 2.3). Where-ever this had been studied in Namibia, predators preferred natural prey to livestock (in some parts of South Africa, black-backed jackals might start to prefer livestock when lambs are available throughout the year – Drouilly et al., 2018, but see Kamler et al., 2012a). Most prey species appear to be taken according to availability, and all four carnivore species are considered generalists to some extent, but all also show some prey preferences within a certain landscape (Avenant and Nel, 2002; Hayward et al., 2006b; Balme et al., 2007; Minnie et al., 2016a). However, the specific preferred prey species in Namibia for each predator species is not known – farmers can thus not be given reliable advice on which prey species will be the best buffer species for each predator species (Mizuntani, 1999). More diet studies on Namibian farmlands are needed, preferably using a combination of techniques such as scat analysis (which cannot tell anything about the age and size of the prey), GPS collar cluster analysis (which often underestimate small prey items) and spoor tracking (which can include both hunting attempts and successes, and can differentiate between scavenging and hunting, but are limited by soil substrate and tracking skills). Specifically, these methods also need to be combined with an index of (seasonal) abundance (availability) of the different prey species in order for actual preferences to be determined.

2. It was seen that individual cheetahs and leopards can specialise on specific prey species, and it is therefore possible that some individuals can start to specialize on livestock (in which case mitigation interventions can concentrate on a single individual). However, it is unknown whether similar individual prey specialisation is also found in jackals and caracals. GPS collar studies are probably the best method currently for determining individual diet preferences. Spoor tracking (where the substrate allows it) of single individuals for multiple days (Bothma and Le Riche, 1984) or the isotope analysis approach used by Voigt et al. (2018) are other approaches that might show the level of individual prey specialisation by these predators.

3. Five relevant aspects of the four predator species' spatial ecology include: 1) home range sizes and territorial behaviour of each predator species, 2) the proportion of resident to non-resident individuals in the predator population, 3) habitat preferences within their home ranges, 4) specialist usage of certain habitat types for certain activities, and 5) typical dispersal distances of each species.

   The home range sizes of jackals and caracals on Namibian farmlands are generally unknown, although from studies in South Africa, it can be expected that caracals will have larger home ranges than jackals. Caracal males in North-Central Namibia had home ranges of 79 – 440 km^2 (mean of 312.6 km^2) in the only published study of caracals in Namibia. Cheetah (515 – 2 324 km^2) and leopard (40 – 1 164 km^2) home ranges had been recorded in Namibia, but only in the north central to north eastern parts of the country. Further GPS home range studies of all four species on farmlands in different biomes would be required for a fuller picture of their spatial ecology.

4. The proportion of resident to non-resident individuals in the Namibian predator populations is currently unknown. A demographic study such as has been done for cheetahs, would probably be the best starting point for such research (Marker et al., 2003a). Combined with camera trap studies, this should give an indication of the ratio of dispersing individuals to breeding, territorial animals and the age-class structure and mortality rates at each age class. This ratio, combined with home range sizes, can enable a good estimation of population densities. If the predator age and sex class that is most likely to kill livestock is also known, this will enable better predation management.

5. No studies in Namibia has been published so far that focussed on identifying preferred habitats within their home ranges for any of the four predator species. GPS collaring (and possibly spoor tracking combined with camera traps in some areas) appears to be currently the most obvious method to do this.

6. While some research has been done on leopards in South Africa and cheetahs in Namibia to identify preferred habitat types for hunting in particular, nothing similar has been done for either of the other two species on Namibian farmlands (Balme et al., 2007; Nghikembua et al., 2016). GPS collars (and spoor tracking where practical) is probably the best way to continue this avenue of research.

7. All four species are known to disperse over large distances, black-backed jackals in South Africa dispersed for up to 150 km, caracals for 138 km, cheetahs for up to 200 km (∼450 km when returning to its home range) and leopards for 353 km (McVittie, 1979; Stuart, 1982; Marker et al., 2008; Humphries et al., 2016; Swanepoel et al., 2016). While generally a minimum distance of 200 km is recommended for translocation of cheetahs and leopards, this would indicate that even 200 km might be too close to prevent homing by these predators (Weise et al., 2015a,b).

8. It has been shown that sometimes one predator species causes much more livestock damage than other predators in the same area. Two aspects of interspecific, intra-guild interactions between predator species can be utilised to decrease such livestock losses: 1) spatial avoidance of some high-use areas of the dominant species by the less dominant predators, and 2) prevention (or reversing) of mesopredator release.

   Spatial avoidance or niche partitioning between any of the four predator species have not been investigated in Namibia yet. Simultaneous GPS collaring of individuals from each predator species in the same area would be required to show spatial or temporal avoidance between the species. To a lesser extent camera traps and spoor tracking can provide additional evidence for such avoidance.

9. While it seems likely that mesopredator release is at least partly to blame for the current high livestock losses from black-backed jackals and caracals in Southern Namibia (e.g. Swanepoel, 2016), no such evidence have been published for Namibia. The only research approach that is likely to provide direct evidence of mesopredator release, is replicated before-and-after manipulation studies, preferably with controls, where the effects of removal of top predators on the mesopredator numbers as well as the result of (re)introducing

Figure 2.3: *Calf killed by leopard on farm Troye, North Central Namibia (Okahandja district).* This calf was already older than nine months (often considered a "safe" age against predators – Rolf Ritter 2015, personal communication).

top predators, are compared (Sinclair et al., 2003; Flagel et al., 2017). However, lower densities of the same mesopredators in the presence of top predators than in areas of the same habitat without them, would provide some evidence for mesopredator release (Terborgh, 2015). Additionally, if it can be shown that mesopredators avoid top predators spatially (using GPS collars as above), it seems likely that their densities would indeed increase in the absence of the top predators when all the habitat becomes available to them.

While the literature review answered many questions about the ecology of the four predators on Nambian farmlands, it is clear that many important facets of their ecology that are relevant to human-wildlife conflict are still unknown and require more research. Nine of these facets that require further research have been identified here and an appropriate research approach for each has been suggested.

References

Adabe, L., Macdonald, D., Dickman, A., 2014. Assessing the relative importance of landscape and husbandry factors in determining large carnivore depredation risk in Tanzania's Ruaha landscape. Biological Conservation 180, 241–248.

Andelt, W., 2004. Use of livestock guarding animals to reduce predation on livestock. Sheep & Goat Research Journal 19, 72–75.

Anthony, B., Swemmer, L., 2015. Co-defining program success: Identifying objectives and indicators for a livestock damage compensation scheme at Kruger National Park, South Africa. Journal for Nature Conservation 26, 65–77.

Athreya, V., Odden, M., Linnell, J., Karanth, K., 2010. Translocation as a tool for mitigating conflict with leopards in human-dominated landscapes of India. Conservation Biology 25 (1), 133–141.

Avenant, N., Drouilly, M., Power, R., Thorn, M., Martins, Q., Neils, A., Du Plessis, J., Do Linh San, E., 2016. A conservation assessment of *Caracal caracal*. In: Child et al. (2016), p. 13 pp.

Avenant, N., Du Plessis, J., 2008. Sustainable small stock farming and ecosystem conservation in southern Africa: a role for small mammals? . Mammalia 72, 258–263.

Avenant, N., Nel, J., 1998. Home-range use, activity, and density of caracal in relation to prey density. African Journal of Ecology 36, 347–359.

Avenant, N., Nel, J., 2002. Among habitat variation in prey availability and use by caracal *Felis caracal*. Mammalian Biology 67, 16–33.

Avgan, B., Henschel, P., Ghoddousi, A., 2016. *Caracal caracal*. In: The IUCN Red List of Threatened Species 2016, Downloaded on 1 March 2018. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T3847A50650230.en, p. e.T3847A102424310.

Bailey, A., Conradie, B., 2013. The effect of predator culling on livestock losses: Caracal control in Cooper hunting club, 1976 – 1981. Tech. rep., Centre for Social Science Research, Cape Town.

Baker, P., Funk, S., Harris, S., White, P., 2000. Flexible spatial organization of urban foxes, *Vulpes vulpes*, before and during an outbreak of sarcoptic mange. Animal Behaviour 59 (1), 127–146.

Baker, S., Johnson, P., Slater, D., Watkins, R., Macdonald, D., 2007. Learned food aversion with and without an odour cue for protecting untreated baits from wild mammal foraging. Applied Animal Behaviour Science 102 (3-4), 410–428.

Balme, G., Batchelor, A., De Woronin Britz, N., Seymour, G., Grover, M., Hes, L., Macdonald, D., Hunter, L., Oct. 2012a. Reproductive success of female leopards *Panthera pardus*: the importance of top-down processes. Mammal Review 43 (3), 221–237.

Balme, G., Hunter, L., Goodman, P., Ferguson, H., Craigie, J., Slotow, R., 2010. An adaptive management approach to trophy hunting of leopards (*Panthera pardus*): a case study from KwaZulu-Natal, South Africa. In: Macdonald and Loveridge (2010), Ch. 14, pp. 341–352.

Balme, G., Hunter, L., Slotow, R., 2007. Feeding habitat selection by hunting leopards *Panthera pardus* in a woodland savanna: prey catchability versus abundance. Animal Behaviour 74, 589–598.

Balme, G., Lindsey, P., Swanepoel, L., Hunter, L., 2014. Failure of research to address the rangewide conservation needs of large carnivores: leopards in South Africa as a case study. Conservation Letters 7 (1), 3–11.

Balme, G., Slotow, R., Hunter, L., 2009. Impact of conservation interventions on the dynamics and persistence of a persecuted leopard (*Panthera pardus*) population. Biological Conservation 142, 2681–2690.

Bartnick, T., Van Deelen, T., Quigley, H., Craighead, D., 2013. Variation in cougar (*Puma concolor*) predation habits during wolf (*Canis lupus*) recovery in the southern Greater Yellowstone Ecosystem. Canadian Journal of Zoology 91 (2), 82–93.

Bauer, H., Müller, L., Van der Goes, D., Sillero-Zubiri, C., 2017. Financial compensation for damage to livestock by lions *Panthera leo* on community rangelands in Kenya. Oryx 51 (1), 106–114.

Begg, C., Kushnir, H., 2013. Human-lion conflict toolkit. Wildlife Conservation Network, www.appleseedesign.com.

Begon, M., Harper, J., Townsend, C., 1996. Ecology: individuals, populations and communities., third edition Edition. Blackwell Science.

Beinart, W., 1998. The night of the jackal: sheep, pastures and predators in the Cape. Past & Present 158, 172–206.

Berger, K., 2006. Carnivore-livestock conflicts: Effects of subsidized predator control and economic correlates on the sheep industry. Conservation Biology 20 (3), 751–761.

Bergman, D., De Waal, H., Avenant, N., Bodenchuk, M., Marlow, M., Nolte, D., 2013. The need to address black-backed jackal and caracal predation in South Africa. In: J.B., A., Gallagher, G. (Eds.), Proceedings of the 15th Wildlife Damage Management Conference. Wildlife Damage Management Conference, pp. 86–94.

Bernard, R., Stuart, C., 1987. Reproduction of the caracal *Felis caracal* from the Cape Province of South Africa. South African Journal of Zoology 22 (3), 177–182.

Bingham, S., 1997. Grassroots restoration: holistic management for villages. Savory Center.

Bissett, C., Bernard, R., 2007. Habitat selection and feeding ecology of the cheetah (*Acinonyx jubatus*) in thicket vegetation: is the cheetah a savanna specialist? Journal of Zoology 271, 310–317.

Bonacic, C., Amaya-Espinel, J., Ibarra, J., 2016. Human-wildlife conflicts - An overview of cases and lessons from the Andean region. In: Aguirre, A., Sukumar, R. (Eds.), Tropical Conservation: Perspectives on Local and Global Priorities., 1st Edition. Oxford University Press, Ch. 8, pp. 109–125.
URL `https://books.google.co.za/books?id=BgsBDQAAQBAJ`

Bosc, C., Roets, F., Hui, C., Pauw, A., 2018. Interactions among predators and plant specificity protect herbivores from top predators. Ecology 99 (7), 1602–1609.

Bothma, J. P., Jul. 2012. Literature review of the ecology and control of the black-backed jackal and caracal in South Africa. Tech. rep., CapeNature.

Bothma, J. P., Knight, M., Le Riche, E., Van Hensbergen, H., 1997. Range size of southern Kalahari leopards. South African Journal of Wildlife Research 27 (3), 94–99.

Bothma, J. P., Le Riche, E., 1984. Aspects of the ecology and the behaviour of the Leopard *Panthera pardus* in the Kalahari desert. Koedoe (Supplement) 27, 259–279.

Bothma, J. P., Le Riche, E., 1994a. Range use by an adult male caracal in the southern Kalahari. Koedoe 37 (2), 105–108.

Bothma, J. P., Le Riche, E., 1994b. Scat analysis and aspects of defecation in Northern Cape leopards. South African Journal of Wildlife Research 24 (1/2), 21–26.

Bouché, P., Crosmary, W., Kafando, P., Doamba, B., Kidjo, F., Vermeulen, C., Chardonnet, P., 2016. Embargo on lion hunting trophies from West Africa: An effective measure or a threat to lion conservation? PLoS ONE 11 (5), e0155763.

Brand, D., Fairall, N., Scott, W., 1995. The influence of regular removal of black-backed jackals on the efficiency of coyote getters. South African Journal of Wildlife Research 25 (2), 44–48.

Broomhall, L., Mills, M., Du Toit, J., 2003. Home range and habitat use by cheetahs (Acinonyx jubatus) in the Kruger National Park. Journal of Zoology, London 261 (2), 119–128.

Brown, J., 1988. Patch use as an indicator of habitat preference, predation risk, and competition. Behavioral Ecology and Sociobiology 22 (1), 37–47.

Cadotte, M., Dinnage, R., Tilman, D., 2012. Phylogenetic diversity promotes ecosystem stability. Ecology 93 (8), S223–S233.

Carbone, C., Teacher, A., Rowcliffe, J., 2007. The costs of carnivory. PLoS Biology 5 (2), 363–368.

Caro, T., 1987. Cheetah mothers' vigilance: Looking out for prey or for predators? Behavioral Ecology and Sociobiology 20 (5), 351–361.

Caro, T., 2011. On the merits and feasibility of wildlife monitoring for conservation: a case study from Katavi National Park, Tanzania. African Journal of Ecology 49, 320–331.

Caro, T., Collins, D., 1986. Male cheetahs of the Serengeti. National Geographic Research 2 (1), 75–86.

Caro, T., Collins, D., 1987. Male cheetah social organization and territoriality. Ecology 74, 52–64.

Carter, J., Jones, A., O'Brien, M., Ratner, J., Wuerthner, G., 2014. Holistic Management: Misinformation on the science of grazed ecosystems. International Journal of Biodiversity 2014 (163431), 1–10.

Carvalho, E., Zarco-González, M., Monroy-Vilchis, O., Morato, R., 2015. Modeling the risk of livestock depredation by jaguar along the Transamazon highway, Brazil. Basic and Applied Ecology 16 (5), 413–419.

Casaway, W., Casaway, K., Berry, H., 1996. Persistent low densities of plains ungulates in Etosha National Park, Namibia: testing the food-regulating hypothesis. Canadian Journal of Zoology 74 (8), 1556–1572.

Caughley, G., 1976. Wildlife Management and the dynamics of ungulate populations. In: Coaker, T. (Ed.), Applied Biology Vol.1. Academic Press, New York, NY, pp. 183–246.

Cayre, P., Michaud, A., Theau, J., Rigolot, C., 2018. The coexistence of multiple worldviews in livestock farming drives agroecological transition. A case study in French Protected Designation of Origin (PDO) cheese mountain areas. Sustainability 10 (4), 1–13.

Chapron, G., Treves, A., 2016. Blood does not buy goodwill: allowing culling increases poaching of a large carnivore. Proceedings of the Royal Society B 283, e20152939.

Child, M., Roxburgh, L., Do Linh San, E., Raimondo, D., Davies-Mostert, H. (Eds.), 2016. The 2016 red list of mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust.

Chiriboga, L., Kilmer, C., Fan, R., Gawande, K., 2008. Does Namibia have a comparative advantage in beef production? Tech. rep., Texas A&M University, http://kishoregawande.net/wp-content/uploads/2008/12/finalreporti.pdf.

Christiansen, P., Wroe, S., 2007. Bite forces and evolutionary adaptations to feeding ecology in carnivores. Ecology 88 (2), 347–358.

Connell, J., 1961. The influence of interspecific competition and other factors on the distribution of the barnacle Chthamalus stellatus. Ecology 42 (4), 710–723.

Conradie, B., Piesse, J., 2013. The effect of predator culling on livestock losses: Ceres, South Africa, 1979 – 1987. Tech. Rep. 319, Centre for Social Science Research, Cape Town.

Conradie, B., Piesse, J., 2016. Ranking perceived risk to farmers: How important is the environment? African Journal of Agricultural and Resource Economics 11 (4), 263–276.

Conway, G., 1985. Agroecosystem Analysis. Agricultural Administration 20, 31–55.

Cooper, A., Pettorelli, N., Durant, S., 2007. Large carnivore menus: factors affecting hunting decisions by cheetahs in the Serengeti. Animal Behaviour 73, 651–659.

Cozzi, G., Broekhuis, F., McNutt, J., Turnbull, L., Macdonald, D., Schmid, B., 2012. Fear of the dark or dinner by moonlight? Reduced temporal partitioning among Africa's large carnivores. Ecology 93 (12), 2590–2599.

Cristescu, B., Bernard, R., Krause, J., 2013. Partitioning of space, habitat, and timing of activity by large felids in an enclosed South African system. Journal of Ethology 31 (3), 285–298.

Daly, B., Davies-Mostert, H., Davies-Mostert, W., Evans, S., Friedmann, Y., King, N., Snow, T., Stadler, H. (Eds.), 2006. Prevention is the cure. Proceedings of a workshop on holistic management of human-wildlife conflict in the agricultural sector of South Africa. Endangered Wildlife Trust.

De Klerk, J., 2004. Bush encroachment in Namibia: Report on phase 1 of the bush encroachment research, monitoring and management project. Tech. rep., Ministry of Environment and Tourism.

De Wet, T., 2002. Doodmaak slaag nie teen probleemdiere nie. (Killing does not succeed against problem animals.). Landbouweekblad 4 Januarie 2002, 32–35.

De Wet, T., 2006. Vermin>problem animal>damage control – A wake-up call. In: Daly, B., Davies-Mostert, H., Davies-Mostert, W., Evans, S., Friedmann, Y., King, N., Snow, T., Stadler, H. (Eds.), Prevention is the cure. Proceedings of a workshop on holistic management of human-wildlife conflict in the agricultural sector of South Africa. Endangered Wildlife Trust, pp. 17–22.

Dhondt, A., 1988. Carrying capacity: a confusing concept. Acta Oecologica 9, 337–346.

Dickman, A., Macdonald, E., Macdonald, D., 2011. A review of financial instruments to pay for predator conservation and encourage human–carnivore coexistence. Proceedings of the National Academy of Sciences of the USA 108 (34), 13937–13944.

Dicks, L., Ashpole, J., Dänhardt, J., James, K., Jönsson, A., Randall, N., Showler, D., Smith, R., Turpie, S., Williams, D., Sutherland, W., 2018. Farmland conservation. In: Sutherland, W., Dicks, L., Ockendon, N., Petrovan, S., Smith, R. (Eds.), What Works in Conservation., third edition Edition. OpenBook Publishers, DOI: 10.11647/OBP.0131, Ch. 4, pp. 245–284.

Dieguez Cameroni, F., Fort, H., 2017. Towards scientifically based management of extensive livestock farming in terms of ecological predator-prey modeling. Agricultural Systems 153, 127–137.

Dingfelder, S., 2010. A second chance for the Mexican wolf. Monitor on Psychology 41 (10), 20.

Dreyer, H. v. A., Nel, J., 1990. Feeding site selection by black-backed jackals on the Namib Desert coast. Journal of Arid Environments 19, 217–224.

Drouilly, M., Nattrass, N., O'Riain, M., 2018. Dietary niche relationships among predators on farmland and a protected area. Journal of Wildlife Management 82, 507–518.

Dröge, E., Creel, S., Becker, M., M'Soka, J., 2017. Spatial and temporal avoidance of risk within a large carnivore guild. Ecology and Evolution 7 (1), 189–199.

Du Plessis, J., 2013. Towards the development of a sustainable management strategy for *Canis mesomelas* and *Caracal caracal* on rangeland. Ph.D. thesis, University of the Free State.

Du Plessis, J., Avenant, N., Botha, A., Mkhize, N., Müller, L., Mzileni, N., O'Riain, M., Parker, D., Potgieter, G., Richardson, P., Rode, S., Viljoen, N., Hawkins, H.-J., Tafani, M., 2018. Past and current management of predation on livestock. In: Kerley et al. (2018), Ch. 6, pp. 125–177.

Du Plessis, J., Avenant, N., De Waal, H., 2015. Quality and quantity of the scientific information available on black-backed jackals and caracals: contributing to human–predator conflict management? African Journal of Wildlife Research 45 (2), 138–157.

Du Toit, J., 2013. Predator wars: Lynx vs Jackal. Internet.

Durant, S., 1998. Competition refuges and coexistence: an example from Serengeti carnivores. Journal of Animal Ecology 67, 370–386.

Durant, S., 2000. Predator avoidance, breeding experience and reproductive success in endangered cheetahs, *Acinonyx jubatus*. Animal Behaviour 60, 121–130.

Durant, S., Mitchell, N., Groom, R., Pettorelli, N., Ipavec, A., Jacobson, A., Woodroffe, R., Böhm, M., Hunter, L., Becker, M., Broekhuis, F., Bashir, S., Andresen, L., Aschenborn, O., Beddiaf, M., Belbachir, F., Belbachir-Bazi, A., Berbash, A., Brandao de Matos Machado, I., Breitenmoser, C., Chege, M., D., C., Davies-Mostert, H., Dickman, A., Fabiano, E., Farhadinia, M., Funston, P., Henschel, P., Horgan, J., De Iongh, H., Jowkar, H., Klein, R., Lindsey, P., Marker, L., Marnewick, K., Melzheimer, J., Merkle, J., M'Soka, J., Msuha, M., O'Neill, H., Parker, M., Purchase, G., Sahailou, S., Saidu, Y., Samna, A., Schmidt-Küntzel, A., Selebatso, E., Sogbohossou, E., Soultan, A., Stone, E., Van der Meer, E., Van Vuuren, R., Wykstra, M., Young-Overton, K., 2017. The global decline of cheetah *Acinonyx jubatus* and what it means for conservation. PNAS 114 (3), 528–533.

Durant, S., Mitchell, N., Ipavec, A., Groom, R., 2015. *Acinonyx jubatus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T219A50649567.en, p. e.T219A50649567.

Eaton, R., 1970. Hunting behavior of the cheetah. Journal of Wildlife Management 34 (1), 56–67.

Eaton, R., 1974. The Cheetah - The biology, ecology, and behavior of an endangered species. Van Nostrand Reinhold Company.

Edwards, S., Aschenborn, O., Gange, A., Wiesel, I., 2016. Leopard density estimates from semi-desert commercial farmlands, south-west Namibia. African Journal of Ecology 54 (1), 103–106.

Ellins, S., 1985. Coyote control and taste aversion: A predation problem or a people problem? Appetite 6, 272–275.

Ellis, J., Swift, D., 1988. Stability of African pastoral ecosystems: Alternate paradigms and implications for development. Journal of Range Management 41 (6), 450–459.

Estes, J., Terborgh, J., Brashares, J., Power, M., Berger, J., Bond, W., Carpenter, S., Essington, T., Holt, R., Jackson, J., Marquis, R., Oksanen, L., Oksanen, T., Paine, R., Pikitch, E., Ripple, W., Sandin, S., Scheffer, M., Schoener, T., Shurin, J., Sinclair, A., Soulé, M., Virtanen, R., Wardle, D., 2011. Trophic downgrading of planet Earth. Science 333 (6040), 301–306.

Estes, R., 1991. The behavior guide to African mammals: Including hoofed mammals, carnivores, primates. University of California Press.

Fabiano, E., 2013. Demografia histórica e contemporânea de guepardos (*Acinonyx jubatus*) na Namíbia, África Austral. Ph.D. thesis, Pontifícia Universidade Católica do Rio Grande do Sul.

Fattebert, J., Balme, G., Robinson, H., Dickerson, T., Slotow, R., Hunter, L., 2016. Population recovery highlights spatial organization dynamics in adult leopards. Journal of Zoology, London 299 (3), 153–162.

Ferguson, J., 1978. Social interactions of black-backed jackals *Canis mesomelas* in the Kalahari Gemsbok National Park. Koedoe 21, 151–162.

Ferguson, J., Nel, J., De Wet, M., 1983. Social organization and movement patterns of black-backed jackals *Canis mesomelas* in South Africa. Journal of Zoology 199, 487–502.

Fischer, J., Lindenmayer, D., 2000. An assessment of the published results of animal relocations. Biological Conservation 96, 1–11.

Flagel, D., Belovsky, G., Cramer, M., Beyer, D. J., Robertson, K., 2017. Fear and loathing in a Great Lakes forest: cascading effects of competition between wolves and coyotes. Journal of Mammalogy 98 (1), 77–84.

Ford, A., Goheen, J., Otieno, T., Bidner, L., Isbell, L., Palmer, T., Ward, D., Woodroffe, R., Pringle, R., 2014. Large carnivores make savanna tree communities less thorny. Science 346 (6207), 346–349.

Forthman, D., 2000. Experimental application of Conditioned Taste Aversion (CTA) to large carnivores. Carnivore Damage Prevention News 2, 2–4.

Forthman Quick, D., Gustavson, C., Rusiniak, K., 1985. Coyote control and taste aversion. Appetite 6, 253–264.

Frank, L., Simpson, D., Woodroffe, R., 2003. Foot snares: an effective method for capturing African lions. Wildlife Society Bulletin 31 (1), 309–314.

Fröhlich, M., 2011. Studying the foraging ecology of leopards (*Panthera pardus*) using activity and location data: an exploratory attempt. Master's thesis, Humboldt-Universität zu Berlin.

Garcia, J., Kimeldorf, D., Koellino, R., 1955. Conditioned aversion to saccharin resulting from exposure to gamma radiation. Science, New Series 122 (3160), 157–158.

Gillson, L., Biggs, H., Smit, I., Virah-Sawmy, M., Rogers, K., 2019. Finding common ground between adaptive management and evidence-based approaches to biodiversity conservation. Trends in Ecology and Evolution 34 (1), 31–44.

Goldenberg, M., Goldenberg, F., Funk, S., Henschel, J., Millesi, E., 2010. Diet composition of black-backed jackals, *Canis mesomelas* in the Namib Desert. Folia Zoologica 59 (2), 93–101.

Grace, J., Anderson, T., Seabloom, E., Borer, E., Adler, P., Harpole, W., Hautier, Y., Hillebrand, H., Lind, E., Pärtel, M., Bakker, J., Buckley, Y., Crawley, M., Damschen, E., Davies, K., Fay, P., Firn, J., Gruner, D., Hector, A., Knops, J., MacDougall, A., Melbourne, B., Morgan, J., Orrock, J., Prober, S., Smith, M., 2016. Integrative modelling reveals mechanisms linking productivity and plant species richness. Nature 529 (7586), 390–393.

Griffith, B., Scott, J., Carpenter, J., Reed, C., 1989. Translocation as a species conservation tool: Status and strategy. Science 245 (4917), 477–480.

Griffith, R. J., Connolly, G., Burns, R., Sterner, R., 1978. Coyotes, sheep and Lithium Chloride. In: Proceedings of the 8th Vertebrate Pest Conference (1978). pp. 190–196.

Grobler, J., 1981. Feeding behaviour of the caracal *Felis caracal* Schreber 1776 in the Mountain Zebra National Park. South African Journal of Zoology 16 (4), 259–262.

Gunter, Q., 2008. A critical evaluation of historical data on two damage causing predators, *Canis mesomelas* and *Caracal caracal*. Master's thesis, University of the Free State.

Gustavson, C., Garcia, J., Hankins, W., Rusiniak, K., 1974. Coyote predation control by aversive conditioning. Science, New Series 184 (4136), 581–583.

Gwynne, M., Bell, R., 1968. Selection of vegetation components by grazing ungulates in the Serengeti National Park. Nature 220, 390–393.

Haas, S., Hooten, M., Rizzo, D., Meentemeyer, R., 2011. Forest species diversity reduces disease risk in a generalist plant pathogen invasion. Ecology Letters 14, 1108–1116.

Hall-Martin, A., Botha, B., 1980. A note on feeding habits, ectoparasites and measurements of the black-backed jackal, *Canis mesomelas*, from Addo Elephant National Park. Koedoe 23, 157–162.

Hangara, G., Teweldemedhin, M., Groenewald, I., 2011a. Major constraints for cattle productivity and managerial efficiency in communal areas of Omaheke Region, Namibia. International Journal of Agricultural Sustainability 9 (4), 495–507.

Hangara, G., Teweldemedhin, M., Groenewald, I., 2011b. Measuring factors that can influence cattle supply response to the market in Namibia: Case study from Omaheke communal farmers. Journal of Agricultural Extension and Rural Development 3 (8), 102–112.

Hangara, G., Teweldemedhin, M., Groenewald, I., 2012. Assessment on the constraints of cattle supply chain management in Namibia: Case study of Omaheke communal farmers. African Journal of Agricultural Research 71 (35), 4876–4884.

Hansen, I., Jan. 2005. Use of livestock guarding dogs in Norway – a review of the effectiveness of different methods. Carnivore Damage Prevention News 8, 2–8.

Harrington, J., Conover, M., 2007. Does removing coyotes for livestock protection benefit free-ranging ungulates? Journal of Wildlife Management 71 (5), 1555–1560.

Haswell, P., Shepherd, E., Stone, S., Purcell, B., Hayward, M., 2019. Foraging theory provides a useful framework for livestock predation management. Journal for Nature Conservation 49, 69–75.

Hautier, Y., Tilman, D., Isbell, F., Seabloom, E., Borer, E., Reich, P., 2015. Anthropogenic environmental changes affect ecosystem stability via biodiversity. Science 348 (6232), 336–340.

Hayward, M., Adendorff, J., Moolman, L., Hayward, G., Kerley, G., 2007a. The successful reintroduction of leopard *Panthera pardus* to the Addo Elephant National Park. African Journal of Ecology 45 (1), 103–104.

Hayward, M., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G., Kerley, G., 2006a. Prey preferences of the leopard (*Panthera pardus*). Journal of Zoology, London 270 (2), 298–313.

Hayward, M., Hofmeyr, M., O'Brien, J., Kerley, G., 2006b. Prey preferences of the cheetah (*Acinonyx jubatus*) (Felidae: Carnivora): morphological limitations or the need to capture rapidly consumable prey before kleptoparasites arrive? Journal of Zoology 270, 615–627.

Hayward, M., Kerley, G., 2008. Prey preferences and dietary overlap amongst Africa's large predators. South African Journal of Wildlife Research 38 (2), 93–108.

Hayward, M., O'Brien, J., Kerley, G., 2007b. Carrying capacity of large African predators: Predictions and tests. Biological Conservation 139, 219–229.

Hayward, M., Porter, L., Lanszki, J., Kamler, J., Beck, J., Kerley, G., Macdonald, D., Montgomery, R., Parker, D., Scott, D., O'Brien, J., Yarnell, R., 2017. Factors affecting the prey preferences of jackals (Canidae). Mammalian Biology 85, 70–82.

Hayward, M., Slotow, R., 2009. Temporal partitioning of activity in large African carnivores: tests of multiple hypotheses. South African Journal of Wildlife Research 39 (2), 109–125.

Hiscocks, K., Perrin, M., 1988. Home range and movements of black-backed jackals at Cape Cross Seal Reserve, Namibia. South African Journal of Wildlife Research 18 (3), 97–100.

Hixon, M., 2008. Carrying capacity. In: Jørgensen, S., B.D., F. (Eds.), Encyclopedia of Ecology. Elsevier Science, pp. 528–530.

Hobbs, N., 1996. Modification of ecosystems by ungulates. Journal of Wildlife Management 60 (4), 695–713.

Hoffman, M., 2014. *Canis mesomelas*. In: The IUCN Red List of Threatened Species 2014, Downloaded on 1 March 2018. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2014-1.RLTS.T3755A46122476.en, p. e.T3755A46122476.

Holling, C., 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4, 1–23.

Holmes, M., Holmes, R., 2006. Can caracals save sheep? Downloaded January 2015.
URL `http://www.farmersweekly.co.za/article.aspx?id=286`

Hoogesteijn, R., Hoogesteijn, A., 2008. Conflicts between cattle ranching and large predators in Venezuela: could use of water buffalo facilitate felid conservation? Oryx 42 (1), 132–138.

Hoogesteijn, R., Hoogesteijn, A., 2010. Conserving wild felids in humanized landscapes - Strategies for reducing conflicts between jaguars and cattle. Wild Felid Monitor 3 (2), 1, 10–13.

Howery, L., DeLiberto, T., 2004. Indirect effects of carnivores on livestock foraging behavior and production. Sheep & Goat Research Journal 19, 53–57.

Howley, A., 2013. Lions no match for young boy and his invention: Richard Turere at TED. Downloaded January 2015.
URL `http://www.cnn.com/2013/02/26/tech/richard-turere-lion-lights`

Huffaker, C., 1958. Experimental studies on predation: Dispersion factors and predator-prey oscillations. Hilgardia 27 (14), 343–383.

Humphries, B., Ramesh, T., Hill, T., Downs, C., 2016. Habitat use and home range of black-backed jackals (*Canis mesomelas*) on farmlands in the Midlands of KwaZulu-Natal, South Africa. African Zoology 51 (1), 37–45.

Hunt, C., 1984. Vol. 1. Behavioral responses of bears to tests of repellents deterrents and aversive conditioning Vol. 2. Deterrents aversive conditioning and other practices: An annotated bibliography to aid in bear management. Ph.D. thesis, University of Montana.

Hunter, J., Durant, S., Caro, T., 2007. To flee or not to flee: predator avoidance by cheetahs at kills. Behavioral Ecology and Sociobiology 61 (7), 1033–1042.

Hunter, L., 1998. Do male cheetahs commit infanticide? Cheetah News 8 (2), 4–5.

Hunter, L., Lindsey, P., Balme, G., Becker, M., Begg, C., Brink, H., Chardonnet, P., Dickman, A., Edwards, C., Frank, L., Funston, P., Henschel, P., Ikanda, D., Kissui, B., Loveridge, A., Mesochina, P.,

Midlane, N., White, P., Whitman-Gelatt, K., 2013. Urgent and comprehensive reform of trophy hunting of lions is a better option than an endangered listing; a science-based consenus. Tech. rep., Panthera, http://www.accuratereloading.com/2012/alusfw2.pdf.

Hunter, L., Skinner, J., 1995. A case of cannibalism in male cheetahs. African Journal of Ecology 33, 169–171.

Hutchinson, G., 1959. Homage to Santa Rosalia or Why are there so many kinds of animals? The American Naturalist 93 (870), 145–159.

IUCN/SSC, 2013. Guidelines for reintroductions and other conservation translocations. Tech. rep., IUCN Species Survival Commission.

Jackson, C., McNutt, J., Apps, P., 2012. Managing the ranging behaviour of African wild dogs (*Lycaon pictus*) using translocated scent marks. Wildlife Research 39, 31–34.

Jacobs, D., 2009. Animal personality and biological markets: rise of the individual. African Zoology 44 (2), 271–282.

Jaeger, M., 2004. Selective targeting of alpha coyotes to stop sheep depredation. Sheep & Goat Research Journal 19, 80–84.

Jansen, C., 2016. Diet of key predators responsible for livestock conflict in Namaqualand, South Africa. Master's thesis, Stellenbosch University.

Jaynes, E., 1968. Prior probabilities. IEEE Transactions on Systems Science and Cybernetics 4 (3), 227–241.

Jenner, N., Groombridge, J., Funk, S., 2011. Commuting, territoriality and variation in group and territory size in a black-backed jackal population reliant on a clumped, abundant food resource in Namibia. Journal of Zoology 284, 231–238.

Johnson, S., Marker, L., Mengersen, K., Gordon, C., Melzheimer, J., Schmidt-Küntzel, A., Nghikembua, M., Fabiano, E., Henghali, J., Wachter, B., 2013. Modeling the viability of the free-ranging cheetah population in Namibia: an object-oriented Bayesian network approach. Ecosphere 4 (7), 90.

Jones, C., 2000. Grazing management for healthy soils. In: Proceedings Stipa Inaugural National Grasslands Conference 'Better Pastures Naturally'. p. 1.

Joubert, E., Mostert, P., 1975. Distribution patterns and status of some mammals in South West Africa. Madoqua 9 (1), 5–24.

Jøsang, A., 2008. Conditional reasoning with subjective logic. Journal of Multiple-Valued Logic and Soft Computing 15 (1), 5–38.

Kamler, J., Davies-Mostert, H., Hunter, L., Macdonald, D., 2007. Predation on black-backed jackals (*Canis mesomelas*) by African wild dogs (*Lycaon pictus*). African Journal of Ecology 45, 667–668.

Kamler, J., Jacobsen, N., Macdonald, D., 2008. Efficiency and safety of soft catch traps for capturing black-backed jackals and excluding non-target species. South African Journal of Wildlife Research 38 (2), 113–116.

Kamler, J., Klare, U., Macdonald, D., 2012a. Seasonal diet and prey selection of black-backed jackals on a small-livestock farm in South Africa. African Journal of Ecology 50, 299–307.

Kamler, J., Stenkewitz, U., Klare, U., Jacobsen, N., Macdonald, D., 2012b. Resource partitioning among Cape foxes, bat-eared foxes, and black-backed jackals in South Africa. Journal of Wildlife Management 76 (6), 1241–1253.

Kamler, J., Stenkewitz, U., Macdonald, D., 2013. Lethal and sublethal effects of black-backed jackals on cape foxes and bat-eared foxes. Journal of Mammalogy 94 (2), 295–306.

Kaunda, S., 2001. Spatial utilization by black-backed jackals in southeastern Botswana. African Zoology 36 (2), 143–152.

Keehner, J., Wielgus, R., Maletzke, B., Swanson, M., 2015. Effects of male targeted harvest regime on sexual segregation in mountain lion. Biological Conservation 192, 42–47.

Kerley, G., Wilson, S., Balfour, D. (Eds.), 2018. Livestock predation and its management in South Africa: A scientific assessment. Centre for African Conservation Ecology.

Khorozyan, I., Ghoddousi, A., Soofi, M., Waltert, M., 2015. Big cats kill more livestock when wild prey reaches a minimum threshold. Biological Conservation 192, 268–275.

King, A., Wilson, A., Wilshin, S., Lowe, J., Haddadi, H., Halles, S., Morton, A., 2012. Selfish-herd behaviour of sheep under threat. Current Biology 22 (14), R561–R562.

King, L., 2006. An armour approach to the prevention of small-stock predation. In: Daly, B., Davies-Mostert, H., Davies-Mostert, W., Evans, S., Friedmann, Y., King, N., Snow, T., Stadler, H. (Eds.), Prevention is the Cure. Proceedings of a workshop on holistic management of human-wildlife conflict in the agricultural sector of South Africa. Endangered Wildlife Trust, pp. 56–59.

Kissui, B., 2008. Livestock predation by lions, leopards, spotted hyenas, and their vulnerability to retaliatory killing in the Maasai steppe, Tanzania. Animal Conservation 11 (5), 422–432.

Klare, U., Kamler, J., Stenkewitz, U., Macdonald, D., 2010. Diet, prey selection, and predation impact of black-backed jackals in South Africa. Journal of Wildlife Management 74 (5), 1030–1042.

Knopff, K., Boyce, M., 2007. Prey specialization by individual cougars in multiprey systems. In: Transactions of the North American Wildlife and Natural Resources Conference 5. Transactions of the North American Wildlife and Natural Resources Conference. pp. 194–220.

Kolar, B., 2005. Black-backed jackals hunt seals on the Diamond Coast, Namibia. Tech. Rep. 2, IUCN/SSC Canid Specialist Group, http://www.canids.org/canidnews/8/Black_backed_jackals_hunt_seals.pdf.

Krafte Holland, K., Larson, L., Powell, R., 2018. Characterizing conflict between humans and big cats *Panthera spp*: A systematic review of research trends and management opportunities. PLoS ONE 13 (9), e0203877.

Krofel, M., 2007. Opportunistic hunting behaviour of black-backed jackals in Namibia. African Journal of Ecology 46 (2), 220–222.

Landi, P., Minoarivelo, H., Brännström, Å., Hui, C., Dieckmann, U., 2018. Complexity and stability of ecological networks: a review of the theory. Population Ecology 60 (4), 319–345.

Lichtenfeld, L., Trout, C., Kisimir, E., 2015. Evidence-based conservation: predator-proof bomas protect livestock and lions. Biodiversity and Conservation 24 (3), 483–491.

Lieury, N., Ruette, S., Devillard, S., Alabaret, M., Drouyer, F., Baudoux, B., Millon, A., 2015. Compensatory immigration challenges predator control: An experimental evidence-based approach improves management. Journal of Wildlife Management 79 (3), 425–434.

Lindeman, R., 1942. The trophic-dynamic aspect of ecology. Ecology 23 (4), 399–417.

Lindeque, P., Nowell, K., Preisser, T., Brian, C., Turnbull, P., 1998. Anthrax in wild cheetahs in the Etosha NP, Namibia. In: ARC- Onderstepoort OIE International Congress, 9-15 August 1998. pp. 9–15.

Lindsey, P., Balme, G., Booth, V., Midlane, N., 2012. The significance of African lions for the financial viability of trophy hunting and the maintenance of wild land. PLoS ONE 7 (1), e29332.

Lindsey, P., Balme, G., Funston, P., Henschel, P., Hunter, L., Madzikanda, H., Midlane, N., Nyirenda, V., 2013a. The trophy hunting of African lions: Scale, current management practices and factors undermining sustainability. PLoS ONE 8 (9), e73808.

Lindsey, P., Havemann, C., Lines, R., Palazy, L., Price, A., Retief, T., Rhebergen, T., Van der Waal, C., 2013b. Determinants of persistence and tolerance of carnivores on Namibian ranches: Implications for conservation on Southern African private lands. PLoS ONE 8 (1), e52458.

Lindsey, P., Havemann, C., Lines, R., Price, A., Retief, E., Rhebergen, T., Van der Waal, C., Romañach, S., 2013c. Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. Oryx 47 (1), 41–53.

Linnell, J., 2000. Taste aversive conditioning: a comment. Carnivore Damage Prevention News 2, 4–5.

Linnell, J., Aanes, R., Swenson, J., 1997. Translocation of carnivores as a method for managing problem animals: a review. Biodiversity and Conservation 6 (9), 1245–1257.

Linnell, J., Odden, J., Smith, M., Aanes, R., Swenson, J., 1999. Large carnivores that kill livestock: Do "problem individuals" really exist? Wildlife Society Bulletin 27 (3), 698–705.

Linnell, J., Smith, M., Odden, J., Swenson, J., Kaczensky, P., 1996. Carnivores and sheep farming in Norway. 4. Strategies for the reduction of carnivore - livestock - conflicts: a review. Tech. rep., Norsk Institutt for Naturforskning.

Loveridge, A., Macdonald, D., 2001. Seasonality in spatial organization and dispersal of sympatric jackals (*Canis mesomelas* and *C. adustus*): implications for rabies management. Journal of Zoology 253 (1), 101–111.

Loveridge, A., Macdonald, D., 2002. Habitat ecology of two sympatric species of jackals in Zimbabwe. Journal of Mammalogy 83 (2), 599–607.

Lubbe, L., 2005. Towards an updated carrying capacity map for Namibia: a review of the methodologies currently used to determine carrying capacity in Namibia. Agricola 15 (6), 33–39.

Lubbe, L., Espach, C., 2006. An accurate grazing capacity map for Namibia – Myth or reality? Spotlight on Agriculture 100, 1–2.

Ludwig, D., Walker, B., Holling, C., 1997. Sustainability, stability, and resilience. Conservation Ecology 1 (1), 7.

Lukarevsky, V., 2003. Saving the Central Asian leopard in Turkmenistan. Carnivore Damage Prevention News 6, 13–15.

Luyt, E. d. C., 2004. Models of bontebok (*Damaliscus pygargus pygargus*, Pallas 1766) habitat preferences in the Bontebok National Park and sustainable stocking rates. Master's thesis, Stellenbosch University.

López-Bao, J., Rodrigues, A., 2014. Revisiting food-based models of territoriality in solitary predators. Journal of Animal Ecology 83 (4), 934–942.

Lötter, P., 2006. The use of cell-phone technology to prevent stock losses / Veldwagter veediefstal alarm. In: Daly, B., Davies-Mostert, H., Davies-Mostert, W., Evans, S., Friedmann, Y., King, N., Snow, T., Stadler, H. (Eds.), Prevention is the Cure. Proceedings of a workshop on holistic management of human-wildlife conflict in the agricultural sector of South Africa. Endangered Wildlife Trust, pp. 46–50.

MacArthur, R., 1955. Fluctuations of animal populations and a measure of community stability. Ecology 36 (9), 533–536.

MacArthur, R., Wilson, E., 1963. An equilibrium theory of insular zoogeography. Evolution 17 (4), 373–387.

Macaskill, C. (Ed.), 2013. Die agri handboek vir Suid-Afrika 2013/14. Department of Agriculture, Forestry and Fisheries (DAFF), http://www.agrihandbook.co.za/.

Macdonald, D., Loveridge, A. (Eds.), 2010. Biology and conservation of wild felids. Oxford University Press.

Madden, F., McQuinn, B., 2014. Conservation's blind spot: The case for conflict transformation in wildlife conservation. Biological Conservation 178, 97–106.

Margalef, R., 1963. On certain unifying principles in ecology. The American Naturalist 97 (897), 357–374.

Marker, L., 1998. Current status of the cheetah (*Acinonyx jubatus*). In: Proceedings of a Symposium on Cheetahs as Game Ranch Animals. pp. 1–17.

Marker, L., 2000a. Aspects of the ecology of the cheetah (*Acinonyx jubatus*) on North Central Namibian farmlands. Journal of Namibia Scientific society 48, 40–48.

Marker, L., 2000b. Donkeys protecting livestock in Namibia. Carnivore Damage Prevention News 2, 7–8.

Marker, L., 2002. Aspects of cheetah (*Acinonyx jubatus*) biology, ecology and conservation strategies on Namibian farmlands. Ph.D. thesis, University of Oxford.

Marker, L., Dickman, A., 2005a. Factors affecting leopard (*Panthera pardus*) spatial ecology, with particular reference to Namibian farmlands. South African Journal of Wildlife Research 35 (2), 105–115.

Marker, L., Dickman, A., 2005b. Notes on the spatial ecology of caracals (*FeIis caracal*), with particular reference to Namibian farmlands. African Journal of Ecology 43, 73–76.

Marker, L., Dickman, A., Jeo, R., Mills, M., Macdonald, D., 2003a. Demography of the Namibian cheetah, *Acinonyx jubatus jubatus*. Biological Conservation 114, 413–425.

Marker, L., Dickman, A., Macdonald, D., 2005a. Perceived effectiveness of livestock-guarding dogs placed on Namibian farms. Rangeland Ecology & Management 58 (4), 329–336.

Marker, L., Dickman, A., Macdonald, D., 2005b. Survivorship and causes of mortality for livestock-guarding dogs on Namibian rangeland. Rangeland Ecology & Management 58 (4), 337–343.

Marker, L., Dickman, A., Mills, M., Jeo, R., Macdonald, D., 2008. Spatial ecology of cheetahs on north-central Namibian farmlands. Journal of Zoology 274, 226–238.

Marker, L., Dickman, A., Mills, M., Macdonald, D., 2003b. Aspects of the management of cheetahs, *Acinonyx jubatus jubatus*, trapped on Namibian farmlands. Biological Conservation 114, 401–412.

Marker, L., Dickman, A., Schumann, M., 2005c. Using livestock guarding dogs as a conflict resolution strategy on Namibian farms. Carnivore Damage Prevention News 8, 28–32.

Marker, L., Mills, M., Macdonald, D., 2003c. Factors influencing perceptions of conflict and tolerance toward cheetahs on Namibian farmlands. Conservation Biology 17 (5), 1290–1298.

Marker, L., Muntifering, J., Dickman, A., Mills, M., Macdonald, D., 2003d. Quantifying prey preferences of free-ranging Namibian cheetahs. South African Journal of Wildlife Research 33 (1), 43–53.

Marker-Kraus, L., Kraus, D., 1994. The Namibian free-ranging cheetah. Environmental Conservation 21 (4), 369–370.

Marker-Kraus, L., Kraus, D., Barnett, D., Hurlbut, S., 1996. Cheetah survival on Namibian farmlands. Cheetah Conservation Fund.

Marks, C., Wilson, R., 2005. Predicting mammalian target-specificity of the M-44 ejector in south-eastern Australia. Wildlife Research 32 (2), 151–156.

Martins, Q., 2010. The ecology of the leopard *Panthera pardus* in the Cederberg Mountains. Ph.D. thesis, University of Bristol.

Martins, Q., Horsnell, W., Titus, W., Rautenbach, T., Harris, S., 2011. Diet determination of the Cape Mountain leopards using global positioning system location clusters and scat analysis. Journal of Zoology 283, 81–87.

Matich, P., Heithaus, M., Layman, C., 2011. Contrasting patterns of individual specialization and trophic coupling in two marine apex predators. Journal of Animal Ecology 80, 294–305.

MAWF, 2017. Agricultural statistics bulletin - 2010-2015. Tech. rep., Directorate of Planning and Business Development, Private Bag 13184, Government Office Park, Windhoek.

McDonald-Madden, E., Probert, W., Hauser, C., Runge, M., Possingham, H., Jones, M., Moore, J., Rout, T., Vesk, P., Wintle, B., 2010. Active adaptive conservation of threatened species in the face of uncertainty. Ecological Applications 20 (5), 1476–1489.

McGranahan, D., 2008. Managing private, commercial rangelands for agricultural production and wildlife diversity in Namibia and Zambia. Biodiversity and Conservation 17 (8), 1965–1977.

McGranahan, D., 2011. Identifying ecological sustainability assessment factors for ecotourism and trophy hunting operations on private rangeland in Namibia. Journal of Sustainable Tourism 19 (1), 115–131.

McLeod, S., 1997. Is the concept of carrying capacity useful in variable environments? Oikos 79 (3), 529–542.

McManus, J., Dickman, A., Gaynor, D., Smuts, B., Macdonald, D., 2014. Dead or alive? Comparing costs and benefits of lethal and non-lethal human–wildlife conflict mitigation on livestock farms. Oryx 49 (4), 687–695.

McVittie, R., 1979. Changes in the social behaviour of South West African cheetah. Madoqua 11 (3), 171–184.

Melville, H., Bothma, J. P., 2006. Using spoor counts to analyse the effect of small stock farming in Namibia on caracal density in the neighbouring Kgalagadi Transfrontier Park. Journal of Arid Environments 64, 436–447.

Melville, H., Bothma, J. P., Mills, M., 2004. Prey selection by caracal in the Kgalagadi Transfrontier Park. South African Journal of Wildlife Research 34 (1), 67–75.

Melzheimer, J., Streif, S., Wasiolka, B., Fischer, M., Thalwitzer, S., Heinrich, S., Weigold, A., Hofer, H., Wachter, B., 2018. Queuing, takeovers, and becoming a fat cat: Long-term data reveal two distinct male spatial tactics at different life-history stages in Namibian cheetahs. Ecosphere 9 (6), e02308.

Mendelsohn, J., Jarvis, A., Roberts, C., Robertson, T., 2003. Atlas of Namibia - A portrait of the land and its people. David Phillip Publishers, http://www.the-eis.com/.

Merrifield, J., 2012. The influence of group size and individual role on aspects of cooperative hunting in the Namibian black-backed jackal.

Miller, J., 2015. Mapping attack hotspots to mitigate human–carnivore conflict: approaches and applications of spatial predation risk modeling. Biodiversity and Conservation 24 (12), 2887–2911.

Miller, J., Jhala, Y., Jena, J., 2016a. Livestock losses and hotspots of attack from tigers and leopards in Kanha Tiger Reserve, Central India. Regional Environmental Change 16 (Suppl 1), S17–S29.

Miller, J., Jhala, Y., Jena, J., Schmitz, O., 2015. Landscape-scale accessibility of livestock to tigers: implications of spatial grain for modeling predation risk to mitigate human–carnivore conflict. Ecology and Evolution 5 (6), 1354–1367.

Miller, J., Stoner, K., Cejtin, M., Meyer, T., Middleton, A., Schmitz, O., 2016b. Effectiveness of contemporary techniques for reducing livestock depredations by large carnivores. Wildlife Society Bulletin 40 (4), 806–815.

Mills, M., 1998. Cheetah ecology and behaviour in East and South Africa. In: Proceedings of a Symposium on Cheetahs as Game Ranch Animals. pp. 18–22.

Mills, M., Broomhall, L., Du Toit, J., 2004. Cheetah *Acinonyx jubatus* feeding ecology in the Kruger National Park and a comparison across African savanna habitats: is the cheetah only a successful hunter on open grassland plains? Wildlife Biology 10 (3), 177–186.

Mills, M., Mills, M., 2014. Cheetah cub survival revisited: a re-evaluation of the role of predation, especially by lions, and implications for conservation. Journal of Zoology 292 (2), 136–141.

Minnie, L., Avenant, N., Drouilly, M., Samuels, I., 2018. Biology and ecology of the black-backed jackal and the caracal. In: Kerley et al. (2018), pp. 178–204.

Minnie, L., Avenant, N., Kamler, J., Butler, H., Parker, D., Drouilly, M., Du Plessis, J., Do Linh San, E., 2016a. A conservation assessment of *Canis mesomelas*. In: Child et al. (2016), p. 14 pp.

Minnie, L., Boshoff, A., Kerley, G., 2015. Vegetation type influences livestock predation by leopards: Implications for conservation in agro-ecosystems. African Journal of Wildlife Research 45 (2), 204–215.

Minnie, L., Gaylard, A., Kerley, G., 2016b. Compensatory life-history responses of a mesopredator may undermine carnivore management efforts. Journal of Applied Ecology 53, 379–387.

Mishra, C., Allen, P., McCarthy, T., Madhusudan, M., Bayarjargal, A., Prins, H., 2003. The role of incentive programs in conserving the snow leopard. Conservation Biology 17 (6), 1512–1520.

Mizuntani, F., 1999. Impact of leopards on a working ranch in Laikipia, Kenya. African Journal of Ecology 37, 211–225.

Moehlman, P., 1987. Social organization in jackals: The complex social system of jackals allows the successful rearing of very dependent young. American Scientist 75 (4), 366–375.

Montag, J., 2003. Compensation and predator conservation: Limitations of compensation. Carnivore Damage Prevention News 6, 2–6.

Moolman, L., 1984. 'n Vergelyking van die voedingsgewoontes van die rooikat *Felis caracal* binne en buite die Bergkwagga Nasional Park. Koedoe 27, 121–129.

Muntifering, J., Dickman, A., Perlow, L., Hruska, T., Ryan, P., Marker, L., Jeo, R., 2006. Managing the matrix for large carnivores: a novel approach and perspective from cheetah (*Acinonyx jubatus*) habitat suitability modelling. Animal Conservation 9, 103–112.

Murray, E., Merrifield, J., Hodge, S., Wiesel, I., 2012. The influence of group size on hunting and feeding success in the cooperatively hunting black-backed jackal. Master's thesis, University of Exeter, Cornwall Campus.

Musiani, M., Mamo, C., Boitani, L., Callaghan, C., Gates, C., Mattei, L., Visalberghi, E., Breck, S., Volpi, G., 2003. Wolf depredation trends and the use of fladry barriers to protect livestock in Western North America. Conservation Biology 17 (6), 1538–1547.

NAMMIC, 2011. Namibian livestock sector strategy - Final report. Tech. rep., Namibian Meat Board.

Nattrass, N., Conradie, B., 2015. Jackal narratives and predator control in the Karoo, South Africa. Journal of Southern African Studies 41 (4), 1–19.

Nattrass, N., Conradie, B., 2018. Predators, livestock losses and poison in the South African Karoo. Journal of Cleaner Production 194, 777–785.

Naughton-Treves, L., Grossberg, R., Treves, A., 2003. Paying for tolerance: Rural citizens' attitudes toward wolf depredation and compensation. Conservation Biology 17 (6), 1500–1511.

Nel, J., Loutit, R., Braby, R., Somers, M., 2013. Resource dispersion, territory size and group size of black-backed jackals on a desert coast. Acta Theriologica 58, 189–197.

Nemecek, T., Huguenen-Elie, O., Dubois, D., Gaillard, G., Schaller, B., Chervet, A., 2011. Life cycle assessment of Swiss farming systems: II. Extensive and intensive production. Agricultural Systems 104 (3), 233–245.

Nghikembua, M., Harris, J., Tregenza, T., Marker, L., 2016. Spatial and temporal habitat use by GPS collared male cheetahs in modified bushland habitat. Open Journal of Forestry 6 (4), 269–280.

Nyhus, P., Fischer, H., Madden, F., Osofsky, S., 2003. Taking the bite out of wildlife damage: The challenges of wildlife compensation schemes. Conservation in Practice 4, 37–40.

Nyhus, P., Osofsky, S., Ferraro, P., Madden, F., Fischer, H., 2005. Bearing the costs of human-wildlife conflict: The challenges of compensation schemes. In: Woodroffe, R., Thirgood, S., Rabinowitz, A. (Eds.), People and Wildlife: Conflict or Coexistence? Vol. 9. Cambridge University Press, Ch. 7, pp. 107–121.

Ogada, D., 2014. The power of poison: pesticide poisoning of Africa's wildlife. Annals of the New York Academy of Sciences 1322 (1), 1–20.

Ogada, M., Woodroffe, R., Oguge, N., Frank, L., 2003. Limiting depredation by African carnivores: the role of livestock husbandry. Conservation Biology 17 (6), 1521–1530.

Orams, M., 2002. Feeding wildlife as a tourism attraction: a review of issues and impacts. Tourism Management 23, 281–293.

Packer, C., Kosmala, M., Cooley, H., Brink, H., Pintea, L., Garshelis, D., Purchase, G., Strauss, M., Swanson, A., Balme, G., Hunter, L., Nowell, K., 2009. Sport hunting, predator control and conservation of large carnivores. PLoS ONE 4 (6), e5941.

Packer, C., Loveridge, A., Canney, S., Caro, T., Garnett, S., Pfeifer, M., Zander, K., Swanson, A., MacNulty, D., Balme, G., Bauer, H., Begg, C., Begg, K., Bhalla, S., Bissett, C., Bodasing, T., Brink, H., Burger, A., Burton, A., Clegg, B., Dell, S., Delsink, A., Dickerson, T., Dloniak, S., Druce, D., Frank, L., Funston, P., Gichohi, N., Groom, R., Hanekom, C., Heath, B., Hunter, L., Delongh, H., Joubert, C., Kasiki, S., Kissui, B., Knocker, W., Leathem, B., Lindsey, P., Maclennan, S., McNutt, J., Miller, S., Naylor, S., Nel, P., Ng'weno, C., Nicholls, K., Ogutu, J., Okot-Omoya, E., Patterson, B., Plumptre, A., Salerno, J., Skinner, K., Slotow, R., Sogbohossou, E., Stratford, K., Winterbach, C., Winterbach, H., Polasky, S., 2013a. Conserving large carnivores: dollars and fence. Ecology Letters 16 (5), 635–641.

Packer, C., Swanson, A., Canney, S., Loveridge, A., Garnett, S., Pfeifer, M., Burton, A., Bauer, H., MacNulty, D., 2013b. The case for fencing remains intact. Ecology Letters 16 (11), 1414–e4.

Paige, C., 2015. A Wyoming landowner's handbook to fences and wildlife: Practical tips for fencing with wildlife in mind. Wyoming Community Foundation, 2nd Edition.

Palmer, R., Fairall, N., 1988. Caracal and African wild cat diet in the Karoo National Park and the implications thereof for hyrax. South African Journal of Wildlife Research 18 (1), 30–34.

Palomares, F., Caro, T., 1999. Interspecific killing among mammalian carnivores. The American Naturalist 153 (5), 492–508.

Peers, M., Thornton, D., Murray, D., 2012. Reconsidering the specialist-generalist paradigm in niche breadth dynamics: Resource gradient selection by Canada lynx and bobcat. PLoS ONE 7 (12), e51488.

Pitman, R., 2012. The conservation biology and ecology of the African leopard *Panthera pardus pardus*. The Plymouth Student Scientist 5 (2), 281–600.

Potgieter, G., 2011. The effectiveness of livestock guarding dogs for livestock production and conservation in Namibia. Master's thesis, Nelson Mandela Metropolitan University.

Potgieter, G., Kerley, G., Marker, L., 2016. More bark than bite? The role of livestock guarding dogs in predator control on Namibian farmlands. Oryx 50 (6), 514–522.

Potgieter, G., Marker, L., Avenant, N., Kerley, G., 2013. Why Namibian farmers are satisfied with the performance of their livestock guarding dogs. Human Dimensions of Wildlife: An International Journal 18 (6), 403–415.

Potgieter, G., Weise, F., Wachter, B., Melzheimer, J., Wiesel, I., Stratford, K., 2017. Comment on Rust et al.: Human–carnivore conflict in Namibia is not simply about black and white. Society & Natural Resources 30 (10), 1299–1303.

Pringle, J., Pringle, V., 1979. Observations on the lynx *Felis caracal* in the Bedford district. South African Journal of Zoology 14 (1), 1–4.

Pulliam, H., 1988. Sources, sinks, and population regulation. The American Naturalist 132 (5), 652–661.

Pulliam, H., Danielson, B., 1991. Sources, sinks, and habitat selection: A landscape perspective on population dynamics. The American Naturalist 137, S50–S66.

Ray, J., Hunter, L., Zigouris, J., 2005. Setting conservation and research priorities for larger African carnivores. Tech. rep., Wildlife Conservation Society.

Reiss, J., Bridle, J., Montoya, J., Woodward, G., 2009. Emerging horizons in biodiversity and ecosystem functioning research. Trends in Ecology and Evolution 24 (9), 505–514.

Reznick, D., Bryant, M., Bashey, F., 2002. r- and K-selection revisited: The role of population regulation in life-history evolution. Ecology 83 (6), 1509–1520.

Rigg, R., Findo, S., Wechselberger, M., Gorman, M., Sillero-Zubiri, C., Macdonald, D., 2011. Mitigating carnivore–livestock conflict in Europe: lessons from Slovakia. Oryx 45 (2), 272–280.

Ripple, W., Estes, J., Beschta, R., Wilmers, C., Ritchie, E., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M., Schmitz, O., Smith, D., Wallach, A., Wirsing, A., 2014. Status and ecological effects of the world's largest carnivores. Science 343 (6167), 1241484.

Ritchie, E. G., Johnson, C. N., 2009. Predator interactions, mesopredator release and biodiversity conservation. Ecology Letters 12, 982–998.

Roberts, D., 1986. Determination of predators responsible for killing livestock. South African Journal of Wildlife Research 16 (4), 150–152.

Rohde, R., Hoffman, M., 2012. The historical ecology of Namibian rangelands: Vegetation change since 1876 in response to local and global drivers. Science of the Total Environment 416, 276–288.

Ropiquet, A., Knight, A., Born, C., Martins, Q., Balme, G., Kirkendall, L., Hunter, L., Senekal, C., Matthee, C., 2015. Implications of spatial genetic patterns for conserving African leopards. Comptes Rendus Biologies 338, 728–737.

Rothauge, A., 2011a. Ecological dynamics of Central Namibia's savannas: Part 1 - Grass ecology. Agricola 21, 7–13.

Rothauge, A., 2011b. Ecological dynamics of Central Namibia's savannas: Part 2 - Bush ecology. Agricola 21, 14–25.

Rowe-Rowe, D., 1982. Home range and movements of black-backed jackals in an African montane region. South African Journal of Wildlife Research 12, 79–84.

Rust, N., 2016. Can stakeholders agree on how to reduce human-carnivore conflict on Namibian livestock farms? A novel Q-methodology and Delphi exercise. Oryx 51 (2), 1–8.

Rust, N., Marker, L., 2013a. Attitudes toward predators and conservancies among Namibian farmers. Human Dimensions of Wildlife: An International Journal 18 (6), 463–468.

Rust, N., Marker, L., 2013b. Cost of carnivore coexistence on communal and resettled land in Namibia. Environmental Conservation 41 (1), 45–53.

Rust, N., Tzanopoulos, J., Humle, T., MacMillan, D., 2016. Why has human-carnivore conflict not been resolved in Namibia? Society & Natural Resources 29 (9), 1079–1094.

Rust, N., Whitehouse-Tedd, K., MacMillan, D., 2013. Perceived efficacy of livestock-guarding dogs in South Africa: Implications for cheetah conservation. Wildlife Society Bulletin 37 (4), 690–697.

Rutherford, M., Mucina, L., Powrie, L., 2006. Biomes and bioregions of Southern Africa. In: The Vegetation of South Africa, Lesotho and Swaziland: Strelitzia 19, strelitzia 19 Edition. South African National Biodiversity Institute, p. 51 pp.

RWCP, IUCN/SSC, 2015. Regional conservation strategy for the cheetah and African wild dog in Southern Africa; revised and updated, August 2015. Tech. rep., IUCN Species Survival Commission.

Sacks, B., Jaeger, M., Neale, J., McCullough, D., 1999. Territoriality and breeding status of coyotes relative to sheep predation. Journal of Wildlife Management 63 (2), 593–605.

Sacks, B., Neale, J., 2002. Foraging strategy of a generalist predator toward a special prey: Coyote predation on sheep. Ecological Applications 12 (1), 299–306.

Santangeli, A., Arkumarev, V., Rust, N., Girardello, M., 2016. Understanding, quantifying and mapping the use of poison by commercial farmers in Namibia – Implications for scavengers' conservation and ecosystem health. Biological Conservation 204 (B), 205–211.

Savory, A., 2013. Response to request for information on the 'science' and 'methodology' underpinning holistic management and holistic planned grazing. http://www.savoryinstitute.com/media/42448/Science-Methodology-Holistic-Mgt_Update_March.pdf.

