Culture, Conflict and Cuisine: A Quantitative Assessment of Terrestrial Vertebrate Off-Take at the Human-Wildlife Interface

by

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Abstract

Three major forms of hunting are believed to be on the increase in the Western Cape Province of South Africa, posing independently and synergistically some of the greatest threats to the continued survival of local wildlife. Firstly, there is growing evidence of the presence and reliance of local communities on bushmeat harvesting by means of wire-snare poaching, potentially implying severe reductions or extirpations of target species, high rates of non-target off-take, and the loss of entire communities. Secondly, human-wildlife conflict poses a threat to the livelihoods and agricultural security of many stakeholders living at the interface of human development and natural habitat in the Boland, resulting in the vast eradication of damage-causing animals (DCA's). Finally, the use of animals and animal-derived materials in traditional medicine constitutes an important part of the belief-systems of indigenous African cultures, and is believed to be rapidly expanding. Due to the severity of the consequences reported elsewhere globally, and the general lack of local information with which to quantify the extent and impact of these hunting practices locally, structured interviews were conducted with farmers (n = 103) and labourers (n = 307) on private agricultural properties bordering protected areas (PA's). In addition, semi-structured interviews were conducted with traditional healers (n = 36) operating from impoverished, rural communities near PA's. Our reliance on the knowledge and experiences of local people elucidated several dynamic and interwoven social, economic and ecological factors underlying wildlife off-take, and subsequently allowed for the quantification, documentation and mapping of vertebrate off-take at the human-wildlife interface. Wire-snare poaching incidence and behaviour was strongly influenced by economic factors relating to poverty, a lack of governing regulations and punitive measures, interpersonal development, and abiotic factors such as proximity to major residential areas, roadways and PA's. Results showed that local, male farmers managing large commercial properties affiliated with regional conservancies were most likely to rely on the lethal control of DCA's. The highest level of tolerance by farmers was shown for primates and ungulates, while tolerance for carnivores, avifauna and invasive or feral species was comparatively lower. The spatial location of observed and expected zones of species-specific risk on a regional level was also mapped using a maximum entropy algorithm. We recorded 26 broad use-categories for 12 types of animal parts or products from 71 species used in traditional medicine. The most commonly sold items were skin pieces, oil or fat, and bones. To conclude, we conducted a synergistic assessment of species' vulnerability to the combined impacts of the above-mentioned hunting practices, and subsequently found that leopard, grey duiker, chacma baboon, caracal, Cape porcupine, aardvark, genet spp., and cape clawless otters experience the highest potential endangerment. This study provided the first demonstration of the multifaceted and complex nature of hunting practices in the Boland Region, opening a dialogue between local communities and conservation agencies. The primary goals being to broaden our understanding of the heterogeneity in local-scale socio-ecological dynamics, to apply policies for effective management and eradication, to prioritize areas and species for intervention, to provide for more accurate allocation of conservation resources, and to provide grounds for future research in the area and elsewhere.

Opsomming

In die provinsie Wes-Kaap in Suid-Afrika is daar tans drie dominante jagpraktyke aan die toeneem, en gevolglik bied hul selfstandig en sinergisties sommige van die grootste bedreigings vir die voortgesette bestaan van plaaslike natuurlewe. Eerstens is daar toenemende bewyse van die teenwoordigheid en afhanklikheid van plaaslike gemeenskappe op die jag van bosvleis met behulp van strikdraad-stropery, wat moontlik kan lei tot drastiese afnames of uitwissings van teikenspesies, groot volumes nie-teiken afname, en die verlies van hele gemeenskappe. Tweedens, die voorkoms van konflik tussen mens en natuur vorm 'n bedreiging vir die lewensbestaan en landboubeveiliging van talle belanghebbendes woonagtig by die koppelvlak tussen menslike ontwikkeling en natuurlike habitat in die Bolandstreek, met die gevolg dat skade-veroorsakende diere (SVD'e) uitgeroei word. Laastens, die gebruik van diere en dier-afgeleide materiale in traditionele medisyne vorm 'n belangrike komponent van die geloofstelsels van inheemse Afrika-kulture, en daar word vermoed dat die praktyk vinnig toeneem. Weens die erns van die gevolge wat elders wêreldwyd gerapporteer word, en die algemene gebrek aan inligting om die omvang van hierdie jagpraktyke plaaslik te kwantifiseer, het ons gestruktureerde onderhoude met boere (n = 103) en arbeiders (n = 307) gevoer op privaatbesit-landboueiendomme aangrensend aan beskermde gebiede (BG'e). Daarbenewens het ons semi-gestruktureerde onderhoude gevoer met traditionele genesers (n = 36) wat in arm, landelike gemeenskappe naby BG'e praktiseer. Ons vertroue op die kennis en ondervindings van plaaslike mense het verskeie dinamiese en verweefde sosiale, ekonomiese en ekologiese faktore onderliggend aan wild-afname uitgelig, en gevolglik die kwantifisering, dokumentasie en kartering van werweldier-afname by die mens-wild-koppelvlak moontlik gemaak. Die voorkoms van, en gedrag gebonde aan strik-stropery was sterk beïnvloed deur ekonomiese faktore wat verband hou met armoede, 'n gebrek aan beheerregulasies en stafmaatreëls, interpersoonlike ontwikkeling, en abiotiese faktore soos die afstand tot groot residensiële gebiede,

paaie en BG'e. Ons het gevind dat plaaslike, manlike boere in beheer van groot kommersiële eiendomme wat met bewaringsinisiatiewe op streek-vlak geassosieer is meer geneig was om letale-beheer van SVD'e uit te oefen. Verdraagsaamheid-vlakke getoon deur boere was die hoogste vir primate en hoefdiere, terwyl verdraagsaamheidvlakke vir karnivore, voëls en indringer- of rondloperspesies relatief laer was. Die ruimtelike ligging van waargenome en verwagte sones van spesie-spesifieke risiko op streeksvlak is ook gekarteer met behulp van 'n maksimum-entropie algoritme. Ons het 26 gebruikskategorieë aangeteken vir 15 soorte dierlike dele of produkte van 71 werweldier-spesies wat in tradisionele medisyne gebruik word. Die mees algemene markitems was velstukke, olies of vette, en bene. Ter afsluiting het ons 'n sinergistiese assessering van die kwesbaarheid van spesies vir die gekombineerde impak van bogenoemde jakpraktykte uitgevoer, en gevolglik gevind dat luiperd, grysduiker, Kaapse bobbejaan, rooikat, ystervark, aardvark, muskeljaatkat spp., en groototters die hoogste potensiële bedreiging ondervind. Hierdie studie het die eerste demonstrasie gebied van die veelsydige en komplekse aard van jagpraktyke in die Bolandstreek, en gevolglik 'n gesprek geopen tussen plaaslike gemeenskappe en bewaringsorganisasies. Die primêre doelwitte was dus om ons begrip van die heterogeniteit in plaaslike sosio-ekologiese dinamika te verbreed, om beleide toe te pas vir effektiewe bestuur en uitroeiing, om areas en spesies te prioritiseer vir intervensie, om voorsiening te maak vir die meer akkurate bedeling van bewaringshulpbronne, en om 'n platform te skep vir toekomstige navorsing in die streek en elders.

"…the Cape Province is largely subdivided into farms and consequently 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."

- Dr. Douglas Hey, 1964.

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"For from Him and through Him and for Him are all things. To Him be the glory forever. Amen" - Romans 11: 36

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List of Abbreviations

AIC	Akaike Information Criterion
ANOVA	Analysis of Variance
AUC	Area Under the Curve
CART	Classification and Regression Trees
CBD	Convention on Biological Diversity
CBM	Community-Based Wildlife Management
CF	Correction Factor
CFR	Cape Floristic Region
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CoCT	City of Cape Town
COMACO	Community Markets for Conservation
CoP	Conference of the Parties
CSI	Cultural Significance Index
CTA	Conditioned Taste Aversion Strategies
CV	Coefficient of Variation
DAFF	Department of Agriculture, Forestry and Fisheries
DCA	Damage-Causing Animal
DEA	Department of Environmental Affairs
EGVV	Elgin, Grabouw, Vyeboom and Villiersdorp
ENM	Ecological Niche Model
ESRI	Environmental Systems Research Institute
F_{ic}	Informant Consensus Factor
FL	Fidelity Level
GIS	Geographic Information System
GLM	Generalised Linear Regression Model
GPS	Global Positioning System
HCC	Human-Carnivore Conflict

HWC	Human-Wildlife Conflict
ICDP	Integrated Conservation and Development Project
ICF	Informant Consensus Factor
IKS	Indigenous Knowledge Systems
IPM	Integrated Strategies of Pest Management
ITCZ	Inter Tropical Convergence Zone
IUCN	International Union for Conservation of Nature
LEK	Local Ecological Knowledge
LEM	Law Enforcement Monitoring
MAP	Mean Annual Precipitation
MIST	Management Information System
NEMBA	National Environmental Management: Biodiversity Act
NGO	Non-Governmental Organization
NTFP	Non-Timber Forest Product
PA	Protected Area
RCI	Relative Cultural Importance
RF	Random Forest
ROC	Receiver Operating Characteristic
ROP	Rank Order Priority
RPL	Relative Popularity Level
RVA	Rapid Vulnerability Assessment
SES	Social-Ecological System
SMART	Spatial Monitoring and Reporting Tool
SU	Specific Uses
TDI	Tolerance to Damage Index
UBPL	Upper-Bound Poverty Line
UNESCO	United Nations, Educational, Scientific and Cultural Organisation
UV	Use-Value
WWF	World Wide Fund for Nature

Chapter 1

Background Literature Review

1.1 Ecological theory

1.1.1 Theoretical framework for ecological interactions

Human activities over the past two centuries have drastically transformed the planet and in so doing started a new era – the Anthropocene (Crutzen 2002). Concurrently, the synthesis of modern ecological understanding of ecosystem functioning gave rise to a widely accepted connectedness ideology wherein it is popularly recognized that the removal of a single species has the potential to elicit numerous responses from sympatric species (see Estes et al. 2011). Therefore, the alteration of fundamental ecosystem dynamics is now more than ever being recognized, and studied as a means of securing the survival of as many wildlife species as possible.

Ecosystem dynamics are dually driven by bottom-up and top-down control systems, and ecologists have debated their relative importance in relation to each other for decades. In this sense, control refers to a qualitative or quantitative effect on ecosystem structure, function, and diversity (Menge 1992). Many argue that bottom-up influences are most important, because removal of the first trophic level (i.e. the producers) would result in the loss of an entire ecosystem (Hunter & Price 1992). In this respect, ecosystems are regulated by energy moving upward through trophic levels, and increases in consumer biomass will thus result in increased productivity (Miller et al. 2001). This viewpoint gives little value to carnivores placed at the top of the food chain, and has led many ecologists to value them as mere ecological passengers hitching a ride atop the trophic pyramid but not actively contributing to the structure below (Estes et al. 2001). Consequently, many past management strategies that focus on the complete eradication or extirpation of carnivores from a given geographical area have been justified. It is however impossible to deny the feedback mechanisms that consumers enact on producers (Henke & Bryant 1999), and the ability of top predators to control and structure herbivore populations (Redford 1992; Estes 1996). The idea that ecosystems are shaped by apex predators was already discussed more than a century ago, but was popularized in the early 1960's (Hairston et al. 1960). In a top-down control system, carnivores regulate herbivore density, and herbivores in turn regulate plant biomass (Fretwell & Barach 1977; Oksanen et al. 1981; Fretwell 1987; Oksanen & Oksanen 2000). From here the term '*trophic cascades*' originated, roughly defined as the dissemination of predatory impacts on prey downward through trophic levels (Paine 1980). Under the paradigm of top-down control, carnivores enact important roles in regulating species interactions despite occurring in relatively low densities, and predation can even have indirect effects on flora and fauna that seem ecologically distant (Terborgh 1988; Beschta & Ripple 2009; Ritchie et al. 2012). The importance and nature of the ecosystem-regulating roles of carnivores (Prugh et al. 2009; Estes et al. 2011) depends on various factors such as population density, individual body size and metabolic constraints (Ripple et al. 2014). The removal of carnivores from the tertiary trophic level, by whatsoever means, therefore affects their social structure and directly influences their ecological role in ecosystems (Wallach et al. 2010). Without carnivores, herbivores and plants are thus released from the predatory effects enacted upon them. Recent research focus is also being extended more towards investigating the effect large carnivores have on sympatric species as mediated by their control on meso-carnivores through intraguild competition, with striking results (Prugh et al. 2009; Ritchie & Johnson 2009; Ripple et al. 2013).

Reducing trophic interactions to a dichotomous rubric of either top-down or bottom-up control is however not beneficial for the development of ecology or conservation, since it is clear that energy flows in both directions simultaneously (Hunter & Price 1992; Menge 1992; Power 1992; Estes 2001). In ecological theory, the concept of multidirectional control systems is known as '*connectivity*', which dictates that each species has within it the holistic potential to influence many other species, linking species together in a highly complex network (Estes et al. 2011). Trophic webs are also not static, but continually vary within a range of bounds to which it has adapted to over evolutionary time (Noss 1999). When trophic dynamics are tampered with, as seen with the anthropogenic removal of species on an unsustainable basis, trophic systems may fluctuate beyond the range of bounds in which it can successfully recover (Miller et al. 2001). An altered structure and function results, promoting secondary extinctions and further ecosystem instability (Miller et al. 2001).

1.1.2 Ecological implications of unsustainable wildlife removal

The pervasive impact of humans on wildlife and how it sequentially affects humans themselves, have been commented on for many decades (Redford 1992; Carson 2013). More recently, these impacts issued through anthropogenic activities such as the overexploitation of animal resources have been increasingly documented to illustrate its negative consequences on biodiversity and associated ecosystem services (Barnosky et al. 2012; Cardinale et al. 2012). It has therefore been suggested, that the human-driven extinction of medium- and large-sized vertebrates represents major global environmental change, and many areas that were once characteristic of ample wildlife have now been hunted to a state of defaunation (Redford 1992; Robinson et al. 1999). The magnitude of the decline and complete loss of animal species is however often overlooked and underestimated due to its cryptic nature, and hunting has thus been coined as the '*invisible threat*' (Phillips 1997). Fortunately, our increased ability to compile global data has allowed us to put defaunation into perspective, and has revealed that approximately 22% of the world's mammal species can now be listed as threatened or extinct (Galetti & Dirzo 2013).

Among anthropogenic activities, hunting, poaching and the illegal trade of animals or animal parts are some of the biggest drivers of defaunation, and can have lasting impacts (see Peres 2000). These impacts are often further exacerbated by accompanying land-use change which reduces the area available for natural wildlife, and isolates populations from one another (Dixo et al. 2009). As a result, numerous biological implications and lasting consequences can now be seen at the plot level (Beck et al. 2013), the local level (Harrison et al. 2013), the regional level (Steinmetz et al. 2013), the ecosystem level (Jorge et al. 2013), and possibly even up to the global level (Poulsen et al. 2013).

Carnivores in particular are largely acknowledged for their ecological, economic and social importance (Estes et al. 2011; Ripple et al. 2014), but continue to decline globally due to anthropogenic off-take (Vitousek et al. 1997; Ripple et al. 2014). The effects of removing large carnivores, known as trophic downgrading (Estes et al. 2011), are however often difficult to observe, but have nonetheless been recorded in all of the world's major biomes. Carnivores are valued for their ability to control prey species both directly and indirectly. Direct methods refer to standard predatory activities, and ultimately results in reduced prey numbers (Estes et al. 1998; Schoener & Spiller 1999), while indirect methods merely alter the behaviour of prey species (Brown 1999). Through effectively reducing the abundance of prey species, or by altering prey behaviour, carnivores create ecological boundaries that give weaker competitors the opportunity to persist (Estes et al. 2001). The anthropogenic removal of predators will thus lead to an overly simplified ecosystem (Miller et al. 2001), and therefore, also the dissolution of these ecological boundaries, increasing the respective niches of prey species and subsequently also competition prevalence. Furthermore, superior competitors may dislocate weaker competitors resulting in decreased diversity through competitive exclusion (Henke & Bryant 1999). Ultimately, decreased species diversity, biomass and interactions ensued by the removal of carnivores directly affects the balance of predator-prey interactions, and likely also the stability of the ecosystem (Galetti & Dirzo 2013) – potentially causing a perturbation reaction leading to species' population explosions or extinctions.

Small- or medium-sized predators often benefit from reduction in large carnivore numbers by increasing their home ranges and population sizes (Treves & Karanth 2003), and the removal of large carnivores will thus also likely affect mesopredators and the abundance and distribution of their respective prey species (Crooks & Soulé 1999; Henke & Bryant 1999; Schoener & Spiller 1999). This phenomenon, known as mesopredator-release, may consequently directly impact various fundamental ecosystem processes, such as seed dispersal, disease epizootics, plant biomass, plant nutrient content, and even soil porosity and chemistry (Whicker & Detling 1988; Hoogland 1995; Keesing 2000).

Conversely, the impact of prey depletion on the health of predator populations has been well documented (du Toit et al. 2004; Fusari & Carpaneto 2006; Datta et al. 2008; Lindsey et al. 2014), as well as the impact of the illegal wildlife trade on both prey and predator species (Datta et al. 2008). Predator density usually coincides with the biomass of their principle prey species across their range (Marker & Dickman 2005; Hayward et al. 2007). Their continued existence is therefore similarly threatened when principle prey species are unsustainably removed through anthropogenic means, such as the bushmeat trade, which has led to the collapse of several prey populations across savanna Africa (Lindsey et al. 2013).

1.1.3 Transformed habitats and concomitant edge effects

In every habitat on earth where agriculture has been introduced, the natural vegetation cover has been extensively modified, usually resulting in a landscape with fragmented vegetation patches (Saunders et al. 1991). Wilcove et al. (1986) defined habitat fragmentation as "*a process during which a large expanse of habitat is transformed* *into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original*". Fragmentation is often defined differently by various authors, making comparisons and consensus challenging. Most authors however agree that fragmentation is a landscape-scale process (McGarigal & Cushman 2002) that inherently leads to four distinct outcomes: (1) total habitat reduction, (2) an increased number of habitat patches, (3) decreased patch size, and (4) increased patch isolation (Fahrig 2003).

Factors associated with land-use change, such as fragmentation and habitat loss, are now widely accepted as some of the main drivers of biodiversity loss (Fahrig 2003; Pardini et al. 2010); especially since they are so easily observed (Ribeiro et al. 2009). The continued persistence of wildlife populations, particularly carnivores, depends on large, inviolate areas (Balme et al. 2010), but due to anthropogenic land-use change and subsequent fragmented landscapes, even protected areas (PA's) are unable to effectively conserve wildlife (Woodroffe & Ginsberg 1998; Balme et al. 2010). Landscape fragmentation usually has two detrimental consequences for native biodiversity. Firstly, the total available habitat is decreased, and secondly, the habitat is broken up into remnants that are isolated to varying degrees (Wilcove et al. 1986). This results in direct negative consequences for biodiversity such as decreases in species richness (Steffan-Dewenter et al. 2002), population distribution and abundance (Best et al. 2001), and genetic diversity (Gibbs 2001), as well as indirect factors affecting biodiversity, such as reproductive success (Robinson et al. 1995).

Due to the effects of fragmented landscapes on key population parameters such as reproduction and survival, the dynamics of populations occurring in fragmented landscapes may also be strongly affected by edge effects (Yahner 1988; Saunders et al. 1991; Murcia 1995). Fragmentation exposes remaining organisms to different conditions in surrounding ecosystems (Saunders et al. 1991), which thus leads to pronounced edge effects near and on the border of different habitat types (Murcia

1995); such as the interface between natural areas and converted agricultural land or urban settlements. Edge effects are defined as the junctions between two landscape elements, and can either be a distinct border or a transition zone (Yahner 1988). Typically, there are three types of edge effects on fragment peripheries, namely: (1) abiotic effects i.e. changes in environmental conditions due to the adjacent dissimilar matrix, (2) direct biological effects i.e. changes in species abundance and distributions, and (3) indirect biological effects i.e. changes in species interactions (Saunders et al. 1991; Murcia 1995). Edge effects are however rarely quantified (Revilla et al. 2001), mainly because there is no standardized protocol in use (Yahner 1988). As a result, the edge effect phenomenon remains poorly understood. For example, Leopold (1933) reported an increased diversity of wildlife at edges, which he concluded was either due to the availability of more than one habitat at edges or the variety of vegetation at edges. Similarly, Yahner (1988) concluded that the creation of more edge in landscapes will always have a positive effect on wildlife. Contrastingly however, edge effects are often noted as detrimental to wildlife, especially to specialist and rare species (Woodroffe & Ginsberg 1998). Carnivores therefore may be particularly sensitive to edge effects caused by anthropogenic activities due to their extensive spatial requirements and low densities (Schonewald - Cox et al. 1991; Clarke et al. 1996; Woodroffe & Ginsberg 1998), and are consequently at greater risk of extirpation (Purvis et al. 2000; Cardillo et al. 2005).

In modern times, most wildlife species reside within and/or adjacent to PA's which are threatened with increasing pressure from human population expansion (Wittemyer et al. 2008). Human encroachment and activities in peripheries can therefore lead to more pronounced edge effects, limiting wildlife populations within PA's (Pulliam 1988; Woodroffe & Ginsberg 1998) or drawing individuals to attractive sinks outside of PA's (Loveridge et al. 2010). The majority of wildlife mortality occurs when animals range beyond the edges of natural areas to encounter people directly

(Castley et al. 2002; Schwartz et al. 2006; Loveridge et al. 2007). In this landscape, human-induced mortality may be unintentional, such as road kill accidents, but the majority seems to be deliberate, such as hunting and poaching activities (Revilla et al. 2001). Consequently, population sinks tend to form around the peripheries of PA's, and if mortality is not balanced with reproduction and immigration in these areas, edge effects are bound to become more pronounced and potentially push the resident wildlife population to extinction (Woodroffe & Ginsberg 1998). In response, buffer zones are often constructed along with the erection of fences, to lessen edge effects and illegal activities in PA's, but the efficacy of these measures are frequently questioned (Geldmann et al. 2013; Lindsey et al. 2014; Durant et al. 2015).

The transformation of natural global landscapes to human-dominated landscapes over the last three centuries (Ellis et al. 2010) has seen competition between humans and wildlife for space and resources reach unprecedented levels (Bulte & Rondeau 2005). Therefore, as the human population continues to increase, resulting in a sharp increase in the conversion of natural habitat to agricultural areas (Brink & Eva 2009), more urgent quantifications and descriptions are needed to be able to understand edge dynamics and to efficiently conserve edge-species.

1.2 Involving local communities in conservation

Conservation groups have traditionally placed little emphasis on the importance of people in conservation compared to ecosystems or species (Mascia et al. 2003), but modern conservation ideology acknowledges the necessity of involving people in conservation programmes (Fa et al. 2000; Robinson & Sasu 2013), especially with the increased recognition shown by scientists for the important role social factors play in determining the success or failure of conservation and management interventions (Mascia et al. 2003; Knight & Cowling 2007). Similarly, the importance of directly involving local communities and cultures in conservation and management is rapidly emerging (Hulme & Murphree 1999; Berkes et al. 2000), as local communities are the immediate custodians of their natural resources (Bowen - Jones et al. 2003), and have the potential to directly assist in the conservation or extinction of wildlife (Child et al. 1997). Conservation actions inherently exist as a product of human decision-making processes, and therefore a change in the behaviour of people is needed for conservation actions to succeed (Knight & Cowling 2007).

This newfound appreciation of social involvement in conservation has subsequently led to the development of a wide array of multidisciplinary and transdisciplinary techniques aimed at encapsulating interactions between people, ecosystems and biodiversity. For example, conservation ideals are often enclosed in the framework of a socio-ecological system (SES) (Jochum et al. 2014). Similarly, many researchers increasingly rely on local ecological knowledge (LEK) and indigenous knowledge systems (IKS) to accrue information relevant to ecological conservation (Gadgil et al. 1993; Berkes et al. 2000). These systems acknowledge the fact that empowering people through conservation initiatives is essential for developing and implementing effective conservation strategies (Smith et al. 2009; Miller et al. 2011). Furthermore, to translate ecological research into applied management strategies, the input of the general publics' perceptions is needed, and hence the incorporation of consultation and education into management planning (Chase et al. 2000; Schusler & Decker 2002; Holsman & Peyton 2003).

1.2.1 Questionnaires in ecology

Central to the incorporation of local communities in conservation is the use of questionnaires as a standardized method to gather ecological information, and the application thereof in ecological research has significantly increased in the last few decades (Drury et al. 2011; St John et al. 2016). The use of questionnaires is traditionally a social science method, but has become particularly valuable in the field of ecology when information is required from a specific human target population in order to test research hypotheses (Bennett et al. 2017; Young et al. 2018). For example, in the context of this thesis, questionnaires were remarkably effective tools for collecting data on public or stakeholder perceptions, data covering a large geographic area and population group, data regarding anthropogenic impacts on wildlife, and data spanning interdisciplinary fields that include both human and non-human aspects (White et al. 2005). Researchers are acknowledging the limitations in traditional monodisciplinary approaches in confronting and prioritizing issues in environmental conservation and management (Riley et al. 2002; O'Connor et al. 2003), and questionnaires are vital in transforming research approaches to an interdisciplinary state which includes human behaviour, attitudes and perceptions (Obiri & Lawes 2002; Liu et al. 2011; Schumann et al. 2012). Questionnaires as an indirect data collection tool further reduces the strain on monetary and time-related resources, especially in large-scale studies (Reading et al. 1996; Macdonald & Johnson 2000; Vaughan et al. 2003). Additionally, direct research means such as field surveys can only provide data on current impacts or management regimes, and lacks the ability to provide information regarding past impacts or management strategies (White et al. 2005).

Questionnaire studies all start by contacting a subset of the target population and asking them to participate in a questionnaire by providing information, after which the information that is collected from the respondents is analysed to test specific hypotheses. The method of data collection can vary depending on factors such as the time and cost constraints and the hypotheses being tested. The most popular forms of questionnaire data collection includes postal surveys, face-to-face interviews, telephone conversations, and web-based surveys. These methods all offer specific advantages and disadvantages, most of which can however be minimized by adequate methodological design (Gomm 2008).

White et al. (2005) reviewed a total of 168 questionnaires used in 127 papers published between 1991 and 2003, and found the mean (±SE) sample size (number of respondents per questionnaire) was 1 422 ± 261 (279 ± 111 for face-to-face interviews, 1 962 ± 383 for postal surveys and 854 ± 357 for telephone interviews). They also found the mean response rate was highest for face-to-face interviews (97.5 ± 1.4%) and lowest for postal surveys (51.9 ± 2.7%). Postal surveys are therefore not ideal, even though they represent the most economical approach to collecting questionnaire data (Weisberg et al. 1996). They are furthermore susceptible to problems associated with self-selection and non-response bias (Lewis & Oppenheim 1992). While many factors can affect response rates, a low response rate typically suggests that the issue researched is not of particular importance to the target population (see Lindner 2002). Conversely, face-to-face interviews are regarded as the preferable sampling method overall in questionnaire methodology despite their expensive nature (Arrow et al. 1993).

Questionnaires can further be divided into closed-format questions (dichotomous, multiple choice, Likert scale or rating scale) or open-ended questions. In general, closed-format questions are preferred as they result in less uncertainty than open-ended questions for both the researcher and the respondent (White et al. 2005). Well-designed open-ended questionnaires can however be valuable in assessing behaviour or thought processes of the respondents (White et al. 2005), but this type of information is generally easier obtained from in-depth interviews rather than survey-type questionnaires (Gomm 2008). Additionally, available sampling procedures include random, systematic, comprehensive, and opportunistic. It is important to thoroughly consider the sampling procedure that will be adopted in any potential study. An incorrect sampling procedure for a particular research question can nullify any conclusions arising from the study since it would be difficult to evaluate the reliability of the data or whether the respondents were sufficiently representative of the target population (White et al. 2005).

Questionnaires embrace a new research paradigm as interdisciplinary tools that are able to mix ecological data with socio-economic or political data. In this way, questionnaires are able to reflect the positivist approach to social science, which follows the traditional approach of natural sciences with the emphasis of gathering quantitative information while the interviewer remains objective and detached (Neuman 1997).

1.2.2 Face-to-face interviews

Interviews have been conducted for centuries, and was at first account defined as an "*interchange in which one person… attempts to elicit information for expressions of opinion or belief from another person or persons*" (Maccoby & Maccoby 1954). This form of mutual learning can range from short conversations to long detailed exchanges that can be repeated over time (Fontana & Frey 2005; Young et al. 2018), and are particularly valuable in filling a knowledge gap (Minichiello et al. 2008) by providing interviewee's perspectives and knowledge on certain topics.

Advantages of face-to-face interviews (Bernard 2006) include: (1) the ability to include people who could not otherwise provide information e.g. illiterate, disabled or hard-to-reach people, (2) the ability to explain a question when a respondent does not fully understand it, (3) the ability to use combinations of several data collection techniques e.g. include visual aids and cue cards, (4) focus on a particular question since the respondent does not have the ability to page through the questions to see what is coming, and (5) knowledge of who the respondent was. Disadvantages include their intrusive nature, high running costs, high travel and time requirements, and possible interruption of data collection due to the extended period of research.

Interviews either follow a structured or unstructured approach, or a variation thereof (semi-structured) based on the amount of control an interviewer wants over people's responses (Spradley 1979). Structured interviews rely on a fixed set of pre-defined questions, and the same script is repeated in each consecutive interview, allowing close comparison between resulting answers (Punch 2013). Contrastingly, unstructured interviews allow the interviewer to spontaneously shape the discussion based on the answers given by the respondent (Bryman 2016). One advantage of such an approach is that it limits pre-conceived researcher bias, but it typically does not cover all relevant issues and presents a problem for comparative data analysis (Bryman 2016). To get the most out of both approaches, many researchers enjoy using a semi-structured approach (Dunn et al. 2003). During a semi-structured interview pre-defined questions are asked, allowing comparison between respondents, but they also allow the interviewer to ask additional questions (Young et al. 2018).

In the end, the sampling strategy that is adopted should be based on the sample size required to be able to collect meaningful and robust data that sufficiently represents the target population (St John et al. 2016). Potential sampling strategies (Newing 2010) include snowball-sampling (the use of a small pool of initial informants to nominate other participants who meet the eligibility criteria of the study), theoretical sampling (key patterns are identified after initially interviewing a few people, after which further participants are approached based on emergent themes), key informant sampling (knowledgeable people are targeted), representative sampling (a stratified sample is chosen to represent the total population), and random sampling (people are spoken to at random). Additionally, some researchers prefer a focus-group approach in which issues are discussed among a small group of respondents. The main advantage of this approach is the opportunity to observe and document a large amount of interaction on a topic in a limited time period, providing direct evidence on similarities and differences in respondent opinions and behaviour, while individual interviews allow for the opportunity to form a more intimate bond with the respondent (Morgan 2011). In terms of popularity, Young et al. (2018) found in a review of 227 interview-based articles published since 2011, an average sample size of 87 with a median of 35; significantly lower than the abovementioned results of White et al. (2005). They also found most interview studies relied on face-to-face interview methods (> 60%) and key informant sampling strategies (46%).

1.2.3 Ethical considerations

St John et al. (2016) states that many conservation researchers undertake questionnaire studies without thoroughly considering the issues surrounding confidentiality and anonymity (non-disclosing the identity of respondents), informed consent (assuring respondents understand the overall purpose of the study and how the information provided by them will be utilized), and compensation (providing some form of tit-for-tat recompense for the information given). However, it is essential that no study involving human participants, questionnaire-based or otherwise, be undertaken without first obtaining ethical clearance from the relevant governing bodies (Silverman 2013), as well as all respondents to the study. Additionally, consideration should be given to the sensitivity of the questions and the level of personal intrusion (Young et al. 2018).

1.3 Bushmeat harvesting

People living in close proximity to natural or PA's often rely on the hunting of wildlife as bushmeat to sustain basic livelihood needs such as food security (Rentsch & Damon 2013). However, the unsustainability of bushmeat harvesting has been well-documented in the literature (Barnett 1997; Fa et al. 2002; Loibooki et al. 2002; Mainka & Trivedi 2002; Robinson & Bennett 2002). Bushmeat harvesting for both commercial as well as self-use purposes is viewed as one of the largest threats to mammals (Fa et al. 2002), particularly in Africa (Knapp 2012; Taylor et al. 2015). The collective consensus drawn from these studies indicate a decline in wildlife as a direct result of bushmeat poaching, and in more extreme cases, extirpation of wildlife species (Wilkie & Carpenter 1999). Direct evidence of the unsustainability of the bushmeat trade is however difficult to obtain, and subsequently most available data consists of qualitative perceptions and observations (Bowen - Jones et al. 2003). Illegal hunting for bushmeat is believed to be predominantly driven by increasing human encroachment of natural wildlife habitat (Kiringe et al. 2007), poverty and associated food insecurity (Brashares et al. 2011), a lack of clear rights over wildlife or property (Lindsey et al. 2013), inadequate legal protection for wildlife (Lindsey et al. 2011a), political instability (Bouché et al. 2012), and cultural demands (Lindsey et al. 2013).

Based on overall trade volume globally, mammalian species, and in particular ungulate species, are the most desired form of bushmeat, with small antelope species such as duikers often predominating (Bowen - Jones et al. 2003; Fa et al. 2005; Taylor et al. 2015). Primate species are the second largest constituent of the trade, while rodent species such as porcupine, as well as other smaller-bodied mammals, are becoming increasingly important elements of the bushmeat trade due to the decrease in large-bodied mammal incidence worldwide (Jones-Bowen & Pendry 1999; Fa et al. 2005). In the Western Cape Province, very few large ungulate species still roam the area without human intervention (Coetzee 2016), and we can thus expect to find local dependency on smaller wildlife species.

Several underlying patterns and trends following bushmeat poaching and trade have been elucidated in the primary literature. Predominantly, bushmeat harvesting is either attributed to economic (Adams et al. 2004; Ndibalema & Songorwa 2008) or cultural (Loibooki et al. 2002; Ndibalema & Songorwa 2008) drivers, and researchers have debated the relative importance of those drivers in relation to each other for decades. From an economic perspective, it is well established that poachers are predominantly poor, uneducated people (Knapp 2009), while culturally poachers are pressured to hunt for the sake of prestige, tradition, or camaraderie (Bitanyi et al. 2012). Furthermore, while economic and cultural factors may shape the ultimate driving forces of bushmeat harvesting, the decision to poach is likely further motivated by a suite of proximate or stochastic short- and long-term factors that may differ substantially between environments. For example, drought conditions (Kaltenborn et al. 2005), resource value, and human-wildlife conflict (Knapp 2009) have also been correlated with increases in bushmeat poaching. Additionally, poaching incidence is believed to be further motivated, at least to some extent, by human virtues such as skill development, boredom, thrill seeking, or authority antagonism (Forsyth & Marckese 1993; Eliason 2003; Knapp 2009).

Several previous studies have also highlighted numerous spatial and temporal patterns associated with bushmeat poaching incidence. Firstly, bushmeat consumption tends to be concentrated around the edges of PA's (Campbell et al. 2001; Knapp et al. 2010; Lindsey et al. 2011a), especially in areas where wildlife numbers have drastically decreased in unprotected habitats (Newmark 2008). Secondly, bushmeat harvesting tends to increase close to residential areas (Hofer et al. 2000; Wato et al. 2006; Marealle et al. 2010), as well as close to areas where wildlife numbers concentrate, such as near water bodies, game trails, and fruiting or flowering trees (Lindsey & Bento 2012; Becker et al. 2013). Finally, hunting tends to peak significantly during the late dry season (Brown 2007; Holmern et al. 2007) and following crop harvest periods (Lindsey et al. 2011a).

In 2011, the problem surrounding illegal bushmeat harvesting was formally acknowledged when the Convention of Biological Diversity (CBD) established a liaison group on bushmeat (COP 11 Decision XI/25). However, finding solutions to lessen the dependency of people on bushmeat is not only of conservation concern, as many people living in rural areas depend largely on hunting, and sometimes also the commercialized trade of bushmeat for food security and income (Nasi et al. 2008; Lindsey et al. 2011a), essential for their survival (Bowen‐Jones et al. 2003; Lindsey et al. 2011a). This often generates conflict between the competing goals of wildlife protection and social sustainability (Rentsch & Damon 2013), preventing the implementation of successful mitigation initiatives. For research on bushmeat to comply or keep abreast with international development policy, a more socio-economic focus is required (Davies and Brown 2007). An interdisciplinary approach of this nature will further allow for a better understanding of the dynamics associated with bushmeat harvesting and provide means for the analysis of the dual objectives set forth in bushmeat mitigation strategies, namely to conserve biodiversity as well as to protect the livelihoods of the people involved in the bushmeat trade (Bowen‐Jones et al. 2003). Previously, outdated and unenforced policies and legislative frameworks for the bushmeat trade was not conducive to the sustainable management of natural wildlife resources (Bowen‐Jones et al. 2003). In recent years, bushmeat legislation has progressed to increasingly accommodate wildlife needs (CoP17 2016), but enforcement and compliance with updated legislative frameworks remain an issue (Davies and Brown 2007). Additionally, the lines of ownership regarding land and wildlife possession, and in particular bushmeat, is often blurred as a result of political instability in many African nations (Bouché et al. 2012), inarguably resulting in increased poaching activities as local communities believe bushmeat has a *res nullius* (without ownership) status (Bulte & Horan 2002).

A vast compilation of studies concerned with aspects of bushmeat harvesting have focused on the greater ranges of West and Central Africa (Barnes 2002a; Fa et al. 2002; Bowen‐Jones et al. 2003; Bennett et al. 2007; Brashares et al. 2011), including specified studies in areas of Gabon (Wilkie & Carpenter 1999; Wilkie et al. 2005), Ghana (Asibey

1974; Cowlishaw et al. 2005), Ivory Coast (Hofmann et al. 1999), Nigeria (Anadu et al. 1988), Equatorial Guinea (Fa et al. 2000; East et al. 2005), and Cameroon (Edderai & Dame 2006), as well as the Central African Republic (Noss 1998) and locations in East Africa, namely Kenya (Fitzgibbon et al. 1995; Okello & Kiringe 2004; Wato et al. 2006), the Serengeti (Hofer et al. 2000; Loibooki et al. 2002; Ndibalema & Songorwa 2008; Marealle et al. 2010; Nuno et al. 2013; Rentsch & Damon 2013), and elsewhere in Tanzania (Nielsen 2006; Wilfred & MacColl 2010; Nyaki et al. 2014; Hegerl et al. 2017). In southern Africa, bushmeat research has been conducted in Namibia (Vaughan & Long 2007), Zambia (Lewis & Phiri 1998; Lewis & Lusaka 2005; Brown 2007; Lewis 2007; Lewis et al. 2011; Becker et al. 2013), Mozambique (Fusari & Carpaneto 2006; Lindsey & Bento 2012), and Zimbabwe (Lindsey et al. 2011a, 2011b; Gandiwa et al. 2013), but reliable data quantifying the extent and scope of bushmeat poaching in South Africa is noticeably lacking (Robinson & Bennett 2002; Lindsey et al. 2011a). Most likely, the lack of research on the topic can be attributed to the general misconception that hunting in South Africa is a sustainable subsistence practice (Barnett 1997). However, even in reasonably affluent countries such as South Africa and Botswana, illegal bushmeat harvesting poses a significant threat to the survival of wildlife populations (Rogan et al. 2015). In the Western Cape Province, numerous anecdotal and observational records suggest that the illegal hunting of bushmeat is becoming progressively apparent (CapeNature data; unpublished), yet a complete informational void exists in the area, thus prohibiting the formation of adequate conservation and management efforts.

Currently, anthropogenic pressures continue to intensify globally as a result of the increasing human population continually encroaching wildlife areas (Kiringe et al. 2007), leading to more rural communities living close to or on the border of PA's (Rentsch & Damon 2013) and increased conversion of pristine habitat for agricultural land-use (Alexandratos & Bruinsma 2012; Rosenblatt et al. 2016). An increase in resource requirements is thus expected, together with an increase in the intensity of threats posed to native wildlife (Wilfred 2010). In the Western Cape Province, similar increases in anthropogenic pressures are expected to result in encroachment of wildlife habitat and the eventual loss of biodiversity (Statistics South Africa 2012). Furthermore, the widespread presence of forestry enterprises (Tewari 2001; Western Cape Forestry Sector Forum 2018) will continue to exacerbate these pressures by continuously causing human influxes to the province (Poulsen et al. 2009) and the conversion of pristine habitat for logging concessions (Armstrong & Van Hensbergen 1996).

1.3.1 Wire-snare poaching

In 2013, Lindsey et al. found illegal wire-snares to be the most prevalent method used for the hunting of bushmeat in South Africa, posing an intense threat to local ecosystems. The popularity of this hunting method is likely founded on its inexpensive and obscure nature (Lindsey et al. 2011a), as well as its ability to successfully capture a wide range of taxa (Noss 1998). Depending on the material, wire-snares have been reported to effectively capture anything ranging from rodents to African elephants (*Loxodonta africana*) (Hofer et al. 2000), and is easily obtained in abundance from amongst others, reserve, farm or garden fences (Lindsey et al. 2011a); making the regulation of snare-hunting extremely difficult (Lindsey et al. 2013). Wire-snares can be made from a wide variety of materials (Lindsey et al. 2011a), but typically, these snares comprise of a piece of wire or cable that is rolled into a noose and anchored to vegetation or fences to catch game (Fig. 2.1) (Hofer et al. 1996; Noss 1998).

In recent years the use of wire-snares have however become progressively unacceptable across a wide range of social classes (Treves & Naughton-Treves 2005), predominantly because of the ethical apprehensions linked to the injuries animals sustain through escaping snares, as well as its delayed efficacy. Lewis & Phiri (1998) argue that wire-snare poaching incidence can be considered a direct indication of the degree of conservation support given by local communities, since increased stewardship of wildlife is directly linked to increased revenue manufactured through community-based conservation. However, the influence of bushmeat market value, the historical or cultural influence prevailing in local communities, and personal motivations cannot be denied (Watson et al. 2013). Further devastation of wire-snares stems from their indiscriminate nature, as they frequently kill or injure non-target and charismatic species (Lindsey et al. 2011b; Vanthomme et al. 2017); inadvertently upsetting functional plant-animal relations (Wright 2003) and ecosystem stability (Galetti & Dirzo 2013). Predator species such as leopard (*Panthera pardus*) and caracal (*Caracal caracal*), which form the bulk of top-down control pressures in the Boland (Martins & Harris 2013), are particularly vulnerable to being captured as by-catch because of their tendency to frequently visit areas of high prey densities, as well as areas with trapped animals or carcasses (Lindsey et al. 2013). Because of the low cost associated with producing snares, hunters also tend to neglect checking on them and often desert them entirely, resulting in unnecessary animal mortalities (Noss 1998). Impacts associated with wire-snare poaching have been shown to include severe reductions or extirpations of target species, high rates of non-target off-take of threatened species, and the loss of entire wildlife communities (Hofer et al. 1996; Lindsey et al. 2011a, 2013; Becker et al. 2013). Consequently, the use of wire-snares is specified as a criminal offence under the Western Cape Nature Conservation Ordinance (No. 19 of 1974).

Research on the extent, distribution, trends and patterns of wire-snare hunting is urgently needed to quantify the socio-economic and ecological impacts (Lindsey et al. 2013), and to allow for the development of efficient management strategies. Such information is however not easily obtained due to the inherent nature of snare poaching, which includes its concealed appearance, low densities compared to landscape spread, heterogeneity, and highly skewed statistical distribution of snare detection data (Becker et al. 2013). An efficient methodological approach is thus required to evaluate trends and patterns in snaring to mitigate poaching activity.

1.3.2 Legislation

Hunting and poaching activities in the Western Cape Province are primarily governed by the Western Cape Nature Conservation Ordinance (No. 19 of 1974), the Western Cape Nature Conservation Laws Amendment Act (No. 3 of 2000), the National Environmental Management: Biodiversity Act (No. 10 of 2004), and the Western Cape Biosphere Reserves Act (No. 6 of 2011), with the former predominating. Under the governance of these documents, enforced by local authoritative and managing bodies, the use of wire-snares to kill or capture any wild animal is specified as a criminal offence. As a result, convicted transgressors may be liable for a fine of up to R10 000, or two years of imprisonment, if a protected species is caught. Alternatively, a fine of up to R100 000, or 10 years of imprisonment, may be awarded if an endangered species is caught in a snare. Furthermore, the possession or selling of the carcass of a dead animal without a valid permit is deemed illegal, and similar penalties will be awarded if a person is found guilty.

1.4 Human-wildlife conflict

In many parts of the world, wildlife frequently cross reserve boundaries into landscapes where they come into contact with local communities (Inskip & Zimmermann 2009). At this interface, people and wildlife compete for the same resources, and inevitably, conflict arises (Woodroffe et al. 2005). These conflicts are collectively referred to as human-wildlife conflict (HWC), and is formally described as an interaction where people and wildlife compete for shared resources in a common environment (Inskip & Zimmermann 2009), typically occurring when "*the needs and* *behaviour of wildlife impact negatively on the goals of humans or when the goals of humans negatively impact the needs of wildlife*" (Madden 2004).

In recent years the existential, economic and ecological value of naturally occurring wildlife is being increasingly recognized and emphasized, providing impetus for biodiversity conservation. Nonetheless, the persecution of both genuine and perceived damage-causing animals (DCA's) is now viewed among the leading world-wide threats to the continued existence of these species (Treves & Karanth 2003; Graham et al. 2005; Inskip & Zimmermann 2009; Stein et al. 2010). The incidence of HWC and the associated losses are essentially economically and spatially heterogeneous (Hill 2000), and is thus particularly substantial and disproportionate for small-scale and subsistence farmers close to PA's, therefore challenging a synergy between rural development and biodiversity conservation (Dar et al. 2009; Loveridge et al. 2010). Additionally, the frequency and associated costs of conflict appears to be on the increase in many areas across the world (Karanth 2002; Treves et al. 2002; McManus et al. 2015), most notably in areas where human populations, farming frontiers and urban developments continue to grow and expand their limits (Woodroffe 2000; Treves & Karanth 2003). As a result, wildlife is forced out of historic ranges into small, fragmented landscapes (Pettigrew et al. 2012; Kiffner et al. 2015), or into conflict with people. In turn, the livelihoods of people, particularly those depending on crop or animal husbandry as their main source of income (Barua et al. 2013), are negatively affected (Marker & Dickman 2005); leading to the widespread classification of numerous wildlife species as "*pests*", "*vermin*" (Beinart 2008), or "*problem animals*" (Linnell et al. 1999).

The National Environmental Biodiversity: Management Act (NEMBA SA, No. 10 of 2004) defines a damage-causing or problem animal as "*a wild vertebrate animal that, when interacting with humans or interfering with human activities, there is substantial proof that it causes losses to stock or other wild specimens, causes damages to cultivated trees, crops,* *natural flora or other property, presents a threat to human life, or is present in such numbers that agricultural grazing is materially depleted*" (DEA 2010; Brassine 2011). Linnell et al. (1999) describes two categories of problem animals, depending on the scale of the agriculture-wildlife distribution matrix. The first type refers to an animal in a coarse-grained matrix, where agricultural activities do not overlap with the home range of the individual, that essentially finds itself "*in the wrong place at the wrong time*" during the process of dispersing between territories or from a PA into human-dominated landscapes, or encounters people in a natural setting. The second type is an animal occurring in a fine-scale matrix, that has a preference for agricultural produce (e.g. crops or livestock) or humans, usually to a greater extent than conspecifics. The definitions underlie the assumption that HWC is confined to agricultural landscapes, but this is not always the case, as seen in the urban settlements of the Cape Peninsula (Hoffman & O'Riain 2012). It further suggests a scenario wherein only a small proportion of individuals in any given population is responsible for conflicts of interest with humans; a verified reality that is rarely acknowledged (Avenant & du Plessis 2008).

The attitudes and tolerance of people towards DCA's may vary significantly, and can largely influence the likelihood of an individual relying on lethal persecution measures (Schumann et al. 2012). For example, Marker et al. (2003) reported that of a group of farmers in Namibia experiencing conflict with cheetahs, 84% removed the cheetahs while 16% chose not to – begging the question as to why inconsistencies in terms of tolerance and attitudes exist among seemingly homogeneous groups of people? It has been established that the degree of tolerance is often related to the magnitude of the damage incurred, as witnessed in the study by Marker et al. (2003), wherein it was evident that more cheetahs were removed by landowners perceiving them as '*problem animals*' than those who did not consider them to be a problem. The relationship between damage and tolerance is however not always linear, and it is impossible to deny the influence of several other socio-political factors on tolerance levels and general attitudes toward wildlife. Hostility towards conflict species is also often based on historical or cultural attitudes, and further influenced by past experiences and personal values (Loveridge et al. 2010). Religious beliefs, income and education levels are also large contributors (Liu et al. 2011). Therefore, people may vary in the level of damage that is tolerated, with some people not tolerating any damage – whether real or perceived (MacGowan et al. 2006). It is therefore important to examine and acknowledge these differences when conservation strategies and policies are being developed.

Apart from experiencing losses due to damaged crops and depredated stock, HWC encounters have also resulted in significant injuries or fatalities to people. Although these encounters are not exceptionally common on a global scale, they erratically occur in mostly low-income countries (Das & Chattopadhyay 2011). In India for example, 295 people were killed by leopards from 2000 to 2007 (Marker & Sivamani 2009), while more than 400 people are killed by elephants annually (Rangarajan et al. 2010). In Mozambique and Namibia, crocodiles are responsible for the deaths of more than 100 people annually (Lamarque et al. 2009), and in Tanzania, 563 people were killed by lions from 1990 to 2005, while 308 added people sustained serious injuries (Packer et al. 2005). Numerous other species such as snakes, hippopotamuses and tigers are also feared internationally (Bonnet et al. 1999; Lamarque et al. 2009). Furthermore, free-ranging wildlife is responsible for a large number of vehicle collisions, resulting in fatalities to both wildlife and people (Morelle et al. 2013), and disease transmission between wildlife and domestic animals further contributes to the prevalence of conflict events (Wilkinson et al. 2004).

Less conspicuous and poorly documented than direct impacts such as fatalities, crop and stock losses are the indirect or '*hidden*' impacts associated with HWC. Indirect impacts are defined as "*psychological or social costs that are typically uncompensated for and temporally delayed*" (Ogra 2008). Indirect impacts are often also termed '*secondary* *impacts*' due to their delayed onset, which usually results from direct impacts of HWC. For example, injuries or fatalities caused by HWC may eventually result in diminished states of psycho-social wellbeing, which may further result in unemployment or livelihood burdens shifting to women and children (Chowdhury et al. 2008; Dixon et al. 2009; Jadhav & Barua 2012). Similarly, crop or livestock losses may eventually result in the disruption of families, livelihoods and food security or agricultural sustainability (Dixon et al. 2009; Barua et al. 2013).

1.4.1 Human-Carnivore Conflict (HCC)

Conflicts with wildlife include a plethora of terrestrial species extending from invertebrates to birds, but conflict with large charismatic carnivores are the most frequently reported (Inskip & Zimmermann 2009). Consequently, carnivores are often viewed as undesirable at local level (Woodroffe et al. 2005), despite being widely regarded as flagship species at global and national levels (Treves & Karanth 2003) due to the important regulatory roles they fulfil in terrestrial ecosystems (Estes et al. 2011). Human-carnivore conflict (HCC) has therefore greatly contributed to the distribution and population declines that carnivore species experienced during the previous century (Woodroffe & Ginsberg 2000), resulting in human persecution being listed as the main threat to carnivores outside of PA's (Friedmann & Daly 2004).

Carnivores are primarily drawn into human-dominated landscapes because of their protein-rich diets and extensive home ranges (Treves & Karanth 2003), and are therefore especially prone to conflict with people, who often share their spatial and dietary requirements (Linnell et al. 2001). This is especially true for males of a particular species, mainly owing to their larger home range requirements, higher dispersal rates and probable larger body size (Linnell et al. 2001; Bunnefeld et al. 2006; Loveridge et al. 2010). Consequently, HCC is particularly rife in non-protected areas bordering reserves (Thorn et al. 2013), where a conflict of interest arises (Woodroffe &

Ginsberg 1998; Stein et al. 2010). Furthermore, carnivores are widely perceived as inimical to animal farming, and as a result they tend to experience the worst consequences of HCC in areas where animal production is prevalent (Treves & Karanth 2003; Graham et al. 2005). Conflicts on agricultural properties often arise due to carnivores preying on domestic livestock or game (Marker et al. 2003; Treves & Karanth 2003; Woodroffe & Frank 2005; Thorn et al. 2012), the incidence of which may be particularly exacerbated when anthropogenic activities have depleted natural prey (Graham et al. 2005; Gusset et al. 2009). Among the most infamous carnivores responsible for conflicts with humans globally are the big cats, namely the African lion (*Panthera leo*), tiger (*Panthera tigris*), leopard (*P. pardus*), jaguar (*Panthera onca*), snow leopard (*Panthera uncia*), puma (*Puma concolor*), and cheetah (*Acinonyx jubatus*) (Inskip & Zimmermann 2009). Felids in particular are however, often cited to prefer preying on natural wildlife to avoid human retribution, despite the greater stocking densities of livestock (Loveridge et al. 2010). It is therefore only when natural prey, typically medium- and large-sized ungulates become scarce due to population declines, that increases in depredation rates are observed (Polisar et al. 2003; de Azevedo & Murray 2007; Mondal et al. 2011; Zhang et al. 2013; Kabir et al. 2014). Smaller carnivore species are also frequently involved in competition with humans over small game, fish, and poultry (Reynolds & Tapper 1996; Freitas et al. 2007). The body size of the predator typically determines which livestock or game species are vulnerable to depredation. For example, smaller predators such as caracal typically prey on smaller livestock size classes such as sheep and goats, while intermediate-sized predators such as leopard have the added ability to prey on juveniles of the larger livestock size-classes, such as cattle. Large predators such as lion have no restrictions on prey size-classes (Loveridge et al. 2010).

The impact of retaliatory hunting on wildlife populations is patently detrimental, but it is however impossible to deny the adverse effects these populations often have on the livelihoods of local communities, and in particular pastoralists. For this reason, the intentional killing of DCA's is often self-justified (Thorn et al. 2012). Due to the opportunistic feeding behaviour of carnivores and the high vulnerability of domesticated stock, carnivores sometimes partake in surplus killings (events wherein a large number of domestic animals are killed in one incident, but only a few are fed on) (Marker & Dickman 2005; Bothma & Walker 2013). Carnivores are also characteristically susceptible to a wide array of infectious diseases (Murray et al. 1999), many of which can be directly transmitted to game and livestock. Consequently, landowners may experience serious adverse effects on their economic security (Treves & Karanth 2003; Graham et al. 2005), and respond with the retaliatory or preventative persecution of carnivores (Sillero-Zubiri & Laurenson 2001; Woodroffe et al. 2005). Furthermore, the total cost of losses due to HCC, as well as the cost of control measures, can be substantial. For example, the total cost of depredation to the livestock industry in South Africa was estimated to be between USD 22 million (Statistics South Africa 2010) and USD 171 million annually (Van Niekerk 2010). The cause of the great range disparity is uncertain and undoubtedly questions the integrity of the methodologies employed to arrive at the estimated net sum. However, even the lowest range estimate indicates high losses to carnivores, fuelling lethal control of DCA's by farmers. It is also worth noting that research into actual and perceived predation has revealed that carnivores are frequently blamed by landowners for more losses than they actually cause, either mistakenly or deliberately (Mishra 1997; Sillero-Zubiri & Laurenson 2001; Chavez et al. 2005; Rigg et al. 2011; Boulhosa & Azevedo 2015). The pursued DCA's are also frequently misidentified (Naughton - Treves et al. 1998; Linkie et al. 2007). Several studies have attempted to quantify the extent of HCC in Southern and Eastern Africa, and consequently reported a mere loss of 1.4% of total stock holdings to large carnivores in Namibia (Marker et al. 2003), compared to 1.8% in Kenya (Kolowski & Holekamp 2006), 2.2% in Botswana (Schiess - Meier et al. 2007), 2.8% in South Africa (Thorn et al. 2013), 4.5% in Tanzania (Holmern et al. 2007), and 5.0% in Zimbabwe (Bagchi & Mishra 2006). Scat analysis of species typically blamed for depredation further revealed that livestock only contributed to minor proportions (< 1.5%) of their diets (Norton 1986; Grey et al. 2017). This suggests that retaliatory responses are not proportional to the extent of damage incurred, as previously also demonstrated across Southern Africa (Constant 2014).

1.4.2 Conflict with crop-raiding species

Compared to HCC, research on conflict with crop-raiding species have been far less abundant. This is likely because of the impression that herbivores are less important ecologically and that their populations are generally stable (Parker et al. 2007). However, despite the lack of available information, crop damage is the most prevalent form of HWC in both Asia and Africa (Parker et al. 2007), and is largely responsible for shaping public perceptions of conservation as it threatens the general well-being of people (Lee & Graham 2006). Crops are particularly attractive to wild animals because of the effect selective breeding has had on the natural physical and chemical defences of crops, as well as their nutritional value (Purseglove 1968). Crops also offer energetic advantages compared to many natural foods, especially during times of food scarcity (Forthman-Quick and Demment 1988; El Alami et al. 2012). The utilization of crops by crop-raiding species however often results in increased levels of intra- and interspecific aggression, especially towards humans (Hsu et al. 2009; El Alami et al. 2012). Furthermore, the loss of crops has a substantial impact on people's livelihoods, food security and agricultural security (Barua et al. 2013). An urgent need thus exists to fill the gap in available knowledge on conflict with crop-raiding species.

In sub-Saharan Africa, baboons (*Papio* spp.) are amongst the large mammals that cause the single most damage to crop production in agricultural landscapes (Ripple et al. 2014), and in many areas baboon raids require families to withhold children from attending school in order to help guard planted crops (Brashares et al. 2011). Baboons exhibit a suite of traits particularly adept to exploiting human-modified landscapes, for instance semi-terrestrial locomotive abilities, adaptability, flexible and diverse diets, large and complex social demographics, advanced intelligence, manual dexterity and agility, and boldness (Else 1991; Knight 1999; Strum 2010). Consequently, nonhuman primates, particularly baboons, are responsible for the most complex and large-scale conflicts in the realm of HWC, reported all across Africa (Strum 1994; Kansky & Gaynor 2000). Currently, human-baboon conflict is expected to increase with human population expansion and increased land development (Hoffman & O'Riain 2012). In the Cape Peninsula of South Africa, for instance, baboons frequently engage in conflict with local people (Beamish 2010; Hoffman & O'Riain 2012), resulting in extensive damage to properties and financial losses due to crop raiding (Kaplan et al. 2011). They are also frequently reported to exhibit intimidating behaviour towards urban people in order to obtain alternative food sources (Hoffman & O'Riain 2011), all of which often results in human-induced injuries or mortality to baboons (Beamish 2010). Consequently, management often relies on the euthanasia of target individuals (Kansky & Gaynor 2000). Legislation to protect baboons has however been on the rise (South Africa 2001), leading to more innocuous control methods, such as regimented waste management (Kaplan et al. 2011), public education, electrified fences, and human shepherding and monitoring of troublesome troops close to urban areas (Kansky & Gaynor 2000) (see http://www.baboonmatters.org.za).

Similarly, elephants across both Asia (*Elephas* spp.) and Africa (*Loxodonta* spp.) are notoriously known for their ability to destroy cultivated land and crops, food stores, fences, and artificial water sources such as water tanks (Hoare 1999; Parker & Osborn 2006; Taruvinga & Mushunje 2014); often causing communities to lose up to 10-15% of their total agricultural output (Lamarque et al. 2009; Madhusudan & Sankaran 2010). Conflict with elephants was probably already a major hindrance and evolutionary driver of arable farming in precolonial Africa (Parker & Graham 1989; Barnes 1996), and elephants have continued to be in conflict with farmers for most of the twentieth century (Eltringham 1990; Barnes 1996). Human-elephant conflict as a primary concern has however previously been overshadowed by other issues related to elephant conservation, such as the overpopulation of elephants in national parks (Barnes 1983) and the illegal hunting of elephant populations for ivory (Spinage 1973; Parker & Graham 1989), but the increased prevalence of human-elephant conflict in the last few decades (Kangwana 1995) has shifted the attention of researchers and conservationists. The cause of the spike in conflict can likely be attributed to increasing elephant densities, accompanied with a shrinkage of their available range (Barnes et al. 1995). As a result, elephants are now widely regarded as pests in many areas (Taruvinga & Mushunje 2014). Despite their reputation however, elephants are an integral component in terrestrial ecosystems where they are largely responsible for the structuring of vegetative cover and diversity (Wing & Buss 1970; Chapman et al. 1992; Kahumbu 2002). Efforts aimed at mitigating conflict with elephants have been largely unsuccessful (Hoare 1999), despite the predictability of their raiding-activity, which usually corresponds with the maturation of food crops, thus spiking markedly during the late wet season in their savanna ranges and at night (Hoare 1995).

1.4.3 Mitigation

Finding solutions for alleviating HWC is pivotal to the survival of carnivore and other wildlife species, but a clear understanding of the extent and drivers of HWC is essential for any interventions to be successful. Adequate conflict mitigation strategies should draw on knowledge concerning wildlife behavioural ecology and public perceptions of wildlife management (Treves & Karanth 2003), thus incorporating both human and biodiversity conservation interests. It is further necessary to incorporate accumulated empirical knowledge and context-specific local experiences. For intervention tools to be able to balance these two aspects sustainably they should thus be persistently efficient, include minimal unintended environmental consequences, be selective towards problematic animals, incur a lower cost than that of the depredation prevented, and be socially acceptable (McManus et al. 2015).

Traditionally, landowners in South Africa have resorted to the lethal control of DCA's through the use of predominantly gin-traps, gun-traps, poison, and hunting with or without dogs (Macdonald et al. 2010). Despite growing scientific evidence highlighting the limitations associated with the use of lethal control methods, such as undesired demographic responses (Prugh et al. 2009), the common misconception that lethal control is the cheapest and most effective method for managing conflict species is still widely accepted (Conover 2001; Mitchell et al. 2004). Increased efforts to balance both the needs of people and wildlife have however led to a considerable proportion of research regarding HWC, focussing on finding alternative solutions to manage problem animals that do not result in lethal outcomes (Treves et al. 2009). The ultimate advantage of non-lethal control initiatives is that it fails to elicit social perturbation on population demographics like lethal control methods, meaning that, although certain behaviours may be altered (e.g. food aversion through conditioned taste aversion strategies – CTA's), individuals are allowed to remain in their territories, thus maintaining ecological relationships (Prugh et al. 2009). Contrastingly, many non-lethal methods are criticized as being nothing more than palliative (Barnes 2002b), implying that they alleviate rather than eradicate problems (O'Connell-Rodwell et al. 2000). Popular non-lethal interventions include the collaring of livestock (Schiess-Meier et al. 2007) with dead-stop-, king-, bell-, cell-phone-, bell and smell-, protective sheep-, and smart technology collars, as well as the planting of buffer crops (Thouless 1994), fencing and kraaling (corralling) (Taylor 1999; Dickman 2010), traditional shepherding or herding (Shivik 2006; Dickman 2010), the use of fladry lines (Davidson-Nelson & Gehring 2010), translocations (Bradley et al. 2005; Athreya et al. 2011), a variety of chemical, visual, olfactory or acoustic repellents (Mason 1998), and guarding animals such as dogs (*Canis lupus familiaris*), donkeys (*Equus africanus asinus*), alpacas (*Lama pacos*) and llamas (*Lama glama*) (Conover 2001; Crawshaw 2004). Additionally an assortment of management practices can be applied, such as seasonal lambing coordination, breed selection, breed mixtures, and stock rotation. There is however rarely a single panacea to resolving the conflict. Instead, a combination of several strategies are usually needed to ameliorate the impact of HWC (Distefano 2005).

1.4.4 Legislation

The draft norms and standards regarding the hunting of DCA's are primarily provided by the National Environmental Management: Biodiversity Act (NEMBA SA, No. 10 of 2004), while additional governance regarding hunting may be obtained from the Western Cape Nature Conservation Ordinance (No. 19 of 1974), the Western Cape Nature Conservation Laws Amendment Act (No. 3 of 2000), and the Western Cape Biosphere Reserves Act (No. 6 of 2011). In summary, these documents provide that DCA's should first be treated with methods to prevent or mitigate recurring damage, and such methods should be proportionate to the extent of damage incurred. These methods include live capture for translocation, euthanasia, or keeping in captivity for registered breeding or scientific research purposes, as well as the direct killing of the DCA, except in instances where the damage caused can be ascribed to human error or negligence. Additionally many methods exist that are designed to deter DCA's, and can be used freely without first obtaining a permit (see 1.4.3). The use of cage traps, poison collars, darting, soft traps and dogs may only be used by an adequately trained professional, and after a permit is obtained. Permits are also required to hunt endangered species (e.g. riverine rabbit or blue crane) any time of the year, and protected species (e.g. guineafowl and duiker) outside of their hunting season. A zero-quota on the hunting of leopards was extended in 2014 (DEA 2014), and therefore no permit can be obtained to hunt damage-causing leopards. Unprotected wild animals are defined as having healthy populations, such as porcupine and rock rabbit, and may be hunted without first obtaining a permit, provided that a hunting licence and written permission from the landowner is first obtained. A thorough explanation for the use of each of the above-mentioned methods is available in the original documents, including general provisions, operational arrangements, minimum requirements, and permit-use.