Scantlebury, D., Mills, M., Wilson, R., Mills, M., Durant, S., Bennett, M., Bradford, P., Marks, N., Speakman, J., 2014. Flexible energetics of cheetah hunting strategies provide resistance against kleptoparasitism. Science 346 (6205), 79–81.

Schumann, M., 2004. Guide to integrated livestock and predator management. Cheetah Conservation Fund, P.O. Box 1755, Otjiwarongo, Namibia.

Sharp, N., 1997. Timed running speed of a cheetah (*Acinonyx jubatus*). Journal of Zoology, London 241, 493–494.

Shivik, J., 2004. Non-lethal alternatives for predation management. Sheep & Goat Research Journal 19, 64–71.

Shivik, J., 2006. Tools for the edge: What's new for conserving carnivores. BioScience 56 (3), 253–259.

Shivik, J., Tregenza, T., Callahan, P., 2003. Nonlethal techniques for managing predation: Primary and secondary repellents. Conservation Biology 17 (6), 1531–1537.

Shortridge, G., 1934. The mammals of South West Africa. Heinemann.

Shwiff, S., Bodenchuk, M., 2004. Direct, spillover, and intangible benefits of predation management. Sheep & Goat Research Journal 19, 50–52.

Simmons, R., Brown, C., Kemper, J., 2015. Birds to watch in Namibia: red, rare and endemic species. Ministry of Environment and Tourism and Namibia Nature Foundation.

Sinclair, A., Mduma, S., Brashares, J., 2003. Patterns of predation in a diverse predator–prey system. Nature 425, 288–290.

Sinclair, A., Mduma, S., Grant, J., Hopcraft, C., Fryxell, J., Hilborn, R., Thirgood, S., 2007. Long-term ecosystem dynamics in the Serengeti: Lessons for conservation. Conservation Biology 21 (3), 580–590.

Skead, C. J., 1980. Historical mammal incidence in the Cape Province. The Department of Nature and Environmental Conservation of the Provincial Administration of the Cape of Good Hope, Cape Town.

Smuts, B., 2008. Predators on livestock farms - A practical farmers' manual for non-lethal, holistic, ecologically acceptable and ethical management. Landmark Foundation, www.landmarkfoundation.org.za.

Smuts, G., 1978. Interrelations between predators, prey, and their environment. BioScience 28 (5), 316–320.

Snow, T., 2009. A systems-thinking based evaluation of predator conflict management on selected South African farms. Master's thesis, University of KwaZulu-Natal.

Soofi, M., Ghoddousi, A., Zeppenfeld, T., Shokri, S., Soufi, M., Egli, L., Jafari, A., Ahmadpour, M., Qashqaei, A., Waltert, M., Khorozyan, I., 2019. Assessing the relationship between illegal hunting of ungulates, wild prey occurrence and livestock depredation rate by large carnivores. Journal of Applied Ecology 56 (2), 365–374.

Spears, C., BFAP, Jul. 2013. JADAFA Agricultural outlook brief: Namibia's livestock markets. Tech. rep., Department of Agriculture, Forestry and Fisheries (DAFF) & Agbiz, http://jadafa.co.za.dedi81.cpt4.host-h.net/wp-content/uploads/2014/06/13_07-Namibia_livestock_brief_Final.pdf.

Stander, P., 1990a. A suggested management strategy for stock-raiding lions in Namibia. South African Journal of Wildlife Research 20 (2), 37–43.

Stander, P., 1990b. Notes on foraging habits of cheetah. South African Journal of Wildlife Research 20 (4), 130–132.

Stander, P., 1998. Spoor counts as indices of large carnivore populations: the relationship between spoor frequency, sampling effort and true density. Journal of Applied Ecology 35, 378–385.

Stander, P., 2008. Tourism and the conservation of desert lions in Namibia. Tech. rep., Desert Lion Conservation Project.

Stander, P., ||au, K., |ui, N., Dabe, T., Dabe, D., 1997a. Non-consumptive utilisation of leopards: Community conservation and ecotourism in practice. In: Proceedings of a Symposium on Lions and Leopards as Game Ranch Animals. Desert Lion Conservation Project, Desert Lion Conservation Project, pp. 50–57.

Stander, P. E., Haden, P. J., Kaqece, ||., Ghau, ||., 1997c. The ecology of asociality in Namibian leopards. Journal of Zoology, London 242, 343–364.

Stein, A., 2008. Ecology and conservation of the leopard (*Panthera pardus* Linnaeus 1758) in Northcentral Namibia. Ph.D. thesis, University of Massachusetts Amherst.

Stein, A., Athreya, V., Gerngross, P., Balme, G., Henschel, P., Karanth, U., Miquelle, D., Rostro-García, S., Kamler, J., Laguardia, A., Khorozyan, I., Ghoddousi, A., 2015. *Panthera pardus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://www.iucnredlist.org/details/summary/15954/0, p. e.T15954A102421779.

Stein, A., Fuller, T., Damery, D., Sievert, L., Marker, L., 2010. Farm management and economic analyses of leopard conservation in north-central Namibia. Animal Conservation 13, 419–427.

Stein, A., Fuller, T., DeStefano, S., Marker, L., 2011. Leopard population and home range estimates in north-central Namibia. African Journal of Ecology 49, 383–387.

Stein, A., Fuller, T., Marker, L., 2013. Brown hyaena feeding ecology on Namibian farmlands. South African Journal of Wildlife Research 43 (1), 27–32.

Stein, A., Hayssen, V., 2013. *Panthera pardus* (Carnivora: Felidae). Mammalian Species 45 (900), 30–48.

Stone, S., Breck, S., Timberlake, J., Haswell, P., Najera, F., Bean, B., Thornhill, D., 2017. Adaptive use of nonlethal strategies for minimizing wolf–sheep conflict in Idaho. Journal of Mammalogy 98 (1), 33–44.

Stone, S., Fascione, N., Miller, C., Pissot, J., Schrader, G., Timberlake, J., 2008. Livestock and wolves - A guide to nonlethal tools and methods to reduce conflicts. Defenders of Wildlife.

Stuart, C., 1976. Diet of the black backed jackal *Canis Mesomelas* in the central Namib Desert, South West Africa. Zoologica Africana 11 (1), 193–205.

Stuart, C., 1981. Notes on the mammalian carnivores of the Cape Province, South Africa. Bontebok 1, 1–59.

Stuart, C., 1982. Aspects of the biology of the Caracal (*Felis caracal* Schreber 1776) in the Cape Province, South Africa. Master's thesis, University of Natal.

Stuart, C., Shaughnessy, P., 1984. Content of *Hyaena brunnea* and *Canis mesomelas* scats from southern coastal Namibia. Mammalia 48 (4), 611–612.

Swanepoel, J., 2016. Habits of the hunters: the biopolitics of combatting predation amongst small stock farmers in southern Namibia. Journal of Contemporary African Studies 34 (1), 129–146.

Swanepoel, L., Balme, G., Williams, S., Power, R., Snyman, A., Gaigher, I., Senekal, C., Martins, Q., Child, M., 2016. A conservation assessment of *Panthera pardus*. In: Child et al. (2016), p. 13 pp.

Swanepoel, L., Lindsey, P., Somers, M., Van Hoven, W., Dalerum, F., 2013. Extent and fragmentation of suitable leopard habitat in South Africa. Animal Conservation 16 (1), 41–50.

Swanepoel, L., Somers, M., Van Hoven, W., Schiess-Meier, M., Owen, C., Snyman, A., Martins, Q., Senekal, C., Camacho, G., Boshoff, W., Dalerum, F., 2015. Survival rates and causes of mortality of leopards *Panthera pardus* in southern Africa. Oryx 49 (4), 595–603.

Sweet, J., 1998. Livestock - Coping with drought: Namibia - A case study. In: Livestock - Coping with drought. FAO, FAO/AGAP, pp. 1–32.

Tainton, N. (Ed.), 1999. Veld management in South Africa. University of Natal Press.

Tambling, C., Avenant, N., Drouilly, M., Melville, H., 2018. The role of mesopredators in ecosystems: Potential effects of managing their populations on ecosystem processes and biodiversity. In: Kerley et al. (2018), Ch. 8, pp. 205–227.

Terborgh, J., 2015. Toward a trophic theory of species diversity. PNAS 112 (37), 11415–11422.

Treves, A., 2009. Hunting for large carnivore conservation. Journal of Applied Ecology 46, 1350–1356.

Treves, A., Karanth, K., 2003. Human-carnivore conflict and perspectives on carnivore management worldwide. Conservation Biology 17 (6), 1491–1499.

Treves, A., Krofel, M., McManus, J., 2016. Predator control should not be a shot in the dark. Frontiers in Ecology and the Environment 14 (7), 380–388.

Turpie, J., Akinyemi, B., 2018. The socio-economic impacts of livestock predation and its prevention in South Africa. In: Kerley et al. (2018), pp. 53–81.

Urbigkit, C., Urbigkit, J., 2010. A review: The use of livestock protection dogs in association with large carnivores in the Rocky mountains. Sheep & Goat Research Journal 25, 1–8.

Van Bommel, L., 2010. Guardian dogs: Best practice manual for the use of livestock guardian dogs. University of Canberra, Kirinari Street, Bruce ACT 2617.

Van Bommel, L., Johnson, C. N., 2012. Good dog! Using livestock guardian dogs to protect livestock from predators in Australia's extensive grazing systems. Wildlife Research 39, 220–229.

Van der Merwe, V., Marnewick, K., Bissett, C., Groom, R., Mills, M., Durant, S., 2016. A conservation assessment of *Acinonyx jubatus*. In: Child et al. (2016), p. 11 pp.

Van Eeden, L., Ecklund, A., Miller, J., López-Bao, J., Chapron, G., Cejtin, M., Crowther, M., Dickman, C., Frank, J., Krofel, M., Macdonald, D., McManus, J., Meyer, T., Middleton, A., Newsome, T., Ripple, W., Ritchie, E., Schmitz, O., Stoner, K., Tourani, M., Treves, A., 2018. Carnivore conservation needs evidence-based livestock protection. PLoS Biology 16 (9), e2005577.

Van Heezik, Y., Seddon, P., 1998. Range size and habitat use of an adult male caracal in northern Saudi Arabia. Journal of Arid Environments 40, 109–112.

Van Niekerk, H., 2010. The cost of predation on small livestock in South Africa by medium-sized predators. Master's thesis, University of the Free State.

Veblen, K., Porensky, L., Riginos, C., Young, T., 2016. Are cattle surrogate wildlife? Savanna plant community composition explained by total herbivory more than herbivore type. Ecological Applications 26 (6), 1610–1623.

Vesey-FitzGerald, D., 1960. Grazing succession among East African game animals. Journal of Mammalogy 41 (2), 161–172.

Voigt, C., Krofel, M., Menges, V., Wachter, B., Melzheimer, J., 2018. Sex-specific dietary specialization in a terrestrial apex predator, the leopard, revealed by stable isotope analysis. Journal of Zoology 306 (1), 1–7.

Von Wehrden, H., Hanspach, J., Kaczensky, P., Fischer, J., Wesche, K., 2012. Global assessment of the non-equilibrium concept in rangelands. Ecological Applications 22 (2), 393–399.

Wachter, B., Jauernig, O., Breitenmoser, U., 2006. Determination of prey hair in faeces of free-ranging Namibian cheetahs with a simple method. Cat News 44, 8–9.

Walker, E., Nghikembua, M., Bibles, B., Marker, L., 2016. Scent-post preference of free-ranging Namibian cheetahs. Global Ecology and Conservation 8, 55–57.

Wallace, K., Leslie, A., Coulson, T., 2011. Living with predators: a focus on the issues of human–crocodile conflict within the lower Zambezi valley. Wildlife Research 38, 747–755.

Wallach, A., Ramp, D., O'Neill, A., 2017. Cattle mortality on a predator-friendly station in central Australia. Journal of Mammalogy 98 (1), 45–52.

Ward, D., 2005. Do we understand the causes of bush encroachment in African savannas? African Journal of Range & Forage Science 22 (2), 101–105.

Weilenmann, M., Gusset, M., Mills, D., Gabanapelo, T., Schiess-Meier, M., 2010. Is translocation of stock-raiding leopards into a protected area with resident conspecifics an effective management tool? Wildlife Research 37, 702–707.

Weise, F., Hayward, M., Casillas Aguirre, R., Tomeletso, M., Gadimang, P., Somers, M., Stein, A., 2018. Size, shape and maintenance matter: A critical appraisal of a global carnivore conflict mitigation strategy - Livestock protection kraals in northern Botswana. Biological Conservation 225, 88–97.

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

Weise, F., Lemeris, J. J., Stratford, K., Van Vuuren, R., Munro, S., Crawford, S., Marker, L., Stein, A., 2015b. A home away from home: insights from successful leopard (*Panthera pardus*) translocations. Biodiversity and Conservation 24, 1755–1774.

Weise, F., Stratford, K., Van Vuuren, R., 2014. Financial costs of large carnivore translocations – Accounting for conservation. PLoS ONE 9 (8), e105042.

Weise, F., Vijay, V., Jacobson, A., Schoonover, R., Groom, R., Horgan, J., Keeping, D., Klein, R., Marnewick, K., Maude, G., Melzheimer, J., Mills, G., Van der Merwe, V., Van der Meer, E., Van Vuuren, R., Wachter, B., Pimm, S., 2017. The distribution and numbers of cheetah (*Acinonyx jubatus*) in southern Africa. PeerJ 5, e4096.

Westgate, M., Likens, G., Lindenmayer, D., 2013. Adaptive management of biological systems: A review. Biological Conservation 158, 128–139.

Westoby, M., Walker, B., Noy-Meir, I., 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42 (4), 266–274.

Wezel, A., Soboksa, G., McClelland, S., Delespesse, F., Boissau, A., 2015. The blurred boundaries of ecological, sustainable, and agroecological intensification: a review. Agronomy for Sustainable Development 35 (4), 1283–1295.

Wilson, A., Lowe, J., Roskilly, K., Hudson, P., Golabek, K., McNutt, J., 2013. Locomotion dynamics of hunting in wild cheetahs. Nature 498, 185–189.

Woodroffe, R., Frank, L., Lindsey, P., ole Ranah, S., Romañach, S., 2007. Livestock husbandry as a tool for carnivore conservation in Africa's community rangelands: a case–control study. Biodiversity and Conservation 16, 1245–1260.

Yarnell, R., Phipps, W., Burgess, L., Ellis, J., Harrison, S., Dell, S., MacTavish, D., MacTavish, L., Scott, D., 2013. The influence of large predators on the feeding ecology of two African mesocarnivores: the black-backed jackal and the brown hyaena. South African Journal of Wildlife Research 43 (2), 155–166.

Zhang, F., Hui, C., 2014. Recent experience-driven behaviour optimizes foraging. Animal Behaviour 88, 13–19.

Chapter 3

Farmers' perspectives on human-wildlife conflict in Namibia

Abstract

Namibian farmers were surveyed to estimate farmer-predator conflict. Since most cheetahs and leopards are found outside protected areas, continued human-wildlife conflict poses a risk to their long-term survival. Livestock depredation was reportedly the major cause of livestock loss on commercial Namibian farms. Leopards (*Panthera pardus*), cheetahs (*Acinonyx jubatus*), caracals (*Caracal caracal*) and black-backed jackals (*Canis mesomelas*) were the primary predators of livestock, with leopards being the primary predator of cattle, and jackals of small livestock. On average 9.2 Livestock Units were killed by predators per farm per year, with a few farms losing more than double this average. Approximately 70% of farmers killed predators as part of management, but the results suggest that this approach is ineffective. Twenty predation management methods used by Namibian farmers were evaluated and most were not statistically effective. Herding led to a significant decrease in livestock losses ($p < 0.05$, resampling t-test, 10 000 iterations) and a small, but significant decrease in predation losses was associated with higher prey diversity ($p < 0.05$, resampling regression test, 10 000 iterations). Each method was also assigned a prior success probability for use in a Decision Support System. With farmers having little idea about the costs of anti-predation methods, cost-effective and ecologically sustainable methods need to be identified, aiming at lower livestock losses and sustainable ecosystems.

Keywords: Brown hyaena, caracal, cheetah, farmer-predator conflict, human-wildlife conflict, jackal, leopard, livestock depredation, mitigation methods, Namibia, questionnaire

3.1 Introduction

Farmers, as the custodians of most of the land in countries where agriculture is a major economic activity, should be natural partners in the conservation of wildlife (Hey, 1964; Hoogesteijn and Hoogesteijn, 2010; WWF, 2016). However, human-wildlife conflict (HWC) between livestock farmers and predators is a worldwide problem (Treves and Karanth, 2003b), and may be the primary threat to carnivores in Africa (Ray et al., 2005). It is also the major cause of death for Namibian cheetahs (*Acinonyx jubatus*) (Marker et al., 2010). Human-wildlife conflict is increasingly responsible for conflict between conservationists and farmers (Treves and Karanth, 2003a; Nattrass and Conradie, 2015). One reason for the breakdown of trust between farmers and wildlife managers is the uncertainty about how great the effect of predators on livestock farming truly is (Madden and McQuinn, 2014). Conservationists often think that livestock farmers are exaggerating their losses, while farmers feel that conservationists are not sympathetic and offer impractical solutions to their very real problems with predators

(Nattrass and Conradie, 2015). An important step in reducing the subjectivity inherent in this conflict is gathering better data on the actual and relative losses to predation.

In 2010, a study was conducted in South Africa to determine the economic effects of depredation on small livestock farming in that country (Van Niekerk, 2010). A telephone survey of the five main small livestock producing provinces estimated a total direct loss of R 1 390 453 062 per year, equivalent to 11.45 livestock units (LSU) per farm per year according to the definition of the Food and Agriculture Organization of the United Nations (FAO) (A North American cow *Bos taurus* is 1 LSU, a sub-Saharan cow is equal to 0.5 LSU and a goat *Capra aegagrus hircus* or sheep *Ovis aries* equal to 0.1 LSU – Chilonda and Otte, 2006). However, many farmers do not keep good records of their losses and typically there is no way to distinguish between reliable, documented data and an exaggerated estimation. While total losses can be verified through the state veterinarian and cattle inspectors, it is not possible for predation losses specifically. In Namibia, there have been only limited farmer surveys to determine depredation rates: 1) in North-Central Namibia (Marker et al., 2003a), with an emphasis on cheetahs, 2) in an area around the Waterberg National Park (Stein et al., 2010), focussing on leopard predation, and 3) a country-wide survey of commercial farmers (Lindsey et al., 2013a), with an emphasis on farmer attitudes and determinants of tolerance towards predators. The potential income for the farmer is actually the percentage of offspring per females and the real impact of predators should be measured by their impact on this number, rather than simply the number of losses. So far only a few surveys have done this. The estimated average loss per farm according to the literature varied between 1.75 LSU/farm/year and 10.65 LSU/farm/year (Table 1.1 on page 5).

In this chapter the focus is on the agricultural management part of the agroecological model (Figure 1.4 on page 12), but with a wider perspective than only mitigation methods (*cf.* Chapter 2). Within a holistic agroecological system a number of important questions were identified that could influence the mitigation of farmer-predator conflict in Namibia (from Section 1.6.1 on page 18): 1) determine which anti-predation methods are currently used by Namibian farmers; 2) determine current livestock depredation losses by Namibian farmers; 3) determine any changes in predator numbers reported by Namibian farmers on their land; 4) determine current stocking rates and/or farming intensity on Namibian farms; 5) determine relative costs of each anti-predation method used in Namibia; 6) determine the advantages and disadvantages of anti-predation methods used worldwide from an agricultural perspective (short-term cost-effectiveness); 7) determine the relative effectiveness of each method from an agricultural viewpoint.

Twenty-nine anti-predation methods which had been used in Namibia, were identified from the literature in Chapter 2 (the first question above). Additionally, the last two questions were addressed from the literature (advantages, disadvantages and prior probability of success of each method), but there was still uncertainty about the effectiveness of many of the methods in the Namibian context (see Table 2.1 on page 47 of Chapter 2). Considering the wide range in estimated predation losses reported in previous studies, there was a need for a new survey to:

1. determine the current stocking rates and farming intensity on Namibian farms (some conflict mitigation techniques require more intensive farming),
2. determine if any changes in predator numbers on their land are seen by Namibian farmers (e.g. trends in threatened predator populations, possibly evidence of mesopredator release),
3. determine the current predation livestock losses by Namibian farmers (higher losses imply that higher-cost mitigation measures will make economic sense),
   (a) compared to other causes of livestock loss,
   (b) summed up to give total annual costs of livestock depredation in the country, and
   (c) compared by district to find possible conflict hotspots in Namibia,

4. determine the relative costs of each anti-predation method used in Namibia (costs differ between countries and are often not reported in the literature),
5. determine the relative effectiveness of each method from an agricultural viewpoint (prior probability of success from a Namibian perspective, rather than general international research).

In a Decision Support System all of these factors could influence the land manager's choice of a predation management method. Thorn et al. (2012, 2015) revealed that livestock depredation was not the most important determining factor of human-predator conflict in the North West Province of South Africa, but cultural group and land-use had the greatest effect on attitudes and actions towards predators. By contrast, a number of studies showed a correlation between livestock losses and predators being killed (Ogada et al., 2003; Conradie and Piesse, 2013; Lindsey et al., 2013a; Miller et al., 2016a; Santangeli et al., 2016; Teichman et al., 2016). A Decision Support System that can help farmers to reduce livestock losses would not really mitigate human-wildlife conflict if a decrease in livestock depredation did not result in less conflict. Since their cultural group is outside the control of farmers, this chapter focuses on those management aspects that are within the control of the individual decision makers.

3.2 Materials and methods

3.2.1 Study area

Namibia is an arid country with one of the lowest human population densities on earth. It is therefore not well suited for crop farming except in the far North where mostly communal subsistence farming can be found (Mendelsohn et al., 2003; MAWF, 2017; see first map – Figure 3.1). Livestock farming is the main source of livelihood for most of Namibia with the agricultural sector being the greatest provider of employment in the country (Chiriboga et al., 2008).

The commercial farmlands of Namibia include parts of 16 districts in eight of Namibia's 13 regions. Commercial farming areas are found mostly in the two biomes of the Nama-Karoo from the south to north-west of the country and the tree-and-shrub savannah in the eastern, central and northern parts of Namibia (Mendelsohn et al., 2003). The mean rainfall varies from 0 – 50 mm in the far south-west to 550 – 600 mm per year in the Tsumeb, Grootfontein and Otavi triangle in the north-east (Mendelsohn et al., 2003). This rainfall pattern also serves as a rough indicator of the kind of livestock farming found on the land – below 300 mm rain per annum small livestock (sheep and goats) predominate while above 300 mm (around Windhoek) cattle become more common (Mendelsohn et al., 2003; NAMMIC, 2011).

3.2.2 Questionnaires

Questionnaires were sent out to all 2500 members of the Namibian Agricultural Union (NAU) as an insert in the May 2015 edition of a monthly magazine, AgriForum (see Appendix A). Ethical clearance was obtained from Stellenbosch University (SU-HSD-000277). The questionnaire included about 30 questions on the general location (district), size and type of farming, followed by questions on numbers of livestock, livestock losses, methods used to prevent livestock depredation, wildlife on the farm (including carnivores), costs of current methods used and the willingness of the farmers to change to different predation management techniques. A short article, explaining the research and asking for information from farmers, was placed in the magazine. The questionnaire had the option to be filled in anonymously; it was made clear that no personal data would be published without the permission of the farmer and that the results of the research would be shared with the NAU. There was also the option of filling in an online version of the questionnaire or downloading the questionnaire from a website. With only about 66% of NAU members having an e-mail address (Wallie Roux,

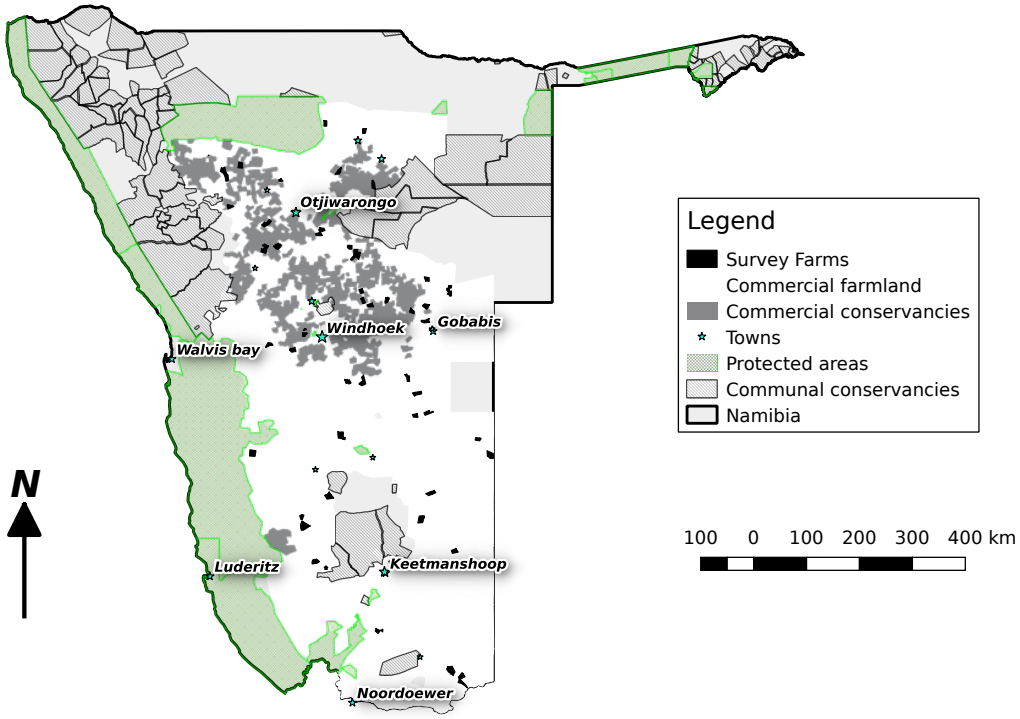


Figure 3.1: *Map showing commercial and communal farming areas of Namibia.* Commercial farming (in white on the map) occupies about 43% of the country (McGranahan, 2011). The protected areas in the west consist mostly of the Namib Desert. Communal conservancies occupy about 16.1% of Namibia's 823 680 km^2 land area, 6.1% of the total land area are freehold conservancies in the commercial farming area, and 16.5% of Namibia's land area consists of national parks & game reserves (Davis et al., 2011).

personal communication), the magazine was considered as the most cost-effective method to reach the majority of the farmers in the country. As farmers are notorious for a low response rate (average of only 25% to NAU questionnaires – Wallie Roux, personal communication), the ideal would have been personal interviews, but the logistics and size of Namibia made this impossible within the financial constraints of the study.

As an indication of how intensive the farming system was, the time since the last livestock count was used (see text box *Intensive vs extensive farming* on page 140). Farmers could also explicitly state when livestock numbers were only an estimate. Similarly, for livestock losses they could fill in both documented livestock losses and estimated livestock losses (e.g. when no remains were found and it was assumed that predators removed the whole carcass). This was to address the methodological flaw of previous studies, which did not differentiate between those farmers who documented losses accurately and those who simply estimated their losses (see Appendix A for the questionnaire used).

Not all 29 predation management methods that had been used in Namibia before (as identified in Chapter 2), are still commonly used. In the questionnaire, farmers were asked which of the following methods they used and how (*cf.* Table 2.1 in Chapter 2):

1. Hunting for extermination (combination of killing predators and more than 50% of the farm having jackal-proof or electric fencing).
2. Hunting with dogs.
3. Poison bait.

4. Carnivore-proof fencing.
5. Hunting predators opportunistically (to reduce density).
6. Trapping.
7. Human herders.
8. Kraaling all livestock at night.
9. Kraaling young livestock permanently.
10. Electric fencing.
11. Calving/lambing camps.
12. Numbers of natural prey (as estimated by farmers – generally only conservancy members are committed to regular game counts).
13. Livestock guarding dogs
14. Guarding donkeys.
15. Avoiding predation hotspots.
16. Seasonal breeding.
17. Livestock type (species or breed).
18. Horned cattle.
19. Large herds.
20. Hunting reactively only.
21. Translocation of predators.

3.2.3 Analysis

The completed questionnaires were translated from Afrikaans (except for the online version, all responses were in Afrikaans), entered into a spreadsheet and imported into QGIS (QGIS-Development-Team, 2017) and R (Kerns, 2010; Peng, 2016). Since the livestock loss data were not normally distributed ($p<0.001$; Anderson-Darling normality test and Shapiro-Wilk normality test – Figure 3.3), resampling (randomization) statistics were used instead of parametric tests to compare livestock depredation losses among farmers using a certain method to those farmers not using the method (Simon, 1997). Resampling statistics do not assume any theoretical distribution (normal or otherwise) of the data in the population and can thus be used to determine the probability of seeing the observed difference between the two groups by random chance alone (Yu, 2003). For all tests the data was resampled 10 000 times. (Parametric statistics were also performed on the data that had a normal distribution when log-transformed, for comparison – only mentioned in the results if it differed from the resampling tests). Although the questionnaires differentiated between documented losses and estimates, in the analysis total losses (including both documented and estimated) were used — except where the two types of data were explicitly compared to each other and when comparing the relative impact of the predator species. The reason for including both documented and estimated losses was to enable comparison with previous studies (which did not make this distinction).

Intensive vs extensive farming — Farming can be categorised on a scale from intensive to extensive (see Section 2.2.3 in Chapter 2). The practicality of various anti-predation methods may be limited by how intensive the farm can be managed (Figure 2.2 in Chapter 2). However, there is no universal classification system. Here intensive livestock farming systems were defined as those where all livestock were seen or handled at least daily; medium intensive systems as those where the livestock owner saw all livestock at least once per week; medium extensive systems as those where the farmer saw all livestock at least once per month; and extensive farming as those where the farmer saw or handled all livestock less than once per month. It should be noted that most Namibian farmers *see* ***some*** *of their livestock* every single day, but are unable *to handle or count* ***all*** their livestock on a daily basis because of the low stocking densities typical of extensive farming systems (various, personal communication).

Stocking rates and farming intensity were reported with descriptive statistics only (mean, standard deviation, range). While it can be assumed that farmers have a fairly good idea of their livestock numbers and losses (in order to be profitable), their estimates of predator numbers on their farms are often based on indirect impressions like livestock losses, rather than on actual observations or signs (Marker et al., 2003a; Conradie and Nattrass, 2017; personal observation). Therefore, trends in predator observations by landowners were not analysed for statistical significance since it does not describe actual numerical data. Estimated predator observations were compared to reported livestock depredation losses using a resampling version of a regression test (see section 3.3.3 below on page 142). Current depredation livestock losses in Namibia were given as descriptive statistics (mean, median, range, with standard deviation (SD) for means and median absolute deviation (MAD) for medians). To compare multiple groups (predation losses) statistically (e.g. between districts/regions/potential conflict hotspots and between causes of livestock losses), resampling on the F statistic was used to determine a non-parametric analysis of variance (ANOVA). Differences in livestock species and age classes taken by different predator species were tested using a Pearson's Chi-squared Test with Monte Carlo simulation (χ^2). Costs for each conflict mitigation method, where available, would be described by using descriptive statistics. For all statistics, using the mean would include the effects of the outliers (those farms with extraordinary livestock losses) while using the median can be considered as excluding their effect.

All mitigation methods except the effect of natural prey were analysed using one-tailed resampling of the difference between the means and medians of total livestock losses (in LSU) per farm. For each anti-predation method reported by the farmers, the null hypothesis was that the method would have no effect (i.e. a random redistribution of the farms between the two groups would show a similar difference in average livestock losses at least 5% of the time). The alternative hypothesis was that the specific method resulted in lower average livestock losses compared to farmers not using the method (than what would be expected from a random assignment of the farms to the two groups using *resampling without replacement*). One-tailed resampling without replacement tests were used because it was hypothesised that an effective method would lead to a decrease in livestock losses (and not just a difference). The independent categorical variable would every time be the method used, with the test repeated for both the percentage of offspring (reproduction) killed and the total livestock losses due to predation (as LSU) as dependent variables. Mann-Whitney U tests were not used, because unlike resampling, it uses ranks rather than actual values and in most cases has less power (Ibrahim, 1991).

Where the "effort" of the method (e.g. wildlife numbers as independent variable) could be represented by a continuous variable (instead of categorical groups of treatment and control), a resampling version of a regression test was used. Here the null hypothesis was that the observed *slope* of the regression will not be greater than that obtained by a random reassignment of the farms. The alternative hypothesis was that a similar or greater negative regression slope has a chance of less than 0.05 to be produced by a random reassignment of the farms between the treatment and control groups. Essentially the same resampling method was used to test for correlation between livestock losses and predators killed, but using the correlation coefficient instead of the slope of the regression. Correlation rather than regression was used, because the dependent *versus* independent variable is uncertain — do killing more predators result in higher livestock losses due to disturbing the territorial

behaviour or compensatory life histories of predators (Conradie and Piesse, 2013; Minnie et al., 2016; Teichman et al., 2016) or do farmers who experience higher livestock losses kill more predators (Ogada et al., 2003; Lindsey et al., 2013a; Miller et al., 2016a; Santangeli et al., 2016)? Experimental studies with adequate controls would be required to explain any correlation between livestock losses and predators being killed – a survey could not address this question satisfactorily (Sinclair et al., 2003; Miller et al., 2016b; Treves et al., 2016; Van Eeden et al., 2018).

As an indication of prior probability of success that could be used in a Decision Support System (Section 1.5 in Chapter 1), the proportion of farmers using a method where the livestock losses (as percentage of increase) were below the median of all farmer losses, was used. The raw data (excluding personal information) and basic r scripts used for the resampling, are included in Appendix B (Sections B.1.0.1, B.1.0.2, B.1.0.3, B.2.0.6).

3.3 Results

3.3.1 General

The farmers response rate was lower than expected (2.1%), with only 52 responses (3 online responses and 49 paper responses). Only 42 respondents completed the questionnaire in full. However, the responses were well distributed across the whole commercial farming area of Namibia (Figure 3.2) with 34 of the 104 farmers associations in Namibia (Lindsey et al., 2013a) and 13 of the 16 districts being represented. Contrary to expectations, only 9 of the 52 responses (17.3%) chose to fill in the questionnaire anonymously. Thirty-three farmers (63.5%) were situated in the traditional cattle farming area of Namibia (Windhoek and Northward) and 19 (36.5%) were from the South (Mendelsohn et al., 2003; NAMMIC, 2011). The farms in the North (Windhoek & Gobabis and Northward) reported a significantly higher number of wild herbivore species ($n = 33$, 6.76 ± 2.41 SD species) on their farms ($n = 47$, $p < 0.05$, using two-tailed resampling of mean) than farms in the South ($n = 13$, 4.86 ± 2.03 SD species).

Only 23 (44.2%) of the respondents were commercial cattle farmers (farming with 200 or more head of cattle), although 40 respondents (77.0%) had some cattle. Twenty-two (42.3%) farmed with game, but only 13 (25.0%) used trophy hunting as a source of income. Nineteen respondents (36.5%) were primarily small livestock farmers (having at least 200 does or at least 400 ewes), while 27 (51.9%) had some small livestock. Nineteen (36.5%) of the farmers described their farming enterprise as being mixed farming with only one farmer (1.9%) also having some crop cultivation. Most of the responding farmers (65.0%) could only estimate their livestock numbers and did not have data from recent counts. Based on farmer responses, livestock species differed in their potential profitability (offspring as percentage of females) as shown in Table 3.1 on page 143. This is one important factor in farmers' livestock choices and may partly explain the reluctance of some farmers to change livestock species or breed as a mitigating method to reduce livestock depredation (NAMMIC, 2011).

3.3.2 Stocking rates and farming intensity

From the responses received only three farmers (5.8%) counted all their livestock at least weekly (= medium intensive farming), nine farmers (17.3%) counted their livestock at least monthly (= medium extensive farming) and 40 farmers (76.9%) counted all their livestock less than once per month (= extensive farming). The mean total farm size was 9 344.69 ha ($\pm$4231.48 SD, $n = 48$), with the smallest farm 4 339 ha and the largest 22 000 ha in size. Livestock stocking densities varied between 0.00114 LSU/ha (=877.19 ha/LSU) and 0.0954 LSU/ha (10.486 ha/LSU) with a mean stocking density of 0.0339 ($\pm$0.022 SD, $n = 40$) LSU/ha (29.52, $\pm$135.23 ha/LSU) (Table 3.1). This wide variation in livestock numbers underlined the different types of farm management options available to farmers — a single herd of 40 goats can be managed much more intensively than 1 300 cattle spread over 22 000 ha in 22 or more camps.

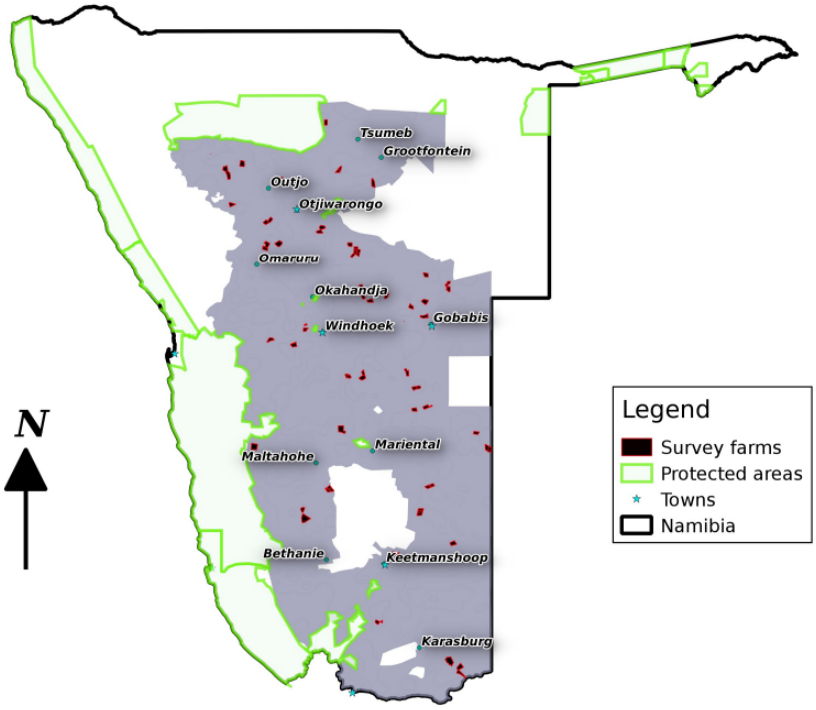


Figure 3.2: *Map indicating the locations of all the farms that completed the questionnaire.* The anonymous responses are in the correct district and of similar size, but not necessarily in their exact position. The towns shown indicate the 13 districts from which responses were received. Two of the farms were on the border of the country in the east and south respectively.

3.3.3 Predator observations

Reported predator observations showed that jackals (46 farms) and caracals (44 farms) had the widest distribution, followed by cheetahs (33 farms) and leopards (24 farms). This is in agreement with the current literature (see Chapter 2; LCMAN Red Data List for Namibian Carnivores, in prep; Lindsey et al., 2013a). The observations included seeing the predator, hearing it, or finding signs such as tracks or scat of the predator. The type of observations could be considered as a proxy for predator densities, with seeing predators considered as more likely at higher densities than only finding signs or hearing them in the distance. It can be assumed that the identification of tracks was not always accurate. Other carnivores that were reported on farms include brown hyaenas (*Parahyaena brunnea*), spotted hyaenas (*Crocuta crocuta*), African wild dogs (*Lycaon pictus*), lions (*Panthera leo*), African wildcats (*Felis silvestris lybica*), Cape foxes (*Vulpes chama*), Servals (*Leptailurus serval*) and Aardwolves (*Proteles cristata*). Although strictly speaking neither a carnivore nor predator, baboons (*Papio ursinus*) were also reported by some farmers who consider them as predators on young small livestock.

There was a positive regression slope = 0.6329 ($n = 45$; $p < 0.01$; correlation coefficient = 0.5) between reported leopard observations and livestock depredation losses to leopards, but no significant effect between leopard observations and either total livestock losses or livestock losses as percentage of offspring. Reported cheetah observations showed no significant effect on livestock depredation ($n = 37$). Caracal observations ($n = 45$) showed no significant effect on livestock depredation, nor did jackal observations ($n = 46$). See Figure 3.5 below for a comparison between livestock depredation and predators *killed*.

Table 3.1: *Livestock numbers reported per farm.* The reproduction percentage (increase) can be considered as an indication of the relative gross profitability (before expenses) of the livestock species in Namibia.

Livestock species	Average number per farm	Reproduction percentage (offspring/female)	Farm sizes (ha)	Stocking densities (LSU/ha)
Cattle	404.9 (±347.96 SD, n=39, range=2 – 1500)	67.2% (±24.7 SD, n=39, range=2% – 100%)	9394.7 (± 4451.25 SD, n=38, range=4339 – 22000)	0.0229 (±0.0196 SD, n=38, range=0.0002 – 0.0862)
Goats	232.8 (±178.52 SD, n=18, range=40 – 604)	95.5% (±37.09 SD, n=18, range=39% – 159%)	9437.8 (±4095.21 SD, n=18, range=4339 – 19400)	0.0025 (±0.00148 SD, n=18, range=0.00044 – 0.00558)
Sheep	1667 (±1625 SD, n=23, range=6 – 5100)	72.7% (±17.46 SD, n=19, range=40% – 110%)	9995.96 (±4284.02 SD, n=23, range=5556 – 19400)	0.0198 (±0.0232 SD, n=23, range=0.000043 – 0.093)
All (LSU)	299.3 (±199.55 SD, n = 41, range=55.5 – 750)	N/A	9344.69 (±4231.48 SD, n=48, range=4339 – 22000)	0.0339 (±0.0214 SD, n=40, range=0.00114 – 0.0954)

The minimum calving percentage of 2% appears to be an outlier due to drought.

3.3.4 Livestock losses

3.3.4.1 Depredation losses compared to other losses

Only 11 respondents (22.9%) actively documented *all* their losses while a further 5 (10.4%) partially documented their losses (i.e. only for some livestock species or documented some losses, but estimated that there were more undocumented losses as well). Thirty-seven respondents (77%) documented all *predation* losses, 10 (21%) documented predation losses partially, with only one respondent estimating all predation losses without any documentation whatsoever. On average, cattle farmers reported losses of 17.7 cattle per year (5.1%) to disease, injuries, drought, theft, predators and other causes, including calves falling into aardvark holes, snake bites, calving problems and abortions, lightning, and poisonous plants. Predation was by far the single greatest cause of cattle losses (Table 3.2).

Sheep farmers lost a mean of 70.9 sheep per year (11.9%) to disease, injuries, drought, theft, predators and other causes (mostly poisonous plants). Similar to cattle, predation was responsible for >50% of all sheep losses. On average 10.4 goats (13.3%) were lost per year to disease, injuries, drought, theft, depredation and other causes, including falling in rock crevices and simply going missing. For all livestock losses the majority of farmers had relatively few losses while a few farmers had very high losses (Figure 3.3), a similar pattern as that reported in other countries (Knowlton et al., 1999; Van Niekerk, 2010; Rigg et al., 2011). The highest cattle loss on a single farm was 21.9% (due to drought), the highest sheep loss was 40.1% (half to drought, the rest to depredation, theft and disease) and the highest goat loss was a substantial 55% (of which 60% was due to depredation and 40% to drought) — in both cases it was an extensive farming system.

3.3.4.2 Total predation losses

Depredation of livestock (Table 3.3 on page 146) demonstrates the difference between documented losses and estimated depredation losses. The predation estimates did not include the predator species responsible; thus, only documented losses were used to compare predator species. The mean estimated predation losses (3.15

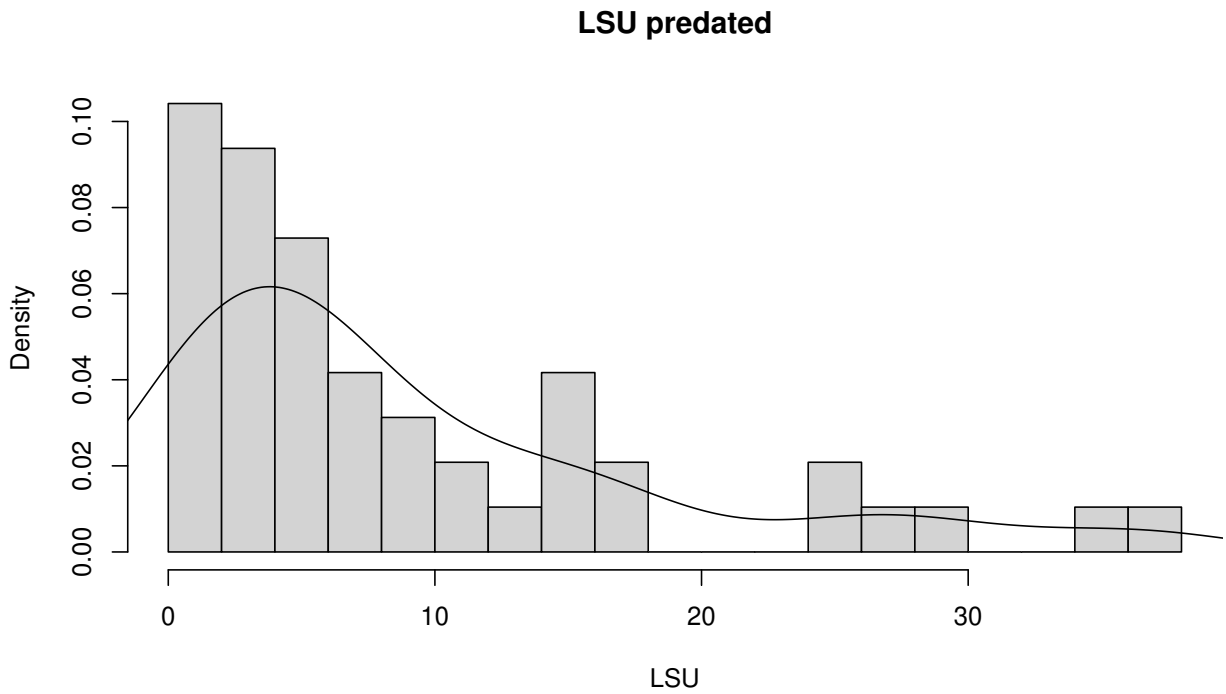


Figure 3.3: *Histogram showing distribution of livestock losses to predation (in LSU) in Namibia.* Distribution density (i.e. proportion of observations within each range of losses reported) is shown on the y axis, with the losses grouped in classes of 2 LSU width, on the x axis. The clear long tail to the right demonstrates very high losses by a few farmers, with most farmers showing losses that are lower than the mean of 9.2 LSU — this pattern has been reported elsewhere in the world as well (e.g. Van Niekerk, 2010; Rigg et al., 2011).

Table 3.2: *Relative contribution to livestock losses by different causes (as mean % of total losses).* Cattle: n=41, Predation: $p < 0.001$, observed $R^2 = 0.28$, observed $F = 19.55$; Goats: n=18, Predation $p < 0.01$, observed $R^2 = 0.16$, observed $F = 3.9$, Disease: $p < 0.05$, Drought: $p < 0.05$; Sheep: n=23, Predation: $p < 0.001$, observed $R^2 = 0.20$, observed $F = 6.76$.

Livestock species	Predation	Disease	Drought	Theft	Injuries	Other
Cattle	57.9%***	18.2%	11.0%	4.5%	3.6%	4.8%
Goats	37.6%**	28.3%*	32.0%*	0.4%	0.4%	1.5%
Sheep	58.9%***	6.2%	22.4%	11.3%	0.3%	1.0%

LSUs, ±7.78 SD, n = 48) were about half of the documented losses (6.05 LSUs, ±7.78 SD, n = 48). The annual cattle losses per farm varied between zero and a documented 35 calves (6.4% of total cattle herd, 66.7% of calves), with the highest undocumented estimated loss per farm being 60 calves (14% of total herd, and 50% of calves). Leopards were responsible for most cattle deaths, followed by cheetahs ($n = 45$ $p < 0.05$, observed $R^2 = 0.05$, observed $F = 2.83$). A number of farmers reported jackals attacking newly-born calves; and one farmer reported jackals killing five calves older than a month. One farmer on the Botswana border reported three calves being killed by lions. For farms with sheep, the losses varied between zero and a documented high of 360 lambs (10 to caracal, 350 to jackal) with a highest percentage of 25.5% of the total herd and 72.7% of lambs lost to predators. Goat farmers lost between zero and a documented high of 70 kids to jackal (35.0% of total goat herd, 100% of kids). Jackals were by far responsible for most small livestock deaths ($n = 15$, $p < 0.01$, observed $R^2 = 0.11$, observed $F = 2.31$) and for total LSUs lost per farm ($n = 48$, $p < 0.01$, observed $R^2 = 0.06$, observed $F = 4.32$). The different predators preferred different livestock in different age classes (Table 3.3). Leopards were the only predators reported to kill cattle older than 12 months. Jackals killed more sheep and goats compared to the other predators, especially lambs younger than six months old. Caracals were reported to kill more goats older than six months than would be expected from their total number of livestock kills. Brown hyaenas were reported to kill calves between one and six months slightly more than expected.

If both the estimated and the documented losses are included (as in most other published studies), an average mean of 9.2 LSUs (±9.48 SD) and median of 6 LSUs (±6.23 MAD) of all livestock species were killed by predators per farm per year. This combined mean was not statistically different from the mean documented livestock losses of 6.05 LSUs ($n = 48$, $p > 0.05$). If it is assumed that the sampled farmers were actually representative of all ~3500 commercial livestock farms in Namibia (Lindsey et al., 2013a), this translates to 32 200 LSUs (= 64 400 cattle or 322 000 small livestock) per year lost to predation, although given the small number of respondents, this estimate is probably inaccurate.

3.3.4.3 Depredation hotspots

Mean predation losses were significantly ($p < 0.001$) less for cattle per farm per year (9.8 ±12.44 SD, n = 45) than for small livestock (74.8 ±110.29 SD, n = 15). The median percentage of the potential profit (reproduction) was also significantly smaller for cattle than for small livestock ($p < 0.05$, cattle: 2.81% ±4.17 MAD, n = 32; small livestock: 11.27% ±8.17 MAD, n = 11, two sample resampling test). No clear depredation "hotspots" could be identified from the survey data (Figure 3.4) possibly due to the small sample size, and it would appear that some farms in most districts had higher depredation losses compared to others in the same areas (Figure 3.3 on page 144). This could be caused by farming with different livestock species (as shown here) or by variability in the use of conflict mitigation methods.

3.3.4.4 Livestock losses and conflict

There was a significant positive correlation between the total numbers of livestock killed by leopards and the total numbers of leopards killed by farmers ($p < 0.05$). Similarly, significant positive correlations existed for cheetahs ($p < 0.01$), for caracals ($p < 0.05$) and jackals ($p < 0.01$). However, there was no similar correlation between livestock depredation by hyaenas and them being killed by farmers. Overall, a significant positive correlation was seen between the total number of livestock killed by predators and the total number of predators killed by farmers ($p < 0.05$). About 71.2% of the farmers (39) killed predators as part of their predator management attempts (Figure 3.5).

Table 3.3: *Average documented depredation loss per predator species and estimated total depredation losses.* For each livestock species, the *documented* mean loss per age group per predator species per farm is given, followed by a row with the total mean loss per farm for that livestock species to each predator species. The second to last column gives the *estimated* total predation loss for each livestock species and the last column gives the mean total (documented + estimated) predation livestock losses for each age class and livestock species. For each livestock species, this total predator losses as a percentage of the offspring (potential income) is shown in brackets (). Other predators refer almost exclusively to brown hyaena, with one report of losses to lions. The last row gives the mean total livestock losses (as LSU) per predator species per farm. The asterisk * indicates a significant difference ($p < 0.05$) between groups using resampling of the F statistic (ANOVA). When only considering the livestock age classes and predator species, from the documented losses, the Chi squared test showed a significant $p < 0.001$ ($\chi^2 = 68.14$, df = 28). The observed - expected values are included here in square brackets [].

Livestock Lost	Leopard	Cheetah	Caracal	Jackal	Other	Estimated	Total
Calves <1 month	2.0 [1.879]	0.47 [0.345]	0.07 [-0.276]	0.29 [-2.008]	0.11 [0.06]	3.51	6.4
Calves 1-6 months	0.33 [0.25]	1.22 [1.138]	0.0 [-0.227]	0.11 [-1.399]	0.27 [0.237]	0.56	2.49
Calves 7-12 months	0.11 [0.096]	0.11 [0.096]	0.0 [-0.039]	0.0 [-0.258]	0.11 [0.104]	0.04	0.38
Cattle >12 months	0.29 [0.274]	0.0 [-0.016]	0.0 [-0.045]	0.0 [-0.297]	0.09 [0.084]	0.11	0.49
All Cattle (as% of calves)	0.68	0.47	0.03	0.20	0.14	1.06	9.8 (9.7%)
Lambs 0-6 months	0.0 [-1.513]	0.0 [-1.559]	4.4 [0.084]	31.64 [2.967]	0.64 [0.021]	15.52	52.4
Sheep >6 months	0.0 [-0.919]	0.72 [-0.227]	2.48 [-0.142]	19.08 [1.664]	0.0 [-0.376]	5.48	27.76
All Sheep (as % of lambs)	0.0	0.36	3.44	25.46*	0.32	10.5	80.16 (16.7%)
Kids 0-6 months	0.15 [-0.114]	0.50 [0.228]	1.05 [0.297]	4.7 [-0.303]	0.0 [-0.108]	1.7	8.1
Goats >6 months	0.1 [0.046]	0.05 [-0.005]	0.5 [0.347]	0.65 [-0.366]	0.0 [-0.022]	0.05	1.35
All Goats (as % of kids)	0.13	0.28	0.78	2.68*	0.0	0.88	9.45 (13.9%)
LSU /farm	1.29	0.95	0.45	3.06*	0.3	3.15	9.20

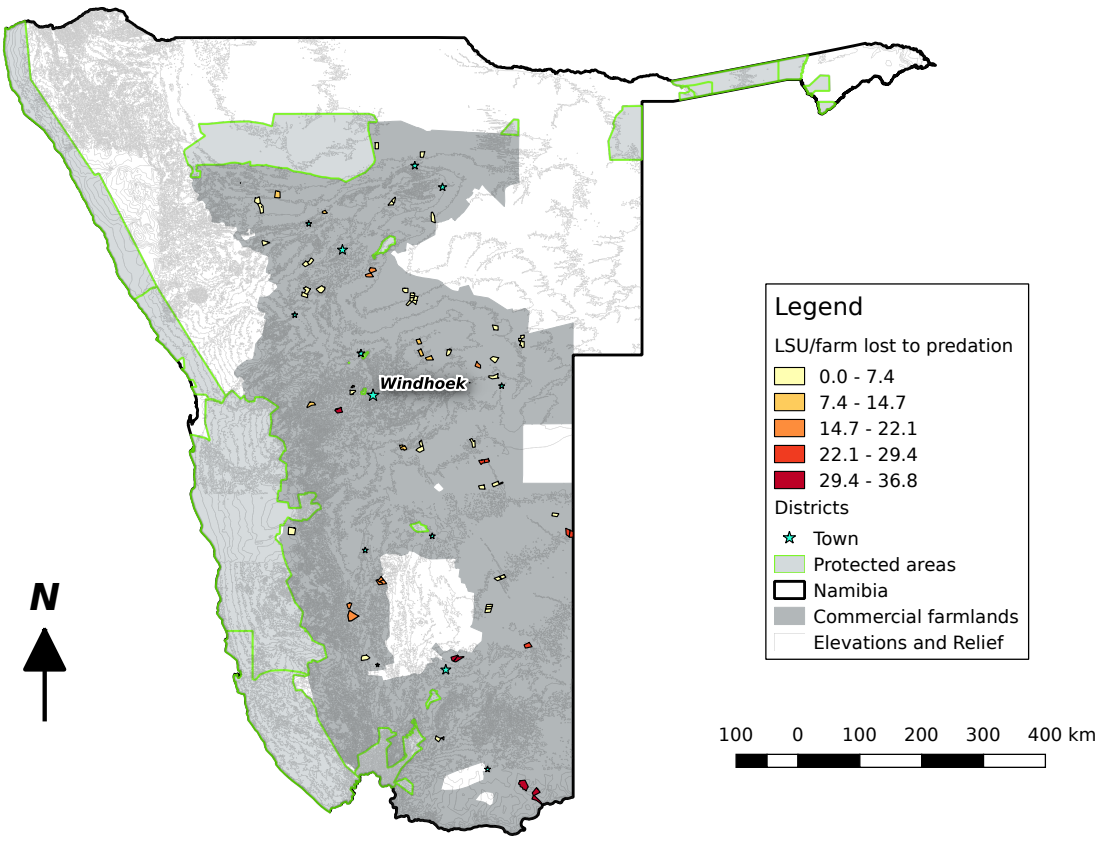


Figure 3.4: *Map showing the total livestock predation losses per farm in LSUs.* Resampling the F statistic of predation losses showed no overall significant difference between the 13 districts ($p > 0.05$, $n = 48$, observed $R^2 = 0.36$, observed $F = 1.65$), with the within-group variation similar or greater than the between-group variation – there was also no difference in livestock losses between the seven regions included in the surveys. If there are predation hotspots in Namibia, it is probably at a finer scale than at district level.

3.3.5 Effectiveness of methods

3.3.5.1 Killing predators

Generally, killing predators could not be shown to be effective and sometimes was even counter-productive (Table 3.4). For exterminating predators, the default one-tailed resampling test showed no significant decrease in livestock predation losses (of either means or medians) ($p > 0.05$), but there was a significant increase in predation losses ($p < 0.01$, using the two-tailed test). Only two respondents (4.8%) actively hunted leopards, one hunted cheetahs (2.4%), nine hunted caracals (21.4%), 15 hunted jackals (35.7%) and three (7.1%) hunted other predators, including African wild cat and brown hyaena. Jackals were often hunted at night, using spotlights and sounds to call them in. Only one farmer (2.4%) hunted leopards using dogs while another farmer hunted caracals and jackals with dogs. Some 7.1% (three respondents respectively) paid hunters to kill leopards, cheetahs, caracals and other predators, including brown hyaena and African wild cats. Additionally, 10 farmers (23.8%) paid hunters to kill jackals. One farmer claimed that a hired hunter shot 30 jackals in a period of two nights. A number of farmers did not actively hunt predators, but would shoot them on sight when encountered by chance. Two farmers (4.8%) killed leopards in this manner, five (11.9%) killed cheetahs, three (7.1%) killed caracals, 11 (26.2%) killed jackals and three (7.1%) killed other predators opportunistically, primarily hyaenas. In one case, a perceived danger to human lives possibly led to two leopards being killed close to the homestead. Some farmers only killed predators during the calving season. Surprisingly, only a single farmer used gin traps (foot traps) to catch and kill leopards and cheetahs, while five farmers (11.9%) used gin traps to catch caracals and eight (19%) to catch jackals. Three farmers also used gin traps to catch African wild cats and brown

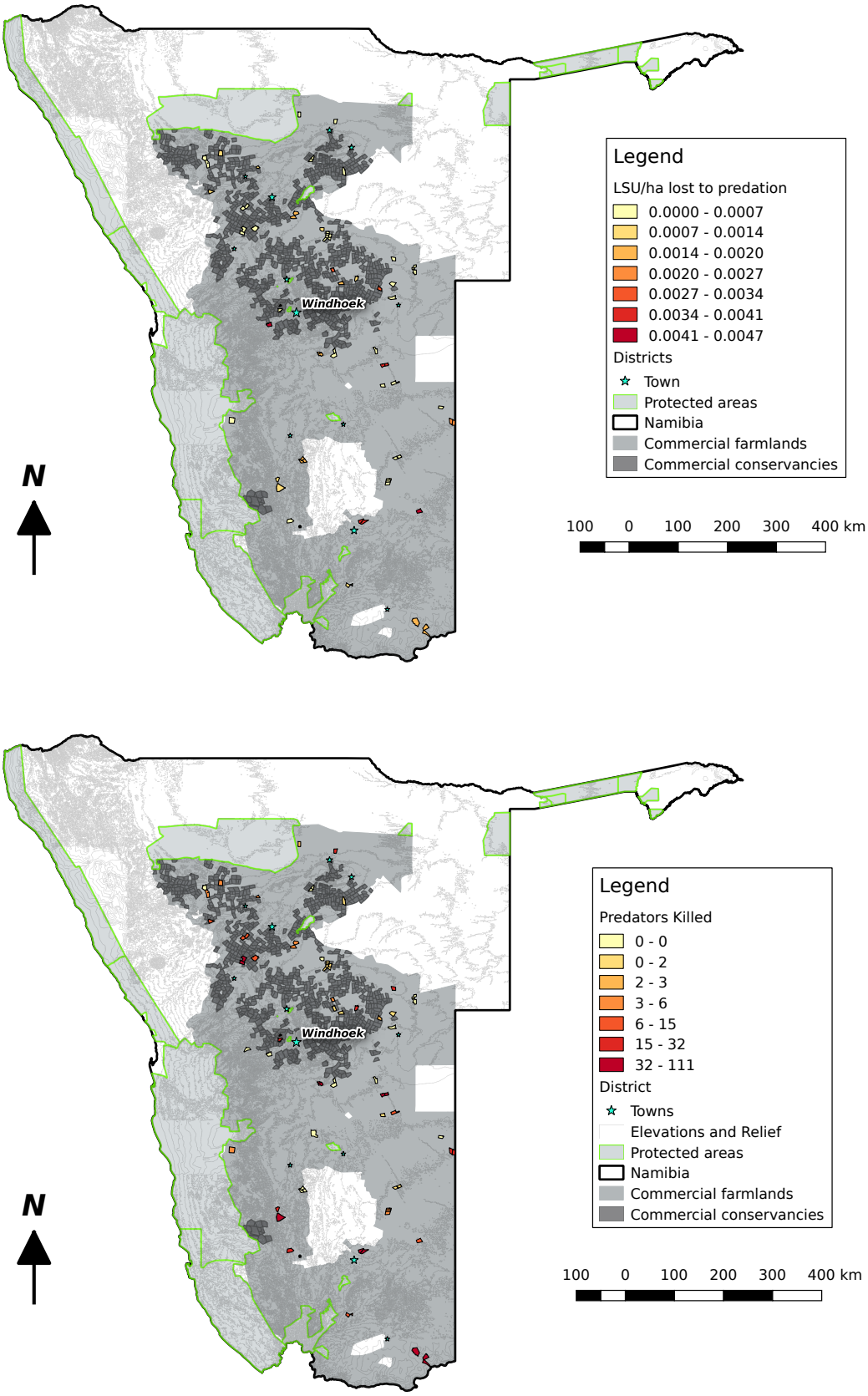


Figure 3.5: *Human-wildlife conflict from two perspectives:* Livestock lost to predation per ha at the top and predators killed by farmers per farm at the bottom. There was a significant positive correlation between the number of livestock losses and predators killed for all predator species. However, there were a few farms where relatively few livestock were lost to predation, which still had high numbers of predators killed.

hyaenas. Eleven farmers (26.2%) used cage traps to catch leopards, four used them to catch cheetahs (9.5%), eight (19%) caught caracals in cage traps, three succeeded in catching jackals in cage traps (7.2%) and a further five (11.9%) used cage traps for catching other predators, such as African wild cats and brown hyaenas. About 4.8% of responding farmers used poison of some kind against leopards, cheetahs and caracals, and five (11.9%) used poison against jackals, while only one farmer used poison to kill brown hyaenas. Similarly to hunting for extermination, poison baiting actually showed statistically significant higher predation losses ($p < 0.01$, using two-tailed test). On average 1.039 leopards, 0.442 cheetahs, 1.173 caracals, 9.519 jackals and 0.96 brown hyaenas or wild cats were removed or killed per livestock farm per year. A number of farmers did not kill any predators at all.

3.3.5.2 Natural prey

For natural prey a regression test was used, since the independent variable was not a categorical variable. In contrast to the previous study by Marker-Kraus et al. (1996), the density of game (wild prey) on farms, could not be shown to have a statistically significant effect on livestock losses, but approached statistical significance ($p \approx 0.06$) with a slightly negative correlation (-0.05) between game density and livestock depredation losses (LSU). There was a significant correlation between reported wild prey densities and prey species richness ($p < 0.001$, $r = 0.68$). Higher prey species richness (number of game species on a farm) led to a statistically significant decrease in livestock losses ($p < 0.05$) (Table 3.4). Reported prey animals did not include species that were smaller than porcupine (*Hystrix africaeaustralis*) (*cf.* Avenant and Du Plessis, 2008). All responding farmers except for two, had at least some game on their farm and 28.9% of them were members of a conservancy (although 40% of these reported that the conservancy was non-functional). Members of conservancies had significantly less depredation losses than non-members ($p < 0.05$) when considering only confirmed losses, which could indicate that conservancies with higher game diversity, also resulted in less predation losses. Conservancy members reported a mean of 7.27 $\pm$1.53 SD game species on their farms, with non-conservancy members having a mean of 5.69 $\pm$2.64 SD game species, significantly higher for conservancies at the $p < 0.05$ level.