1.5 Traditional cultural medicine: Ethnopharmacology

Natural resources such as plant and animal parts or derivatives have been incorporated into traditional medicines by various cultures for millennia (Lev 2003), and many of these uses continue to form a part of modern-day ethnomedicine practice. The pharmacopoeia of traditional medicines is made up of herbal, faunal and mineral material that is actively used for physiological as well as symbolic or psychological purposes (Cunningham 1991). Direct harvesting of natural resources for human consumption however poses a unique problem for biodiversity (Cunningham & Zondi 1991; Pimentel et al. 1997), especially when rare or endangered (EN) species are preferred (Mander et al. 2007).

Despite of the lack of formal records, the trade and procurement of natural products in South Africa is deemed to be substantial (Shackleton 2009; Whiting et al. 2011), and pharmacopoeias of traditional healers in South Africa consist of a wide array of natural products (Berglund 1976). In South Africa, traditional medicine typically falls into two distinct classes. The first is used in the treatment of medical afflictions, and referred to as '*white medicine*'. Conversely, the second is termed as '*black magic*', and is used for dealing with spiritual or ancestral conflict (Bye & Dutton 1991). Health and welfare issues are considered to be intimately connected with supernatural forces, social relationships and ancestral relationships by traditional healers in Southern Africa and many other African cultures (Berglund 1976; Bye & Dutton 1991; Simelane 1996). Traditional medicines are highly valued by these cultures and are regularly used (McKean et al. 2013). Per capita, South Africa has few western doctors (Williams et al. 2007). Contrastingly, traditional healers are far more abundant, outnumbering western doctors 2000:1 in some parts of the country (Levitz 1992). As a result, a large proportion of the country's population (60 – 80%) (Cunningham 1991; Mander 1998; Philander 2011) believe in the efficacy of traditional medicine and have at some time either purchased traditional medicine or consulted with a traditional healer (Cunningham & Zondi 1991; Mander et al. 2007). This is particularly true for communities in rural areas, where less opportunity exists to consult with university-educated doctors, while traditional healers in comparison are easily accessible (Bye & Dutton 1991), culturally more familiar, and able to treat ailments that western doctors can't. Almost 60% (55.5%) of South Africans live below the upper-bound poverty line (UBPL) (Statistics South Africa, 2015), and traditional medicine thus also often offers a solution when pharmaceutical drugs are too expensive (Cunningham 1991). As a result, far more African laymen are responsible for treating illnesses and other ailments than have been recognized in the past (Hardon 1994).

1.5.1 Ethnozoology

Ethnozoology refers to the interrelationships between human cultures and animals, and often constitutes zootherapeutic practices; roughly referred to as the healing of human ailments with the use of traditional non-Western medicine, specifically through the use of whole animals, animal parts, or animal-derived products (Costa-Neto 1999). Zootherapy and associated practices have a rich history that has existed in traditional folk medicinal practices and rituals in many cultures throughout history (Lev 2003; Betlu 2013). For example, Chinese culture have relied on the harvesting and use of bear gall bladders for over 1 300 years (Li et al. 1995), while the use of rhinoceros horn in treating various medical afflictions dates back to more than 2 000 years ago (But et al. 1990). Interestingly however, there is no clear evidence of the use of either animal parts or animal products for medicine in prehistoric times (Lev 2003). Instead, evidence of prehistoric societies limit their reliance on animals to the making of tools, consuming animals as food, making clothes, and as sacrifices for religious purposes (Holland 1994). Nonetheless, historical information on ancient Egypt mention the use of animal derivatives such as cattle milk, bee honey and lizard blood in medicine (Stetter 1993; Lev 2003), while in archives of ancient Mesopotamia fish oil, bee wax and honey, mongoose blood and animal fat are recorded (Thompson 1923; Ritter 1965).

For traditional healers, medicine requires a strong symbolic meaning (Berglund 1976; Bye & Dutton 1991; Simelane 1996). Therefore the choice of animals used in traditional medicine is often found to be based on the complete or partial resemblance of animal body forms or behaviour to specific parts of the human body, organs, or responses due to ailments (Williams & Whiting 2016). This phenomenon, known as the Doctrine of Signatures, thus poses that corresponding body parts or behaviour (such as diurnal or nocturnal movements, feeding habits, reproduction, and prey capture) can be exploited to treat disorders relating to similar body parts or behaviours (Voeks 1996; Lev 2002; Douwes et al. 2008; Pandita et al. 2016). For example, South African healers often prescribe the bones of baboons to patients suffering from arthritis, due to the dexterity exhibited by these animals (Pujol 1990). Similarly, the vast number of feet and articulated body segments of centipedes results in Korean traditional healers prescribing them to treat leg, foot, and joint ailments (Pemberton 1999). In Tswana-culture, crocodile skin is prescribed for treating fevers because the aquatic crocodile symbolizes cooling-off (Krige 2009), and in Zulu and Xhosa cultures, physical strength of animals such as lions, pythons, elephants, buffalos and crocodiles are often thought to be transferrable to humans to impart comparable strength to overcome adversity (Williams & Whiting 2016). The Doctrine of Signatures is also occasionally referred to as '*suggestive forms*' (Hutchings 1989) or the Doctrine of Correspondence (Pandita et al. 2016). Traditional healers often also rely on animal colouration as indicators of potential treatments (Krige 2009). For example, Cuttlefish (*Sepia* spp.) is used in teeth discoloration treatments due to their white skeletal colour (Pandita et al. 2016).

Traditional healers in South Africa can be broadly divided into two categories, namely *diviners* and *herbalists* (classification and terminology provided during interviews). Diviners (referred to as '*witchdoctors*' in Western culture, but also known throughout African cultures by the Zulu-word *isangoma,* and Xhosa-word *amgqirha*) use a supernatural approach to make diagnoses and treat ailments, while herbalists (referred to as either *isinyanga* [Z] or *amaxwhele* [X]) dispenses traditional medicines (*umuthi*) made from natural substances derived from plant, animal, and mineral materials (Ngubane 1977). Diviners are often also trained as herbalists, and can practice both vocations either simultaneously or separately (Hutchings 1989; Krige 2009). In many instances no distinction is made between the two vocations, as in both instances a '*divine calling*' is received (Mtshali 2004). Although '*fake sangomas*' are notorious in communities, traditional healers are commonly viewed with high regard in their respective communities (Hutchings 1989).

In the Western Cape, the use of animals in traditional medicine or cultural practice is largely dominated by Xhosa-speaking people (Petersen et al. 2014), and, as with many African cultures, traditional medicine constitutes an important component of indigenous Xhosa people's culture (Simelane & Kerley 1997, 1998; Cocks & Dold 2000). Xhosa medicines, *Amazeya esiXhosa*, are predominantly herbal medicines, but a strong emphasis is placed on animal parts or products either incorporated alongside herbal medicines or used separately (Cocks & Dold 2000). One of the first documentations of animals used by Xhosa communities dates back to the 1930's (Cawston 1933), but the practice is likely much older. Xhosa people had no contact with Western doctors and medical procedure prior to the 19th century (Simon & Lamia 1991), hence they had to rely on the abilities and knowledge of local healers. Similar to other indigenous African cultures in South Africa, such as the Zulu, Sotho and Tswana denominations, traditional Xhosa healing and other cultural practices place an equal value on the use of animal constituents and derivatives for the curing and prevention of non-medical issues, such as protection against bad luck and witches (Anyinam 1995). In fact, it is becoming increasingly evident that animal parts are mainly used for '*symbolic magical purposes*' and the treatment of '*magico-medical*' ailments, with only a few exceptions (Cunningham & Zondi 1991). Animal-derived medicine is further also used for protection against physical and spiritual enemies and entities, as love charms and aphrodisiacs, for increased intelligence, to acquire wealth and prosperity, and to aid in pastoral activities (Simelane 1996; Cocks & Dold 2000; White et al. 2004; Mander et al. 2007). Unlike the more obvious and prominent *muthi* markets found in Johannesburg and Durban, managed by transient commercial harvesters, hawkers and traditional healers (Williams et al. 1997, 2007; Mander 1998; Whiting et al. 2011; Williams & Whiting 2016), traditional healing in the Western Cape is much more concealed and discreet (Petersen et al. 2014). One likely contributing factor for this trend is the absence of vacant premises for setting up *umuthi* stores in townships, resulting in the majority of traditional healers consulting patients at home (Williams 1996).

1.5.2 Existing research and research paucity

The importance of animals in traditional medicine to indigenous communities was already formally acknowledged in 1950 by Watt and Breyer-Brandwijk. Compared to ethnobotanical research however, ethnozoological studies have been largely subjected to paucity (Herbert et al. 2003; Betlu 2013), and in most existing research, only baseline data have been collected (Whiting et al. 2011). Research on traditional medicinal plants has been conducted for > 200 years (Williams & Whiting 2016), and subsequently more than 2 000 botanical species used or traded as traditional medicine have been inventoried (Williams et al. 2013). In comparison, ethnozoological studies in South Africa only started increasing in the past few decades amid growing concerns that the hunting and trade of animals for traditional purposes is largely unsustainable (Williams & Whiting 2016), and subsequently a mere 232 vertebrate species have been enumerated (Whiting et al. 2011). The large dedication to ethnobotanical research is likely due to its potential in advancing both health as well as accompanying economic sectors (Herbert et al. 2003), while the lack of ethnozoological data in comparison is mainly attributed to its small claim on the greater spectrum of the *Materia Medica* of indigenous cultures (Betlu 2013). Furthermore, the large-scale association of ethnozoology with '*witchcraft*' and the Doctrine of Signatures (Lev 2002), especially in Africa, have withdrawn credibility from zootherapeutics as a realistic scientific pursuit (Williams & Whiting 2016). Bioprospecting for new medicines has therefore placed far greater emphasis on traditional medicine of botanical origin, since the spiritual component of traditional zootherapeutics cannot be accurately translated into scientifically screened and medically approved patent medicines (Williams & Whiting 2016).

Despite research on ethnozoology in South Africa being largely sporadic and subject to neglect (Betlu 2013), there has been an upsurge in available information in the last few decades, drastically improving our knowledge on the subject. For example, Williams & Whiting (2016) reported on the use of animals at the Faraday traditional medicine market in Johannesburg, South Africa. During this study they recorded uses for 52 terrestrial vertebrate species and 18 morphospecies (i.e. typological species that can only be identified as genet spp., eagle spp., snake spp. etc.); 42% of which were mammals, 29% reptiles and 8% birds. Of the individual species, the most cited uses were for the Southern African python (*Python natalensis*), Cape porcupine (*Hystrix africaeaustralis*), chacma baboon (*Papio ursinus*), Nile crocodile (*Crocodylus niloticus*), and African elephant (*Loxodonta africana*). Uses attributed to morphospecies lizard (incl. *Cordylus, Smaug, and Agama* spp.), jackal (*Canis* spp.), tortoise (incl. *Kinixys, Stigmochelys,* and *Chersina* spp.), and monitor lizard (*Varanus albigularis* and *V. niloticus*) were also frequently cited. By virtue of the diversity of their uses, these species are thus likely in danger of being overharvested. Animals were predominantly used to acquire increased physical or mental strength and for protection against evil spirits, as well as for medical treatment of skin problems, headaches and strokes. An earlier study by Whiting et al. (2011) listed 147 vertebrate species at the Faraday market. Of those, rock and water monitor, Nile crocodile, Southern African python, puff adder (*Bitis arietans*), chacma baboon, Cape porcupine, vervet monkey (*Chlorocebus pygerythrus*), and warthog (*Phacochoerus africanus*) were the species most traded (> 50% of stalls). In other South African studies, Simelane & Kerley (1998) reported 44 vertebrate species being sold at 19 herbalist shops in the Eastern Cape Province, compared to 79 (Cunningham & Zondi 1991) and 132 vertebrate species in Kwazulu-Natal Province (Ngwenya 2001). In all instances mammals were the most commonly traded taxon. Reptiles are generally not well recognized or understood by indigenous African communities (Simelane & Kerley 1997), and consequently have much lower harvest rates compared to mammal species. Among birds, vultures are among the most frequently traded species in South Africa (Whiting et al. 2011), and based on recent evidence it has been suggested that traditional use is at least partly responsible for the rapid decline of vulture populations in the country (McKean et al. 2013). See also: McKean (1995) (Kwazulu-Natal), Simelane (1996) (Eastern Cape) and Derwent & Mander (1997) (Kwazulu-Natal).

The demand for animal products for traditional cultural uses is both extensive and expanding (Hutchings 1989, 1996; Cunningham & Zondi 1991; La Cock & Briers 1992). This is in part stimulated by rapid urbanization and high levels of unemployment, resulting in an increasing number of people with no other income resorting to the harvesting and sale of animals to support their livelihoods (Cunningham 1991). The animal components used in ethnozoology are secretive and often illegal in South Africa (McKean et al. 2013), as many of these species are presented on the IUCN Red List of threatened species (Williams & Whiting 2016), thus posing major conservation implications (Simelane & Kerley 1998). Negative stigma surrounding the selling and buying of such products is also common, and the markets are largely unregulated by provincial and national authorities (Castells and Portes 1989). Furthermore, the immigration rate of Xhosa people from the Eastern Cape Province into the Western Cape Province has drastically increased in the past few years (Jacobs 2014), potentially exerting novel and unexamined pressures on local wildlife populations.

1.5.3 Legislation

The socio-economic structure in South Africa can be viewed as being dually divided into formal and informal contexts, both spatially and legally (May & Meth 2007). Traditional medicine markets mainly occur in the informal section of the South African economy, where a blind eye is often given to activities, such as medicinal trade, that are heavily regulated by societal institutions in formalized economies through structured, legal frameworks (Castells and Portes 1989). The overbearing framework for legalisation regarding traditional medicinal practices is found in the overarching Traditional Health Practices Act (No. 22 of 2007), and aided by general hunting regulations set forth in the Western Cape Nature Conservation Ordinance (No. 19 of 1974) and the Western Cape Nature Conservation Amendment Act (No. 3 of 2000). These regulatory frameworks aim to ensure the efficacy, safety and quality of traditional health care services, and thus recognises traditional healing as a legal health practice in South Africa. The current policy however only allows western medicine to be practiced officially, but traditional medicine is not prohibited (Krige 2009). While the practising of traditional medicine is not illegal *per se*, several species involved in the practice are endangered or protected animals, and are thus illegally harvested.

1.6 Greater study area

1.6.1 Location

The study encompasses the Cape Winelands and Overberg districts, collectively referred to as the central and southern divisions of the Boland Region, part of the

Western Cape Province of South Africa. The area (Fig. 1.1) is roughly 4 000 km^2 in size, and contains within its core various open and closed provincial nature reserves or PA's. Most notably is the Nature Reserves of Limietberg, Hottentots-Holland, Jonkershoek, Theewaters, Hawequa, Groenlandberg, and Kogelberg (managed by the public statutory conservation agency CapeNature), as well as the Nature Reserves of Helderberg and Steenbras (managed by the City of Cape Town Biodiversity Management – CoCT). The study area also overlaps with two UNESCO designated Biosphere Reserves, namely the Cape Winelands- and Kogelberg Biosphere Reserves, and is part of the Cape Floristic Region (CFR) of South Africa. The CFR is a globally recognized biodiversity hotspot and one of the world's six floral kingdoms (Myers

1990), containing exceptional plant diversity and endemism (Cowling & Holmes 1992).

The reserves and PA's in the study area are almost exclusively enveloped by transformed lands and are thus seen as biodiversity islands with few corridors remaining between them. For a lot of charismatic and functionally important species these critical refuges are thus a last stronghold for their continued existence.

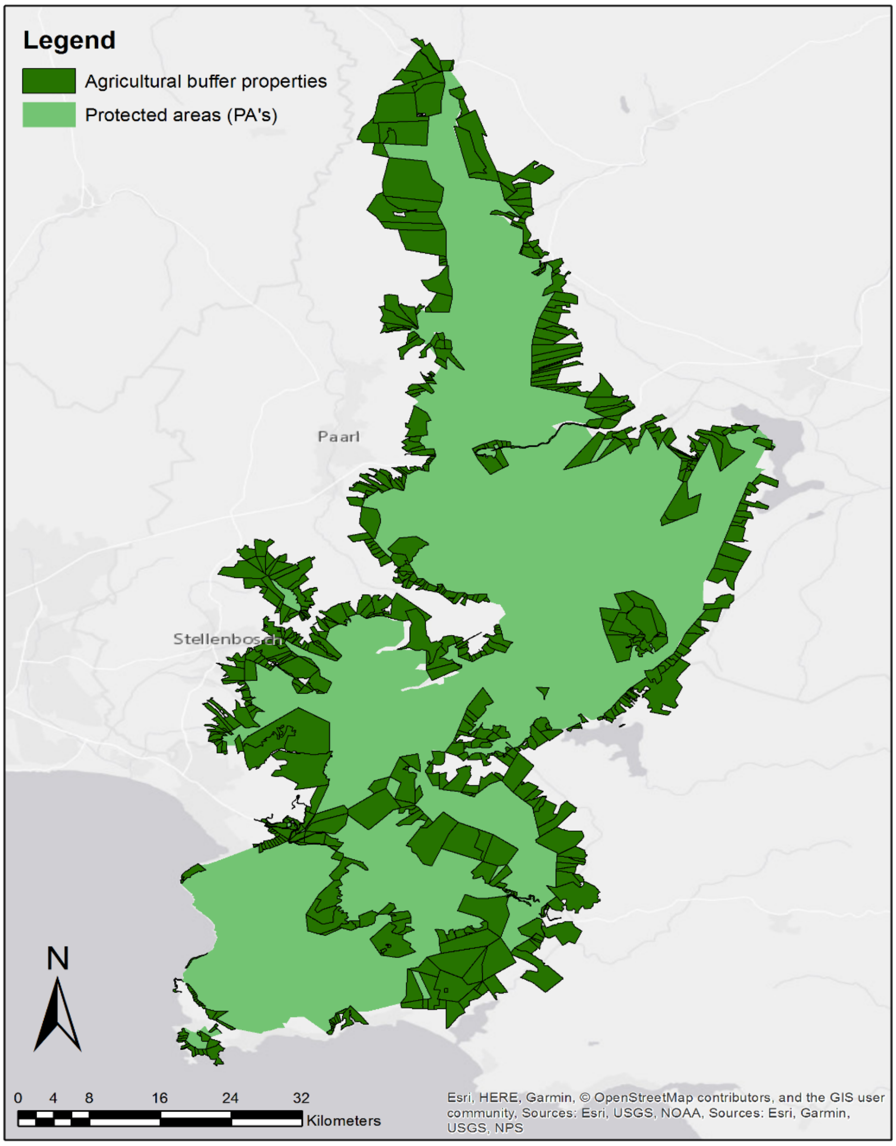


Figure 1.1. The greater study region. Light green areas denote protected areas (PA's), and dark green patches indicate private agricultural properties bordering PA's. North-westernmost point: 33°25'51.3"S 18°49'28.3"E; South-easternmost point: 34°24'59.4"S 19°25'44.3"E.

In some areas, privately owned natural lands adjacent to commercially modified land, is governed and protected as an ecological unit through regional conservancies to achieve specific conservation goals. Most notably is the Theewaters, Greater Simonsberg, Klapmutskop, Groenlandberg, Renosterveld, Franschhoek and Breedekloof Conservancies (See http://www.conservationatwork.co.za). This study focused predominantly on the private properties directly bordering PA's. These properties consist of a wide array of agricultural enterprises, as well as ecotourism and accommodation, and consequently provide working and housing opportunities for a large proportion of the regions' human population. Approximately 52% of the greater study region consisted of formally PA's, while the remainder constituted transformed agricultural lands, urban development and infrastructure, or commercially modified land.

1.6.2 History

Natural resources in areas surrounding Cape Town have been utilised and to some extent even exploited, for thousands of years by first San-gatherers and thereafter Khoikhoi pastoralists who trekked through the Western Cape Province in search of seasonal grazing pastures and water points for their livestock (Deacon 1992; Giliomee & Mbenga 2007). The environmental impact at this stage might have been minimal, despite their use of fire and hunting to alter the landscape (Deacon 1992), but the Khoikhoi's trade in meat, along with the prospect of fresh water, attracted seafarers during the fifteenth century, followed by settlers in 1652 (Worden et al. 1998). During this time human impact was accelerated, ecosystems were suppressed through infrastructure development (Anderson & O'Farrell 2012), and wildlife was reduced to a commodity; plant species were sent to Europe, agricultural land was proclaimed, wood was harvested for self-use and export (Worden et al. 1998), and pest species such as leopard (labelled as *vermin*) were hunted (Martins & Martins 2006). By the 1800's extensive livestock farming became an environmentally exhausting practice

(Giliomee 1989), resulting in the transformation of most lowland vegetation (Allsop et al. 2014). The CFR, an area almost synonymous with the fynbos region, housed 13 million vines in 1819, almost 70 million vines in 1875, and by 1880 almost 30% of the region's population was involved either directly or indirectly with wine farming (Giliomee 1989). The nineteenth century further experienced an upsurge in flora and fauna harvesting for export to Europe (Beinart 2008).

1.6.3 Climate

The Western Cape has a fairly wide climatic range underpinned by synoptic drivers, with rainfall seasons varying to some extent between winter, summer and year-round rainfall patterns. In the Boland Region however, the majority of precipitation occurs during winter months, and the mountains are characterised by persistent cloud caps caused by south-easterly winds during the summer months (Bradshaw & Cowling 2014). Due to the large amounts of moisture produced by these clouds (Marloth 1903), summer drought may be ameliorated, but all natural vegetation is nevertheless likely to experience some degree of water shortage at some point in the year (Kruger 1979). The Boland Region's climate is driven by circumpolar, westerly frontal systems (Tyson 1986) that deliver rain to the region during winter months when these cyclonic air mass systems make landfall, shifting northwards of their summer track, when the Inter Tropical Convergence Zone (ITCZ) moves 10 – 15 degrees northwards in winter. The bioregions in the study area all have Mediterranean rainfall regimes that receive more than 60% of their rainfall in winter (Keeley et al. 2011). As a consequence, the solar radiation for winter is lower than anywhere else in South Africa during any time of the year (Rebelo et al. 2006). Mean annual precipitation (MAP) changes with latitude (Campbell 1983) and altitude (Van Wilgen et al. 2010). Generally, MAP averages between 2 000 – 3 000 mm, however it can reach levels as low as 200 mm annually in the northwest regions (Coetzee 2016). Rainfall reliability, best calculated as the relationship between rainfall amount and variability i.e. the coefficient of variation (CV), is greater on the exterior (coastal) areas of the study area compared to inland areas. An important additional source of moisture and cooling comes from fog, which is also more abundant near the coast due to the large difference between cool ocean temperature and warm air temperature (Tyson 1986), and snow falls sporadically on peaks (> 1 000m altitude) of the Cape Fold Mountains in the southwest and southeast regions of the study area (Rebelo et al. 2006). Relative humidity is highest (> 70%) along the coastal areas, but also extends inland during winter months (Rebelo et al. 2006). Frost occurs in the higher-lying regions of the study area, with average heavy-frost days (< 0°C) ranging between 0.3 – 12 annually (Rebelo et al. 2006).

1.6.4 Geology and soils

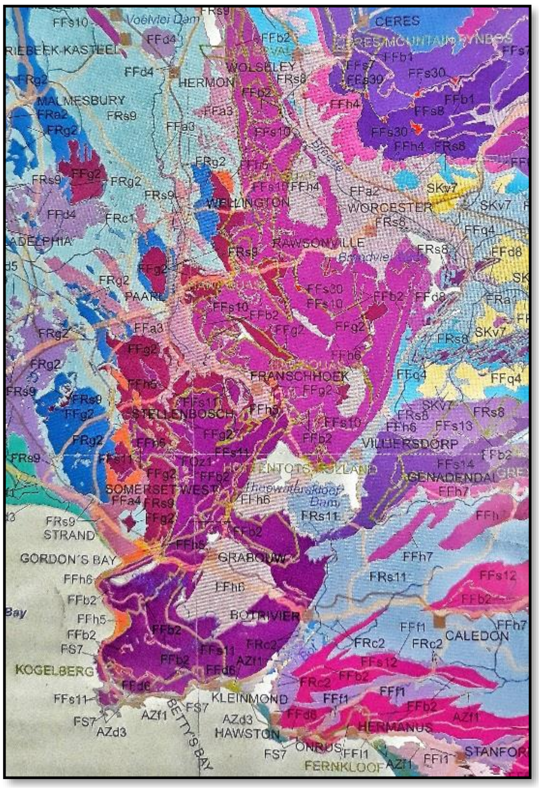
When compared to many other areas across the globe, the Boland Region is an ancient landscape whose geological history spans 2 600 myr (Bradshaw & Cowling 2014). Although the region has been subjected to countless geological events over the past two and a half billion years, it has been experiencing relative tectonic stability since the fragmentation of western Gondwana which initiated approximately 140 myr (Partridge 1987). The Boland region is characteristic of nutrient-poor, sandy soils consisting of either acidic sands of quartzite origin, or calcareous or leached coastal sands (Rebelo et al. 2006). These soils are mainly sourced from the hard quartzite and sandstone rocks of the Table Mountain and Witteberg geological groups (Rebelo et al. 2006; Coetzee 2016). The groups produce shallow, acidic rocky sands, and thus the availability of key nutrients needed for plant growth, such as phosphorus (P) and nitrogen (N), is particularly low across the region (Rebelo et al. 2006; Coetzee 2016). The soils on or close to mountainous areas may appear particularly skeletal, while lowland fynbos soils (with the exception of limestone substrata), are usually deeper (Bergh et al. 2014). The dominant rock types are quartzose sandstone and shale (Mucina et al. 2014). These rock types are known collectively as the Cape Supergroup, and is associated with the Cape Fold Mountains (Bradshaw & Cowling 2014). There are two vegetation units in the fynbos biome, each with characteristic soils components. The first unit, the Fynbos and Western Strandveld areas, consist of nutrient-poor sandy soils. Contrastingly, the second unit, the Renosterveld areas, consist of more fine-grained (silt), clayey soils, predominantly derived from soft shales of the Bokkeveld, Malmesbury, and Karoo sequence rocks, and is thus slightly more nutrient rich and fertile (Coetzee 2016).

1.6.5 Topography and landscape

The break-up of western Gondwana was arguably the greatest event shaping the Boland Regions' landscape by creating the Great Escarpment (Partridge 1987) and exhuming the quartzitic sandstone core of the Cape Fold Belt (Tinker et al. 2008). Contemporary landscapes are topographically largely heterogeneous, ranging from mountainous outcrops to heavily-vegetated valleys and commercialized land. While there are many impressive mountainous areas within the Boland Region (maximum elevation ~1 750 m), overall elevation is relatively subdued, with the majority of the region lying below 300 m (Bradshaw & Cowling 2014).

1.6.6 Vegetation

The southern and south-western Cape boasts an exceptional rich variety of floral species occurring as schlerophyllous shrublands and heathlands that are collectively known as fynbos (Campbell et al. 1981). Within the study area, there are three distinct bioregions, namely the fynbos bioregion, the fynbos-renosterveld bioregion, and the renosterveld bioregion, each with characteristic vegetation types ranging from sandstone and shale fynbos to silcrete and shale renosterveld (Fig. 1.2) (Mucina et al. 2014). This extraordinary diversity in ecosystems has been of considerable scientific and aesthetic interest for decades (Campbell et al. 1981).



Vegetation unit	Description
AZd	Seashore vegetation var.
AZf	Freshwater wetland var.
FFb	Shale band vegetation var.
FFd	Sand fynbos var.
FFf	Ferricrete fynbos var.
FFg	Granite fynbos var.
FFh	Shale fynbos var.
FFs	Sandstone fynbos var.
FOz	Zonal and intrazonal forest var.
FRc	Silcrete renosterveld var.
FRg	Granite renosterveld var.
FRs	Shale renosterveld var.
FS	Western strandveld var.

Figure 1.2. The underlying vegetation units occurring in the study area. Adopted from: Mucina & Rutherford 2006.

Fynbos is the vegetation type largely responsible for the uniqueness of the Boland's floral characteristics. In terms of vegetation, it is characterised by an unparalleled floral diversity accompanied by extremely high levels of endemism (Cowling & Holmes 1992; Bergh et al. 2014), large numbers of plant species coexisting in relatively small areas (up to 121 species per 100 m^2 – exceeded only in tropical rainforests) (Taylor 1978), fire-driven vegetation growth (Van Wilgen et al. 2010), and particularly infertile soils (Campbell et al. 1981). Naturally, the vegetation physiognomy is almost entirely treeless (Specht 1979), modified only after the introduction and spread of several woody invasive tree species, such as *Acacia, Eucalyptus, Hakea and Pinus* spp. (Le Maitre et al. 2000). The remaining vegetation structure is dominated by proteioid, ericoid and restoid components, with grasses less abundant (Cowling & Holmes 1992).

Due its rich species diversity, it is difficult to nominate any one species as particularly important to the area. Some characteristic and valuable flora (Bergh et al. 2014) however include: The fine-leaved ericoid component: Ericaceae (*Erica*), Rutaceae (*Diosmeae*, specifically *Agathosma*), Bruniaceae (*Brunia, Berzelia*), Polygalaceae (*Muraltia*), Thymelaeaceae (*Struthiola, Gnidia*), and Rhamnaceae (*Phylic*a). As well as Fabaceae (*Aspalathus*), Rosaceae (*Cliffortia*), and Asteraceae (*Stoebe, Metalasia*). The Renosterbos (*Dicerothamnus rhinocerotis*) in particular dominates large expanses. The proteoid component, forming the dominant overstorey in the fynbos, consists of flowers that are characterised by their broad and leathery leaves, and include the genera *Protea, Leucadendron, Leucospermum*, and *Mimetes*, as well as some less prominent, fine-leaved members such as *Paranomus, Serruria*, and *Spatalla*. Grasses are less conspicuous, but the restoid component (Cape reeds) of fynbos is well-known, and is made up predominantly of the genera *Restio* and *Thamnochortus*, as well as the families Restionaceae and schoenoid Cyperaceae. A final and equally diverse component of fynbos vegetation comes from the geophytes, particularly the families Iridaceae, Orchidaceae, Hyacinthaceae, and Amaryllidaceae.

1.6.7 Fauna

The fynbos biome supports low animal biomass, but high levels of richness and endemism is seen in all major taxonomic groups (Coetzee 2016). This is largely due to the vegetation in the fynbos being largely unpalatable and nutrient-deficient, and thus unable to support large herbivores adequately (Coetzee 2016). The large herbivores (>20 kg) that historically occurred in the region were therefore mainly restricted to the Renosterveld vegetation units due to their partly greater soil fertility and more palatable vegetation (Rebelo 1992). Historic large herbivore and concomitant large predators that occurred in the region include the Cape Mountain zebra (*Equus zebra zebra*), quagga (*Equus quagga quagga*), blue antelope (*Hippotragus leucophaeus*), red hartebeest (*Alcelaphus buselaphus caama*), eland (*Taurotragus oryx*), bontebok

(*Damaliscus pygargus*), African buffalo (*Syncerus caffer*), African elephant (*L. africana*), black rhinoceros (*Diceros bicornis*), Cape lion (*Panthera leo melanochaita*), spotted hyena (*Crocuta crocuta*), wild dog (*Lycaon pictus*), and leopard (*P. pardus*) (Skead 1980; Boshoff & Kerley 2001; Rebelo et al. 2006). Of these, only the leopard, zebra and bontebok still persist without human intervention. The Western Cape boasts high levels of endemism of mammal, amphibian, reptile and avifauna taxa (11 – 54% endemism), and supports approximately 50% of all terrestrial vertebrate species found in South Africa (Turner 2012). This includes an estimated 54 frog species (> 50% endemic, 14 threatened spp.), 153 reptile species (22 endemic spp., 26 threatened spp.), 507 – 599 bird species (45% resident spp.), and 172 mammal species (9 endemic spp., 10 near-endemic spp., 37 threatened spp.) (Birss and Palmer 2012; Shaw and Waller 2012; Turner and de Villiers 2012; Turner et al. 2012). Furthermore, an exceptional diversity of pollination systems with high levels of specialisation is present in the area (Johnson 1996).

1.6.8 Agrarian Economy

The agricultural sector of the Western Cape is a significant component of the net economy of the province, with a total value-addition of R14.7 billion in 2011 (DAFF 2014). It further contributes greatly to the collective economic growth of South Africa by creating job employment and ensuring food security. The Western Cape Province is the 4th largest province in the country in terms of land area (12 938 600 ha), 89.3% of which is farmed land, and only 5.6% is nature conservation (Stats SA 2012). Farming systems in the Western Cape typically occur in close proximity to PA's (Fig 1.3), and are distinct from other provinces in South Africa because of their unique physical resource differences, such as winter rainfall. The largest proportion of agricultural activity, both in farm units (41%) and hectares (46%) consists of horticultural activity, while the remaining major agricultural activities are relatively equally divided between field crops (9%; 14%), animal production (10%; 10%), and mixed farming

(14%; 15%). More than 135 000 people were employed by the agricultural sector in the Western Cape in 2013, constituting 23% of the total national agricultural workforce; more than any other province in South Africa (Stats SA 2012).