3.3.5.3 Other preventative management methods

The 11.5% of farmers with herders did not have significantly less median depredation losses compared to those without herders, but the means of their losses were significantly less – the only other method besides natural prey species that could be shown as effective from the data of this survey (Table 3.4). Seasonal breeding on its own had no significant effect on depredation losses with 66.0% of cattle farmers, 72.0% of goat farmers and 61.0% of sheep farmers implementing seasonal breeding. However, no single breeding season was used by all farmers. Farms using predator-proof fencing (including jackal-proof fencing, electrified fencing and special camps for young livestock) had a *higher* mean livestock depredation rate than those with only livestock fencing. This was the opposite of the expected effect, and statistically significant for jackal-proof fencing ($p < 0.05$, using two-tailed test). Keeping young livestock in small camps close to the farm homestead was used by 18 farmers all over the country, but it had no significant effect on livestock losses. Keeping lambs in a kraal during the day (used by 21.0% of sheep farmers) caused no significant decrease in depredation losses. Similarly, for goats (used by 33.0% of goat farmers) and cattle (used by 13.8% of cattle farmers), keeping the young ones permanently in a kraal had no significant effect on depredation losses. Keeping all livestock in a kraal at night (used by 28.9% of farmers) had no significant effect on depredation losses, and for goats and sheep could even result in greater losses than free-roaming livestock (Table 3.4). Livestock guarding dogs, used by 15.0% of farmers, had no significant effect on small livestock depredation, but all farmers in this survey only used small or medium-sized dogs. Moving livestock to avoid depredation did not have any significant effect on the numbers of livestock lost to predators. However, most farmers that used this method, only moved livestock in response to actual losses, and not pro-actively. Approximately 25.0% of farmers used donkeys as guarding animals against predators, primarily in the north, but with no significant difference in depredation livestock losses compared to

Table 3.4: *Effectiveness of methods according to Namibian farmer survey.* For each method the mean (with Standard Deviation) and median (with Median Absolute Deviation) total livestock predation losses are given in LSU (for both those farmers who used the method, and those who did not and were used as the control group). The prior probability p(success) for each method is given by the proportion of farmers using the method that had less livestock losses than the overall median of 6 LSUs (*cf.* Chapter 2) or a smaller percentage of offspring lost than the median of 4.43% (in brackets). Statistical significance indicated with *.

Method	Method mean LSU	Control mean LSU	p (mean)	n (use)	n (control)	Method median LSU	Control median LSU	p (median)	p(success)
1.Exterminate hunt	17.075 ±12.94 SD	6.88 ±6.85 SD	>0.05	12	26	15 ±16.53 MAD	5.45 ±4.74 MAD	>0.05	0.25 (0.35)
2.Hunting dogs	3.3 ±2.4 SD	9.46 ±9.59 SD	>0.05	2	46	3.3 ±2.52 MAD	6 ±6.23 MAD	>0.05	1.0? (0.0)
3.Poison bait	13.35 ±7.84 SD	8.61 ±9.62 SD	>0.05	6	42	14.5 ±4.6 MAD	5.25 ±4.97 MAD	>0.05	0.17 (0.25)
4.Jackal-proof fence	13.25 ±12.31 SD	6.78 ±6.37 SD	>0.05	18	30	9 ±11.05 MAD	5.25 ±4.45 MAD	>0.05	0.39 (0.41)
5.Shoot on sight	8.92 ±7.81 SD	9.36 ±10.40 SD	>0.05	17	31	6.2 ±6.38 MAD	5 ±5.19 MAD	>0.05	0.35 (0.46)
6.Trapping	10.55 ±10.67 SD	8.07 ±8.39 SD	>0.05	22	26	6.5 ±7.26 MAD	5.25 ±5.04 MAD	>0.05	0.22 (0.39)
7.Herders*	3.67 ±2.01 SD	9.995 ±9.87 SD	<0.05*	6	42	3.1 ±1.93	6.05 ±6.67	>0.05	0.67 (0.6)
8.Kraaling at night	9.09 ±9.03 SD	9.26 ±9.81 SD	>0.05	15	33	6 ±6.23 MAD	6 ±6.52 MAD	>0.05	0.47 (0.38)
9.Kraaling young	11.37 ±11.73 SD	8.31 ±8.32 SD	>0.05	15	33	4.9 ±5.78 MAD	6 ±5.19 MAD	>0.05	0.53 (0.36)
10.Electric fence	11.61 ±12.12 SD	8.65 ±8.86 SD	>0.05	9	39	6.1 ±4.15 MAD	6 ±6.52 MAD	> 0.05	0.44 (0.38)
11.Calf/Lamb camps	10.78 ±9.07 SD	8.26 ±9.74 SD	>0.05	18	30	7.5 ±6.30 MAD	4 ±4.08 MAD	>0.05	0.28 (0.4)
12.Natural prey (n)	N/A	N/A	>0.05	47	47	N/A	N/A	≈0.06	0.52 (0.52)
12. Natural prey (sp)*	N/A	N/A	<0.05*	47	47	N/A	N/A		0.53 (0.47)
13.LGDs	10.40 ±11.54 SD	8.97 ±9.17 SD	>0.05	8	40	5.1 ±5.34 MAD	6 ±6.23 MAD	>0.05	0.5? (0.57)
14.Guard donkeys	6.08 ±4.31 SD	10.37 ±10.61 SD	>0.05	13	35	5.5 ±4.45 MAD	6 ±6.67 MAD	>0.05	0.54 (0.6)
15.Avoid hotspots	11.96 ±10.89 SD	6.87 ±7.55 SD	>0.05	22	26	8 ±7.19 MAD	4.05 ±3.71 MAD	>0.05	0.32 (0.32)
16.Seasonal breeding	8.18 ±8.39 SD	11.68 ±11.69 SD	>0.05	34	14	5.5 ±5.63 MAD	6.75 ±7.56 MAD	>0.05	0.5 (0.47)
17.Livestock type (sp)	5.05 ±6.38 SD	8.08 ±11.01 SD	>0.05	41	27	3 ±4.45 MAD	2.8 ±4.15 MAD	>0.05	0.68 (0.49)
17. Cattle breed	8.33 ±7.53 SD	8.89 ±10.08 SD	>0.05	16	21	5.75 ±3.56 MAD	6 ±6.82 MAD	>0.05	0.5 (0.5)
17. Sheep breed	8.19 ±9.62 SD	6.74 ±4.65 SD	>0.05	6	10	5.73	6.84	>0.05	0.5 (0.2)
18.Horned cattle	9.34 ±5.96 SD	9.18 ±10.01 SD	>0.05	7	41	6.2 ±4.74 MAD	5.5 ±5.78 MAD	>0.05	0.29 (0.25)
19.Large herds	7.51 ±5.17 SD	10.14 ±11.13 SD	>0.05	17	31	6 ±5.19 MAD	4.9 ±5.78 MAD	>0.05	0.41 (0.44)
20.Hunt reactively	5.67 ±4.58 SD	9.71 ±9.92 SD	>0.05	6	42	5.5	6.05	>0.05	0.5 (0.6)

farmers without donkeys. Almost half (46.0%) of the farmers selected female livestock animals that were able to protect offspring until weaning age, but it could not be shown to have a significant effect on depredation losses compared to those who did not report selecting female livestock animals for this protective instinct. Unlike the absolute number of individual animals killed (see Section 3.3.4.3 on depredation hotspots), there was no significant difference in terms of LSUs (as proxy for "metabolic biomass") between cattle and small livestock predation losses. Only about 18.0% of cattle farmers allowed their livestock's horns to grow for protection, and it did not appear to have any significant advantage in preventing livestock depredation. While 32.7% of farmers made use of the natural protective behaviour of livestock in large herds to protect against depredation, this method could not be shown to have a significant influence on depredation numbers.

3.3.5.4 Reactive management methods

These methods are generally not meant so much to prevent livestock losses, but are rather implemented in reaction to current livestock losses and to prevent further losses. They are generally cheaper to implement than the preventative methods, but carry a higher risk (see Chapter 2). A few farmers only actively hunted predators in retaliation for livestock depredation, with four killing leopards in response to losses (9.5%), one (2.4%) killing cheetahs, two (4.8%) killing jackals and one (2.4%) killing brown hyaenas. Reactionary killing was one of the few lethal methods that did not appear counter-productive (see Section 3.3.5.1), but was not statistically significant (Table 3.4). One brown hyaena and one cheetah were removed by the Ministry of Environment and Tourism (MET). Only one farmer who had predators removed also reported livestock losses and this method could not be analysed for statistical significance (and is therefore not included in Table 3.4).

3.3.6 Predation management costs

The majority of farmers did not know what the methods of preventing livestock depredation was costing them per year, and only 15 (37.5%) provided a rough estimate of costs, which averaged about N$ 70 392.86 and ranged between N$ 1 000.00 per month for some extra labour, to N$ 400 000.00 for electrification of fences (Table 3.5). Almost 70% of the farmers (29) were willing to change their farming methods if it would result in less livestock losses or conflict with predators.

3.4 Discussion

The lower than expected response rate, means that the results of this survey should be used with caution. However, the results of this study were primarily used to provide a first prior success probability for the various conflict mitigation methods used by Namibian farmers for use in a Decision Support System and are meant to be updated for greater accuracy over time as more farmers use the system (Chapter 5). Specifically, not too much should be concluded from the fact that many management methods could not be shown to have a statistically significant effect on livestock depredation. A similar result was found by Marker-Kraus et al. (1996), who investigated mostly the same methods and found the density of wild prey species as the only management technique to make a significant difference in livestock depredation. Their study was more limited in geographical extent to predominately cattle farming areas, but included face-to-face interviews with 241 farmers in total, recruited from attending farmers association meetings. Their survey was similar to the present one, but with a focus on cheetahs and farmer tolerance (Marker et al., 2003a). Subsequent studies showed that 73% of farmers that received an Anatolian shepherd dog experienced a decrease in livestock losses (Marker et al., 2005) — compared to the Marker-Kraus et al. (1996) study that found no significant effect for guarding dogs. Similarly, the study by Weise et al. (2018) in Botswana emphasized the fact that it is often not the method used that makes the difference, but how it is implemented (more on this in the discussion of the management method results below).

Table 3.5: *Some cost estimates for mitigation methods used by Namibian farmers.* Estimates by 12 Namibian farmers. Some methods were combined and the farmer could only give a combined cost estimate. These estimates were used as the basis of the cost classes used in the DSS in Chapter 5.

Method	Implementation (N$)	Maintenance (N$)	Notes
Jackal-proof sheep kraal	20 000		
Herder		1 000 / month	Combined with small camps, kraal at night, & small dogs
Electrifying fences	10 000, 100 000, 350 000, 400 000		
Foot traps, moving large herd		3 000 / month	2 extra labourers
Predator hunter		30 000 / year	Hunt self, paid hunter & hunting dogs
Electric fence, kraal kids, hunt & trap		3 000 / month	
Large horned herd, kraaling calves, small camps		30 000 / year	Trap reactively
Hunt predators		3 500 / month	Patrolling, spoor & hunt
Donkeys	10 000		

There was also the danger of bias in this project, with those farmers having more predation problems being more likely to respond (*cf.* Ioannidis, 2007). However, the reported losses to leopards of 1.29 LSU (Table 3.3), were lower than both Stein et al. (2010) (1.75 LSU) and Lindsey et al. (2013a) (1.82 LSU – see Table 1.1 on page 5). This would suggest a lack of self-select bias for high predation losses. A more likely reason for the low response rate, is that many farmers simply neglect to read the AgriForum magazine (personal observation). An additional possible reason for the low farmer response rate could be farmer frustration with the results of past research, which they felt did not adequately address the issues of farmer-predator conflict (see text box *Some farmer comments* on page 160) and a general breakdown in trust of researchers (*cf.* Chapter 1, Section 1.2.1; Madden and McQuinn, 2014).

3.4.1 Stocking rates and farming intensity

The mean stocking density of 0.0339 LSU/ha (= 29.5 ha/LSU) was similar to the estimated median "carrying capacity" of 12 kg/ha (= 30 ha/LSU) in Lubbe and Espach (2006) for Namibia (Lubbe, 2005). Even though most farmers could not give an accurate recent count of their livestock (due to 76.9% extensive farmers counting all their livestock less than once per month), the reported livestock numbers seem to be fairly reliable. Due to the legal requirement of every single transported animal having a unique identifying ear tag and being able to track individual cattle back to their farm of origin for EU export purposes (Chiriboga et al., 2008; NAMMIC, 2011; Francois Kok 2015, personal communication), livestock farmers generally still had a good grasp on their livestock numbers in spite of extensive farming systems (e.g. it is possible to keep record of every single animal, even if seen inside a kraal only once every three months – personal observation). The more important result from this part of the survey was that any predator conflict mitigation method that requires intensive management, would not be possible for 76.9 % of farmers to implement. Some farmers (17.3%) can use more intensive management methods and only about 5.8% of Namibian farmers can use management methods that require them to count or handle all their livestock at least weekly. Some of the respondents explicitly mentioned no longer using certain management methods (e.g. kraaling and lambing camps) since the number and size of their herds have increased too much for it to be practical.

3.4.2 Predator observations

With a few notable exceptions (e.g. the claim of a cheetah observation in the far south-east of the country – almost certainly a misidentification of a leopard) the reported predator distribution by and large agreed with the published past and present distributions (Shortridge, 1934; Brown, 2006; Lindsey et al., 2013a; Weise et al., 2017). While there were some individual reports of certain species increasing on some farms (see below in text box on page 160), there was no general trend to be seen from the data itself. Thus, no definite support or repudiation for either mesopredator release or a change in distribution of the predators could be found from the survey data. Some alternative approaches that might get better data on this aspect, are discussed in Chapter 2 on page 112.

3.4.3 Predation losses

Most farmers (77%) documented all their predation losses and a further 21% documented their predation losses partially (i.e. estimated that there were more losses to predation than they documented or only documented depredation of some livestock species). Total predation losses (including estimates) were about 1.5 times that of the documented livestock losses alone (Table 3.3). Previous surveys did not make this distinction between recorded livestock losses and estimated losses. Therefore, some caution should be applied when using data from surveys that depend on self-reporting of livestock depredation by farmers. However, Conradie and Nattrass (2017) showed that in at least one case in the Karoo, self-reported data on livestock depredation were robust and surprisingly consistent. The high percentage of farmers who recorded rather than estimated their losses in our Namibian survey, provides a possible explanation for their result. Errors in reported livestock losses as shown by the difference between documented losses and estimated self-reported losses can be ascribed to three different reasons:

1. Farmers sometimes over-estimate their losses or report exaggerated predation losses in order to engender sympathy and support for their plight (Knowlton et al., 1999). Conradie and Nattrass (2017) suggest that this is not the main reason for the observed difference between documented and estimated livestock losses. Where there is good reason to suspect that this was indeed the reason for higher estimated losses, only documented losses should be used.

2. The estimates actually include losses that could simply not be confirmed by the farmer and using only documented losses would be an underestimation of the real situation — e.g. when the predator removed or ate the whole carcass of a small animal with no remains to be found (Montag, 2003). In this case the advantage of using both documented and estimated livestock losses, is in making this factor explicit.

3. Farmers misinterpret carcass evidence (especially at old carcasses) (Camacho, 2006). This happens most often where predators scavenged on the carcass of an animal that died from other causes and then are blamed for killing it. This source of error cannot be addressed by improving questionnaires to differentiate between documented and estimated depredation losses, however.

While the average losses to predators as a percentage of potential income were relatively low (9.7% of calves, 13.9% of goat kids and 16.7% of lambs), there were a few farms on which these losses were up to 100% of the potential income in the past year. From the available data no clear-cut factors could be found to explain why these specific farms had such high levels of predation compared to other farms. The small livestock farmers with highest depredation losses, all killed high numbers of predators, but this was not true for the cattle farmers with the highest losses. About half of the highest loss cattle farmers had a low number of natural prey species on their land and most did not belong to a conservancy. The answer to these "problem farms" possibly lies in some ecological factors that could not be captured by a questionnaire, like fine-scale habitat or other factors (e.g. one farmer complaining of his neighbour shooting all the kudus — the major cheetah prey, see Chapter 2;

McVittie, 1979; Marker et al., 2003b). While this observation of a few farms with exceptionally high predation levels (*cf.* Figure 3.3) has been made in a number of studies (Davies, 1999; Knowlton et al., 1999; Van Niekerk, 2010; Rigg et al., 2011), its explanation remains a research question that begs for further investigation.

The percentage of total livestock lost to predators was 2.48% for cattle, which compares well to the study by Thorn et al. (2012) of 2.77%, but the 4.23% for goats and 6.65% for sheep were higher. The total average of 9.2 LSUs lost per farm per year, was higher than most previously published results for Namibia, with only the 1991-1993 "cheetah problem"[1] farmers of the Marker et al. (2003a) study higher, but still lower than the 11.45 LSU/farm/annum of the South African study by Van Niekerk (2010). The study by Lindsey et al. (2013a) was most similar to the present one in geographic extent, but with a focus on farmer tolerance and the larger carnivores only – cheetahs (*Acinonyx jubatus*), leopards (*Panthera pardus*), brown hyaenas (*Parahyaena brunnea*), spotted hyaenas (*Crocuta crocuta*), wild dogs (*Lycaon pictus*) and lions (*Panthera leo*). It ignored two of the agriculturally important predators (black-backed jackals and caracals). It did not include conflict mitigation methods or most livestock losses due to other causes. However, with a sample size of 250 farmers, it should be considered as more reliable for leopard depredation losses in Namibia (1.82 LSU/farm/annum) than this survey. The survey by Stein et al. (2010) was geographically limited to the area around the Waterberg National Park and had a sample size (farmers) of only 19. Their 1.75 LSU/farm/annum could be typical for leopard predation in the Waterberg Conservancy only (compared to the 1.29 LSU/farm/annum by leopards in the present survey). In terms of questions asked, the surveys by Marker et al. (2003a) were closer to the present one. However, they had a focus on cheetahs and were restricted to those parts of the country with higher cheetah densities (Marker et al., 2003a). Most previous questionnaires in Namibia did not have an agroecological approach and did not compare depredation losses to other losses. Similar to the study by Van Niekerk (2010) in South Africa, this survey found predation to be the largest single cause of livestock losses in Namibia (Table 3.2 on page 144). This implies that predation management should be relatively important in the farm management part of the agroecosystem (Figure 1.4 on page 12). Other significant causes of livestock losses were disease and drought for goats — of which drought is almost impossible to manage successfully.

The most important implication of using an agroecological approach to predation losses and also comparing it to other causes of livestock losses, is that predation management should be an integrated part of farm management and not, as it has too often been in the past, an afterthought. Moreover, it should be managed within a holistic view of the whole agroecological system, including aspects of both short-term practicality and profitability of the farming system as a whole and the likely long-term ecological effects of the different management options (Mason et al., 2018). In Chapter 5 a Probabilistic Graphical Model (PGM) is presented that can be used for decision-making in holistic predation management, using the results of this survey as one of its main data inputs.

In terms of livestock species and age class killed, there were significant differences between the predator species (Table 3.3). Leopards predated on calves younger than a month more than expected, and would also predate on calves between one and six months and on cattle older than 12 months, slightly more than expected. On the other hand, leopards killed sheep less than expected. One possible explanation for this may be the habitat preferences of leopards, preferring more rocky mountainous areas and sheep being mostly grazed on flatter areas, with goats preferring steeper slopes (personal observation). Most other livestock species and age classes were taken at close to expected rates by leopards. Cheetahs took calves between one and six months more than expected, but killed sheep lambs less than expected and compared to other predator species. Cheetahs also predated on small calves (younger than a month) and goat kids slightly more than expected. The fact that leopards preferred calves that were younger than a month old, while cheetahs appeared to prefer slightly older calves (between 1-6 months old) could also be explained by their way of hunting. Cheetahs typically prefer to hunt prey that is running away, catching them from behind, while leopards are typically stealth hunters (Estes,

[1]The study by Marker et al. (2003a) divided farmers into two groups: those who claimed to have problems with cheetahs and those who did not. It then compared the two groups in terms of actual claimed livestock losses, tolerance of predators in general and removal of cheetahs in particular.

1991). Cows often hide their very young calves which do not typically run away when approached by a predator — this suits leopards who prefer stalking and pouncing onto an unsuspecting prey animal. When calves become older, they join their mothers and are more likely to run away from predators, thus becoming more attractive prey to cheetahs. Caracals mostly took livestock in proportion to their overall depredation rates. However, they showed a slight avoidance of calves and a slight preference for goats, which could be the result of their habitat preferences (see Chapter 2). Jackals took all cattle less than expected from their overall predation rates, while showing a clear preference for sheep, both young lambs and those older than six months. They also showed a slight avoidance of goats. Like for leopards, this could be the result of habitat preferences, with jackals generally preferring flat areas and goats preferring steeper hills (see Chapter 2).

There has been no attempt to estimate the total depredation livestock losses in Namibia before. Thus, irrespective of the small sample size, this study represents an important baseline for livestock predation in Namibia. As a first approximation this survey estimated a total loss of 32 200 LSUs per year to predation in Namibia's commercial farming area of 356 533 km^2. Using the same (by now outdated) values as Van Niekerk (2010) of R600 per small livestock unit, this translates into N$ 193 200 000 per year. From a Gross Domestic Product (GDP) of N$ 2 767 000 000 in 2015 by the livestock farming sector in Namibia, this translates into about 7% of the livestock sector's GDP (MAWF, 2017). This does not include the impact of predators on communal farming. Two important conclusions follow from this: 1) Finding solutions to human-wildlife conflict in Namibia should be a priority from both an agricultural and a conservation perspective. Farmers have a legitimate problem with livestock depredation. 2) Current methods used by most farmers do not prevent predation from having a significant impact on livestock production. Therefore, farmers need to adapt to more effective predation management methods.

No conflict hotspots could be identified from the data at the regional or district level (Figure 3.4 on page 147). This was probably due to the small sample size, because districts around Etosha, for example, is known to have high human-wildlife conflict with other predators like lions and spotted hyaena as well, which was not reflected in the data from this survey (Stander, 1990a; Lindsey et al., 2013a). This lack of hotspots is in contrast to the findings of Van Niekerk (2010) in South Africa where certain districts had higher losses than most of the rest of their province. Some single farms did have exceptionally high depredation losses and it is possible that predator conflict hotspots can be found at a finer scale. Miller et al. (2015) showed that a 20 m spatial resolution was required to identify conflict hotspots for tigers (*Panthera tigris*) in India.

While Marker et al. (2003a) could initially find no significant relationship between livestock losses and cheetah removal, but only found such a relationship in follow-up surveys, here a significant relationship between total livestock losses and predators killed was found for all four predators species considered. However, brown hyaenas were killed more than expected from their contribution to livestock depredation — more research of brown hyaenas on Namibian farmlands is suggested. While a survey like this cannot show conclusively whether it is higher livestock losses that cause more predators being killed (Marker et al., 2003a; Ogada et al., 2003; Shivik et al., 2003; Lindsey et al., 2013a; Thorn et al., 2015; Miller et al., 2016a; Santangeli et al., 2016) or the killing of predators being counter-productive and causing more livestock losses (Conradie and Piesse, 2013; Minnie et al., 2016; Teichman et al., 2016), it is likely that both factors are involved. This results in a "vicious circle" (Snow, 2009). The killing of predators has been shown to sometimes result in higher livestock losses *in the following year* (Conradie and Piesse, 2013; Nattrass and Conradie, 2015). It seems safe to say that *if* killing predators was effective, it should by now have resulted in less livestock losses compared to other methods (Swanepoel, 2016). From another perspective, correlation between livestock depredation and predators killed also implies that there is reason to expect that farmer-carnivore conflict will be mitigated if livestock depredation losses are reduced (Miller et al., 2016a).

3.4.4 Predation management costs

From the responses it became clear that most farmers did not separate their predation management costs from their general expenses. Only 37.5% could give an estimate of the costs associated with implementing specific anti-predation measures, with the majority having no idea of the costs of their predation management. A follow-up study would require helping farmers to keep record of their expenses on predation management specifically, since this was not information that even those farmers with good record-keeping, had available.

3.4.5 Effectiveness of predation management

One slightly surprising result was that most of the anti-predation methods used by Namibian farmers had no statistically significant effect on livestock depredation (Table 3.4 on page 150). However, this result is similar to results of the survey by Marker-Kraus et al. (1996), which had more participating farmers (241), but in a smaller area. There could be three possible reasons for the apparent lack in effectiveness and these methods should not simply be dismissed as ineffective based only on the obtained p-values (Johnson, 1999; Nuzzo, 2014):

1. The specific method might actually be ineffective. For some of the more common methods this is probably the simplest and best explanation for the lack of difference in livestock losses where the method is used. For the methods where livestock losses were significantly higher, it is fairly safe to say that they are currently ineffective (e.g. hunting to exterminate, poison – Conradie and Piesse, 2013; Nattrass and Conradie, 2018). It is very difficult to determine the cause and effect relationship when looking at a single snapshot in time, as mentioned earlier (Sutherland et al., 2004; Sinclair et al., 2007). In addition, several methods are usually implemented concurrently, negating our ability to indicate which methods is producing the effect. Jackal-proof fencing for example, is usually combined with other methods and only expected to be effective in such combination (see methods 1 & 4 in Section 2.2.3 of Chapter 2, Beinart, 1998). Moreover, some methods are required for other methods to be effective at all (e.g. herding needs to be combined with kraaling at night or a LGD to protect livestock while the herder is sleeping — whereas kraaling alone *might* have little effect on livestock losses). Most previous analysis of mitigation methods did not investigate this interaction between different methods where some methods cannot be used together (Section 2.2.3) while other methods needs to be combined to be effective at all (Van Eeden et al., 2018).

2. The small sample size and low response rates led to low statistical power for evaluating the different methods. Some methods only being used with certain livestock (e.g. dogs only used with sheep or goats), made the available samples for testing even lower than the total number of completed questionnaires (e.g. a standard parametric t-test with 6 samples per group would have a power of only 0.1 with a standard deviation of 9 and delta = 2 at the 0.05 level – *cf.* sheep breed effects in Table 3.4).

3. Additionally, some methods work well in some circumstances, but can be ineffective or even counter-productive if incorrectly implemented (Marker and Boast, 2015). For instance fences (and kraals), which are supposed to protect livestock, sometimes result in surplus killing or a killing frenzy by feline predators when they enter the enclosure and prey cannot escape when cornered against the fence (Linnell et al., 1999; Ray et al., 2005; Weise et al., 2018; personal observation). Jackal-proof fences do not stop feline predators such as leopards or caracals, and were reported by one farmer to be used by leopards as part of their hunting strategy (Pringle and Pringle, 1979; Stuart, 1982). The *mean* livestock losses which include the effect of exceptionally high losses such as surplus killing, were higher for those farmers using kraaling of young. The *median* livestock losses, which is not influenced by a few cases of exceptionally high livestock losses, did decrease when using kraaling (Table 3.4). This can be explained if it was mostly the effect of occasional surplus killing that causes kraaling to appear "not effective". It becomes clear that instead of only asking about kraals and their usage, the questionnaire should have asked more details on the extent to which the kraals were actually predator-proof (Weise et al., 2018). None of the supposed "Livestock Guarding Dogs"

reported were large breeds such as the Anatolian shepherd, Rhodesian ridgeback or Kangal, but were all small or medium sized. There is little reason to suppose that smaller dogs would keep a predator like a leopard at bay, when dogs themselves (like jackals, Bothma and Le Riche, 1994) are often leopard prey (Athreya et al., 2016). If the dog size had not been included in the questionnaire, this detail would have been missed and it would have seemed like livestock guarding dogs were ineffective, contra Marker et al. (2005) and Potgieter (2011). At least one farmer in our survey from the Windhoek district complained that donkeys used to work well when cheetahs were the major danger to his livestock, but since leopards moved onto his land in the past few years and *the situation changed,* they are no longer effective. Seasonal breeding can be useful to reduce livestock depredation by limiting livestock as a food source for predators throughout most of the year (*cf.* Drouilly et al., 2018). It can become counter-productive when, for example, small livestock are managed to have their young in the same season of higher grazing quality in which the natural prey of the jackals (springbok) would have had their young. Livestock young are then used as a convenient replacement (since there are no natural prey) to provide the extra nutritional requirement of jackal pups (which are born in the same season – see Chapter 2; Kamler et al., 2012).

In spite of its relative ineffectiveness, this survey showed that 71.0% of the farmers still used lethal methods to deal with livestock depredation. The most popular method was proactive hunting (mostly for jackals by night), followed by cage traps (mostly leopards) and opportunistic shooting (mostly jackals), followed by paying hunters to kill predators (jackals mostly), and then foot (gin) traps (mostly for jackals). Worryingly, the use of poison is the fifth most popular method of killing predators (mostly jackals), in spite of being illegal in Namibia and having major negative ecological impacts (see section 2.2.3.1, Poison bait, on page 50 of Chapter 2; Ogada, 2014; Simmons et al., 2015; Santangeli et al., 2016). This was followed in popularity by retaliatory hunting after livestock depredation (mostly leopards), while hunting with dogs (for leopards, caracals and jackals) and removal by MET (for cheetahs and brown hyaena) were rarely used. The use of lethal methods suggests that the levels of conflict between predators and farmers are still high, as is also shown by the relatively high percentages of livestock losses (Messmer, 2000; Ogada et al., 2003; Thorn et al., 2012, 2015). Although there is little evidence that hunting is effective in reducing livestock losses (Berger, 2006; Conradie and Piesse, 2013; Swanepoel, 2016), one possible reason for its popularity is the fact that "results" can be seen immediately – the numbers of predators killed. A reduction in livestock losses, on the other hand, needs to be measured over time before the effectiveness of a predation management method can be observed. Cultural factors. like the gratification of "punishing the guilty party", no doubt plays an important role as well in the current popularity of lethal methods in Namibia (Thorn et al., 2012, 2015; Swanepoel et al., 2016). It is therefore important that the correct metric is used by farmers for measuring the cost-effectiveness of management actions.

It has been shown that repeated hunting of red foxes (*Vulpes vulpes*) in the correct season causes a short-term decrease in their densities (Lieury et al., 2015). Because of compensatory immigration, the foxes had to be culled after the dispersal period (winter for red foxes in France) and repeated annually to have any effect on fox densities. Similar compensatory immigration and breeding is probably important for all four of the main Namibian predator species on livestock, especially the mesopredators (see Chapter 2, Minnie et al., 2016, 2018). This is one explanation for the observed increase in livestock losses a year after hunting of predators (Bailey and Conradie, 2013; Conradie and Piesse, 2013). However, the short-term effectiveness seen in red foxes also depended on them having a specific dispersal season, which is not true for all Namibian predator species. More importantly, predator species density, rather than a reduction in livestock losses, was used as the metric of success. Nattrass and Conradie (2018) has shown that poison, like other lethal methods, can increase livestock losses in the following year. Additionally, poison has several knock-on effects on the ecology of rangelands in killing non-target species, including threatened species and beneficial species (Ogada, 2014; Simmons et al., 2015; Santangeli et al., 2016; see section 2.2.3.1, Poison bait, on page 50).

One encouraging result of the survey was that almost 70.0% of the respondent farmers were willing to change their management practices if it would result in less livestock losses. Similarly, a study by Rust (2016) in Namibia

showed that about 66.0% of farmers preferred non-lethal methods to prevent livestock depredation, which implies that most of the farmers currently using lethal methods would prefer non-lethal alternatives. McManus et al. (2014) showed that non-lethal options can be more cost-effective, but they might have overestimated the real costs of lethal predation management methods by estimating it while using legal requirements. As seen here, farmers do not always keep within the boundaries of environmental law, which emphasises the need for farmers as willing partners in conservation since it is almost impossible to police effectively otherwise (Hoogesteijn and Hoogesteijn, 2010).

In order to help decision-making to be more cost-effective and ecologically sustainable, the reasons why farmers may be unwilling to change their current predation management methods need to be addressed (see text box *Some farmer comments* on page 160). Some of the reasons given by farmers for preferring their current methods included the following: 1) they see no alternative or do not believe an alternative will be effective; 2) their current method is effective and working well enough or considered as the best option; 3) they are already investing heavily in infrastructure, specifically electrified fences; 4) some were losing high numbers of game and lethal mitigation methods were seen as the only option; 5) they considered their current methods as being the most sustainable; 6) some farmers cited high labour costs and labour legislation; 7) a lack of time requirements was mentioned by one farmer.

When considering the prior success probabilities calculated here for use in the Decision Support System (Table 3.4; Section 1.5 in Chapter 1), there were a number of observations:

1. Some methods differed in their prior success probability from that in the published international literature (cf. Table 2.1 in Chapter 2). For example, most lethal methods (hunting to exterminate, poison, shoot on sight to reduce numbers) were shown to have much lower success probabilities than the generalized p(success) of 0.57 for all lethal methods determined by Van Eeden et al. (2018). This is probably more realistic for Namibia. The only lethal method where the success probability approaches the one inferred from Van Eeden et al. (2018), was reactive hunting. Similarly, kraaling was much less effective than the published success ratio of enclosures (0.87) in Van Eeden et al. (2018) — this management method should probably be split into predator-proof kraals and other kraals.

2. In general, the prior success probabilities calculated here was lower than the international probabilities inferred from the published literature. This could be simply a coincidence due to the low sample size, but is more likely a reflection of the extensive farming forced by the low sustainable stocking rates in Namibia — methods that require more intensive management are more likely to fail in Namibia.

3. While similar for most methods, there were a number of management methods where the effect on total livestock losses were not similar to the effect on the depredation losses as a percentage of the population increase (young). E.g. calving/lambing camps were more likely to be successful in reducing losses of the young than overall losses. Because the success probabilities were generally low, the higher of the two prior probabilities are to be used in a probabilistic graphical model for decision-making.

4. For hunting dogs, the sample size of two was simply too small to determine prior success. It had a probability of 1 for reducing total livestock depredation numbers and simultaneously a probability of 0 for reducing the percentage of livestock population increase that are predated — the published prior probability of lethal methods internationally of 0.57 should be used instead until more information is gained.

5. Since the guarding dogs mentioned in the survey responses were all probably too small to be effective, the prior probability of success of livestock guarding dogs should be determined from the previously published Namibian studies (e.g. Marker et al., 2005; Potgieter, 2011; Potgieter et al., 2013) — p(success) = 0.91).

6. A number of methods for which no prior probability of success could be determined from the literature (and thus had a 0.5 success probability), could now be given a data-based prior probability. For instance

increasing natural prey species (p(success) = 0.53), moving livestock away from depredation "hotspots" (p(success) = 0.32), seasonal breeding (p(success) = 0.5), livestock type change (p(success) = 0.68 for small livestock to cattle, p(success) = 0.5 for breed change), horned cattle (p(success) = 0.29), large herds (p(success) = 0.44).

Some farmer comments (directly translated) — Reasons for low participation in surveys and suspicion of HWC research:

- "Still holding out and bearing with cheetahs, but their numbers are becoming too much (not threatened) and losses more. D-day is approaching. Even game numbers are drastically decreasing. My opinion is that the organizations working for the "conservation" of cheetahs, only work for their foreign investors. Why are animals that are caught in cages, simply released out of sight on the same farm? Somewhere somebody is lying horribly about "threatened" numbers, and that for the sake of money. Between rabies and cheetahs our kudu population has been almost exterminated."
- "Do you know how many students have already been here over the years with the same form and nothing is being done? We are wasting our time filling in forms."

As well as more positive responses:

- "We have to try farming with our natural enemies."
- "Do not want to kill predators!"
- "1. With leopards I only catch the problem animal, using the caught carcass as bait. 2. With jackals you can give me advice."

Some reasons given for why some management methods do not work:

- Fences & horned cattle: "Initially the leopard tried to catch calves late in the afternoons at the cattle posts. Nguni cows' horns were helping then. Leopards have changed their strategy since, and now hunt in the mornings between 5 a.m. and 6 a.m. where the cows are sleeping against the fence or in the corner of a camp. There he grabs the calf and pulls it through underneath the fence where the cow cannot do anything to it. Clever, hey?"
- Donkeys: It used to work well against cheetahs, but when "leopards moved in, the donkey experiment doesn't seem to work any-more!!"

Reasons why they would not consider changing their current predation management techniques:

- "How? Moving livestock is not always possible. No methods. Come catch cheetahs so they will become less and reduce losses. My neighbour does it for me. Hunts with dogs: cheetahs, kudus, oryx, warthogs. Report to Nature Conservation & Police, but they do nothing; just ask where the meat is."
- "Not an insurmountable problem. Cage traps are working well at the moment.";
- "Cannot. Most effective."
- "From years of experience it is best to hunt at night with spotlights. Poison and gin traps do not work, not even with people claiming themselves as experts in their use. Our jackals all have doctorate degrees in being clever. They learn from the two-legged jackals." (two-legged jackals = stock thieves)
- "Already in the process of electrifying fences."
- "Lost 350 springbok over a three to four year period to cheetahs. Hunt."
- "Ideal sheep with sustainable grazing methods. Sustainable for whole ecosystem!"
- "Tried leaving horns on cattle. Government antagonism against farmers makes employing extra workers a last option."
- "Not time consuming and can be carried out in conjunction with security (theft) patrols."

One "method" that was not specifically analysed because it showed a significant correlation with prey species richness, was membership of a conservancy — it was assumed that prey species would be more likely to have a direct effect on predator behaviour than the conservancy membership of the farmer. Most conservancies were situated in the northern half of the country (see Figure 3.1 on page 138) where the numbers of natural game species on farms are also higher (Mendelsohn et al., 2003). In contrast to most trophy hunting game farms, members of conservancies usually do not keep game in game-fenced camps (Lindsey et al., 2013b). It is thus interesting that members of conservancies had significantly lower livestock depredation losses than non-members. Conservancy membership had a significant positive effect on wildlife species richness, which also had a significant effect on livestock depredation. Most of the additional prey species were medium to large herbivores, similar in size to livestock and reduced livestock depredation could thus be attributed to a dilution effect (Haas et al., 2011; King et al., 2012). However, it is also possible that some of the natural prey species were actually preferred prey for the predators (Hayward et al., 2006a,b, 2017). It does appear that a higher prey species richness and not only prey density, led to lower livestock predation (Table 3.4). Khorozyan et al. (2015) found that below a specific wild prey threshold density, large feline predators started to kill more livestock ("cattle predation is high when prey biomass is $<812.41 \pm 1.26$ kg/km^2, whereas sheep and goat predation is high at $<544.57 \pm 1.19$ kg/km^2") which is in agreement with the findings of Marker-Kraus et al. (1996). It has been demonstrated before that Namibian farmers were more tolerant of predators when biodiversity on their land was higher (Lindsey et al., 2013a). However, a number of farmers were concerned that predators killed game species in addition to livestock. This could become a further cause of conflict when farmers within conservancies depend on game for a larger portion of their income (*cf.* Lindsey et al., 2013b). Marker and Boast (2015) mentions a number of other ways in which conservancies can benefit cheetah survival, including better livestock management, economies of scale causing lower costs, and decreased livestock losses — a significantly lower livestock depredation rate was also found on conservancies in the present survey (see page 149).

3.4.6 Addressing limitations

One possible way to overcome the low response rate in future would be personal interviews, preferably at farmer days or events where a personal relationship with farmers can be built (Marker et al., 2003a). Feedback from HWC research should also be presented at such events and through popular media, instead of just being published in scientific journals which are not freely available to most farmers. Many farmers complained that they do not know what happens with the information that they supply to researchers (personal communication). A follow-up, detailed survey obtaining data from at least four times as many farmers (n=200, 8% of NAU members), but with an emphasis on the cost-effectiveness of the different methods, would be needed to provide a better understanding of the real effectiveness of the different methods. This will bring the sample size closer to previous surveys in Namibia, without becoming too large (it would also enable large enough sample sizes for most t-tests at power level of 0.9 to show significance if a specific method caused a significant decrease in livestock losses). Approximately half of all of the current respondents indicated that they would be willing to participate in further research about predators on farmlands. This option of personal interviews was prevented in the current study by repeated vehicle breakdowns and funding constraints. An alternative is to use the approach of McManus et al. (2014) and have before-and-after trials on certain farms. This will enable study of the cause-and-effect relationship. However, this approach will probably depend on an even smaller sample size of participating farmers willing to risk their livestock and would therefore not be able to evaluate the whole range of methods currently used by Namibian farmers (Conradie and Piesse, 2016).

As noted above, more and detailed surveys cannot address the issue of farmers mistakenly assigning livestock losses from other causes to predation or blaming the wrong predator species. One common way to address this is by first training (and testing) farmers and farm workers, or by personally investigating and confirming every single livestock depredation claim (Camacho, 2006). Using more detailed surveys is also unlikely to increase information on the costs of predation prevention methods. For this reason, a CyberTracker app (`http:`

//www.cybertracker.org/) was developed in consultation with the NAU for farmers to use for recording some of their daily activities (see Appendix C; Steventon, 2015). Specifically, the app includes a carcass investigation option. Here farmers are not only guided through the process of identifying the probable predator species responsible for a kill, but also take pictures at each stage that can be used afterwards to confirm or reject their assessment. It also has a monthly predation expenses option, where farmers can record what they spent on predation management on a monthly basis, enabling a more accurate assessment of method costs.

Returning to the original aims of the survey:

1. It was seen that most Namibian farming systems are extensive, which limits the use of some HWC mitigation methods (see Table 2.1 in Chapter 2). The stocking densities reported was mostly in line with the published "carrying capacity" of Namibian farmlands and slightly lower. There was thus little evidence of overgrazing if the ecologically dubious assumption of a single carrying capacity is accepted (Dhondt, 1988; McLeod, 1997; Luyt, 2004; Lubbe, 2005; Hixon, 2008; Terborgh, 2015).

2. No obvious trends in predator densities or evidence of mesopredator release could be determined from the survey data.

3. A mean of 9.2 LSU/farm/year and a median of 6 LSU/farm/year was reportedly lost to predation by Namibian farmers,

   (a) which was the highest single cause of livestock losses for all livestock,

   (b) extrapolated from the survey results to all of Namibia's commercial farms, it gave a first estimate of about 32 200 LSUs lost per year to predation in total, which is a substantial proportion of the livestock farming contribution to Namibia's GDP (MAWF, 2017),

   (c) and no districts or regions were found with exceptionally high livestock losses (hotspots).

4. The costs of the different anti-predation methods used in Namibia were mostly unknown to the farmers and could be estimated for only a few methods.

5. The relative probability of success of the different methods used in Namibia were determined by using the proportion of those farmers using the method with livestock losses below the median as an estimate of its prior probability of success (Jøsang, 2008). Switching from small livestock to cattle (p(success) = 0.68) and using herders (p(success) = 0.67) were the two methods with the highest probability of success. Increasing the number of wild prey species led to a statistically significant decrease in livestock depredation, but farmers with higher than the median number of species had livestock losses lower than the median only 53% of the time — this p(success) of 0.53 is about the same as that of using guarding donkeys or kraaling the young livestock permanently, but lower than that from the literature (Chapter 2, Table 2.1).

3.5 Conclusions

Livestock depredation is thus a real and widespread issue for Namibian livestock farmers. It needs to be addressed properly for any realistic chance at the future survival of currently threatened predators on farmlands. This survey provided an example of the difference between documented, confirmed livestock depredation numbers and the total numbers that farmers report lost to predators. It was clear that not all farmers are able or willing to accurately report losses and this should be taken into account in future research by explicitly differentiating between documented, accurate data and estimates.

Conservancies and incorporating some way of increasing prey biodiversity on farmlands appear to be a consistent method for decreasing livestock losses and human-wildlife conflict (current chapter; McGranahan, 2008; Stein

et al., 2010; Lindsey et al., 2013b; Marker and Boast, 2015). Increasing herbivore biodiversity often increase the total prey species densities (see Section 3.3.5.2 above), since grazers and browsers differ in their ecological niches (Peel et al., 1998; Veblen et al., 2016). Khorozyan et al. (2015) have shown that wild prey densities directly impact livestock depredation by large felid carnivores. Miller et al. (2016b) found five studies in which increasing wild prey decreased livestock depredation and two in which it actually increased livestock depredation — prior success probability = 0.71 (Chapter 2; Marker-Kraus et al., 1996; Shivik et al., 2003; Avenant and Du Plessis, 2008; see also text box below: *Why increasing wild prey could increase livestock depredation*).

Why increasing wild prey could increase livestock depredation — It is suggested here that livestock depredation losses actually forms a n-shaped curve, with highest predation at intermediate densities (Gail Potgieter, personal communication). If there is no natural prey at all, predators are unlikely to survive on livestock alone, and occur at such low densities that there is little livestock depredation. At very high natural prey densities, by providing predators with preferred prey species and through a general dilution effect, livestock depredation also decreases. At intermediate densities and species richness of natural prey, predators are more likely to encounter livestock by chance and the natural prey community might not provide enough preferred prey species to the predator, possibly causing higher levels of livestock depredation.

In addition to possibly less livestock losses, increasing biodiversity generally increase ecosystem stability (Cadotte et al., 2012), benefiting the farming enterprise indirectly as well (see Chapter 2). These herbivores could compete with livestock for grazing. However, there is generally enough differentiation between the different wild herbivores (especially browsers) and livestock in their plant part and type selection that it is possible to keep both without grazing competition becoming a major problem (Lindsey et al., 2013b; Veblen et al., 2016). Wild herbivores on Namibian farms are generally used as an additional source of income and food. The possible overlap in diet between wild herbivores and livestock can be considered as a hidden cost of using this method, however. This predation management "method" has the added advantage that it can be combined with most other methods as well. From the survey data, herding had the highest probability of success, which is not totally unexpected (Ogada et al., 2003; Woodroffe et al., 2007; Van Eeden et al., 2018). A similar prior success probability was found for switching to cattle from small livestock, but it was not statistically significant and depending on the grazing, could be impractical (NAMMIC, 2011). Using the ratio of successful to unsuccessful farmers for each method, gives an indication of the *probability* of success, but no indication of *the size of its effect* (*cf.* Miller et al., 2016b). For example, the difference between livestock losses of the half of the farmers with the greatest number of natural prey species compared to the half with the least number of prey species, were only a mean decrease of 4.55 LSU per farm and a median decrease of 1.45 when having more prey species. Compare that to the decrease in means of 6.33 LSU per farm and of 2.95 in the median LSU per farm when using herders *versus* not using herders (Table 3.4).

With about 70.0% of farmers using primarily lethal methods to attempt to reduce livestock depredation, and an average of 9.2 LSUs (3.9%) lost per farm annually, it should be clear that current methods are not addressing the human-wildlife conflict adequately. While approximately 70.0% of farmers are willing to change their management if they can be shown more cost-effective farming methods to prevent livestock depredation, many also suggest that they do not know how to improve matters and feel like they are fighting a losing battle. With farmers having little idea about the costs of the different methods used, it would appear that there is a real need for cost-effective methods that result in higher prey biodiversity, lower livestock losses and more sustainable ecosystems on Namibian farmlands. Sharing the result of this research project widely with both participating farmers and the wider livestock producer community in Namibia, will be an important first step to reduce barriers between farmers and the research community. It will also address current farmer frustrations.

There is a need for future research to focus on the cost-effectiveness of the different methods by which farmers attempt to protect livestock. The sample size needs to be big enough so that the effectiveness of the methods can be teased apart and analysed per livestock species, per predator species and per ecosystem attribute such as vegetation type, topography, heterogeneity or homogeneity of the terrain, availability of drinking water and seasonal changes. Although questionnaires alone might provide insufficient data for this analysis, using an

online Decision Support System (DSS) will enable farmers to give feedback on the effectiveness and costs of any new methods which they had implemented. By using the DSS farmers will thus engage in an ongoing trial, following an adaptive management approach. A major remaining research question is to identify the ecological or other factors that result in some farms having such disproportionately high livestock losses. Farmers should be encouraged and helped to keep record of livestock losses and the costs of specific predation management techniques (see e.g. Appendix C). The implementation and maintenance costs of the various methods can be combined with the probability of effectiveness and actual decrease in predation loss, for a proper cost-benefit analysis. The results can then be widely disseminated amongst farmers. An online DSS that acquires knowledge over time by using farmer input, is a promising way to disseminate these results (Chapter 5). It will provide a service (advice on predation management) to farmers, but simultaneously use their input to improve the reliability of the advice.

References

Athreya, V., Odden, M., Linnell, J., Krishnaswamy, J., Karanth, K., 2016. A cat among the dogs: leopard *Panthera pardus* diet in a human-dominated landscape in western Maharashtra, India. Oryx 50 (1), 156–162.

Avenant, N., Du Plessis, J., 2008. Sustainable small stock farming and ecosystem conservation in southern Africa: a role for small mammals? . Mammalia 72, 258–263.

Bailey, A., Conradie, B., 2013. The effect of predator culling on livestock losses: Caracal control in Cooper hunting club, 1976 – 1981. Tech. rep., Centre for Social Science Research, Cape Town.

Beinart, W., 1998. The night of the jackal: sheep, pastures and predators in the Cape. Past & Present 158, 172–206.

Berger, K., 2006. Carnivore-livestock conflicts: Effects of subsidized predator control and economic correlates on the sheep industry. Conservation Biology 20 (3), 751–761.

Bothma, J. P., Le Riche, E., 1994. Scat analysis and aspects of defecation in Northern Cape leopards. South African Journal of Wildlife Research 24 (1/2), 21–26.

Brown, C., Jul. 2006. Historic distribution of large mammals in the Greater Fish River Canyon Complex, southern Namibia, and recommendations for re-introductions. Tech. rep., Namibia Nature Foundation.

Cadotte, M., Dinnage, R., Tilman, D., 2012. Phylogenetic diversity promotes ecosystem stability. Ecology 93 (8), S223–S233.

Camacho, G., 2006. Perceptions regarding livestock losses in Lydenburg and surrounding area. In: Daly, B., Davies-Mostert, H., Davies-Mostert, W., Evans, S., Friedmann, Y., King, N., Snow, T., Stadler, H. (Eds.), Prevention is the cure. Proceedings of a workshop on holistic management of human-wildlife conflict in the agricultural sector of South Africa. Endangered Wildlife Trust, pp. 35–42.

Child, M., Roxburgh, L., Do Linh San, E., Raimondo, D., Davies-Mostert, H. (Eds.), 2016. The 2016 red list of mammals of South Africa, Swaziland and Lesotho. South African National Biodiversity Institute and Endangered Wildlife Trust.

Chilonda, P., Otte, J., 2006. Indicators to monitor trends in livestock production at national, regional and international levels. Livestock Research for Rural Development 18 (117).

Chiriboga, L., Kilmer, C., Fan, R., Gawande, K., 2008. Does Namibia have a comparative advantage in beef production? Tech. rep., Texas A&M University, http://kishoregawande.net/wp-content/uploads/2008/12/finalreporti.pdf.

Conradie, B., Nattrass, N., 2017. The robustness of self-report data on predation: A comparison of two Karoo surveys. African Journal of Agricultural and Resource Economics 12 (3), 217–229.

Conradie, B., Piesse, J., 2013. The effect of predator culling on livestock losses: Ceres, South Africa, 1979 – 1987. Tech. Rep. 319, Centre for Social Science Research, Cape Town.

Conradie, B., Piesse, J., 2016. Ranking perceived risk to farmers: How important is the environment? African Journal of Agricultural and Resource Economics 11 (4), 263–276.

Davies, A., 1999. The extent, cost and control of livestock predation by eagles with a case study on black eagles (Aquila verreauxii) in the Karoo. Journal of Rapture Research 33 (1), 67–72.

Davis, A., Jarvis, A., Viall, W., Symonds, A., Kelly, M., 2011. Namibia's communal conservancies - a review of progress - 2010. NACSO (Namibian Association of CBNRM Support Organisations).

Dhondt, A., 1988. Carrying capacity: a confusing concept. Acta Oecologica 9, 337–346.

Drouilly, M., Nattrass, N., O'Riain, M., 2018. Dietary niche relationships among predators on farmland and a protected area. Journal of Wildlife Management 82, 507–518.

Estes, R., 1991. The behavior guide to African mammals: Including hoofed mammals, carnivores, primates. University of California Press.

Haas, S., Hooten, M., Rizzo, D., Meentemeyer, R., 2011. Forest species diversity reduces disease risk in a generalist plant pathogen invasion. Ecology Letters 14, 1108–1116.

Hayward, M., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G., Kerley, G., 2006a. Prey preferences of the leopard (*Panthera pardus*). Journal of Zoology, London 270 (2), 298–313.

Hayward, M., Hofmeyr, M., O'Brien, J., Kerley, G., 2006b. Prey preferences of the cheetah (*Acinonyx jubatus*) (Felidae: Carnivora): morphological limitations or the need to capture rapidly consumable prey before kleptoparasites arrive? Journal of Zoology 270, 615–627.

Hayward, M., Porter, L., Lanszki, J., Kamler, J., Beck, J., Kerley, G., Macdonald, D., Montgomery, R., Parker, D., Scott, D., O'Brien, J., Yarnell, R., 2017. Factors affecting the prey preferences of jackals (Canidae). Mammalian Biology 85, 70–82.

Hey, D., 1964. The control of vertebrate problem animals in the province of the Cape of Good Hope, Republic of South Africa. In: Proceedings of the 2nd Vertebrate Pest Control Conference (1964). Vertebrate Pest Conference Proceedings. http://digitalcommons.unl.edu/vpc2/11, pp. 57–70.

Hixon, M., 2008. Carrying capacity. In: Jørgensen, S., B.D., F. (Eds.), Encyclopedia of Ecology. Elsevier Science, pp. 528–530.

Hoogesteijn, R., Hoogesteijn, A., 2010. Conserving wild felids in humanized landscapes - Strategies for reducing conflicts between jaguars and cattle. Wild Felid Monitor 3 (2), 1, 10–13.

Ibrahim, I., 1991. Evaluating the power of the Mann-Whitney test using the bootstrap method. Communications in Statistics - Theory and Methods 20 (9), 2919–2931.

Ioannidis, J. P. A., 2007. Why most published research findings are false: Author's reply to Goodman and Greenland [7].

Johnson, D., 1999. The insignificance of statistical significance testing. Journal of Wildlife Management 63 (3), 763–772.

Jøsang, A., 2008. Conditional reasoning with subjective logic. Journal of Multiple-Valued Logic and Soft Computing 15 (1), 5–38.

Kamler, J., Klare, U., Macdonald, D., 2012. Seasonal diet and prey selection of black-backed jackals on a small-livestock farm in South Africa. African Journal of Ecology 50, 299–307.

Kerley, G., Wilson, S., Balfour, D. (Eds.), 2018. Livestock predation and its management in South Africa: A scientific assessment. Centre for African Conservation Ecology.

Kerns, G., 2010. Introduction to probability and statistics using R. R-project.

Khorozyan, I., Ghoddousi, A., Soofi, M., Waltert, M., 2015. Big cats kill more livestock when wild prey reaches a minimum threshold. Biological Conservation 192, 268–275.

King, A., Wilson, A., Wilshin, S., Lowe, J., Haddadi, H., Halles, S., Morton, A., 2012. Selfish-herd behaviour of sheep under threat. Current Biology 22 (14), R561–R562.

Knowlton, F., Gese, E., Jaeger, M., 1999. Coyote depredation control: An interface between biology and management. Journal of Range Management 52 (5), 398–412.

Lieury, N., Ruette, S., Devillard, S., Alabaret, M., Drouyer, F., Baudoux, B., Millon, A., 2015. Compensatory immigration challenges predator control: An experimental evidence-based approach improves management. Journal of Wildlife Management 79 (3), 425–434.

Lindsey, P., Havemann, C., Lines, R., Palazy, L., Price, A., Retief, T., Rhebergen, T., Van der Waal, C., 2013a. Determinants of persistence and tolerance of carnivores on Namibian ranches: Implications for conservation on Southern African private lands. PLoS ONE 8 (1), e52458.

Lindsey, P., Havemann, C., Lines, R., Price, A., Retief, E., Rhebergen, T., Van der Waal, C., Romañach, S., 2013b. Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. Oryx 47 (1), 41–53.

Linnell, J., Odden, J., Smith, M., Aanes, R., Swenson, J., 1999. Large carnivores that kill livestock: Do "problem individuals" really exist? Wildlife Society Bulletin 27 (3), 698–705.

Lubbe, L., 2005. Towards an updated carrying capacity map for Namibia: a review of the methodologies currently used to determine carrying capacity in Namibia. Agricola 15 (6), 33–39.

Lubbe, L., Espach, C., 2006. An accurate grazing capacity map for Namibia – Myth or reality? Spotlight on Agriculture 100, 1–2.

Luyt, E. d. C., 2004. Models of bontebok (*Damaliscus pygargus pygargus*, Pallas 1766) habitat preferences in the Bontebok National Park and sustainable stocking rates. Master's thesis, Stellenbosch University.

Madden, F., McQuinn, B., 2014. Conservation's blind spot: The case for conflict transformation in wildlife conservation. Biological Conservation 178, 97–106.

Marker, L., Boast, L., 2015. Human–wildlife conflict 10 years later: Lessons learned and their application to cheetah conservation. Human Dimensions of Wildlife: An International Journal 20 (4), 302–309.

Marker, L., Dickman, A., Macdonald, D., 2005. Perceived effectiveness of livestock-guarding dogs placed on Namibian farms. Rangeland Ecology & Management 58 (4), 329–336.

Marker, L., Dickman, A., Mills, M., Macdonald, D., 2010. Cheetahs and ranchers in Namibia: a case study. In: Macdonald, D., Loveridge, A. (Eds.), Biology and Conservation of Wild Felids. Oxford University Press, Ch. 15, pp. 353–372.

Marker, L., Mills, M., Macdonald, D., 2003a. Factors influencing perceptions of conflict and tolerance toward cheetahs on Namibian farmlands. Conservation Biology 17 (5), 1290–1298.

Marker, L., Muntifering, J., Dickman, A., Mills, M., Macdonald, D., 2003b. Quantifying prey preferences of free-ranging Namibian cheetahs. South African Journal of Wildlife Research 33 (1), 43–53.

Marker-Kraus, L., Kraus, D., Barnett, D., Hurlbut, S., 1996. Cheetah survival on Namibian farmlands. Cheetah Conservation Fund.

Mason, T., Pollard, C., Chimalakonda, D., Guerrero, A., Kerr-Smith, C., Milheiras, S., Roberts, M., Ngafack, P., Bunnefeld, N., 2018. Wicked conflict: Using wicked problem thinking for holistic management of conservation conflict. Conservation Letters 82 (6), e12460.

MAWF, 2017. Agricultural statistics bulletin - 2010-2015. Tech. rep., Directorate of Planning and Business Development, Private Bag 13184, Government Office Park, Windhoek.

McGranahan, D., 2008. Managing private, commercial rangelands for agricultural production and wildlife diversity in Namibia and Zambia. Biodiversity and Conservation 17 (8), 1965–1977.

McGranahan, D., 2011. Identifying ecological sustainability assessment factors for ecotourism and trophy hunting operations on private rangeland in Namibia. Journal of Sustainable Tourism 19 (1), 115–131.

McLeod, S., 1997. Is the concept of carrying capacity useful in variable environments? Oikos 79 (3), 529–542.

McManus, J., Dickman, A., Gaynor, D., Smuts, B., Macdonald, D., 2014. Dead or alive? Comparing costs and benefits of lethal and non-lethal human–wildlife conflict mitigation on livestock farms. Oryx 49 (4), 687–695.

McVittie, R., 1979. Changes in the social behaviour of South West African cheetah. Madoqua 11 (3), 171–184.

Mendelsohn, J., Jarvis, A., Roberts, C., Robertson, T., 2003. Atlas of Namibia - A portrait of the land and its people. David Phillip Publishers, http://www.the-eis.com/.

Messmer, T., 2000. The emergence of human-wildlife conflict management: turning challenges into opportunities. International Biodeterioration & Biodegradation 45, 97–102.

Miller, J., Jhala, Y., Jena, J., Schmitz, O., 2015. Landscape-scale accessibility of livestock to tigers: implications of spatial grain for modeling predation risk to mitigate human–carnivore conflict. Ecology and Evolution 5 (6), 1354–1367.

Miller, J., Jhala, Y., Schmitz, O., 2016a. Human perceptions mirror realities of carnivore attack risk for livestock: Implications for mitigating human-carnivore conflict. PLoS ONE 11 (9), e0162685.

Miller, J., Stoner, K., Cejtin, M., Meyer, T., Middleton, A., Schmitz, O., 2016b. Effectiveness of contemporary techniques for reducing livestock depredations by large carnivores. Wildlife Society Bulletin 40 (4), 806–815.

Minnie, L., Avenant, N., Drouilly, M., Samuels, I., 2018. Biology and ecology of the black-backed jackal and the caracal. In: Kerley et al. (2018), pp. 178–204.

Minnie, L., Gaylard, A., Kerley, G., 2016. Compensatory life-history responses of a mesopredator may undermine carnivore management efforts. Journal of Applied Ecology 53, 379–387.

Montag, J., 2003. Compensation and predator conservation: Limitations of compensation. Carnivore Damage Prevention News 6, 2–6.

NAMMIC, 2011. Namibian livestock sector strategy - Final report. Tech. rep., Namibian Meat Board.

Nattrass, N., Conradie, B., 2015. Jackal narratives and predator control in the Karoo, South Africa. Journal of Southern African Studies 41 (4), 1–19.

Nattrass, N., Conradie, B., 2018. Predators, livestock losses and poison in the South African Karoo. Journal of Cleaner Production 194, 777–785.

Nuzzo, R., Feb. 2014. Statistical errors. Nature 506, 150–152.

Ogada, D., 2014. The power of poison: pesticide poisoning of Africa's wildlife. Annals of the New York Academy of Sciences 1322 (1), 1–20.

Ogada, M., Woodroffe, R., Oguge, N., Frank, L., 2003. Limiting depredation by African carnivores: the role of livestock husbandry. Conservation Biology 17 (6), 1521–1530.

Peel, M., Biggs, H., Zacharias, P., 1998. Perspective article: The evolving use of stocking rate indices currently based on animal number and type in semi-arid heterogeneous landscapes and complex land-use systems. African Journal of Range & Forage Science 15 (3), 117–127.

Peng, R., May 2016. R programming for data science. Leanpub.

Potgieter, G., 2011. The effectiveness of livestock guarding dogs for livestock production and conservation in Namibia. Master's thesis, Nelson Mandela Metropolitan University.

Potgieter, G., Marker, L., Avenant, N., Kerley, G., 2013. Why Namibian farmers are satisfied with the performance of their livestock guarding dogs. Human Dimensions of Wildlife: An International Journal 18 (6), 403–415.

Pringle, J., Pringle, V., 1979. Observations on the lynx *Felis caracal* in the Bedford district. South African Journal of Zoology 14 (1), 1–4.

QGIS-Development-Team, Apr. 2017. QGIS user - guide release 2.18. http://qgis.osgeo.org.

Ray, J., Hunter, L., Zigouris, J., 2005. Setting conservation and research priorities for larger African carnivores. Tech. rep., Wildlife Conservation Society.

Rigg, R., Findo, S., Wechselberger, M., Gorman, M., Sillero-Zubiri, C., Macdonald, D., 2011. Mitigating carnivore–livestock conflict in Europe: lessons from Slovakia. Oryx 45 (2), 272–280.

Rust, N., 2016. Can stakeholders agree on how to reduce human-carnivore conflict on Namibian livestock farms? A novel Q-methodology and Delphi exercise. Oryx 51 (2), 1–8.

Santangeli, A., Arkumarev, V., Rust, N., Girardello, M., 2016. Understanding, quantifying and mapping the use of poison by commercial farmers in Namibia – Implications for scavengers' conservation and ecosystem health. Biological Conservation 204 (B), 205–211.

Shivik, J., Tregenza, T., Callahan, P., 2003. Nonlethal techniques for managing predation: Primary and secondary repellents. Conservation Biology 17 (6), 1531–1537.

Shortridge, G., 1934. The mammals of South West Africa. Heinemann.

Simmons, R., Brown, C., Kemper, J., 2015. Birds to watch in Namibia: red, rare and endemic species. Ministry of Environment and Tourism and Namibia Nature Foundation.

Simon, J., Oct. 1997. Resampling: The new statistics., 2nd Edition. Resampling Stats, http://www.resample.com/intro-text-online/.

Sinclair, A., Mduma, S., Brashares, J., 2003. Patterns of predation in a diverse predator–prey system. Nature 425, 288–290.

Sinclair, A., Mduma, S., Grant, J., Hopcraft, C., Fryxell, J., Hilborn, R., Thirgood, S., 2007. Long-term ecosystem dynamics in the Serengeti: Lessons for conservation. Conservation Biology 21 (3), 580–590.

Snow, T., 2009. A systems-thinking based evaluation of predator conflict management on selected South African farms. Master's thesis, University of KwaZulu-Natal.

Stander, P., 1990a. A suggested management strategy for stock-raiding lions in Namibia. South African Journal of Wildlife Research 20 (2), 37–43.

Stein, A., Fuller, T., Damery, D., Sievert, L., Marker, L., 2010. Farm management and economic analyses of leopard conservation in north-central Namibia. Animal Conservation 13, 419–427.

Steventon, J., 2015. CyberTracker 3 control reference manual - CyberTracker version 3.268.

Stuart, C., 1982. Aspects of the biology of the Caracal (*Felis caracal* Schreber 1776) in the Cape Province, South Africa. Master's thesis, University of Natal.

Sutherland, W., Pullin, A., Dolman, P., Knight, T., 2004. The need for evidence-based conservation. Trends in Ecology and Evolution 19 (6), 305–308.

Swanepoel, J., 2016. Habits of the hunters: the biopolitics of combatting predation amongst small stock farmers in southern Namibia. Journal of Contemporary African Studies 34 (1), 129–146.

Swanepoel, L., Balme, G., Williams, S., Power, R., Snyman, A., Gaigher, I., Senekal, C., Martins, Q., Child, M., 2016. A conservation assessment of *Panthera pardus*. In: Child et al. (2016), p. 13 pp.

Teichman, K., Cristescu, B., Darimont, C., 2016. Hunting as a management tool? Cougar-human conflict is positively related to trophy hunting. BMC Ecology 16 (1), 44.

Terborgh, J., 2015. Toward a trophic theory of species diversity. PNAS 112 (37), 11415–11422.

Thorn, M., Green, M., Dalerum, F., Bateman, P., Scott, D., 2012. What drives human-carnivore conflict in the North West Province of South Africa? Biological Conservation 150, 23–32.

Thorn, M., Green, M., Marnewick, K., Scott, D., 2015. Determinants of attitudes to carnivores: implications for mitigating human–carnivore conflict on South African farmland. Oryx 49 (2), 270–277.

Treves, A., Karanth, K., 2003a. Human-carnivore conflict and perspectives on carnivore management worldwide. Conservation Biology 17 (6), 1491–1499.

Treves, A., Karanth, K., 2003b. Special section: Human-carnivore conflict: Local solutions with global applications. Conservation Biology 17 (6), 1489–1490.

Treves, A., Krofel, M., McManus, J., 2016. Predator control should not be a shot in the dark. Frontiers in Ecology and the Environment 14 (7), 380–388.