Figure 1.3. Images depicting the human-wildlife interface in the Boland Region, with agricultural development typically pushing the boundaries of protected areas (PA's). Photo credits: W.A. Nieman.

1.7 Research aims and objectives

The overall aim of this study was to gain a novel and improved understanding of the extent, dynamics and drivers of three major hunting practices in the Boland Region of South Africa, and to provide much anticipated recommendations for future research, management and conservation.

The main objectives of the study were to:

1. Identify the socio-economic and biophysical determinants of wire-snare poaching incidence and behaviour among farm labourers employed permanently on agricultural properties buffering PA's.

a) *What are the socio-economic characteristics of labourers more likely to be involved in wire-snare poaching activities?*
b) *What are the motivations for wire-snare poaching among labourers?*
c) *What are the socio-ecological attributes of properties more susceptible to wire-snare poaching activities?*
d) *How is wire-snare poaching activity distributed in relation to abiotic geographic features, such as permanent water bodies and residential areas?*
e) *Are there seasonal peaks in wire-snare poaching activity?*
f) *Which species are most affected by wire-snare poaching? Specifically, which species are the most desired by poachers, and which species are the most frequently caught in wire-snares?*
g) *Which specific areas in the study region experience the highest level of wire-snare poaching?*

2. Identify and map the current hotspots and drivers of human-wildlife conflict between farmers and DCA's on agricultural properties buffering PA's, and provide predictions for zones of conflict risk beyond the scope of this study.
    a) *How many farmers rely on the use of lethal and non-lethal control methods, respectively, and what non-lethal control methods are the most popular?*
    b) *What are the characteristics of farmers that are more likely to rely on the lethal control of DCA's?*
    c) *What is the combined level of tolerance shown by farmers towards damage-causing species, and how do they differ among taxa?*
    d) *For each of the 15 most abundant damage-causing species documented in the study, which specific areas in the region experiences the highest level of conflict?*
    e) *For each of the 10 most abundant damage-causing species documented, what are the environmental and agricultural characteristics of properties more susceptible to conflict?*

   f) *For each of the eight most abundant damage-causing species documented, which geographical zones are most at risk of future conflict?*

3. Document and describe the use of animals and animal-derived constituents in African traditional medicine in informal communities.
   a) *What are the species most commonly sold as traditional medicine, which species have the most uses attributed to them, and how many are of conservation concern?*
   b) *What are the animal parts or products most frequently sold as traditional medicine, and which are the most expensive?*
   c) *Was the sampling performance adequate to conduct reliable analyses?*
   d) *What was the observed species richness, diversity and evenness of species documented in the study, and how many more species can be expected to be found if sampling continued indefinitely?*
   e) *What was the degree of heterogeneity or homogeneity in the selection of animals to treat certain conditions by traditional healers?*
   f) *What species can be ranked as most important for the treatment of specific ailments?*
   g) *Which species are the most valuable to traditional healers in the cultural groups sampled in the study?*

4. Provide a synergistic assessment of animal species' vulnerability to the three major hunting practices assessed in previous chapters of this study.
   a) *Is the proposed novel vulnerability-assessment test statistically valid?*
   b) *Which species are most at risk of potential endangerment from the combined effects of the three major hunting pressures?*

1.8 Thesis outline

This thesis is composed of six chapters. Chapter one provides a comprehensive literature review of the study topics, methodology and study location. Chapters two, three, four and five are written as individual manuscripts to assist publication in peer-reviewed journals. Some level of repetition may thus be expected between chapter one and each of the data-chapters. The final chapter is prepared for managers, policy-makers and conservationists in the Boland Region, and potentially elsewhere, to provide recommendations for the management of the three hunting pressures assessed in this study.

Chapter 1: Background literature review: This chapter provides an in-depth description of the ecological theory substantiating our research, our sampling approach, bushmeat harvesting, human-wildlife conflict, the use of animals in traditional medicine, and the greater study area.

Chapter 2: Socio-economic and biophysical determinants of wire-snare poaching incidence and behaviour in the Boland Region of South Africa: This chapter examines human behaviour and biophysical elements to describe their influences on the distribution and incidence of wire-snare poaching.

Chapter 3: Farmer attitudes and regional risk modelling of human-wildlife conflict on South African farmlands: This chapter identifies species-specific hotspots of HWC on agricultural properties, as well as the drivers and characteristics of those conflicts, to map the zones of species-specific conflict risk in the Boland Region.

Chapter 4: The use of animals and animal-derived constituents in African traditional medicine and other cultural applications: townships in the Western Cape: This chapter documents the species used in traditional medicinal practices in the Boland Region, and quantifies differences in social coherence, species value and importance, parts or products used, and ailments treated.

Chapter 5: A synergistic assessment of animal species' vulnerability to three major hunting practices in the Western Cape Province: This chapter combines the incidence of all three hunting practices documented in chapters two, three and four, to provide combined analysis of the species currently most vulnerable to hunting pressures.

Chapter 6: Project summary with implications for management, conservation and policy development: This chapter summarises the key findings presented in this thesis, provides management recommendations based on the study findings, and gives a final conclusion for the study.

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Chapter 2
Socio-Economic and Biophysical Determinants of Wire-Snare Poaching Incidence and Behaviour in the Boland Region of South Africa

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2.1 Abstract

Bushmeat harvesting, fuelled by wire-snare poaching, has long been recognized as a severe threat to biodiversity throughout East and Central Africa, and has been directly linked to severe reductions or extirpations of target species, high rates of non-target off-take of threatened species, and the loss of entire wildlife communities. Studies dedicated to assessing the extent and underlying dynamics of wire-snare poaching in South Africa have however been lacking, and no formal research has been conducted in the Boland Region, despite growing evidence of wire-snare incidence. Through structured interviews with landowners or farm managers and labourers on private agricultural properties bordering protected areas (PA's), this study quantified the influence of several socio-economic and biophysical determinants on the distribution of wire-snare poaching across the Boland. Wire-snare poaching incidence and behaviour was strongly influenced by economic factors relating to poverty, a lack of governing regulations and punitive measures, interpersonal development, and abiotic factors such as proximity to major residential areas, roadways and PA's. The animal species reportedly most affected by wire-snare poaching were identified, with small antelope and porcupine predominating. Seasonal peaks in wire-snare activity were highlighted and several activity hotspots across the region were identified. This study provided the first demonstration of the multifaceted and complex nature of wire-snare poaching in the Boland Region, opening a dialogue between rural communities and conservation agencies to broaden our understanding of the heterogeneity in local-scale socio-ecological dynamics, to apply policies for effective management and eradication, and to provide grounds for future research in the area and elsewhere.

Keywords: Agriculture, buffer zones, bushmeat, community conservation, ecological questionnaires, illegal hunting, rural communities

2.2 Introduction

During the past few decades, natural habitats have been continually converted for agricultural land-use to sustain the needs of an ever-increasing human population in sub-Saharan Africa (Alexandratos & Bruinsma 2012; Rosenblatt et al. 2016). Subsequently, many rural communities have been established on the borders of protected areas (PA's) where they are often forced to rely on natural resources to sustain their own livelihood needs (Wittemyer et al. 2008), leading to more pronounced edge effects and increased strains on natural wildlife populations (Pulliam 1988; Woodroffe & Ginsberg 1998; Watson et al. 2013).

One of the most devastating effects of human encroachment on natural areas has been the rapidly escalating bushmeat trade. People living in close proximity to PA's often rely on the hunting of wildlife, as well as the commercialized trade of bushmeat, for improved food security (Rentsch & Damon 2013) and alternative income (Lindsey et al. 2011a). Bushmeat harvesting for both commercial as well as self-use purposes is however largely unsustainable (Fa et al. 2002; Loibooki et al. 2002; Mainka & Trivedi 2002; Robinson & Bennett 2002; Milner-Gulland & Bennett 2003), and has been described as one of the greatest contemporary threats to the continued existence of natural wildlife (Fa et al. 2002), especially in Africa (Knapp 2012; Taylor et al. 2015). A decline in wildlife numbers is often observed in areas where bushmeat harvesting is excessive (Lindsey et al. 2013; Kragt et al. 2016; Hegerl et al. 2017), threatening native wildlife species with local extinction (Wilkie & Carpenter 1999).

In 2013, Lindsey et al. found illegal wire-snares (Fig. 2.1) to be the most prevalent method used for the hunting of bushmeat in South Africa. The popularity of wire-snares is most likely founded in its inexpensive and obscure nature (Lindsey et al. 2011a), as well as its ability to successfully capture a wide range of taxa (Noss 1998). Depending on the material, wire-snares have been reported to effectively capture anything ranging from rodents to African elephants (*Loxodonta africana*) (Hofer et al. 2000). The wire used for snares is also easily obtainable from primarily reserve, farm or garden fences (Lindsey et al. 2011b), further impeding effective eradication of its use (Lindsey et al. 2013).

Figure 2.1. Typical wire-snares, consisting of a cable or fence wire that is rolled into a noose and anchored to vegetation or fences to catch small game. Photo credits: W.A. Nieman.

In recent years the use of wire-snares has however become progressively unacceptable across a wide range of social classes (Naughton-Treves and Treves 2005), predominantly owing to the ethical apprehensions linked to the injuries animals sustain through escaping snares, as well as its delayed efficacy. Further concern stems from their indiscriminate nature, as they are frequently reported to kill non-target and charismatic species (Lindsey et al. 2011b; Vanthomme et al. 2017), inadvertently disrupting functional plant-animal relations (Wright 2003) and ecosystem stability (Galetti & Dirzo 2013). Predator species such as leopard (*Panthera pardus*) and caracal (*Caracal caracal*), which are the primary apex predators in the Boland Region (Martins & Harris 2013), are particularly vulnerable to being captured as by-catch because of their tendency to frequently visit areas of high prey densities, as well as areas with trapped animals or carcasses (Lindsey et al. 2013). Due to the low cost associated with producing snares, hunters also tend to neglect checking on them and often desert them entirely, resulting in unnecessary animal mortalities (Noss 1998). Impacts associated with wire-snare poaching have been shown to include severe reductions or extirpations of target species, high rates of non-target off-take of threatened species, and the loss of entire communities (Hofer et al. 1996; Lindsey et al. 2011a, 2011b, 2013; Becker et al. 2013). Consequently, the use of wire-snares is specified as a criminal offence under the Western Cape Nature Conservation Ordinance (No. 19 of 1974).

Research on bushmeat harvesting and the associated use of wire-snares has mainly been concentrated in areas of East and Central Africa (Noss 1998; Bowen‐Jones et al. 2003; Brashares et al. 2011; Nuno et al. 2013; Rentsch & Damon 2013; Nyaki et al. 2014; Hegerl et al. 2017), and reliable data quantifying the extent and underlying dynamics of bushmeat harvesting in Southern Africa is lacking (Robinson & Bennett 2002; Lindsey et al. 2011a). To allow for the development of efficient management strategies, urgent quantification of bushmeat off-take is thus required locally.

This study focused on describing the various socio-economic and biophysical factors influencing bushmeat harvesting by means of wire-snare poaching on private properties bordering PA's in the Boland Region. Specifically, human behaviour was examined to ascertain what socio-economic attributes were more likely to influence an individual's involvement in wire-snare activity, and what motivates them to rely on this form of hunting. Biophysical and other socio-ecological factors were also considered to identify attributes of properties subjected to poaching activity, and the spatio-temporal distribution of wire-snare incidence. Finally, the wildlife species and geographical areas most at risk of wire-snare poaching were identified. Identifying the suite of factors underlying wire-snare poaching patterns will likely expose areas where exigent conservation action is required, and allow for better resource allocation of anti-snaring initiatives.

2.3 Materials and methods

2.3.1 Study area

The Boland Region forms part of the Western Cape Province of South Africa, and is characteristic of a wide array of land cover types ranging from mountainous outcrops to heavily-vegetated valleys and commercially modified land. The region's mountainous core consists of various open and closed PA's which are almost exclusively enveloped by transformed lands and are thus seen as biodiversity islands, with few corridors remaining between them. For some functionally important and charismatic species these critical refuges represent a last stronghold for their continued existence.

The study focused specifically on private properties bordering PA's, covering an area of approximately 3 500 km^2 (Fig. 2.2). These properties consist of a wide array of agricultural enterprises, as well as ecotourism and accommodation, and consequently provide working and housing opportunities for a large proportion of the region's people. The Mediterranean climate is typified by hot and dry summers, with cool and wet winters (Keeley et al. 2011). To accommodate potential seasonal fluctuations in poaching trends, data collection included both seasonal extremes, occurring from July 2017 to May 2018.

2.3.2 Data collection

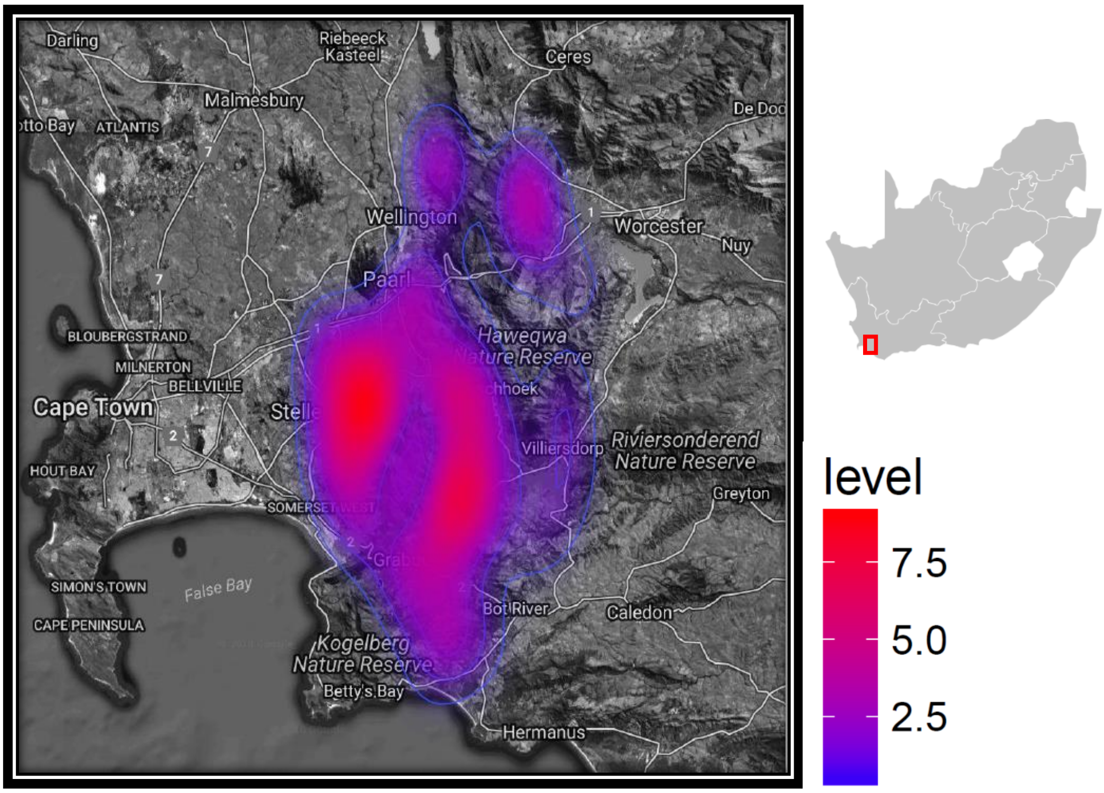

Figure 2.2. The study area shown according to sampling effort. Sampling intensity (i.e. number of respondents) is indicated by a colour change between shades of blue (low intensity) and red (high intensity). Coordinates: Southeastern extreme: 34°20′51.7″S 19°08′26.9″E; North-western extreme: 33°29′41.0″S 19°00′13.7″E.

Closed-format questionnaires (Appendix 6.1 & 6.2) were used to conduct one-on-one, face-to-face, structured interviews with two separate groups of respondents: (1) permanently employed labourers on agricultural properties bordering PA's, and (2) the managers or owners (hereafter collectively referred to as farmers) of those properties (Fig. 2.14). This form of data collection was relied on due to the relatively large size of the study area, and the interdisciplinary design that included both ecological as well as sociological aspects (White et al. 2005; Becker et al. 2013). Face-to-face interviews are also viewed as the preferable sampling method overall in questionnaire methodology (Arrow et al. 1993), and closed-format questions tend to result in less uncertainty than open-ended questions for both the researcher and the respondent (White et al. 2005).

To ensure that all properties had a relatively equal probability of being sampled, surveyed sites were selected *a priori* using a stratified random sample from the greater study area (Fig. 1.1), provided that the properties were all adjacent to PA's and had permanent labourers in employment. Small properties (< 50 ha) directly adjacent to one another were manually excluded from the sample to avoid spatial autocorrelation. Based on the criteria, a total of 232 properties were eligible for selection, of which 103 were included in the study. Only five farmers refused to participate (nonresponse < 4.7%), and nonresponse bias was thus minimal (Lindner 2002). Interviews with farmers ranged between 30 minutes to two hours. Participating labourers were selected based on availability at the time of the visit, and the prerequisite that the respondent had to spend considerable periods of time working outdoors, covering large parts of the property in the process. Sample size was kept relatively consistent across properties. A total of 307 workers were interviewed, and 4 refused (nonresponse < 3.9%). In all cases where respondents refused to participate it occurred before any questions were asked. Labourer interviews lasted between 10 – 15 minutes each.

The interviewer was kept consistent throughout the duration of the study, and preceding each interview a written consent was signed by the respondent to demonstrate their willingness to participate and to provide permission for the data to be used for further analyses (Appendix 6.4). Ethical clearance was obtained from the Human Research (Humanities) Council at Stellenbosch University (Reference: SU-HSD-004696). Given the clandestine nature of the activities in question, all accounts were kept anonymous to improve the likelihood of participation (Whelan 2007). As it however remains impossible to assess respondents' *bona fides*, labourers involved in wire-snare poaching that had less trust in visiting researchers may nonetheless be underrepresented in this study. To ensure continued confidentiality and to minimize survey sensitivity, no personal or geographic information was collected that could be used to identify specific properties or individuals.

2.3.3 Statistical analysis

To evaluate the effects of various predictor variables on wire-snare activity, three separate sets of generalised linear regression models (GLM's) assuming a Poisson family distribution with a "log"-link function were fitted. The first set evaluated the effect of several socio-economic predictors relating to personal characteristics of labourers on their participation in wire-snare activities, to identify the nature of individuals more likely to be involved in wire-snare activities. A total of 10 plausible predictor variables were postulated to explain the characteristics of the individuals associated with the use of wire-snares. Secondly, an attempt was made to identify the motivations driving wire-snare behaviour by asking respondents which of a possible six postulated motivations were applicable to them. For both these analyses, the mean number of wire-snares set by respondents on a monthly basis was used as the response variable. The third set used information provided during farmer interviews to identify the socio-ecological attributes of properties more susceptible to wire-snare poaching using the mean number of snares found by labourers on the property on a monthly basis. Candidate models from all three sets contained all additive combinations of predictors as well as two-way interactions. Standard information-theoretic multi-model inference techniques were then used to interpret the results (Burnham & Anderson 2003; Dochtermann & Jenkins 2011), and models within each set were then ranked in order of parsimony using Akaike's Information Criterion values adjusted for sample size (AICc). For the first two model sets, models within two ΔAICc units ($\Delta AICc \leq 2$) of the model with the lowest AICc were regarded as having equal support (Akaike 1974; Burnham et al. 2011). For the third set, models within four ΔAICc units ($\Delta AICc \leq 4$) were considered to have sufficient relative support to be included in the final set of explanatory models (Burnham & Anderson 2003; Burnham et al. 2011). The probability of each model being the most parsimonious in the candidate model set was determined by the Akaike model weight, and model statistics were obtained with an analysis of deviance for general linear model fits (ANOVA.GLM).

Decision trees founded on the Breiman's (2001) Random Forest (RF) algorithm were created based on an ensemble of best-fit variables produced in above-mentioned GLM model-fits, to ascertain which variable(s) drive or enhance the probability of correctly predicting the (1) individuals involved in wire-snare behaviour and (2) the agricultural properties that have wire-snare activities. Random forests add an additional layer to the traditional bagging of classification trees (Breiman 1996), and is a type of data mining technology which combines information from a collection of virtually grown trees, resulting in a '*forest*' of decision trees. The forests are then grown out to their maximum possible size based on CART methodology (Biau 2012) and the most significant predictors (R statistical software). The RF algorithm has been shown to be consistent and well able to adapt to sparsity when the rate of convergence depends only on the number of significant variables and not on the number of noisy or insignificant variables (Biau 2012). Each split was chosen using the best fit among a subset of predictors randomly chosen at the node. Using this method, significant progress can be made in classification and regression accuracy (Biau 2012).

To evaluate the effect of several abiotic variables on the distribution of wire-snares across a geographic landscape, a multiple linear regression analysis was used. The proximity of each postulated variable to the centroid of the property section on which snares were reportedly found was measured '*as the crow flies*' on ESRI-mapped GIS layers, and correlated with the mean number of snares found. The regression coefficient (β) was used to determine the likelihood of the wire-snares' location for each predictor variable. The most parsimonious models were based on the lowest

AICc values, and models within four AICc units of the lowest AICc (ΔAICc ≤ 4) were included in the final set of explanatory models (Burnham et al. 2011). Models that were simply a more complicated version of a nested model with lower AICc were excluded to reduce the selection of unnecessarily complex models (Burnham & Anderson 2003).

To evaluate temporal trends in wire-snare activity, a one-way analysis of variance (ANOVA) was performed, followed by Tukey's post-hoc honestly significant difference (HSD) test, to compare the number of snares found monthly by labourers on properties. Differences were considered significant at $P < 0.05$.

Two separate non-parametric Kruskal-Wallis rank sum tests were conducted to identify the species most caught in wire-snares, as well as those most desired by wire-snare poachers. To determine this respondents were asked about their own preferences and successful captures, as well as those of their colleagues, neighbours and/or strangers. Individual answers were then pooled together to obtain a single value describing the species most vulnerable to wire-snare poaching. Several species were pooled together as '*morphospecies*' (i.e. a typological species) in instances where they were not identifiable to species-level (e.g. species that could only be identified as genet spp., hare spp., mongoose spp., etc.). Differences between species were identified with a post-hoc Dunn test, wherein p-values were adjusted according to the Benjamini-Hochberg method. Finally, a two-way chi-squared test (χ^2) was conducted to test for differences between the species desired by poachers and the species caught by poachers.

Poaching activity is presumably unevenly spread across the surveyed area, with some areas experiencing a greater than average number of wire-snare activities (hotspots). Spatial visualization of wire-snaring hotspots was achieved with the construction of a heat map, wherein a static '*hybrid*' map from Google Maps was overlaid with layered matrix data containing the location (property coordinates) of snares that were found by respondents throughout the study area. The Moran's I spatial autocorrelation statistic was applied to compare the value of one location to all other locations, to assign an intensity value to each geographic area (Levine 2004).

All data analyses were performed in RStudio v.3.5.0. Spatial analyses were achieved with the GIS package CapeFarmMapper v.2.1.0.2, ESRI satellite imagery and ArcGIS v.10.3.1.

2.4 Results

2.4.1 Characteristics of the sampled community

Most labourers (77%) were part of the Afrikaans-speaking cultural group. The remainder were Xhosa-speaking (16%), and 7% spoke other African languages. The majority of labourers were also full-time residents on the property on which they were employed (65%), with a mean job tenure of 15 years (±SE 0.6). Most labourers were born in the Western Cape (74%), while the remainder migrated from other South African provinces (21%) and other African nations (5%). Interviewed labourers further consisted of 82% males, were of a mean age of 42 years (±SE 0.7), and had a mean immediate family size of 4 individuals (±SE 0.1).

The combined area of the farms surveyed was 41 706 ha (median = 158 ha, range = 5 – 4 107 ha). This represents 44.4% of the total farmland bordering PA's in the study area. The majority of farms were used primarily for vineyards (45%) and orchards (43%), with the remaining 12% used for livestock and game rearing, as well as aquaculture and apiaries. Median elevation was 279 m (range = 34 – 620 m). For a full description of respondent and survey location characteristics, see Appendix 2.1.

2.4.2 Socio-economic determinants of wire-snare involvement

Almost fourteen percent (13.7%, n = 42) of the labourers admitted to regular involvement in wire-snare poaching activity, setting a mean number of 4 snares per month (±SE 0.4). Five of the 63 candidate models of the global additive model were within 2 Akaike units (AICc) of the top ranked model (Table 2.1). A number of socio-economic variables explained involvement in wire-snare activity, namely job tenure, respondent age, family size, migratory status, historical influences, and the perceived presence or absence of prohibiting regulations ($F_{(15,306)} = 34.65$, $P < 0.0001$).

Table 2.1. Generalized linear models that were within 2 AICc units of the highest ranking model (lowest AICc), with associated degrees of freedom (df), the number of parameters in the model (k), AICc, ΔAICc and Akaike model weights.

Model	df	k	AIC_c	ΔAIC_c	Weight
Response variable: number of snares set in the month prior to the interview					
1. Age + Gnd[1] + Hst[2] + Mgr[3] + Rgl[4] + EmP[5] + FmS[6]	15	16	414.1	0.00	0.204
2. Gnd + Hst + Mgr + Rgl + FmS	13	14	414.4	0.23	0.182
3. Gnd + Hst + Mgr + Rgl + EmP + FmS	14	15	415.5	1.31	0.106
4. Gnd + Hst + Mgr + Rgl + Rsd[7] + FmS	14	15	415.9	1.79	0.083
5. Age + Gnd + Hst + Mgr + Rgl + FmS	14	15	416.0	1.89	0.079

[1]Gnd = Gender
[2]Hst = Historical influences
[3]Mgr = Migratory status
[4]Rgl = Governing regulations
[5]FmP = Period of employment (job tenure)
[6]Fms = Family size
[7]Rsd = Place of residency

Local respondents (β = 0.68; SE = 0.31; z = 2.25; $P < 0.05$), as well as respondents that migrated from the Northern Cape Province (β = 1.51; SE = 0.36; z = 4.17; $P < 0.0001$), were significantly more likely to be involved in snaring activity than respondents that migrated from other regions across Southern Africa (Fig. 2.4d), and set significantly more wire-snares per month. Most respondents who regularly set snares (69%) learned snaring behaviour from past family-related activities (Fig. 2.4a), and set significantly more snares on a monthly basis (β = 1.63; SE = 0.22; z = 7.31; $P < 0.0001$). Of those respondents setting snares, 33.3% did so despite being aware of governing regulations prohibiting the use of snares on the site, but significantly more snares were set during a monthly period by individuals who claimed that there were no anti-snaring regulations on the property (β = 1.18; SE = 0.25; z = 7.31; $P < 0.0001$; Fig. 2.4b).

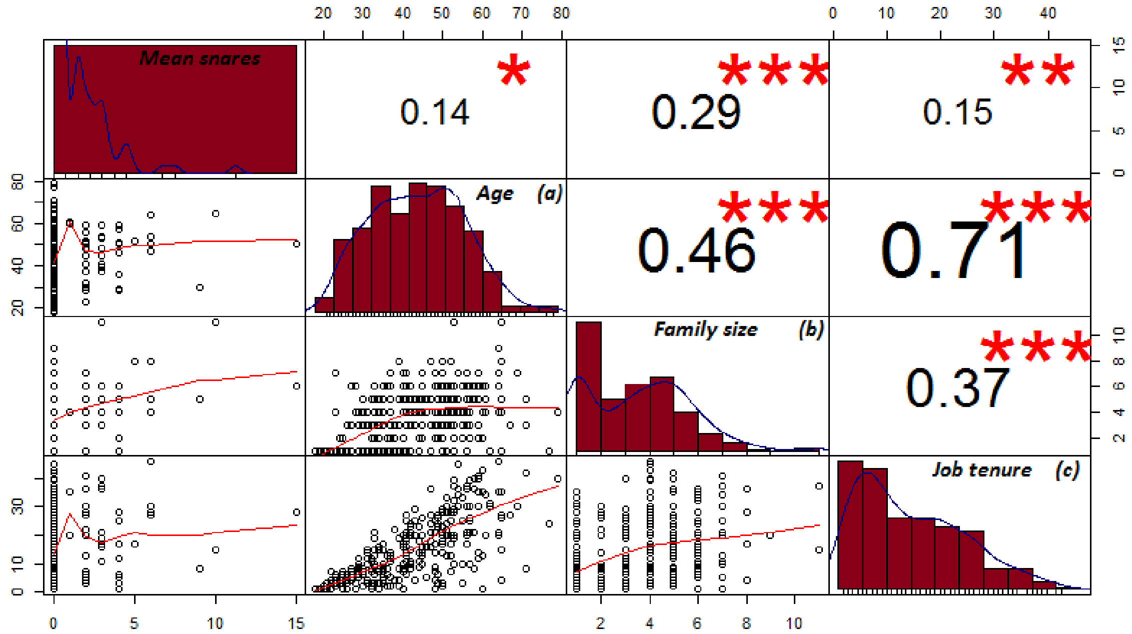

Figure 2.3. The relationship between the mean number of snares set by respondents (n = 307) in the month preceding the interview, and the corresponding (a) age, (b) family size, and (c) job tenure of respondents.

The size of a respondent's family significantly influenced the number of wire-snares set in a monthly period (Fig. 2.3b), with more snares being set as family size increased (β = 0.23; SE = 0.04; z = 5.65; $P < 0.0001$). Mean monthly snares set also increased significantly as a respondent's age (β = 0.02; SE = 0.01; z = -1.84; $P < 0.05$; Fig. 2.3a) and job tenure (β = 0.01; SE = 0.01; z = 2.10; $P < 0.05$; Fig. 2.3c) increased. No female respondents were involved in wire-snare behaviour, and males thus set more snares on a monthly basis (β = 0.02; SE = 0.02; z = 2.20; P < 0.05; Fig. 2.4c). The respondent's home language, race, and place of residency had no significant influence on their snaring behaviour.

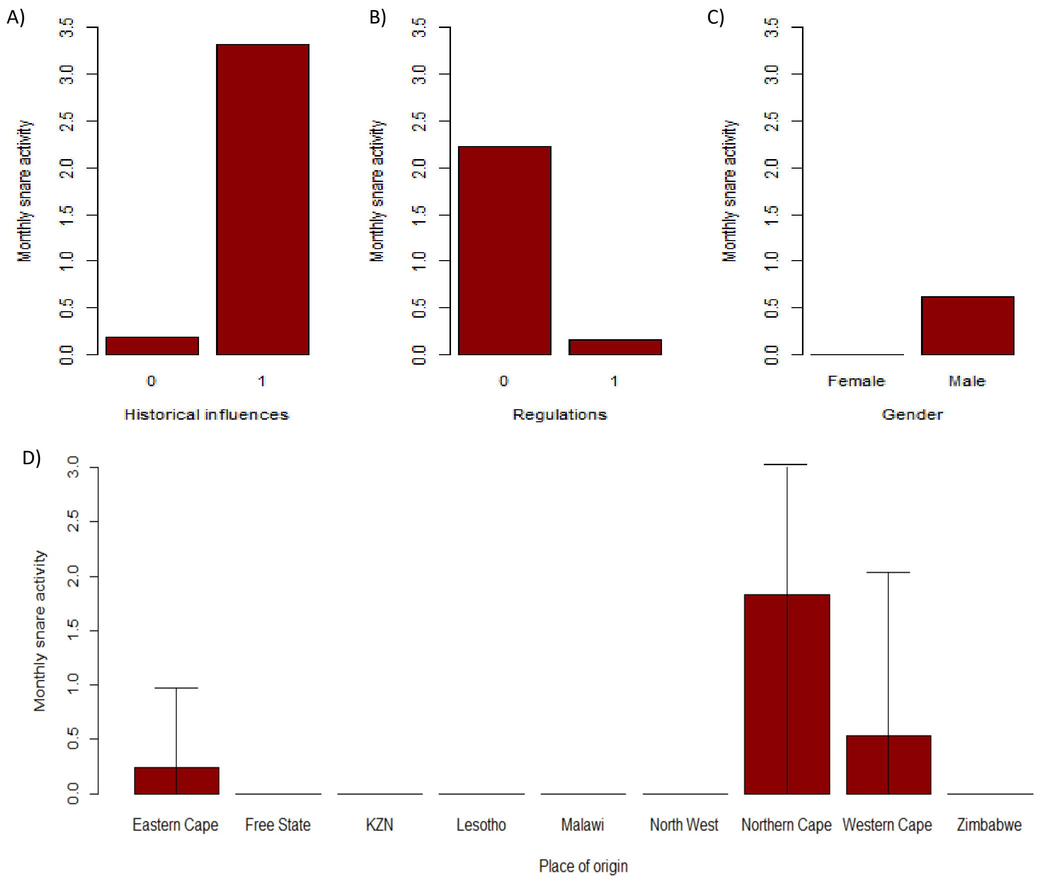

Figure 2.4. Mean monthly snares set by labourers (a) that described their ability to snare as either peer-learned or family-learned, (b) that reported the absence or presence of anti-snaring regulations at their hunting sites, (c) that were female or male, and (d) that originated locally or migrated from various regions across Southern Africa (±SE).

2.4.2.1 Random forest prediction

Random forests produce decision trees for determining the targets of conservation actions more accurately based on the most important defining factors. The algorithm provided three node splits based on the five most integral factors determining an individual's involvement in wire-snare activities (Fig. 2.5) (lm, n = 500). These were, in order: historical influences (MSE = 2.52; SD = 0.32; n = 307), period of employment and governing regulations (MSE = 1.28; SD = 0.30; n = 276), and age and family size (MSE = 4.73; SD = 0.30; n = 31).

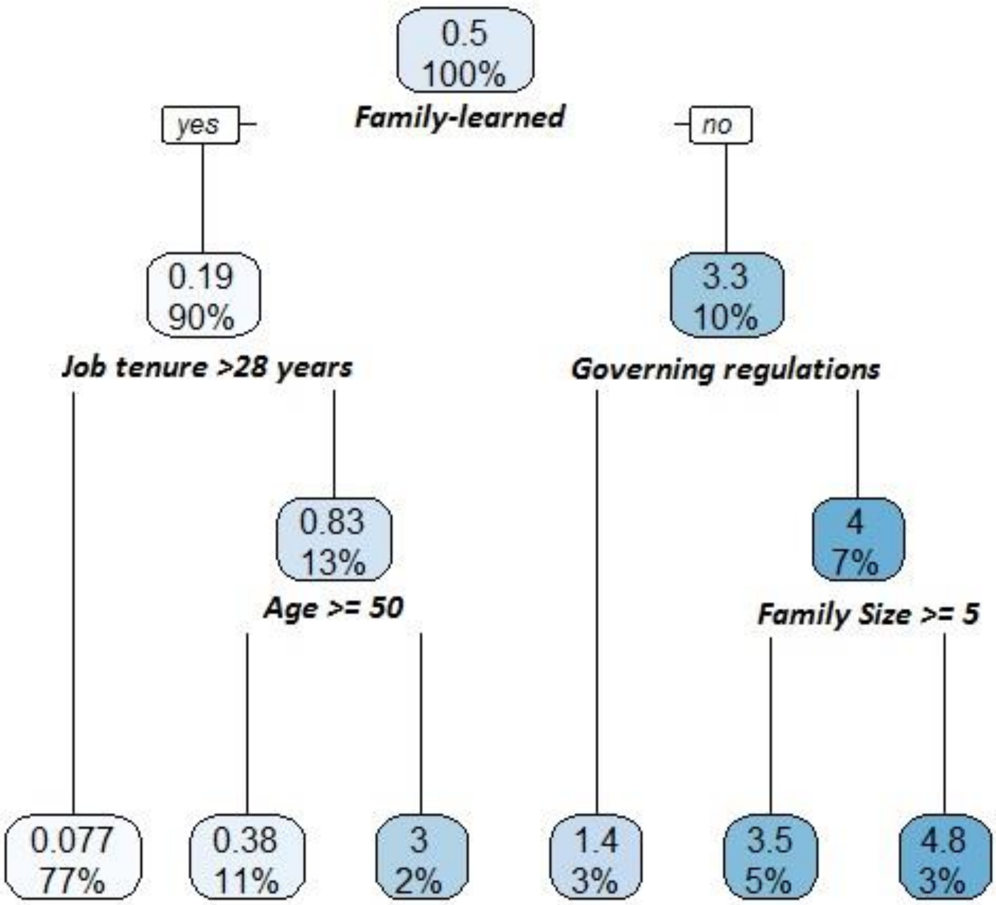

Figure 2.5. Decision tree for determining whether an individual is likely to be involved in wire-snare poaching activities.

2.4.3 Motivations for using wire-snares

Labourers that admitted to involvement in regular snaring activity (n = 42) were personally motivated by 3 of the 4 variables included in the candidate models (Table 2.2; Fig. 2.6) that were within 2 Akaike units of the top ranked model ($F_{(3,306)} = 63.67$; $P < 0.0001$).

Table 2.2. Generalized linear models that were within 2 AICc units of the highest ranking model (lowest AICc), with associated degrees of freedom (df), number of parameters in the model (k), AICc, ΔAICc and Akaike model weights.

Model	df	k	AIC_c	ΔAIC_c	Weight
Response variable: number of snares set in the month prior to the interview					
1. CnV[1] + FdI[2] + PsC[3]	4	5	163.4	0.00	0.167
2. CnV + PsC	3	4	163.6	0.18	0.153
3. CnV + Clt[4] + FdI + PsC	5	6	165.4	2.00	0.061

[1]CnV = Convenience of the method compared to other hunting methods
[2]FdI = Food insecurity
[3]PsC = Pest Control
[4]Clt = Cultural requirements

Most respondents involved in wire-snare activity (81%) cited the relative convenience of this hunting method (i.e. ease of use, production cost, obscurity and the low risk of arrest it poses) as a primary motivator for their use ($\beta = 0.94$; SE = 0.35; z = 2.68; $P <$ 0.01). The majority (95.2%) also disclosed that food insecurity is a major motivator for their decision to use wire-snares ($\beta = 4.83$; SE = 0.67; z = 7.18; $P < 0.0001$), and significantly more snares are set for self-nutritional purposes per month. A smaller proportion of the respondents (12%) use wire-snares as a form of pest control against destructive problem species occurring on agricultural properties, but set significantly high numbers of monthly snares for this purpose ($\beta = 1.26$; SE = 0.21; z = 6.02; $P <$ 0.0001; Fig. 2.7).

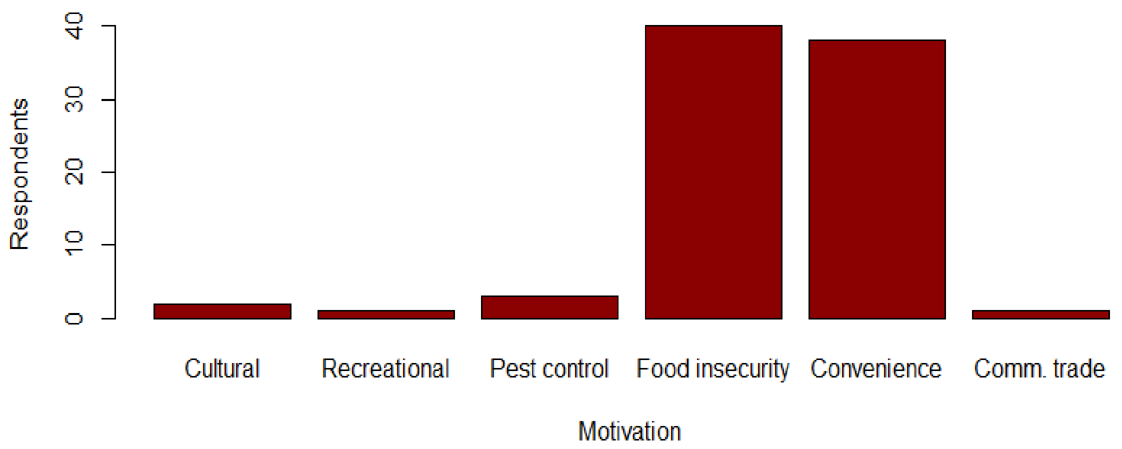

Figure 2.6. Motivations disclosed by respondents for setting snares. Cultural requirements (medicinal, spiritual, religious, or traditional), recreational or hobbyist hunting, and commercialized trade of bushmeat did not explain the motive for wire-snaring. The control of pest species, food insecurity, and the convenience of the method compared to other hunting techniques were significant motivations for wire-snaring.

No distinguishable commercial trade sector was identified, and consequently the commercialized buying and selling of poached bushmeat was not a significant driver of wire-snare activity. Wire-snares set due to cultural requirements (the use of bushmeat for medicinal, traditional, spiritual or religious purposes) did not significantly contribute to wire-snare incidence, nor did hunting as a recreational or pastime activity.

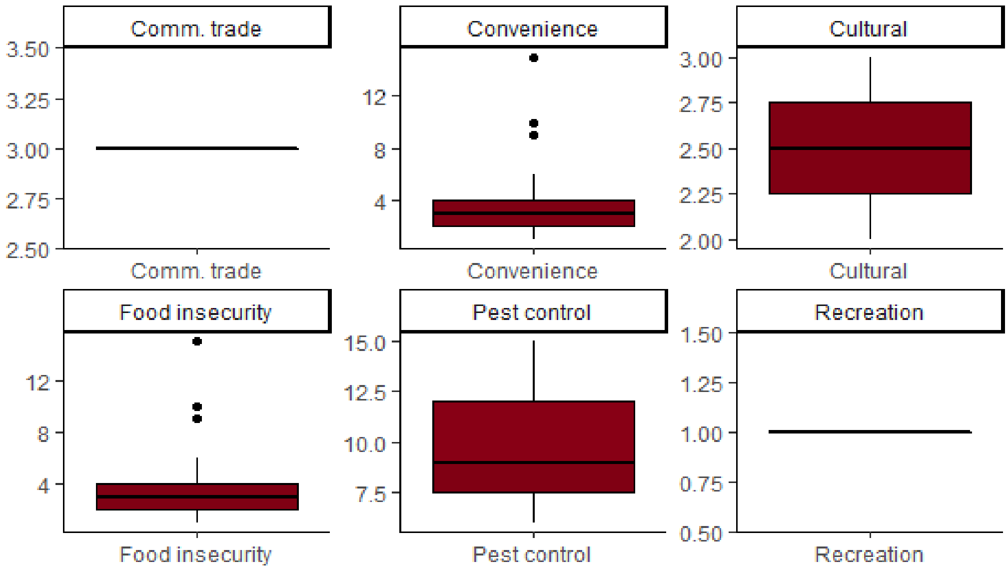

Figure 2.7. The mean number of snares set per month for each of the motivations disclosed by respondents as driving their decision to use wire-snares. A large number of wire-snares were set out for pest control purposes, while respondents predominantly driven by food insecurity, cultural requirements, commercial trade and the relative convenience of the method set comparatively fewer snares per month. Few snares were set for purely recreational purposes.

2.4.4 Socio-ecological attributes of properties susceptible to poaching

Five of the 76 candidate models were within 4 Akaike units of the model with the lowest AICc value (Table 2.3), containing seven variables. Based on significant values, the average number of snares found on properties on a monthly basis was significantly influenced by the number of families living on the property, the employment of seasonal workers, and the absence or presence of enforced punitive measures ($F_{(3,102)} = 19.33$; $P < 0.0001$).

Table 2.3. Generalized linear models that were within 4 AICc units of the highest ranking model (lowest AICc), with associated degrees of freedom (df), number of parameters in the model (k), AICc, ΔAICc and Akaike model weights.

Model	df	k	AIC_c	ΔAIC_c	Weight
Response variable: Average monthly number of snares found on properties					
1. PnM[1] + SsW[2] + FmP[3]	5	6	417.9	0.00	0.345
2. PnM + SsW + FmP + PrS[4]	6	7	420.1	2.22	0.114
3. PnM + SsW + FmP + NtV[5]	6	7	420.1	2.26	0.112
4. OwR[6] + PnM + SsW + FmP	6	7	420.2	2.28	0.111
5. PaO[7] + PnM + SsW + FmP	7	8	421.0	3.11	0.073

[1]PnM = Punitive measures
[2]SsW = Seasonal or contract workers
[3]FmP = Families living on the property
[4]PrS = Property size
[5]NtV = Proportion of the property consisting of natural, untransformed land
[6]OwR = Farmer's place of residency
[7]PaO = Primary agricultural output

The number of snares reportedly present on the property was significantly influenced by the number of families living on the property (Fig. 2.8a), with more snares being found on properties with more families ($\beta = 0.02$; SE = 0.01; t = 3.07; $P < 0.01$). Farms that employed seasonal or contract workers during any time of the year also had significantly more snares present ($\beta = 1.89$; SE = 0.62; t = -3.07; $P < 0.01$) (Fig. 2.8b), as well as farms that had no enforced punitive measures ($\beta = 2.42$; SE = 0.49; t = -4.93; $P <$ 0.0001) (Fig. 2.8c). The primary agricultural output of the property (vineyards, orchards or animal production), the size of the property, the proportion of the property consisting of natural, untransformed land, and the place of residency of the manager/owner had no significant influence of the mean number of snares found on the property on a monthly basis.

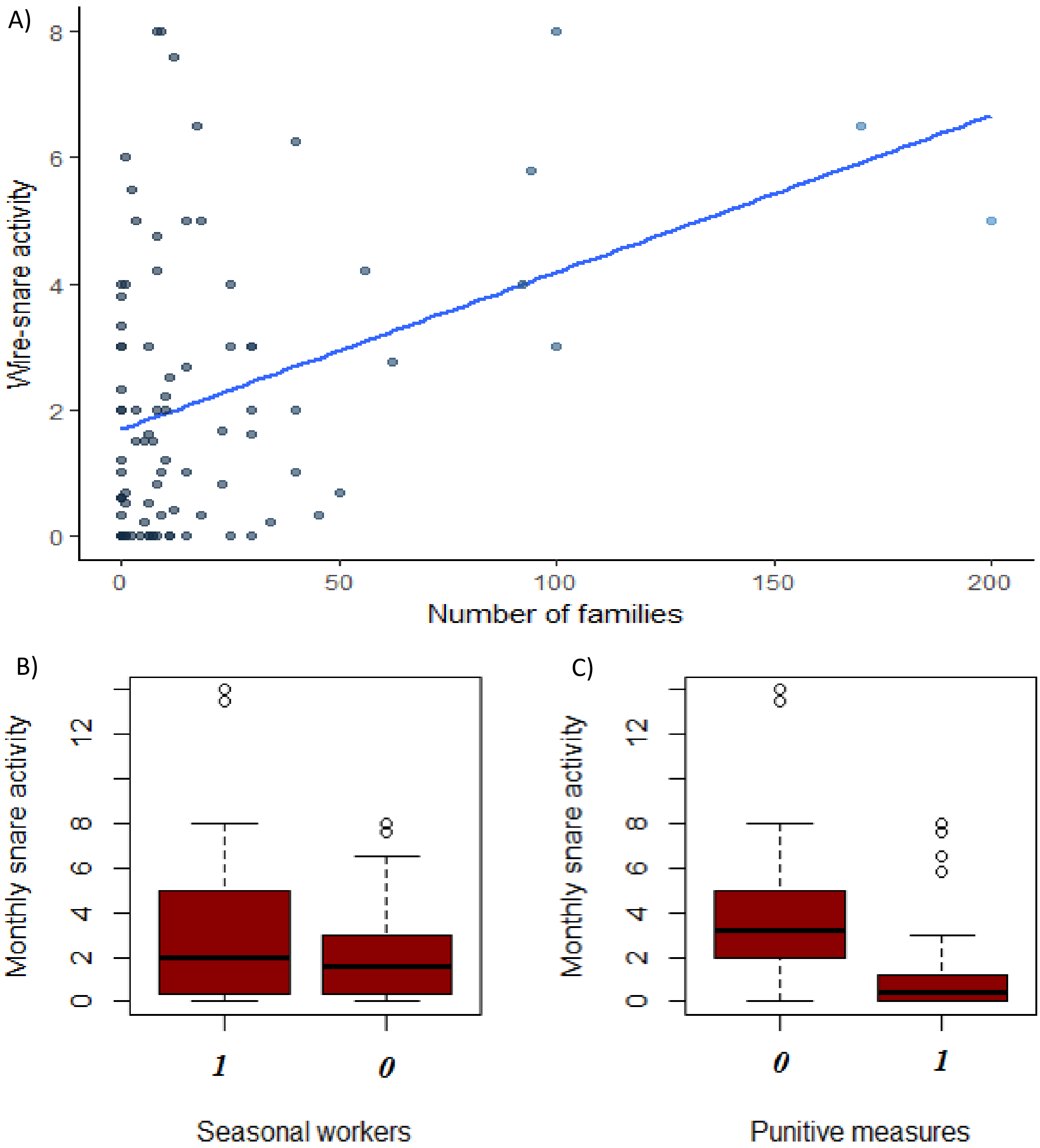

Figure 2.8. The relationship between the mean number of snares found on properties (n = 103) per month (snaring activity) and (a) the number of families permanently living on the property, (b) the absence or presence of seasonal/contract workers on the property (±SE), and (c) the absence or presence of enforced punitive measures on the property.

2.4.5 Wire-snare locations in relation to geographic abiotic variables

Reported wire-snare density was significantly influenced by proximity to PA's, proximity to major residential areas (> 5 000 residents) and proximity to major roadways (N and R), as well as the mean elevation of the site ($F_{(4,102)} = 23.15$; $P < 0.0001$) (Table 2.4; Fig. 2.9).

Table 2.4. Generalized linear models that were within 4 AICc units of the highest ranking model (lowest AICc), with associated degrees of freedom (df), number of parameters in the model (k), AICc, ΔAICc and Akaike model weights.

Model	df	k	AIC_c	ΔAIC_c	Weight
Response variable: Mean number of snares found at each location in the month prior to the interviews					
1. Cts[1] + Elv[2] + Pas[3] + Rdws[4]	6	7	400.0	0.00	0.367
2. Elv + Pas + Rdws	5	6	401.5	1.53	0.171
3. Cts + Elv + Pas + Rvs[5] + Rdws	7	8	402.3	2.33	0.114
4. Cts + Elv + Pas + Rvs + Wtr[6]	7	8	402.3	2.34	0.114
5. Elv x Pas	17	18	403.3	3.33	0.069
6. Cts x Rdws	17	18	403.6	3.62	0.069

[1]Cts = Residential areas
[2]Elv = Elevation
[3]Pas = Protected areas (PA's)
[4]Rdws = Major roadways
[5]Rvs = Riparian corridors
[6]Wtr = Permanent water bodies

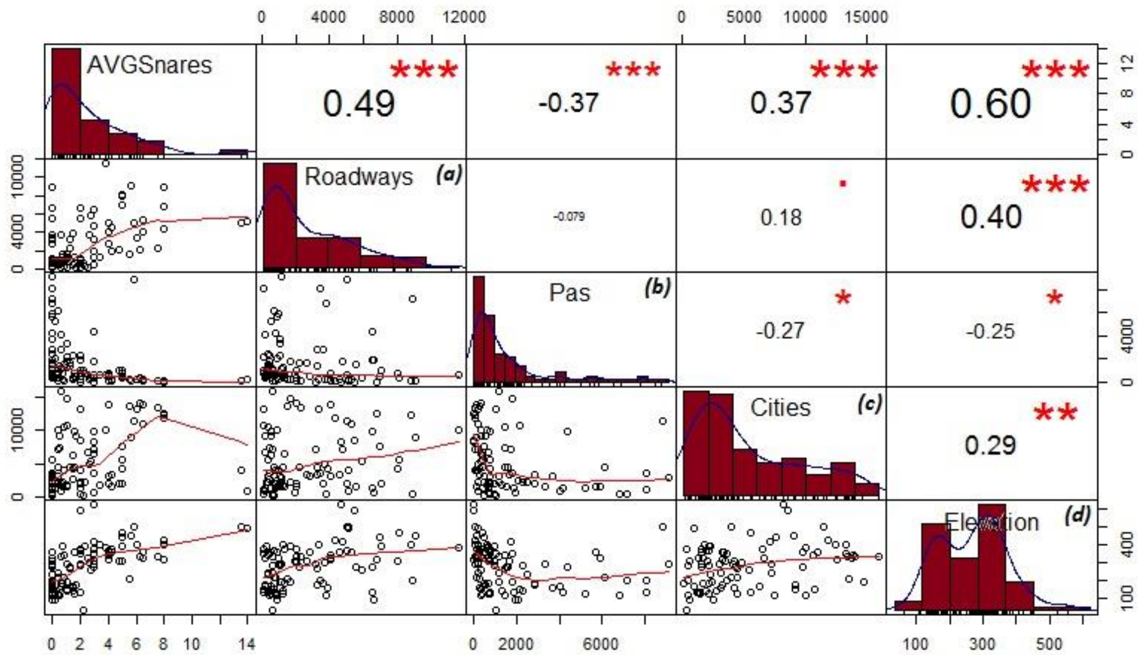

Figure 2.9. The relationship between snaring activity (measured at the centroid coordinate of wire-snare landscapes) and the geodesic distance (m) to the nearest (a) major roadways (N and R), (b) protected area, or (c) major residential areas (> 5 000 residents). (d) The relationship between snaring activity and elevation (m).

Reported wire-snare density was high close to or on the borders of PA's, and decreased with distance away from PA's ($\beta < 0.0001$; SE < 0.0001; t = -2.64; $P < 0.0001$). Snare density increased as distance from major residential areas increased ($\beta < 0.0001$; SE < 0.0001; t = 1.92; $P < 0.05$). Snared density increased further away from major roadways ($\beta < 0.001$; SE = 0.0001; t = 3.61; $P < 0.001$), and snare density was highest at high elevations ($\beta < 0.01$; SE < 0.001; t = 4.56; $P < 0.01$). Permanent water bodies and riparian corridors had no significant effect on the distribution of wire-snares.

2.4.5.1 Random forest prediction

The algorithm provided four node splits based on the five most integral factors determining properties with high incidence of wire-snare activity (Fig. 2.10) (lm, n = 500). These were, in order: the presence or absence of punitive measures (MSE = 7.71; SD = 0.26; n = 93), the number of families residing on the property and the absence or presence of seasonal workers (MSE = 3.76; SD = 0.28; n = 49), the mean elevation of the property (MSE = 8.30; SD = 0.20; n = 44), and the farm's proximity to major residential areas (MSE = 0.66; SD = 0.19; n = 34).

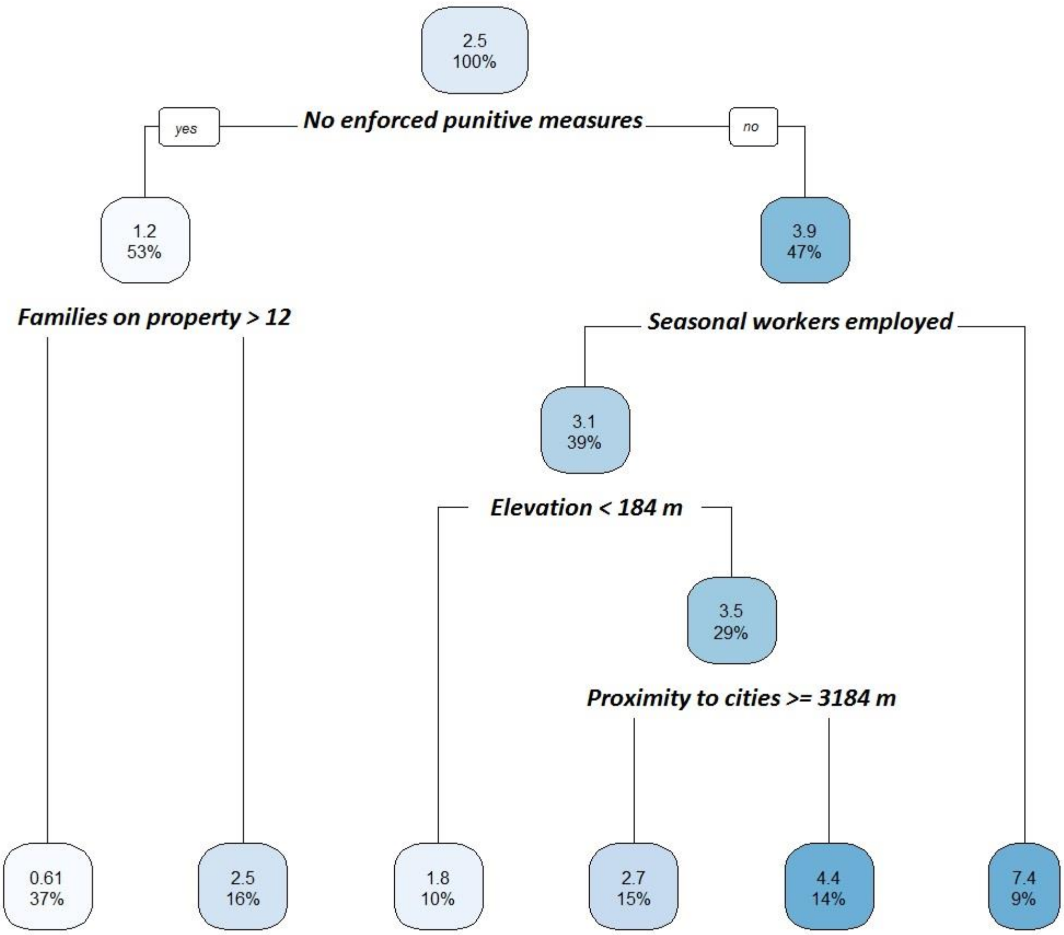

Figure 2.10. Decision tree for determining whether a given property is likely to be susceptible to wire-snare poaching.

2.4.6 Temporal variation in wire-snare abundance

Most respondents (59.2%) reported that there was no specific time during the year when they found more snares, with 32.7% suggesting there are more in summer, 4.1% in autumn, 3.4% in winter, and 0.7% in spring (Fig. 2.11). The variation in peak times during the year when wire-snares were found by labourers was found to be significant (anv, $F_{(4,142)} = 3.88$; $P < 0.05$), and a post-hoc test revealed that significantly more snares were found during summer months compared to other seasons and each other ($P < 0.05$; Tukey HSD).

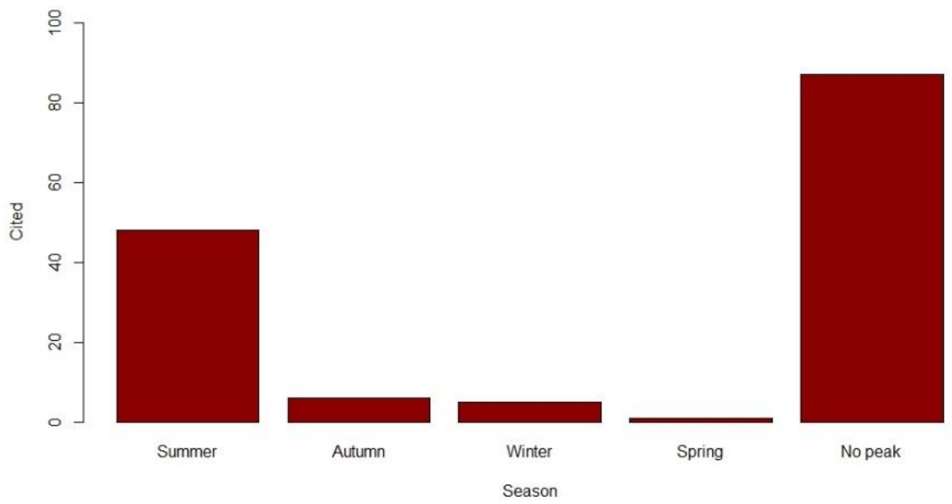

Figure 2.11. Seasonal peaks in wire-snare activity observed by respondents. The majority of respondents observed no discernible peaks in wire-snare activity during any given season. Hot and dry summer months nonetheless had more wire-snare activity than wet and cool winter months.

2.4.7 Species affected by wire-snare poaching

There was significant variation in the species most desired by wire-snare poachers (H = 10.82; df = 10; $P < 0.05$), as well as the species most frequently caught in snares (H = 18.25; df = 10; $P < 0.05$). There were no significant differences between the species desired by wire-snare poachers and the species frequently caught by poachers (Chisq, $\chi^2 = 26.50$; df = 21, $P = 0.19$). A post-hoc test revealed grey duiker (*Sylvicapra grimmia*), Cape grysbok (*Raphicerus melanotis),* Cape porcupine (*Hystrix africaeaustralis*), landfowl (includes most notably helmeted guineafowl – *Numida meleagris*), feral pigs (*Sus scrofa*), grey rhebok (*Pelea capreolus*), and klipspringer (*Oreotragus oreotragus)* were the most desired species (Fig. 2.12a), and were significantly different from other species or morphospecies and each other ($P < 0.05$; Dunn posthoc comparisons). Similarly, post-hoc analysis revealed rates of capture differed among species ($P < 0.05$; Dunn posthoc comparisons), with grey duiker, Cape porcupine, Cape grysbok, landfowl, feral pigs, grey rhebok and feral/domestic dogs (*Canis lupus familiaris*) caught the most (Fig. 2.12b).

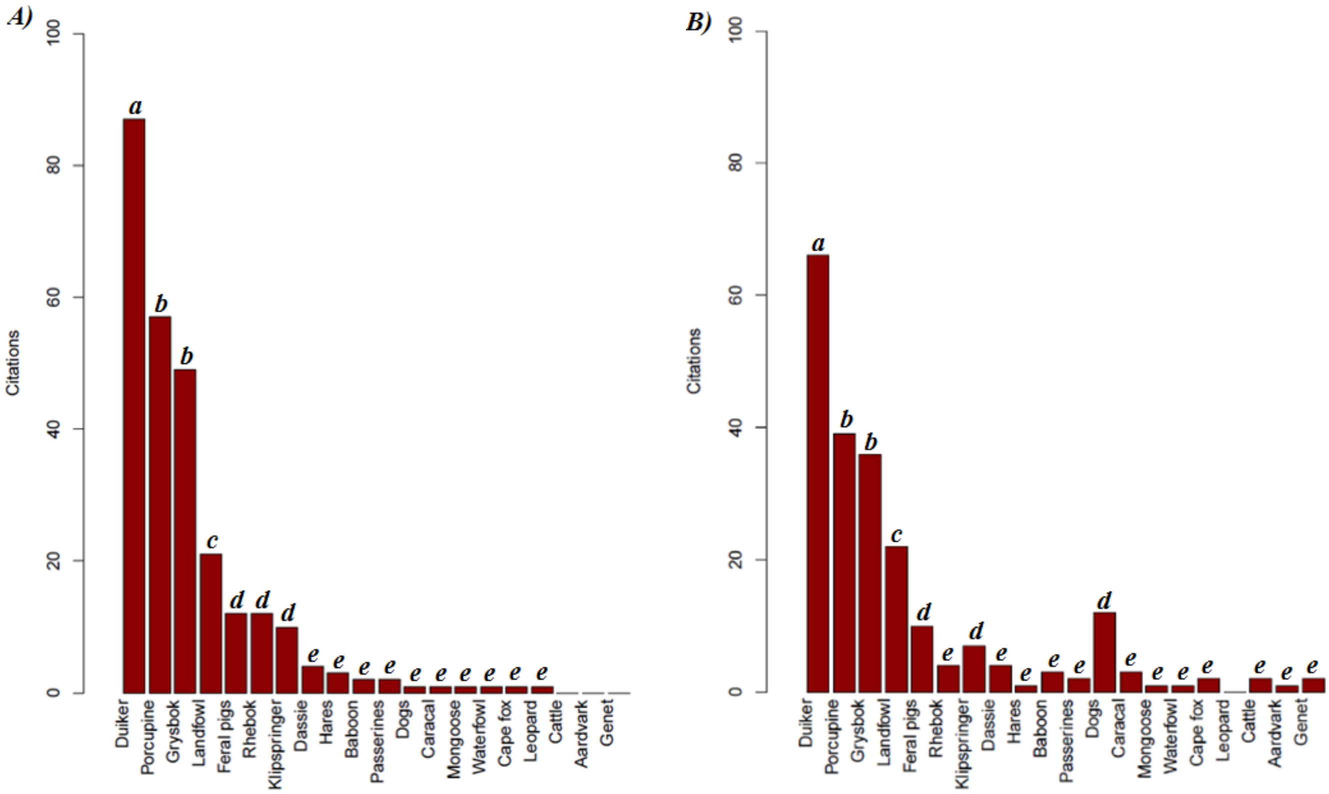

Figure 2.12. The animal species and morphospecies (A) most desired by wire-snare poachers, and (B) most frequently caught in wire-snares across the study area. Significant differences among groups revealed by Dunn's post-hoc comparisons are denoted by letters a - e.

2.4.8 Wire-snare activity hotspots

A clear pattern emerged in the mean number of snares that were reported to be set and found by labourers on surveyed properties in the month prior to interviews. Four distinct hotspots appeared with varying intensities (Fig. 2.13). The (1) Agter-Groenberg farming area had the highest concentration of wire-snares, with the most number of wire-snares found in this area close to the Groenberg Nature Reserve near Wellington. In decreasing intensity, the remaining hotspots were concentrated in (2) the Slanghoek farming valley adjacent to the Hawequa Nature Reserve and close to Rawsonville, as well as opposite the N1 roadway closer to the Brandvlei Nature Reserve, (3) the Elandskloof farming valley situated between the Theewaters Nature Reserve and the Southern end of the Hawequa Nature Reserve, and (4) the North-eastern slopes of the Kogelberg Nature Reserve, close to Grabouw.