Van Eeden, L., Ecklund, A., Miller, J., López-Bao, J., Chapron, G., Cejtin, M., Crowther, M., Dickman, C., Frank, J., Krofel, M., Macdonald, D., McManus, J., Meyer, T., Middleton, A., Newsome, T., Ripple, W., Ritchie, E., Schmitz, O., Stoner, K., Tourani, M., Treves, A., 2018. Carnivore conservation needs evidence-based livestock protection. PLoS Biology 16 (9), e2005577.

Van Niekerk, H., 2010. The cost of predation on small livestock in South Africa by medium-sized predators. Master's thesis, University of the Free State.

Veblen, K., Porensky, L., Riginos, C., Young, T., 2016. Are cattle surrogate wildlife? Savanna plant community composition explained by total herbivory more than herbivore type. Ecological Applications 26 (6), 1610–1623.

Weise, F., Hayward, M., Casillas Aguirre, R., Tomeletso, M., Gadimang, P., Somers, M., Stein, A., 2018. Size, shape and maintenance matter: A critical appraisal of a global carnivore conflict mitigation strategy - Livestock protection kraals in northern Botswana. Biological Conservation 225, 88–97.

Weise, F., Vijay, V., Jacobson, A., Schoonover, R., Groom, R., Horgan, J., Keeping, D., Klein, R., Marnewick, K., Maude, G., Melzheimer, J., Mills, G., Van der Merwe, V., Van der Meer, E., Van Vuuren, R., Wachter, B., Pimm, S., 2017. The distribution and numbers of cheetah (*Acinonyx jubatus*) in southern Africa. PeerJ 5, e4096.

Woodroffe, R., Frank, L., Lindsey, P., ole Ranah, S., Romañach, S., 2007. Livestock husbandry as a tool for carnivore conservation in Africa's community rangelands: a case-control study. Biodiversity and Conservation 16, 1245–1260.

WWF, 2016. Living Planet Report 2016. Risk and resilience in a new era. Tech. rep., WWF International.

Yu, C., 2003. Resampling methods: Concepts, Applications, and Justification. Practical Assessment, Research & Evaluation 8 (19), 1–16.

Chapter 4

Comparative spatial ecology of cheetahs and leopards on Namibian farmlands

Abstract

For the ecological sustainability of any conflict mitigation method, we require a good insight into the ecology of the predator species involved, as well as how they fit within the wider agroecological system. Global Positioning System (GPS) collar data were used from predators that had been captured as a result of farmer conflict and re-released to investigate various aspects of the spatial ecology of two predator species (leopards, *Panthera pardus* and cheetahs, *Acinonyx jubatus*) found on Namibian farmlands. This included home range sizes, methodological aspects of determining home range sizes, habitat preferences, spatial use of home ranges, spatial evidence of competition and differences between the two species. Specific habitats were identified as hotspots that are preferred by these predators and can be considered as having a high potential for conflict with farmers. From the actual conflict data, two methods of conflict mitigation were also evaluated, namely translocation and collar-and-release back at the capture site. No significance differences in the success of the methods were found when considering only the release site. The large home ranges found for both predator species also suggested that multi-farm management, such as is found in conservancies, are ecologically more likely to succeed.

Keywords: *Acinonyx jubatus*, behaviour, cheetah, home range size estimates, leopard, Namibia, *Panthera pardus*, spatial ecology

4.1 Introduction

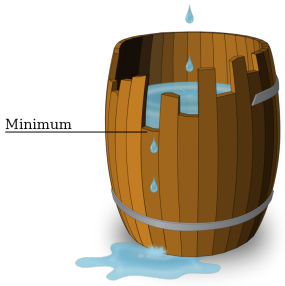
An estimated one in four mammal species are threatened with extinction and one in two are showing declining global populations (Schipper et al., 2008; Di Marco et al., 2014). For many predator species, conflict with humans is the major threat to their future survival (Messmer, 2000; Treves and Karanth, 2003b; Ray et al., 2005; Treves and Bruskotter, 2014; Ripple et al., 2014). Two predator species, the cheetah (*Acinonyx jubatus*, Schreber, 1775) and the leopard (*Panthera pardus*, Linnaeus, 1758), commonly found on Namibian farmlands, are classified as vulnerable by the IUCN (Durant et al., 2015; Stein et al., 2015). With an estimated 90% of Namibia's cheetahs on farmlands, occurring outside protected areas (Marker, 2000; Marker et al., 2003a; Ray et al., 2005) and the majority of leopard habitat in southern Africa occurring outside of protected areas (Swanepoel et al., 2013), this highlights the importance of livestock and game farmers in their continued survival (Ray et al., 2005; Balme

et al., 2014; Swanepoel et al., 2015). Not only do farmers manage most of their habitat, but conflict with farmers is also the greatest threat to their continued persistence on farmlands (Marker, 2000; Ray et al., 2005; Marker et al., 2010). The status of the cheetah in Namibia has recently been updated to endangered in the country, which has the world's largest remaining free-ranging cheetah population (Ray et al., 2005; Durant et al., 2015). Proposals have been forthcoming that their global status also be updated to endangered (Durant et al., 2017; Weise et al., 2017). This chapter focuses on the ecological part of the Namibian agroecosystem (Figure 1.4 on page 12) that plays a role in mitigating human-wildlife conflict on Namibian farms. The spatial ecology of predators can influence both the effectiveness of specific conflict mitigation methods (section 2.2.3 on page 39 of Chapter 2) and the sustainability of predation management techniques (section 2.3 on page 86). It is therefore to be included in the Decision Support System (DSS) to help farmers experiencing livestock depredation make informed management decisions (Chapter 5).

Balme et al. (2014) highlighted the fact that most research on large carnivores failed to address their most pressing conservation needs, human-wildlife conflict being especially important for leopards and cheetahs in Namibia and globally (Ray et al., 2005). Marker et al. (2008) showed the importance of the spatial ecology of cheetahs on Namibian farmlands in human-wildlife conflict. Stein et al. (2011) used the spatial ecology of leopards to address conflict mitigation around the Waterberg Plateau in Namibia. Although relatively few studies directly address the importance of home range size in human-wildlife conflict, a number of studies focussed on identifying conflict hotspots using the spatial ecology of predators (Carvalho et al., 2015; Miller, 2015; Miller et al., 2015, 2016).

Burt (1943) stated, "How can we manage any species until we know its fundamental behaviour pattern?". One of the most basic questions regarding animal ecology concerns home range size, habitat use and territoriality. Space is one important aspect of the ecological niche (Hutchinson, 1957) that is required to meet the limiting factors (Liebig's Law of the Minimum) for an individual.

Liebig's Law of the Minimum states that of all the different requirements for the survival of an individual or species (its fundamental ecological niche), only one factor will be limiting at any specific time (e.g. water, or sunshine, or a specific soil nutrient, or space, or soil pH, etc. for a plant; or herbaceous nutrient quality or quantity or drinking water availability or shade for a herbivore) — that limiting factor will prevent the population density to increase enough for the limits imposed by other factors to be reached. These n number of factors that can limit (at different times or places) the persistence of an organism and together form the n-dimensional hypervolume that is its ecological niche (Hutchinson, 1957; Blonder et al., 2014), are known as the limiting factors for the species or individual. For territorial predators, "safe" space itself can be one such limiting factor and space is always required to provide other potential limiting factors like prey or denning sites.



Burt (1943) first defined home range as "that area traversed by the individual in its normal activities of food gathering, mating, and caring for young. Occasional sallies outside the area, perhaps exploratory in nature, should not be considered as part of the home range. Often animals will move from one area abandoning the old home range and setting up a new one. Migratory animals have different home ranges in summer and winter - the migratory route is not considered part of the home range of the animal." He further clarified the relationship between home range and territory, "Home range then is the area, usually around a home site, over which the animal normally travels in search of food. Territory is the protected part of the home range, be it the entire

home range or only the nest", quoting the definition of Nobler (1939), "territory is any defended area". The spatial ecology of cheetahs and leopards in Namibia as applicable to farmer-predator conflict is investigated in this context of fundamental behavioural patterns.

4.1.1 Conflict mitigation methods

For two conflict mitigation methods used on Namibian farmlands, namely translocation and on-site collar and release no prior success probability could be found from a survey of Namibian farmers (Chapter 3). Translocation had a 0.45 prior success probability determined from the global literature (Chapter 2), but on-site collar and release had not been evaluated before. These two methods were evaluated for their effectiveness in reducing conflict with the two predator species. It was expected that translocation should have greater success than on-site collar and release, while the null hypothesis was that there would be no significant difference between the two methods.

4.1.2 A methods review of home range estimates

There have been many recent developments in obtaining accurate spatial behavioural data and their analysis (Kie et al., 2010; Gregory, 2017). However, many ecologists still use analysis methods that were developed for and are much better suited to the use of radio telemetry with fewer and less accurate data points (e.g. Melzheimer et al., 2018). Widely different home range sizes have been reported for cheetahs (Marker et al., 2008; Weise et al., 2015a; Melzheimer et al., 2018) and leopards (Stander et al., 1997; Marker and Dickman, 2005; Stein et al., 2011) in Namibia (Table 4.1 on page 175). Two contrasting hypotheses were considered to explain these differences:

1. The differences in reported home range sizes reflect actual differences in the home ranges of the two species in Namibia in different areas of the country.
2. The reason for the differences in reported home range sizes are primarily because of using different methods to calculate home ranges and the inherent biases and inaccuracies of these methods.

Moreover, from the published literature on Namibian home ranges of leopards and cheetahs, there was no way to determine why a specific method was chosen and what the possible advantages and disadvantages of the different methods could be. In this chapter some of the strengths and weakness of the different home range estimation methods are illustrated, hopefully causing less frustration in future for others having to decide on how to analyse their spatial data.

4.1.3 The spatial ecology of two Namibian predators

In Section 1.6.2 of Chapter 1 nine ecological factors with applications in farmer-predator conflict were identified, two regarding diet, five regarding spatial factors and two regarding intra-guild interspecific interactions. A literature review was used to answer as many of these questions as possible (Chapter 2). For Namibian leopards and cheetahs, the most important questions regarding prey preferences relevant to conflict with livestock farmers have been answered from the literature — both species prefer natural prey to livestock and both species show some individual prey specialisation. However, most of the spatial questions remained unanswered: 1) home range sizes and territorial behaviour of each predator species, 2) the proportion of resident to non-resident individuals in the predator population, 3) habitat preferences within their home ranges, 4) specialist usage of certain habitat types for certain activities, and 5) typical dispersal distances of each species. Of these questions, the proportion of resident to non-resident individuals would require a more detailed demographic study of the predator species than possible within the constraints of this project (Marker et al., 2003a). This proportion is

important, because combining it with home range sizes (and overlap) of the resident individuals enables us to calculate a density estimate of the species. Predator densities in turn, are important for two reasons: 1) for threatened species, lower densities means higher extinction risk and 2) higher densities of predators increases the frequency of chance encounters with livestock, even for opportunistic livestock killers and those species that prefer wild prey (Linnell et al., 1996, 1999). Dispersal distances are an important aspect of translocation success (see above; Linnell et al., 1997). Regarding interspecific interactions one relevant question is if there exists spatial avoidance of some high-use areas of the dominant species by the less dominant predators. The spatial ecological questions that were addressed in this chapter:

1. What are the actual home ranges of leopards and cheetahs in Namibia (descriptive)?
   (a) How do the home range sizes differ between the two species and sexes? — Hypothesis: they are different for different species and sex.
   (b) Does rainfall influence the home range sizes in different parts of the country, acting as proxy for prey biomass as claimed by Marker and Dickman (2005)? — Hypothesis: Home range sizes differ between rainfall classes.
   (c) Do different habitat types influence home range sizes (i.e. habitat rather than prey biomass determines home range sizes)? — Hypothesis: Home range sizes are different in different broad, coarse-scaled habitat types.
2. Do the species differ in their habitat preferences within their home ranges? — Hypothesis: Different habitat preferences of the species will provide evidence for possible niche partitioning between the two species (Loveridge and Macdonald, 2002; Bartnick et al., 2013; Dröge, E. and Creel, S. and Becker, M.S. and M'Soka, J., 2017; Flagel et al., 2017).
3. Do the species use their habitat differently, preferring certain habitats for certain activities? — Hypothesis: The two predator species will not use habitat according to availability, but will prefer different habitats for specific activities, like hunting (Balme et al., 2007)
4. Is there any evidence for spatial avoidance and/or intra-guild interference competition between the two species that is not explained by different habitat preferences? — Hypothesis: Cheetahs will tend to avoid the more dominant leopards irrespective of habitat (or be killed by leopards).

4.2 Materials and methods

4.2.1 Review of methods for home range estimates

Since it was suspected as one of the reasons for the great variance in published home ranges of Namibian cheetahs and leopards (Table 4.1), the methods used to determine the home ranges were investigated in the literature and compared with some newer alternatives. Different methods to estimate home ranges from GPS data were evaluated for advantages and limitations as a first step to decide which methods to use for describing Namibian cheetah and leopard spatial ecology. The importance of and the current lack of reproducibility in reported home range estimates were emphasized. Additionally, our own data was used to evaluate some of the common published methods, illustrate their strengths and weaknesses and suggest other alternatives. This was considered important because of the common lack of enough information in many published home range studies on the reasons why a specific method was chosen, as mentioned in the introduction.

Table 4.1: *Published home range sizes for Cheetahs and leopards in km²*. The variation in reported home range sizes is very large (e.g. the mean home range size of male leopards reported by Stander et al. (1997) are more than double the size of home range estimates by Stein et al. (2011) and Marker and Dickman (2005)).

Species	Author	n	Mean MCP	Range MCP	Mean 95% MCP	Range 95% MCP	CI	Mean 95% KDE	Mean 50% KDE
Leopard	Stander et al., 1997	4	188.5	182.9 - 294.4					
female	Stein et al., 2011	2			53	40- 66		109	12
	Marker & Dickman, 2005	5			179.0	52.4- 393.5			
Leopard	Stander et al., 1997	9	451.2	210 - 1163.5					
male	Stein et al., 2011	1			108			49.5	10.5
	Marker & Dickman, 2005	6			229.0	125.2 - 311.9			
Cheetah male	Marker et al., 2008	26			1664.9	266.3 - 5658.2		1436.5	215.2
territorial male	Melzheimer et al., 2018	30			379		313 - 441		
floater male	Melzheimer et al., 2018	28			1595		1156 - 2033		
Cheetah all	Weise, Lemeris, Munro et al., 2015	7	1558.8	217.0 - 2624.8					
Cheetah	Marker et al., 2008	15			1836.3	306.6 - 6353.7		2160.7	397.8
female	Melzheimer et al., 2018	17			650		507 - 793		

4.2.2 Data acquisition

Spatial data from the N/a'an ku sê Research Programme for cheetahs and leopards as described in Weise et al. (2014, 2015a) and Weise et al. (2015b) were used, from 2011 to the start of July 2018 and restricted to GPS collar data (i.e. excluding the less accurate VHF radio collar data to avoid triangulation inaccuracies). All data collection was done with the permission of the Ministry of Environment and Tourism and with ethics approval from Manchester Metropolitan University (clearance: RD1-06979434-20130923). Various collar types were used and GPS readings were taken at different time intervals ranging from hourly readings to 30 hourly, with a median of 8-hourly readings providing the best trade-off between useful data and collar longevity. This variation existed both between collars and sometimes for the same collar with higher frequency monitoring just after release, followed by lower frequency data once the animals appeared to settle in its home range. The data were from 16 cheetahs (with one premature collar failure) and 34 leopards (with 12 premature collar failures). Except for eight animals that were captured specifically for research purposes on N/a'an ku sê land, the other individuals were all the result of conflict calls received from farmers. Twenty-three calls were due to livestock losses, eight as a result of game losses, four accidental captures where the farmer was targeting another species, three animal rescues (e.g. where animals were kept in captivity illegally), and seven carnivores captured for

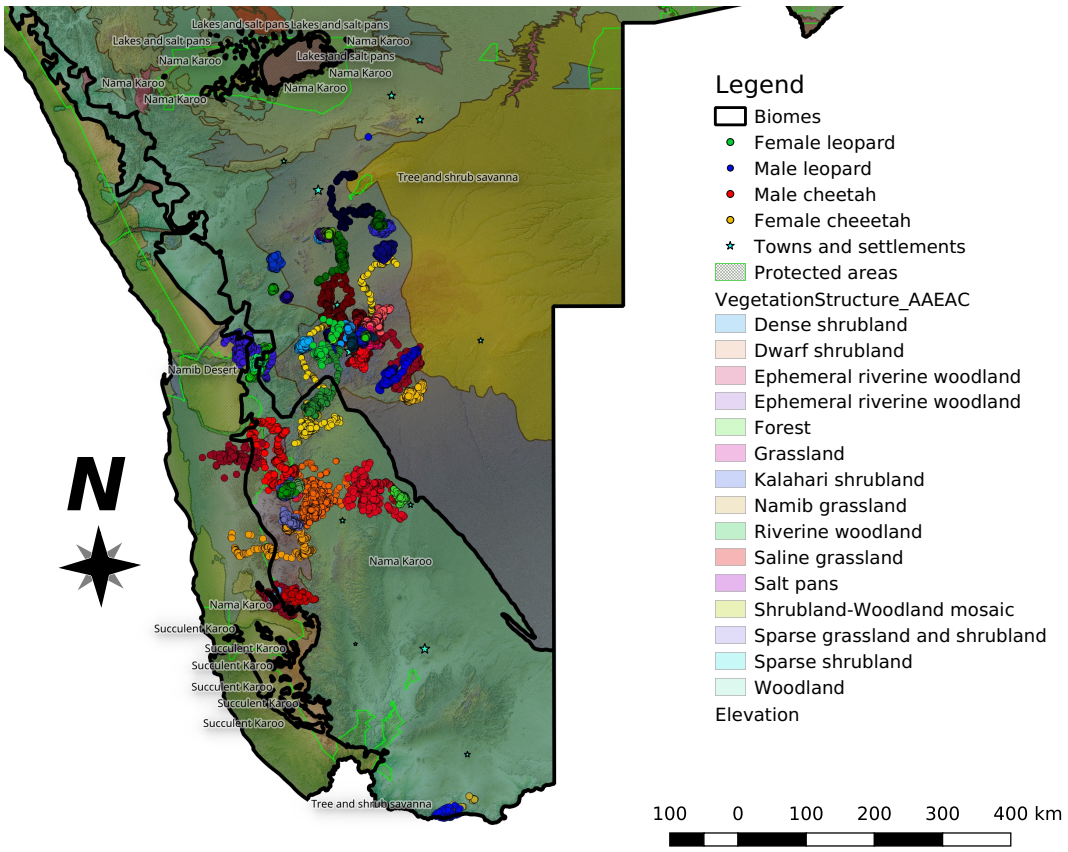


Figure 4.1: *Locations of released predators in Namibia.* Leopards are green (female) and blue (male) and cheetahs red (males) and orange-yellow (females). Biomes are indicated with labels on the map, vegetation structure by colours.

unknown reasons.

Initially, most predators were translocated to protected areas as a conflict mitigation method (Weise et al., 2015a,b). However, removal of predators (by killing or translocation) is known to have questionable long-term success in conflict resolution (Linnell et al., 1997; Fischer and Lindenmayer, 2000; Athreya et al., 2010; Weilenmann et al., 2010; Weise et al., 2014, 2015a,b; Ropiquet et al., 2015). Also, as most predators are known to prefer natural prey rather than livestock (Marker-Kraus et al., 1996; Stein, 2008), and because farmers can not always be sure that the trapped individual was responsible for livestock losses, an alternative mitigating method that was implemented at a later stage, was to convince the farmer to have the predator collared and released, with daily updates on the predator's movements. In this way the farmer could manage his livestock to avoid areas frequented by the predator, or to protect his livestock more intensively whenever the predator was present on the farm. Farmers were also able to check locations of GPS point clusters to determine if a predator had made a kill and what it had killed (Anderson and Lindzey, 2003; Fröhlich, 2011; Martins et al., 2011). This was used primarily as a conflict mitigation method (see Chapter 2) and most farmers did not keep record of what the predators killed (as long as it was not livestock). The normal procedure was for N/a'an ku sê personnel to investigate if there was no GPS collar movement for more than 24 hours and retrieve collars to confirm the cause of death within a week after the collar stopped moving.

Eight cheetahs and seven leopards were translocated to new locations (47 - 754 km from the capture site, median = 257 km, mean = 98 km, n=15), one leopard was re-wilded (i.e. a rescued cub released after a year in captivity allowing it to first mature), and eight cheetahs and 25 leopards were collared and released on-site (i.e. at the capture site, presumably inside their original home range). The releases were classified as either successful or not, using the criteria of at least one successful reproduction event after release, or surviving more

than 300 days within an established home range, without killing livestock or conflict with farmers[1]. Conflict (including human-caused death) before having the opportunity to breed, or killing livestock more than once (i.e. habitual livestock killers) after release, were considered as an unsuccessful release. Where the collar died prematurely (before 300 days), it was classified as an unknown outcome. Using these criteria, the effectiveness of translocation *versus* collar and release on-site were compared as conflict mitigating measures by comparing the proportion of successful releases in each group (see analysis below).

All geographical data for habitat analysis were sourced from freely available Geographical Information System (GIS) files (`http://www.the-eis.com/`) from the Atlas of Namibia project (Mendelsohn et al., 2003).

4.2.3 Data analysis

All data analyses were carried out using QGIS (QGIS-Development-Team, 2017), GRASS (Neteler et al., 2012), and R (R Development Core Team, 2011). Spatial data were all projected to the Africa Albers Equal Area Conic projection. Significance testing was done using permutation tests (resampling/Monte Carlo) (Simon, 1997; Yu, 2003; Hothorn et al., 2010; Puth et al., 2015) in R (R Development Core Team, 2011). For comparing two groups (e.g. number of successful releases for translocated *versus* on-site releases, or home range sizes between two species or two sexes), two-tailed resampling without replacement of the difference between the mean and median of the two groups was used (a resampling version of the t-test). The median was used instead of the mean when there were one or two outliers in the data that could have a large effect on the means. For those factors where the independent variable was continuous, a resampling version of regression was used (using the slope of the regression line). For comparing multiple groups, resampling the F statistic was used (resampling analysis of variance). For comparing habitat use *versus* habitat availability a contingency table was used and resampled to compare the observed use:availability ratio to 10 000 random use:availability ratios possible from shuffling the same data as a permutation version of the Chi-squared test. All resampling tests were randomised 10 000 times. The resampling script examples can be found in Appendix B.

For determining home ranges, the 100% MCP and the Concave Hull methods were implemented and mapped directly in QGIS. We used the QGIS Concave Hull plug-in, with a setting of three neighbours for all Minimum Concave Hull estimates. The number of neighbours chosen had no perceivable effect on the resulting map or home range size. The rhr package in R (Signer and Balkenhol, 2015) was used for the 50% (core area in home range) KDE and 95% KDE estimations. Using the rhr package in R (Signer and Balkenhol, 2015), the 100% MCP gave exactly the same results as QGIS and rhr was considered as consistent with QGIS for MCP. The KDE smoothing parameter (h) was determined using rhrHrefScaled in rhr (the rule-based *ad hoc* method as described by Kie, 2013). For comparison, using the same value for h**,** QGIS and GRASS (using heatmap and point sampling in QGIS, followed by r.reclass in GRASS and polygonize in QGIS) were used to determine and map KDE. For low convex hull methods (LoCoH) rhr was used and compared with the T-LoCoH R package (Lyons et al., 2013). T-LoCoH was also used for classifying predator movements within their home ranges. Following the advice of Getz et al. (2007), the a-LoCoH method was used as being the most robust for determining LoCoH with the rule-of-thumb of the greatest distance between any two locations in the home range being used for the value of a (kernels are constructed from all points within the radius a such that the distances of all points within the radius to the reference point sum to a value less than or equal to a).

Following the recommendations of Laver and Kelly (2008), we tested whether and when the home range estimates reached an asymptote using rhrAssymptote in the rhr package for R. This would indicate the number of locations for determining a home range size where more location data does not increase the accuracy of the estimate.

Various researchers have used cluster analysis successfully to illuminate hunting behaviour (kill frequency, habitat and prey preferences) of predators (Anderson and Lindzey, 2003; Martins et al., 2011; Fröhlich, 2011).

[1]Conflict was defined as either a farmer killing the predator (or trapping it) or calling a conservation organisation asking for the predator to be removed.

The T-LoCoH package in R was used to build a predictive model from the data identifying locations of different behaviour (hr_in.r, HomeRanges.r and hr_out.r). Probable kill sites of leopards and cheetahs were identified by T-LoCoH as clusters (more than 12 hours spent around the same point). A visit was classified as 48 hours for cheetahs and five days for leopards before becoming a re-visit, the maximum observed time that they would stay on a single carcass during the spoor tracking pilot study period. In T-LoCoH, locations were classified as 1) probable kills, showing locations that were visited only once, but where the predator stayed around the same area for a long period; 2) preferred locations which were locations that were most often visited; 3) corridors those single GPS locations that were far removed from both the previous and next GPS reading; 4) exploratory movement indicating points that were only visited once and where no time was spent; 5) and sometimes marking/denning sites, those locations that were both relatively frequently visited and where the animal spent long periods of time. Possible denning sites of females were identified as sites with both frequent usage and long periods (more than 12 hours) spent there. Many locations were also unclassified.

Habitat types were created by intersection of ArcGIS shape file habitat layers in QGIS using layers from the Atlas of Namibia project (Melzheimer et al., 2018). The biome, vegetation structure, rainfall and topography (mountains, rocky hills, dunes, plains, drainage lines) were combined by intersection to classify the available habitat into 95 unique habitat types. A second combination of factors excluding rainfall as a factor, resulted in 51 larger, unique habitat types. The core area of the predator was used to assign it to a certain habitat, in order to compare home range sizes between habitats in Namibia.

The most straightforward and usual way to look at habitat use and preference in a single study site, is Chi-square analysis where you compare the relative availability of the different habitat types in the home range of the animal to actual use (or relative time spent) in each habitat (Luyt, 2004). If the animals spend relatively more time in a certain habitat than its available area, it is a preferred habitat; if it spends less time in a habitat type than its available area, it is evidence of avoidance (if the observed difference between the time spend in a certain habitat and its availability is less extreme than would be expected from random usage, the null hypothesis of no preference or avoidance cannot be rejected and it is accepted that animals use habitat according to availability). If the animal spends more time on specific activities in certain habitat types in its home range (out of proportion to its availability), it is evidence of being a preferred habitat for that specific activity. In this study an alternative approach was followed. Within the home range of each animal, core areas (50% KDE) and high usage areas (from T-LoCoH) were used to identify preferred habitat. Then the habitat type of the core areas was determined (out of the available habitat types within the home range).

This approach was considered as more appropriate for cases where different individuals at different study sites are used to investigate habitat use within the range of the species in a region or country. Because the whole commercial farming area of Namibia as well as parts of protected areas (see Figure 4.1) were the "study site" of this project, the usual method of determining preference as area used *versus* available surface area of the different habitats were not considered realistic. This was because not all the "available habitat" would truly be available to an individual animal (e.g. the Shrubland-woodland Mosaic vegetation structure is a large proportion of the available commercial farmlands, but very few animals were released in an area where it was an available habitat choice; even though having a "large available area" it should be considered as having a low availability for the released animals). Similarly, measuring the size of the core areas within a certain habitat as an indication of usage, would also give a skewed result (home ranges and core areas would be *smaller* in higher quality landscapes). Instead, the *frequency* of occurrence of a habitat type within the *home ranges* of the released predators were used as its "availability" and the frequency of its occurrence within the *core or high usage areas* as "use". If a large number of animals are used and a greater proportion of their core areas are found in certain habitat types than could be explained by randomly reassigning core areas to habitat types (resampling without replacement), this would indicate a preference of those habitat types. I used a resampling version of the χ^2 test to test for any significant preference of habitat for specific behaviour (including "preferred habitat") — see Section B.2.0.7 in Appendix B.

The lack of overlap between individuals of the same species and sex were used as an indication of territoriality

(Fieberg and Kochanny, 2005). This was more complicated for male cheetahs as they use two different spatial strategies with territorial males having relatively small, defended territories and "floaters" having overlapping home ranges that are often larger than that of females (Melzheimer et al., 2018). If the two species showed evidence of different habitat preferences, combined with a lack of overlap between individuals of different species in the same general area and time, it can be seen as evidence for spatial niche partitioning. If farmers have livestock that are more vulnerable to one predator than the other (see Table 3.3 in the previous chapter), such spatial niche partitioning can be used to reduce livestock losses by moving vulnerable livestock away from the preferred habitats of the more dangerous predator. Where both species overlap, it was hypothesised that when interspecific competition lead to exclusion of cheetahs from some areas by leopards, the overlap between leopards and cheetahs should be smaller than that between cheetahs (who are mostly not territorial) and more similar to that between leopards of the same sex (who are territorial). The percentage intraspecific overlap between cheetahs was compared to the interspecific overlap between cheetahs and leopards using a one-tailed resampling test of means.

The change in relative numbers of the two species involved in conflict calls over time was tested using a resampling regression of year (independent variable) against species (coded as leopard = 1, cheetah = -1) (resampled regression 1-tailed, Section B.2.0.5 in Appendix B). Although strictly speaking conflict calls included both farmers asking for advice to prevent livestock depredation calling after having caught a predator in a cage trap, only the data from captured animals were used for this analysis (conflict calls in general showed a similar pattern, however).

Finally, an indication of the overall density of each species on Namibian farmlands could be calculated using the following formula assuming that all resident individuals within a certain area TotArea have been collared (see Figure 4.2):

$density = n/area = ((TotArea + TotOverlapArea)/AverageHomeRangeSize)/TotArea$

with

- TotArea = Total area within the outer boundaries of the overlapping/touching home ranges. This was done by calculating the geometric union of all the home ranges using the "union" function in QGIS.
- TotOverlapArea = Sum of overlap areas (where each overlap area = area × (number of individuals overlapping-1)) This was done in QGIS by using the "intersection" function to calculate each area of overlap and using the number of layers intersecting there - 1 (as well as the "difference" function to prevent the same area from being counted more than once), summed to include all overlapping areas.
- AverageHomeRangeSize = mean home range size.

4.3 Results

4.3.1 Translocation vs. On-site collar and release

Of the 15 translocated animals in this study, eight had a clear initial phase of exploratory movements (9 – 176 days, mean = 53 days) before settling in a home range (e.g. Figure 4.3b)) while of the 33 animals that were released back to the same farm where it was captured, only three showed exploratory movements (1 - 19 days, mean = 8 days). One animal clearly showed an initial home range (territory) followed by moving in a large arc of about 130 km over 19 days, before setting up a new home range 55 km to the South (Figure 4.3a) on page 184).

There was no significant difference in the success rate of the translocation vs. on-site collar and release methods for mitigating human-wildlife conflict ($p > 0.05$, resampling mean difference 2-tailed, Section B.2.0.4 of Appendix

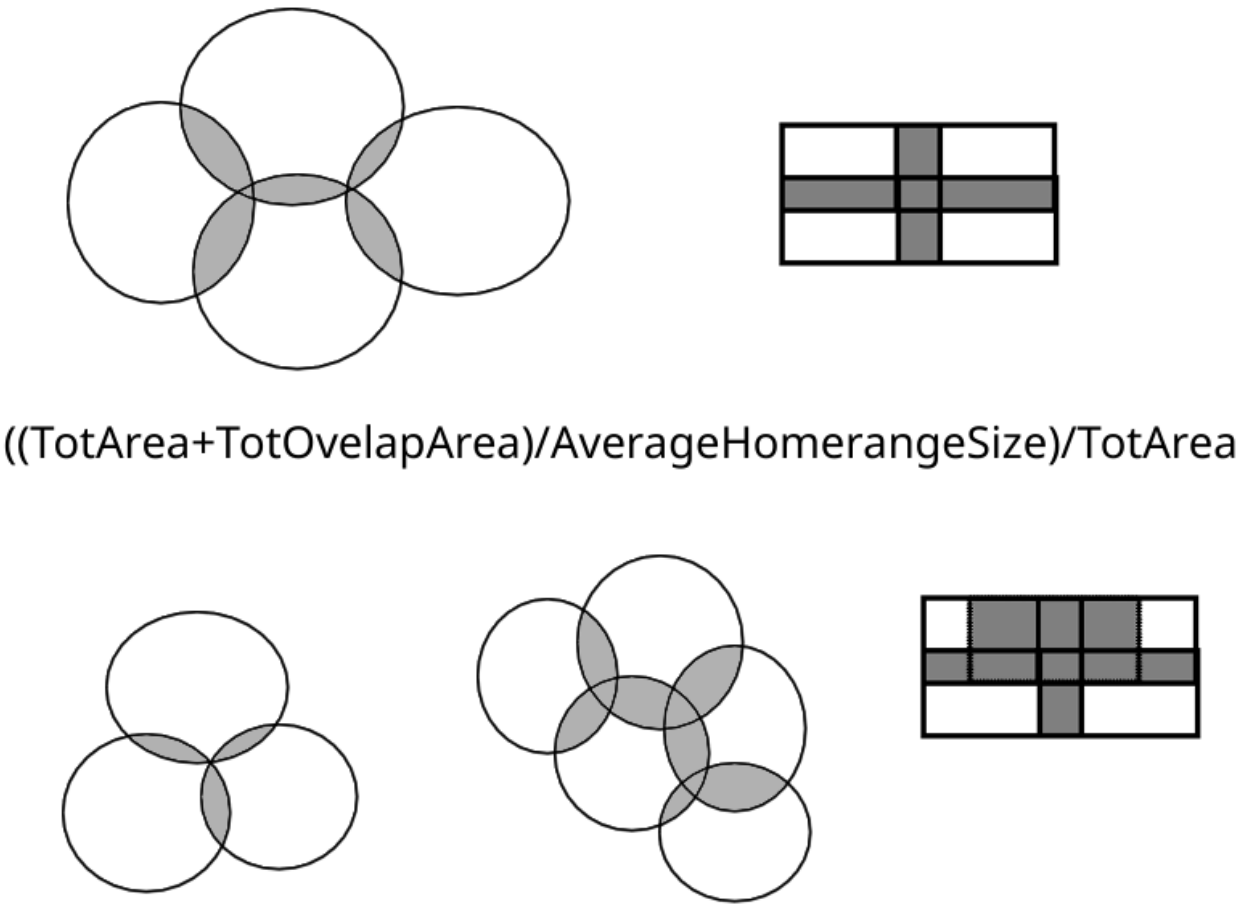


Figure 4.2: *A schematic representation of the generalised density estimate for solitary animals with overlapping home ranges.* The grey areas are the total overlap. The total area is everything inside the outer borders of the individual home ranges. With greater sample size and if all individuals within the outer boundaries are collared, the density estimate should become increasingly accurate. For a larger area (e.g. all of Namibia) the mean of such density estimates in different parts of the country (e.g. the three groups of circles in the figure) can be used to extrapolate an overall mean density.

B), with both methods being successful more than 60% of the time as defined above in Section 4.2.2. However, translocated animals showed significantly larger home ranges when including initial exploratory movements than animals that were released on-site ($p < 0.01$ for both Concave Hull *and* 95% KDE and for both species, resampling mean and median difference 1-tailed, Section B.2.0.4 of Appendix B). Of the 35 released leopards, four (11.42%) killed multiple livestock within 300 days after their release; i.e. they were probably habitual livestock killers (Stander, 1990; Linnell et al., 1999). Two cheetahs out of the 16 released in total (12.5%), killed multiple livestock within 300 days after their release.

For the on-site collar and release method this resulted in a prior success probability of 0.64. Translocation had a prior probability of success of 0.78 according to the success criteria used here.

4.3.2 Evaluating methods of home range estimates

4.3.2.1 Literature overview

One of the oldest, simplest and most widely used estimators of home range is the Minimum Convex Polygon (MCP). However, Burt (1943) already stated that because of the different shapes of home ranges "to connect the outlying points give a false impression of the actual area covered. Not only that, it may indicate a larger range than really exists." "Overlapping of home ranges, based on these calculated areas, thus may at times be

exaggerated." (see also Getz and Wilmers, 2004). Home range estimates using MCP are also very sensitive to inaccurate triangulation when using radio VHF collars and to outlying points that might reflect exploratory movement rather than "normal activities of food gathering, mating, and caring for young" (Burt, 1943; Getz et al., 2007). On the other hand, MCP is often more accurate when the sample size is small (Gregory, 2017) and the actual home range has a fairly regular convex shape. As sample size increases, the Minimum Concave Hull should give a better estimation of the actual shape and size of home ranges when compared to MCP (e.g. see Figure 4.4).

The definition of home range explicitly excludes migratory routes and exploratory movements (Burt, 1943). Typically, this issue of exploratory movement being included by accident in the home range estimate, is handled by using only the inner 95% of the points for the area. However, if an animal spent a long period in exploratory movement outside its home range, it will still overestimate the home range size, whereas if the animal did not leave its home range, it will underestimate the home range size. Kie et al. (2010) mentioned that home range estimation techniques such as this common use of an arbitrary inner 95% of points, were originally developed for VHF telemetry technology and other field methods with limited accuracy – and commonly with only one or even less observations per day. They are therefore often not well suited to modern GPS telemetry data. It is proposed here that instead of using a fixed 5% exclusion of data points, where enough data exists the actual geographical data should be used to identify exploratory movements. We used the criterion of "revisit"[2] to distinguish between exploratory and home range movement. If the animal did not return to the same general area within a year, it was classified as exploratory movement since it could not be part of the area "over which the animal normally travels in search of food" (Burt, 1943). This typically appeared as a single line of movement points, often in the same general direction (and any clusters on the line did not show the animal leaving and returning to the same cluster – no revisits). In contrast to the recommendation by Laver and Kelly (2008), removing data in order to reach "statistical independence" of points or avoiding serial autocorrelation will preclude the use of this method — identifying single lines of movement will become impossible.

Laver and Kelly (2008) made a plea for consistency in home range studies, that the methods used should be described in enough detail to be replicable and comparable across studies. Mitchell (2006) showed that even when using the "same" method, different software packages could give different results. Signer and Balkenhol (2015) described an open source software package (rhr) written in R, that can be used to help ensure reproducible home range estimates. Various authors (Laver and Kelly, 2008; Gregory, 2017) made the point that more than one method may be necessary for home range estimation and utilization distributions, depending on the species and questions asked. Kie et al. (2010) argued that as the accuracy of spatial data points and the number of readings increase with modern technology, traditional estimators of home range become less useful.

Minimum Convex Polygon (MCP) and Concave Hull methods only delimit an animal's outer home range boundaries, but give no indication of how the home range is used or where the animal spends most of its time. Utilization Distributions (UD) are commonly used to identify high usage within the home ranges (Powell, 2000). It is most often interpreted as a probability distribution of where the animal is most likely to be found, with the 95% probability contour as the outer boundary of the home range and the area within the 50% probability contour as the core range. The most commonly used Utilization Distribution estimates are various versions of the Kernel Density Estimate (KDE) (Gregory, 2017). Laver and Kelly (2008) recommended that KDE rather than MCP should always be used for a number of reasons. Furthermore, Getz and Wilmers (2004) argued that with the increasing number of location readings and accuracy of GPS, the shape of the home range can be the more important problem with using MCP – predators often have home ranges with sharp edges along natural barriers like rivers or mountains, resulting in home ranges with irregular shapes, holes and sharp edges or corners, unlike the convex shape assumed by MCP or smooth shapes assumed by KDE (*cf.* buffalo *Syncerus caffer* home ranges, in Getz et al., 2007). The issue of sensitivity to outliers can be handled by excluding areas that are not revisited (within a year). KDE in turn, has been criticized mostly for two weaknesses: 1)

[2]Any cluster of points covering more than 24 hours, which has at least one point that is more than a week apart from the others, is defined as a revisit

being too sensitive to the smoothing parameter/bandwidth and exact selection method (e.g. fixed kernel vs. adaptive kernel, and user-defined vs. reference vs. least-squares cross-validation selection of the bandwidth) and 2) including areas within the home range estimate that are known not to form part of the home range (i.e. a Type II error, similar to MCP) (Getz and Wilmers, 2004; Benhamou and Cornélis, 2010; Gregory, 2017). As an alternative, Getz and Wilmers (2004) proposed the Local Convex Hull (LoCoH) method. Like KDE, this also highlights areas of high use within the home range, while simultaneously giving the advantage of identifying sharp home range boundaries (Getz and Wilmers, 2004; Getz et al., 2007). However, LoCoH methods are more likely to exclude areas that are actually part of the home range (Type 1 error) (Lichti and Swihart, 2011). KDE should thus be preferred when wanting to make sure that no part of the home range is mistakenly excluded, while LoCoH should be more important when wanting to ensure that no part of the available habitat that is not used, is included by mistake, (e.g. it is more likely to show territoriality as a lack of overlap in home range). From the literature it is difficult to determine which of the KDE or LoCoH is more accurate, with contradictory results being reported (e.g. Getz et al., 2007 vs. Lichti and Swihart, 2011). For this reason we used and compared both methods below.

Both methods have also been improved more recently, taking into account the improvements in data quality from using GPS-based location measurements (Kie et al., 2010). The traditional KDE has been criticised for not taking into account the actual direction of movement of the animal provided by serial correlation between GPS measurements and considering serial correlation as a flaw (Laver and Kelly, 2008), rather than useful data. Independently, two alternative methods for KDE Utilization Distribution estimation were developed based on using the line of movement between recorded locations as the basic unit of analysis, rather than single points (Benhamou and Cornélis, 2010; Steiniger and Hunter, 2013). However, since neither method has been implemented to work with an Open Source GIS and would need some programming to adapt for use with an equal-area projection, they were not included in our analysis, but are mentioned here as fruitful avenues for future analysis. The LoCoH approach has been similarly extended to include time as another dimension (T-LoCoH) with the ability to distinguish between areas of high (repeated) use (an indicator of high habitat quality or core territory within the home range) and areas where the animal spent an extended period near one point, but only infrequently (e.g. where it made a kill) (Lyons, 2014; Lyons et al., 2013).

4.3.2.2 Comparing home range size estimates using data

Most home range studies fail to mention the projection used for the geographical data. We found that the popular WGS 84 UTM zone 33 S projection gave an area 5 km^2 smaller than the actual 461 km^2 for a home range in the south of Namibia, while giving a home range size 5 km^2 larger than the actual 405 km^2 in the north of the country. It is therefore important to use area-preserving projections when doing home range estimates.

Mitchell (2006) demonstrated that KDE estimates often differed when using different software. Similar results were seen here. When estimating the 100% LoCoH using the rule-of-thumb value of a in rhr still resulted in a LoCoH estimate that was smaller than when using the tlocoh R package with a time value of 0 and similar a (where a is the adaptive radius as defined by Getz et al. (2007) and chosen as the maximum distance between any two points in the data set). Similarly, the KDE smoothing parameter (h) was determined using rhrHrefScaled in rhr (see Materials and methods above). However, using the same value for h in QGIS (using heatmap, point sampling, r.reclass in GRASS, and polygonize) gave KDE values that were less than half of that calculated by rhr in R. This highlights the importance of reporting the software used for all analysis (Laver and Kelly, 2008; Mitchell, 2006) — even when using the exact same method, the implementation used by different software can give different results.

In the literature overview the importance of not including exploratory movement in the home range estimate was mentioned (Burt, 1943). Figure 4.3 illustrates the effect of including exploratory movement. Figure 4.3a shows a leopard male (N094) that was released in the same area where he was captured. He remained in that area for 73 days before moving for approximately 150 km over a period of 19 days to set up a new territory 55

km to the south. As can be seen, if the exploratory movement had not been identified and excluded, the MCP would give a highly overestimated 3291.0 km² home range. The Concave Hull would still give an overestimated 1387.2 km², but its shape would alert one to an irregularity. If the movement from the first territory to the second territory is excluded (as it should be per definition of home range), an initial northern Concave Hull home range size of 25.9 km² (33.4 km² MCP) and a southern home range of 158.8 km² (224.8 km² MCP) were found. This is a difference of more than an order of magnitude. Figure 4.3b shows a leopard female (N162) translocated less than the 200 km minimum recommended by Weise et al. (2015b), travelling for about 23 days, covering 95 km measured in a straight line south south-west, to return to her original home range. If this travel time had not been identified, a MCP home range of 3490.99 km² would be estimated, instead of the actual Concave Hull territory estimate of 551.3 km² (803.6 km² MCP). From Table 4.2 it was also clear that including the initial exploratory movement as part of the home range, would result in significantly larger home range estimates (For 95% KDE: $p < 0.05$ for all predators, resampling mean difference 1-tailed, R script: Exploratory-or-not-KDE.r; for Concave Hull: $p < 0.05$ for leopards, $p \approx 0.06$ for all predators, resampling mean difference 1-tailed, R script: Exploratory-or-not.r).

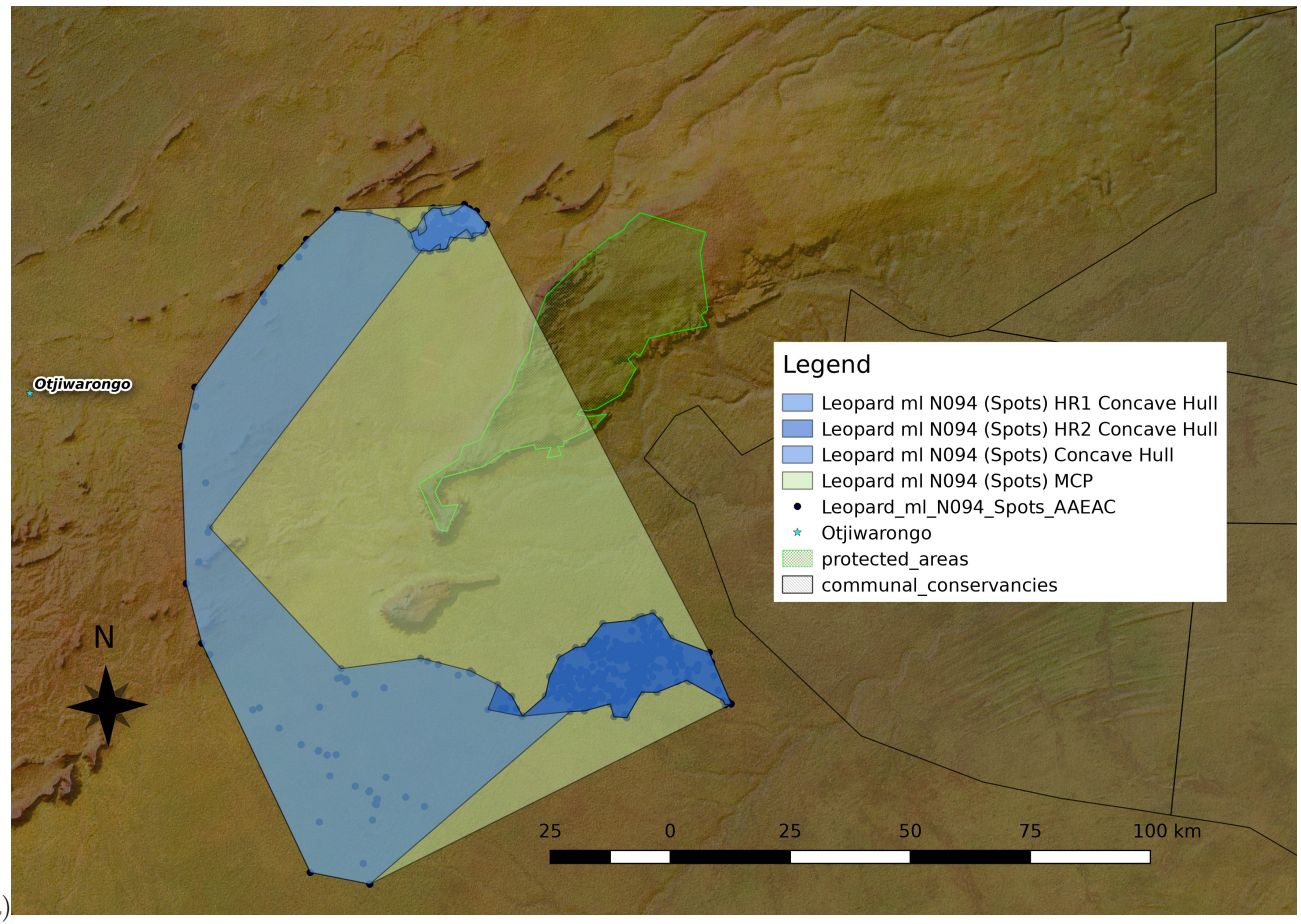
It is clear from Table 4.2 that the different estimators of home range size gave different results. The (100%) Minimum Convex Polygon (MCP) consistently gave the largest home range estimate. This is probably an overestimate of the actual home range (Type II error – see Figure 4.4) including areas that are not used — there was a significant difference between the estimated home range sizes using the different methods (observed $F = 2.83$, $p < 0.05$, resampling for F value between groups MCP, 95% MCP, Concave Hull, 95% KDE, LoCoH and T-LoCoH). The 95% MCP (mean = 556.09 ± 955.58 SD km²) estimates were mostly not too different from the Concave Hull estimates (mean = 538.12 ±937.13 SD km²) (Two-tailed resampling of means, $p > 0.05$). However, as the most visible exploratory movements were already excluded (using the "revisit" criterion), it could be prone to Type I errors by excluding some areas that should be included.

The median home range estimates were consistently smaller than the mean, showing that the mean is skewed by a few very large home ranges and that most predators have home ranges that is smaller than the mean (Table 4.2). For comparison with previous studies that have not explicitly excluded initial exploratory movements from the calculating the home range, we also indicated the average home ranges estimated by Concave Hull and 95% fixed-kernel KDE when such exploratory movements were included.

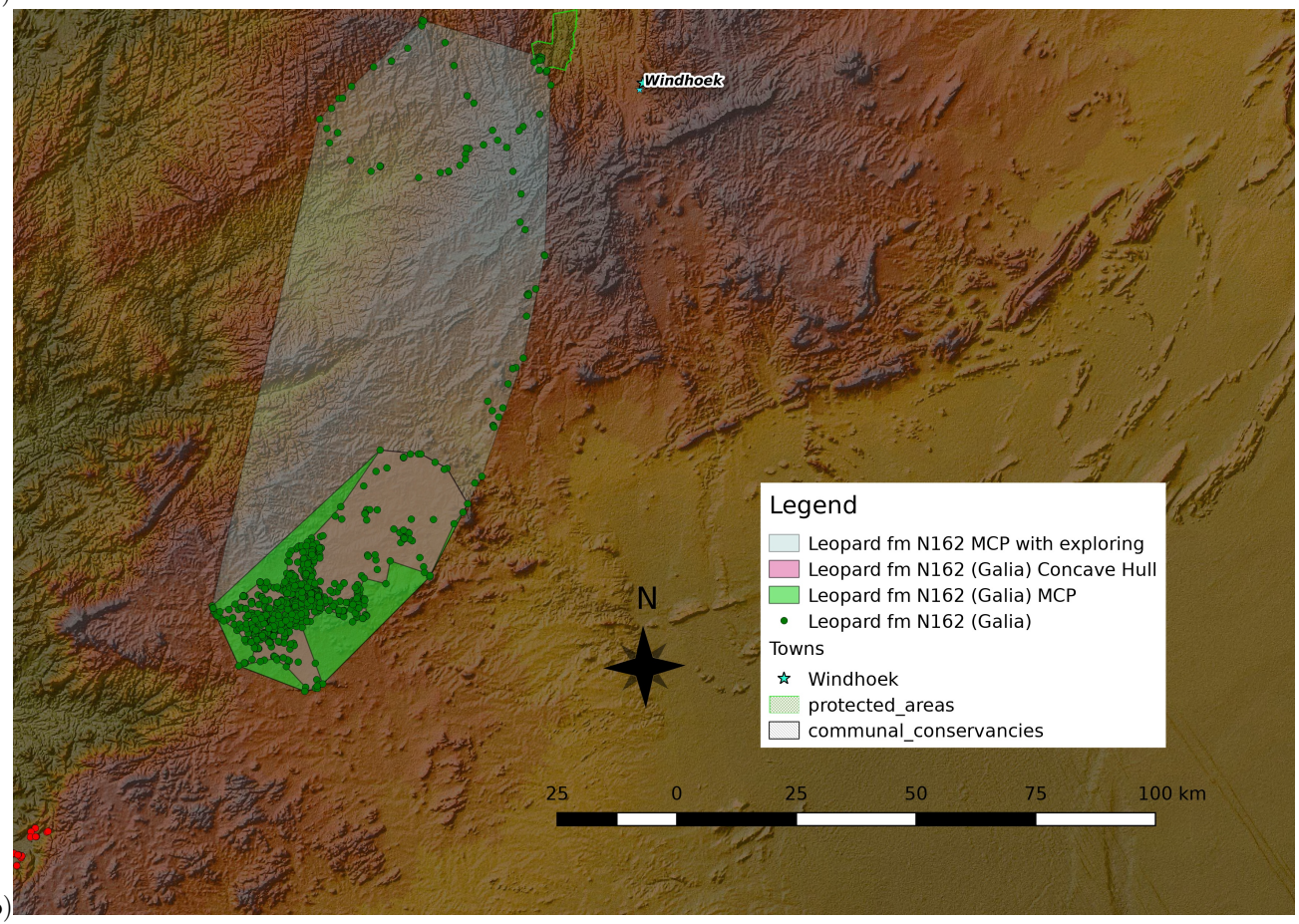
At least for those animals where a reasonable number of location data points (at least 300 days and 600 points) were recorded, the Concave Hull estimate appears accurate (as long as exploratory movements were excluded). Its coefficient of variation was similar to that of the other methods and generally improved as the sample size increased (text box *Coefficient of variation for different home range estimators* below).

Coefficient of variation for different home range estimators:
Female leopard: Concave hull cv = 95.2% | MCP cv= 90.9% | 95% MCP cv = 104.8% | 95% KDE cv = 102.4% | 50% KDE cv = 90.6% | LoCoH cv = 77.7% | T-LoCoH cv = 75.8%
Male leopard: Concave hull cv = 98.7% | MCP cv= 101.7% | 95% MCP cv = 101.1% | 95% KDE cv = 121.85% | 50% KDE cv = 121.9% | LoCoH cv = 86.5% | T-LoCoH cv = 82.0% .
Male cheetah: Concave hull cv = 87.1% | MCP cv= 91.7% | 95% MCP cv = 104.0% | 95% KDE cv = 108.5% | 50% KDE cv = 140.7% | LoCoH cv = 97.3% | T-LoCoH cv = 75.9% .
Female cheetah: Concave hull cv = 153.3% | MCP cv= 135.3% | 95% MCP cv = 115.6% | 95% KDE cv = 105.6% | 50% KDE cv = 118.1% | LoCoH cv = 145.3% | T-LoCoH cv = 146.0% .

For example, Figure 4.4 shows that the MCP of a male cheetah (N080) included the city of Windhoek as part of the home range, which is unlikely as there has been no known record of cheetahs using urban habitat in Namibia contra leopards in India (*cf.* Athreya et al., 2016). Moreover, the shape of the Concave Hull suggested essentially two separate home ranges connected by a relatively small corridor which is not shown by the Minimum Convex Polygon (MCP). This was confirmed by further investigation of the data: From the start of the monitoring on 29 May 2014 until 9 September 2014, the cheetah was in the highlands to the East of Windhoek, then he



a)



b)

Figure 4.3: *Effect of exploratory movements on home range estimates*: *a)* Leopard male N094 (Spots) was released on the farm where he was first captured. *b)* Leopard female N162 was translocated less than 200 km and subsequently returned from the protected area where she was released to the same area where she had been captured.

Table 4.2: *Home range sizes in km² for the different predator species and sexes using different estimators.* Shown as median with range and mean with standard deviation.

Species	Measure	MCP	95% MCP	Concave Hull	95% KDE	50% KDE	LoCoH	T-LoCoH	Exploratory Concave Hull	Exploratory 95% KDE
Leopard Female	Median	215.498 (28.74 - 803.59)	90.434 (22.72 - 491.52)	113.291 (21.43 - 551.31)	129.550 (25.89 - 668.44)	32.390 (7.08 - 118.05)	57.845 (13.45 - 189.14)	86.420 (18.60 - 264.54)	140.162 (21.43 - 1435.00)	135.763 (25.89 - 2157.61)
(n = 13)	Mean	249.729 (± 226.94)	150.388 (± 157.56)	161.577 (± 153.85)	188.554 (± 193.64)	35.658 (± 32.31)	70.895 (± 55.13)	103.496 (± 78.49)	395.965 (± 527.52)	492.562 (± 761.04)
Leopard Male	Median	220.923 (33.42 - 1324.12	141.919 (25.26 - 896.87)	170.031 (25.93 - 1068.71)	164.689 (25.78 - 1400.72)	40.228 (3.77 - 243.35)	92.604 (3.45 - 404.28)	139.651 (7.42 - 548.47)	247.990 (43.15 - 1458.73)	181.638 (25.78 - 2488.25)
(n = 21)	Mean	320.770 (± 326.30)	225.354 (± 227.73)	243.393 (± 240.25)	272.197 (± 331.69)	59.183 (± 60.39)	112.138 (± 97.00)	174.779 (± 143.34)	406.542 (± 452.09)	549.714 (± 779.64)
Cheetah Male	Median	1565.722 (317.60 - 6748.90	1113.760 (199.31 - 5626.98)	1059.952 (211.38 - 4028.47)	930.784 (161,88 - 5688.62)	140.138 (13.53 - 1225.13)	320.935 (78.35 - 1669.21)	631.373 (165.42 - 2178.55)	1249.515 (211.38 - 4028.47)	1045.21 (161.88 - 5688.62)
(n = 11)	Mean	2065.066 (± 1894.15)	1433.995 (± 1491.02)	1307.574 (± 1139.55)	1401.716 (± 1520.98)	239.451 (± 336.82)	475.980 (± 463.21)	818.057 (± 621.00)	1785.543 (± 1398.84)	1798.691 (± 1690,22)
Cheetah Female	Median	497.023 (203.80 - 5962.02)	405.410 (191.97 - 3225.48)	290.432 (117.09 - 4860.85)	644.799 (196.05 - 2851.04)	112.436 (30.42 - 557.48)	147.601 (29.43 - 1325.18)	250.124 (25.41 - 2124.33)	2575.076 (156.12 - 4860.85)	2851.042 (330.65 - 17155.23)
(n = 5)	Mean	1799.067 (± 2433.85)	1110.816 (± 1283.70)	1321.331 (± 2025.97)	1020.751 (± 1078.35)	184.078 (± 217.47)	373.040 (± 542.17)	597.021 (± 871.61)	2418.971 (± 2171.09)	4848.771 (± 7000.79)

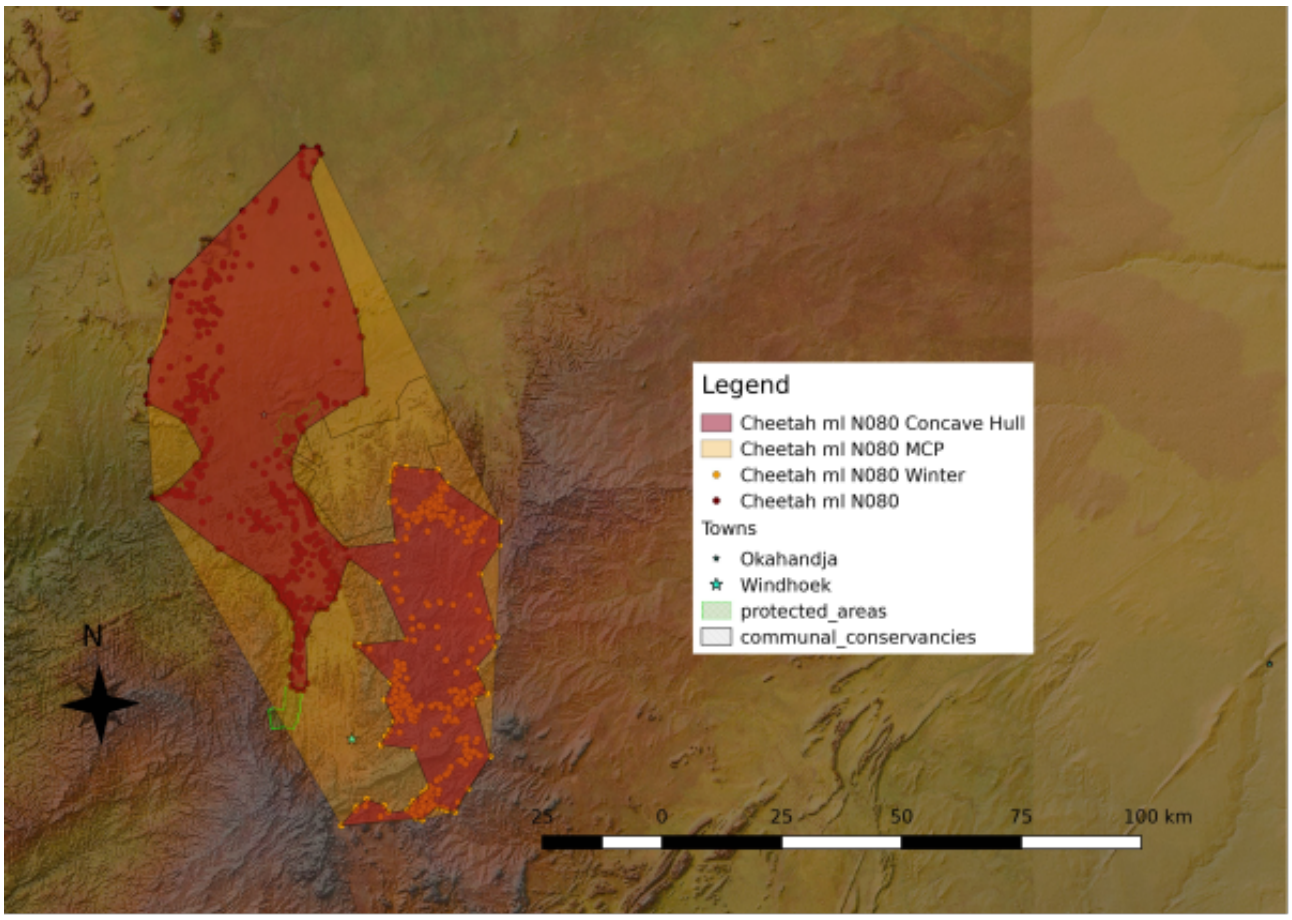


Figure 4.4: *Map of Cheetah male N080 GPS readings (n = 1499), illustrating the difference between the Minimum Convex Polygon (MCP) and Minimum Concave Hull home range estimates.* The MCP home range estimate of 6748.90 km^2 was a third larger than that of the Minimum Concave Hull (4028.47 km^2). Moreover, it failed to even suggest the possible existence of two seasonal home ranges.

moved to the plains North of Windhoek and around Okahandja, before moving back to the mountainous eastern home range on 4 March 2015, where he remained until the GPS collar stopped working on 20 June 2015. The monitoring period was too short to determine if this was a regular seasonal movement or a once-off movement (e.g. if he set up a territory and then lost it again), but only the Concave Hull revealed a home range consisting of two distinctive parts. However, this method was still unable to identify actual "holes" in the home range (e.g. the town of Okahandja in Figure 4.4). Like the Convex Hull (MCP), the Concave Hull did not show how the animals used their home ranges or identified core areas.

Both the Kernel Density Estimates (KDE) and the Local Convex Hull estimates (LoCoH and T-LoCoH) could identify areas of high usage inside the home range. The 95% KDE tended to over-estimate the home range size and in a few cases included areas that were known not to form part of the true home range (Type II errors: e.g. Figure 4.5). One of the claimed advantages of the LoCoH approach is that it is more effective at identifying physical barriers and excluding them from the home range (Getz and Wilmers, 2004; Getz et al., 2007), but in this case (Leopard male N045) the Concave Hull performed better in identifying the river as a physical barrier to the leopard's territory (Figure 4.5). The LoCoH core area (red) showed a better fit to the actual home range usage than the 95% LoCoH. For this leopard (N045 in Figure 4.5) the Concave Hull home range estimate was very similar to the MCP (probably because of relatively few location points), but it followed the actual home range usage much more closely in the south where there were enough location points and correctly excluded most of the area to the south of the river. Both the MCP and Concave Hull methods, using 100% of all points, could possibly include exploratory movements which were correctly excluded by the 95% LoCoH and 95% KDE methods. The 95% LoCoH isopleth consistently underestimated the home range size when compared to the Concave Hull estimate. Consequently, the Concave Hull was considered as the least likely to overestimate or

underestimate home range size and is proposed as the preferred method for home range estimates when the outer boundary of the home range is important (e.g. for calculating overlap between individual home ranges). The T-LoCoH method for determining a-LoCoH gave an estimate that was closer to that of the Concave Hull method than when using the rhr estimate (LoCoH in Table 4.2).

Comparing our results (Table 4.2) with the published home ranges of cheetahs and leopards in Namibia (Table 4.1), it did not appear as if the analysis method alone could be responsible for the wide range in published home range sizes, but that the large variation in home range sizes of cheetahs and leopard on Namibian farmlands is real. When comparing only those published methods used in Namibia for our two focal species, there was no statistically significant difference (observed $F = 0.77$, $p > 0.05$, resampling for F value between groups MCP, 95% MCP, Concave Hull, 95% KDE – 50% KDE was excluded since it estimates only the core home range and are thus expected to be smaller than the other measures).

4.3.3 Home ranges

Only for the Minimum Convex Polygon and only for three animals (male cheetah at 1 907.5 data points, female cheetah at 2 107.8 points, and female leopard at 1 070.4 points) was an asymptote reached in home range estimates. All three of these animals were collared for more than 415 days and with GPS location readings more frequent than 8-hourly (i.e. more than three readings per day).