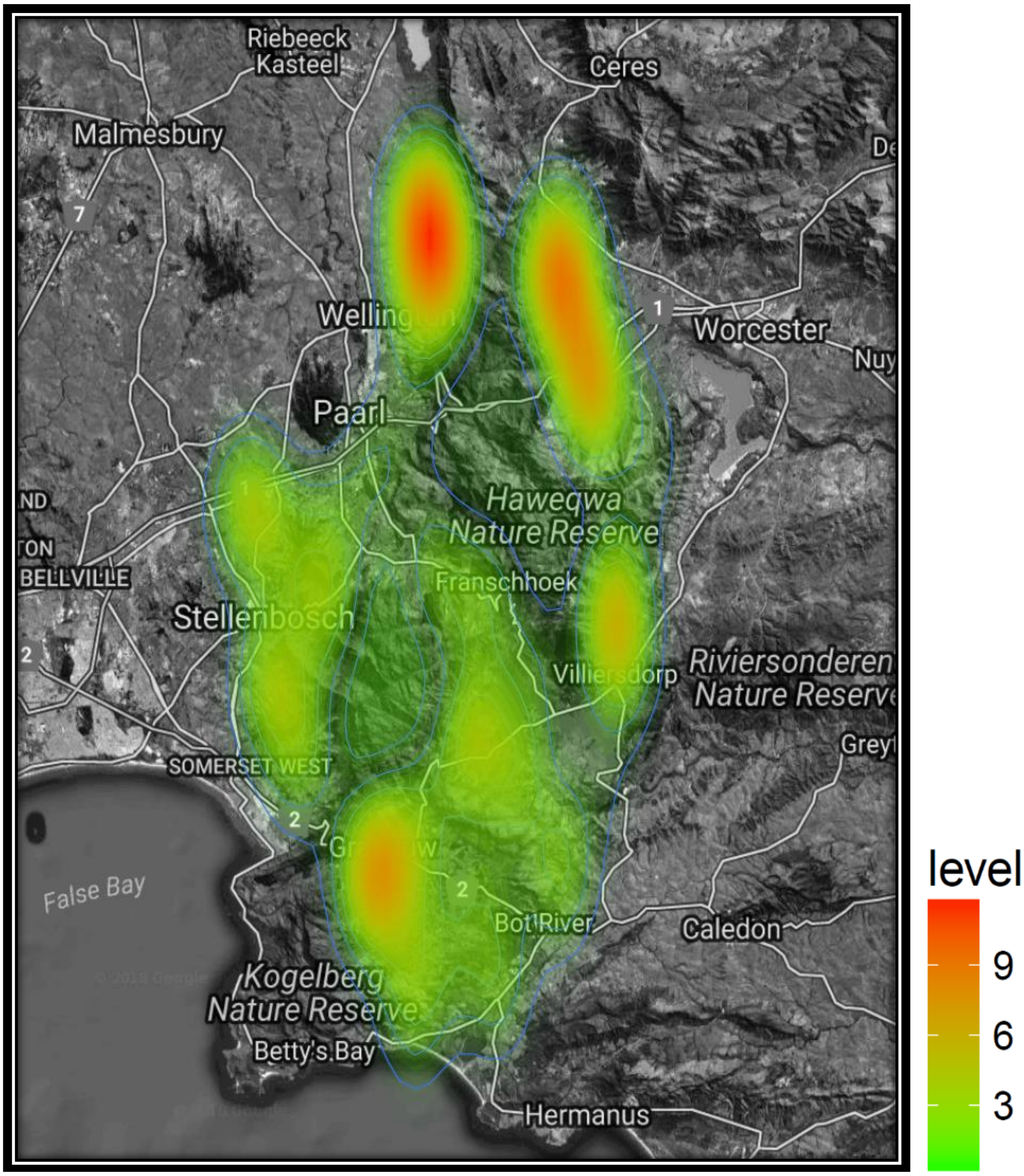

Figure 2.13. Heat map displaying wire-snare activity hotspots, measured as the mean number of wire-snares found on properties by labourers. The intensity of the hotspot is indicated by a colour change between shades of green (least concern) and red (highest concern).

2.5 Discussion

The poaching of bushmeat with the use of wire-snares has become an increasing and devastating threat to the persistence of an array of species within their natural ranges (Watson et al. 2013). Therefore, identifying individuals and areas most involved in snaring activity, as well as assessing the extent and nature of wire-snare activities, seems to be the logical first step in devising effective ameliorating strategies. Quantifying accurate data on any illegal activity is however difficult due to people's fear of prosecution, as well as the assumed cryptic nature of the activity (Gavin et al. 2010). Similarly, accurately characterising the underlying dynamics of illegal wire-snare poaching is difficult, especially on the individual level. Questionnaires offer an anonymous and intimate alternative to collecting sensitive information, and have gained popularity during the last few decades as an effective means for obtaining quantifiable, large-scale data in ecology (White et al. 2005), especially in heterogeneous landscapes (Becker et al. 2013). The reliability of information collected for this study was however dependent on the fidelity of the responses provided. The study nonetheless provides valuable insight on the dynamics of several interwoven factors that determines wire-snare poaching patterns, such as economics, politics, ethnicity and ecology. Wire-snare activity in the Boland Region is driven largely by a need for food supplementation, fuelled by factors such as family size, ineffective or absent governing regulations set forth by landowners, and interpersonal development.

Figure 2.14. The interview process. Photo credits: B.C. Schultz.

2.5.1 Poaching economics

A number of factors indicated that poverty might have a substantial effect on the prevalence of wire-snare incidence. The concept of poverty is multifaceted and is thus defined by various social, economic and political elements. By definition, poor communities are often unable to sustain basic survival needs with their monthly income alone, and consequently resort to illegal activities to subsist – such as wire-snare poaching (Lindsey et al. 2011a; Gandiwa et al. 2013). Brashares et al. (2011) summarized two contrasting hypotheses (Robinson & Bennett 2002; Milner-Gulland & Bennett 2003; Brashares et al. 2004) in an attempt to explain the bushmeat harvesting patterns that might emerge from a state of poverty. Firstly, the *"bushmeat as inferior good"* hypothesis states that poorer, rural households will typically consume more bushmeat since it provides an inexpensive and accessible source of food and income during times of economic adversity (De Merode et al. 2004). This hypothesis further argues that bushmeat is consumed as a measure of last resort, and the dependency thereon will thus decrease as household wealth increases. Secondly, the *"bushmeat as normal good"* hypothesis states that bushmeat consumption increases as wealth increases, similar to patterns of supply-and-demand observed in other household commodities (Brashares et al. 2004). Similar to studies previously conducted in the Serengeti National Park, Tanzania (Loibooki et al. 2002; Kaltenborn et al. 2005), the current study supported the former by showing that people in the Boland mainly resort to harvesting bushmeat with wire-snares to sustain their own dietary needs. Contrastingly, other studies have found that food insecurity was either not linked to hunting activities (Brashares et al. 2011), or that hunting increased as food security lessens (Nuno et al. 2013), supporting the *"bushmeat as normal good"* hypothesis.

Larger family sizes typically imply that labourers have more dependents to support, and would potentially experience greater poverty. Therefore, as expected, our study revealed that individuals from larger families are more frequently involved in wire-snare activities. Hunting activities can however decrease as family size increases (Johannesen 2005), and it is assumed that in these instances more family members translates to more income sources.

It is further commonly recognized that poor communities do not only rely on hunting for self-nutritional purposes, but tend to form vast commercialized sectors for the trade of bushmeat in order to supply additional monetary support (Nasi et al. 2008; Brashares et al. 2011), or replace other revenue-generating activities altogether (Knapp 2012). These trade sectors are commonly found in central African countries (Bowen-Jones et al. 2003; Nyaki et al. 2014), but recently they have also been identified in South Africa (Warchol & Johnson 2009; Grey-Ross et al. 2010), as well as in neighbouring countries (Fusari & Carpaneto 2006; Lindsey et al. 2011a, 2011b; Lindsey & Bento 2012). The current study was however unable to identify a discernible trade sector for bushmeat in the Boland Region, concluding that if any such sectors should exist in the area, it is likely not driven by permanently employed farm labourers.

Seasonal workers often lack a reliable income and are consequently expected to face periods of economic adversity during which they struggle to sustain their own dietary needs. It is therefore unsurprising that we found properties relying on this form of labour force had on average more snares present. Since hunting is often driven by a lack of jobs (Gandiwa et al. 2013), as substantiated in the findings, a potential bias is expected in the results since the surveyed population predominantly included permanently employed labourers with a fixed income, and did not consider unemployed people living in informal settlements in close proximity to properties. These people are expected to experience greater economic pressures associated with food insecurity (Gandiwa et al. 2013), and were often blamed by respondents in the study as being responsible for snares found on the outermost fences of farmed properties. Poor rural communities are further expected to rely on inexpensive hunting techniques. Wire-snares are cheaply and effortlessly made from any wire-related materials, such as fence and electricity lines (Lindsey et al. 2011b), and thus offer an attractive means of obtaining bushmeat for poor communities (Lewis & Phiri 1998). The current study corroborated that the low-cost and simple nature of wire-snares are dominant drivers motivating their use.

This study thus strongly supported the notion that economic factors relating to poverty influences patterns in wire-snare poaching. However, poverty alleviation may not necessarily translate to a decrease in wildlife off-take, since communities that transition from a state of poverty often merely transition to more efficient and direct hunting techniques (Damania et al. 2005; Nielsen et al. 2012; Nuno et al. 2013). Furthermore, in some African cultures hunters are often wealthier than non-hunters (Knapp 2007), enjoy elevated social status (Brown 2007) and are preferred by women (Lowassa et al. 2012). Economic theory does however suggest that provisioning rural communities with access to affordable and conventional substitutes to bushmeat may promote wildlife conservation by reducing wire-snare poaching (Wilkie et al. 2005).

2.5.2 Inadequate policy development, law enforcement and penal systems

Increased illegal bushmeat off-take has been linked to a lack of clear regulations in several African countries, for example in Gabon (Fitzgibbon et al. 1995), Tanzania (Hofer et al. 2000; Haule et al. 2002), Mozambique (Fusari & Carpaneto 2006; Lindsey & Bento 2012), and Kenya (Saru 2012). Similarly, increased wire-snare use was observed by individuals employed on properties that they claimed were lacking appropriate anti-snaring regulations associated with adequate enforcement and punitive measures. However, despite labourers often believing they are allowed to use wire-snares, property owners tend to claim otherwise. This is indicative of a lack of communication existing between landowners and their labourers. Substantiating this finding, properties lacking enforced punitive measures had, on average, more snares present. Although more strongly regulated governing regulations are therefore suggested, there is a paradox in existence wherein a blatant dismissal or monetary fine is potentially counterproductive if it increases economic hardship for the individual or his family and friends, potentially exacerbating the issue by encouraging the need for additional resources obtainable through bushmeat poaching (Knapp 2012). Therefore, stronger law enforcement measures should be coupled with efforts to extend benefits in the form of alternative livelihoods or protein sources (Keane et al. 2008; Brashares et al. 2011).

To resolve this issue, a top-down management regime adopting the following proposals is suggested: (1) Conservation bodies and agencies should educate and encourage landowners to become involved in conservation-orientated initiatives, to promote a holistic approach to farming, and thereby combat issues such as wire-snare poaching. This study further showed that wire-snare poaching was more prevalent on properties with contract workers in employment, presumably at least partially due to the lack of repercussions faced when not in permanent employment at a specific farm, and it would therefore be of further benefit for conservation bodies to approach third party employment agencies issuing these contract workers. (2) Farmers should in turn develop or join numerous existing regional conservancies in their area, which urges them to adhere to various issues of conservation concern. These conservancies also provide a valuable platform to form communication networks between landowners, as well as conservation agencies. Disparity in the perceptions on conservation problems between farmer groups often impedes the application of conservation initiatives (Biggs et al. 2011), and it is therefore important to facilitate communication between landowners. (3) Landowners should further ensure that their labourers are educated with regards to the issue of wire-snare poaching, and that they comply with the anti-snaring regulations set forth. On 51.4% of the properties included in the study and on which regular snaring activity occurred, authorities (owners and managers) were unaware of any such activity. To ensure better communication and compliance, regular information sessions should be held by property owners. To further ensure compliance, landowners are encouraged to include labourers in anti-snaring initiatives, for example by giving them the responsibility to monitor fence lines on a regular basis, and thereby also removing all snares encountered.

In some regions surveyed in the study, anti-snaring regulations set forth by landowners were abandoned in an attempt to rid human-wildlife conflict-inducing problem species that cause large-scale destruction to property infrastructure and crops. In these instances the use of wire-snares by labourers was permitted or overlooked by farmers. This behaviour specifically pertained to the invasive European boar *(S. scrofa)*, and included the Northern regions of the study area, such as the farmlands surrounding Wellington and du Toitskloof. European boars, or feral pigs, are a problem in many parts of the world (Hone 2002), and were likewise introduced to south-western South Africa, where they have since escaped confinement and spread beyond their intended boundaries (Skead et al. 2011). They are largely detrimental due to their tendency to till large areas of soil in search for roots, stems and macroinvertebrates (Kotanen 1995), and have been reported to destroy croplands (Schley & Roper 2003) and transmit diseases to livestock (de la Fuente et al. 2014). Therefore their eradication is a top priority for many agricultural stakeholders. The use of wire-snares is however not an effective method for combatting this issue, primarily due to its indiscriminate nature which directly affects native wildlife (Lindsey et al. 2011a). Hence this form of control should be abandoned and regulations should be reinstated.

2.5.3 Circumstances that provide opportunities for wire-snare use

An individual's ability to access areas suitable for wire-snare use is presumed to strongly influence their involvement in snaring activities, as well as the likelihood of being caught (Felson & Cohen 2017). The number of families living on a property was used as a proxy for accessibility, since the vast majority of snares are presumably set after work hours and over weekends. Permanent residents on the property would thus have more opportunity to set wire-snares. Results showed that wire-snare incidence increased with an increased number of families living on farms. However, the majority of labourers that did not live on the property were resident in nearby informal settlements or small residential communities, and in walking distance from properties with natural resources, and were just as likely to set snares as workers living on farms. Wire-snares were also less likely to be set in close proximity to major residential areas (> 5 000 permanent residents), as well as closer to major roadways, most likely avoiding witnesses, as increased human traffic poses a risk of getting reported or apprehended (Haines et al. 2012). Loibooki et al. (2002) found similar trends, and it can further be assumed that wildlife abundance would be greater further away from residential areas and highly frequented roads. A lack of snaring near residential areas also further substantiates the absence of commercialized trade of bushmeat among farm labourers, as bushmeat prices and accessibility typically increases with proximity to urban areas (Brashares et al. 2011), and communities further away from residential areas would thus be more likely to hunt solely for self-nutritional purposes. Potentially, it further indicates that wire-snares are not set by residents of major residential areas, as effective snaring would require frequent checking of snare lines to prevent losses to scavenging species and decay (Watson et al. 2013), therefore wire-snares would occur in greater densities closer to residential areas.

Wildlife abundance is also presumed to be greatest close to natural and protected areas, promoting accessibility and opportunity for hunting activities (Hofer et al. 1996; Campbell et al. 2001; Nielsen 2006; Wato et al. 2006; Brashares et al. 2011; Watson et al. 2013). Likewise wire-snare densities were high close to PA's, and decreased with distance away from them. The proportion of the farm covered with natural, untransformed vegetation however had no effect on wire-snare occurrence.

2.5.4 Ethnicity, interpersonal development and past experiences

Ethnicity is determined by the interrelationship between various factors such as race, language and culture. In general, the findings showed that these types of factors do not play a role in wire-snare involvement. This study was partially biased towards men since the number of female respondents was far less, mainly because female labourers tend to be primarily involved in indoor activities, thereby excluding them as potential candidates for the study based on participant criteria. Nonetheless, hunting is predominantly a male activity (Brown 2007; Mfunda & Røskaft 2010; Lindsey et al. 2011b), and similarly this study revealed no female participants involved in any wire-snare activities.

Numerous studies conducted across Southern Africa indicated that traditional medicinal uses and other cultural aspects strongly correlate with increased bushmeat off-take (Van der Westhuizen 2007; Warchol & Johnson 2009; Grey-Ross et al. 2010). However, we found no significant cultural motivation for the use of wire-snares in the Boland Region. The geographic region in which you mature may also strongly affect the culture you adopt, and immigrants are often found to participate more in hunting activities (Mfunda & Røskaft 2010). The current study revealed that local people, as well as people from the Northern Cape Province of South Africa are more involved in wire-snare practices than immigrants from other regions of Southern Africa.

The decision to be involved in any kind of activity is further likely to be shaped by experiences during one's formative years, as shown with regards to wire-snare behaviour. Far less people seem to learn snaring behaviour from peer group activities during mature life-stages, but rather learn this behaviour through observing older relatives during their formative years. This suggests that educational interventions should focus on older people participating in wire-snare activities, and in doing so inform and appeal to them that such behaviour is not passed on to their kin. Older respondents were also more likely to be involved in snaring practices, as well as respondents with a longer job tenure, further substantiating this claim.

2.5.5 Ecological aspects of wire-snare patterns

A wide range of animal species affected by wire-snare poaching were recorded in this study. Snares were predominantly placed along game paths that are frequented by a vast number of taxa, especially where these trails intersect fences – with the exception of funnel-shaped wire-snares, which are placed exclusively inside the burrows of porcupines. Even though snaring is largely non-selective, in the majority of cases, the intended species were caught. Small antelope such as grey duiker, Cape grysbok, klipspringer and grey rhebok were the most desired species, and correspondingly had high off-take rates. Cape porcupine, landfowl spp. and feral pigs were also highly sought after, and had significantly high rates of off-take. These findings are significant because small antelope and porcupine forms the primary prey base of the only remaining apex predator in the Western Cape, namely the Cape leopard (*P. pardus*) (Norton 1986; Martins et al. 2011). Without its primary prey base, leopard populations will likely decline (Ray et al. 2005), and several detrimental ecosystem responses will be elicited, such as secondary extinctions and trophic imbalances (Miller et al. 2001; Fryxell et al. 2007). The unsustainable removal of these prey species are thus bound to have severe consequences. Reports of unintended species off-take, most notably feral and domesticated dogs, genet spp. (*Genetta genetta* and *G. tigrina*), caracal (*C. caracal*), aardvark *(Orycteropus afer*) and Cape fox (*Vulpes chama*), are also worrying.

An increase in wire-snare activity is often observed during hot and wet summer months when there is an increase in wildlife activity (Kaltenborn et al. 2005; Holmern et al. 2007; Becker et al. 2013). The study area is however subject to a Mediterranean climate, which is typified by dry summer months. However, Knapp (2007) suggested that the decision to poach may be largely predicated on time availability. Summer months in the Boland are thus nonetheless expected to experience more snaring activity purely because it provides more daylight opportunity at the end of the work-day to set and inspect snares. Furthermore, due to the hot and dry nature of the summer months in our study area, coinciding with the peak harvest periods, wildlife may actually be more abundant on properties due to a lack of food and water in natural fynbos-dominated areas during the summer, as opposed to the food- and water-rich farmlands. Although most of our respondents reported that there is no observable difference in the amount of snares found on properties at any time of the year, the majority of those that did make a distinction cited summer months as having a greater wire-snare presence. This may be partially biased however due to the nature of our data collection i.e. respondents may simply see more snares due to a more visible landscape during the dry summer months facilitated by decreased vegetation cover due to veldfires and senescence, irrespective of the true representation of wire-snare incidence.

Wildlife numbers presumably concentrate around reliable water sources, and snares are therefore expected to occur in greater densities close to permanent water bodies (Haines et al. 2012; Watson et al. 2013). However, no significant influence of permanent water bodies or riparian corridors on wire-snare distribution was found in the current study. This result is potentially explained by the fact that all agricultural properties have permanent water supplies in close proximity.

2.6 Conclusion

Collecting information on sensitive topics from community members to describe anthropogenic impacts on the environment will remain a complex endeavour, and many have found such approaches to underestimate (Knapp et al. 2010) or overestimate (Loibooki et al. 2002) actual pressures. Concurrently, this study leaves little doubt that the use of wire-snares remains a multifaceted and complex issue that is unlikely to be resolved in the near future. The approach to this study opened a dialogue between rural communities and conservation agencies on an issue that has received no formal attention to date, and it is suggested that the application of unilateral policies be implemented in conjunction with further studies to empirically validate these findings and broaden the understanding of the heterogeneity in local scale socio-ecological dynamics. Conservation policies and governing regulations are often seen as enigmatic and not easily understandable by local people (Kideghesho 2008), especially when those policies are developed without a clear understanding of the needs and challenges rural communities face. By providing a novel and valuable assessment of an array of socio-economic and biophysical factors influencing wire-snare poaching, this study provides information that can potentially be used to develop effective management and eradication frameworks focused on an ever-increasing threat to wildlife populations in the Boland Region of South Africa.

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2.8.1 Appendix 2.1. Supplementary Tables for chapter 2 and chapter 3

Supplementary Table 2.1. Personal characteristics of all labourers (n = 307), labourers involved in wire-snare poaching activities (n = 42), and landowners or managers (n = 103) involved in the study. *INR* = Information not recorded.

Personal characteristics	Labourers (n = 307)	Poachers (n = 42)	Farmers (n = 103)
Gender	82.4% male	100% male	94.2% male
Age			
(years, mean ± SE)	42.4 ± 0.70	47.0 ± 0.62	48.9 ± 1.24
Age class			%
< 20	0.7	0	0
20-29	16.6	9.5	3.9
30-39	23.4	16.7	23.3
40-49	26.7	19.0	27.2
> 49	32.6	54.8	45.6
Family size			
(mean ± SE)	3.6 ± 0.12	5.0 ± 0.11	*INR*
Employment period			
(years, mean ± SE)	15.1 ± 0.62	19.8 ± 0.70	22.7 ± 1.70
Place of residency	64.5% on property	66.7% on property	89.3% on property
Race			%
Black African	32.9	23.8	0
Coloured	66.8	76.2	1.9
White	0.3	0	98.1
Home language			%
Afrikaans	77.2	90.5	63.1
English	0	0	36.9
Xhosa	16.0	7.1	0
Other (Shona, Chewa, Chitumbuka, Chilomwe, Zulu, Sotho, Tumbuka)	6.8	2.4	0
Place of origin			%
Western Cape	73.9	81.0	82.5
Northern Cape	3.9	7.1	1.9
Eastern Cape	16.6	11.9	1.9
Elsewhere in Africa	4.6	0	1.1

Elsewhere in South Africa	1.0	0	9.7
Elsewhere worldwide	0	0	2.9

Supplementary Table 2.2. Characteristics of properties included in the study (n = 103).

Property characteristics	
Primary agricultural output	**%**
Vineyards	45.2
Orchards	43.0
Animal production	11.8
Punitive measures present	52.7% of properties
Property size (ha, mean + SE, range)	436.3 ± 71.8, 5-4107
Proportion natural vegetation (%, mean + SE, range)	48.3 ± 2.85, 0-100
Families on property (mean ± SE, range)	22.8 ± 4.43, 0-270
Owner residency	88.2% on property
Seasonal workers present	81.7% of properties

Supplementary Table 2.3. All species cited by labourers as desired by wire-snare poachers (n = 123 respondents), and species caught by wire-snare poachers within one month prior to the interview (n = 110 respondents).

	Species desired (n = 123 respondents)		**Species caught (n = 110 respondents)**	
Species	**Cited**	**Respondents (%)**	**Cited**	**Respondents (%)**
Sylvicapra grimmia	87	70.73	66	60.00
Hystrix africaeaustralis	57	46.34	39	35.45
Raphicerus melanotis	49	39.84	36	32.73
Landfowl spp.	21	17.07	22	20.00
Sus scrofa	12	9.76	10	9.09
Pelea capreolus	12	9.76	4	3.64
Oreotragus oreotragus	10	8.13	7	6.36
Procavia capensis	4	3.25	4	3.64
Lepus spp.	3	2.44	1	0.91
Papio ursinus	2	1.63	3	2.73
Passerine spp.	2	1.63	2	1.82
Alopochen aegyptiaca	2	1.63	1	0.91

Caracal caracal	1	0.81	3	2.73
Mongoose spp.	1	0.81	1	0.91
Vulpes chama	1	0.81	2	1.82
Panthera pardus	1	0.81	0	0.00
Canis lupus familiaris	1	0.81	12	10.91
Orycteropus afer	0	0.00	1	0.91
Genetta spp.	0	0.00	2	1.82
Bos spp.	0	0.00	1	0.91

Chapter 3
Farmer Attitudes and Regional Risk Modelling Of Human-Wildlife Conflict on South African Farmlands

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3.1 Abstract

Human-wildlife conflict in unprotected areas, especially those bordering reserves, has resulted in the global home range and population size reduction of naturally occurring wildlife. Simultaneously, rural communities and commercial farmlands at the interface of human development and natural habitat face severe threats to their livelihoods and agricultural security, often resulting in the vast eradication of real or perceived damage-causing animals (DCA's). The knowledge of local people was relied on to elucidate the dynamic and interwoven social, economic and ecological factors giving rise to the largely undocumented conflict between landowners and wildlife in the Boland Region of South Africa. Subsequently, the spatial location of observed and expected zones of species-specific risk on a regional level was anticipated and mapped using a maximum entropy algorithm. Local, male farmers managing large commercial properties affiliated with regional conservancies were most likely to rely on the lethal control of DCA's. The highest level of tolerance by farmers was shown for primates and ungulates, while tolerance for carnivores, avifauna and invasive or feral species were comparatively lower. Zones of conflict varied significantly between species and areas, but were most intense for baboons, porcupines, caracal, feral pigs and dogs, otters, duikers, and honey badgers. The results presented in this document will enable the prioritisation of locations and species to create improved mitigation and management plans, as well as provide for more accurate allocation of conservation resources to minimize conflicts, optimize

agricultural yield, reduce wildlife off-take, and ultimately ameliorate human-wildlife conflict.

Keywords: Agricultural security, crop-raiding species, damage-causing animals, depredation, ecological niche modelling, human-carnivore conflict, retaliatory killing

3.2 Introduction

At the interface of agricultural or urban expansion and natural habitats, local communities and wildlife frequently compete for shared resources, and consequently enter into conflict scenarios (Woodroffe 2000; Woodroffe et al. 2005; Inskip & Zimmermann 2009). The incidence of human-wildlife conflict (HWC) is believed to be on the increase globally (Karanth 2002; Treves et al. 2002; Anthony et al. 2010) and is especially substantial and disproportionate for small-scale and subsistence farmers close to protected areas (PA's); therefore challenging a synergy between rural development and biodiversity conservation. At one end, the livelihoods of people depending solely or partially on crop or animal husbandry as a source of income (Barua et al. 2013) are negatively affected (Marker & Dickman 2005). At the other end, wildlife is forced to enter into conflict with people, or take refuge in small, fragmented landscapes (Pettigrew et al. 2012; Kiffner et al. 2015); ultimately reducing population size and viability. Naturally occurring wildlife are increasingly being acknowledged for their existential, economic and ecological value, providing impetus for biodiversity conservation, but are stymied by their persecution as damage-causing animals (DCA's) on a global scale (Treves & Karanth 2003; Graham et al. 2005; Inskip & Zimmermann 2009; Stein et al. 2010).

Among the plethora of terrestrial species associated with HWC, large charismatic carnivores are inarguably the most frequently reported (Inskip & Zimmermann 2009). Consequently, large carnivores are often described as inimical to animal farming and viewed as undesirable at a local level (Woodroffe et al. 2005), where they tend to experience the worst consequences of human-carnivore conflict (HCC) (Treves & Karanth 2003; Graham et al. 2005). The high incidence of HCC is largely attributed to the protein-rich diets and extensive home ranges of large carnivore species (Treves & Karanth 2003), drawing them into areas where their spatial and dietary needs overlap with those of humans (Linnell et al. 2001). The important regulatory roles fulfilled by carnivores in terrestrial ecosystems are however undisputed (Estes et al. 2011), resulting in carnivores being widely regarded as flagship species at national and global levels (Treves & Karanth 2003), and thus requiring substantial conservation interventions to promote their continued existence. Nonetheless, HCC has greatly contributed to the distribution and population declines that carnivore species experienced during the previous century (Woodroffe & Ginsberg 2000), resulting in human persecution being listed as the main threat to carnivores outside PA's (Friedmann & Daly 2004).

Comparatively, little research has been done on conflict between people and crop-raiding species, despite crop damage being described as the most prevalent form of HWC in both Asia and Africa (Parker et al. 2007) – thus largely shaping public perceptions of conservation as it threatens the general welfare of people (Lee & Graham 2006). This is likely due to the general impression that herbivores are less important ecologically and typically have stable populations (Parker et al. 2007), while this may not necessarily be the case. Crops are particularly attractive to wild animals because of the effect selective breeding has had on the natural physical and chemical defences of crops, as well as its nutritional value (Purseglove 1972). It should therefore be no surprise that wildlife frequently raid croplands, especially during times of food scarcity, when crops offer additional energetic advantages (Forthman-Quick and Demment 1988; Naughton-Treves et al. 1998; El Alami et al. 2012). Consequently, the livelihoods, food security and agricultural security of many people become severely threatened (Kaplan et al. 2011; Barua et al. 2013; Taruvinga & Mushunje 2014), resulting in the direct persecution of DCA's.

Few prior studies have addressed HWC in the Western Cape Province, preventing accurate development of policies as well as effective management and conservation schemes aimed at preserving biodiversity and human livelihoods. The aim of this study was to fill this knowledge gap by investigating conflict between co-existing stakeholders and wildlife in the Boland Region of South Africa. It has been suggested that conflict arises from a number of interwoven social, economic, and environmental factors whose effects may vary spatially as well as temporally (Graham et al. 2005; Woodroffe et al. 2006; Inskip & Zimmermann 2009). Identifying and implementing accurate mitigation strategies will thus vary between geographical regions, and depends on an accurate understanding of the underlying dynamics of the area (Thorn et al. 2013). The expertise of the largely underrepresented (Hill 2004) local people engaging in conflict scenario's on a regular basis was relied on to determine (1) the characteristics of people more likely to rely on the lethal control of DCA's, and (2) the level of tolerance shown towards DCA's by farmers in the Boland Region. A spatial analysis (3) in the form of hotspot-mapping was conducted to identify areas where exigent conservation action is required, (4) regression models were used to identify the characteristics of properties likely to experience conflict on a species-level, and (5) the zones of potential conflict risk based on a maximum entropy (MaxEnt) modelling algorithm were predicted.

3.3 Materials methods

3.3.1 Study area

The Boland Region forms part of the Western Cape Province of South Africa, and is subdivided into the Northcentral Cape Winelands and Southern Overberg regions. The area is typified by a Mediterranean-type climate and accommodates a substantial

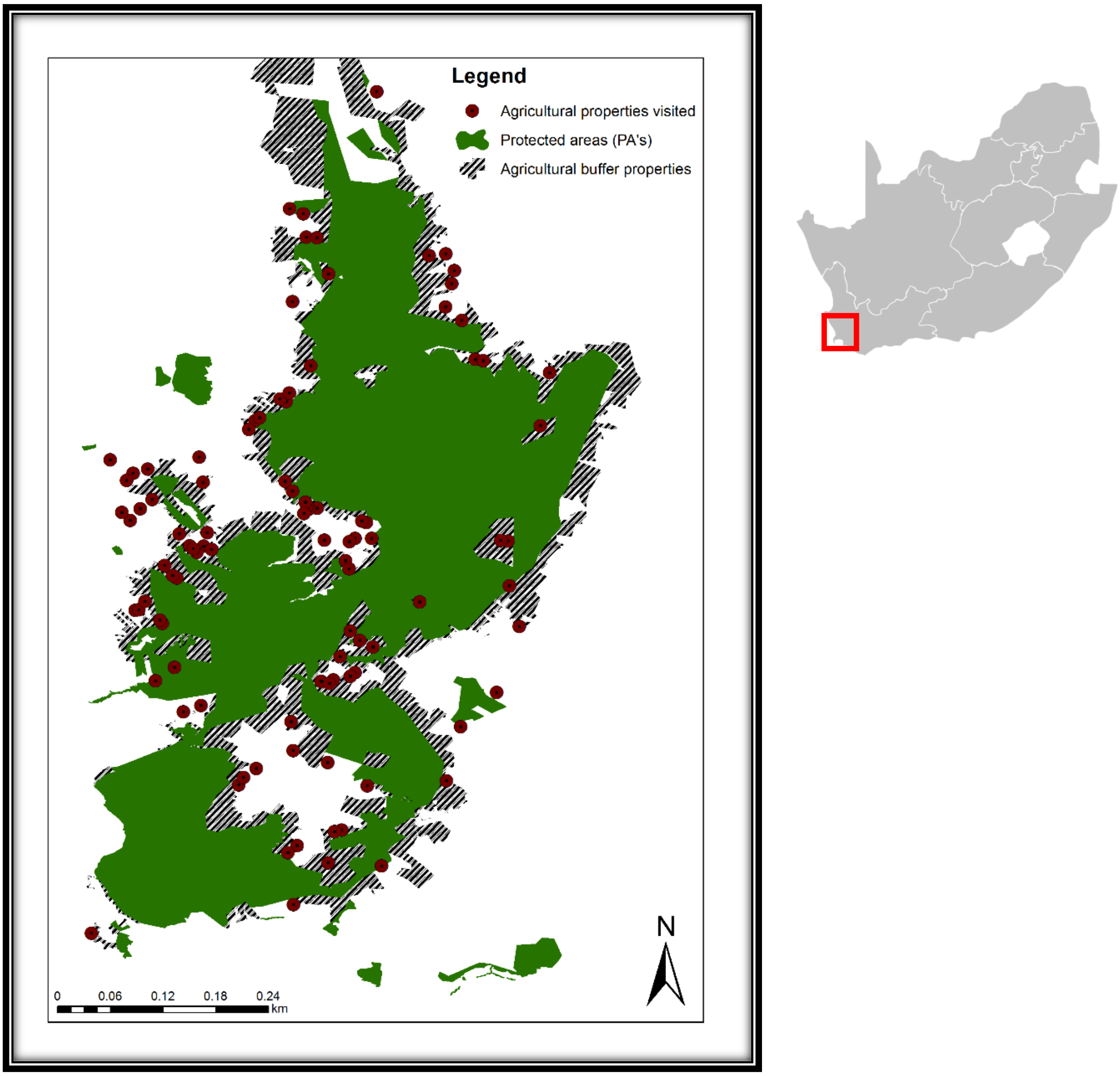


Figure 3.1. The study area. Green shaded areas denote PA's, and striped shading denote agricultural land bordering PA's. Interview locations are marked with red points. Coordinates: South-eastern extreme: 34°20′51.7″S 19°08′26.9″E; North-western extreme: 33°29′41.0″S 19°00′13.7″E.

component of the agrarian economy of the province. It further has within its core large expanses of protected and unprotected natural habitat where large numbers of terrestrial wildlife are present. Consequently, the human-wildlife interface is a prominent feature of the area, and high accounts of conflict can thus be expected – making it the perfect place to conduct this study. Furthermore, the agricultural sector of the Western Cape is a significant component of the net economy of the province, and provides employment and often also housing opportunities for approximately 135 000 people (Statistics South Africa 2012), thus accentuating the economic importance of agriculture in the region. The largest proportion of agricultural activity, both in farm units (41%) and hectares (46%) consists of horticultural activity, while the remaining major agricultural activities are relatively equally divided between field crops (9%; 14%), animal production (10%; 10%), and mixed farming (14%; 15%) (DAFF 2014). This study focused specifically on intensively and extensively farmed properties bordering protected areas (~3 500 km^2) (Fig. 3.1).

3.3.2 Data collection

To obtain information on human-wildlife conflict in the Boland Region, face-to-face structured interviews were conducted based on closed-format questionnaires with landowners or managers of agricultural properties (hereafter collectively referred to as farmers) between July 2017 and May 2018. This form of data collection was relied on due to the relatively large size of the study area, and the interdisciplinary design that included both ecological as well as sociological aspects (White et al. 2005; Becker et al. 2013). Face-to-face interviews are viewed as the preferable sampling method overall in questionnaire methodology (Arrow et al. 1993), and closed-format questions tend to result in less uncertainty than open-ended questions for both the researcher and the respondent (White et al. 2005). The targeted properties were selected *a priori* using a stratified random sample of the complete list of properties in the study area that fit the criteria. Selection criteria dictated that properties had to (1) be directly adjacent to PA's, (2) have a general manager in permanent employment at the time of the interview, and (3) actively rely on some form of crop or animal husbandry as the main source of income. Small properties (< 50 ha) directly adjacent to one another were manually excluded to avoid spatial autocorrelation between sampling sites. Ultimately, 232 properties matched the criteria, and 103 were selected.

The questionnaire (Appendix 6.2) consisted of three sections. The first section focussed on information relating to the managed property. Questions included for example farm size, the division and management of agricultural produce, the presence and maintenance of natural areas, and affiliation with conservational agencies. The second section focused on general respondent demographics, such as age, ethnicity, job title and tenure, and residential status. The last section focused on conflict with terrestrial vertebrate species. Questions included for example the species causing financial damage to the property and the extent thereof, the species hunted on the property and the motivations thereof, the methods of hunting, the use of non-lethal control methods, and the respondents' knowledge of existing hunting legislation.

Farmers were contacted telephonically to arrange an interview, and non-response bias was assumed to be minimal due to a refusal rate of < 4.7% (Lindner 2002). Participation was completely voluntary, and all accounts were kept entirely confidential and anonymous. Hence we do not present any information that can be used to identify specific properties or individuals. Each interview lasted between 30 minutes and two hours and was administered in the respondent's preferred language (either Afrikaans or English). Preceding all interviews the overall purpose of the study was thoroughly communicated to the respondents, and formal consent was obtained to allow further analysis of their accounts (Appendix 6.4). Ethical clearance was received from the Research Ethics Commission (Humanities) of Stellenbosch University (Reference: SU-HSD-004696).

3.3.3 Statistical analysis

Characteristics of human-wildlife conflict were summarised, as well as characteristics of the study participants, using standard descriptive statistics. To identify the characteristics of farmers more likely to rely on the lethal control of DCA's, a set of generalised linear regression models (GLM's) assuming a Poisson family distribution with a "*log*"-link function was fitted. A total of nine plausible variables were postulated, and the mean number of DCA's hunted annually was used as a response variable. Candidate models contained all additive combinations of predictors as well as two-way interactions. Standard information-theoretic multi-model inference techniques were used to interpret the results (Burnham & Anderson 2003; Dochtermann & Jenkins 2011), and models within each set were ranked in order of parsimony using Akaike's Information Criterion values adjusted for sample size (AICc). Models within three ΔAICc units (ΔAICc ≤ 3) of the model with the lowest AICc were regarded as having equal support (Akaike 1974; Burnham et al. 2011). The probability of each model being the most parsimonious in the candidate model set was determined by the Akaike model weight, and model statistics were obtained with an analysis of deviance for generalised linear model fits (ANOVA.GLM). Additionally, nine independent logistic regression models were applied to identify the property and environmental variables determining likelihood of HWC. Presence or absence of species-specific conflict was presented as binary data and used as predictor variables. Model validity was assessed with the Chi-squared likelihood ratio test, and the predictive power was given by the MacFadden pseudo R^2-value, where values between 0.2 – 0.4 were taken to present a good model fit (Domenich & McFadden 1975). Analyses were conducted in RStudio v3.5.0.

Tolerance levels shown towards DCA's by farmers in the study region were thought to vary substantially with regards to the species in question. To evaluate this, the tolerance to damage index (TDI) developed by Kansky et al. (2014), was applied. Using this index, it was possible to determine the proportion of farmers expressing positive attitudes towards DCA's, despite incurring damages. The formula is given as follows:

$$TDI = X - (1 - Y)$$

Where, X = the proportion of the respondents suffering damage, and Y = the proportion not resorting to lethal control methods. A resulting TDI value of 0 indicates neutrality (i.e. the proportion of respondents resorting to lethal control is relative to the proportion of respondents suffering damage). Negative TDI values indicate low tolerance, and positive TDI values indicate high tolerance.

Spatial visualization of conflict hotspots was achieved with the construction of a heat map, wherein a static "*hybrid*" map from Google Maps was overlaid with layered matrix data containing the location (property coordinates) of conflicts. The Moran's I spatial autocorrelation statistic was applied to compare the value of one location to all other locations, to assign an intensity value to each geographic area (Levine 2004).

To spatially predict the zones of conflict risk beyond the scope of the interview locations, ecological niche models (ENM's) were applied from the location and environmental characterization of the sites where species-specific conflict occurred. A total of 19 bioclimatic variables related to temperature and precipitation were applied (https://www.bioclim.org), as well as vegetative and landscape-appropriate variables (https://www.landcover.org). RStudio v3.5.0 was used to identify variables with correlations less than 0.5 (Zarco-González et al. 2012), and were retained in the models. The MaxEnt machine learning algorithm (Phillips et al. 2006, 2009) was applied to learn the mapping function and classification rules inductively from the training data (Franklin et al. 2009), therefore enabling the modelling of conflict risk for each species. Using MaxEnt, the best approximation of an unknown distribution of a species over a geographical range with maximum entropy, subject to predetermined parameters obtained from a sample of occurrence data, was used to estimate the multivariate distribution of suitable habitat conditions in environmental feature-space (Zarco-González et al. 2012). The algorithm has proven efficacy when applied to large geographic regions and with use of relatively small samples sizes (Hernandez et al. 2008), and was therefore chosen for this study. The individual performance of each model was evaluated from the value of the area under the curve (AUC) receiver operating characteristic (ROC). Models were considered poor performers when AUC values were low (0.5 – 0.7), adequate at moderate AUC values (0.7 – 0.9), and good at high AUC values (> 0.9) (Manel et al. 2001). Separate analysis for each species (total of eight) was performed in the Maximum Entropy Species Distribution Modelling software v3.4.1.

3.4 Results

3.4.1 Descriptive characteristics of the sampled community

Of the 103 properties included in the study, 47 farms (45.6%) relied on a single form of agricultural output, while the remainder (54.4%, n = 56) had several farming systems in place. The majority farmed with variations of orchards (57.3%, n = 59) and vineyards (56.3%, n = 58). To a lesser extent, farms consisted of livestock rearing (18.4%, n = 19), poultry production (10.7%, n = 11), protea (*Proteaceae*) farming (10.7%, n = 11), field crops (7.8%, n = 8), trout farming (5.8%, n = 6), apiaries (5.8%, n = 6), and game farming (4.9%, n = 5). This was relatively proportional to trends observed in the total agricultural activity of the Western Cape Province (DAFF 2014). The total area of farm surveyed was 41 706 ha (median = 158 ha, range = 5 – 4 107 ha). This represents 44.4% of total farmland (n = 232 properties) in the study area, calculated as the area within the minimum convex polygon formed from the outermost interview locations.

The respondent group had low ethnic variation, as expected from typical commercial farmland tenure patterns of South Africa (Tladi et al. 2002). We therefore assume that the homogenous study population (98% white) represents the wider population of farmers in the area (Marker et al. 2003). Most respondents belonged to the Afrikaans-speaking cultural group (63.1%), and the rest were English-speaking (36.9%). The majority of respondents were also of Western Cape origin (82.5%), while the remainder migrated either intranationally (13.7%) or internationally (3.8%) to the

Western Cape Province. Respondents were mostly male (94.2%), ranged between ages 27 – 80 years (mean = 48.9 ±SE 1.24) and had a mean tenure of 23 years (±SE 1.70). For a full description of respondent and survey location characteristics, see Appendix 2.1.

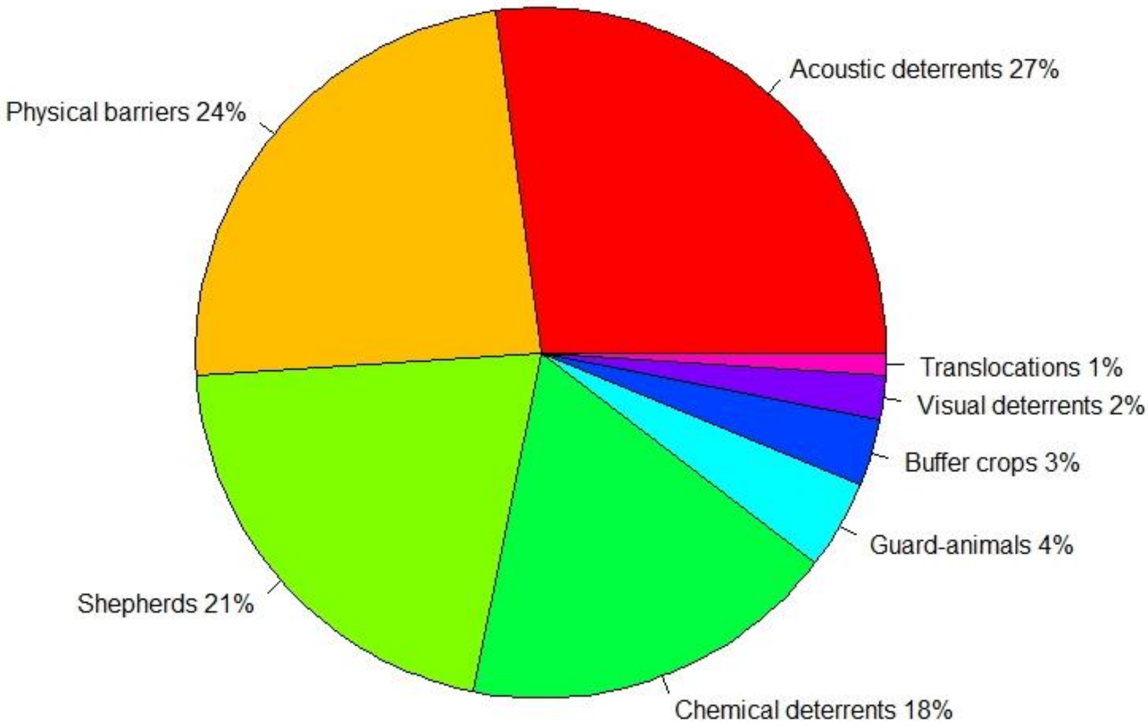


Figure 3.2. Breakdown of the most popular non-lethal damage-preventing methods employed on sampled properties in the Boland Region.

The lethal control of any DCA was reportedly practiced on 51.5% (n = 53) of the sampled properties, on all of which the preferred method of off-take was via firearms. Of these, six properties additionally used baited traps and caging methods prior to removing the DCA. No farmers reported the use of hunting dogs or poison for removing DCA's. The most extensively persecuted species were chacma baboons (*Papio ursinus*; mean = 6.3 animals/year, range 1 – 40; n = 32 respondents), feral dogs (*Canis lupus familiaris*; mean = 3.6 animals/year, range 1 – 6; n = 24 respondents), and feral pigs (*Sus scrofa*; mean = 12 animals/year, range 3 – 42; n = 14 respondents).

Non-lethal damage-preventing methods were employed on 60% (n = 62) of the surveyed properties (Fig. 3.2), the most popular methods being acoustic deterrents (27%) such as wind cannons and virtual fences, physical barriers (24%) such as electrified fencing and night bomas, human shepherds (21%) and chemical deterrents (18%) such as semiochemical substances (e.g. predator hair and faeces). Less prevalent were the use of guard-animals (4%), buffer crops (3%), visual deterrents (2%), and translocations (1%). On 21% (n = 22) of the sampled properties, no form of either lethal or non-lethal control methods were implemented.

3.4.2 Characteristics of farmers relying on lethal control methods

Lethal control methods were reportedly administered for the control of 19 species of DCA's by 53 farmers (51.5%) who shot on average 15 individuals per annum (±SE 15.7). Three of the 65 candidate models of the global additive model were within 3 Akaike units (AICc) of the top ranked model (Table 3.1). Reliance on lethal control methods was explained by the farm's conservation status and size (ha), as well as the respondents' gender, migratory status, place of residency and age ($F_{(15, 102)} = 1163.2$; $P < 0.0001$).

Table 3.1. Generalized linear models that were within 3 AICc units of the highest ranking model (lowest AICc), with associated degrees of freedom (df), the number of parameters in the model (k), AICc, ΔAICc, and Akaike model weights.

Model	**df**	**k**	**AIC$_c$**	**ΔAIC$_c$**	**Weight**
Response variable: number of individuals shot in past year					
1. CnS[1] + Gnd[2] + MgS[3] + Rsd[4] + PrS[5]	15	16	1325.1	0.00	0.462
2. CnS + MgS + Rsd + PrS	14	15	1325.8	0.63	0.338
3. Age + CnS + Gnd + MgS + Rsd + PrS	16	17	1327.9	2.81	0.114

[1]CnS = Conservancy affiliation
[2]Gnd = Respondent gender
[3]MgS = Migratory status of respondent
[4]Rsd = Respondent place of residency
[5]PrS = Property size (ha)

Local respondents ($\beta = 3.11$; SE = 1.00; z = 3.11; $P < 0.01$), as well as migrants from other provinces within South Africa ($\beta = 4.32$; SE = 1.01; z = 4.26; $P < 0.0001$), were more likely to rely on lethal control methods and shot significantly more DCA's on an annual basis than respondents who immigrated to South Africa internationally (Fig. 3.3e). Male respondents were also more likely to rely on lethal control ($\beta = -1.48$; SE = 0.20; z = 7.30; $P < 0.0001$) than women (Fig. 3.3d), as well as respondents who resided on the property on a full-time basis ($\beta = -0.59$; SE = 0.51; z = -1.16; $P < 0.05$), as opposed to living on a neighbouring farm or in a nearby town (Fig. 3.3a). The number of DCA's shot annually also increased as property size increased ($\beta < 0.004$; SE < 0.003; z = 10.51; $P < 0.0001$), with more animals being shot on larger properties (Fig. 3.3b). Farms that were affiliated with regional conservancies were more likely to rely on lethal control methods ($\beta = 0.30$; SE = 0.08; z = 3.54; $P < 0.001$) for the control of DCA's (Fig. 3.3c). The respondents' age, race, language and tenure had no significant influence on their likelihood to rely on lethal control methods.

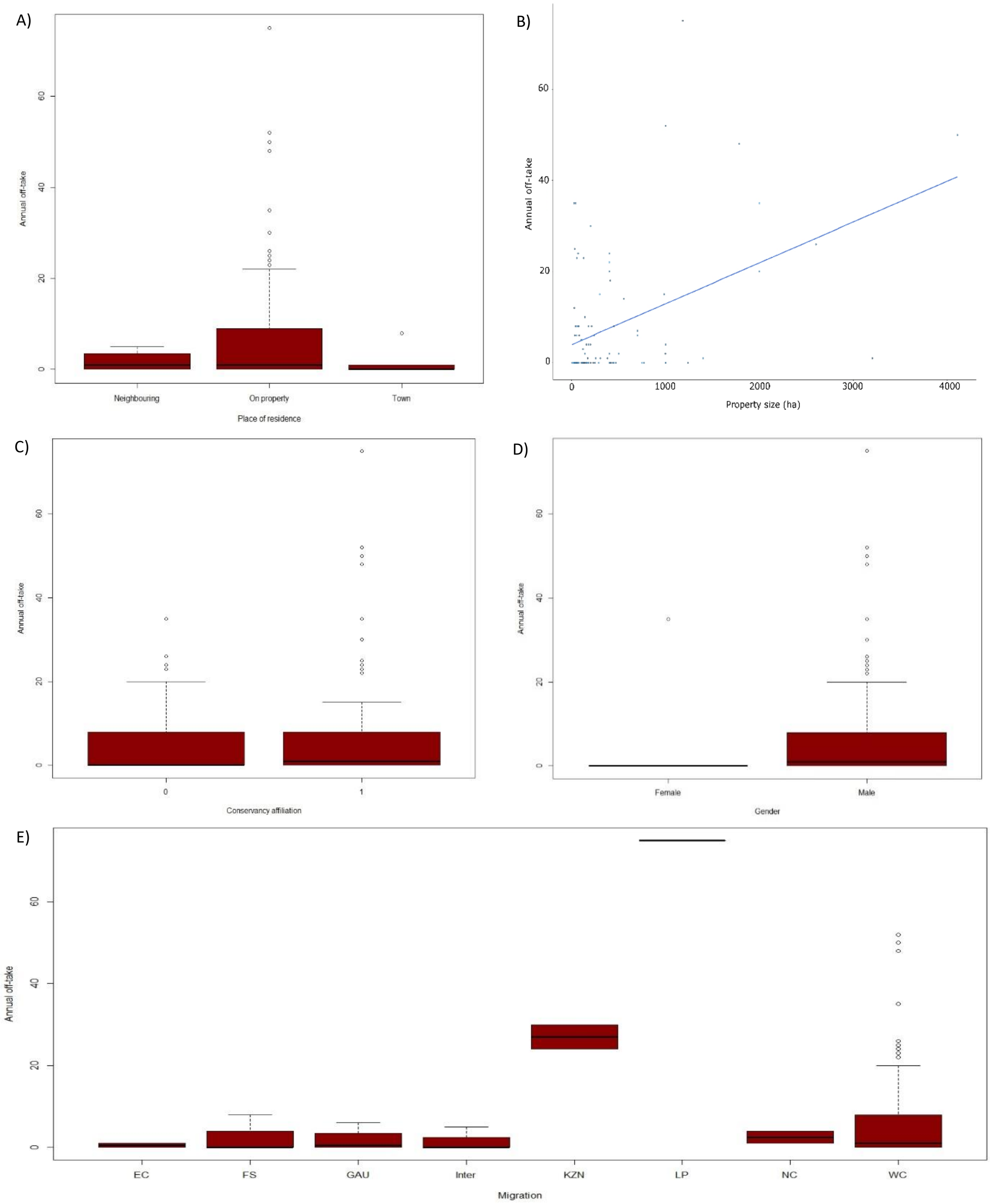


Figure 3.3. Mean annual off-take of farmers relying on lethal control methods, compared by (a) place of residency, i.e. farmers living on neighbouring properties, on the property itself, or in town; (b) property size (ha), (c) conservancy affiliation, (d) gender, and (e) place of origin (i.e. Eastern Cape Province, Free state, Gauteng, International, Kwazulu-Natal, Limpopo, Northern Cape Province, and Western Cape Province).

3.4.3 Tolerance levels

The TDI values revealed levels of tolerance exerted by farmers towards 27 DCA's in the Boland Region (Table 3.2). Respondents were most tolerant of baboons (TDI = 0.17) and honey badgers (*Mellivora capensis*; TDI = 0.08). The lowest level of tolerance was shown for domestic/feral cats (*Felis catus*; TDI = -2.00) and domestic/feral dogs (TDI = -2.50). Comparatively, tolerance of ungulates (TDI_{AVG} = 0.02) was higher than tolerance of carnivores (TDI_{AVG} = -0.23). The lowest level of tolerance was shown towards birds (TDI_{AVG} = -1.31), snakes (TDI = -2.00), and invasive, extralimital or feral species (TDI_{AVG} = -1.48).

Table 3.2. The relative tolerance exerted by farmers towards 27 DCA's in the Boland Region, quantified by the tolerance-to-damage index (TDI). X = the proportion of the respondents suffering damage, Y = the proportion not resorting to lethal control methods.

Species	Common name	X	1-Y	TDI
Papio ursinus	Chacma baboon	0.65	0.48	0.17
Mellivora capensis	Honey badger	0.08	0.00	0.08
Atilax paludinosus	Marsh mongoose	0.05	0.00	0.05
Herpestidae spp.	Mongoose spp.	0.04	0.00	0.04
Taurotragus oryx	Common eland	0.04	0.00	0.04
Sylvicapra grimmia	Grey duiker	0.26	0.22	0.04
Raphicerus melanotis	Cape grysbok	0.02	0.00	0.02
Lepus spp.	Hare spp.	0.02	0.00	0.02
Oreotragus oreotragus	Klipspringer	0.01	0.00	0.01
Pelea capreolus	Grey rhebok	0.01	0.00	0.01
Hystrix africaeaustralis	Cape porcupine	0.21	0.23	-0.01
Genetta spp.	Genet spp.	0.04	0.25	-0.21
Aonyx capensis	Cape clawless otter	0.05	0.40	-0.35
Caracal caracal	Caracal	0.08	0.50	-0.42
Panthera pardus	Leopard	0.02	0.50	-0.48
Procavia capensis	Rock hyrax	0.05	0.60	-0.55
Sciurus carolinensis[a]	Eastern grey squirrel	0.04	1.00	-0.96
Pavo cristatus[a]	Indian peafowl	0.03	1.00	-0.97
Dama dama[a]	Fallow deer	0.01	1.00	-0.99
Bostrychia hagedash[a]	Hadeda ibis	0.00	1.00	-1.00
Numida meleagris	Helmeted guineafowl	0.11	1.27	-1.17

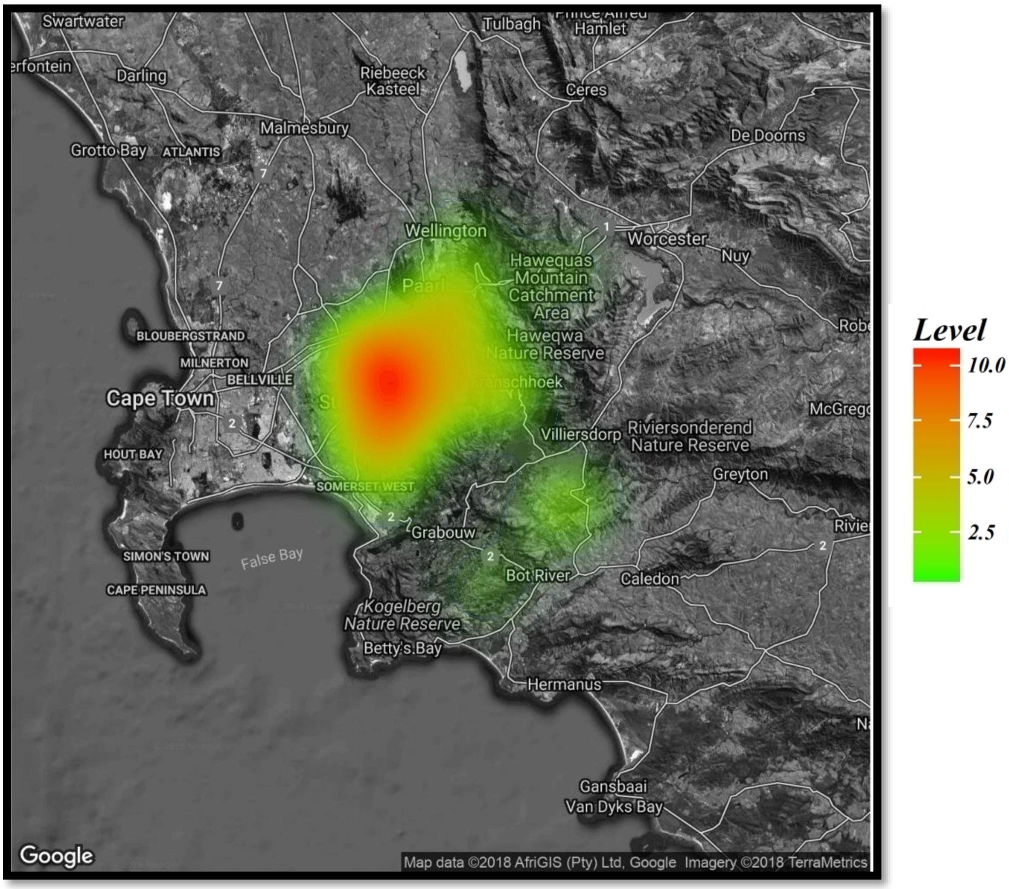
Sturnus vulgaris[a]	European starling	0.08	1.50	-1.42
Sus scrofa[a]	Feral pig	0.07	1.71	-1.65
Alopochen aegyptiaca	Egyptian goose	0.03	2.00	-1.97
Alethinophidia spp.	Snake spp.	0.00	2.00	-2.00
Felis catus	Domestic/feral cat	0.00	2.00	-2.00
Canis lupus familiaris	Domestic/feral dog	0.10	2.60	-2.50

[a]Alien invasive or extralimital species

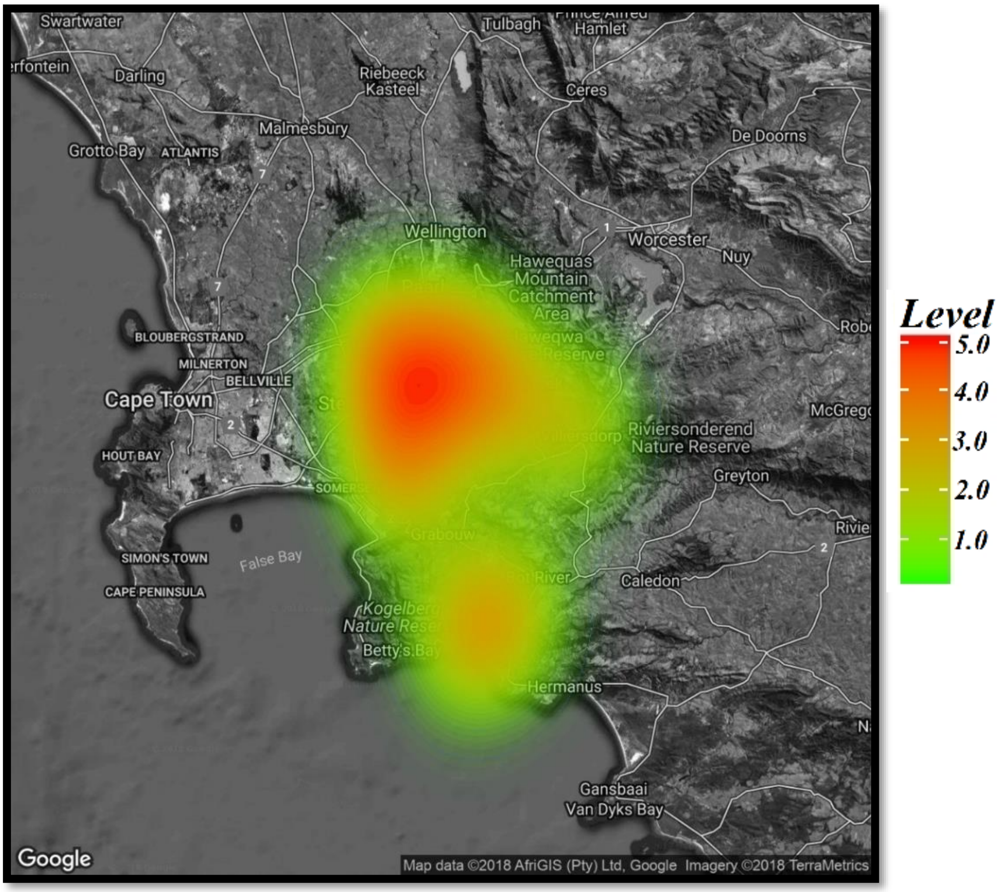
3.4.4 Conflict occurrence hotspots

Clear patterns emerged in the areas most vulnerable to species-specific human-wildlife conflict (Fig. 3.4). Conflict with Cape porcupine (*Hystrix africaeaustralis*) was most prevalent on farms between Stellenbosch, Paarl and Franschhoek, increasing closer to Stellenbosch. Conflict with honey badgers (*M. capensis*) was highly prevalent in the central and southern divisions of the study area, most notably near the towns of Paarl, Stellenbosch, Franschhoek, and in the Highlands farming area east of Kogelberg Nature reserve. Conflict with caracal (*Caracal caracal*) was mostly confined to the Northern areas, especially in the Agter-Groenberg areas near Wellington and near Paarl. Conflict with baboons (*P. ursinus*) was extremely widespread, occurring in all parts of the study area, but predominating in the Franschhoek valley, near Paarl, and in the Elgin and Vyeboom farming areas. Conflict with rock hyrax (*Procavia capensis*) was scarce, only occurring near Franschhoek and in the Slanghoek farming community near Rawsonville. Conflict with grey duikers (*Sylvicapra grimmia*) predominated on the farms surrounding Stellenbosch and in the Vyeboom and Elgin farming communities. Conflict with feral dogs (*C. familiaris*) were the most widespread form of conflict, occurring in all major study regions, especially between the towns of Paarl, Stellenbosch and Somerset-West. Conflict with common eland (*Taurotragus oryx*) were mostly confined to the plains between Villiersdorp and the Theewaterskloof dam. Conflict with genet (*Genetta* spp.) occurred mostly on the farms north of Stellenbosch and towards Franschhoek. Conflict with helmeted guineafowl (*Numida meleagris*) was widespread, but most intense in the Franschhoek, Vyeboom, Elgin and Grabouw farming communities. Conflict with Cape clawless otter (*Aonyx capensis*) was highly concentrated in the mountainous areas near Franschhoek and du Toitskloof pass. Conflict with Indian peafowl (*Pavo cristatus*) was highest near Stellenbosch, Somerset-West, Elgin, Vyeboom, and Bot River. Conflict with feral pigs (*S. scrofa*) occurred mostly on farms near Wellington, and decreased towards Paarl and Franschhoek. Conflicts with rodents (excl. porcupine) occurred in confined areas, mostly in the Franschhoek and Banhoek valleys, near Villiersdorp and near Botriver. Conflict with marsh mongoose (*Atilax paludinosus*) was concentrated in du Toitskloof, Franschhoek and Stellenbosch.