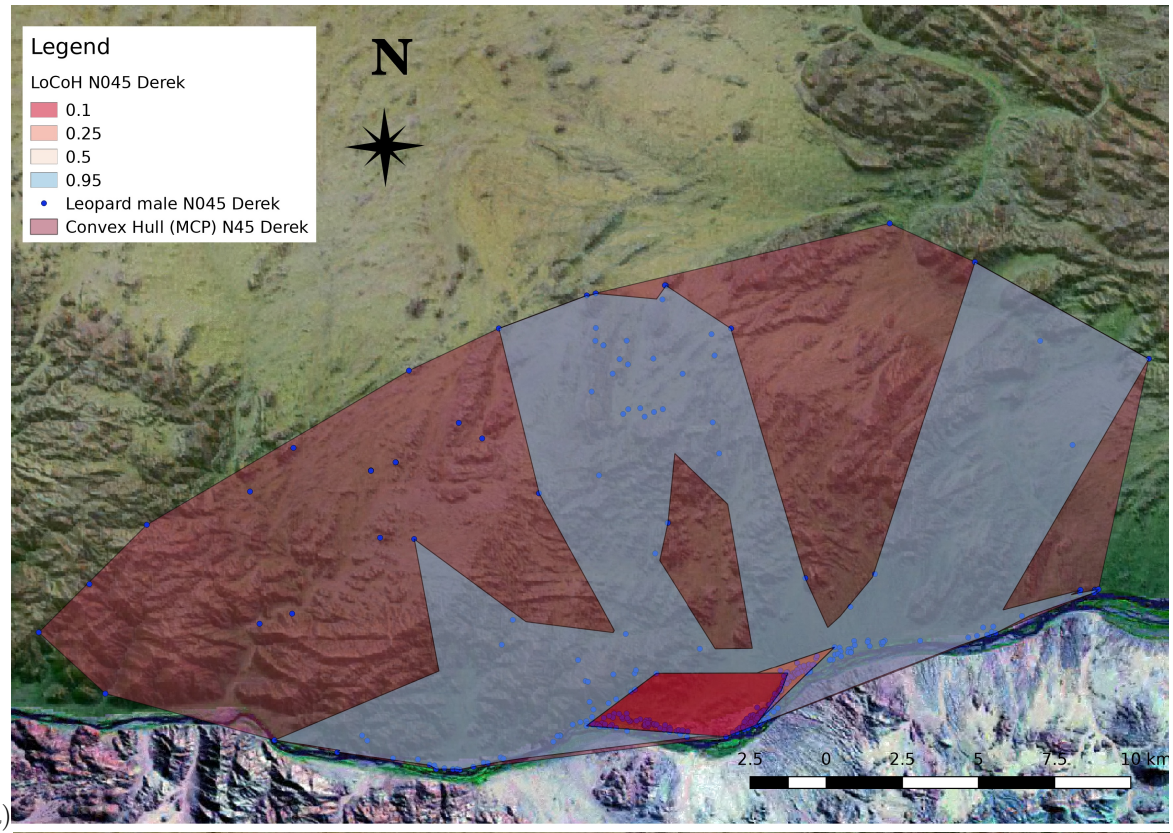
4.3.3.1 Effect of species and sex on home range sizes

Cheetahs (median = 819,47 km^2) in general had significantly larger home ranges than leopards (median = 147,68 km^2)($p < 0.0001$, resampling mean and median difference 2-tailed, Section B.2.0.4 in Appendix B). However, no significant difference was shown between male (median = 1059,95 km^2) and female (median = 290.43 km^2) cheetahs ($p > 0.05$, resampling mean and median difference 2-tailed, Section B.2.0.4 in Appendix B. This is probably because no distinction could be reliably made between floater males and territorial males. No significant difference between the home ranges of male (median = 170.03 km^2) and female (median = 113.29 km^2) leopards could be shown ($p > 0.05$, resampling mean and median difference 2-tailed, SpeciesSex-HR.r). In the present study, this was likely because of the widely different habitats and parts of Namibia in which these animals were released. The study area was divided roughly by half into the semi-arid South and the savannah North, using the central part of the country around Windhoek as a convenient natural boundary. When considering only those leopards ($n = 9$) from the more arid areas receiving less than an average maximum of 300 mm rain[3] (the lower average of Windhoek), males had significantly larger home ranges than females ($p < 0.01$, resampling mean and median difference 1-tailed, Section B.2.0.4 in Appendix B).

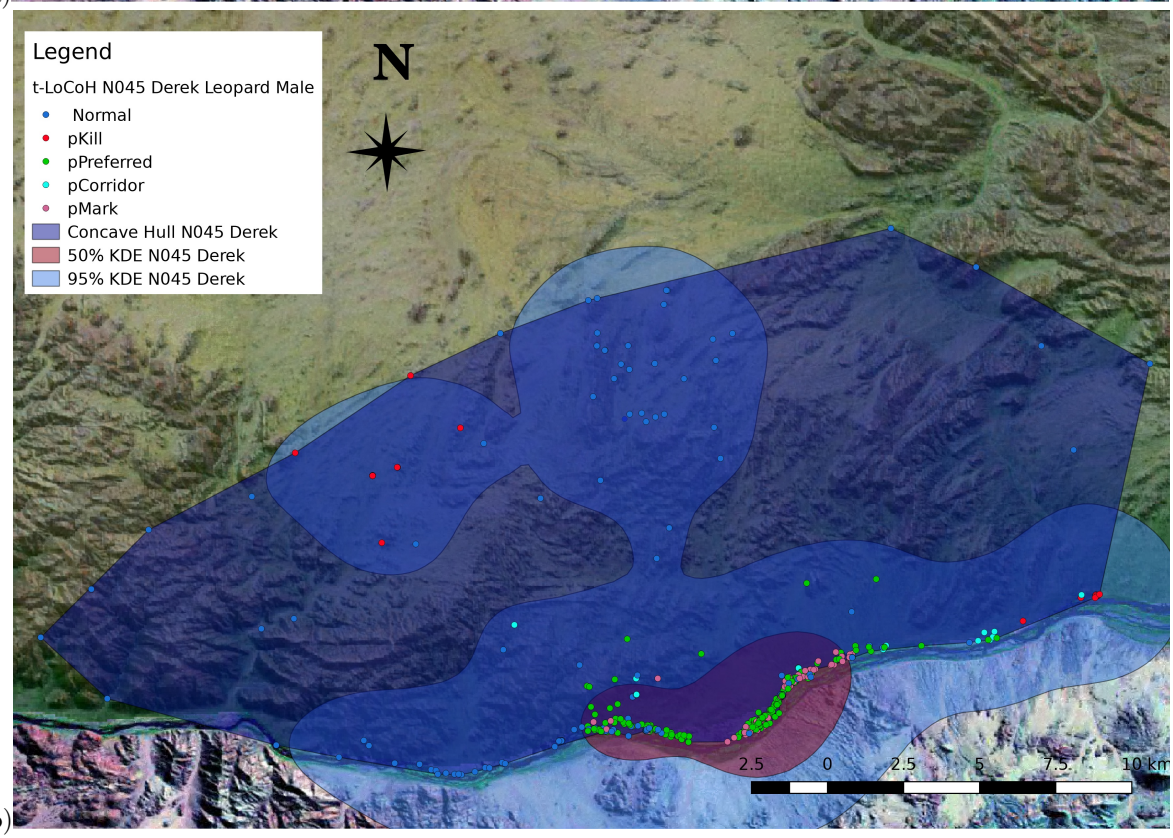
4.3.3.2 Effect of rainfall on home range sizes

Both minimum average rainfall ($r = -0.6684182$, $p < 0.01$) and maximum average rainfall ($r = -0.5942155$, $p < 0.05$) in the area where the animal was released, showed a statistically significant negative correlation to the home range size of male leopards (resampling regression, Section B.2.0.5 in Appendix B), but neither could be shown to have a significant effect on female leopards ($p > 0.05$) (see Marker and Dickman, 2005). Rainfall did have a statistically significant effect on home range size of all leopards combined (min rainfall: $r = -0.518599$, $p < 0.01$, max rainfall: $r = -0.4103061$, $p < 0.05$, resampling regression, Section B.2.0.5 in Appendix B). Rainfall had no significant effect on the home range size of female or male cheetahs ($p > 0.05$), and actually showed a weak positive correlation with home range size for male cheetahs (i.e. an increase in home range size at higher rainfall).

[3]This is about the rainfall area where three successive above-average rainfall years can result in bush encroachment (Rothauge, 2011b). It has been suggested that habitat changes because of bush encroachment in Namibia can influence leopard densities (see Section 2.3 in Chapter 2).



a)



b)

Figure 4.5: *Home range estimates for a leopard male N045 with a known physical boundary (the Orange River) to the South.* a) Both the Convex Hull methods (MCP & LoCoH) included an area to the south of the river which is clearly not part of the leopard home range (not a single GPS point south of the river in spite of a clear preference for the area just north of the river, and leopards are not known for their love of water, making the wide and deep river an effective physical barrier). b) Unlike the Concave Hull, the KDE 95% and 50% estimates both included very large areas south of the river which is a clear boundary of the leopard's home range. (A similar result was seen for female leopard N131 where both KDE estimates included part of the Hardap Dam in her home range).

4.3.3.3 Effect of habitat on home range sizes

Both cheetahs and leopards had their largest home ranges in the Namib Desert Biome, medium sized home ranges in the Nama-Karoo and smallest home ranges in the Tree & Shrub Savannah Biome, but the differences were only statistically significant for leopards ($F = 22.33778$, $p < 0.01$, resampling ANOVA, Section B.2.0.6 in Appendix B). Cheetahs had their smallest home ranges in the Grassy Shrublands, Mixed Tree & Shrub Savannah and Semi-desert & Savannah transition vegetation types (117 – 616 km^2) and their largest home ranges in the Dwarf Shrub Savannah and Southern Namib vegetation types (1182 – 4860 km^2), but the differences were not statistically significant ($F = 0.94$, $p > 0.05$, resampling ANOVA, Section B.2.0.6 in Appendix B). Cheetahs also showed no significant preferences for any specific vegetation structure or when combining biome, vegetation and vegetation structure into a combined habitat type.

Leopards showed a significant effect of vegetation type on home range size ($F = 7.584238$, $p < 0.05$, resampling ANOVA, Section B.2.0.6 in Appendix B). The smallest leopard home ranges were found in the Dwarf Shrub Savannah and Highland Shrubland (21 – 551 km^2), and the largest leopard home range was found in the Swakop and Khan river canyons in the Namib (1068 km^2). Leopards had significantly smaller home ranges in Woodland, Dense and Dwarf Shrubland (21 – 442 km^2) and their largest home ranges in the Ephemeral Riverine Woodland (in the Namib: 1068 km^2), showing a significant effect of vegetation structure on home range size ($F = 11.53196$, $p < 0.05$, resampling ANOVA, Section B.2.0.6 in Appendix B). The combined habitat type (biome, vegetation, vegetation structure) had a significant effect on leopard home range size ($F = 7.584238$, $p < 0.05$, resampling ANOVA, Section B.2.0.6 in Appendix B).

4.3.4 Predictive model of spatial behaviour using T-LoCoH

For each collared animal their GPS positions were classified into 1) probable kills, showing locations that were visited only once, but where the predator stayed around the same area for a long period; 2) preferred locations, which were locations that were most often visited; 3) corridors, those single GPS locations that were far removed from both the previous and next GPS reading; 4) exploratory movement indicating points that were only visited once and where no time was spent; 5) and sometimes marking/denning sites, those locations that were both relatively frequently visited and where the animal spent long periods of time. Because it was unclear why the predator sometimes spent much time in the same regularly visited area, these classifications should be seen as predictive and probabilistic, rather than definite. This model only identified possible large kills (where the predator stayed on the kill for relatively long) and unquestionably missed some kill sites, since it showed an average inter-kill period of 45.4 days for leopards and 52.9 days for cheetahs. Figure 4.5b on page 188 shows an example for leopard male N045 (Derek).

The preferred class were validated by comparing it to the core areas of the LoCoH and KDE 50% Utilization Distributions — in every case more than 80% of the preferred points were inside one of the core areas. Similarly, the exploratory movement class could be checked against known exploratory movement for some animals. For kill site prediction, only one female leopard was checked a posteriori against previous data from the farmer, which cannot be considered as actual validation of the model. The T-LoCoH model identified seven probable kill sites. Of three possible kill sites identified by eye during the daily farmer updates and confirmed by the farmer (a kudu, an oryx and a horse foal), the model had found one. However, the other six probable kill sites identified by the model, had not been investigated.

4.3.5 Habitat preferences and home range use

Home ranges of collared leopards ($n = 33$) included 31 of the 51 combined habitat types (and 51 of the original 95 habitat types). They used five habitat types preferentially for core home ranges and four as marking/denning habitat (Table 4.3).

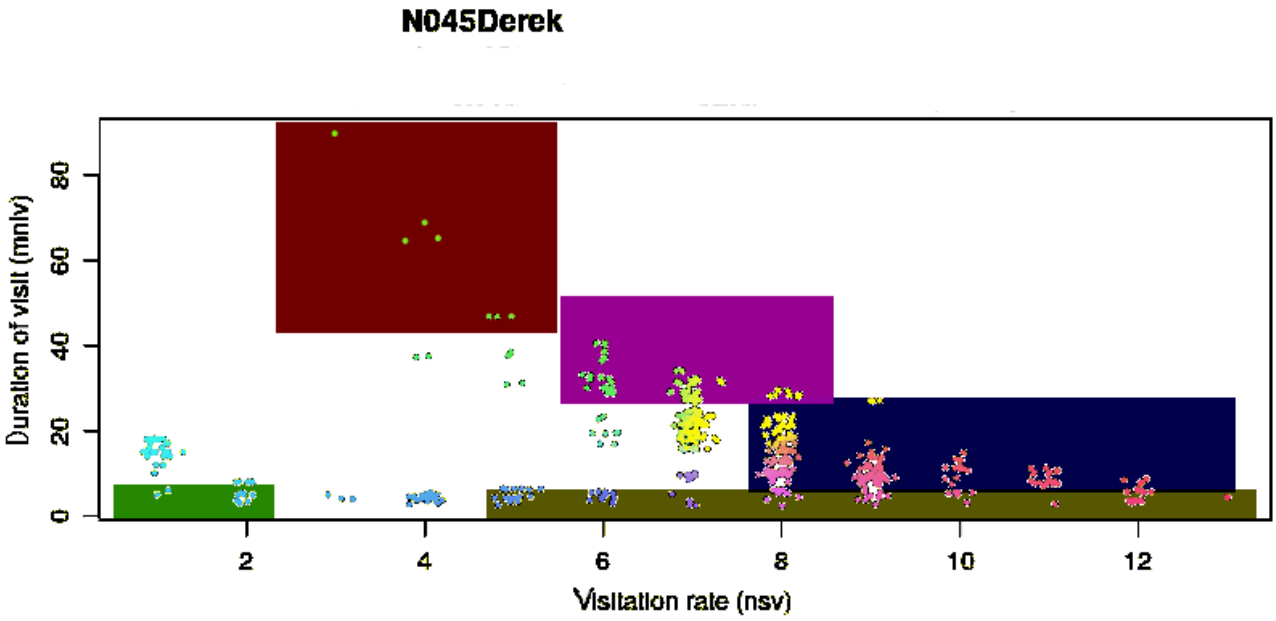


Figure 4.6: *The T-LoCoH graph used for classification of spatial use for male leopard N045.* It plots the number of visits (delimited by an intervisit gap period) against the duration of the visit. Different parts of the graph were then selected as typical for specific behaviour. This is the same leopard as on the map of Figure 4.5. In this example the selected green box to the left denotes areas that were visited only once or twice and for a very short period (typical of exploratory movement) – these points were not indicated on the map. The red block denotes the areas that were very infrequently visited (only once or twice), but where the leopard spent a long time period (it would typically be more to the top left, but shown here as an example). This is typical of kill/feeding sites which are unlikely to happen at the same spot twice, but where the animal will stay for a while after a kill (also red points on the map). The pink block in the top middle would bracket those areas that were visited with medium frequency, but where the animal would spend a relatively long time – typical of denning sites and marking trees (also pink point on the map). The long olive rectangle to the bottom right denotes those areas that were visited frequently, but for a very short duration (typically one GPS point only) — this is typical of frequent movement corridors (shown as cyan points on the map). The blue block to the top right shows those areas where the animal would both frequently visit and for a relatively long duration — this were typical of his core area, shown as green points on the map. This leopard was only collared for 266 days before the collar failed, but was chosen to illustrate the process by which specific spatial behaviour was selected and mapped spatially because of being one of only two leopards with a known physical barrier as a boundary of their home ranges.

Cheetahs (n = 16) were found in 41 of the 51 combined habitat types (and in 76 of the original 95 habitats). They showed no significant preference for any of the 51 combined habitat types as part of their core home range. While this could be simply because they have no preference, when subdividing the the habitat types by rainfall (the original 95 habitat types) two of these habitat types were significantly preferred (resampling "χ^2" proportion use, $p < 0.05$), suggesting that the smaller sample size than the leopards' is the reason for being unable to show stronger preference for certain habitat types (Table 4.3). For their denning/marking areas only two combined habitats were significantly preferred (resampling "χ^2" proportions used, $p < 0.05$), while six different habitat types were preferred when sub-dividing the combined habitats by rainfall (resampling "χ^2" proportion use, $p < 0.05$).

Leopards preferred the same five habitat types that they preferred as core areas of their home ranges, for "hunting" (i.e. clusters where they stayed longer than 24 hours in the same area, but with no regular revisits after leaving – Table 4.3). Cheetahs showed preference for hunting in one combined habitat type (resampling "χ^2" proportion use, $p < 0.05$) and in eight of the original habitat types classified by rainfall (Table 4.3; resampling "χ^2" proportion use, $p < 0.05$).

The only habitat that was preferred by both cheetahs and leopards for core home range areas, were drainage lines in Tree and Shrub Savannah Dense Shrubland (300-400 mm) (resampling "χ^2" proportion use, $p < 0.05$). Drainage and flat plains in Tree and Shrub Savannah Dense Shrubland were also shared as denning / marking habitat. At least two habitats that were preferred by leopards as core habitat (mountains and rocky hills in Tree and Shrub Savannah Dense Shrubland) were avoided (used significantly less than their availability) by cheetahs. Mountains in the Tree and Shrub Savannah Dense Shrubland were also preferred hunting habitat for leopards, but were avoided by cheetahs. The only preferred hunting habitat that was shared as such by both species, was rocky hills in Nama-Karoo Sparse Shrubland (100-200 mm rain) and drainage lines in Tree and Shrub Savannah Dense Shrubland (two out of the five leopard and eight cheetah hunting habitats). The two species thus show both overlap and some specialisation in their preferred habitats.

Table 4.3: *Habitat preferences of two predator species.* The proportion is the number of times the habitat was used for a specific t-LoCoH modelled purpose relative to the number of times that habitat was available to the species. Significant p-values are indicated by *. Of the 95 unique (including rainfall) and 51 combined (excluding rainfall area) habitats available, only statistically significant or near-significant preferred habitats are shown.

		Core		Den / Mark		Hunt	
Species	**Habitat**	**Proportion**	**p-value**	**Proportion**	**p-value**	**Proportion**	**p-value**
Leopard	**Rocky hills in Nama-Karoo Sparse Grassland & Shrubland**	66.67	0.6913	66.67	0.4351	33.33	0.9159
	Flat plains in Nama-Karoo Sparse Shrubland	20.00	0.9151	20.00	0.7854	20.00	0.9194
	Rocky hills Nama-Karoo Sparse Shrubland	70.00	0.0061*	50.00	0.0117*	80.00	0.0015*
	Drainage in Tree & Shrub Savannah Dense Shrubland	57.89	<0.0001*	52.63	<0.0001*	57.89	<0.0001*
	Flat plains in Tree & Shrub Savannah Dense Shrubland	76.92	<0.0001*	61.54	0.0001*	84.62	<0.0001*
	Mountains in Tree & Shrub Savannah Dense Shrubland	75.00	0.0243*	25.00	0.4297	75.00	0.0281*
	Rocky hills in Tree & Shrub Savannah Dense Shrubland	50.00	0.0258*	33.33	0.0487*	75.00	0.0004*
Cheetah	**Drainage in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)**	33.33	0.1039	33.33	0.0443*	50.00	0.041*
	Flat plains in Nama Karoo Sparse Grassland & Shrubland (100-200 mm)	33.33	0.1046	33.33	0.0452*	50.00	0.0431*
	Ecotone mountain-flat in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	40.00	0.1072	40.00	0.05	60.00	0.0454*
	Ecotone rocky-flat in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	60.00	0.016*	40.00	0.0507	80.00	0.0067*
	Rocky hills in Nama Karoo sparse grassland & shrubland (100-200 mm)	14.29	0.4449	28.57	0.0457*	42.86	0.0404*

Species	Habitat	Core Proportion	Core p-value	Den / Mark Proportion	Den / Mark p-value	Hunt Proportion	Hunt p-value
	Drainage in Nama-Karoo Sparse Shrubland (100-200 mm)	33.33	0.1031	16.67	0.3132	50.00	0.0428*
	Rocky hills in Nama-Karoo Sparse Shrubland (100-200 mm)	33.33	0.11	16.67	0.3082	50.00	0.045*
	Drainage in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	60.00	0.016*	60.00	0.0047*	80.00	0.0089*
	Flat plains in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	50.00	0.1031	100.00	0.0006*	50.00	0.204
	Ecotone rocky-flat in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	50.00	0.1084	50.00	0.0482*	25.00	0.5787
	Ecotone rocky-flat in Nama-Karoo Sparse Grassland & Shrubland	50.00	0.0653	33.33	0.138	66.67	0.0571
	Drainage in Nama-Karoo Sparse Shrubland	37.50	0.063	12.50	0.4924	50.00	0.0507
	Flat plains in Nama-Karoo Sparse Shrubland	28.57	0.2402	0.00	>0.9999	57.14	0.0515
	Ecotone rocky-flat in Nama-Karoo Sparse Shrubland	14.29	0.6304	14.29	0.4917	57.14	0.0547
	Rocky hills Nama-Karoo Sparse Shrubland	37.50	0.062	12.50	0.4919	50.00	0.0534
	Drainage in Tree & Shrub Savannah Dense Shrubland	50.00	0.0603	50.00	0.024*	66.67	0.0494*
	Flat plains in Tree & Shrub Savannah Dense Shrubland	40.00	0.1765	80.00	0.0026*	60.00	0.1765
Both	Drainage in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	55.56	0.0072*	44.44	0.0055*	55.56	0.0154*
	Flat plains in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	25.00	0.3501	25.00	0.1786	50.00	0.0569
	Ecotone mountain-flat in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	37.50	0.1195	37.50	0.0381*	50.00	0.0588

Species	Habitat	Core Proportion	Core p-value	Den / Mark Proportion	Den / Mark p-value	Hunt Proportion	Hunt p-value
	Ecotone rocky-flat in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	62.50	0.0077*	37.50	0.0413*	87.50	0.0004*
	Rocky hills in Nama-Karoo Sparse Grassland & Shrubland (100-200 mm)	30.00	0.1248	40.00	0.0061*	40.00	0.0549
	Drainage in Nama-Karoo Sparse Shrubland (100-200 mm)	30.77	0.0341*	15.38	0.1744	46.15	0.0034*
	Rocky hills in Nama-Karoo Sparse Shrubland (100-200 mm)	50.00	0.0002*	28.57	0.0062*	64.29	<0.0001*
	Drainage in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	54.55	<0.0001*	50.00	<0.0001*	63.64	<0.0001*
	Flat plains in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	68.75	<0.0001*	62.50	<0.0001*	75.00	<0.0001*
	Ecotone Rocky-Flat in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	62.50	0.0063*	37.50	0.0394*	50.00	0.058
	Mountains in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	60.00	0.0012*	20.00	0.1776	60.00	0.0028*
	Rocky hills in Tree & Shrub Savannah Dense Shrubland (300-400 mm)	33.33	0.0061*	26.67	0.0069*	60.00	0.0001*
	Drainage in Nama-Karoo sparse grassland & shrubland	55.56	0.0618	44.44	0.0521	55.56	0.1227
	Ecotone rocky-flat in Nama-Karoo Sparse Grassland & Shrubland	55.56	0.0641	33.33	0.1634	77.78	0.015*
	Rocky hills in Nama-Karoo Sparse Grassland & Shrubland	30.00	0.3809	40.00	0.0471*	40.00	0.2716
	Drainage in Nama-Karoo Sparse Shrubland	27.78	0.0624	16.67	0.1544	44.44	0.0045*

Species	Habitat	Core Proportion	Core p-value	Den / Mark Proportion	Den / Mark p-value	Hunt Proportion	Hunt p-value
	Ecotone rocky-flat in Nama-Karoo Sparse Shrubland	30.77	0.1699	23.08	0.1741	46.15	0.0474*
	Rocky hills Nama-Karoo Sparse Shrubland	55.56	<0.0001*	33.33	0.002*	66.67	<0.0001*
	Drainage in Tree & Shrub Savannah Dense Shrubland	56.00	<0.0001*	52.00	<0.0001*	60.00	<0.0001*
	Flat plains in Tree & Shrub Savannah Dense Shrubland	66.67	<0.0001*	66.67	<0.0001*	77.78	<0.0001*
	Ecotone rocky-flat in Tree & Shrub Savannah Dense Shrubland	55.56	0.06	33.33	0.1632	55.56	0.1247
	Mountains in Tree & Shrub Savannah Dense Shrubland	54.55	0.0204*	18.18	0.4232	54.55	0.0491*
	Rocky hills in Tree & Shrub Savannah Dense Shrubland	33.33	0.0204*	22.22	0.0528	61.11	<0.0001*

When managing their livestock to prevent depredation, finding generic "danger zones" of high predator probability or hunting activity could be more important to farmers than species-specific information. For this reason, we also included a habitat preference model using both species pooled. This model identified nine of the original and six of the combined habitat types as being preferred for core areas, nine of the original and four of the combined habitat types as preferred for repeated long visits (denning/marking) and eight of the original and eight (slightly different) of the combined habitat types as preferred hunting habitat (last part of Table 4.3).

4.3.6 Spatial avoidance and interspecific competition

Evidence was found for both interspecific interference competition and co-existence between cheetahs and leopards. At least one cheetah male (out of 16 cheetahs released) was killed by a leopard (one was also killed by hyaena). For leopards on farmlands, intraspecific competition appeared more important than for cheetahs, with at least two (possibly three, out of 35 released) being killed by other leopards. For both species human-caused death was apparently more important than interference competition (of 35 leopard releases 10 died, seven caused by humans; of 16 cheetah releases, seven died of which four deaths were caused by humans).

However, there was also evidence for extensive simultaneous overlap in home ranges between the two species. Five leopards had an average of 39.1 km^2 (18.3 - 87.8 km^2) of their home ranges overlapping with cheetahs, an average of 24% (SD = ±20%) of their total home range sizes. These leopards overlapped with six cheetahs with an average of 33.7 km^2 (6.7 - 87.8 km^2) per cheetah home range (15.7 ± 22.8 % of the cheetahs' average home range sizes). There was inadequate data to determine intraspecific overlap in leopard home ranges, but data showed that seven cheetah males' home ranges overlapped with other cheetahs for an average of 393.2 km^2 per cheetah (55.2 - 901.4 km^2) (39.9 ± 23.9% of their average home range sizes). If leopards are actively excluding cheetahs from some areas, one would expect a decrease in interspecific overlap (Rosenzweig, 1981). The degree of interspecific overlap between leopards and cheetahs should then be less than the intraspecific overlap between individual non-territorial cheetahs (no exclusion) and more similar to that between individual

territorial leopards (exclusion). The probability that the observed difference between the 39.9% and 15.7% was caused by random chance was > 0.05 (resampling mean difference 1-tailed, Section B.2.0.4 in Appendix B), so one could not reject the null hypothesis of no difference between interspecific and intraspecific home range overlap. There is thus little evidence from the data that leopards are having a real effect on cheetah distribution. A change in their relative abundance on Namibian farmlands, is thus more likely to be the result of habitat differences than direct interference competition.

The data provided some indirect evidence for an increase in leopard numbers relative to cheetah numbers on Namibian farmlands. This increase in the numbers of captured leopards relative to cheetahs does not necessarily indicate an increase in leopard densities or a decrease in cheetah densities, even though there is a positive correlation between conflict (livestock losses) and predator observations (see Chapter 3). Almost all of the collared predators were captured as a result of conflict with farmers. There was a significant positive correlation between the year and the relative number of leopards to cheetahs ($r = 0.4711032$, $p < 0.001$, resampled regression 1-tailed, Section B.2.0.5 in Appendix B). From 2011 to 2014 most conflict reporting calls involved cheetahs (2014 equal numbers of leopards and cheetahs), but since 2014 most conflict calls have involved leopards.

As no overlap data was available for male and female leopards respectively, we could not use our formula

$$density = n/area = ((TotArea + TotOverlapArea)/AverageHomeRangeSize)/TotArea$$

to calculate leopard densities. For cheetahs, the data from east of Windhoek (n=4) gave a density estimate of 0.0825 / 100 km^2 for male cheetahs and the data from next to the Namib (n=2) 0.039 / 100 km^2. Using the unrealistic assumption that females have similar home range sizes and similar overlaps in home range sizes and a 50/50 sex ratio, this gave an estimate of 0.165 and 0.078 resident cheetahs per 100 km^2 respectively.

4.4 Discussion

4.4.1 Translocation vs. On-site collar and release

Translocation of predators following conflict with farmers was no more successful than releasing it back on the same farm with a GPS collar and updating farmers on the movement of the predator. It should be emphasized that success here was only measured at the site where the collared animal was released. In the case of translocation, there is also the question about how successful the translocation was in decreasing livestock losses at the site from where the animal was removed (*cf.* Weise et al., 2015a,b who looked at what happened at both locations for translocated animals). Unless the removed animal was a habitual livestock killer (i.e. killing livestock on more than three occasions – Stander, 1990), translocation will probably have little long-term effect on predation of livestock (Linnell et al., 1999). This was shown in our data by leopard male N062 taking part of his previous neighbour's territory within less than a month after the neighbouring male was shot. It was also seen that less than 15% of the captured "problem" predators showed any evidence of being habitual livestock killers. Therefore, the high "success" rate of p(success) = 0.78 found from this data for translocation, account for only half of the actual success. The previous prior success probability as determined for Namibia from Weise et al. (2014) in Chapter 2 is therefore considered more accurate and should be used when evaluating this method — p(success) = 0.45. It should be mentioned that some past authors have similarly to this present project, not considered what happens in the areas from where the animal is removed, by measuring success only at the release site of translocation (Linnell et al., 1997; Hayward et al., 2007a; Weilenmann et al., 2010; Ropiquet et al., 2015). The most important issue at the release site is that the animals should not return to the home range from which it was removed (Linnell et al., 1996). In our data there was at least one example of this happening (Figure 4.3a). Another possible issue is territorial individuals that are already at the release site, resulting in intra-specific conflict and sexually selected infanticide if the translocated individual wins or alternatively, being killed or driven out into an area of renewed conflict with farmers (Weilenmann et al., 2010). Additionally,

translocation can dilute the genetic adaptation of the sub-population to which it is moved (Ropiquet et al., 2015). In Section 2.2.3 of Chapter 2 the advantages and disadvantages of translocation is discussed in more details as applicable to Namibian farmers.

When releasing the predator at its capture site with a GPS collar, farmers could physically go to any kill site and find and identify prey remains. Thus they were able to confirm for themselves that livestock is not the preferred prey of leopards and cheetahs and the occasional habitual livestock killers could be identified with certainty (Marker et al., 2003b; Stein, 2008; Stander, 1990; Linnell et al., 1999). Additionally, because the home ranges of both predators were much larger than a single farm and they thus spent only part of their time on any particular farm, some farmers used the predator movement data to manage their livestock – moving livestock closer to the homestead and protecting them when the predator approached the farm, or even moving the livestock to a different part of the farm than the area occupied at the time by the collared predator. Both the goodwill generated and the improved livestock management probably contributed to the prior success probability of this method of 0.64.

The significant differences in home ranges (when including initial exploratory movements), even when restricted to only 95% of the points in KDE, between translocated and on-site released animals introduces another possible explanation for the published variation in home range sizes. In some previous home range studies in Namibia (e.g. Marker et al., 2008) almost half of the animals used had been translocated. Other home range studies, almost exclusively released animals on or near the same site where they had been captured (Melzheimer et al., 2018). This could very well explain the observed differences in home range estimates between these studies (Table 4.1). This re-emphasizes that caution should be used when using translocated animals in home range studies. It can be expected that translocated animals will be more likely to exhibit exploratory movement after their release and care should be taken to exclude these explorations from any home range analysis or estimations. Such exploratory movements are also more likely to result in the predator coming into contact with livestock, often resulting in conflict with farmers (Weise et al., 2015a,b). This is one important reason why translocation has been advocated as a last resort only (Weilenmann et al., 2010; Weise et al., 2014, 2015a,b), and our data support this emerging consensus.

4.4.2 Evaluating home range estimation methods

Although home range estimation methods significantly affect the resulting estimate, the data showed that the variation in home ranges reported in the literature for the two focal species is probably the result of real variation, rather than being an artefact of different home range estimation methods (Table 4.1 & Table 4.2). Although no direct statistical test was possible using the data from the literature, this would follow from the fact that in our own data there was no significant difference when comparing the methods used in the published Namibian studies and a similarly large range in home range sizes in our data. The home range sizes measured for leopards ranged from 21.43 km^2 to 1068.71 km^2 in our data (using a single method to estimate it). In the literature, it ranged between 40 km^2 and 1136.5 km^2 (using different methods). Similarly, our cheetah home range sizes ranged from 117.09 km^2 to 4860.85 km^2 (using only Concave Hull estimates). In the literature it ranged from 217 km^2 to 6353 km^2 using different estimation methods.

One exception where the analysis method could make a significant difference, is in home range estimates for male cheetahs. In our data the male home ranges could be classified into two distinct groups (associated with floaters: 2009.13 $\pm$ 1123.75 km^2 and territorial males: 465.70 $\pm$ 209.04 km^2) based on size alone (which were significantly different: one tailed resampling of means, $p < 0.05$). However, a male coalition, including the collared six to seven-year old cheetah (N066), that returned to a previous home range after translocation of 70 km, had a home range of 1501.64 km^2 (95% MCP). The fact that it was a coalition, with individuals old enough to have confidence for setting up a territory and the fact that they returned to their previous home range, all seemed to indicate that it was likely that this coalition would be territorial. However, the home range

size was more similar to that of a floater (Melzheimer et al., 2018). Any classification of cheetah males into territorial residents and floaters based only on home range size, is questionable. In our case "correcting" the data so that this coalition would be part of the territorial group, did not have a significant effect (they were still significantly different – floaters: 2168.74 ± 1177,91 km^2 and territorial males: 589.93 ± 357,15 km^2). However, unless behavioural data (e.g. territorial marking and aggression) is observed and also used for classification of cheetah males, there may be other unknown individuals that were also incorrectly classified and this would actually make a significant difference if correctly classified. The data from Marker et al. (2008) showed that there was no significant difference between the home range sizes of male cheetah coalitions and single males. Melzheimer et al. (2018) explained this observation by showing that not all male coalitions are territorial and on the other hand, that some single males are territorial as well. The importance of a good understanding of the spatial ecology of the predator species is illustrated by this, since it would have been easy to conclude from the Marker et al. (2008) study that Namibian male cheetahs did not show the typical dual strategy of being either floaters or territorial, seen previously in East Africa (Caro and Collins, 1985, 1986, 1987). When using home ranges of cheetahs in farmer-predator conflict management, this dual strategy needs to be taken into account — e.g. territorial males will be more frequently on the same farms than floaters, who will generally cover more farms and therefore spend less time on any single farm (thus enabling temporal avoidance strategies by livestock farmers), but territorial males are more likely to restrict their movements to only a certain portion of any specific farm (thus enabling spatial avoidance strategies).

4.4.3 Home range sizes of Namibian leopards and cheetahs

Except for three cases, none of our home range estimates reached an asymptote as recommended by Laver and Kelly (2008). Possible reasons include the obvious cases where the collared animal died or the collar failed prematurely. There is also the possibility that occasional exploratory movements were sometimes not identified or, in at least one case (leopard male N062), where a territorial animal expanded its territory to include part of the territory previously belonging to a neighbouring animal that had been killed. For male cheetahs, a switch between a floater strategy and setting up a territory might be another reason for home range estimates not reaching an asymptote (Melzheimer et al., 2018).

Returning to the applied questions on the spatial behaviour of leopards and cheetahs as it relates to human-wildlife conflict on Namibian farmlands, it was found that both species have large home ranges, with a cheetah home range of 1311.87 km^2 on average covering 14.04 farms and a leopard home range of 209.16 km^2 covering 2.24 farms ranging up to 4860.85 km^2 (52 farms) for a cheetah and 1068.71 km^2 (11 farms) for a leopard home range (Table 4.2 above & Table 3.1 in Chapter 3). This has many important implications for the conservation of the two species on farmlands in the majority of their range in Namibia — 1) farmers often overestimate the densities of predators because of these large home ranges (Marker et al., 2008); 2) a single farmer that kills predators indiscriminately has a larger impact on the total predator population than would be expected (Marker et al., 2003a); 3) the predators do not spend all of their time on a single farm and if their movements are known, livestock can be managed accordingly (as mentioned above).

4.4.3.1 Effect of species and sex on home range sizes

Cheetahs generally had larger home ranges than leopards (Hayward et al., 2007b). This implies that cheetahs are more vulnerable to being killed (moving over more farms) and their densities are also more likely to be over-estimated by farmers compared to leopards. The data showed no significant differences in home range sizes between cheetah males and females (Marker et al., 2008). This was most likely because of the different spatial strategies of male cheetahs and our inability to identify which strategy most males were following from the collar data alone (*cf.* Melzheimer et al., 2018; Chapter 2).

In agreement with Marker and Dickman (2005), but contra Stander et al. (1997) and Stein et al. (2011), there was no significant difference between the home ranges of male and female leopards when looking at the country as a whole. This was also true for leopards in the more mesic Savannah Biome. However, in drier areas below a maximum average rainfall of 300 mm, male leopards had significantly larger home ranges than females, as is more typical for the species in general (Estes, 1991; Bothma et al., 1997; Stander et al., 1997; Martins, 2010; Stein et al., 2011). Fattebert et al. (2016) showed that contrary to expectations, female leopards often have larger home ranges than required for them to raise their cubs. López-Bao and Rodrigues (2014) had similarly shown that for Iberian Lynx (*Lynx pardinus*) home range sizes were not determined by food availability and that most individuals had home ranges that were larger than required for providing food alone. As their population grows female leopards would cede part of their home ranges to their female offspring (Balme et al., 2012a; Fattebert et al., 2016). However, male leopard home ranges would not decrease and young males had to disperse and find their own home ranges (or die during the process – Stander et al., 1997). The population studied by Fattebert et al. (2016) was growing and recovering after a period of heavy hunting; it is possible that these relatively larger female home ranges in the northern parts of Namibia might similarly indicate a recovering or growing population, in agreement with farmer perceptions (Chapter 3) and the increasing farmer-leopard conflict reported here (page 195).

4.4.3.2 Effect of rainfall on home range sizes

Cheetah home range sizes did not change significantly with rainfall. However, leopard males had smaller home ranges and higher densities as rainfall increases (>300 mm average rainfall/annum) while this effect was much weaker and insignificant for female leopard home range sizes. As leopard home ranges became smaller with increasing rainfall, the difference between males and females also became less pronounced. This partly explains the lack of significant differences between male and female leopard home range sizes in high-rainfall areas. This result is consistent with the hypothesis by Marker and Dickman (2005), that rainfall acts as a proxy for prey density which in turn determines leopard home range sizes (see previous section).

In general, the larger home range sizes of male leopards have been considered as a reproductive strategy (the larger the territory, the more females are available for mating), while females likely have a minimum territory size requirement for raising cubs successfully dependent on available cover and prey (Stander et al., 1997; but see Fattebert et al., 2016). Their home range use patterns also differ, with females more likely to criss-cross their territories and males more likely to travel in straight lines patrolling the borders of their territory (Martins, 2010; Fröhlich, 2011). It can be hypothesised that the reason for the decrease in male home range sizes with higher rainfall is caused by taller and denser vegetation in high rainfall areas, making it more difficult to patrol and defend larger territories, rather than prey density alone. This would fit the viewpoint of male leopard territory size as primarily a reproductive strategy (Fattebert et al., 2016). Thus in the northern higher-rainfall areas with generally higher prey density and biodiversity (see Chapter 3), leopard males exert top-down control and are density limited through intra-specific competition alone, while in the less biodiverse South they could be bottom-up controlled by prey availability (see Sinclair et al., 2003) — however, this does not explain why the decrease in female home range sizes are so much smaller. Alternatively, the decreasing male home range sizes may indeed be reacting to higher prey densities as suggested by Marker and Dickman (2005) and females are the ones not reacting to prey densities and keeping similar home range sizes in high rainfall areas than in more arid areas in agreement with Fattebert et al. (2016). A similar spatial pattern was seen in Iberian Lynx (*Lynx pardinus*) where females did not decrease their home range sizes to increasing densities in periods of high prey availability (López-Bao and Rodrigues, 2014). When looking at this result dynamically, it should also be kept in mind that through bush encroachment in Northern Namibia (>300 mm rainfall/annum), the habitat is becoming more suited to leopards (De Klerk, 2004; see below). This makes the "recovering population" scenario more likely, with females having home ranges that are larger than required for raising their cubs, but which decrease over time in size as their daughters take parts of the home ranges. This female family cluster forming

behaviour as explained by Fattebert et al. (2016), also explains the observation by Estes (1991) that female leopards often have overlapping home ranges. However, neither hypothesis can be confirmed from our data and research over time of leopards in the same area would be required in order to confirm if female leopards show matrilineal kin clusters with higher home range overlap between related females and if males keep stable home range sizes over time as would be typical for the recovering population scenario described by Fattebert et al. (2016). If male leopards are density-dependent, their home range sizes should become smaller over time when they settle in a new area (or recover from heavy persecution) and "pack" the landscape until they become resource limited, while females should have stable home range sizes over time (López-Bao and Rodrigues, 2014; Fattebert et al., 2016).

For cheetahs, the lack of higher densities at higher rainfall areas could be explained by either the effect of intra-specific competition resulting in more floaters in high-rainfall areas or by inter-specific competition with leopards, resulting in cheetahs requiring larger home ranges to avoid leopards and still find enough prey. This would fit with Scantlebury et al. (2014), who showed that for cheetahs most of their energy was spent finding prey rather than in the actual hunt. More likely than avoiding leopards, the more open vegetation in lower rainfall areas may simply be better suited habitat for cheetahs (Muntifering et al., 2006; Nghikembua et al., 2016).

4.4.3.3 Effect of habitat on home range sizes

Habitat had no significant effect on cheetah home range size, while leopards have significantly larger or smaller home ranges in different habitat types. In general, leopard home ranges were smaller in higher-rainfall habitats (within the Tree & Shrub Savannah Biome) and largest in low-rainfall habitats (in the Namib Desert Biome). For biomes (but not for vegetation types or vegetation structure), leopards and cheetahs showed a similar pattern of preference and the lack in significance for cheetahs could be due to a small sample size (see also Muntifering et al., 2006; Lindsey et al., 2011; Swanepoel et al., 2013; Nghikembua et al., 2016).

4.4.4 Predictive model of spatial behaviour

While the predictive kill site T-LoCoH model, probably missed many kill sites, it is relatively unlikely to have false positives. There are few other reasons for a predator to spend a long time around the same point and not revisit it again, unless it had made a kill. One of the major reasons for missing kill sites, is that currently all kills made within the preferred areas or core areas of the home range are not identified as kill sites, but rather as "preferred habitat". To my knowledge, this was the first use of T-LoCoH for evaluating predator behaviour (it has been used for herbivores before – Lyons et al., 2013). It is important to realize that unless the model is followed by using another method like spoor tracking to confirm the actual behaviour, currently it only uses the time spent by predators at certain sites and their spatio-temporal movement patterns to predict probable behaviour.

This model presents a relatively quick and easy predictive model to identify probable kill sites using the T-LoCoH package in R. If kills are confirmed by follow-up visits or farmer reports, this can provide useful prey preference data in addition to other methods like scat analysis and also at a quicker rate (e.g. Foster et al., 2010 recommend a minimum sample size of 70 to 100 scats for diet analysis of pumas and jaguars). This method could potentially identify possible conflict hotspots and kill sites for identifying prey usage, much quicker.

There was not enough data available for a true ground-truthing of the model, since most farmers did not record the results of their kill site checks and showing possible kill sites to farmers was mostly used as a conflict mitigation method. Still, one of seven predicted kill sites was confirmed, while two known kills were missed (14% confirmed). It should be clarified that the farmers were not sent data from this model, but clusters that were identified visually during the daily map update. Therefore, it is possible that not all or even most of the

seven predicted kill sites were visited for confirmation. Anderson and Lindzey (2003) found 13 confirmed kills from 38 predicted potential kill sites for cougars in their study (34% confirmed), but their model predicted a mean of a kill every seven days, which is more realistic than this study's T-LoCoH model. The method used by Fröhlich (2011) (see also Martins et al., 2011) identified 78 potential kill sites of which 31 was confirmed (∼8 days per kill, 40% confirmed). Both of these studies were done at single study sites and use much more complicated and detailed calculations (and more frequent GPS readings). Our model could be improved by also including locations that had been revisited (from the preferred/core areas), but then runs the risk of more false positives. As it is, it is better suited for quickly finding the most likely potential kill sites where GPS data is more widely spaced in time. Like other methods to determine prey use by using GPS clusters, this method is unable to identify most of the smaller prey items (Pitman et al., 2013b; Jansen, 2016).

Typically, most models that find and predict conflict hotspots, use actual spatial data on livestock losses (or other conflict resources), to build the model (Miller, 2015). Then more conflict data is used to validate it afterwards. The current T-LoCoH model avoided the need to use livestock loss data to *build* the model, but it cannot be *validated* without this data (which were not available). The novelty of using the T-LoCoH model compared to past cluster analysis models however, lies in the fact that it did not only model probable kill sites (classified as "hunting habitat"), but also areas of general high usage (classified as "core habitat") and regular resting sites (classified as "den/marking tree" sites). In terms of managing human-wildlife conflict, it means that all these habitat types should be considered as high risk for livestock depredation.

4.4.5 Home range use

While the size of their home ranges can be used to give some indication of habitat preference (smaller home ranges = higher densities = preferred habitat), it does not tell us anything about how predators use their home ranges (Fretwell and Lucas, 1969; Luyt, 2004).

Some evidence of possible niche separation between leopards and cheetahs was shown by cheetahs evincing a statistically significant avoidance of rocky hills and mountains in dense shrublands, which is significantly preferred by leopards, also for hunting (Bartnick et al., 2013). On the other hand, both species preferred riverbeds and drainage lines in dense shrublands. The reasons could be either because of higher prey encounter rates (kudu is a major prey source of both species and are often found along riverbeds with good browsing and hiding possibilities – Estes, 1991) or simply because the riverbeds provide easy movement paths through the thick bush (McVittie, 1979; Marker et al., 2003b; Stein, 2008). Although the majority of cheetahs in Namibia are found in the Tree and Shrub Savannah Biome, and they are well adapted to hunting in the bush, the flat plains in the Nama-Karoo Sparse Grassland and Shrubland was a preferred cheetah habitat, avoided by leopards (Bissett and Bernard, 2007; Nghikembua et al., 2016; Weise et al., 2017).

The two species also differed not only in their preferred habitat, but also in their preferred use of habitat. Mountains in dense shrubland, preferred hunting habitat of leopards, were avoided by cheetahs for hunting. This makes sense when considering the hunting methods of the two species (Estes, 1991). It is difficult for cheetahs to use their speed in order to catch prey on uneven mountain slopes in shrublands (Hildebrand, 1961; Sharp, 1997; Kaplan, 2013; Wilson et al., 2013). On the other hand, mountains in dense shrubland provide plenty of hiding opportunities for a stealth hunter like a leopard to stalk or ambush prey (Balme et al., 2007; Pitman et al., 2013a; Minnie et al., 2015). However, both species preferred to hunt along the rocky hills of the Nama-Karoo Sparse Shrubland and along drainage lines in the dense shrublands of the North. In the open shrublands of the Nama Karoo, the rocky hills will provide cheetahs with some cover, enabling them to get close to their prey before using their speed to capture prey on the adjacent open plains. These hills would therefore play a similar role in cheetah hunting than the edges of grassy clearings in Northern Namibia as described by Nghikembua et al. (2016). For both leopards and cheetahs, rocky hills in the open shrublands also provide an elevated position from which to spot potential prey. Given that finding prey is more energetically demanding

for cheetahs than the actual hunt, this is an important role of these rocky hills (Wilson et al., 2013; Scantlebury et al., 2014). For leopards, these rocky hills will provide better cover for stealth hunting than the open, flat plains around them. The preference for drainage lines and river beds in the dense shrublands could either be because of easier movement, more preferred prey (kudu) or also because typically larger trees rather than the dense medium-sized Acacia bush are found in these habitats — medium density woody vegetation has been identified as preferred leopard hunting habitat elsewhere (Balme et al., 2007). Balme et al. (2007) explained this preference in leopards as choosing vegetation that provide enough cover for stalking, but also open enough to enable early detection of prey and without obstructing vegetation during the chase phase of the hunt. These same factors might be important for cheetahs as well, who showed higher success rate when hunting in dense cover than in the open (Wilson et al., 2013). These are therefore habitat types that can be considered as potential conflict hotspots and should be avoided by vulnerable livestock (Carvalho et al., 2015; Minnie et al., 2015; Miller et al., 2015; Miller, 2015; Miller et al., 2016).

This analysis presents a more complicated picture of the habitat preferences of the two species than commonly accepted: Cheetahs may not always prefer open Savannah habitat as had been presumed before (Mills et al., 2004; Muntifering et al., 2006; Bissett and Bernard, 2007). Leopards may not always prefer dense vegetation, as presumed (Balme et al., 2007). There were some shared preferred habitat types, as well as some habitats that were strongly preferred by one species, that were avoided by the other species. Cheetahs have a few habitat types that approach statistical significance as core and hunting areas, which might actually be preferred habitat if analysed using a larger sample size. As it stands, leopards show a wider range of preferred habitat than cheetahs for their core area. While previous studies have focussed on the habitat preferences of a single predator species, often in a single part of Namibia, this study enabled us to compare the habitat preferences over most of the habitat types available on Namibia's commercial farmlands and to compare the two species to each other. In contrast to what would be expected from their dissimilar hunting strategies, leopards and cheetahs share some preferred habitat types in Namibia.

4.4.6 Spatial avoidance and interspecific competition

Except for the habitat preference differences between the species, there was little evidence that direct interference competition (leopards killing cheetahs) prevents co-existence and overlapping home ranges between the two species. It is known that leopards kill cheetahs (Estes, 1991), and that kleptoparasitism by leopards contribute to cheetah prey losses (Hayward et al., 2006b,a; Hunter et al., 2007). However, it is possible that coexistence on Namibian farms occur through temporal avoidance (Hayward and Slotow, 2009).

The farmers conflict data from N/a'an ku sê supports the claim that there has been an increase in leopard numbers relative to cheetahs on Namibian farmlands — unlike the survey data of Chapter 3, this data was not simply based on farmers' perceptions, but on the number of predators actually caught in cage traps as a result of the conflict (and then translocated or released on-site). While there could be other reasons for the observed change in the number of conflict calls for each predator species, the simplest explanation is that the numbers of leopards on Namibian farmlands have increased relative to the numbers of cheetahs. If direct interference competition is not responsible for this relative increase in leopard numbers, it might be the result of bush encroachment and differences in habitat preferences between the two species (Estes, 1991; Boshoff and Kerley, 2001; Bissett and Bernard, 2007; Cristescu et al., 2013; Swanepoel et al., 2013). This illustrates the importance of considering the farm as a full agroecosystem (Chapter 1, Figure 1.4) — veld management is largely responsible for the increasing bush encroachment, which in turns influence the predation patterns of livestock (De Klerk, 2004; McGranahan, 2008; Rothauge, 2011b).

A formula was presented for estimating of predator densities from GPS collar data, if enough animals in an arbitrary study site are collared. However, if this density estimate is calculated from a limited number of individuals it should not be considered as reliable or accurate. Preferably all the individuals in that site (of

whom the home ranges overlap) needs to be collared (Balme et al., 2009). If *all* individuals in a certain area have been collared, excluding non-resident nomads, it can be considered as the best density estimate possible for that species *in that area* (Balme et al., 2009). Only if this is done in a number of different study sites showing similar densities, can the resulting density estimate be extrapolated to the whole area between study sites (Balme et al., 2009; Weise et al., 2017; Durant et al., 2017; Figure 4.2). Furthermore, by focussing on adults only, using GPS collars to estimate home range sizes using this formula will give a better estimate of the breeding population (effective population size), but sub-adult and juvenile numbers will remain unknown (Lande, 1995; Lynch and Lande, 1998). Global Positioning System (GPS) collars cannot be used safely for growing sub-adult and juvenile individuals because of the danger of choking. Using this method on a group of male cheetahs near Windhoek, resulted in a density estimate (0.165 / 100 km^2) that was lower than the 0.20 / 100 km^2 of Weise et al. (2017). However, no overlapping female cheetahs were collared (they were assumed to occur at the same density as the collared males for this density estimate), nor is it certain that all or most males were included (*cf.* Melzheimer et al., 2018). The estimate by Weise et al. (2017) was largely an extrapolation from studies done elsewhere and could over-estimate the density of cheetahs. The total number of cheetahs in Namibia might therefore be even lower than the estimate by Weise et al. (2017) and their plea for the conservation status of cheetahs to be updated to "endangered" is supported here.

4.5 Conclusions

With the majority of Namibian cheetahs and leopards found on farmlands, outside protected areas, their long-term conservation is dependent on the sustainable management of farmland ecosystems. The spatial behaviour of predators is an important part of ecologically sustainable management. Using data from leopards and cheetahs that were collared with GPS collars as a result of human-wildlife conflict on farmlands, we investigated their spatial behaviour in widely-distributed locations throughout Namibian farmlands.

The plethora of different methods used to estimate home ranges of the two predator species in Namibia and the lack of justification for the method choices, led to an in-depth investigation of the literature on home range estimation methods. From the literature a number of themes could be identified:

- There is a need for consistency in home range studies, and the methods used should be described in enough detail to be replicable and comparable across studies (Laver and Kelly, 2008).
- Older measures of home range like MPC are becoming increasingly inappropriate for analysing high-frequency and accurate GPS data (Kie et al., 2010).
- One method is seldom adequate to define all the relevant aspects of spatial ecology and home range size, and different home range estimates should be used to answer different questions (Gregory, 2017). For some questions the outer boundaries of home ranges are important (e.g. when looking at territorial behaviour) while for other questions the utilization distribution (use of the habitat within the home range) is more important. Reporting multiple home range estimates also enables better comparison between studies.
- Newer methods include using the direction of travel of the animal as well as including time as another factor (e.g. T-LoCoH) that describe more of the spatial behaviour than older utilization distributions or home range estimates (Benhamou and Cornélis, 2010; Steiniger and Hunter, 2013; Lyons, 2014; Lyons et al., 2013).

Other aspects of the the methodology used to estimate home range sizes, had a significant effect:

- When comparing home range sizes determined by the different methods used (using resampling analysis of variance) they gave significantly different estimates ($p < 0.05$).

- Most importantly, using an arbitrary 95% cut-off isopleth (Kenward et al., 2001) to exclude exploratory movements from the home range estimate, was seen to still include most exploratory movements and thus be significantly larger than those home ranges where exploratory movement were manually excluded (one-tailed resampling of means, $p < 0.5$ – see Section 4.2.3 above). Visual identification and exclusion of exploratory movement using our new "rule of no revisit" (i.e. that the animal has not returned to the same general area within a year's period) was more accurate in excluding exploratory movements as required by the very definition of "home range" (Burt, 1943). It is therefore proposed as the method of choice if there are enough data points to ensure that all general areas within the home range that was visited by the animal would be recorded (the exact numbers of points required will depend on the species, the size of its home range and how far it moves every day).

While the present study used simple visual identification for excluding exploratory movement (see Figure 4.3), defining the size of the "same general area" (either as specific units of area, in terms of average daily movement of the species, or as a specific percentage of the total area of the GPS data points) would enable automatic exclusion of such exploratory movements. However, it is uncertain if this is really a worthwhile gain over manual inspection, given the fact that the number of GPS collared animals are usually small due to costs (Hebblewhite and Haydon, 2010; Tomkiewicz et al., 2010).

- With more data points, the Minimum Concave Hull method can overcome many of the problems of the Minimum Convex Polygon (MCP) method for obtaining a 100% outline of the home range, once exploratory movements have been excluded. It also performed better[4] in at least some cases than the LoCoH method by correctly excluding areas that were known to be outside the home range (Figure 4.5 in Section 4.3.2.2). Its main drawbacks are that it is probably prone to Type I errors when there are too few data points and that it tells us nothing about how the animal use the area inside its home range (i.e. no utilization distribution). While LoCoH home range estimates also have the danger of Type I errors, they have the advantage of providing a utilization distribution (Lichti and Swihart, 2011). Since different methods focus on answering different questions, I concur with Laver and Kelly (2008) and Gregory (2017) that home range studies should include at least two different estimates of home range size as well as providing all the relevant variables and software used (Mitchell, 2006), in order to ease comparisons between studies.

While different home range estimation methods can explain some of the variation in reported home range sizes for these two predators in Namibia, the data supports the claim that at least some real variation is involved. Cheetahs had larger home ranges than leopards, making them more vulnerable to conflict with farmers. That said, for both species a single farmer out of every 10 killing predators indiscriminately, can have a negative effect on the long-term survival of the species, possibly creating a population sink and continuously disrupting the spatial structure of the predators (Pulliam, 1988; Minnie et al., 2016). As a conflict mitigation method, re-releasing the predator back into its own home range with a GPS collar, was found to be just as effective as translocating the animal. This method, while not really a method that the farmer can implement on his own farm because of the high costs, is still given a prior success probability of 0.64, while for translocation the prior probability (p(success) = 0.45) from the literature (Chapter 2) should be used, as it measured success at both the capture and the release site. Although not included in the Decision Support System for farmers and land managers, this conflict mitigation method should be kept in mind as an option by conservation organisations dealing with HWC.

Rainfall has a significant effect on home ranges size of male leopards, (much less on females), with higher rainfall areas having smaller home ranges and thus expected to have higher densities of leopards. Different habitat types also had a significant effect on leopard home ranges, but can be expected to be influenced by

[4]Here "perform better" is understood as correctly excluding areas that are not part of the home range (avoiding Type II errors).

rainfall. The observed effect of rainfall can thus be either because of higher prey densities at higher rainfall (as hypothesised by Marker and Dickman, 2005) or because the habitat type improves the hunting success of leopards (Balme et al., 2007). The fact that rainfall does not have a similar effect on cheetahs (who should also benefit from higher prey densities), suggests that it is the latter, favourable hunting habitat, rather than simply prey densities that is responsible for this observation (Muntifering et al., 2006; Balme et al., 2007). The difference in the effect of rainfall between male and female leopards, could indicate a population dynamic similar to that studied by Fattebert et al. (2016), which might be typical of recovering leopard populations, with males retaining their home range sizes through intra-specific competition, while females at low densities initially have larger home ranges than they need, which shrink over time as they cede more of their home ranges to their daughters when densities increase until they become resource limited and their home range sizes stabilize as well. At this point they probably return to the typical feline pattern where males attempt to maximize their breeding opportunities by having large territories, and the smaller female home ranges are resource limited (Estes, 1991; Fattebert et al., 2016).

A new and fast approach to predicting probable kill sites by larger predators using a conceptual model based on predator movement analysed by T-LoCoH is described. This model has not been ground-truthed yet and is based on the spatial behaviour of collared predators, rather than on confirmed kill sites. However, it is essentially the same approach as that of typical cluster analysis, which has been ground-truthed and repeatedly shown as effective for identifying kill sites (Anderson and Lindzey, 2003; Fröhlich, 2011; Martins et al., 2011; Pitman et al., 2013b). The major advantage of the T-LoCoH model over typical cluster analysis is that the time spent around a cluster is taken into account. Repeat visits more than 5 days apart to the same place (e.g. a den site), will not register as a probable kill site by T-LoCoH (since the visits are removed *in time*), while it would look like a cluster (and thus possible kill site) when using traditional cluster analysis. While underestimating the number of kill sites and undoubtedly missing many, especially within the preferred habitat with repeat visits, T-LoCoH is thus less likely to have false positives than traditional cluster analysis. The two predator species are predicted to preferentially hunt in different habitat types, although they share some hunting habitat (like water drainage lines). Because of not clearly discerning hunting habitat when it is within the preferred core of the home range, it is advised that not only the preferred hunting habitats, but also the generally preferred habitats of the predator species should be avoided by vulnerable livestock. The two species mostly prefer different habitat types within their home range, measured as either the 50% KDE core home range or as areas of long repeat visits (identified in T-LoCoH). However, in the dense shrubland of the tree and shrub savannah biome, they shared both the flat plains and dry river drainage lines as preferred areas.

There was a significant increase in leopards compared to cheetahs captured on farms as a result of conflict, providing indirect evidence that the numbers of leopards on Namibian farmlands have increased relative to the numbers of cheetahs. The number of farmers calling with cheetahs caught in cage traps to be translocated (or later collared and released on-site) decreased, while the numbers of farmers with captured leopards increased significantly over time (2011-2018). The most parsimonious explanation for this is that there has been an actual increase in leopard numbers relative to cheetah numbers. There is little evidence that this is due to leopards successfully excluding cheetahs from some areas, but it is more likely to be the result of habitat changes, specifically bush encroachment, on much of Namibian farmlands (De Klerk, 2004; Swanepoel et al., 2013). The size of the predator home ranges means that this change in habitat is influenced not only by what happens on a single farm, but rather by what happens on the level of 2 to 11 neighbouring farms. For this reason, it is advised that wildlife management (including predator management) should happen at the level of conservancies which include multiple farms managed as a single ecological unit (McGranahan, 2011; Lindsey et al., 2013b,a; Kinnaird and O'Brien, 2012). Thus the size of the management unit is increased, while the basic principles of a holistic agroecological management approach still stands.

References

Anderson, C. J., Lindzey, F., Apr. 2003. Estimating cougar predation rates from GPS location clusters. Journal of Wildlife Management 67 (2), 307–316.

Athreya, V., Odden, M., Linnell, J., Karanth, K., 2010. Translocation as a tool for mitigating conflict with leopards in human-dominated landscapes of India. Conservation Biology 25 (1), 133–141.

Athreya, V., Odden, M., Linnell, J., Krishnaswamy, J., Karanth, K., 2016. A cat among the dogs: leopard *Panthera pardus* diet in a human-dominated landscape in western Maharashtra, India. Oryx 50 (1), 156–162.

Balme, G., Batchelor, A., De Woronin Britz, N., Seymour, G., Grover, M., Hes, L., Macdonald, D., Hunter, L., Oct. 2012a. Reproductive success of female leopards *Panthera pardus*: the importance of top-down processes. Mammal Review 43 (3), 221–237.

Balme, G., Hunter, L., Slotow, R., 2007. Feeding habitat selection by hunting leopards *Panthera pardus* in a woodland savanna: prey catchability versus abundance. Animal Behaviour 74, 589–598.

Balme, G., Hunter, L., Slotow, R., 2009. Evaluating methods for counting cryptic carnivores. Journal of Wildlife Management 73 (3), 433–441.

Balme, G., Lindsey, P., Swanepoel, L., Hunter, L., 2014. Failure of research to address the rangewide conservation needs of large carnivores: leopards in South Africa as a case study. Conservation Letters 7 (1), 3–11.

Bartnick, T., Van Deelen, T., Quigley, H., Craighead, D., 2013. Variation in cougar (*Puma concolor*) predation habits during wolf (*Canis lupus*) recovery in the southern Greater Yellowstone Ecosystem. Canadian Journal of Zoology 91 (2), 82–93.

Benhamou, S., Cornélis, D., Aug. 2010. Incorporating movement behavior and barriers to improve kernel home range space use estimates. Journal of Wildlife Management 74 (6), 1353–1360.

Bissett, C., Bernard, R., 2007. Habitat selection and feeding ecology of the cheetah (*Acinonyx jubatus*) in thicket vegetation: is the cheetah a savanna specialist? Journal of Zoology 271, 310–317.

Blonder, B., Lamanna, C., Violle, C., Enquist, B., May 2014. The n-dimensional hypervolume. Global Ecology and Biogeography 23 (5), 595–609.

Boshoff, A., Kerley, G., Oct. 2001. Potential distributions of the medium- to large-sized mammals in the Cape Floristic Region, based on historical accounts and habitat requirements. African Zoology 36 (2), 245–273.

Bothma, J. P., Knight, M., Le Riche, E., Van Hensbergen, H., 1997. Range size of southern Kalahari leopards. South African Journal of Wildlife Research 27 (3), 94–99.

Burt, W., 1943. Territoriality and home range concepts as applied to mammals. Journal of Mammalogy 24 (3), 346–352.

Caro, T., Collins, D., 1985. Territoriality in male cheetah.

Caro, T., Collins, D., 1986. Male cheetahs of the Serengeti. National Geographic Research 2 (1), 75–86.

Caro, T., Collins, D., 1987. Male cheetah social organization and territoriality. Ecology 74, 52–64.

Carvalho, E., Zarco-González, M., Monroy-Vilchis, O., Morato, R., 2015. Modeling the risk of livestock depredation by jaguar along the Transamazon highway, Brazil. Basic and Applied Ecology 16 (5), 413–419.

Cristescu, B., Bernard, R., Krause, J., 2013. Partitioning of space, habitat, and timing of activity by large felids in an enclosed South African system. Journal of Ethology 31 (3), 285–298.

De Klerk, J., 2004. Bush encroachment in Namibia: Report on phase 1 of the bush encroachment research, monitoring and management project. Tech. rep., Ministry of Environment and Tourism.

Di Marco, M., Boitani, L., Mallon, D., Hoffmann, M., Iacucci, A., Meijaard, E., Visconti, P., Schipper, J., Rondinini, C., 2014. A retrospective evaluation of the global decline of carnivores and ungulates. Conservation Biology 28, 1109–1118.

Dröge, E. and Creel, S. and Becker, M.S. and M'Soka, J., 2017. Spatial and temporal avoidance of risk within a large carnivore guild. Ecology and Evolution 7 (1), 189–199.

Durant, S., Mitchell, N., Groom, R., Pettorelli, N., Ipavec, A., Jacobson, A., Woodroffe, R., Böhm, M., Hunter, L., Becker, M., Broekhuis, F., Bashir, S., Andresen, L., Aschenborn, O., Beddiaf, M., Belbachir, F., Belbachir-Bazi, A., Berbash, A., Brandao de Matos Machado, I., Breitenmoser, C., Chege, M., D., C., Davies-Mostert, H., Dickman, A., Fabiano, E., Farhadinia, M., Funston, P., Henschel, P., Horgan, J., De Iongh, H., Jowkar, H., Klein, R., Lindsey, P., Marker, L., Marnewick, K., Melzheimer, J., Merkle, J., M'Soka, J., Msuha, M., O'Neill, H., Parker, M., Purchase, G., Sahailou, S., Saidu, Y., Samna, A., Schmidt-Küntzel, A., Selebatso, E., Sogbohossou, E., Soultan, A., Stone, E., Van der Meer, E., Van Vuuren, R., Wykstra, M., Young-Overton, K., 2017. The global decline of cheetah *Acinonyx jubatus* and what it means for conservation. PNAS 114 (3), 528–533.

Durant, S., Mitchell, N., Ipavec, A., Groom, R., 2015. *Acinonyx jubatus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T219A50649567.en, p. e.T219A50649567.

Estes, R., 1991. The behavior guide to African mammals: Including hoofed mammals, carnivores, primates. University of California Press.

Fattebert, J., Balme, G., Robinson, H., Dickerson, T., Slotow, R., Hunter, L., 2016. Population recovery highlights spatial organization dynamics in adult leopards. Journal of Zoology, London 299 (3), 153–162.

Fieberg, J., Kochanny, C., Oct. 2005. Quantifying home-range overlap: The importance of the utilization distribution. Journal of Wildlife Management 69 (4), 1346–1359.

Fischer, J., Lindenmayer, D., 2000. An assessment of the published results of animal relocations. Biological Conservation 96, 1–11.

Flagel, D., Belovsky, G., Cramer, M., Beyer, D. J., Robertson, K., 2017. Fear and loathing in a Great Lakes forest: cascading effects of competition between wolves and coyotes. Journal of Mammalogy 98 (1), 77–84.

Foster, R., Harmsen, B., Doncaster, C., 2010. Sample-size effects on diet analysis from scats of jaguars and pumas. Mammalia 74, 1–5.

Fretwell, S., Lucas, H., 1969. On territorial behavior and other factors influencing habitat distribution in birds. I. Theoretical Development. Acta Biotheoretica 19 (1), 16–36.

Fröhlich, M., 2011. Studying the foraging ecology of leopards (*Panthera pardus*) using activity and location data: an exploratory attempt. Master's thesis, Humboldt-Universität zu Berlin.

Getz, W., Fortmann-Roe, S., Cross, P., Lyons, A., Ryan, S., Wilmers, C., 2007. LoCoH: Nonparameteric kernel methods for constructing home ranges and utilization distributions. PLoS ONE 2 (2), e207.

Getz, W., Wilmers, C., 2004. A local nearest-neighbor convex-hull construction of home ranges and utilization distributions. Ecography 27, 489–505.

Gregory, T., 2017. Home range estimation. In: Bezanson, M., MacKinnon, K., Riley, E., Campbell, C., Nekaris, K., Estrada, A., Di Fiore, A., Ross, S., Jones-Engel, L., Thierry, B., Sussman, R., Sanz, C., Loudon, J., Elton, S., Fuentes, A. (Eds.), The International Encyclopedia of Primatology, 1st Edition. John Wiley & Sons Ltd, doi:10.1002/9781119179313.wbprim0177, pp. 1–4.

Hayward, M., Adendorff, J., Moolman, L., Hayward, G., Kerley, G., 2007a. The successful reintroduction of leopard *Panthera pardus* to the Addo Elephant National Park. African Journal of Ecology 45 (1), 103–104.

Hayward, M., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G., Kerley, G., 2006a. Prey preferences of the leopard (*Panthera pardus*). Journal of Zoology, London 270 (2), 298–313.

Hayward, M., Hofmeyr, M., O'Brien, J., Kerley, G., 2006b. Prey preferences of the cheetah (*Acinonyx jubatus*) (Felidae: Carnivora): morphological limitations or the need to capture rapidly consumable prey before kleptoparasites arrive? Journal of Zoology 270, 615–627.

Hayward, M., O'Brien, J., Kerley, G., 2007b. Carrying capacity of large African predators: Predictions and tests. Biological Conservation 139, 219–229.

Hayward, M., Slotow, R., 2009. Temporal partitioning of activity in large African carnivores: tests of multiple hypotheses. South African Journal of Wildlife Research 39 (2), 109–125.

Hebblewhite, M., Haydon, D., 2010. Distinguishing technology from biology: a critical review of the use of GPS telemetry data in ecology. Philosophical Transactions of the Royal Society B 365, 2303–2312.

Hildebrand, M., 1961. Further studies on locomotion of the cheetah. Journal of Mammalogy 42 (1), 84–91.

Hothorn, T., Hornik, K., Van de Wiel, M., Zeileis, A., 2010. coin: A computational framework for conditional inference.

Hunter, J., Durant, S., Caro, T., 2007. To flee or not to flee: predator avoidance by cheetahs at kills. Behavioral Ecology and Sociobiology 61 (7), 1033–1042.

Hutchinson, G., 1957. Concluding remarks. In: Cold Spring Harbor Symposia on Quantitative Biology. 22. doi:10.1101/SQB.1957.022.01.039, pp. 415–427.

Jansen, C., 2016. Diet of key predators responsible for livestock conflict in Namaqualand, South Africa. Master's thesis, Stellenbosch University.

Kaplan, M., Jun. 2013. Speed test for wild cheetahs. State-of-the-art collar reveals animal's quick reflexes and phenomenal acceleration. Nature 498, 150.

Kenward, R., Clarke, R., Hodder, K., Walls, S., Jul. 2001. Density and linkage estimators of home range: Nearest-neighbor clustering defines multinuclear cores. Ecology 82 (7), 1905–1920.

Kie, J., 2013. A rule-based ad hoc method for selecting a bandwidth in kernel home-range analyses. Animal Biotelemetry 1 (13), 1–12.

Kie, J., Matthiopoulos, J., Fieberg, J., Powel, R., Cagnacci, F., Mitchell, M., Gaillard, J., Moorcroft, P., 2010. The home-range concept: are traditional estimators still relevant with modern telemetry technology? Philosophical Transactions of the Royal Society B 365, 2221–2231.

Kinnaird, M., O'Brien, T., 2012. Effects of private-land use, livestock management, and human tolerance on diversity, distribution, and abundance of large African mammals. Conservation Biology 26 (6), 1026–1039.

Lande, R., Aug. 1995. Mutation and conservation. Conservation Biology 9 (4), 782–791.

Laver, P., Kelly, M., Jan. 2008. A critical review of home range studies. Journal of Wildlife Management 72 (1), 290–298.

Lichti, N., Swihart, R., 2011. Estimating utilization distributions with kernel versus local convex hull methods. Journal of Wildlife Management 75 (2), 413–422.

Lindsey, P., Havemann, C., Lines, R., Palazy, L., Price, A., Retief, T., Rhebergen, T., Van der Waal, C., 2013a. Determinants of persistence and tolerance of carnivores on Namibian ranches: Implications for conservation on Southern African private lands. PLoS ONE 8 (1), e52458.

Lindsey, P., Havemann, C., Lines, R., Price, A., Retief, E., Rhebergen, T., Van der Waal, C., Romañach, S., 2013b. Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. Oryx 47 (1), 41–53.

Lindsey, P., Tambling, C., Brummer, R., Davies-Mostert, H., Hayward, M., Marnewick, K., Parker, D., 2011. Minimum prey and area requirements of the Vulnerable cheetah *Acinonyx jubatus*: implications for reintroduction and management of the species in South Africa. Oryx 45 (4), 587–599.

Linnell, J., Aanes, R., Swenson, J., 1997. Translocation of carnivores as a method for managing problem animals: a review. Biodiversity and Conservation 6 (9), 1245–1257.

Linnell, J., Odden, J., Smith, M., Aanes, R., Swenson, J., 1999. Large carnivores that kill livestock: Do "problem individuals" really exist? Wildlife Society Bulletin 27 (3), 698–705.

Linnell, J., Smith, M., Odden, J., Swenson, J., Kaczensky, P., 1996. Carnivores and sheep farming in Norway. 4. Strategies for the reduction of carnivore - livestock - conflicts: a review. Tech. rep., Norsk Institutt for Naturforskning.

Loveridge, A., Macdonald, D., 2002. Habitat ecology of two sympatric species of jackals in Zimbabwe. Journal of Mammalogy 83 (2), 599–607.

Luyt, E. d. C., 2004. Models of bontebok (*Damaliscus pygargus pygargus*, Pallas 1766) habitat preferences in the Bontebok National Park and sustainable stocking rates. Master's thesis, Stellenbosch University.

Lynch, M., Lande, R., 1998. The critical effective size for a genetically secure population. Animal Conservation 1 (1), 70–72.

Lyons, A., Mar. 2014. T-LoCoH - A spatiotemporal home range estimator for GPS data. Remotely Wild 30, 7–9.

Lyons, A., Turner, W., Getz, W., Jul. 2013. Home range plus: a space-time characterization of movement over real landscapes. Movement Ecology 1 (1), 2.

López-Bao, J., Rodrigues, A., 2014. Revisiting food-based models of territoriality in solitary predators. Journal of Animal Ecology 83 (4), 934–942.

Marker, L., 2000. Aspects of the ecology of the cheetah (*Acinonyx jubatus*) on North Central Namibian farmlands. Journal of Namibia Scientific society 48, 40–48.

Marker, L., Dickman, A., 2005. Factors affecting leopard (*Panthera pardus*) spatial ecology, with particular reference to Namibian farmlands. South African Journal of Wildlife Research 35 (2), 105–115.

Marker, L., Dickman, A., Jeo, R., Mills, M., Macdonald, D., 2003a. Demography of the Namibian cheetah, *Acinonyx jubatus jubatus*. Biological Conservation 114, 413–425.

Marker, L., Dickman, A., Mills, M., Jeo, R., Macdonald, D., 2008. Spatial ecology of cheetahs on north-central Namibian farmlands. Journal of Zoology 274, 226–238.

Marker, L., Dickman, A., Mills, M., Macdonald, D., 2010. Cheetahs and ranchers in Namibia: a case study. In: Macdonald, D., Loveridge, A. (Eds.), Biology and Conservation of Wild Felids. Oxford University Press, Ch. 15, pp. 353–372.

Marker, L., Muntifering, J., Dickman, A., Mills, M., Macdonald, D., 2003b. Quantifying prey preferences of free-ranging Namibian cheetahs. South African Journal of Wildlife Research 33 (1), 43–53.

Marker-Kraus, L., Kraus, D., Barnett, D., Hurlbut, S., 1996. Cheetah survival on Namibian farmlands. Cheetah Conservation Fund.

Martins, Q., 2010. The ecology of the leopard *Panthera pardus* in the Cederberg Mountains. Ph.D. thesis, University of Bristol.

Martins, Q., Horsnell, W., Titus, W., Rautenbach, T., Harris, S., 2011. Diet determination of the Cape Mountain leopards using global positioning system location clusters and scat analysis. Journal of Zoology 283, 81–87.

McGranahan, D., 2008. Managing private, commercial rangelands for agricultural production and wildlife diversity in Namibia and Zambia. Biodiversity and Conservation 17 (8), 1965–1977.

McGranahan, D., 2011. Identifying ecological sustainability assessment factors for ecotourism and trophy hunting operations on private rangeland in Namibia. Journal of Sustainable Tourism 19 (1), 115–131.

McVittie, R., 1979. Changes in the social behaviour of South West African cheetah. Madoqua 11 (3), 171–184.

Melzheimer, J., Streif, S., Wasiolka, B., Fischer, M., Thalwitzer, S., Heinrich, S., Weigold, A., Hofer, H., Wachter, B., 2018. Queuing, takeovers, and becoming a fat cat: Long-term data reveal two distinct male spatial tactics at different life-history stages in Namibian cheetahs. Ecosphere 9 (6), e02308.

Mendelsohn, J., Jarvis, A., Roberts, C., Robertson, T., 2003. Atlas of Namibia - A portrait of the land and its people. David Phillip Publishers, http://www.the-eis.com/.

Messmer, T., 2000. The emergence of human-wildlife conflict management: turning challenges into opportunities. International Biodeterioration & Biodegradation 45, 97–102.

Miller, J., 2015. Mapping attack hotspots to mitigate human–carnivore conflict: approaches and applications of spatial predation risk modeling. Biodiversity and Conservation 24 (12), 2887–2911.

Miller, J., Jhala, Y., Jena, J., 2016. Livestock losses and hotspots of attack from tigers and leopards in Kanha Tiger Reserve, Central India. Regional Environmental Change 16 (Suppl 1), S17–S29.

Miller, J., Jhala, Y., Jena, J., Schmitz, O., 2015. Landscape-scale accessibility of livestock to tigers: implications of spatial grain for modeling predation risk to mitigate human–carnivore conflict. Ecology and Evolution 5 (6), 1354–1367.

Mills, M., Broomhall, L., Du Toit, J., 2004. Cheetah *Acinonyx jubatus* feeding ecology in the Kruger National Park and a comparison across African savanna habitats: is the cheetah only a successful hunter on open grassland plains? Wildlife Biology 10 (3), 177–186.

Minnie, L., Boshoff, A., Kerley, G., 2015. Vegetation type influences livestock predation by leopards: Implications for conservation in agro-ecosystems. African Journal of Wildlife Research 45 (2), 204–215.

Minnie, L., Gaylard, A., Kerley, G., 2016. Compensatory life-history responses of a mesopredator may undermine carnivore management efforts. Journal of Applied Ecology 53, 379–387.

Mitchell, B., 01 2006. Comparison of programs for fixed kernel home range analysis. Remotely Wild 21.

Muntifering, J., Dickman, A., Perlow, L., Hruska, T., Ryan, P., Marker, L., Jeo, R., 2006. Managing the matrix for large carnivores: a novel approach and perspective from cheetah (*Acinonyx jubatus*) habitat suitability modelling. Animal Conservation 9, 103–112.

Neteler, M., Bowman, M., Landa, M., Metz, M., 2012. GRASS GIS: A multi-purpose open source GIS. Environmental Modelling & Software 31, 124–130.

Nghikembua, M., Harris, J., Tregenza, T., Marker, L., 2016. Spatial and temporal habitat use by GPS collared male cheetahs in modified bushland habitat. Open Journal of Forestry 6 (4), 269–280.

Pitman, R., Kilian, P., Ramsay, P., Swanepoel, L., 2013a. Foraging and habitat specialization by female leopards (*Panthera pardus*) in the Waterberg Mountains of South Africa. South African Journal of Wildlife Research 43 (2), 167–176.

Pitman, R., Mulvaney, J., Ramsay, P., Jooste, E., Swanepoel, L., 2013b. Global Positioning System-located kills and faecal samples: a comparison of leopard dietary estimates. Journal of Zoology 11 (3), 169–178.

Powell, R., 2000. Animal home ranges and territories and home range estimators. In: Boitani, L., Fuller, T. (Eds.), Research Techniques in Animal Ecology - Controversies and Consequences. Columbia University Press, Ch. 3, pp. 65–110.

Pulliam, H., 1988. Sources, sinks, and population regulation. The American Naturalist 132 (5), 652–661.

Puth, M., Neuhäuser, M., Ruxton, G., 2015. On the variety of methods for calculating confidence intervals by bootstrapping. Journal of Animal Ecology 84 (4), 892–897.

QGIS-Development-Team, Apr. 2017. QGIS user - guide release 2.18. http://qgis.osgeo.org.

R Development Core Team, R., 2011. R: A language and environment for statistical computing. URL `http://www.r-project.org`

Ray, J., Hunter, L., Zigouris, J., 2005. Setting conservation and research priorities for larger African carnivores. Tech. rep., Wildlife Conservation Society.

Ripple, W., Estes, J., Beschta, R., Wilmers, C., Ritchie, E., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M., Schmitz, O., Smith, D., Wallach, A., Wirsing, A., 2014. Status and ecological effects of the world's largest carnivores. Science 343 (6167), 1241484.

Ropiquet, A., Knight, A., Born, C., Martins, Q., Balme, G., Kirkendall, L., Hunter, L., Senekal, C., Matthee, C., 2015. Implications of spatial genetic patterns for conserving African leopards. Comptes Rendus Biologies 338, 728–737.

Rosenzweig, M., Apr. 1981. A theory of habitat selection. Ecology 62 (2), 327–335.

Rothauge, A., 2011b. Ecological dynamics of Central Namibia's savannas: Part 2 - Bush ecology. Agricola 21, 14–25.

Scantlebury, D., Mills, M., Wilson, R., Mills, M., Durant, S., Bennett, M., Bradford, P., Marks, N., Speakman, J., 2014. Flexible energetics of cheetah hunting strategies provide resistance against kleptoparasitism. Science 346 (6205), 79–81.

Schipper, J., Chanson, J., Chiozza, F., Cox, N., Hoffmann, M., Katariya, V., Lamoreux, J., Rodrigues, A., Stuart, S., Temple, H., Baillie, J., Boitani, L., Lacher, T. J., Mittermeier, R., Smith, A., Absolon, D., Aguiar, J., Amori, G., Bakkour, N., Baldi, R., Berridge, R., Bielby, J., Black, P., Blanc, J., Brooks, T., Burton, J., Butynski, T., Catullo, G., Chapman, R., Cokeliss, Z., Collen, B., Conroy, J., Cooke, J., Da Fonseca, G., Derocher, A., Dublin, H., Duckworth, J., Emmons, L., Emslie, R., Festa-Bianchet, M., Foster, M., Foster, S., Garshelis, D., Gates, C., Gimenez-Dixon, M., Gonzalez, S., Gonzalez-Maya, J., Good, T., Hammerson, G., Hammond, P., Happold, D., Happold, M., Hare, J., Harris, R., Hawkins, C., Haywood, M., Heaney, L., Hedges, S., Helgen, K., Hilton-Taylor, C., Hussain, S., Ishii, N., Jefferson, T., Jenkins, R., Johnston, C., Keith, M., Kingdon, J., Knox, D., Kovacs, K., Langhammer, P., Leus, K., Lewison, R., Lichtenstein, G., Lowry, L., Macavoy, Z., Mace, G., Mallon, D., Masi, M., McKnight, M., Medellin, R., Medici, P., Mills, G., Moehlman, P., Molur, S., Mora, A., Nowell, K., Oates, J., Olech, W., Oliver, W., Oprea, M., Patterson,

B., Perrin, W., Polidoro, B., Pollock, C., Powel, A., Protas, Y., Racey, P., Ragle, J., Ramani, P., Rathbun, G., Reeves, R., Reilly, S., Reynolds, J. I., Rondinini, C., Rosell-Ambal, R., Rulli, M., Rylands, A., Savini, S., Schank, S., Sechrest, W., Self-Sullivan, C., Shoemaker, A., Sillero-Zubiri, C., De Silva, N., Smith, D., Srinivasulu, C., Stephenson, P., Van Strien, N., Talukdar, B., Taylor, B., Timmins, R., Tirira, D., Tognelli, M., Tsytsulina, K., Veiga, L., Vié, J., Williamson, E., Wyatt, S., Xie, Y., Young, B., 2008. The status of the world's land and marine mammals: Diversity, threat, and knowledge. Ecology 322 (5899), 225–230.

Sharp, N., 1997. Timed running speed of a cheetah (*Acinonyx jubatus*). Journal of Zoology, London 241, 493–494.

Signer, J., Balkenhol, N., Mar. 2015. Reproducible Home Ranges (rhr): A New, User-Friendly R package for analyses of wildlife telemetry data. Wildlife Society Bulletin 39 (2), 358–363.

Simon, J., Oct. 1997. Resampling: The new statistics., 2nd Edition. Resampling Stats, http://www.resample.com/intro-text-online/.

Sinclair, A., Mduma, S., Brashares, J., 2003. Patterns of predation in a diverse predator–prey system. Nature 425, 288–290.

Stander, P., 1990. A suggested management strategy for stock-raiding lions in Namibia. South African Journal of Wildlife Research 20 (2), 37–43.

Stander, P. E., Haden, P. J., Kaqece, ||., Ghau, ||., 1997. The ecology of asociality in Namibian leopards. Journal of Zoology, London 242, 343–364.

Stein, A., 2008. Ecology and conservation of the leopard (*Panthera pardus* Linnaeus 1758) in Northcentral Namibia. Ph.D. thesis, University of Massachusetts Amherst.

Stein, A., Athreya, V., Gerngross, P., Balme, G., Henschel, P., Karanth, U., Miquelle, D., Rostro-García, S., Kamler, J., Laguardia, A., Khorozyan, I., Ghoddousi, A., 2015. *Panthera pardus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://www.iucnredlist.org/details/summary/15954/0, p. e.T15954A102421779.

Stein, A., Fuller, T., DeStefano, S., Marker, L., 2011. Leopard population and home range estimates in north-central Namibia. African Journal of Ecology 49, 383–387.

Steiniger, S., Hunter, A., 2013. A scaled line-based kernel density estimator for the retrieval of utilization distributions and home ranges from GPS movement tracks. Ecological Informatics 13, 1–8.

Swanepoel, L., Lindsey, P., Somers, M., Van Hoven, W., Dalerum, F., 2013. Extent and fragmentation of suitable leopard habitat in South Africa. Animal Conservation 16 (1), 41–50.

Swanepoel, L., Somers, M., Van Hoven, W., Schiess-Meier, M., Owen, C., Snyman, A., Martins, Q., Senekal, C., Camacho, G., Boshoff, W., Dalerum, F., 2015. Survival rates and causes of mortality of leopards *Panthera pardus* in southern Africa. Oryx 49 (4), 595–603.

Tomkiewicz, S., Fuller, M., Kie, J., Bates, K., 2010. Global positioning system and associated technologies in animal behaviour and ecological research. Philosophical Transactions of the Royal Society B 365, 2163–2176.

Treves, A., Bruskotter, J., 2014. Tolerance for predatory wildlife. Science 344, 476–477.

Treves, A., Karanth, K., 2003b. Special section: Human-carnivore conflict: Local solutions with global applications. Conservation Biology 17 (6), 1489–1490.

Weilenmann, M., Gusset, M., Mills, D., Gabanapelo, T., Schiess-Meier, M., 2010. Is translocation of stock-raiding leopards into a protected area with resident conspecifics an effective management tool? Wildlife Research 37, 702–707.

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

Weise, F., Lemeris, J. J., Stratford, K., Van Vuuren, R., Munro, S., Crawford, S., Marker, L., Stein, A., 2015b. A home away from home: insights from successful leopard (*Panthera pardus*) translocations. Biodiversity and Conservation 24, 1755–1774.

Weise, F., Stratford, K., Van Vuuren, R., 2014. Financial costs of large carnivore translocations – Accounting for conservation. PLoS ONE 9 (8), e105042.

Weise, F., Vijay, V., Jacobson, A., Schoonover, R., Groom, R., Horgan, J., Keeping, D., Klein, R., Marnewick, K., Maude, G., Melzheimer, J., Mills, G., Van der Merwe, V., Van der Meer, E., Van Vuuren, R., Wachter, B., Pimm, S., 2017. The distribution and numbers of cheetah (*Acinonyx jubatus*) in southern Africa. PeerJ 5, e4096.

Wilson, A., Lowe, J., Roskilly, K., Hudson, P., Golabek, K., McNutt, J., 2013. Locomotion dynamics of hunting in wild cheetahs. Nature 498, 185–189.

Yu, C., 2003. Resampling methods: Concepts, Applications, and Justification. Practical Assessment, Research & Evaluation 8 (19), 1–16.

Chapter 5

Building a Probabilistic Graphical Model for Human-Wildlife Conflict

Abstract

Although many methods have been suggested to mitigate human-predator conflict, it is still a worldwide and increasingly important problem, both for predator conservation and for livestock farming. Ultimately farmers remain the custodians of predators on their land and need to be empowered to do a better job of managing their land, including both agricultural and ecological aspects of farmer-predator conflict. An online Decision Support System can put the relevant knowledge into the hands of farmers who are struggling with livestock depredation on their land. The design for a Decision Support System (DSS) is presented based on a Probabilistic Graphical Model that can be used by decision makers to find the most appropriate conflict mitigation technique for farmer-predator conflict situations which will also have the greatest utility on Namibian farmlands. This will address the common problem of conflict mitigation methods being proposed that are impractical for a certain farm. It will also avoid the problem of such a wide range of different methods being presented, that the farmer has no guidance on how to pick an affordable, practical, effective and sustainable solution.

Keywords: Decision Support System, Human-wildife conflict, HWC, farmer-predator conflict, mitigating methods, Probablistic Graphical Model, PGM, Namibia, sustainability

5.1 Introduction

Human-wildlife conflict (HWC) is a growing global problem (Messmer, 2000; Treves and Karanth, 2003; Ripple et al., 2014). Land managers, specifically farmers, are key to finding sustainable solutions to this conflict. One major reason why attempts at mitigating farmer-predator conflict have failed in the past, is because the local conditions on a specific farm have not sufficiently been taken into account (Section 1.2.2; Rigg et al., 2011). Farmers often have to make decisions on a predator management strategy that will be both cost-effective and sustainable in the long run, but without having the relevant information (Conradie and Piesse, 2013; Nattrass and Conradie, 2018; Haswell et al., 2019). Farmlands are also part of a wider ecosystem, which is one part of reducing HWC that is often missing in the decision-making process. Thus, an agroecological approach, taking into account both the ecological sustainability and the agricultural cost-effectiveness of any proposed solution, is critical to its success (Figure 1.4 on page 12).

Typically, ecological modelling has fallen into two broad groups aiming to address different questions (Rosenzweig and MacArthur, 1963).

1. Simulation models typically attempt to incorporate as many different parameters as possible in order to be as realistic and accurate as possible (Parrott, 2011). Because of the wide number of parameters, some of which might not be relevant, these kinds of models often end up as a "black box" with little guarantee that all the relevant factors have been included in the model or that it is actually a true reflection of reality (e.g. Young and Harcourt, 1997; Erasmus et al., 2002; Hannah et al., 2005; many climate change models) and can sometimes give counter-intuitive results (Gelman, 2011). They are often no longer understandable from a human perspective because of this complexity, but also because the main objective is the ability to describe or predict (the output), rather than understanding causality (how the output is produced) (Jørgensen, 2008).

2. The opposite approach is to simplify matters. It is accepted that the model is not a realistic representation of an ecosystem (Rosenzweig and MacArthur, 1963). Instead, the aim of the model is to gain a better understanding of the ecosystem (Jørgensen, 2008). It is a conditional model, only claiming that *if* certain conditions and assumptions are true *then* certain effects can be expected as a result (Rosenzweig and MacArthur, 1963). It concentrates only on those parts of the model that are relevant to the specific question being asked and ignores all other influences. A major aim of the model is to keep it simple enough to be understood by humans (examples include the well-known Lotka-Volterra growth model, applied by Caughley, 1976 for defining two types of carrying capacity, predator-prey interactions Huffaker, 1958; Rosenzweig and MacArthur, 1963; Dieguez Cameroni and Fort, 2017; Fort et al., 2017, etc.).

These are not two absolute categories, but rather two extremes on a scale with various options in-between. Jørgensen (2008) identified 11 different categories of models used for ecological modelling falling somewhere between a total black box and a clear, easy-to-understand but less accurate causal model. For livestock farmers to decide on the best solution for preventing livestock depredation in their specific situation, it is important that they are able to trust the output of a specific model. A model that is more easily understandable is therefore more useful in this context than a possibly more realistic, but incomprehensible black box model (Jørgensen, 2008). Specifically, the one main disadvantage of black box models is that they do not include causality – we can predict *what* will happen, but not *why* it will happen (Jørgensen, 2008). It is also important that those agricultural aspects of the farming enterprise that will play an important role in the decision making, are included in the model. Farmers have to survive in uncertain conditions and explicit recognition of this uncertainty as part of all farming, can help them to realize the limitations of management decisions based on uncertain facts (Ascough et al., 2008; Conradie and Piesse, 2016).

A model designed for mitigating livestock farmer-predator conflict in Namibia is presented here, combining both agricultural and ecological information in a holistic agroecological model. It is included within the design for a Decision Support System (DSS) to provide farmers with the relevant information for managing livestock depredation on their land (Keen, 1980; Sprague, 1980). The model avoids the trap of proposing a one-size-fits-all "silver bullet" solution on the one hand, or the "scattergun" approach of giving a long list of mitigating methods with no guidance on how to choose one, on the other (Section 1.2.2 in Chapter 1). It is envisioned that the same approach can be taken in other HWC situations and refined over time.

5.1.1 Why use Probabilistic Graphical Models (PGMs)?

There are knowledge gaps with regards to agroecosystems and this leads to uncertainty (Allen and Fildes, 2001; Ascough et al., 2008; Turpie and Akinyemi, 2018). Historically there have been three approaches to model uncertainty in computer systems.

1. Fuzzy logic (Bullinaria, 2005; Tangestani, 2009; Comber et al., 2010; Shafer and Pearl, 1990) does not consider statements as either true (1) or false (0), but as having a certain degree of truth between 0 and 1, resulting in a multi-valued logic (Shafer and Pearl, 1990; Bullinaria, 2005). This leads to the

counter-intuitive result that $f(A \bigvee \neg A) \neq f(True)$ (see text box *Logical operators* below; Bullinaria, 2005). Whereas normally either a statement A or its negation NOT A, must be true, this is not necessarily true in fuzzy logic (Shafer and Pearl, 1990; Koller and Friedman, 2005). As a result, although fuzzy logic based expert systems have been very successful in commercial applications, these have tended to be rather specialized, with limited levels of inference, and the various parameters often need to be tuned using machine learning techniques (Bullinaria, 2005). The fact that it includes some obviously counter-intuitive statements, makes it difficult to understand or explain and thus not a good choice for our use case. Additionally, farming systems and ecosystems are complicated, making it difficult to model by using fuzzy logic that are better suited to smaller systems (Bullinaria, 2005). It is tricky to design a model using fuzzy logic to avoid some of the counter-intuitive results mentioned above. This becomes more difficult as the system being modelled becomes larger and more complicated.

Logical operators:

$\bigvee$	(logical) OR (disjunction)
$\neg$	(logical) NOT (negation)
$\bigcap$	(logical) AND (conjuction)
$\mid$	(logical) given (conditional)

2. Dempster-Shafer modelling (Shafer, 1976; Shafer and Pearl, 1990; Luyt, 2004) is another approach to dealing with uncertainty. Instead of dealing directly with probabilities, it deals with the degree of evidence that we have for certain truth values, called belief functions (Shafer, 1976; Pearl, 1990). This can be a very useful approach in some circumstances. However, the theoretical basis of this model and the situations for which it is a good fit, are still debated (Shafer, 1981; Zadeh, 1984; Pearl, 1990; Wakker, 2000; Cobb and Shenoy, 2003a,b; Wilson, 2014). It is best used when there are multiple independent sources of evidence with some overlap concerning the same basic question (i.e. an accumulation of evidence) (Benyon et al., 2000; Bullinaria, 2005; Jøsang et al., 2010). As it looks only at evidence, rather than at probabilities, it cannot deal directly with cause and effect (Shafer, 1976; Provan, 1990; Pearl, 1990). It has been considered as a generalization of probabilistic theory, and can be simpler to implement than the Bayesian Network approach (Bullinaria, 2005). However, it has no way to model cause and effect naturally, and can only combine various pieces of evidence for a certain outcome, without implying any causal relationship (Pearl, 1990; Nilsson, 2010). One advantage of this approach, is that an explicit value can be assigned to uncertainty (Shafer, 1981, 1990, 1992). However, it is easily misused for situations where it is not a good fit and can give counter-intuitive or untrustworthy results in such cases (Jøsang et al., 2010).

3. Bayesian probability theory (Barber, 2015) has a long history and is much better understood than the previous two approaches from a mathematical point of view (Koller and Friedman, 2005). One important advantage compared to the other two types of models, is that it provides a very natural method to model dependencies and conditional dependencies, making it very intuitive to model real cause and effect relationships between different factors (Charniak, 1991). However, it can be computationally expensive (Bullinaria, 2005). Until recently, it was impossible to model the joint probability distribution over any reasonable number of n random variables taking x different values per variable (Koller and Friedman, 2005). It would require x^n different probability assignments, which are both computationally expensive and too large to fit into computer memory, and are also incomprehensible to the human brain (Koller and Friedman, 2005). This remained the main barrier to using probabilistic methods for solving problems dealing with uncertainty, until fairly recently when the current methods of Probabilistic Graphical Models (PGMs) were developed (Barber, 2015). The recent developments have resulted in PGMs based on Bayesian probability theory becoming the dominant paradigm in most modern expert systems in general and Decision Support Systems in particular (Koller and Friedman, 2005). For these reasons, this approach was chosen as the preferred one for modelling the different human-wildlife mitigation methods.

There are many different Probabilistic Graphical Models, but the best fit for proposing solutions to farmer-predator conflict, would be an Influence Diagram (Koller 2016, personal communication), an extension of the Bayesian Network. A Bayesian Network is a Directed Acyclical Graph (DAG), consisting of a number of nodes, representing random variables, connected by directed edges, representing dependencies between the nodes and a table with the conditional probability distribution for each node given its parents (Koller and Friedman, 2005; Barber, 2015). For the marginally independent nodes (those without any parents) their probabilities are given by a prior probability distribution. Figure 5.1 illustrates a basic Bayesian Network for the predation management part of an agroecological system on a livestock farm. This basic Bayesian Network has been extended to a form known as Influence Diagrams or Decision Networks (Charniak, 1991; Koller and Friedman, 2005). In this type of model, not only the various random variables and their conditional probability functions are described, but additionally, there are:

1. action variable nodes, denoting actions or decisions that can be taken by humans; and
2. utility nodes, denoting the expected utility that will result from the chosen action.

Running the model for each choice of conflict mitigation method, a Maximum Expected Utility (MEU) value can be calculated for that method. The Expected Utility is a single value combining the probability of a certain outcome, multiplied by the value for the user of that outcome (e.g. the income increases due to the decrease in livestock losses in our case). The Maximum Expected Utility is the Expected Utility resulting from the user choice with the greatest value (greatest probability of "best" outcome). While this MEU may not be an exact value denoting the true maximum expected utility of each choice, it is robust for comparing different human-wildlife conflict mitigation methods, since essentially the same model is used for each method and just the parameter values change, depending on the method. As in any other computer program, the GIGO (garbage in, garbage out) principle still holds (Gelman, 2011). The results are dependent on the reliability of the data provided by the farmer and the robustness of the assumptions built into the model — but this will be true for any alternative approach as well. Using a Probabilistic Graphical Model, these assumptions are stated explicitly as part of the model and can be updated over time in order to improve the model as more data becomes available from users of the DSS. A first conceptual Bayesian Network model is given in Figure 5.1 as an example of the first step in designing the final Decision Network. A basic Bayesian Network would stop at this step (Johnson et al., 2013). It needs to be mentioned that this Bayesian Network can produce a *probability* of success (i.e. some decrease in livestock losses) for a certain method, but it gives no indication of the size of the effect (i.e. by how much it will decrease livestock losses). For including this information in the PGM, a utility function is needed to combine with the probabilities in order to determine the expected utility (including effectiveness in terms of reduction in losses).

5.2 The model - What data do we need to make an informed and useful decision?

Looking at Figure 5.1, the model consists of two elemental types of data (probabilities):

1. Nodes connected by conditional probabilities that are unchanged by the conditions on a specific farm (white nodes in Figure 5.1). This will form the basic core of the model. The directed edges (arrows) between the nodes basically indicates causality with an associated conditional probability distribution, showing how likely each outcome in the child node is, given certain values in the parent nodes. These probability distributions and causal links are the core of the model that remain constant from farm to farm and will be determined by using the literature and existing data as far as possible (Chapters 2 to 5). For some links the probabilities will need to be updated over time to make the model more accurate.

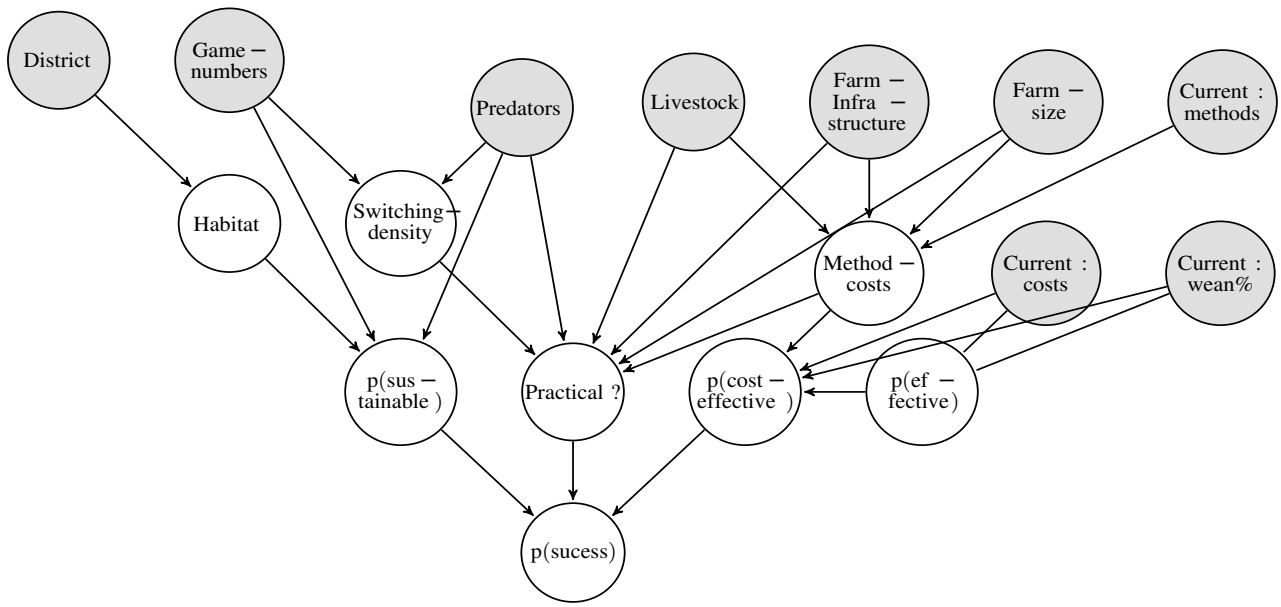


Figure 5.1: *First conceptual Bayesian network PGM illustrating the basic principles of how to determine success probability of farmer-predator conflict resolution methods.* Grey nodes indicate data required from the farmer, white nodes the pre-existing model. The final value is a probability that a specific method will be successful for a specific farmer, but without any representation of decisions or utility (the relative value of success). The model is therefore incomplete for our purpose.

2. Variables that are specific to a specific farm and for which the manager (farmer) will provide the actual observed values (grey nodes in Figure 5.1). These values will form the basic input to the model by the user (farmer or manager) which in turn will determine the actual maximum expected utility of each method.

For livestock farmers to make an informed decision on the best anti-predation method for their specific circumstances, both ecological and agricultural information should be combined in a single agroecological model (Francis et al., 2003; Chapter 1, Figure 1.4).

5.2.1 Agricultural: Cost effectiveness, limitations and relevance

The agricultural data will concern itself primarily with the cost-effectiveness of the different methods, the limitations to where it remains useful, and the practicality of applying the method in different circumstances (i.e. the agricultural or ecological circumstances that will make the method either more or less likely to be effective) (Du Plessis et al., 2018). Specifically, the model needs input on the following aspects for each method being evaluated:

1. The prior probability of effectiveness (success defined as resulting in lower than average livestock depredation levels). [1]

[1] Jøsang (2008) mentions that where the actual prior probabilities of base rates cannot be known with absolute certainty, frequentist approximation through prior empirical observation can be used. We could determine these frequentist approximations

2. The initial implementation costs of each method (relatively to each other). Because of uncertainty in these estimates, cost classes instead of absolute costs can be used to increase robustness. [2]

3. The maintenance costs of each method once implemented (relatively to each other). Like implementation costs, cost classes instead of absolute cost estimates van be used to increase robustness in spite of uncertainty.

4. In addition to the prior probabilities (based on averages), a number of inputs about the specific situation on the farm (e.g. current methods used, predators found on the farm, existing infrastructure on the farm, livestock species and breed, current livestock losses, etc.) and their likely influence (positive or negative) on the probable effectiveness or costs of each method will be required.

5.2.2 Ecological sustainability

The ecological data will concern itself primarily with the sustainability (and effect on the biodiversity) of each predation management method. It will also include the behavioural ecology of the different predators and the effect of the different ecological habitats on the practicality of the different methods (Blackwell et al., 2016). More importantly, ecological data can be used to determine *why* a specific conflict mitigation method is likely to be successful (or not) for a specific predator species or habitat. As input to the model, it will require the following probabilities:

1. The prior probability to be sustainable for each conflict mitigation method. Unlike agricultural cost-effectiveness, which is a short-term measure of success, ecological sustainability implies the indefinite continuation of the ecosystem without loss in biodiversity. This requires long-term research (Sinclair et al., 2007) which has not been done for all four predator species on Namibian farmlands (LCMAN Red Data List for Namibian Carnivores, in prep). In general, the long-term sustainability of various conflict mitigation methods have not been studied (Van Eeden et al., 2018). For this reason, these probabilities will need to be assigned subjective values based on our current understanding of Namibian farmland ecosystems, the importance of high biodiversity for ecosystem stability and resilience, the ecology of the predators and their conservation status. The level of and duration of disturbance to the ecosystem caused by each management technique can be used as a proxy for its sustainability – interventions that cause less disturbance for a shorter period are less likely to have a negative effect on the persistence of a community. For a robust result, the *relative* expected sustainability of the methods will be more important than the absolute probability assigned.

2. The effect of the current state of affairs on the sustainability of each method will also need to be modelled, including existing game fences, current biodiversity on the farm (number of carnivore and herbivore species), and the expected effect of the method on wildlife numbers.

5.2.3 Agricultural data used

From a strictly (and short-term) agricultural point of view, the cost and relative effectiveness of the different methods used to protect livestock from predation are the most important factors. Also important, are the

from 3 sources: 1) where the method had been used by enough Namibian farmers, the *percentage of individuals* from the Farming with Predators project data who had success with a specific conflict mitigation method can be used, 2) for methods not used in Namibia yet, the *percentage of published papers* where that specific method had been shown to be successful can be used (an evidence-based conservation approach, but with the danger of publication bias and the issue of different definitions of "success"), 3) where a specific method had been evaluated for its effectiveness in Namibia before, the *percentage of individual farmers* who had success with the specific method can be used (with data from a single or few published studies). In each case the percentage can be converted directly to a probability (between 0 and 1).

[2] Where available, this data would come directly from the average costs reported by farmers using the specific method. Otherwise these costs will need to be estimated from published studies (if available) or calculated using a best estimate (e.g. using the minimum wage for expected additional labour costs).

limitations of each method (e.g. some methods cannot be combined, or some methods are not practical on a low-stocking rate, extensive farming system) (Chapters 2 & 3).

Other agricultural aspects that will influence the cost, effectiveness and practicality of each method on a specific farm, include the current infrastructure on the land, the availability of capital for farm infrastructure improvements, the sustainable stocking rates of the land, the long-term effects of the different techniques on the veld quality, the kind and breeds of livestock (and the marketing and other reasons for using these specific types), the topography and vegetation types of the land, the density of available water points and the quality and quantity of drinking water on the land, as well as the size of the farm, etc. The current mitigation method used (i.e. how much will it cost to change?) and the current livestock losses (i.e. will the cost be worthwhile?) are also important agricultural inputs.

5.2.4 Ecological data used

For the long-term sustainability of whichever method is used, a number of ecological aspects are important as well (informed by inputs under district, game numbers and predators in Figure 5.1). The predator species on the farm (and their known hunting behaviour), current game densities and current livestock stocking rates will all influence the dietary choices of the predators (see Chapter 2). Predator diet preferences can influence the management of a livestock farmer in two ways:

1. The diet preferences (if any) of the predator species on the farm impacting livestock or game production (i.e. if the farm can be stocked with cheaper, but preferred prey species, the high-value game or livestock animals are less likely to be preyed on by these predators) and
2. the importance of individual prey preference of the predators on his land (i.e. if there are only some individuals that are preferring livestock as prey, removing them from his land might be a viable management option).

Knowing the district of the farm, a general idea of the available habitat types on the farm can be deduced (see Chapter 4). The habitat preferences of predators can influence the farmer's management options in two ways:

1. If different predator species (that presumably prey on different kinds of livestock or age classes – see Table 3.3 on page 146) prefer different habitat types, it may be possible to keep vulnerable livestock away from those high-risk habitats on the farm (provided there is enough low-risk alternative areas available) or
2. if the reason why certain predators prefer certain habitat types within their wide-ranging home ranges can be established (e.g. for hunting or by females with higher prey requirements while having their cubs), vulnerable livestock can be kept away from these habitats (Section 4.4.5 on page 201).

Finally, the interactions between the predator species can also influence management options. It has been claimed that caracals can deter jackals from some areas (Du Toit, 2013). It is also known that leopards will kill and eat black-backed jackals (Bothma and Le Riche, 1994), and that caracals and jackals will occasionally kill each other's young ones (Pringle and Pringle, 1979; Melville et al., 2004; Ray et al., 2005; Bothma, 2012; Tambling et al., 2018). One of the major reasons for the increase in problems with jackals and caracals could be mesopredator release (Beinart, 1998; Ritchie and Johnson, 2009), but the extent and importance of these interactions have not been studied in enough detail to know if it could be a significant reason for the reported increase in livestock depredation. If mesopredator release plays a role in Namibian ecosystems, it becomes important for farmers to maintain enough larger predators on their land to prevent future livestock losses from too many mesopredators. If one species of predator is the major livestock killer on the farm, increasing the numbers of another species that kill or outcompete the livestock killing predator through interference

competition, could decrease livestock losses (alternatively, artificial bio-boundaries using some natural bio-deterrent could have a similar effect – Holmes and Holmes, 2006; Jackson et al., 2012).

While there are probably other aspects of predator ecology on farmlands that will also play a role in HWC (see Balme et al., 2014; Du Plessis et al., 2015), the above were considered as the most important in a Namibian context. It was also important to keep the model as simple as possible in order to remain comprehensible.

5.3 The model

From the literature, 40 different farmer-predator conflict mitigation methods have been identified. Some of these methods are commonly considered as ineffective on their own and normally combined with others (Table 2.1; see Chapter 3), reducing the number of methods to 20. A few methods were considered as so ineffective (e.g. horse patrols) or harmful to the farm ecosystem (e.g. the illegal use of poison), that they were excluded altogether. Some (e.g. most compensatory schemes) cannot be implemented by a single farmer or land manager and can therefore be excluded unless there is at least district-wide cooperation between farmers. Some methods are only likely to be effective if combined with other complementary methods (e.g. herding is unlikely to be effective of not combined with kraaling – having the livestock graze freely at night when most predators are active, is likely to decrease the effectiveness of herding). Effectiveness was calculated from Namibian farmer survey data (Chapter 3), using a frequentist approach to assign it a probability value and from the internationally published literature (Chapter 2) when Namibian data was insufficient (Jøsang, 2008). The proportion of Namibian farmers using a certain method that had lower livestock losses than the median was taken as the prior probability of success of the specific method (see Table 3.4 in Chapter 3). The median was used, because the data was skewed with the mean influenced strongly by a few farmers with very heavy livestock losses. Where the Namibian sample size was small and published data existed, the mean of the two was used. For internationally published data, the proportion of previous studies that reported a method as successful, was taken as the prior success probability (Jøsang, 2008). It is assumed that all of the relevant publications describing a method had been found in the literature search, and that the articles which had been found are representative of the real success rate of the specific mitigation method. The methods included in the model are detailed in Table 5.1.

Table 5.1: *The different mitigation methods used in the model.* Some of the cheaper methods that were reported as relatively ineffective on their own (Chapter 2 & 3) are combined with more effective methods. Limitations are both positive (requirements for implementation, indicated with +) and negative (restrictions that preclude the use of this method, indicated with -). It also includes both agricultural and ecological limitations. Annual costs (Namibian dollars) are classified in four classes rather than absolute costs: 1. N$ 0 - 12 000, 2. N$12 000 - N$120 000, 3. N$ 120 000 - N$ 1 200 000, 4. > N$1 200 000 per year. The prior probability that a method will be effective, is given by p(Effective) based on the farmer survey (Chapter 3) combined with the published literature where Namibian data were not sufficient (Chapter 2). Two methods (CTA, repellent collars) were given a p(Effective) value of 0.5, indicating no current knowledge with regard to its effectiveness (i.e. a 50-50% chance of success). The effectiveness probability of two methods (GPS collars, Translocation) was determined based on information gathered in Chapter 4 (also see Chapter 2, Section 2.2.3). The probability of being sustainable (in brackets) is based on the authors' subjective estimate of the ecological disturbance caused by the method to the farm ecosystem (less disturbance = more sustainable). The last column lists the farm factors which were identified in the literature to either improve or decrease the effectiveness of the method (indicated with a + or -) for which the DSS may need to ask more questions in order to determine its effectiveness (see Figure 5.3 below).

ID	Method	Limiting factors	Implement Cost (N$)	Maintain Cost (N$)	p (Effect) p(Sustain)	Influencing factors
1.	Exterminate predators using:	1. Districtwide +	3. 120 000 - 1 200 000	1. 0 - 12 000	0.44	Small prey -

ID	Method	Limiting factors	Implement Cost (N$)	Maintain Cost (N$)	p (Effect) p(Sustain)	Influencing factors
	Hunting Dogs Predator-proof fencing **Proactive:** (Only combination with historical success against jackals).	2. Threatened predator - 3. Natural prey - 4. Predator-proof fencing + 5. Reactive: dogs + 6. Terrain flat and open +			(0.01)	Jackal & Caracal - Dog training time - Fences surplus killing by felids -
2.	Kill on sight	1. Habituated predators -	1. 0 - 12 000	1. 0 - 12 000	0.35 (0.35)	Wrong individual(s) killed -
3.	Trapping for extermination or reducing numbers	1. Large farm - 2. Extensive farming - 3. Districtwide + 4. Threatened predator -	1. 0 - 12 000	2. 12 000 - 120 000	0.39 (0.06)	District wide + Jackal -
4.	Herding Kraaling (nights) Livestock Guarding dogs	1. Finding herders (p=0.3 for ecoranger project) - 2. Large dogs breeds available	2. 12 000 - 120 000	2. 12 000 - 120 000	0.68 (0.8)	Grazing (trampling) - Herder own livestock + Training time -
5.	Kraaling (nights) Young kraaled permanently Seasonal breeding (required for kraaling young)	1. Predator-proof kraal + or LGD + 2. Extensive farming - 3. Seasonal breeding + 4. Seasonal livestock losses + 5. Wild prey +	2. 12 000 - 120 000 1. 0 - 12 000	2. 12 000 - 120 000 1. 0 - 12 000	0.53 0.6 (0.6)	Goats + Cattle + Felid predators (surplus killing)- Grazing (trampling) - Jackal + Wild prey + Seasonal veld - Birth % -

ID	Method	Limiting factors	Implement Cost (N$)	Maintain Cost (N$)	p (Effect) p(Sustain)	Influencing factors
6.	Predator-proof breeding camps Lambs, kids or calves kept in small, camps with predator-proof fencing, including electrified fencing if required Repellents in camp Repellents on fences (fladry)	1. Terrain flat and open + 2. Extensive farming - 3. Enough grazing near houses+	3. 120 000 - 1 200 000	2. 12 000 - 120 000	0.64 (0.6)	Grazing (trampling) - High maintenance -
7.	Repellent collars (E-shepherd)	1. Cattle - 2. Breeds in small herds - 3. Wild prey + 4. Extensive farming -	2. 12 000 - 120 000	2. 12 000 - 120 000	0.5? No data (0.8)	Growing lambs -
8.	Protective collars	1. Extensive farming - (Labour intensive)	2. 12 000 - 120 000	2. 12 000 - 120 000 (Labour)	0.5 (0.75)	Growing lambs - Jackal - Labour scarce -
9.	Conditioned Taste Aversion	1. Cheetahs - 2. Untreated livestock carcasses - 3. Wild prey + 4. Terrain not too dense or rugged + 5. LiCl available + 6. Killing / Removing predators - 7. Very large farm -	1. 0 - 12 000	1. 0 - 12 000	0.5 Unknown (0.8)	Training for correct dosages - Inexpensive +
10.	Natural repellents against predators protecting small camps	1. Finding enough scat & urine - 2. Small area +	1. 0 - 12 000	1. 0 - 12 000	0.5 +Breed camps↓	Labour scarce - Longterm +

ID	Method	Limiting factors	Implement Cost (N$)	Maintain Cost (N$)	p (Effect) p(Sustain)	Influencing factors
		3. Removing predators -			0.69 (0.8)	
11.	Natural prey High biodiversity	1. Transformed landscape - 2. Very low game stocking rate -	2. 12 000 - 120 000	1. 0 - 12 000	0.62 (0.9)	Few game left - Combined + Game-proof fencing -
12.	Guarding animals (Donkeys, Alpacas Llamas) (LGD after year with herder)	1. Predator > Cheetah - 2. Agressive individual + 3. Moderate herd size (<200) + 4. Guarding animals availability - 5. Bonding & training + 6. Goats/sheep (dogs) + 7. Mesopredators (alpacas) +	1. 0 - 12 000	1. 0 - 12 000	0.57 (Donkeys↑) 0.91 (Dogs↑) (0.8)	Seasonal breeding + Other methods combined +
13.	Depredation hotspots	1. Heterogeneous habitat + 2. Known habitat preferences +	1. 0 - 12 000	1. 0 - 12 000	0.32 (0.9)	As reactive method + Known hotspot +
14.	GPS collar on predators with daily update to farmers	1. Mesopredators -	2. 12 000 - 120 000	1. 0 - 12 000	0.64 (0.85)	Predators removed - Biodiversity + Combined +
15.	Change livestock breed, species or age (buy weaners and raise them) Horned cattle	1. Availability of livestock to buy + 2. Wild prey + 3. Grazing too short/sparse for cattle 4. Feedlot market -	1. 0 - 12 000	1. 0 - 12 000	0.5 0.68 (species) 0.29 (0.5)?	Time to implement - Breed livestock - Adaptive + Cash-flow - Horned breeds +
16.	Holistic grazing	1. Arid -	3. 120 000 - 1 200 000	2. 12 000 - 120 000	0.44	Mixed livestock +

ID	Method	Limiting factors	Implement Cost (N$)	Maintain Cost (N$)	p (Effect) p(Sustain)	Influencing factors
	Big mixed herds Small camps	2. Removing predators - 3. Intensive management +			(0.7)	Production - Existing small camps +
17.	Trophy hunting of predators Tourism (combine with 14.)	1. High losses, many predators - 2. Problem individual (old male) +	1. 0 - 12 000	1. 0 - 12 000	0.85 (0.8)	Biodiversity +
18.	**Compensatory:** Compensation Price premiums Insurance	1. Region-wide / Country-wide + 2. NAU +	1. 0 - 12 000	1. 0 - 12 000	0.64 (0.7)	
19.	Ecotourism Hunting game Combine with 11. & 14.	1. Can track + 2. Infrastructure (capital) + 3. Far from main roads -	3. 120 000 - 1 200 000	2. 12 000 - 120 000	0.68 (0.7)	Farmer disinclination for tourism - Infrastructure not for tourism -
20.	Translocate **Reactionary:** Hunting Hunting dogs Trapping	1. Problem individual + 2. Available release space - 3. Release location > 200 km +	2. 12 000 - 120 000	1. 0 - 12 000	0.44 (0.6)	

5.3.1 The Decision Support System

Any algorithm needs to be implemented in a full DSS for it to become useful to farmers and wildlife managers dealing with HWC. The DSS will basically consist of four parts (Figure 5.2): 1) Input about the farm; 2) Prior information; 3) The algorithms for calculating the most appropriate methods (best chance of success given current livestock losses and situation); 4) Output showing the preferred mitigation method, its pros and cons and two alternative methods (2nd and 3rd most likely to succeed).

1. Input: As a front-end, a web application that farmers can access from a smartphone or on their personal computer, is considered as the best option. The major reason for this choice, rather than a desktop application or phone app, is that it enables all users to always have access to the latest and up-to-date version of the program. It also makes updates to the DSS program easier compared to the alternatives. Having the farmer or manager select the farm(s) on a GIS (with pre-existing layers for topography,

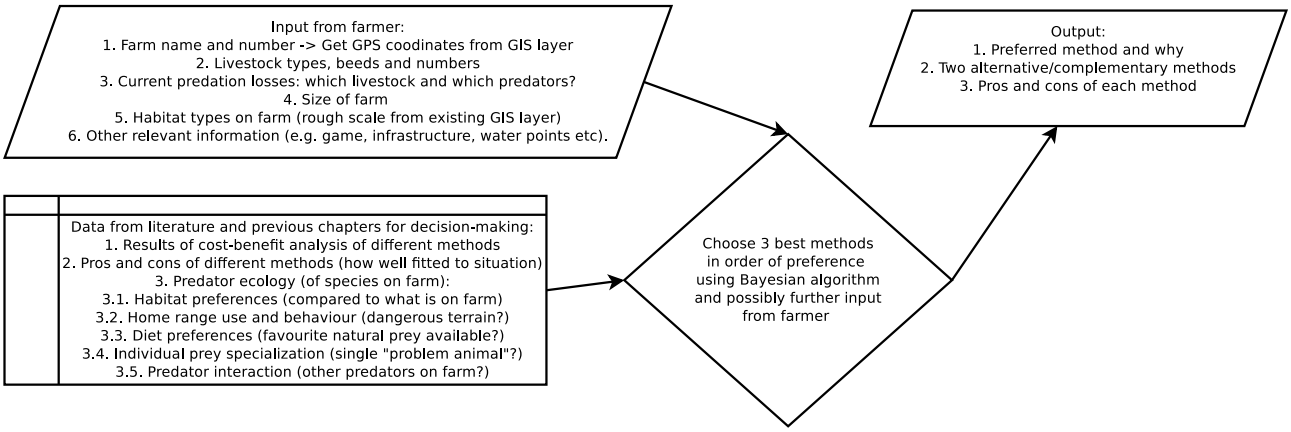


Figure 5.2: *Basic design of the Decision Support System (DSS).*

vegetation etc.), will alleviate the need for the farmer to input all the information manually. Alternatively, if the farmer is using a smartphone with built-in GPS, this information can be learned from the phone.

2. Prior information: Table 5.1 gives a summary of the prior information that will be combined with farmer input to build the model.

3. The actual processing of the data (both input and prior) can be broken up into a two-step algorithm. It should be obvious that if a certain method is not practical on a specific farm because of its limitations (e.g. using Livestock Guarding Dogs is not a practical solution if no dogs are available to be trained or if there is nobody to train the dog), there is no need to continue calculating its cost-effectiveness and probability of success — it already has a 0 probability of success for that specific farm. Therefore, the first step of the DSS processing stage, is the elimination of all those mitigation methods that are known to be impractical in a specific agroecological system. The second step is to run basically the same PGM for each of the remaining mitigation methods, resulting in a Maximum Expected Utility value for each action (mitigation method chosen to implement).

4. Output. The aim of the DSS (unlike some other expert systems), is not to take a decision, but to present the decision maker (actor) with the most relevant information for deciding which mitigation method(s) to use. Some factors included in the model that are likely to influence the decision includes: 1) reduction in livestock losses and thus an increase in production and profit; 2) costs of implementing and maintaining the mitigation method (and financial loss if it is unsuccessful); 3) the risk of an increase or no effect on livestock losses and of financial loss – alternatively, the probability of a favourable outcome in the first two aspects; 4) probability of the long-term sustainability of the method chosen (maximising biodiversity on the farm, and reducing the extirpation risk of wildlife species). However, there might be other factors specific to a decision maker which would not be included in a general model such as this. By providing the pros and cons of the three recommended methods, (and which one has the maximum expected utility) the farmer can make a decision based on the specific farm circumstances.

To keep the whole model transparent, easy to check and easy to update and extend, it is proposed that such an implementation has to be Open Source, with the source code freely available. For the actual probabilistic model implementation, it is proposed to use the pgmpy Python library module, which was written for creating PGMs (Ankan and Panda, 2015).

5.3.2 The elimination algorithm

Based on user input, the DSS can eliminate some of the mitigation methods as impractical on a specific farm using the following algorithm:

1. Input **Farm** name (and district) or select farm on GIS.
   (a) Determine farm **Size** (from GIS or ask user)
      i. **Large farm** (> 10 000 ha – Unable to clean farm from all untreated livestock carcasses)?
         A. YES: Eliminate **Method 9**
   (b) Determine **Vegetation** on farm from GIS (Biome, Vegetation type, Vegetation structure)
   (c) Determine average **Rainfall** of farm from GIS
   (d) Determine **Terrain** on farm from GIS (Mountains, Rocky hills, Drainage lines / rivers, Flat plains, Rolling hills) and user input
      i. Is the **Habitat** (Vegetation & Terrain) mostly the same (**homogeneous**)?
         A. YES: Eliminate **Method 13**
      ii. Is the Terrain very **rough** and/or the Vegetation very **dense?**
         A. YES: Eliminate **Methods 1, 6, 9**
2. Input current **Farm Infrastructure**: Water points & Fences (Stock-proof, Game, Electrified, Jackal-proof, Predator-proof kraals).
   (a) Has or can put up **jackal-proof fencing** (for jackals) or **electrified fencing** for camps?
      i. NO: Eliminate **Methods 1, 6**
   (b) Has or can put up **predator-proof kraals**?
      i. NO: Eliminate **Methods 4, 5** (unless using Livestock Guarding Dogs)
3. Input **Predator species** on farm.
   (a) **Cheetahs and/or Leopards**? (Threatened species)
      i. YES: Eliminate **Method 1, 3** (if trapping to kill or traps can injure animal)
   (b) Access to **scat** and/or **urine** of **dominant predator** species?
      i. NO: Eliminate **Method 10**
   (c) Predators **previously hunted** and still survived on farm?
      i. YES: Eliminate **Method 2**
4. Input **Livestock** species, breeds, age classes and numbers of each.
   (a) **Cattle only**?
      i. YES: Eliminate **Methods 4, 7, 12 (Dogs)**
   (b) Cattle sold primarily to **feedlots**?
      i. YES: Eliminate **Method 15 (horned cattle)**
   (c) Willing and able to **Change Livestock Species** or **Breed**?
      i. NO: Eliminate **Method 15**
   (d) Livestock in **large herds** (>200 and not practical to change)?
      i. YES: Eliminate **Method 12**
   (e) Livestock breed tends to **scatter** into **small herds** < 10?
      i. YES: Eliminate **Methods 7, 12**
5. Input how often livestock is counted to determine **Farming Production System**: (At least every 2 days: Intensive farming, Every week: Medium intensive farming, Every month: Medium extensive farming, Less than once a month: Extensive farming).

(a) **Extensive farming**?

i. YES: Eliminate: **Methods 3, 5, 6, 7, 8, 10**

(b) Reliable **Labour Available**?

i. NO: Eliminate: **Methods 4, 5 (with LGD), 12 (Dogs)**

ii. YES: Allowed to keep **own livestock** on farm?

A. NO: Eliminate **Method 4**

6. Input **Current Livestock Depredation** for each livestock species and age class and each predator species.

(a) Does Livestock Depredation happen **seasonally**?

i. NO: Eliminate **Method 5**

(b) Input **Predator(s)** responsible for **Livestock Losses** (Mostly lambs: probably jackals, Small livestock only: jackals and/or caracals, Small livestock and calves $<$ 6 months: cheetahs and leopards, Calves $>$ 6 months: leopards).

i. **Cheetahs**?

A. YES: Eliminate **Method 9**

ii. **Jackals**?

A. YES: Eliminate **Method 2**

B. NO: Eliminate **Methods 1 (jackal-proof fencing), 6 (jackal-proof fencing)**

iii. **Leopards**?

A. YES: Eliminate **Method 12 (Donkeys / Alpacas)**

iv. **Jackals and caracals only**?

A. NO: Eliminate **Methods 12 (Alpacas)**

B. YES: Eliminate **Method 14**

v. Predator **habitat preference** in habitat (from Input 1.) **unknown**?

A. YES: Eliminate **Method 13**

(c) **Individual predator** suspected of habitually killing livestock?

i. NO: Eliminate **Method 20**

7. Input current state of **Grazing** (veld condition) on the farm (majority palatable grasses, trampling, bush encroachment, transformed veld – sown grasses).

(a) **Trampled veld** around kraals or water points a problem?

i. YES: Eliminate **Methods 5, 6, 10**

(b) **Transformed veld** (artificial grazing with little natural vegetation)?

i. YES: Eliminate **Method 11**

8. Input **Game** numbers and species on farm (now *or planned*).

(a) More than **10 game species**?

i. YES: Eliminate **Method 1**

(b) Less than **5 game species**?

i. YES: Eliminate **Methods 5 (Seasonal), 9, 11, 14, 15**

(c) **Game density** above 812 kg/km^2 (for cattle) or 544.57 kg/km^2 (for small livestock)?

i. YES: Eliminate **Method 1**

9. Input current **Farm Management.**
   (a) **Seasonal breeding** (and livestock species)?
      i. NO: Eliminate **Method 5**
   (b) **Trained hunting dogs** (trained to follow tracks from a kill site only and not kill any small game or other predators)?
      i. NO: Eliminate **Method 1 (Dogs)**
   (c) Input current livestock **Predation Management** used with an annual cost estimate.
      i. Eliminate **Methods already used**.
10. Input **Available Material and Livestock**.
   (a) **LiCl** (Lithium Chloride) available?
      i. NO: Eliminate **Method 9**
   (b) Enough **newly weaned livestock** for sale locally?
      i. NO: Eliminate **Method 15 (change age)**
   (c) **Donkeys/Alpacas available** for sale?
      i. NO: Eliminate **Method 12 (donkey/alpacas)**
   (d) **Large dogs** available (Anatolian/Kangal/Ridgebacks)?
      i. NO: Eliminate **Methods 4, 12 (Dogs)**
11. Input **Cooperation** (with government, NAU and other farmers).
   (a) Will cooperate for **district-wide extermination** of predator species?
      i. NO: Eliminate **Method 1, 3 (exterminate)**
   (b) Will cooperate on national marketing level to charge **price premium** on "predator-friendly" meat?
      i. NO: Eliminate **Method 18**
   (c) Member of **NAU**?
      i. Can establish country-wide **insurance scheme** for predation losses?
         A. NO: Eliminate **Method 18**
      ii. Would NAU set up **national compensation scheme** for predator losses?
         A. NO: Eliminate **Method 18**
   (d) Member of **Conservancy**?
      i. YES: Would conservancy (or CANAM) set up **local/regional compensation scheme** for predator losses?
         A. NO: Eliminate **Method 18**

5.3.3 The PGM

The Probabilistic Graphical Model (Figure 5.3) will form the heart of the DSS. When comparing it to our first conceptual Bayesian Network (Figure 5.1), it can be seen that the whole inner trail of probabilities (to the "Practical?" node) has been removed from the PGM and put in the preceding elimination algorithm. The resulting Decision Network (Influence Diagram) starts with a single Decision Node with all the methods (excluding the previously eliminated ones) as its decision variables. It has a decision rule of simply choosing every practical (non-eliminated) method one by one.

Most of the input required for determining the most appropriate method of mitigating predator conflict is already provided by the input as described above for the elimination algorithm, yielding 25 input variables in total. However, for any chosen method, a maximum of eight farmer inputs will be used to change its Expected Utility from that of the prior Expected Utility (calculated from the general prior probability of effectiveness as calculated in Table 5.1). However, in addition to the input data provided by the user, seven more questions are required to evaluate the different methods:

1. Do you have the time (and a reliable herder) to train a Livestock Guarding Dog?
2. Do you have any problems with surplus killings (i.e. where a predator uses a fence to kill multiple livestock in one event, and only eats one – *cf.* Linnell et al., 1996)?
3. Are you willing to attend training to learn about Conditioned Taste Aversion and how to apply it effectively to keep predators from killing livestock?
4. Do you know of any predation hotspot areas on your farm where predators are more common or more likely to kill livestock, from past experience?
5. How critical is the implementation time before a method starts to work for you? Would you use methods that will only become effective after a year?
6. Are you willing to cater for tourists as an extra source of income?
7. Do you have the necessary infrastructure (buildings, water, electricity etc.) allowing tourism to be financially profitable?

Livestock losses as a percentage of offspring were classified in four classes and each class was assigned a subjective utility value (u): 1) $<$ 3% = Low losses, u = -5; 2) 3 – $<$10% = Medium losses, u = -10; 3) 10 - 20% = High losses, u = -30; 4) $>$ 20% = Very high losses, u = -50. The cut-off of 3% for low losses was chosen as the rounded off median percentage of calves lost to predators from our previous farmer survey (the mean was a much higher 9.66%). This value approximates the findings of Stein et al. (2010) showing that Namibian farmers will tolerate losses of up to 3.3% of their calves to predators. The utility values are all negative, expressing the intuitive sense that livestock loss has a negative usefulness. Each method was also assigned a cost class (see Table 5.1) which was also given similar utility values: 1) N$0 - N$12 000: -5, 2) N$12 000 - N$120 000: -10, 3) N$120 000 - N$1 200 000: -30, 4) $>$ 1 200 000: -50. If the infrastructure does not already exist, the implementation cost class is used, otherwise the maintenance cost class is used. The actual utility of a method for a specific farmer is the *change* in current losses (and in costs) when switching to that method. This change (new utility - current utility and new cost class utility - current cost class utility) results in a change utility that ranges from -45 (worse case) to 45 (highest utility) for each of the costs and losses. Sustainability has a binary value of true or false with associated utility values of 100 and -100. The final Expected Utility of each method consists of the sum of (the product of the utility of each possible outcome and the probability of that outcome):

$$EU = \sum P(\varsigma)\, U(\varsigma)$$

The Maximum Expected Utility (MEU) is the EU value from the method choice of the highest expected utility. For each farmer the expected utility is first reset to 0 given the inputs from the farmer, indicating the choice of changing nothing. For this null model, the expected costs become the current maintenance costs; the probability of effectiveness become the probability of the current method used with the highest prior p(Effective); expected livestock losses become the current losses; expected sustainability becomes the p(sustainable) of the current method with the lowest sustainability.

A PGM does not only consist of a graphical Bayesian decision network, but each node also has a table with the conditional probability distribution (CPD) for the node given its parents. Those nodes without parents, either have a prior probability or require input from the user. The CPDs for the final MEU probabilities are linked

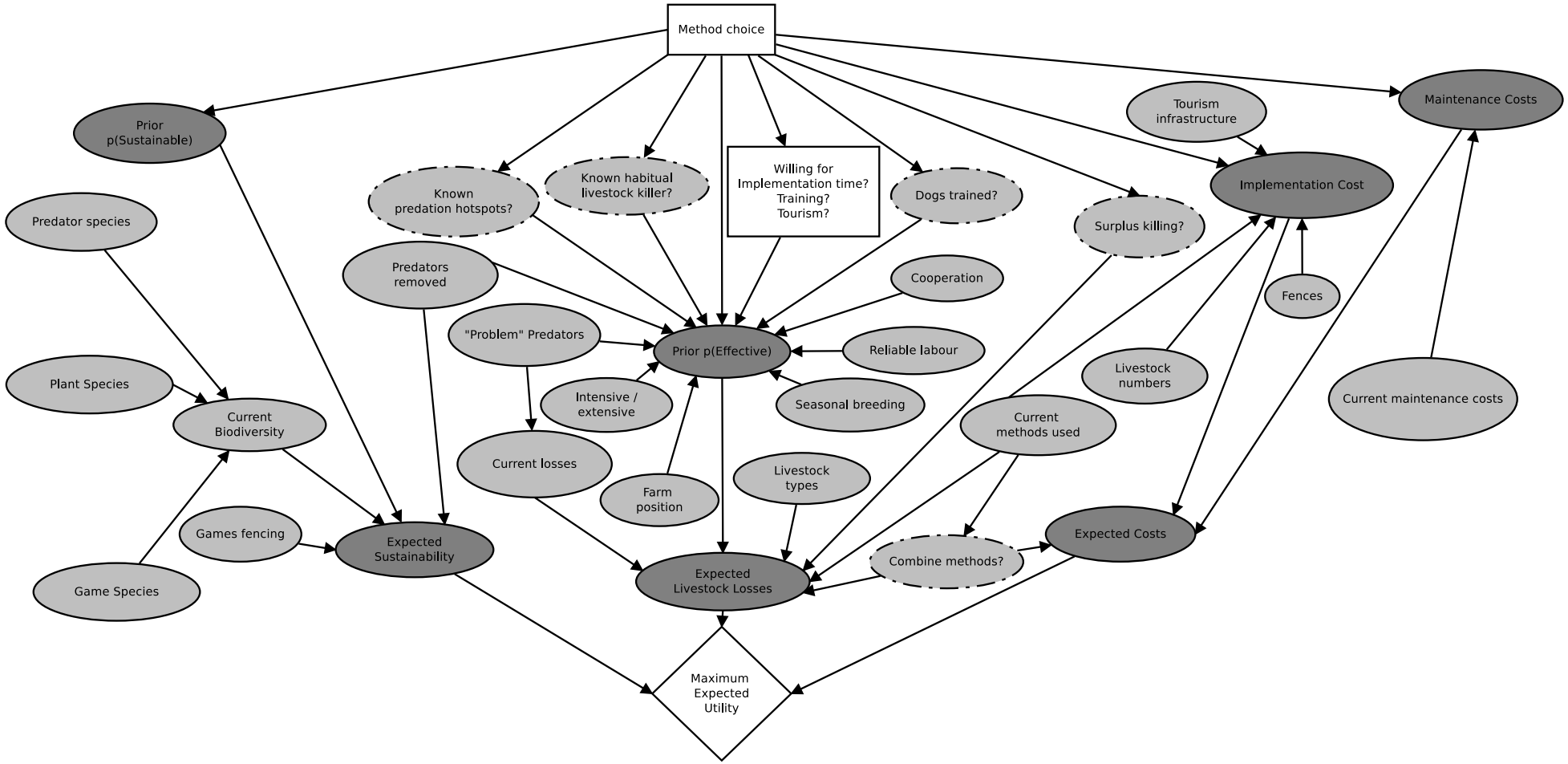


Figure 5.3: *The decision network (influence diagram) PGM for evaluating predator-farmer conflict mitigation methods on Namibian farmlands.* Decisions are indicated by two rectangular nodes (one optional depending on the chosen method). The nodes with prior values and specific Conditional Probability Diagrams (the invariant part of the model) are shown as dark grey ovals and the farm information requiring input by the farmer, as light grey ovals. Extra information required from the farmer for specific methods only, has a dashed line around the oval. The (Maximum) Expected Utility is indicated by a diamond shaped node.

together using the chain rule (generalisation of Bayes rule). The CPDs for the model are given below in section 5.3.5.2 with a step-by-step calculation of the MEU for one method.

5.3.4 Constraints and limitations

There are two sources of error in a model of this kind (Koller and Friedman, 2005):

1. The model itself might not adequately or accurately capture the system or question it wanted to model (i.e. it is not a correct reflection of our domain knowledge). Some of the behavioural ecological data of predators and their natural prey on Namibian farmlands, are simply not known at the moment. For instance, no studies on jackals have been published and only one on caracals. So, it is possible that some vital information, in particular about the interaction between the various predators, has not been included in the model which could have an influence on its accuracy. Such unknowns should be explicitly and transparently provided to the user as part of the final output of the DSS.

2. The approximations used in the model introduce overly large errors. Specifically, the actual costs of most methods are not known (even by the farmers using them). The approximation using cost classes instead of actual costs, may produce results that are too uncertain to be useful. It is also possible that the classes are simply too large and a larger number of smaller classes may provide results that are good enough for decision making under uncertainty. However, the use of large classes also makes the model more robust to "garbage data" that could foul the model (GIGO – Gelman, 2011). It should also make the model more robust to changing circumstances (e.g. price changes and inflation).

The current model was specifically designed for Namibian farms. The basic structure of the model, including Expected Sustainability, Expected Losses (change in losses) and Expected Costs (change in costs) in order to find the Maximum Expected Utility, would remain the same for other species and situations (e.g. more predator species, communal farmers, other continents and biomes). However, the actual probabilities and utility values will be unique for each situation. What the model provides us with in these cases, is a baseline from which the most important directions and information required for future research on HWC can be determined.

5.3.5 The model in action:

Here we will step through the algorithm of the PGM for one iteration (one method), demonstrating how the DSS will calculate the Maximum Expected Utility.

Scenario
"I am a farmer in Southern Namibia, farming with cattle and goats. My farm is 16 000 ha, on which I keep about 150 head of cattle and 200 goats. I farm extensively and only gather the cattle once every three months, but the goats sleep in a kraal every night. I implement no other mitigation methods, except for occasionally setting traps to catch predators. My farm is mountainous and I have leopards and caracals that killed about 10 goats and 1 calf in the past year. What are the best method(s) to prevent livestock losses?"

5.3.5.1 Elimination algorithm

The farmer answers the DSS questions as follows:

1. Input **Farm** name (and district) or select farm on GIS. Farmer response:

   (a) Large farm

   (b) Nama Karoo short shrubs & grass, riverbeds
   (c) $<$ 200 mm rain
   (d) Mountains, rocky hills & drainage lines

2. Input current **Farm Infrastructure**: Water points & Fences (Stock-proof, Game, Electrified, Jackal-proof, Predator-proof kraals). Farmer response:
   (a) No jackal-proof fencing
   (b) One Predator-proof kraal

3. Input **Predator species** on farm. Farmer response:
   (a) Leopard
   (b) No urine or scat available
   (c) Not hunting

4. Input **Livestock** species, breeds, age classes and numbers of each. Farmer response:
   (a) Cattle (150 cows, 120 calved) & goats (200 does, 200 kids)
   (b) Not feedlots
   (c) Not willing to change breed or species
   (d) Large herds, but can change
   (e) Scatter

5. Input date of last Livestock Count to determine **Farming Production System**: ($<$ 2 days ago: Intensive farming, Within past week: Medium intensive farming, Within past month: Medium extensive farming, More than a month ago: Extensive farming). Farmer response:
   (a) Extensive
   (b) Reliable labour with own livestock (few)

6. Input **Current Livestock Depredation** for each livestock species and age class and each predator species. Farmer response:
   (a) 20 goats, 1 calf: Not seasonal
   (b) Caracals and leopard (mostly leopard)
   (c) No individual problem predator suspected

7. Input current state of **Grazing** (veld condition) on the farm (majority palatable grasses, trampling, bush encroachment, transformed veld – sown grasses). Farmer response:
   (a) Trampled veld
   (b) Only natural

8. Input **Game** numbers and species on farm (now *or planned*). Farmer response:
   (a) Many kudus, klipspringers and dassies. Fewer steenbok. Would like to increase.

9. Input current **Farm Management.** Farmer response:
   (a) Seasonal breeding (goats)
   (b) No hunting dogs
   (c) Trapping: N$ 0 - N$ 12 000

Table 5.2: *Prior p(Sustainable).* p(S) is the probability that the outcome will be ecologically sustainable (from Table 5.1), given the choice of the method number above. $p(notS) = 1 - p(S)$ is the probability that the outcome will be unsustainable in the long run (given the choice of method in the top row).

Method	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
p(S)	0.01	0.35	0.05	0.8	0.6	0.6	0.8	0.75	0.8	0.8	0.9	0.8	0.9	0.85	0.5	0.7	0.8	0.7	0.7	0.6
p(not S)	0.99	0.65	0.95	0.2	0.4	0.4	0.2	0.25	0.2	0.2	0.1	0.2	0.1	0.15	0.5	0.3	0.2	0.3	0.3	0.4

10. Input **Available Material and Livestock**. Farmer response:
    (a) No LiCl
    (b) No weaners for sale
    (c) Donkeys available
    (d) Large dogs available
11. Input **Cooperation** (with government, NAU and other farmers). Farmer response:
    (a) No cooperation for extermination
    (b) No market for "predator-friendly" meat
    (c) NAU member
    (d) Not conservancy

This will eliminate Methods 1, 3 (already used & ineffective), 5 (for cattle), 6, 7, 8, 9, 10, 12 (for cattle), 15, 18, 20; leaving Methods 2 (opportunistic shooting of predators), 4 (Herding with kraaling and/or LGDs), 5 (kraaling young, for small livestock), 11 (Increase natural prey), 12 (Guarding dogs), 13 (Avoiding depredation hotspots), 14 (GPS collared predator), 16 (Holistic grazing in mixed herds), 17 (Trophy hunting with tourism) and 19 (Ecotourism) — 10 methods in total to test.

5.3.5.2 PGM algorithm

1. The PGM will step through the remaining methods (that were not eliminated by the elimination algorithm in Step 1) one by one and determine the MEU for *each,* given our input. For this demonstration we will only step through one method. Since Method 1 is already excluded by the elimination algorithm (Section 5.3.2), we use the next Method (Method 2 in Table 5.1) for which we will determine the MEU. This HWC mitigation method is one of the most common (and cheapest) methods currently used on Namibian farms: Killing predators on sight, but not actively hunting them.

CPDs for each node: The Conditional Probability Distribution tables:

In our case, we chose method 2, so we have a prior probability of 0.35 that it will be sustainable, and a probability of 0.65 that it will be unsustainable (Table 5.2).

Our other prior table (Prior p(Effective) from Table 5.1) will look similar to Table 5.2, but with more variables influencing it. Since we already know that method 2 has been chosen, we can ignore the prior effective probability value of the other methods. We also know that intensive/extensive, farm position, seasonal breeding, reliable labour, cooperation, trained dogs, and known predation hotspots will have no effect on the effectiveness of this specific method. However, the "wrong" (innocent) predators being killed, will make this method less effective. If there was a single known habitual livestock killer, then the chances of killing that specific predator by chance is low (given the known home range sizes of the predators) and it would increase the probability that we may kill the wrong predator (Table 5.3 and 5.4). In our simulated case, we do not know of any habitual livestock

Table 5.3: *Part of the CPD for prior p(Effective) given choice of method 2* $p(E|M \bigcap prPred)$. Given method 2, we also know from Table 5.1 in Chapter 6 that it is influenced negatively by the probability of the wrong predator (one that does not currently kill livestock) being killed. Where there is no problem predator, the probability of being effective (E1) is equal to the prior probablity from Table 5.2. [prPr1 = probability of there being a "problem predator"; m2PrPr1 = Method 2 with a Problem predator]

prPr0	prPr1
0.89	0.11

Method & prPred	p(Effective) E1	p(Ineffective) E0
m2prPr0	0.35	0.65
m2prPr1	0.3	0.7

Table 5.4: *p(Effective) joint distribution.* The joint distribution p(E, M, prPr) = p(prPr).p(E|M,prPr). This is the probability for any specific combination of outcomes for E (effectiveness), M (method) and prPr (problem predator). Each variable has only one of two outcomes (binary), *True* of *False.*

Joint distribution	prPr	pE|M.prPr	E,M,prPr	
E0.m2.prPr0	0.89	0.65	0.5785	E0:
E0.m2.prPr1	0.11	0.7	0.077	0.6555
E1.m2.prPr0	0.89	0.35	0.3115	E1:
E1.m2.prPr1	0.11	0.3	0.033	0.3445
				1

Table 5.5: *Utility classes.* Instead of using actual values, outcomes and possible changes in outcomes are coded as classes representing utility. Livestock Loss is given as percentage of annual livestock births, cost class, as annual cost in Namibian dollars (maintenance and implementation), and sustainability as True (1) or False (0). MEU is the sum of the probabilities of each outcome with the highest total utility.

LS Loss %	Loss Utility	Cost class (N$)	Cost utility	Sustainability	Sust. utility
0	0	0	0	0	-100
0 - 3 %	-5	0 - 12 000	-5	1	100
3 - 10 %	-10	12 000 - 120 000	-10		
10 - 20 %	-30	120 000 - 1 200 000	-30		
> 20 %	-50	> 1 200 000	-50		

killers and must give a prior probability to there being a habitual livestock killer. About 4 out of 35 collared leopards made more than one livestock kill after their release (Chapter 4) which gives a marginal p = 0.11 (of there being a "problem animal" on the farm and probability of 0.89 of there being no problem animal – first part of Table 5.3).

Both our current method (trapping) and the method we are investigating (M2, shoot on sight without active hunting) are in the lowest cost class (Table 5.5), so the chances of a change to either higher or lower cost when using the new method are very low. However, we could use both methods, meaning that implementing Method 2 can shift us to a higher cost class (e.g. the price of a new rifle). The various utility classes used (for livestock losses, costs and sustainability) are shown in Table 5.5. The costs for Method 2 (compared to the current situation) and its estimated expected costs are shown in Tables 5.6 and 5.7.

The expected livestock losses (Table 5.8) are actually the expected change in livestock losses as a result of using the new method. It gives a probability (from p(Effective)) of each outcome. Current livestock losses (20 from 200 kids and 1 from 120 calves) is 10.8% of the offspring (gross farm income).

Expected sustainability (Table 5.9) are influenced by the prior p(Sustainable), increased by biodiversity (here simply taken as species richness of predators and game), decreased by game fencing (preventing migratory game movements), and decreased if predators are removed (Table 5.10).

The final expected utility is given by the sum of the three utility values (see Figure 5.3). For Method 2 in this scenario: $ExpectedUtility = CostUtility + EffectiveUtility + SustainUtility$

$= (0.3 + -0.2) + (-6.555 + 2.2967 + 2.871 + 3.445) + (-67 + 33) = -31.8423$

Table 5.6: *Implementation cost.* Implementation is basically the new mitigation method cost minus the cost of the current mitigation method (assuming that the farmer will stop the current method). The same formula is used for the maintenance cost. Instead of using the actual cost, we use the cost utility. For each outcome, there is also an associated probability. Because the price difference of the two methods are similar, the possible changes (New - Old) are limited. Here, there are basically three outcomes: 1) The farmer, already having a rifle and enough bullets for a whole year, simply keeps them in his vehicle when driving and implementing as well as maintaining the new method costs nothing (p = 0.01), 2) the farmer spends some money to implement the method, but basically in the same price class as his current method of trapping (p = 0.95), 3) the farmer buys an expensive new rifle, drives more on the farm and implementation costs are slightly higher than the current method (p = 0.04). The options are basically the same for maintenance, but with slightly different probabilities (e.g. small chance of costing more than N$12 000 per annum (Table 5.5). While estimates are made here about the probabilities of each scenario, this information could also be provided by the farmer.

New cost	Current Cost	New-Current	p (implement)	p (maintain)
0	-5	5	0.01	0.05
-5	-5	0	0.95	0.95
-10	-5	-5	0.04	0
-30	-5	-25	0	0
-50	-5	-45	0	0

Table 5.7: *Expected cost.* Expected cost is simply the utility of implementation + maintenance costs. Expected utility (is the product of utility and probability).

Implement util.	p(impl)	Expected util	Maintenance util	p(maint)	Expect.util	Expected cost
5	0.01	0.05	5	0.05	0.25	0.3
0	0.95	0	0	0.95	0	0
-5	0.04	-0.2	-5	0	0	-0.2
Total (expected cost):						0.1

Table 5.8: *Expected livestock losses.* Each possible outcome (change) in livestock losses are given a probability based on p(Effective) of the method. We use the p(Effective) from the joint distribution and distribute p(ineffective) equally among the ineffective (change in utility <= 0) and p(effective) among the effective outcomes. Expected livestock loss utility = Utility * p.

New	Current	New - Current	p	Expected LS loss utility
-50	-30	-20	0.32775	-6.555
-30	-30	0	0.32775	0
-10	-30	20	0.11483	2.2967
-5	-30	25	0.11483	2.871
0	-30	30	0.11483	3.445
			1	

Table 5.9: *Conditional expected sustainability.* $p(eS|pR\bigcap gF\bigcap S)$ Predators removed (pR) and game fencing (gF) decrease probability of sustainability. Prior p(Sustainable) from Table 5.2 above (S1 & S0).

S1	S0
0.35	0.65

	eS0	eS1
S0pR0gF0	0.65	0.35
S0pR0gF1	0.7	0.3
S0pR1gF0	0.7	0.3
S0pR1gF1	0.75	0.25
S1pR0gF0	0.65	0.35
S1pR0gF1	0.7	0.3
S1pR1gF0	0.7	0.3
S1pR1gF1	0.75	0.25

Table 5.10: *Expected sustainability joint distribution.* p(eS, S, pR, gF) = p(S).p(eS|S,pR,gF). p(eS) * 100 = Utility (see Table 5.5). Species richness is added directly to the expected sustainability utility.

	S	eS|S,pR,gF	eS,S,pR,gF	eS	Utility	+ 6 Species
eS0,S0,pR0,gF0	0.65	0.65	0.4225			
eS0,S0,pR0,gF01	0.65	0.7	0.455			
eS0,S0,pR1,gF0	0.65	0.7	0.455			
eS0,S0,pR1,gF1	0.65	0.75	0.4875			
eS0,S1,pR0,gF0	0.35	0.65	0.2275			
eS0,S1,pR0,gF1	0.35	0.7	0.245			
eS0,S1,pR1,gF0	0.35	0.7	0.245			
eS0,S1,pR1,gF1	0.35	0.75	0.2625	0.73	-73	-67
eS1,S0,pR0,gF0	0.65	0.35	0.2275			
eS1,S0,pR0,gF1	0.65	0.3	0.195			
eS1,S0,pR1,gF0	0.65	0.3	0.195			
eS1,S1,pR0,gF0	0.35	0.35	0.1225			
eS1,S1,pR0,gF1	0.35	0.3	0.105			
eS1,S1,pR1,gF0	0.35	0.3	0.105			
eS1,S1,pR1,gF1	0.35	0.25	0.0875	0.27	27	33
			3.8375	1		

The maximum expected utility (MEU) of changing to this method for the farmer in this scenario is -31.8423. The great advantage of this method is that it is very cheap, but it is neither very sustainable, nor very effective. Thus it might be an option for a farmer currently spending a lot of money on predator management with little results, but in few other scenarios.

One shortcoming of stopping the current model here, is that it does not include the sustainability utility of the *current* method(s) used, in the calculation (e.g. Method 3, eliminated in this scenario). If the current method is even less sustainable than the tested method (and equally ineffective), it might still be worthwhile to change — in this case the Expected Utility of keeping the current method: $ExpectedUtility = CostUtility + EffectiveUtility + SustainUtility$
$= 0 + 0 + (SustainUtility)$ is determined only by its sustainability. It would be even lower than the -31.8423 of Method 2 (since currently used Method 3 has a prior probability of only 0.05 of being sustainable, while the tested Method 2 has a prior probability of 0.35 of being sustainable and all other variables are the same - see Table 5.2). If no higher MEU is found for another method, it will still be worthwhile for the farmer in this scenario to switch to shooting predators on sight only (Method 2 that was tested here and had a negative EU of -31.8423), rather than actively trapping them (Method 3, currently used by the farmer).

This exact same algorithm will need to be repeated for every single method of the 10 that was not eliminated during the elimination phase and the three methods with the highest expected utility values may be presented to the farmer as the methods of choice.

5.3.6 Refining the model

Currently, the probability of a HWC mitigation method being sustainable, is based solely on a subjective estimate of its ecological impact (the more impact, the less chance of it being sustainable). As more research is being undertaken on the ecology of Namibian farmlands, this estimate can be updated to become more accurate and robust.

Every time a farmer uses the DSS, the current predation state of affairs has to be used as input. Included in this input, will be the current predation management methods used, an estimate of the costs of using the methods, and the current livestock losses (an indication of the effectiveness of those methods). Over time, this data can be used to update the current prior probabilities for the effectiveness and costs of the different methods.

More importantly, instead of the current model which uses a simplified prior probability of success (defined as lower livestock losses than the median), feedback from the farmers on the effectiveness of the methods used and eventually direct feedback on the results of a change in methods, can be used for a better probability estimate. Feedback by many farmers on methods that have been used for a sufficient time period (at least 10 years) can be used to update the sustainability probability for each method based on actual data instead of the current subjective estimate.

The program can ask the farmer, after presenting the method with the Maximum Expected Utility and the two alternatives, which method will be used and why. Analysing these data (and seeing if there are any changes in preferred methods over time), can also be used to refine the model (especially with regards to "show-stoppers" for farmers that were possibly not included in the current model).

Since the DSS will be Open Source, it can be used and improved by anybody. For this reason, the actual software has not been written as part of this dissertation, as the copyright would then belong to the University. It should be easy for other researchers and farmers to examine the model and identify any unrealistic assumptions or prior probabilities with which they disagree. More importantly, this model can serve as the basis for other Decision Support Systems involving other human-wildlife conflict situations, or even agroecological systems in other parts of the world. It is envisioned that the model itself should be shown explicitly in the DSS (as a decision network / influence diagram) and provide an easy way to suggest updates to specific values or probabilities with which a farmer or researcher disagrees, without even having to read or understand the source code (while the source code would always remain available as well). Suggested updates should also include a motivation for the change (e.g. anecdotal personal observations, new research publications or raw data).

5.4 Conclusions

Many different approaches to human-wildlife conflict have been used in the past, with partial success. Many farmers still prefer to kill predators on their land or to engage in unsustainable farming practices, without the issue being resolved. Human-wildlife conflict still remains the major cause of death for many predators, but ultimately farmers remain the custodians of predators on their land and need to be empowered to do a better job of managing their land, including both agricultural and ecological aspects of farmer-predator conflict.

An online Decision Support System (DSS) can put the relevant knowledge into the hands of farmers who are struggling with livestock depredation on their land. This is in effect using an ecologically holistic view of the farming system, looking at the whole ecosystem and not considering livestock depredation as an isolated problem (Bingham, 1997). By presenting only the top three methods with their limitations, pros and cons, the farmer will have the required information available to make informed decisions on what to do to decrease livestock losses, without being overwhelmed with irrelevant data. A DSS can also be updated periodically, making sure that it remains current. In time, the model can be expanded or adapted to include communal farming systems (Blackburn et al., 2016) and other predators, other wildlife species or other parts of the world. The major advantages of using an Influence Diagram PGM as the model used by the DSS, is that 1) it shows the causal links (Figure 5.3) used to make the decisions and can thus be understood by humans (i.e. not a black box) and 2) it can be updated with new information as the system is used, improving the prior probabilities used here (either manually, or by using machine learning techniques).

References

Allen, P., Fildes, R., 2001. Econometric forecasting. In: Armstrong, J. (Ed.), Principles of forecasting: A handbook for researchers and practitioners. Kluwer Academic Publishers, p. 45 pp.

Ankan, A., Panda, A., 2015. pgmpy: Probabilistic Graphical Models using Python. In: Proc. of the 14th Python in Science Conf. (Scipy 2015). https://www.youtube.com/watch?v=Vcmjqx7lht0, pp. 6–11.

Ascough, J. I., Maier, H., Ravalico, J., Strudley, M., 2008. Future research challenges for incorporation of uncertainty in environmental and ecological decision-making. Ecological Modelling 219 (3-4), 383–399.

Balme, G., Lindsey, P., Swanepoel, L., Hunter, L., 2014. Failure of research to address the rangewide conservation needs of large carnivores: leopards in South Africa as a case study. Conservation Letters 7 (1), 3–11.

Barber, D., 2015. Bayesian reasoning and machine learning. www.cs.ucl.ac.uk/staff/D.Barber/brml.

Beinart, W., 1998. The night of the jackal: sheep, pastures and predators in the Cape. Past & Present 158, 172–206.

Benyon, M., Curry, B., Morgan, P., 2000. The Dempster-Shafer theory of evidence: an alternative approach to multicriteria decision modelling. Omega 28, 37–50.

Bingham, S., 1997. Grassroots restoration: holistic management for villages. Savory Center.

Blackburn, S., Hopcraft, J., Ogutu, J., Matthiopoulos, J., Frank, L., 2016. Human-wildlife conflict, benefit sharing and the survival of lions in pastoralist community-based conservancies. Journal of Applied Ecology 53, 1195–1205.

Blackwell, B., Devault, T., Fernández-Juricic, E., Gese, E., Gilbert-Norton, L., Breck, S., 2016. No single solution: application of behavioural principles in mitigating human-wildlife conflict. Animal Behaviour 120, 245–254.

Bothma, J. P., 2012. Literature review of the ecology and control of the black-backed jackal and caracal in South Africa. Tech. rep., CapeNature.

Bothma, J. P., Le Riche, E., 1994. Scat analysis and aspects of defecation in Northern Cape leopards. South African Journal of Wildlife Research 24 (1/2), 21–26.

Bullinaria, J., 2005. IAI : Treatment of uncertainty. http://www.cs.bham.ac.uk/ jxb/IAI/w10.pdf.

Caughley, G., 1976. Wildlife Management and the dynamics of ungulate populations. In: Coaker, T. (Ed.), Applied Biology Vol.1. Academic Press, New York, NY, pp. 183–246.

Charniak, E., 1991. Bayesian networks without tears. AI Magazine 12 (4), 50–63.

Cobb, B., Shenoy, P., 2003a. A comparison of Bayesian and Belief Function reasoning. Information Systems Frontiers 5 (4), 345–358.

Cobb, B., Shenoy, P., 2003b. A comparison of methods for transforming belief function models to probability models. In: Nielsen, T., Zhang, N. (Eds.), Symbolic and quantitative approaches to reasoning with uncertainty. Springer, pp. 255–266.

Comber, A., Carver, S., Fritz, S., McMorran, R., Washtell, J., Fisher, P., 2010. Different methods, different wilds: Evaluating alternative mappings of wildness using fuzzy MCE and Dempster-Shafer MCE. Computers, Environment and Urban Systems 34 (2), 142–152.

Conradie, B., Piesse, J., 2013. The effect of predator culling on livestock losses: Ceres, South Africa, 1979 – 1987. Tech. Rep. 319, Centre for Social Science Research, Cape Town.

Conradie, B., Piesse, J., 2016. Ranking perceived risk to farmers: How important is the environment? African Journal of Agricultural and Resource Economics 11 (4), 263–276.

Dieguez Cameroni, F., Fort, H., 2017. Towards scientifically based management of extensive livestock farming in terms of ecological predator-prey modeling. Agricultural Systems 153, 127–137.

Du Plessis, J., Avenant, N., Botha, A., Mkhize, N., Müller, L., Mzileni, N., O'Riain, M., Parker, D., Potgieter, G., Richardson, P., Rode, S., Viljoen, N., Hawkins, H.-J., Tafani, M., 2018. Past and current management of predation on livestock. In: Kerley et al. (2018), Ch. 6, pp. 125–177.

Du Plessis, J., Avenant, N., De Waal, H., 2015. Quality and quantity of the scientific information available on black-backed jackals and caracals: contributing to human–predator conflict management? African Journal of Wildlife Research 45 (2), 138–157.

Du Toit, J., 2013. Predator wars: Lynx vs Jackal. Internet.

Erasmus, B., Van Jaarsveld, A., Chown, S., Kshatriya, M., Wessels, K., 2002. Vulnerability of South African animal taxa to climate change. Global Change Biology 8, 679–693.

Fort, H., Dieguez, F., Halty, V., Soares Lima, J., 2017. Two examples of application of ecological modeling to agricultural production: Extensive livestock farming and overyielding in grassland mixtures. Ecological Modelling 357, 23–34.

Francis, C., Lieblein, G., Gliessman, S., Breland, T., Creamer, N., Harwoord, R., Salomonsson, L., Helenius, J., Rickerl, D., Salvador, R., Wiedenhoeft, M., Simmons, S., Allen, P., Altieri, M., Flora, C., Poincelot, R., 2003. Agroecology: The ecology of food systems. Journal of Sustainable Agriculture 22 (3), 99–118.

Gelman, I., 2011. GIGO or not GIGO: The accuracy of multi-criteria satisficing decisions. ACM Journal of Data and Information Quality 2 (2), 9.

Hannah, L., Midgley, G., Hughes, G., Bomhard, B., Mar. 2005. The view from the Cape: Extinction risk, protected areas, and climate change. BioScience 55 (3), 231–242.

Haswell, P., Shepherd, E., Stone, S., Purcell, B., Hayward, M., 2019. Foraging theory provides a useful framework for livestock predation management. Journal for Nature Conservation 49, 69–75.

Holmes, M., Holmes, R., 2006. Can caracals save sheep? Downloaded January 2015.
URL `http://www.farmersweekly.co.za/article.aspx?id=286`

Huffaker, C., 1958. Experimental studies on predation: Dispersion factors and predator-prey oscillations. Hilgardia 27 (14), 343–383.

Jackson, C., McNutt, J., Apps, P., 2012. Managing the ranging behaviour of African wild dogs (Lycaon pictus) using translocated scent marks. Wildlife Research 39, 31–34.

Johnson, S., Marker, L., Mengersen, K., Gordon, C., Melzheimer, J., Schmidt-Küntzel, A., Nghikembua, M., Fabiano, E., Henghali, J., Wachter, B., 2013. Modeling the viability of the free-ranging cheetah population in Namibia: an object-oriented Bayesian network approach. Ecosphere 4 (7), 90.

Jørgensen, S., 2008. Overview of the model types available for development of ecological models. Ecological Modelling 215 (1-3), 3–9.

Jøsang, A., 2008. Conditional reasoning with subjective logic. Journal of Multiple-Valued Logic and Soft Computing 15 (1), 5–38.

Jøsang, A., Diaz, J., Rifqi, M., 2010. Cumulative and averaging fusion of beliefs. Information Fusion 11 (2), 192–200.

Keen, P., 1980. Decision support systems: a research perspective. Tech. rep., Alfred P. Sloan School of Management. Center for Information Systems Research.

Kerley, G., Wilson, S., Balfour, D. (Eds.), 2018. Livestock predation and its management in South Africa: A scientific assessment. Centre for African Conservation Ecology.

Koller, D., Friedman, N., 2005. Probabilistic Graphical Models: Principles and techniques. Massachusetts Institute of Technology.

Linnell, J., Smith, M., Odden, J., Swenson, J., Kaczensky, P., 1996. Carnivores and sheep farming in Norway. 4. Strategies for the reduction of carnivore - livestock - conflicts: a review. Tech. rep., Norsk Institutt for Naturforskning.

Luyt, E. d. C., 2004. Models of bontebok (*Damaliscus pygargus pygargus*, Pallas 1766) habitat preferences in the Bontebok National Park and sustainable stocking rates. Master's thesis, Stellenbosch University.

Melville, H., Bothma, J. P., Mills, M., 2004. Prey selection by caracal in the Kgalagadi Transfrontier Park. South African Journal of Wildlife Research 34 (1), 67–75.

Messmer, T., 2000. The emergence of human-wildlife conflict management: turning challenges into opportunities. International Biodeterioration & Biodegradation 45, 97–102.

Nattrass, N., Conradie, B., 2018. Predators, livestock losses and poison in the South African Karoo. Journal of Cleaner Production 194, 777–785.

Nilsson, N., 2010. Questions and answers. In: Dechter, R., Geffner, H., Halpern, J. (Eds.), Heuristics, probability and causality: A tribute to Judea Pearl. College Publication, pp. 547–551.

Parrott, L., 2011. Hybrid modelling of complex ecological systems for decision support: Recent successes and future perspectives. Ecological Informatics 6 (1), 44–49.

Pearl, J., 1990. Reasoning with belief functions: An analysis of compatibility. International Journal of Approximate Reasoning 4, 363–389.

Pringle, J., Pringle, V., 1979. Observations on the lynx *Felis caracal* in the Bedford district. South African Journal of Zoology 14 (1), 1–4.

Provan, G., 1990. A logic-based analysis of Dempster- Shafer theory. International Journal of Approximate Reasoning 4, 451–495.

Ray, J., Hunter, L., Zigouris, J., 2005. Setting conservation and research priorities for larger African carnivores. Tech. rep., Wildlife Conservation Society.

Rigg, R., Findo, S., Wechselberger, M., Gorman, M., Sillero-Zubiri, C., Macdonald, D., 2011. Mitigating carnivore–livestock conflict in Europe: lessons from Slovakia. Oryx 45 (2), 272–280.

Ripple, W., Estes, J., Beschta, R., Wilmers, C., Ritchie, E., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M., Schmitz, O., Smith, D., Wallach, A., Wirsing, A., 2014. Status and ecological effects of the world's largest carnivores. Science 343 (6167), 1241484.

Ritchie, E. G., Johnson, C. N., 2009. Predator interactions, mesopredator release and biodiversity conservation. Ecology Letters 12, 982–998.

Rosenzweig, M., MacArthur, R., 1963. Graphical representation and stability conditions of predator-prey interactions. The American Naturalist 97 (895), 209–223.

Shafer, G., 1976. A theory of statistical evidence. In: Harper, W., Hooker, C. (Eds.), Foundations of Probability Theory, Statistical Inference, and Statistical Theories of Science. Vol. II. Reidel, pp. 365–436.

Shafer, G., 1981. Constructive probability. Synthese 48, 1–60.

Shafer, G., 1990. Perspectives on the theory and practice of belief functions. International Journal of Approximate Reasoning 4, 323–362.

Shafer, G., 1992. Dempster-Shafer theory. In: Shapiro, S. (Ed.), Encyclopedia of Artificial Intelligence., 2nd Edition. Wiley, pp. 330–331.

Shafer, G., Pearl, J. (Eds.), 1990. Readings in uncertain reasoning. Morgan Kaufmann Publishers Inc.

Sinclair, A., Mduma, S., Grant, J., Hopcraft, C., Fryxell, J., Hilborn, R., Thirgood, S., 2007. Long-term ecosystem dynamics in the Serengeti: Lessons for conservation. Conservation Biology 21 (3), 580–590.

Sprague, R. J., 1980. A framework for the development of decision support systems. MIS Quarterly 4 (4), 1–26.

Stein, A., Fuller, T., Damery, D., Sievert, L., Marker, L., 2010. Farm management and economic analyses of leopard conservation in north-central Namibia. Animal Conservation 13, 419–427.

Tambling, C., Avenant, N., Drouilly, M., Melville, H., 2018. The role of mesopredators in ecosystems: Potential effects of managing their populations on ecosystem processes and biodiversity. In: Kerley et al. (2018), Ch. 8, pp. 205–227.

Tangestani, M., 2009. A comparative study of Dempster–Shafer and fuzzy models for landslide susceptibility mapping using a GIS: An experience from Zagros Mountains, SW Iran. Journal of Asian Earth Sciences 35 (1), 66–73.

Treves, A., Karanth, K., 2003. Special section: Human-carnivore conflict: Local solutions with global applications. Conservation Biology 17 (6), 1489–1490.

Turpie, J., Akinyemi, B., 2018. The socio-economic impacts of livestock predation and its prevention in South Africa. In: Kerley et al. (2018), pp. 53–81.

Van Eeden, L., Ecklund, A., Miller, J., López-Bao, J., Chapron, G., Cejtin, M., Crowther, M., Dickman, C., Frank, J., Krofel, M., Macdonald, D., McManus, J., Meyer, T., Middleton, A., Newsome, T., Ripple, W., Ritchie, E., Schmitz, O., Stoner, K., Tourani, M., Treves, A., 2018. Carnivore conservation needs evidence-based livestock protection. PLoS Biology 16 (9), e2005577.

Wakker, P., 2000. Dempster Belief Functions are based on the principle of complete ignorance. International Journal of Uncertainty, Fuzziness and Knowledge-Based Systems 8 (3), 217–28.

Wilson, N., 2014. The assumptions behind Dempster's rule. In: Heckerman, D., Mamdani, A. (Eds.), Uncertainty in Artificial Intelligence - Proceedings of the Ninth Conference on Uncertainty in Artificial Intelligence, The Catholic University of America, Washington, D.C. 1993, 1st Edition. Morgan Kaufmann Publishers Inc., pp. 527–534.

Young, T., Harcourt, H., 1997. iVivaCaughley! Conservation Biology 11 (4), 831–832.

Zadeh, L., 1984. Reviews of books - A mathematical theory of evidence. The AI Magazine 5 (3), 81–83.

Chapter 6

Principles for agricultural ecology - Applying the theory

Abstract

Human-wildlife conflict (HWC) is an important threat to the survival of vulnerable and endangered predator species on Namibian farmlands. High depredation levels are becoming a threat to the sustainability of Namibian farming systems. For both reasons finding cost-effective and practical solutions to human-wildlife conflict is important. Forty different conflict mitigation methods from the literature were reviewed and evaluated in terms of pros, cons and limitations. Thereafter a survey was conducted of Namibian farmers to determine the extent of predator conflict and effectiveness of the methods currently used to mitigate the conflict. An agroecological, integrated approach to solving farmer-predator conflict was proposed. Literature on the ecology of the four main predators of livestock in Namibia, was reviewed, with an emphasis on ecological questions that can help to solve the conflict with farmers. Global Positioning System (GPS) collar data from cheetahs and leopards was used to answer some of these ecological questions for the two larger species. Following this a predictive Probabilistic Graphical Model (PGM) was built as the basis of a future Decision Support System (DSS) that integrates agricultural and ecological aspects of finding the best (most sustainable and cost-effective) conflict mitigation methods.

Keywords: Black-backed jackals, caracals, cheetahs, Decision Support System, Human-wildlife conflict, leopards, Namibia, Probabilistic Graphical Models, spatial ecology

6.1 Introduction

This research project was started in order to address human-wildlife conflict (HWC) on Namibian farmlands. Finding practical solutions to human-wildlife conflict is important for two reasons: 1) livestock farming is only sustainable in the long term if livestock depredation is kept at a low enough level to enable profitable farming and 2) the long-term survival of two iconic predator species with the majority of their populations in Namibia found on farmlands, is effectively in the hands of farmers. Both leopards' (*Panthera pardus*) (Swanepoel et al., 2013; Balme et al., 2014) and cheetahs' (*Acinonyx jubatus*) (Marker, 2000; Marker et al., 2003; Ray et al., 2005) Namibian populations are mainly on farmlands, outside of protected areas. Moreover, both species are classified as vulnerable in the IUCN red data list (Durant et al., 2015; Stein et al., 2015) with cheetahs being updated to endangered in Namibia and calls have been made that they should be considered as endangered globally, since the largest remaining population of cheetahs in the world is found in Namibia (Weise et al., 2017; Durant et al., 2017). Two other species, black-backed jackals (*Canis mesomelas*) and caracals (*Caracal caracal*), are considered

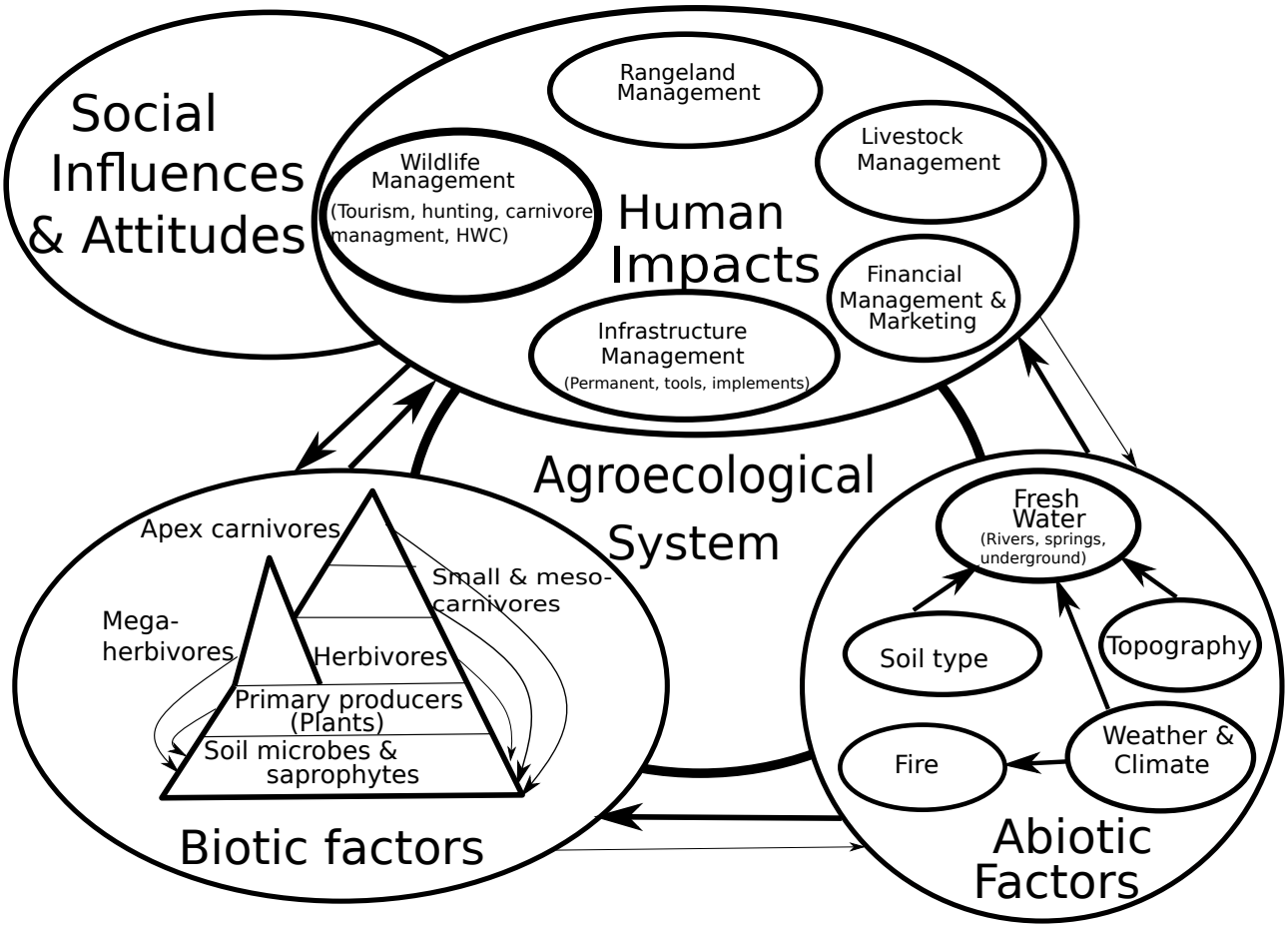


Figure 6.1: *The conceptual model of human-wildlife conflict from an agroecological perspective.* Chapters 2 (Mitigation methods) and 3 (Agricultural aspects) focus on the Human Impacts, with Chapter 1 also mentioning the importance of social influences and attitudes. Chapters 2 (Ecological literature review) and 4 (Spatial ecology) focus on the Biotic factors of the (agricultural) ecosystem. Abiotic factors are part of the habitat models in Chapter 4, but fundamentally cannot be altered by management decisions. Chapter 5 (Probabilistic Graphical Model) brings both Agricultural and Ecological data together in a single model for HWC.

as major livestock predators on farmlands, mostly of small livestock. While the survival of neither species is at stake (Hoffman, 2014; Avgan et al., 2016), there is evidence that current (mostly lethal) conflict mitigation methods are not working to decrease livestock losses and conflict (Van Niekerk, 2010; Conradie and Piesse, 2013; Du Plessis et al., 2015). After investigating the literature, it was proposed that a holistic agroecological approach (Figure 6.1) is needed to address this human-wildlife conflict, for a variety of reasons (Chapter 1).

6.2 Project aims

The main aim of this project has been to design a Decision Support System that can address the current impasse in farmer-predator conflict in Namibia. To address this aim, a number of objectives addressing both the agricultural side of the agroecosystem and the ecological side were defined. From the agricultural perspective, the effectiveness, limitations, advantages and disadvantages of different anti-predation methods were to be determined.

What anti-predation methods are currently used by Namibian farmers? In Chapter 2, forty methods used internationally were identified from the literature. Those methods that had been used in Namibia before, were identified and marked. In Chapter 3 a questionnaire survey sent to Namibian farmers was used to further investigate the use of these anti-predation methods. Twenty-one different methods, of the forty used internationally, were reported by farmers. This implies that there are at least 19 potential methods which have not been used

in Namibia and might be useful. It also means that only the 21 methods in use, can currently be evaluated for their success and applicability in Namibian circumstances.

How effective were these methods in preventing livestock depredation? Current livestock losses by Namibian farmers gave us some insight in how effective current predation management methods are. This was done in Chapter 3 from the questionnaire, where farmers claimed fairly high losses to predation with a mean of 9.2 livestock units (LSU) per farm per year. This was higher than in most previous surveys in Namibia, but still lower than the most recent large-scale survey in South Africa by Van Niekerk (2010). Although based on a limited sample, this was an important baseline for livestock predation in Namibia, showing that about 7% of the Namibian livestock sector's GDP are lost to predators every year. Livestock depredation in Namibia is therefore quite significant from an agricultural perspective, leading to the conclusion that currently used methods are generally ineffective. One interesting observation was that there were a few farmers with exceptionally high predation losses, while most farmers had livestock losses that were lower than the average (mean). This same pattern has been seen internationally and still remains unexplained.

Have there been any changes in predator numbers reported by Namibian farmers on their land? The predator observation data from the farmers in the survey was considered as too subjective to be of much use for answering this question (Chapter 3). In terms of distribution, the reported distribution of the predator species agreed with the recent LCMAN Red Data List for Namibian Carnivores (in prep). In Chapter 4, data from conflict calls received by N/a'an ku sê was used to determine that the relative numbers of leopards involved in conflict have increased relative to cheetahs since 2011, while before 2011 most calls involved cheetahs. No similar data were available for black-backed jackals or caracals.

Not only the density of predators, but also the density of livestock and thus the likely encounter rates between livestock and predators, are important in HWC. What are the current stocking rates and/or farming intensity on Namibian farms? From the farmer survey (Chapter 3), a mean stocking rate of 299.3 LSU per farm was determined for Namibian farms (29.52 ha / LSU). This was below the current long-term "carrying capacity" estimate for Namibia. The great majority of Namibian farms were extensive farming systems, limiting the practicality of some conflict mitigation methods that require intensive management. This could partly explain why the current predation management methods were mostly ineffective in Namibia.

The other side to effectiveness of any cost-benefit analysis of conflict mitigation methods, is the costs (Figure 2.2 in Chapter 2). What were the relative costs of each anti-predation method used in Namibia? No absolute costs could be determined for anti-predation methods, mostly because farmers did not classify their expenditure in terms of predation management (Chapter 3). There was some general idea of the costs for some methods. This, together with an estimation was used to classify costs into cost classes (rather than trying to determine exact costs) in Chapter 5.

In addition to its basic cost-effectiveness, any predation management method would have additional advantages and disadvantages. In a DSS, the farmer could potentially make his final decision on which method to choose, based on these advantages and disadvantages. So what are the advantages and disadvantages of anti-predation methods used worldwide from an agricultural perspective (short-term cost-effectiveness)? All the published anti-predation methods that could be found in the literature were evaluated in terms of advantages, disadvantages and limitations for Namibian circumstances and the result summarised in Table 2.1 (Chapter 2).

What is the relative effectiveness of each method from an agricultural viewpoint? This is perhaps the most important question for writing a DSS to mitigate farmer-predator conflict. Mostly the literature (Chapter 2), but also the farmer surveys (Chapter 3) were used to determine a prior success probability for each conflict mitigation method where possible. This prior success probability can be changed by a number of additional factors (making it more or less likely to be effective) which results in a final success probability for the DSS. It was found that in Namibia, increasing the diversity of game species was one of the simplest methods and most likely to reduce livestock depredation. The only other method that could be shown (with a 67% probability of success) to reduce livestock depredation, was using herders (Table 3.4). Switching from small livestock to

cattle, also had a 68% probability of success, depending on the predator (more successful against jackals and caracals) and if the farmer is actually farming with goats or sheep.

From the ecological viewpoint of sustainability, a number of ecological factors that are relevant to HWC were addressed. The most obviously relevant ecological facets were diet and prey preference. Factors like habitat and prey availability can also influence individual predator diet. The prey preferences of the four predator species were determined mostly from the literature in Chapter 2. It was seen that at least two species (leopards and cheetahs) did not prefer livestock as prey (taking them at lower than available levels). What was the individual degree of variance in prey preference within each species? If individuals of a certain predator species show preferences for specific prey, it becomes more likely that some individuals could prefer livestock as a preferred prey species, even in generalist species. Management options that targets a specific predator individual then becomes more likely to be effective. For both leopards and cheetahs the literature showed that some individual prey specialisation could be found (Chapter 2).

The spatial ecology of the predator species also has obvious implications for predation management. The home range sizes for breeding/territorial males and females of each predator species were determined. For jackals and caracals this was not possible. A rough estimate could be found for male caracals in North-central Namibia from the literature (Chapter 2). For leopards and cheetahs, GPS data from conflict individuals were used to get an indication of how their home ranges varied throughout the country (Chapter 4). This was the first time that home ranges of both species were determined and compared over almost all of the Namibian commercial farmlands. From the home range sizes, a density estimate of cheetahs in Namibia on commercial farmlands could be done using a relatively simple formula: $density = n/area = ((TotArea + TotOverlapArea)/AverageHomeRangeSize)/TotArea$. It was shown that while the GPS collar data in this case were not well suited to this method (since individuals were mostly not collared and released in the same landscape, and thus there was almost no home range overlap for determining the total overlap area), it could be used to estimate the densities of resident cheetahs (and other solitary and potentially territorial species). Because there were no sites with leopards overlapping, this method of density estimation was impossible for leopards from the N/a'an ku sê GPS collar data. However, for cheetahs this density estimate supported the call to update the conservation threat of cheetahs to endangered. Any conflict mitigation methods involving cheetahs (and probably leopards too) should take into account its local and global conservation status.

While the GPS collar data could be used to determine the density of the breeding resident population, it cannot include young non-resident individuals (since growing animals cannot be collared safely). Therefore, it was needed to determine the relative proportion of resident and non-resident individuals in the population of each predator species if a full density estimate is to be done. It was seen that this would require extensive capturing of individuals which was not possible within the limitations of the present project (Marker et al., 2003; Balme et al., 2012a).

In order for livestock to avoid possible predation hotspots, it is not only important in which habitats the different predators occur, but also to determine their habitat use within their home ranges. Does each species have areas within its home range or territory that are used more than the rest (and could be HWC hotspots)? Using GPS collar data, this was done for leopards and cheetahs in Chapter 4. Both species had preferred habitats within their home ranges (Table 4.3). Cheetahs were found to prefer the ecotone between rocky hills and flat plains in the Nama-Karoo Sparse Grassland and Shrubland getting 100-200mm rain per year. In the more mesic (300-400 mm rain / year) Tree and Shrub Savannah Dense Shrubland, they preferred drainage lines and dry riverbeds. Leopards, on the other hand, preferred the rocky hills in the Nama-Karoo Sparse Shrubland. In the more mesic northern parts of Namibia, leopards preferred drainage lines, flat plains, mountains and rocky hills in the Tree and Shrub Savannah Dense Shrubland. In general, dense shrubland in the Tree and Shrub Savannah, especially riverbeds and drainage lines, as well as the transition areas between rocky hills and flat plains in the southern Nama-Karoo areas should be avoided by vulnerable livestock.

Potential hotspots of conflict do not only include core preferred predator habitat types, but also those specific

habitat areas which are preferred for specific purposes, like hunting. Using a new application of T-LoCoH it was shown in Chapter 4 (see Table 4.3) that cheetahs and leopards had specific parts of their home ranges that they preferred for certain behaviour (Lyons, 2014; Lyons et al., 2013). Leopards preferred to hunt in the same habitats which they preferred for their core territories. Cheetahs preferred hunting along drainage lines in both dry and more mesic areas, and along ecotones between mountains or rocky hills and flat areas in the drier areas. In the Nama-Karoo, they also liked to hunt on the open plains. Using the information of which habitat types occurs on a particular farm, and which predator species is causing most livestock losses, the DSS can potentially identify specific areas from which vulnerable livestock should be excluded, and if this is realistic for the specific farm.

If predators are removed as part of predation management, it helps to know the dispersal distances for each predator species for obvious reasons. This was done from the literature for all four species in Chapter 2, although for jackals and caracals, it could not be confirmed for Namibian farmlands. Maximum reported dispersal distances varied between 126 km for black-backed jackals, 138 km for caracals, up to 200 km for leopards and cheetahs (although cheetahs some returned to their home ranges from 450 km away). The need for a minimum translocation distance of 200 km, was demonstrated in Chapter 4.

Where a single predator species is responsible for the vast majority of livestock losses, predator intra-guild interactions can be relevant to livestock depredation management. Was there evidence of less dominant predator species showing spatial avoidance of more dominant species or of spatial niche partitioning between the predator species (the effect of "fear and loathing" – Ritchie and Johnson, 2009)? There was limited evidence of spatial niche separation between leopards and cheetahs (Chapter 4), shown by the fact that while both species showed some overlap in some of their preferred habitat types, there were also habitats that were preferred by one species, while being significantly avoided by the other species.

To determine if there is any evidence of mesopredator release where larger predators have been extirpated (Palomares and Caro, 1999; Ritchie and Johnson, 2009), there were basically two approaches possible: 1) Experimental manipulation and long-term research where the larger predators are actively removed from the land and the effect on the mesopredator densities observed, or 2) not interfering with the ecosystem and determining the spatial movement of both species in the same landscape and observing if there is evidence that areas of high usage by the dominant predator species are avoided by the subordinate species (Flagel et al., 2017). The original proposal included the option of GPS collaring all four predator species in the same area to investigate any such interactions. Technology costs, time constraints and permit issues, prevented this from happening. Spoor tracking was attempted as an alternative approach, but did not succeed on the soil substrates available on the farms where this research was to be done. It was therefore not possible to properly investigate mesopredator release with the data we had.

6.3 Main contributions to resolving human-wildlife conflict

In the process of designing the Decision Support System (*DSS – the* main contribution of this research project to resolving HWC), a number of additional contributions to resolving human-wildlife conflict were made.

A convenient collection of conflict mitigation methods that can be used as a reference in the future, including the both the advantages and disadvantages of the various methods, especially as applicable to Namibia, is provided in Chapter 2. This will be used in the DSS to provide the farmer with the pros and cons of each of the three methods with the highest expected utility.

Predation losses in Namibia was quantified from a farmer survey (Chapter 3). It was shown that high prey diversity resulted in lower livestock depredation. This was the first attempt at a baseline total for quantifying the economic impact of livestock predation in the agricultural sector of Namibia. Predation was seen to have a

significant perceived impact compared to other causes of livestock loss (9.2 livestock units (LSU) per farm per year).

The spatial ecology of leopards and cheetahs in Namibia was investigated using GPS collar data (Chapter 4). A comprehensive literature review of the various home range estimators used in Namibia before was done. The Concave Hull was proposed as a useful and accurate home range estimator with advantages over both the MCP and KDE methods (given enough GPS points). A formula for determining densities from home range data, including range overlaps, was presented.

The use of t-LoCoH for modelling predator movement patterns was demonstrated for the first time. This model was applied as a predictive model for identifying the habitat preferences of predators, especially for different behavioural preferences. An easy method for identifying possible kill sites (with low probability of false positives) using t-LoCoH was proposed. However, it still needs to be properly ground-truthed.

It was shown that translocated animals had significantly larger home ranges than animals that were released back into their own home range. This implies that translocated animals, coming into contact with more farms (and farmers) are often more likely to be involved in human-wildlife conflict afterwards. We did not find any significant difference between the success rates of translocating predators compared to collaring and releasing them back at the same capture site for mitigating HWC.

An agroecological conceptual model that shows the entire system that needs to be taken into account for solving human-wildlife conflict, was presented. This research argues for a whole-system approach, including both agricultural and ecological principles holistically, as the basis for solving human-predator conflict. Based on this conceptual model, a Decision Support System was designed, using a probabilistic graphical model (decision network, an extension of Bayesian networks) and presented in sufficient detail so that an open-source collaborative program can be written based on it (Chapter 5). While Bayesian networks have been used before in modelling large carnivore behaviour, this was the first known description of the use of influence diagrams (decision network) for mitigating human-wildlife conflict.

6.4 Decision Support System

In Chapter 5, a basic algorithm was created for the Decision Support System. It is important that any model for such-decision making should not be a "black box", where the programme simply provides a recommendation at the end, but should be clear and make logical sense (transparent). This is especially important, because many of the basic parameters of the model are still uncertain (e.g. the costs of various conflict mitigation methods, and some basic ecological factors, like the home range size of black-backed jackal). If the model remains comprehensible and clear, it can always be improved, changed and expanded as more information becomes available.

The best way to design such a model, was found to be the use of Probabilistic Graphical Models (PGM's), consisting of Directed Acyclical Graphs indicating the influence of variables on each other, as well as Conditional Probability Diagrams for each node, showing various probabilities for a node given its parents. More specifically, we used Influence Diagrams (also called Decision Networks, an extension of Bayesian Networks), which in addition to the probabilities of a typical Bayesian Network, also include the possible decisions that an actor (farmer, in our case) can take and the utility of the various possible outcomes of the decisions.

The design of the basic Decision Support System (DSS) includes prior probabilities for success included in the PGM for each conflict mitigation method. This prior model will also include the known or estimated costs and effectiveness of each method. Then the farmer or manager would input some data describing his farm or management unit, which will change the probabilities of success either positively or negatively when combined with the prior probabilities. The HWC situation on the farm (e.g. pre-existing infrastructure, currently used methods, current livestock losses etc.) will in most cases also influence the costs and effectiveness of any switch

to a new mitigation method. Finally, for each method the final probability of success is combined with the estimated cost and effectiveness of that method in order to define an influence diagram (Koller and Friedman, 2005) combining the probability of success with the potential economic gain (decrease in livestock losses relative to implementation costs) in a single Maximum Expected Utility value.

Two things are important to note: 1) Even if the prior probabilities were assigned subjectively and the penultimate "probability of success" would not represent an actual probability of success, for this model it is less important than the fact that the *relative* probabilities of success of the different methods compared to each other are in the right order. The published success rates as well as the Namibian farmer feedback on the methods they use from the survey (Chapters 2 & 3) is expected to be accurate enough to at least ensure that the relative order assigned to the success probabilities would be correct. As long as the more practical and likely to be successful mitigation method is assigned a higher probability than the less likely to succeed method, the model will find the "best" mitigation method for the situation. 2) The initial probability values are tentative and the long-term success of this approach will depend on the feedback and updating of the model as it is used. It should thus be seen as a first attempt, but still an improvement on most current one-size-fits-all "silver bullet" solutions, or the "scattergun" (try all of these, one of them should work) advice so often given to mitigate human-wildlife conflict.

6.5 Conclusion

Finding sustainable, practical, affordable, effective, safe, and economically viable solutions to human-wildlife conflict with predators, requires a holistic and integrated agroecological approach. In this thesis, I highlighted some of the agricultural and ecological issues that are important for mitigating farmer-predator conflict. Some basic principles are that of 1) using the whole-ecosystem viewpoint, instead of focusing only on the predator and livestock, 2) including both the pros and cons of the various mitigation methods, 3) acknowledging both the ecological and the agricultural limitations of some mitigation methods, which make them impractical in some situations, 4) acknowledging that every farm and every conflict situation is unique and that there are no silver bullet solutions that will work for everyone.

The ultimate stewards of the land where most surviving cheetahs and leopards in Namibia can be found, are farmers, putting the future survival of these big cats squarely into their hands. Human-wildlife conflict is not only a problem for these threatened predator species, but also for the sustainability of livestock and game farming in Namibia. I present this thesis with the sincere hope that it will make a real contribution to solving human-wildlife conflict in a way that will ensure the sustainability of the agricultural and ecological processes on Namibian farmlands, the wildest, most beautiful country I know.

References

Avgan, B., Henschel, P., Ghoddousi, A., 2016. *Caracal caracal*. In: The IUCN Red List of Threatened Species 2016, Downloaded on 1 March 2018. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T3847A50650230.en, p. e.T3847A102424310.

Balme, G., Batchelor, A., De Woronin Britz, N., Seymour, G., Grover, M., Hes, L., Macdonald, D., Hunter, L., Oct. 2012a. Reproductive success of female leopards *Panthera pardus*: the importance of top-down processes. Mammal Review 43 (3), 221–237.

Balme, G., Lindsey, P., Swanepoel, L., Hunter, L., 2014. Failure of research to address the rangewide conservation needs of large carnivores: leopards in South Africa as a case study. Conservation Letters 7 (1), 3–11.

Conradie, B., Piesse, J., 2013. The effect of predator culling on livestock losses: Ceres, South Africa, 1979 – 1987. Tech. Rep. 319, Centre for Social Science Research, Cape Town.

Du Plessis, J., Avenant, N., De Waal, H., 2015. Quality and quantity of the scientific information available on black-backed jackals and caracals: contributing to human–predator conflict management? African Journal of Wildlife Research 45 (2), 138–157.

Durant, S., Mitchell, N., Groom, R., Pettorelli, N., Ipavec, A., Jacobson, A., Woodroffe, R., Böhm, M., Hunter, L., Becker, M., Broekhuis, F., Bashir, S., Andresen, L., Aschenborn, O., Beddiaf, M., Belbachir, F., Belbachir-Bazi, A., Berbash, A., Brandao de Matos Machado, I., Breitenmoser, C., Chege, M., D., C., Davies-Mostert, H., Dickman, A., Fabiano, E., Farhadinia, M., Funston, P., Henschel, P., Horgan, J., De Iongh, H., Jowkar, H., Klein, R., Lindsey, P., Marker, L., Marnewick, K., Melzheimer, J., Merkle, J., M'Soka, J., Msuha, M., O'Neill, H., Parker, M., Purchase, G., Sahailou, S., Saidu, Y., Samna, A., Schmidt-Küntzel, A., Selebatso, E., Sogbohossou, E., Soultan, A., Stone, E., Van der Meer, E., Van Vuuren, R., Wykstra, M., Young-Overton, K., 2017. The global decline of cheetah *Acinonyx jubatus* and what it means for conservation. PNAS 114 (3), 528–533.

Durant, S., Mitchell, N., Ipavec, A., Groom, R., 2015. *Acinonyx jubatus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T219A50649567.en, p. e.T219A50649567.

Flagel, D., Belovsky, G., Cramer, M., Beyer, D. J., Robertson, K., 2017. Fear and loathing in a Great Lakes forest: cascading effects of competition between wolves and coyotes. Journal of Mammalogy 98 (1), 77–84.

Hoffman, M., 2014. *Canis mesomelas*. In: The IUCN Red List of Threatened Species 2014, Downloaded on 1 March 2018. IUCN, http://dx.doi.org/10.2305/IUCN.UK.2014-1.RLTS.T3755A46122476.en, p. e.T3755A46122476.

Koller, D., Friedman, N., 2005. Probabilistic Graphical Models: Principles and techniques. Massachusetts Institute of Technology.

Lyons, A., Mar. 2014. T-LoCoH - A spatiotemporal home range estimator for GPS data. Remotely Wild 30, 7–9.

Lyons, A., Turner, W., Getz, W., Jul. 2013. Home range plus: a space-time characterization of movement over real landscapes. Movement Ecology 1 (1), 2.

Marker, L., 2000. Aspects of the ecology of the cheetah (*Acinonyx jubatus*) on North Central Namibian farmlands. Journal of Namibia Scientific society 48, 40–48.

Marker, L., Dickman, A., Jeo, R., Mills, M., Macdonald, D., 2003. Demography of the Namibian cheetah, *Acinonyx jubatus jubatus*. Biological Conservation 114, 413–425.

Palomares, F., Caro, T., 1999. Interspecific killing among mammalian carnivores. The American Naturalist 153 (5), 492–508.

Ray, J., Hunter, L., Zigouris, J., 2005. Setting conservation and research priorities for larger African carnivores. Tech. rep., Wildlife Conservation Society.

Ritchie, E. G., Johnson, C. N., 2009. Predator interactions, mesopredator release and biodiversity conservation. Ecology Letters 12, 982–998.

Stein, A., Athreya, V., Gerngross, P., Balme, G., Henschel, P., Karanth, U., Miquelle, D., Rostro-García, S., Kamler, J., Laguardia, A., Khorozyan, I., Ghoddousi, A., 2015. *Panthera pardus*. In: The IUCN Red List of Threatened Species, Downloaded on 16 February 2017. IUCN, http://www.iucnredlist.org/details/summary/15954/0, p. e.T15954A102421779.

Swanepoel, L., Lindsey, P., Somers, M., Van Hoven, W., Dalerum, F., 2013. Extent and fragmentation of suitable leopard habitat in South Africa. Animal Conservation 16 (1), 41–50.

Van Niekerk, H., 2010. The cost of predation on small livestock in South Africa by medium-sized predators. Master's thesis, University of the Free State.

Weise, F., Vijay, V., Jacobson, A., Schoonover, R., Groom, R., Horgan, J., Keeping, D., Klein, R., Marnewick, K., Maude, G., Melzheimer, J., Mills, G., Van der Merwe, V., Van der Meer, E., Van Vuuren, R., Wachter, B., Pimm, S., 2017. The distribution and numbers of cheetah (*Acinonyx jubatus*) in southern Africa. PeerJ 5, e4096.

Appendix A

Appendix: Human-wildlife conflict in Namibia: an agricultural perspective - Questionnaire

UNIVERSITEIT·STELLENBOSCH·UNIVERSITY

Vraelys oor roofdiere op plase vir doktorale studie:

"Boere en roofdiere"

Beste Produsent

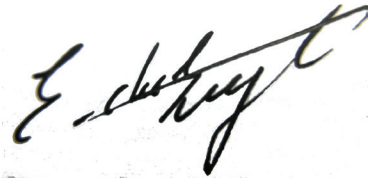
Ek is tans besig met navorsing deur die Universiteit van Stellenbosch oor die omvang en moontlike oplossings vir boere se konflik met roofdiere in Namibië. 'n Soortgelyke opname deur die RPO in Suid-Afrika het gewys dat boere in Suid-Afrika 'n geskatte R 1 390 453 062,00 per jaar aan roofdiere kan verloor. Dit lyk dus asof huidige roofdier-bestuur maatreëls nie werklik doeltreffend is nie en dat daar na alternatiewe gekyk moet word. Sommige bewaringsinstansies reken egter dat boere oordryf en dat roofdiere nie 'n noemenswaardige probleem is nie. Die aangehegte vraelys is deel van 'n proses om die werklike koste van roofdiere op plase te bepaal en praktiese, lang-termyn en koste-doeltreffende oplossings te vind. Hoe meer boere dit invul, hoe betroubaarder sal die resultate wees en kan beter oplossings gevind word. Die vraelys kan anoniem ingevul word sonder om u naam of plaasnaam te noem.

Vir daardie boere wat graag verder wil deelneem aan die navorsing (deur bv. roofdiere op u plaas GPS halsbande om te sit en hul bewegings te monitor of meer besonderhede oor roofdier-beheermaatreels te verskaf) of persoonlike terugvoer oor die vordering van die projek wil ontvang, is daar wel plek om u kontak-besonderhede te verskaf. ALLE DATA SAL KONFIDENSIEEL HANTEER WORD en geen besonderhede sal sonder die boer se uitruklike toestemming gepubliseer word nie. Die resultate sal ook eers aan die deelnemende boere gestuur word, voor dit wyer gepubliseer sal word.

Die vraelys is ook op die Internet beskikbaar by http://www.newdevelopment.co.za/FarmingPredator/boere.html of kan aanlyn ingevul word by http://www.surveyface.com/study/Farming-with-predators-survey-9699.php?l=y .

U kan die voltooide vraelys na my stuur by: Posbus 1755, Otjiwarongo, Namibië. E-pos: chavoux@gmail.com of bel my by 081 312 4678 as daar enige vrae is.

Die uwe



Chavoux Luyt

*Kontakbesonderhede (Opsioneel)

*Naam (opsioneel):_____ *Datum: _____

*Plaasnaam: _____ *Plaasnommer(s): _____

*Telefoon: _____ *E-pos: _____

Wil u deelneem aan verdere navorsing? Ja: ☐ Nee: ☐

Plaasbesonderhede (Vul asb. alles in tensy gemerk as opsioneel met 'n '*')

Distrik:_____ *Boerevereniging:_____

Hoeveel jare is u al op die plaas? _____ Grootte van plaas / plase (ha):_____

Tipe plaas - Beeste: ☐ Wild: ☐ Trofee-jag: ☐ Kleinvee: ☐ Gemeng: ☐

*Veeheining (ha): _____ *Wildheining (hoog): ____ *Wildheining (laag/medium): _____

*Jakkalsproef (ha): _____ *Geëlektrifiseerde heinings (ha): ____

Is u lid van 'n bewaringsgebied (conservancy)? Ja: ☐ Nee: ☐

Indien ja, funksioneer die bewaringsgebied soos verwag? Ja: ☐ Nee: ☐

* Bewaringsgebied Naam:_____

Het u enige bestuurspraktyke op u plaas verander sedert u by die bewaringsgebied aangesluit het?

Ja: ☐ Nee: ☐ Indien ja, wat het u verander? _____

Vee en wild

Veegetalle: Datum van laaste telling [_____] OF Skatting: Ja: ☐ Nee: ☐

Vee	Manlik (vir teel)	Vroulik	Aanwas in afgelope jaar	Totaal	Primêre Ras
Beeste					
Bokke					
Skape					
Ander (bv. Perde)					

Het u kalfseisoen(e) vir u beeste? Ja: ☐ Nee: ☐ Indien wel, wanneer? _____

Het u lamseisoen(e) vir u skape? Ja: ☐ Nee: ☐ Indien wel, wanneer? _____

Het u lamseisoen(e) vir u bokke? Ja: ☐ Nee: ☐ Indien wel, wanneer? _____

Gebruik u klein lam-/kalfkampies naby die huis of veeposte? Ja: ☐ Nee: ☐

Vir watter veetipes? _____

Hou u lammers/kalwers op kraal deur die dag? Ja: ☐ Nee: ☐

Indien wel, watter veetipes? _____

Hou u diere op kraal deur die nag (insluitend volwasse diere)? Ja: ☐ Nee: ☐

Indien wel, watter vee? _____

Gebruik u herdershonde (bv. Anatoliese berghond)? Ja: ☐ Nee: ☐

Hoeveel en hoe groot? Groot: ____ Medium: ____ Klein: ___ Opreg geteel? Ja: ☐ Nee: ☐

Ras(se):_____

Gebruik u veewagters? Ja: ☐ Nee: ☐

Hoeveel en vir watter vee?_____

Watter van die volgende metodes gebruik u om veeverliese deur roofdiere te voorkom?

Skuif vee na area met min/geen roofdiere: Ja: ☐ Nee: ☐

Gebruik donkies as "veewagters": Ja: ☐ Nee: ☐

Verkoop swak moeders: Ja: ☐ Nee: ☐

Laat horings groei: Ja: ☐ Nee: ☐

Rasse met tropinstink / groot troppe: Ja: ☐ Nee: ☐

Hoeveel vee het u die afgelope jaar verloor?

Veeverliese	Skatting (S) OF Gedokumenteer (D)	Siekte	Beserings	Weens droogte	Diefstal	Roofdiere	Ander	Beskryf ander (bv. erdvarkgate)
Volwasse Bees								
Kalwers								
Volwasse Skaap								
Skaaplammers								
Volwasse Bokke								
Boklammers								
Ander								

Sou u bereid wees om van veetipe of ras te verander as dit tot minder roofdier-verliese lei?

Beslis: ☐ Net as dit ander voordele inhou: ☐ Nee: ☐

Waarom (nie)? _____

Enige wild in wildkampe of binne wildwerende heining? Ja: ☐ Nee: ☐

Watter tipe wild op die plaas en hoeveel van elke tipe? Hoe het getalle verander oor die afgelope 5 jaar?

Wildgetalle	**Binne wildwerende heining**	***Buite wildwerende heining**	**Toegeneem:+** **Afgeneem: -** **Dieselfde: =**
Koedoe			
Gemsbok			
Springbok			
Vlakvark			
Eland			
Steenbok (Baie: >50; Paar; Min)			
Duiker (Baie: >50; Paar; Min)			
Ander (Wat?)			
Ander (Wat?)			
Ander (Wat?)			
Ander (Wat?)			
Ander (Wat?)			

Roofdiere en roofdierverliese

Watter roofdiere kom op u plaas voor?

Roofdier	**Gesien (S)** **Spore/Tekens (T)** **Gehoor (H)** **Geen (N)**	**Meer (+)** **Minder (-)** **Dieselfde (=) as 5 jaar gelede**	**Meer (+)** **Minder (-)** **Dieselfde (=) as verlede jaar**
Luiperd (Tier)			
Jagluiperd (Cheetah)			
Rooikat			
Rooijakkals			
Bruin Hiëna (Strandwolf)			
Gevlekte Hiëna			
Leeu			
Wildehond			
Ander			

Het u vee verloor deur roofdiere in die afgelope jaar? Ja: ☐ Nee: ☐

Veeverliese in die afgelope jaar	**Beslis roofdier (Gedokumenteer)**				**Vermoedelik roofdier**				**Vergelyk met 5 jaar gelede:**
	Luiperd	Jagluiperd	Rooikat	Jakkals	Luiperd	Jagluiperd	Rooikat	Jakkals	Meer (+), Minder (-) of Dieselfde (=)
Beeste < 1 maand									
Beeste 1-6 maande									
Beeste 7-12 maande									
Beeste > 12 maande									
Skape 0-6 maande									
Skape > 6 maande									
Bokke 0-6 maande									
Bokke > 6 maande									
Ander									

Het u roofdiere verwyder vanaf u plaas oor die afgelope jaar? Ja: ☐ Nee: ☐

Roofdiere verwyder	**Luiperds**	**Jagluiperds**	**Rooikatte**	**Jakkalse**	**Ander**	**Opmerkings**
Skatting (S) OF Gedokumenteer (D)						
Jag self aktief						
Jag met honde						
Betaal jagter						
Skiet as toevallig gesien						
Jag slegs as daar veeverliese was						
Slagysters & strikke (bv. Doodslaners)						
Vanghokke						
Gif						
Jakkalskanon						
Deur MET verwyder						
Ander? Beskryf						
Totaal verwyder						

Weet u wat die ekstra koste is om u vee teen roofdiere te beskerm? Geen idee nie: ☐ Skatting: ☐

Gedokumenteer: N$_____

Sou u bereid wees om u boerderymetodes te verander as dit to minder veeverliese weens roofdiere lei?

Ja: ☐ Nee: ☐ Waarom (nie)?_____

Waarom verkies u die metodes wat u tans gebruik om roofdierverliese te beperk?

_____

Baie dankie vir u bydrae!

Pos asb. voltooide vorm na: Chavoux Luyt

Posbus 1755

Otjiwarongo

UNIVERSITEIT-STELLENBOSCH-UNIVERSITY

Questionnaire on predators on farms for doctoral research:

"Farming with predators"

Dear Producer

I am currently doing research through Stellenbosch University on the on the magnitude and possible solutions to farmer conflict with predators in Namibia. A similar survey by the RPO in South Africa showed that farmers in South Africa could be losing an estimated R 1 390 453 062.00 to predators. It seems thus as if current predator management approaches are not really effective and that we need to look at alternatives. Some conservation organizations think that farmers are exaggerating and that predators are not really a problem worth mentioning. The attached questionnaire is part of a process to determine the real costs of predators on farms and to find practical, long-term and cost-effective solutions. The more farmers filling it in, the more reliable the results will be and the better the solutions that can be found. The questionnaire can be filled in anonymously without mentioning of your own name or your farm's name.

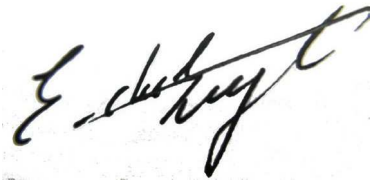
For those farmers who want to participate in further research (e.g. GPS collaring of predators on your farm to monitor their movements or by providing more details on your predation prevention methods) or to receive personal feedback on the progress of the project, there is space to fill in your contact details. ALL DATA WILL BE HANDLED AS CONFIDENTIAL and no details will be published without the express permission of the farmer.

The results will also firs be sent to the participating farmers, before being published more widely.

This questionnaire is also available to download on the Internet at http://www.newdevelopment.co.za/FarmingPredator/boere.html or can be filled in online at http://www.surveyface.com/study/Farming-with-predators-survey-9699.php?l=y .

You can send the completed questionnaire to me at: P.O. Box 1755, Otjiwarongo, Namibia. E-mail: chavoux@gmail.com or call me at 081 312 4678 if there are any questions.

Yours faithfully,



Chavoux Luyt

*Contact details (Optional)

*Name (optional):_____ *Date: _____

*Farm name: _____ *Farm number(s): _____

*Telephone: _____ *E-mail: _____

Do you want to participate in further research? Yes: ☐ No: ☐

Farm details (Please fill in everything unless marked as optional with a '*')

District:_____ *Farmers association:_____

How many years have you been on the farm? _____ Size of farm(s) (ha):_____

Type of farm: Cattle: ☐ Game: ☐ Trophy hunt: ☐ Small livestock: ☐ Mixed: ☐

*Cattle fence (ha): _____ *Game fence (high): _____ *Game fence (non-jumping game): ___

*Jackal proof (ha): _____ *Electrified fencing (ha): _____

Are you a member of a conservancy? Yes: ☐ No: ☐

If yes, does the conservancy function as expected? Yes: ☐ No: ☐

* Conservancy Name:_____

Did you change any management practices since becoming a conservancy member?

Yes: ☐ No: ☐ If yes, what did you change? _____

Livestock and game

Livestock numbers: Date of last count [_____] OR Estimate: Yes: ☐ No: ☐

Livestock	Male breeding	Female	Young born in past year	Total	Primary Breed
Cattle					
Goats					
Sheep					
Other (e.g. Hoerses)					

Do you have calving season(s) for your cattle? Yes: ☐ No: ☐ If yes, when? _____

Do you have lambing season(s) for your sheep? Yes: ☐ No: ☐ If yes, when? _____

Do you have kidding season(s) for your goats? Yes: ☐ No: ☐ If yes, when? _____

Do you use small lambing/calving camps near the house or livestock posts? Yes: ☐ No: ☐

If yes, for which livestock? _____

Do you keep calves/lambs/kids in kraal during the day? Yes: ☐ No: ☐

If yes, for which livestock? _____

Do you kraal livestock at night (including adults)? Yes: ☐ No: ☐

If yes, which livestock?_____

Do you use livestock guarding dogs (e.g. Anatolians)? Yes: ☐ No: ☐

How many and what size? Large:_Medium:_____Small:___Pure bred? Yes: ☐ No: ☐

Breed(s):_____

Do you use herders? Yes: ☐ No: ☐

How many and for which livestock?_____

Which of the following methods do you use to prevent livestock losses to predators?

Move livestock to areas with few or no predators: Yes: ☐ No: ☐

Use donkeys as livestock guarding animals: Yes: ☐ No: ☐

Sell off bad mothers: Yes: ☐ No: ☐

Let horns grow: Yes: ☐ No: ☐

Breeds with herding instinct/large herd: Yes: ☐ No: ☐

How much livestock did you lose over the past year?

Livestock losses	Estimate (S) OR Documented (D)	Disease	Injuries	Through drought	Theft	Predators	Other	Describe other
Adult cattle								
Calves								
Adult sheep								
Lambs								
Adult goats								
Kids								
Other								

Would you be willing to change your livestock type or breed if it led to less predator losses?

Definitely: ☐ Only if it has other advantages: ☐ No: ☐

Why (not)? _____

Any game in game camps or game fenced areas? Yes: ☐ No: ☐

What game occur on the farm and how many? How did the numbers change over the past 5 years?

Game numbers	**Inside game fenced area**	***Outside game fence**	**Increase: +** **Decrease: -** **Same: =**
Kudu			
Oryx			
Springbok			
Warthog			
Eland			
Steenbok (Many: >50; Some; Few: <10)			
Duiker (Many: >50; Some; Few: <10)			
Red Hartebeest			
Other (What?)			
Other (What?)			
Other (What?)			
Other (What?)			

Predators and predation losses

Which predators are found on your farm?

Predator	**Seen (S)** **Track/Signs (T)** **Heard (H)** **None (N)**	**More (+)** **Fewer (-)** **Same (=) as 5 years ago**	**More (+)** **Fewer (-)** **Same (=) as last year**
Leopard			
Cheetah			
Caracal			
Black Backed Jackal			
Brown Hyaena			
Spotted Hyaena			
Lion			
Wild Dog			
Other			

Did you lose livestock to predators in the past year? Yes: ☐ No: ☐

Livestock losses in the past year	**Definitely predator (Documented)**				**Suspected predator**				**Compared to 5 years ago:**
	Leopard	Cheetah	Caracal	Jackal	Leopard	Cheetah	Caracal	Jackal	More (+), Less (-) or Same (=)
Cattle < 1 month									
Cattle 1-6 months									
Cattle 7-12 months									
Cattle > 12 months									
Sheep 0-6 months									
Sheep > 6 months									
Goats 0-6 months									
Goats > 6 months									
Other									

Did you remove predators from your farm in the past year? Yes: ☐ No: ☐

How many predators removed	**Leopards**	**Cheetahs**	**Caracals**	**Jackals**	**Other**	**Notes**
Estimate (S) OR Documented (D)						
Hunt actively yourself						
Hunt with dogs						
Pay hunter						
Shoot when seen by chance						
Only hunt when livestock lost						
Gin traps & snares (e.g. “Doodslaners”)						
Trap cages						
Poison						
Coyote getter						
Removed by conservationists						
Other? Describe						
Total removed						

Do you know what the extra costs are to prevent predation losses? No idea: ☐ Estimate: ☐

Documented: N$_____

Would you be willing to change your farming methods if it led to less predation losses?

Yes: ☐ No: ☐ Why (not)?_____

Why do you prefer the methods you currently use to prevent predation losses?

_____

Thank you for your time!

Please mail completed form to: Chavoux Luyt

P.O. Box 1755

Otjiwarongo

Appendix B

Appendix: Chapters 3, 4 & 5: Resampling R script examples

B.1 Surveys (Chapter 3)

B.1.0.1 Survey data

(http://www.newdevelopment.co.za/FarmingPredators/SurveysCleaned.csv)

B.1.0.2 Comparing the mean of two groups for statistical significance

```
#TwoSampleTestR.r
#Randomization test for difference between two means
#mean difference and t statistic
#Pooled variances should give the same results

#Enter data:
library(readr) Surveys <- read_delim("~/doks/PhD/Data/GIS/Gis/QGIS_Projects/My_Projects/Su
        ";", escape_double = FALSE, locale = locale(grouping_mark = ""),
trim_ws = TRUE) View(Surveys)

#*************************************************************
#Extract data, clean data, prepare data, Split into groups
#changed for each variable to be tested...

#Extract data:
#CattlePredPerc = (Surveys$Catl_Pred/Surveys$Cattle)*100
#SheepPredPerc = (Surveys$Shep_Pred/Surveys$Sheep)*100
#GoatPredPerc = (Surveys$Goat_Pred/Surveys$Goats)*100
#Surveys <- Surveys[which(Surveys$Ewes_Sheep > 400), ]
Selected <- Surveys[which(Surveys$Conservanc > 0), ]
Alone <- Surveys[which(Surveys$Conservanc == 0),]

#Clean data:
#Select columns:
Selected <- Selected$Sh_Pred_Total+Selected$GoatPredTot+Selected$PredationC
Alone <- Alone$Sh_Pred_Total+Alone$GoatPredTot+Alone$PredationC
#Other options commented out:
#Selected <- Selected[!is.na(Selected$Sheep), ]
#Selected <- (Selected$LSU_Total) #Selected <- abs(Selected$Tot_Leoprd) + abs(Selected$Tot_
#Alone <- Alone[!is.na(Alone$Sheep) | !is.na(Alone$Goats | !is.na(Alone$Cattle)), ] #Alone
#Alone <- Alone[!is.na(Alone$Sheep), ]
#Alone <- (Alone$LSU_Total) Selected <- Selected[!is.na(Selected)]
#Kraaled <- Kraaled[!is.infinite((Kraaled))] Alone <- Alone[!is.na(Alone)] #FreeRoam <- Fr

#Prepare data:
#CattleSheepPredPerc = c(CattlePredPerc, SheepPredPerc) #concatenate
#SmallPredPerc = c(SheepPredPerc, GoatPredPerc)

#Split into groups:
SmallGrp = Selected #expect to have lower mean
```

```
LargeGroup = Alone #expect to have higher mean

#Change above for each variable to test;
#code below stays the same**********************************
#Observed:
n1 <- length(SmallGrp)
n2 <- length(LargeGroup)
N <- n1 + n2
meanSmall <- mean(SmallGrp);
meanLarge <- mean(LargeGroup)
diffObs <- (meanLarge - meanSmall)
#combine samples: Combined <- c(SmallGrp, LargeGroup) #c = concatenate
#Pooled variances:
s2p <- (var(SmallGrp)*(n1-1) + var(LargeGroup)*(n2-1))/(n1 + n2 - 2)

#Write results:
cat( "Larger mean observed: ", meanLarge, " with the smaller mean: ", meanSmall, '\n')
cat("The one-tailed test will be used when the alternative hypothesis is that the larger m
cat("The two-tailed test will be used when the alternative hypothesis is simply that there
cat("The observed difference between the means is = ", diffObs, "for ", n1, " and ", n2, "
cat("The pooled variance is = ", s2p, '\n')
cat("Now we will run a standard parametric t test \n")
ttest <- t.test(SmallGrp, LargeGroup)
cat("The resulting t test on raw data is = ", ttest$statistics, '\n')
print(ttest)
tObs <- ttest$statistic

#resampling:
nreps <- 10000 #Using 10000 resampling of the data
cat("Now we will resample with ", nreps, "repetitions.\n")
meanDiff <- numeric(nreps)
#Setting up arrays to hold the results:
t <- numeric(nreps)
#set.seed(1086) #To get exact same result every time
for (i in 1:nreps) {
data <- sample(Combined, N, replace = FALSE)
grp1 <- data[1:n1]
grp2 <- na.omit(data[n1+1:N])
meanDiff[i] <- mean(grp2) - mean(grp1)
test <- t.test(grp1,grp2)
t[i] <- test$statistic }

#Results:
#Just to demonstrat the equivalence of mean differences and t:
cat("The correlation between mean differences and t is = ", '\n') print(cor(meanDiff, t))
cat('\n')
#One-tailed test:
cat("One-tailed test\n")
cat("================")
```

```
cat("The number of times when the mean difference exceeded the observed difference (diffOb
print(length(meanDiff[meanDiff >= diffObs]))
cat("\nThe proportion of resamplings when the mean diff exceeded the observed value (p) =
print(length(meanDiff[meanDiff >= diffObs])/nreps)
cat('\n')
cat('\n')
#Two-tailed test:
cat("Two-tailed test\n")
cat("================")
#Rather than incrementing counter, count at the end (two-tailed test):
cat("The number of times when the absolute mean difference exceeded diffObs = ") absMeanD
absDiffObs <- abs(diffObs)
print(length(absMeanDiff[absMeanDiff >= absDiffObs]))
cat("\nThe number of times when the resampled t values exceeded the obtained t = ") abst <
abstObs <- abs(tObs)
print(length(abst[abst >= abstObs]))
cat("\nThe proportion of resamplings when the absolute mean diff exceeded the observed valu
print(length(abs(absMeanDiff[absMeanDiff >= absDiffObs]))/nreps)
cat('\n')

#Plots data:
#par(mfrow = c(2,2))
#Set up graphic window
#hist(meanDiff, breaks = 25, ylim = c(0,425))
#hist(t, breaks = 50, xlim = c(-5,5), ylim = c(0,425))
# The following plots the empirical distribution of t from resampling
# against the actual Student's t distribution.
# Notice the similarity even though the data are badly skewed
hist(t, freq = FALSE, xlim = c(-5,5), ylim=c(0,.4), col = "lightblue",      ylab = "density
par(new = T)
den <- seq(-5, 4.998, .002)
tdens <- dt(den, 36)
plot(den, tdens, type = "l", ylim = c(0,.4), xlim=c(-5,5),      col = "red", ylab="", xlab

#Following D.C. Howell: http://www.uvm.edu/~dhowell/StatPages/
#*****************************************************************************
```

B.1.0.3 Resampling regression test

```
#RegressionResample.r
#Randomization test for difference between two regression slopes

#Enter data:
library(readr)
Surveys <- read_delim("~/doks/PhD/Data/GIS/Gis/QGIS_Projects/My_Projects/Surveys/SurveysCl

#View data:
View(Surveys)
```

```
#Clean data and read into variables (vectors):
Surveys <- Surveys[!is.na(Surveys$Game_Specs),]
#Surveys <- Surveys[!is.na(Surveys$LSU_Ha),]
IndependentVar <- Surveys$Game_Specs
DependentVar <- Surveys$LSU_Total

#Viewdata (Plot):
plot(IndependentVar,DependentVar, main="Regression Plot")
cat("Correlation: ", cor(IndependentVar, DependentVar), '\n')

#Observed values:
Observed = lm(DependentVar ~ IndependentVar)
cat("Observed: ", Observed$coefficients, '\n')
ObservedSlope <- Observed$coefficients[[2]]
Negative = (ObservedSlope < 0)
cat("with an observed regression slope of ", ObservedSlope, '\n')

#Regressionline: abline(Observed)
#Resample:
nreps <- 10000 #Using nreps resampling of the data
SlopeDiff <- numeric(nreps)
sampleSize = length(IndependentVar)
cat("Will repeat resampling ", nreps, " times for ", sampleSize, " samples.\n")

for (i in 1:nreps)   {
DepResamp <- sample(DependentVar, sampleSize, replace = FALSE)
#IndResamp <- sample(IndependentVar, sampleSize, replace = FALSE)
IndResamp <- IndependentVar
RandR = lm(DepResamp ~ IndResamp)
RandSlope <- RandR$coefficients[[2]]
if (Negative)
Score <- (RandSlope < ObservedSlope)
else      Score <- (RandSlope > ObservedSlope)
if (Score)      SlopeDiff[i] = 1    else      SlopeDiff[i] = 0 }

#Results:
p = sum(SlopeDiff)
p <- p/nreps
cat("p: ", p)
#*******************************************************************************
```

B.2 Spatial Ecology (Chapter 4)

B.2.0.1 Reading GPS collar data for individual animal

```
# Libraries used:
```

```
library(rhr)
library(rgdal)
library(tlocoh)
#=======Changed for every animal=========================================== #Read data
#1. Read data (into class SpatialPointsDataFrame) from OnsiteCollarRelease or Translocated
GPS_Pred <- readOGR("~/GPSCollars/OnsiteCollarRelease", "Leopard_ml_N196_Storm_AAEAC")
#2. Read data for tLoCoH:
PROJ <- proj4string(GPS_Pred)
xys <- coordinates(GPS_Pred)
#Timestamp column:
#GPS_Pred$Timestamp <- paste(GPS_Pred$Local.Date, GPS_Pred$Local.Time)
#timestamps <- GPS_Pred$Timestamp
timestamps <- GPS_Pred$timestamp
#3. Create tLoCoH object
pred.lxy <- xyt.lxy(xys, dt = timestamps, tz=Sys.timezone(), id="N196Storm", proj4string =
#pred.lxy <- xyt.lxy(xys, dt = timestamps, tz="GMT", id="N196Storm", proj4string = CRS(PRO
#Remove bursts?
#Identify bursts
# lxy.plot.freq(pred.lxy, cp=T)
#Remove:
#pred.lxy <- pred.thin.bursts(pred.lxy, thresh=0.2)
#4. Take subset of data (if too many points => very slow)
#pred.lxy.every3rd <- lxy.subset(pred.lxy, idx = seq(from=1, to=5775, by=3)) #pred.lxy <-
#Alternative TLoCoH read:
#GPS_points <- readShapePoints("~/GPSCollars/OnsiteCollarRelease/Leopard_ml_N196_Storm_AAEA
#proj4string(GPS_points) <- PROJ
#5. Graph to see space-time balance:
#pred.lxy <= lxy.ptsh.add(pred.lxy)
#(From the graph read the value of s where the proportion of time selected hulls is betwee
#6. Find balance point (s) for frequency (time of interest 6, 8, 12, 24, 48, 72, 120, 168)
# N196: s = 0.03
S <- 0.03
A <- 31000
#"start with the k-method, see what those isopleths look like,
#and then if you see outlying points connected with really large hulls that cut across
#where the animal clearly did not go, try the a-method"
#(k = SqrRt(n), r = 1/2 x max nearest neighbour distance,
#a = max distance between 2 points) : Getz et al. 2007
#7.Select number of nearest neighbours (k)
K <- 34
pred.lxy <- lxy.nn.add(pred.lxy, s=(S), k=(K))
#8. Select nearest neighbours using auto a (at least 98% of points have K neighbours)
pred.lxy <- lxy.nn.add(pred.lxy, s=(S), a=auto.a(nnn=(K), ptp=0.98))
#t-LoCoH preparation:
#Create t-LoCoH hull with a range up to the k we used above:
#pred.lhs.k <- lxy.lhs(pred.lxy, k=3*2:8, s=0.0002)
#Create hulls around points:
#pred.lhs.k <- lhs.iso.add(pred.lhs.k)
#9.Create t-LoCoH using a:
```

```
pred.lhs.amixed <- lxy.lhs(pred.lxy, s=(S), a=28:34*1000, iso.add=T) lhs.plot.isoarea(pred
pred.lhs.amixed <- lxy.lhs(pred.lxy, s=(S), a=(A), iso.add=T)
#10. For LoCoH (without time estimate) 95% (0.95) default, add iso.levels = 1 pred.lhs <-
#Interval before a location in the same area is counted as a new visit (longest duration o
CheetahInterval <- 48
LeopardInterval <- 120 #==========================================================================
```

B.2.0.2 Output results of analysis to .shp file for GIS per individual animal

```
#=======Changed for every animal=========================================== #Save res
#Precondition: All wanted r calculations done (and libraries loaded) #Postcondition: Writt
#Save:
lxy.exp.kml(pred.lxy, "~/GPSCollars/Analysis/N196.kml")
writeOGR(iso95, "~/GPSCollars/Analysis", "N196_Isop95", driver = "ESRI Shapefile", overwri
writeOGR(iso50, "~/GPSCollars/Analysis", "N196_Isop50", driver = "ESRI Shapefile", overwri
writeRaster(ud95, filename = "N196_UD95.tif", format = "GTiff", overwrite = TRUE) #Hulls:
lhs.exp.shp(pred.lhs.amixed, id = "N196Storm", a = (A), s = (S), hpp=TRUE, hulls=TRUE, iso
```

B.2.0.3 Analysing GPS collar data (use previous 2 scripts)

```
# Libraries used:
library(rhr)
library(rgdal)
library(tlocoh)
#=======Changed for every animal=========================================
# Input data and store values in variables
# Steps 1. - 10. in hr_in.r
source("~/GPSCollars/Analysis/hr_in.r", echo = TRUE) #=================================
setwd("~/GPSCollars/Analysis")
#12. Determine smoothing/bandwidth parameter for KDE:
#Usually smoothest (Reference):
href <- rhrHref(GPS_Pred[,1:2])
#13. hlscv sometime does not converge. Smallest/most detailed:
hlscv <- rhrHlscv(GPS_Pred[,1:2])
#14. Method of Kie 2013... best?
hRefSc <- rhrHrefScaled(GPS_Pred[, 1:2])$h
#15. KDE: Could use GIS (qGIS) instead:
#1. Heatmap (with h = hRefSc, and quartic(biweight) kernel shape)
#2. Point sampling tool plugin (using original points and heatmap)
#3. Sort points, determine 50% and 95% values
#4. Use GRASS r.reclass to determine 50% and 95% rasters
#5. Polygonize (raster to vector)
#6. Remove 0 values (outside home range)
#KDE (raster):
kde95 <- rhrKDE(GPS_Pred[, 1:2], h = hRefSc, levels = 95)
kde50 <- rhrKDE(GPS_Pred[, 1:2], h = hRefSc, levels = 50)
#16. Utility Distributions (= heatmap?)
```

```
ud95 <- rhrUD(kde95)
ud50 <- rhrUD(kde50)
#17. Vector isopleths:
iso95 <- rhrIsopleths(kde95)
iso50 <- rhrIsopleths(kde50)
#18. LoCoH (a-type, n = max distance between 2 points (from GIS):
LoCoH <- rhrLoCoH(GPS_Pred[,1:2], type = "a", n = (A), levels = 95, minPts = 3) #19. 95% M
MCP95 <- rhrMCP(GPS_Pred[, 1:2], 95)
#For comparison with qGIS:
MCP <- rhrMCP(GPS_Pred[, 1:2], 100)
#20. Area:
Area95_km2 <- (rhrArea(kde95)$area)/1000000
Area50_km2 <- (rhrArea(kde50)$area)/1000000
#21.
AreaCore <- (rhrArea(core$iso)$area)/1000000
AreaMCP95_km2 <- (rhrArea(MCP95)$area)/1000000
AreaLocoH_km2 <- (rhrArea(LoCoH)$area)/1000000
#For comparison with QGIS:
AreaMCP_km2 <- (rhrArea(MCP)$area)/1000000
#22. Core as percentage of home range
core <- rhrCoreArea(kde95, method="powell90")
#23. Asymptote?
MCP_Asympt <- rhrAsymptote(MCP)
KDE_Asympt <- rhrAsymptote(kde95)
KDE50_Asympt <- rhrAsymptote(kde50)
#LoCoH_Asympt <- rhrAsymptote(LoCoH)
#=======To be updated per animal ==================================
#t-LoCoH without time = LoCoH:
#plot(pred.lhs.amixed, s=0, iso=T, record=T, ufipt=F)
#24. t-LoCoH movement corridors:
pred.lhs.amixed <- lhs.ellipses.add(pred.lhs.amixed)
pred.lhs.amixed <- lhs.iso.add(pred.lhs.amixed, sort.metric="ecc") plot(pred.lhs.amixed, is
#25. t-LoCoH (ivg: time before counting as revisit (48 hours for cheetah, 5days (120h) fo
# ivg = inter-visit gap (48 hours/120 hours):
pred.lhs.amixed <- lhs.visit.add(pred.lhs.amixed, ivg=3600*LeopardInterval)
#26. t-LoCoH spending time in one place (possible kills):
# mnlv = mean number of location (readings) per visit:
hist(pred.lhs.amixed, metric="mnlv", ivg=3600*LeopardInterval) plot(pred.lhs.amixed, hpp=T
#27. t-LoCoH repeat visits to same area: preferred habitat:
hist(pred.lhs.amixed, metric="nsv") plot(pred.lhs.amixed, hpp=T, hpp.classify="nsv", ivg=3
#28. t-LoCoH Both:
#Auto (not saved)
hsp <- lhs.plot.scatter(pred.lhs.amixed, x="nsv", y="mnlv", col="spiral", bg="black")
plot(pred.lhs.amixed, hpp=T, hsp=hsp, hpp.classify="hsp")
#29. Manually choose regions:
#1: Kill sites (Top Left), long period, no/few revisits
#2: Preferred habitat (Middle Right), Frequent revisits
#3: Regular Corridors (Bottom Right), Frequent short revisits
#4: Exploratory (Bottom Left), Short period, no/few revisits
```

```
#5: Denning (Top Right)/ Marking (Top Middle), Long period, revisits
hsp <- lhs.plot.scatter(pred.lhs.amixed, x="nsv", y="mnlv", col="spiral", bg="black", regio
pred.lhs.amixed <- lhs.hsp.add(pred.lhs.amixed, hsp=hsp)
plot(pred.lhs.amixed, hpp=T, hsp=hsp, hpp.classify="hsp")
#========Changed for every animal========================================== #30. Save
source("~/GPSCollars/Analysis/hr_out.r", echo = TRUE) #=================================
plot(kde95)
plot(kde50)
plot(KDE_Asympt)
plot(MCP_Asympt)
plot(KDE50_Asympt)
#plot(LoCoH_Asympt)
print(KDE_Asympt)
print(MCP_Asympt)
print(KDE50_Asympt)
#print(LoCoH_Asympt)
summary(pred.lxy)
#summary(pred.lhs)
summary(pred.lhs.amixed)
#plot(pred.lhs, iso=T, record=T, ufipt=F)
plot(pred.lhs.amixed, iso=T, a=(A), allpts=T, cex.allpts=0.1, col.allpts="gray30", ufipt=T
#used to find optimum value for k to exclude exploratory movements... #lhs.plot.isoarea(pre
#lhs.plot.isoarea(pred.lhs.amixed)
print(core)
print(Area50_km2)
print(Area95_km2)
print(AreaMCP95_km2)
print(AreaLocoH_km2)
#*****************************************************************************
```

B.2.0.4 Resampling mean and/or median difference between 2 groups

```
#Following D.C. Howell: http://www.uvm.edu/~dhowell/StatPages/
#Clear all data: rm(list = ls())
#Import data:
library(readr)
CollaredPredators <- read_csv("/windows/Users/boer/Documents/PhD/Data/GIS/Gis/QGIS_Project
#Look at data:
View(CollaredPredators)

#***************************************************************************** #This p
#manually excluded
# Statistic tested: Difference between mean of home ranges
#Can the observed difference in means be the result of random chance? 0 hypothesis #H1: In
#estimate of home range sizes
#Thus One-tailed test #*****************************************************************
#Select groups (specific rows) to be compared:
CollaredPredators <- CollaredPredators[which(CollaredPredators$Species == "Leopard"),]
```

```
#CollaredPredators <- CollaredPredators[which(CollaredPredators$Sex == "fm"),] #TrueHomera
#Exploratory <- CollaredPredators[which(CollaredPredators$`Treatment (T/CR/RW)` == "Onsite
TrueHR <- CollaredPredators$`HR (Concave Hull)`
Exploratory <- CollaredPredators$HR_CH_Incl_exploration
#Clean data from illegal values if required:
TrueHR <- TrueHR[!is.na(TrueHR)]
Exploratory <- Exploratory[!is.na(Exploratory)]
#Select data (columns) to be compared:
#Translocated <- Translocated$`Success?`
#OnsiteRelease <- OnsiteRelease$`Success?`

#Generic groups, so that rest of the code can remain unchanged between tests
#Group 2 chosen to have larger values (for doing 1-tailed test)
Group1 <- TrueHR
Group2 <- Exploratory
#Size of groups:
n1 = length(Group1)
n2 = length(Group2)
N = n1 + n2
#Get observed test statistic values (usually mean):
ObsTestStat1 <- median(Group1)
ObsTestStat2 <- median(Group2)
meanSmall <- mean(Group1);
meanLarge <- mean(Group2)
#For 2-tailed test: H1 = there is a difference, use absolute values
#For 1-tailed test: H1 = Group2 is larger than Group1, actual difference ObservedDiff <- (
diffObs <- (meanLarge - meanSmall) #used for mean
#Combine samples: Combined <- c(Group1, Group2) #c = concatenate

#************************************************************************** #Paramet
#While this test might be unsuitable to non-parametric data, it can be used for
#comparison with permutation/Monte Carlo test
#Pooled variances
s2p <- (var(Group1)*(n1-1) + var(Group2)*(n2-1))/(n1 + n2 - 2)
#Write results:
cat( "Larger mean observed: ", meanLarge, " with the smaller mean: ", meanSmall, '\n')
cat("The one-tailed test will be used when the alternative hypothesis is that the larger m
cat("The two-tailed test will be used when the alternative hypothesis is simply that there
cat("The observed difference between the means is = ", diffObs, "for ", n1, " and ", n2, "
cat("The pooled variance is = ", s2p, '\n')
cat("Now we will run a standard parametric t test \n")
ttest <- t.test(Group1, Group2)
cat("The resulting t test on raw data is = ", ttest$statistics, '\n')
print(ttest)
tObs <- ttest$statistic #*************************************************************
#Start resampling ********************************************************** #Create
nreps <- 10000 #Using 10000 resampling of the data
cat("Now we will resample with ", nreps, "repetitions.\n")
#Setting up arrays to hold the results:
```

```
Diff <- numeric(nreps) # For test statistic (other than mean)
meanDiff <- numeric(nreps) #For test statistic using mean
t <- numeric(nreps) # For t test values from random reshuffling of values
#set.seed(1086) #To get exact same result every time
#*********************************************************************
#Loop:
for (i in 1:nreps) {
data <- sample(Combined, N, replace = FALSE)
grp1 <- data[1:n1] #resampled group 1
grp2 <- na.omit(data[n1+1:N]) #resampled group2
meanDiff[i] <- mean(grp2) - mean(grp1)
# To be updated for every test statistic (other than mean)...
Diff[i] <- median(grp2)-median(grp1)
test <- t.test(grp1,grp2)
t[i] <- test$statistic
if((i %% 1000) == 0) cat("=") }
cat("\n")
#Optional******************************************************************* #Just to de
cat("The correlation between mean differences and t is = ", '\n') print(cor(meanDiff, t))
cat("The correlation between test stat differences and t is = ", '\n') print(cor(Diff, t))

#One-tailed test:
cat("One-tailed test\n")
cat("=================\n")
cat("The number of times when the mean difference exceeded the observed difference (diffOb
print(length(meanDiff[meanDiff >= diffObs]))
cat("\nThe proportion of resamplings when the mean diff exceeded the observed value (p) =
print(length(meanDiff[meanDiff >= diffObs])/nreps)
cat('\n')
#When using other statistic (like sum or median) instead of mean:
cat("The number of times when the difference exceeded the observed difference (diffObs) =
print(length(Diff[Diff >= ObservedDiff]))
cat("\nThe proportion of resamplings when the difference exceeded the observed value (p) =
print(length(Diff[Diff >= ObservedDiff])/nreps)
cat('\n')
cat('\n')

#Two-tailed test:
cat("Two-tailed test\n")
cat("=================\n")
#Rather than incrementing counter, count at the end (two-tailed test):
cat("The number of times when the absolute mean difference exceeded diffObs = ") absMeanD
absDiffObs <- abs(diffObs)
print(length(absMeanDiff[absMeanDiff >= absDiffObs]))
cat("\nThe number of times when the resampled t values exceeded the obtained t = ") abst <-
abstObs <- abs(tObs)
print(length(abst[abst >= abstObs]))
cat("\nThe proportion of resamplings when the absolute mean diff exceeded the observed valu
print(length(absMeanDiff[absMeanDiff >= absDiffObs])/nreps) cat('\n')
```

```
#When using other statistic (like sum or median) instead of mean cat("The number of times
absDiff <- abs(Diff)
absObservedDiff = abs(ObservedDiff)
print(length(absDiff[absDiff >= absObservedDiff]))
cat("\nThe proportion of resamplings when the absolute difference exceeded the observed va
print(length(abs(absDiff[absDiff >= absObservedDiff]))/nreps) cat('\n')

#Plots data:
#par(mfrow = c(2,2))
#Set up graphic window
#hist(meanDiff, breaks = 25, ylim = c(0,425))
#hist(t, breaks = 50, xlim = c(-5,5), ylim = c(0,425))
# The following plots the empirical distribution of t from resampling
# against the actual Student's t distribution.
# Notice the similarity even though the data are badly skewed
hist(t, freq = FALSE, xlim = c(-5,5), ylim=c(0,.4), col = "lightblue",      ylab = "resamp
par(new = T)
den <- seq(-5, 4.998, .002)
tdens <- dt(den, 36)
plot(den, tdens, type = "l", ylim = c(0,.4), xlim=c(-5,5),      col = "red", ylab="", xlab
#*****************************************************************************
```

B.2.0.5 Resampled regression

```
#Following D.C. Howell: http://www.uvm.edu/~dhowell/StatPages/
#Clear all data:
rm(list = ls())

#Import data:
library(readr)
CollaredPredators <- read_csv("/windows/Users/boer/Documents/PhD/Data/GIS/Gis/QGIS_Project

#Look at data:
View(CollaredPredators)
#***************************************************************************** #This p
# and home range size
# Statistic tested: cc of (slope and intercept) for Concave Hull home ranges
#Can the observed regression slope be the result of random chance? 0 hypothesis #H1: Rainf
# (higher rainfall, smaller HR)  #*****************************************************
#Select groups (specific rows) to be compared:
CollaredPredators <- CollaredPredators[which(CollaredPredators$Species == "Leopard"),]
CollaredPredators <- CollaredPredators[which(CollaredPredators$Sex == "ml"),] #Males <- Co
#Females <- CollaredPredators[which(CollaredPredators$Sex == "fm"),]
#Cheetahs <- CollaredPredators[which(CollaredPredators$Species == "Cheetah"),]
#Leopards <- CollaredPredators[which(CollaredPredators$Species == "Leopard"),] #Translocate
#OnsiteRelease <- CollaredPredators[which(CollaredPredators$`Treatment (T/CR/RW)` == "Onsi
#TrueHR <- CollaredPredators$`HR (Concave Hull)` #Exploratory <- CollaredPredators$HR_CH_
```

```
#Clean data from illegal values if required:
CollaredPredators <- CollaredPredators[which(!is.na(CollaredPredators$`HR (t-LoCoH 95%)`))
#CollaredPredators <- CollaredPredators[which(CollaredPredators$Premature_Fail == 0),]

#Select data (columns) to be compared into generic groups:
IndependentVar <- CollaredPredators$Rainfall_max
DependentVar <- CollaredPredators$`HR (Concave Hull)`
#DependentVar <- CollaredPredators$`95% KDE HR_Size (FK)`

#Clean data from illegal values if required:
#Males <- Males[!is.na(Males)]
#Females <- Females[!is.na(Females)]

#Plot data
#Viewdata (Plot):
plot(IndependentVar,DependentVar, main="Regression Plot")

#Size of data:
sampleSize <- length(IndependentVar)

#Observed values:
Observed = lm(DependentVar ~ IndependentVar) #Linear model(y : x)
cat("Observed (intercept slope): ", Observed$coefficients, '\n')
ObservedSlope <- Observed$coefficients[[2]]
Negative <- (ObservedSlope < 0)
cat("with an observed regression slope of ", ObservedSlope, "\n")
ObservedCorr <- cor(IndependentVar, DependentVar)
cat("Observed correlation: ", ObservedCorr, '\n')
#Regressionline:
abline(Observed)

#Start resampling ************************************************************ #Create
nreps <- 10000 #Using nreps resampling of the data
#Setting up arrays to hold the results:
SlopeDiff <- numeric(nreps)
RandCorr <- numeric(nreps)
# Done already: sampleSize = length(IndependentVar)
cat("Will repeat resampling ", nreps, " times for ", sampleSize, " samples.\n")
#set.seed(1086) #To get exact same result every time
#*********************************************************************

#Loop:
for (i in 1:nreps) {
DepResamp <- sample(DependentVar, sampleSize, replace = FALSE)
#  IndResamp <- sample(IndependentVar, sampleSize, replace = FALSE)# necessarry?
IndResamp <- IndependentVar
RandR = lm(DepResamp ~ IndResamp)
RandSlope <- RandR$coefficients[[2]]
RandCorr[i] <- cor(IndResamp, DepResamp)
```

```
#If randomised slope is more extreme than observed slope (in the same direction: 1 tailed)
if (Negative)     Score <- (RandSlope < ObservedSlope)    else     Score <- (RandSlope > Ob
#Store Score:
{  if (Score)
        SlopeDiff[i] = 1
else      SlopeDiff[i] = 0 }
if((i %% 1000) == 0) cat("*") # Show progress }
cat("\n") #*********************************************************************

#Results
p = sum(SlopeDiff)
p <- p/nreps
cat("p (slope): ", p, "\n")

#One-tailed correlation:
{if (Negative)
 pCorr1 <- (length(RandCorr[RandCorr <= ObservedCorr]))/nreps
else    pCorr1 <- (length(RandCorr[RandCorr >= ObservedCorr]))/nreps }
cat("One-tailed p (correlation): ", pCorr1, "\n")
#Two-tailed:
pCorr2 <- length(RandCorr[abs(RandCorr) >= abs(ObservedCorr)])/nreps
cat("Two-tailed p (correlation): ", pCorr2, "\n")
#*************************************************************************
```

B.2.0.6 Resampling ANOVA

```
#Following D.C. Howell: http://www.uvm.edu/~dhowell/StatPages/ #http://www.uvm.edu/~dhowel

#Clear all data:
rm(list = ls())
#Import data:
library(readr)
CollaredPredators <- read_csv("/windows/Users/boer/Documents/PhD/Data/GIS/Gis/QGIS_Project

#Look at data:
View(CollaredPredators)

#************************************************************************** #This p
# (equivalent to one way ANOVA)
# Statistic tested: F for Concave Hull home ranges grouped by Vegetation type
#Can the observed F be the result of random chance? 0 hypothesis
#H1: Predators have different home range sizes in different habitats #*******************

#Select groups (specific rows) to be compared (we compare species separately): CollaredPre
#CollaredPredators <- CollaredPredators[which(CollaredPredators$Sex == "fm"),] #Males <- C
#Females <- CollaredPredators[which(CollaredPredators$Sex == "fm"),]
#Clean data from illegal values if required:
CollaredPredators <- CollaredPredators[which(!is.na(CollaredPredators$ `HR (t-LoCoH 95%)`))
```

```
#CollaredPredators <- CollaredPredators[which(CollaredPredators$Premature_Fail == 0),]

#Select data (columns) to be compared into generic groups:
#CollaredPredators$Habitat <- paste(CollaredPredators$Vegetation, CollaredPredators$Biome,
#Groups <- as.factor(CollaredPredators$Habitat)
Groups <- as.factor(CollaredPredators$Veg_Structure)
DependentVar <- CollaredPredators$`HR (Concave Hull)`
#DependentVar <- CollaredPredators$`95% KDE HR_Size (FK)`
#DependentVar <- CollaredPredators$Rainfall_max

#Clean data from illegal values if required:
#Males <- Males[!is.na(Males)]
#Females <- Females[!is.na(Females)]
#Size of data:
sampleSize <- length(DependentVar)
n.i <- as.vector(table(Groups))
# Create vector of sample sizes
n.Groups <- length(n.i)
k <- length(n.i)

#Observed values:
model <- anova(lm(DependentVar ~ Groups)) obt.F <- model$"F value"[1]
# Our obtained F  statistic:
obt.p <- model$"Pr(>F)"
cat("The obtained value of F from the standard F test is ", obt.F, "\n")
cat("This has an associated probability of ", obt.p,  "\n")
#Plot observed data
#Viewdata (Plot):
plot(Groups,DependentVar, main="Groups:HomeRange")

#Start resampling ************************************************************ #Create
nreps <- 10000 #Using nreps resampling of the data
#Setting up arrays to hold the results:
Resamp.F <- numeric(nreps)
Score.counter <- 0
# Done already: sampleSize = length(DependentVar)
cat("Will repeat resampling ", nreps, " times for ", sampleSize, " samples in ", n.Groups,
#set.seed(1086) #To get exact same result every time
#*********************************************************************
#Loop:
updateCounter <- nreps/10
for (i in 1:nreps)  {
newScore <- sample(DependentVar)
newModel <- anova(lm(newScore~Groups))
Resamp.F[i] <- newModel$"F value"[1]
if (Resamp.F[i] > obt.F)
 {Score.counter = Score.counter + 1}
if((i %% updateCounter) == 0) cat("*") # Show progress }
cat("\n") #*********************************************************************
```

```
#Results
p <- Score.counter/nreps
cat("\nThe calculated value of p from randomized samples is ", p, "\n \n")
#Plot results:
#par(mfrow = c(2,1))
hist(Resamp.F, breaks = 50, main = "Histogram of F on Randomized Samples",      xlab = "F
cex = 0.8) legend("right",paste("p-value = ",round(p, digits = 4))) arrows( 5.5, 0.8,obt.F
f <- seq(0, 7,.01)
dens <- df(f,3,41)
par(new = T)
plot(f,dens, col = "red", type = "l", xlim = c(0,7), ylim = c(0,1), xlab = "", ylab = "")
#polygon(f,dens, col = "red")
#*****************************************************************************
```

B.2.0.7 Resampling Chi-Square

```
#Resampling "Chi-square": Test for proportion used : available

#Clear all data:
rm(list = ls())

#Functions used:
is.even <- function(x) x %% 2 == 0 # Will return TRUE if x is even, else FALSE is.odd <- f
is.available <- function(x) {if (is.odd(x)) return(1) else return(0)}
is.core <- function(x) {if (is.even((x+10)%/%10)) return(1) else return(0)}
is.hunt <- function(x) {if (is.even((x+100)%/%100)) return(1) else return(0)} is.sleepmark
proportion <- function(small, large) (small*100)/large #percentage
larger.eq.than <- function(left, right) {if(left >= right) return(1) else return(0)}
smaller.than <- function(left, right) {if(left < right) return(1) else return(0)}

#Import data:
library(readr)
PredatorHabitat <- read_csv("/windows/Users/boer/Documents/PhD/Data/GIS/Gis/QGIS_Projects/

#Look at data:
View(PredatorHabitat)
#Set working directory setwd("~/GPSCollars/Analysis")
#*************************************************************************** #This p
# habitat in their home ranges (equivalent to Chi-squared test)
# Statistic tested: Proportion used : available
#Can the observed proportion be the result of random chance? 0 hypothesis
#H1: Predators have different habitat preferences #*************************************

#Select groups (specific rows) to be compared (we compare species separately): PredatorHab

#Clean data from illegal values if required:
PredatorHabitat <- PredatorHabitat[which(!is.na(PredatorHabitat[,10])),] PredatorHabitat <-
```

```
#Create (contingency) table from observed data
#columns <- 104
#Total number of columns
columns <- ncol(PredatorHabitat)
#To be resampled later:
AvailHabitat <- apply(PredatorHabitat[10:columns], c(1,2), is.available) CoreHabitat <- ap
HuntHabitat <- apply(PredatorHabitat[10:columns], c(1,2), is.hunt)
SleepHabitat <- apply(PredatorHabitat[10:columns], c(1,2), is.sleepmark)

#Clean data from illegal values if required:

#Sums of observed data
Avail <- apply(AvailHabitat, 2, sum)
Core <- apply(CoreHabitat, 2, sum)
CoreAvail <- cbind(Avail, Core)
# View(CoreAvail)
Hunt <- apply(HuntHabitat, 2, sum)
Observed <- cbind(CoreAvail, Hunt)
# View(Observed)
Den <- apply(SleepHabitat, 2, sum)
Observed <- cbind(Observed, Den)
#View(Observed)
ObservedData <- as.data.frame(Observed)
ObservedData$CorePerc <- (ObservedData$Core*100)/ObservedData$Avail
ObservedData$HuntPerc <- (ObservedData$Hunt*100)/ObservedData$Avail
ObservedData$DenPerc <- (ObservedData$Den*100)/ObservedData$Avail

#Data into generic groups:
#df0 = AvailHabitat
#df1 = CoreHabitat
#df2 = HuntHabitat
#df3 = SleepHabitat

#Size of data
Habitats <- nrow(ObservedData)
sampleSize <- nrow(AvailHabitat)
#Observed values:
View(ObservedData)
#Plot observed data
#Viewdata (Plot):

#Start resampling ************************************************************ #Create
nreps <- 0000 #Using nreps resampling of the data
#Setting up arrays to hold the results:
cat("Will repeat resampling ", nreps, " times for ", sampleSize, " samples of ", Habitats,
CoreScore <- numeric(Habitats)
HuntScore <- numeric(Habitats)
DenScore <- numeric(Habitats)
```

```
resampledCore <- CoreHabitat
resampledHunt <- HuntHabitat
resampledSleepmark <- SleepHabitat
countSignificant <- numeric(Habitats)
#set.seed(1086) #To get exact same result every time
#*********************************************************************
#Loop:
updateCounter <- nreps/10
for (i in 1:nreps)  {
#resample:
sapply(1:sampleSize, function (row) resampledCore[row,]<<-sample(resampledCore[row,]))
sapply(1:sampleSize, function (row) resampledHunt[row,]<<-sample(resampledHunt[row,]))
sapply(1:sampleSize, function (row) resampledSleepmark[row,]<<-sample(resampledSleepmark[ro
#  resampledCore <- CoreHabitat[sample(nrow(CoreHabitat)),]
#  resampledHunt <- HuntHabitat[sample(nrow(HuntHabitat)),]
#  resampledSleepmark <- SleepHabitat[sample(nrow(SleepHabitat)), ]
#Sums of resampled data
#  Avail <- apply(AvailHabitat, 2, sum) Not changed
rCore <- apply(resampledCore, 2, sum)
resampledData <- cbind(Avail, rCore)
rHunt <- apply(resampledHunt, 2, sum)
resampledData <- cbind(resampledData, rHunt)
rDen <- apply(resampledSleepmark, 2, sum)
resampledData <- cbind(resampledData, rDen)
resampledData <- as.data.frame(resampledData)
resampledData$rCorePerc <- (resampledData$rCore*100)/resampledData$Avail   resampledData$rI
resampledData$rDenPerc <- (resampledData$rDen*100)/resampledData$Avail
#Update scores if resampled proprtion equal or larger than observed
for(x in 1:Habitats){
 CoreScore[x] <- CoreScore[x] + larger.eq.than(resampledData$rCorePerc[x], ObservedData$Co
 HuntScore[x] <- HuntScore[x] + larger.eq.than(resampledData$rHuntPerc[x], ObservedData$Hu
 DenScore[x] <- DenScore[x] + larger.eq.than(resampledData$rDenPerc[x], ObservedData$DenPe
}#inner for (p value for each habitat)
if((i %% updateCounter) == 0) cat("*") # Show progress }#for
cat("\n") #*********************************************************************

#Results:
p_corePreference <- CoreScore/nreps
p_huntPreference <- HuntScore/nreps
p_Den_markPreference <- DenScore/nreps
ObservedData <- cbind(p_corePreference, ObservedData)
ObservedData <- cbind(p_huntPreference, ObservedData)
ObservedData <- cbind(p_Den_markPreference, ObservedData)
#p <- Score.counter/nreps
for(y in 1:Habitats){countSignificant[y] <- smaller.than(ObservedData$p_corePreference[y],
countSignificant <- sum(countSignificant)
cat("\nCalculated ", countSignificant, " significant p values of core area preference from
print(ObservedData$p_corePreference)
for(y in 1:Habitats){countSignificant[y] <- smaller.than(ObservedData$p_huntPreference[y],
```

```
countSignificant <- sum(countSignificant)
cat("\nCalculated ", countSignificant, " significant p values of hunting area preference fr
print(ObservedData$p_huntPreference)
for(y in 1:Habitats){countSignificant[y] <- smaller.than(ObservedData$p_Den_markPreference
countSignificant <- sum(countSignificant)
cat("\nCalculated ", countSignificant, " significant p values of resting area preference fr
print(ObservedData$p_Den_markPreference)

#Show results
View(ObservedData)
#Export data (Update per species!)
#write.csv(ObservedData,'HabitatCheetah3.csv')
#Plot results: #?
#*****************************************************************************
```

Appendix C

Appendix: CyberTracker App for farm predation management

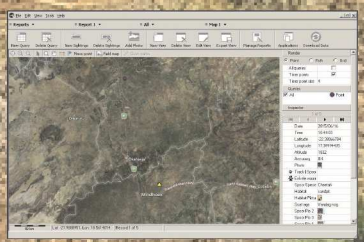
A CyberTracker app for predator management

Human-wildlife conflict, especially between livestock farmers and predators, is increasing in both South Africa and Namibia. Since arid areas are not really suitable for other agricultural activities, (excluding game farming, with its own predator issues) predators can have a significant economic effect on the profitability of farming activities in arid areas. This issue is all the more important, because some predator species might be dependent on farmlands for their long-term survival (e.g. 90% of Namibia's cheetahs are found outside formally protected areas). Predators are know to be important for biodiversity (and thus sustainability)[1]. The main predators on Namibian farms include Cheetahs, Leopards, Caracals, Black-backed Jackals and sometimes, Brown Hyaenas.

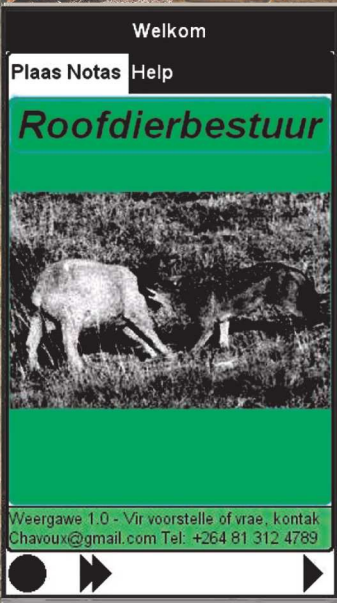
[1] Terborgh, J.W., 2015, Toward a trophic theory of species diversity, PNAS 112(37): 11415–11422.

A major problem is that the actual numbers of predators and depredation losses on livestock farms are not known, but often based on farmers' guesses or extrapolation from limited scientific studies. Farmers don't always keep good records of losses, the numbers of predators killed, or predator sightings on their land. Questionnaire surveys depend on farmers being able to give reasonable estimates, but are limited by the fact that it is difficult to know if a farmer was guessing or kept good records. Farmers are also frequently unwilling to share information with conservationists for a number of reasons. A country-wide survey sent to 2500 farmers in Namibia resulted in a response of only about 50 (1:50 response rate).

The proposed solution? A mobile application which farmers can use to record predator sightings (including tracks), livestock losses and livestock protection costs in order to get more accurate numbers. The CyberTracker ™ app can also be used by the farmer to record some general farming activities for his own farm management including date, time and GPS point for each data point. Farmers can use it anonymously with a pre-assigned App Number, without needing to divulge their personal information. The app has been set up to automatically send the data to an online ftp server from where it can be recovered and imported for analysis.



Download CyberTracker at http://www.cybertracker.org/software/free-download and the Predator app at https://dl.dropboxusercontent.com/u/72165175/RoofdierBestuur.



The application includes a method for recording tracks which can be used to identify individual cheetahs (and in future possibly leopards and Hyaenas as well) (FIT). The left hind footprint is normally used, together with 2 perpendicular rulers. For more info: http://wildtrack.org

Luyt, E.du C.[1], Leslie, A.J.[2]*

1 Farming with Predators project, Department of Conservation Ecology & Entomology, Stellenbosch University
2 Department of Conservation Ecology & Entomology, Stellenbosch University
**Chavoux@gmail.com*