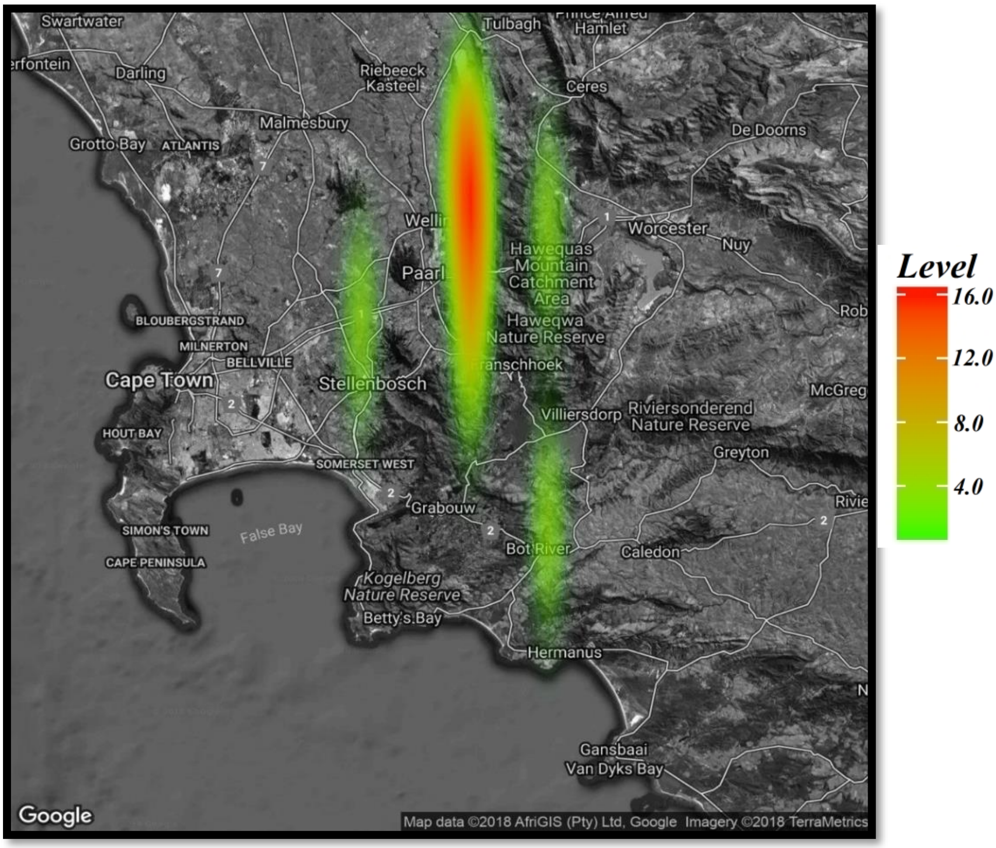
a) Cape Porcupine (*H. africaeaustralis*)



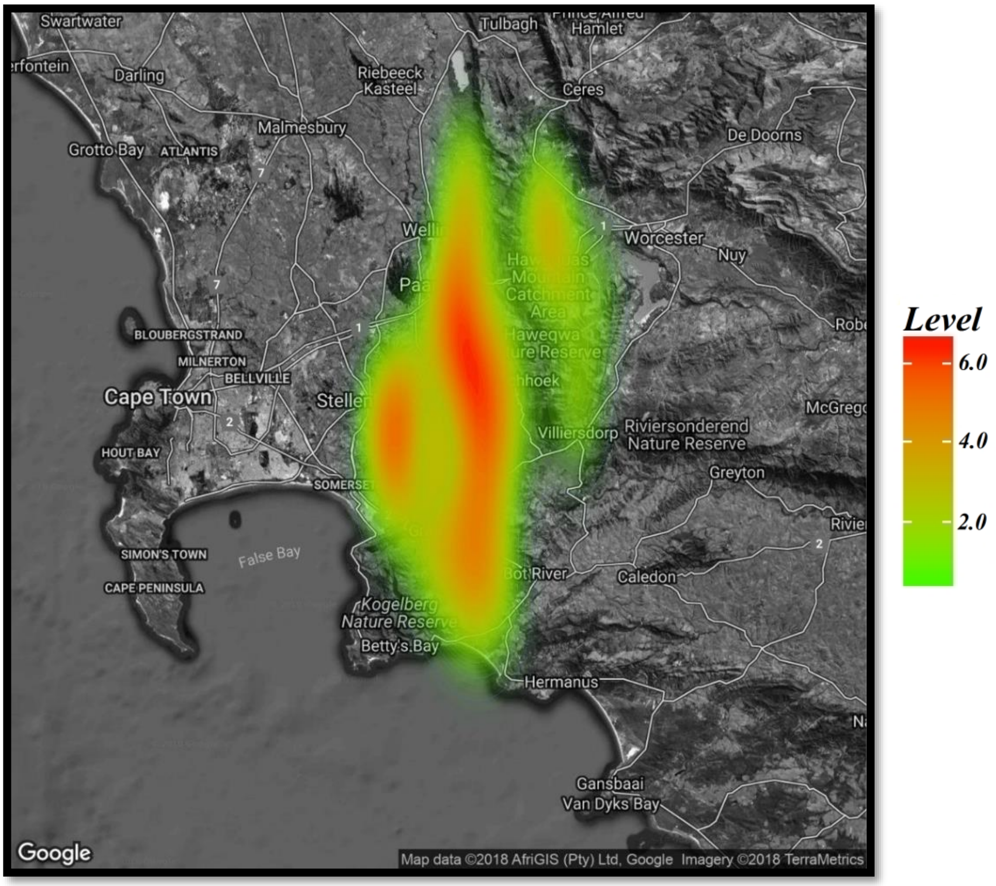
b) Honey badger (*M. capensis*)



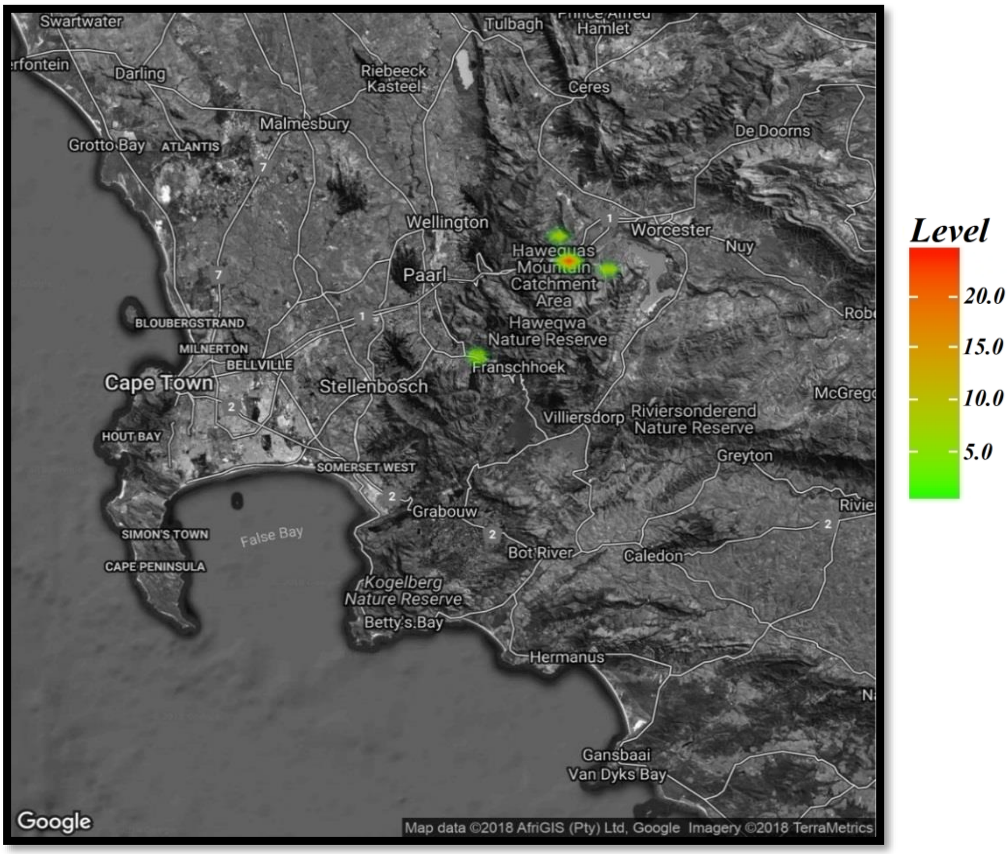
c) Caracal (*C. caracal*)



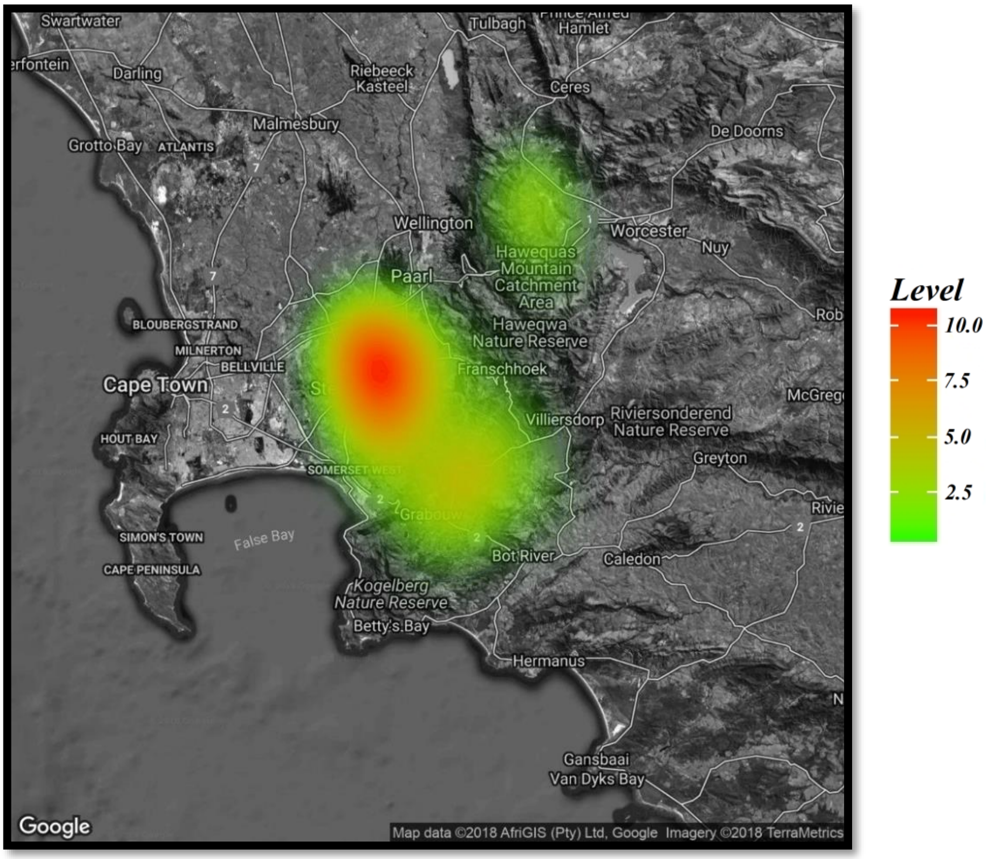
d) Chacma baboon (*P. ursinus*)



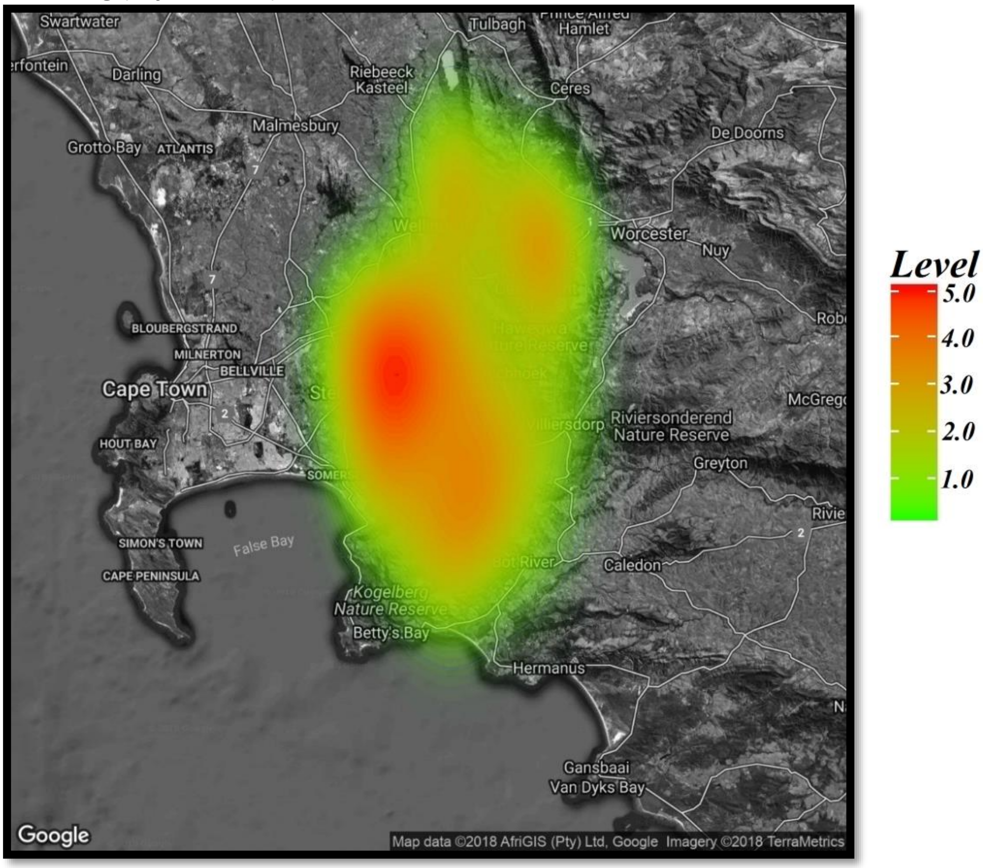
e) Rock Hyrax (*P. capensis*)



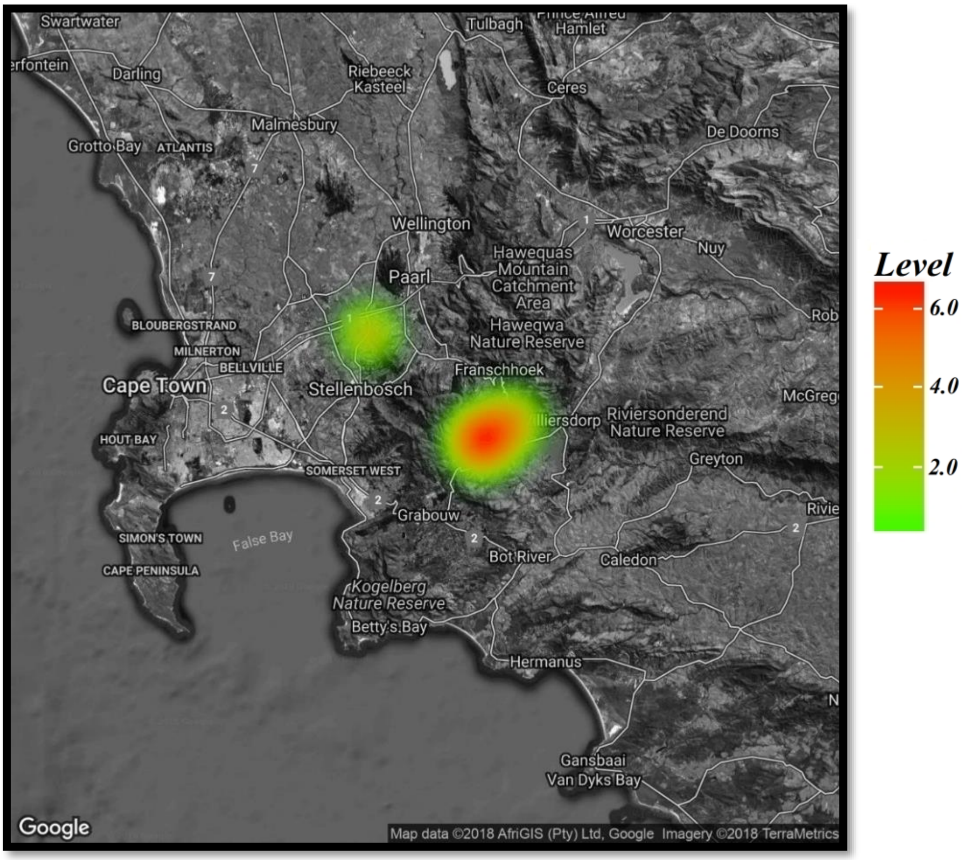
f) Grey duiker (*S. grimmia*)



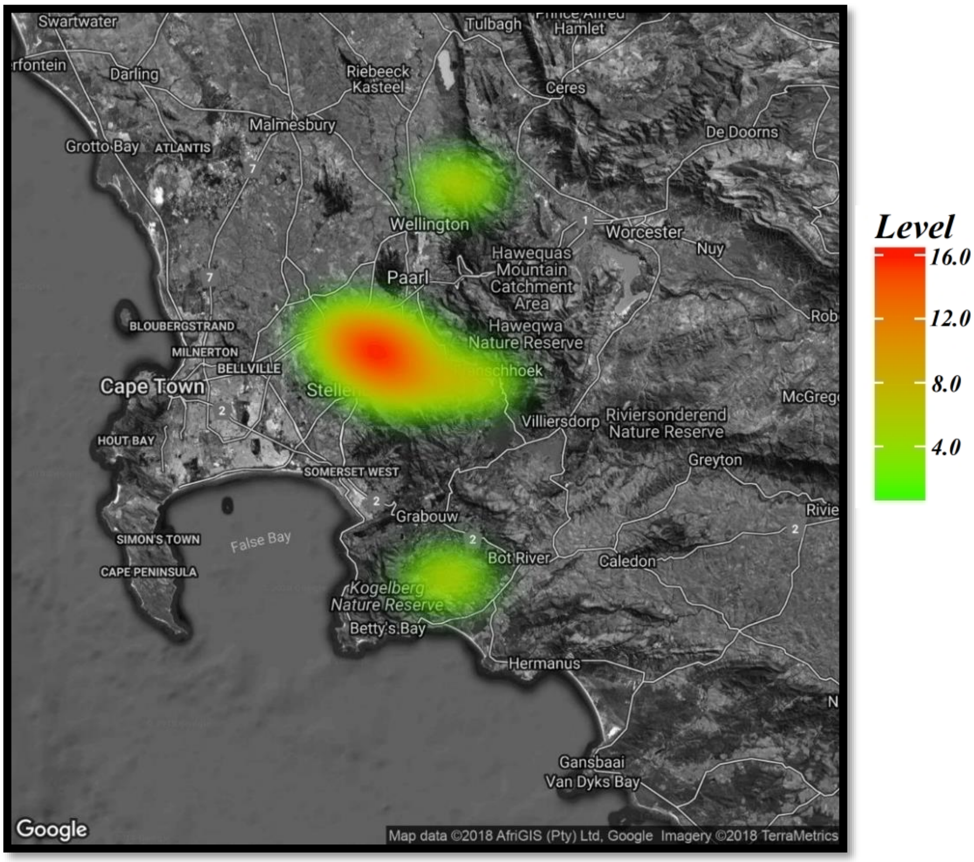
g) Feral dog (*C. familiaris*)



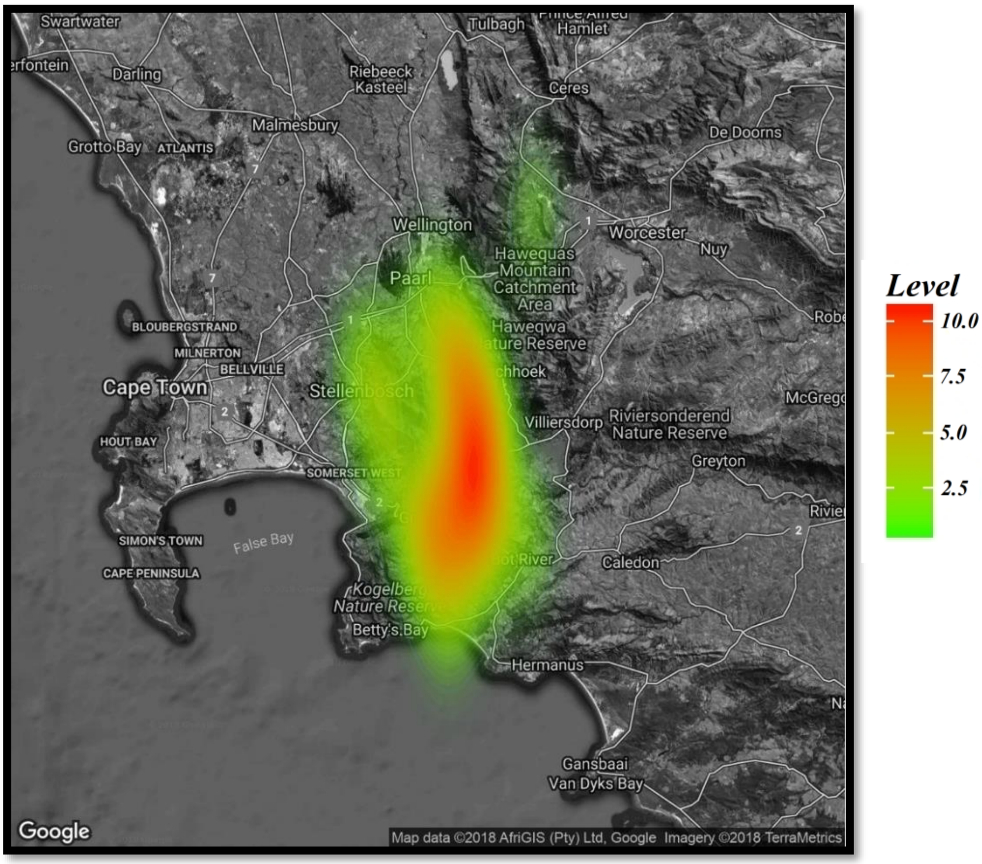
h) Common eland (*T. oryx*)



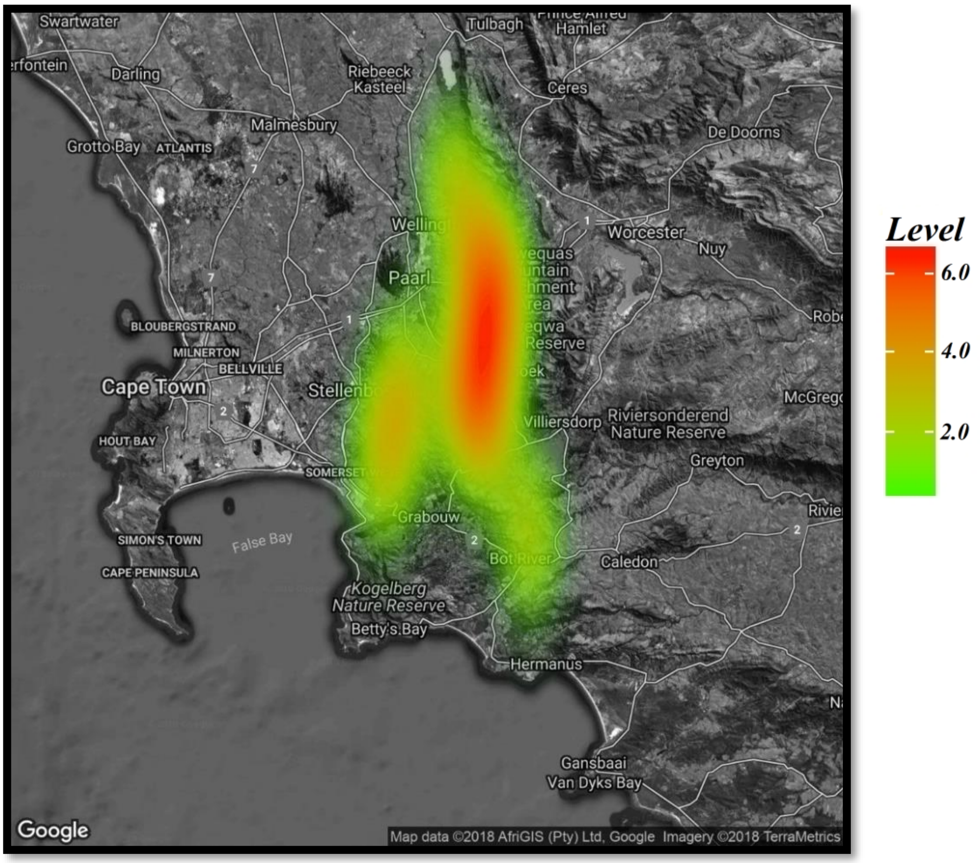
i) Genet (*Genetta* spp.)



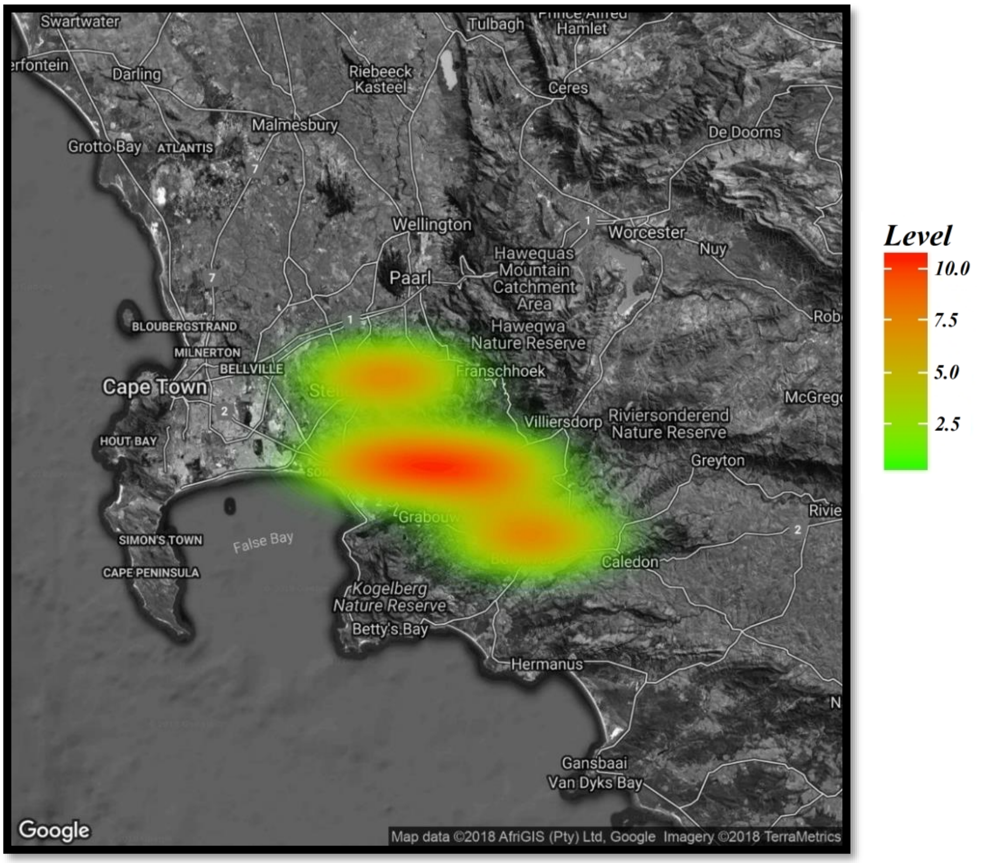
j) Helmeted guineafowl (*N. meleagris*)



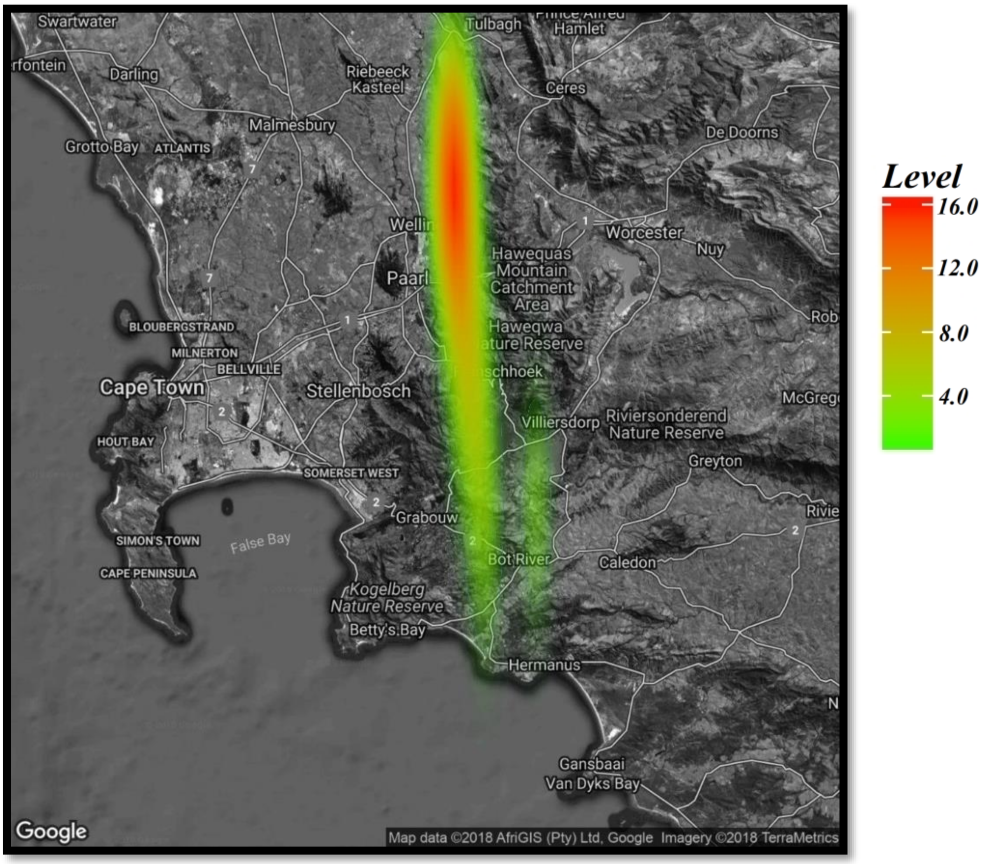
k) Cape clawless otter (*A. capensis*)



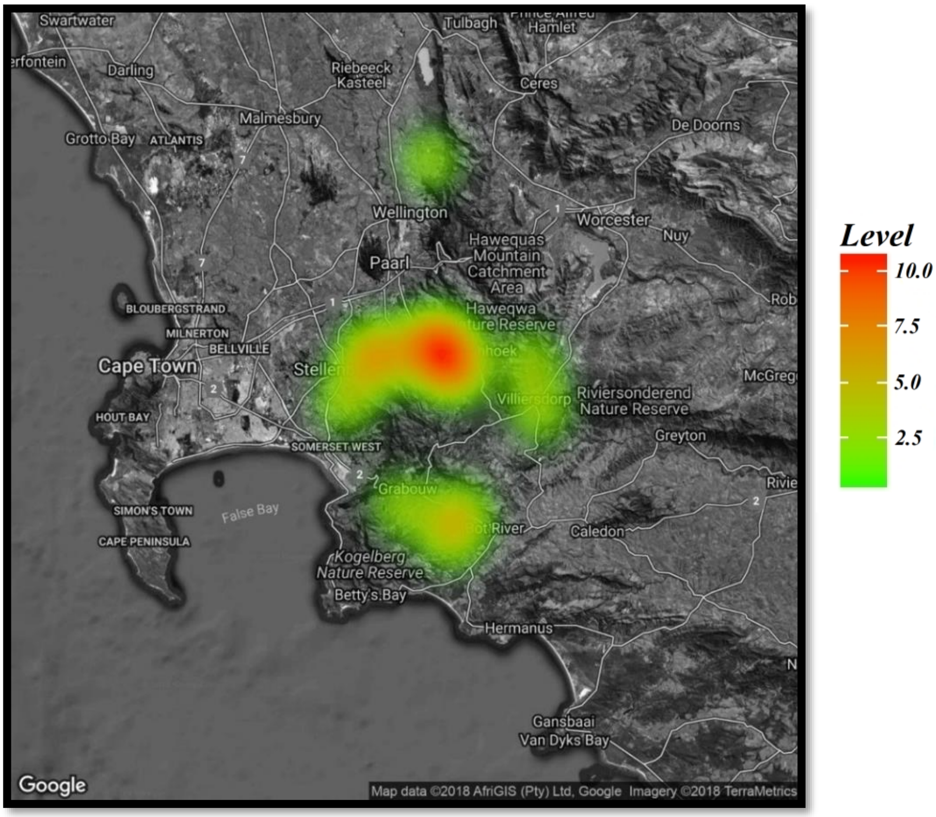
l) Indian peafowl (*P. cristatus*)



m) Feral pigs (*S. scrofa*)



n) <u>Rodents (excl. porcupine)</u>



o) <u>Marsh mongoose (*A. paludinosus*)</u>

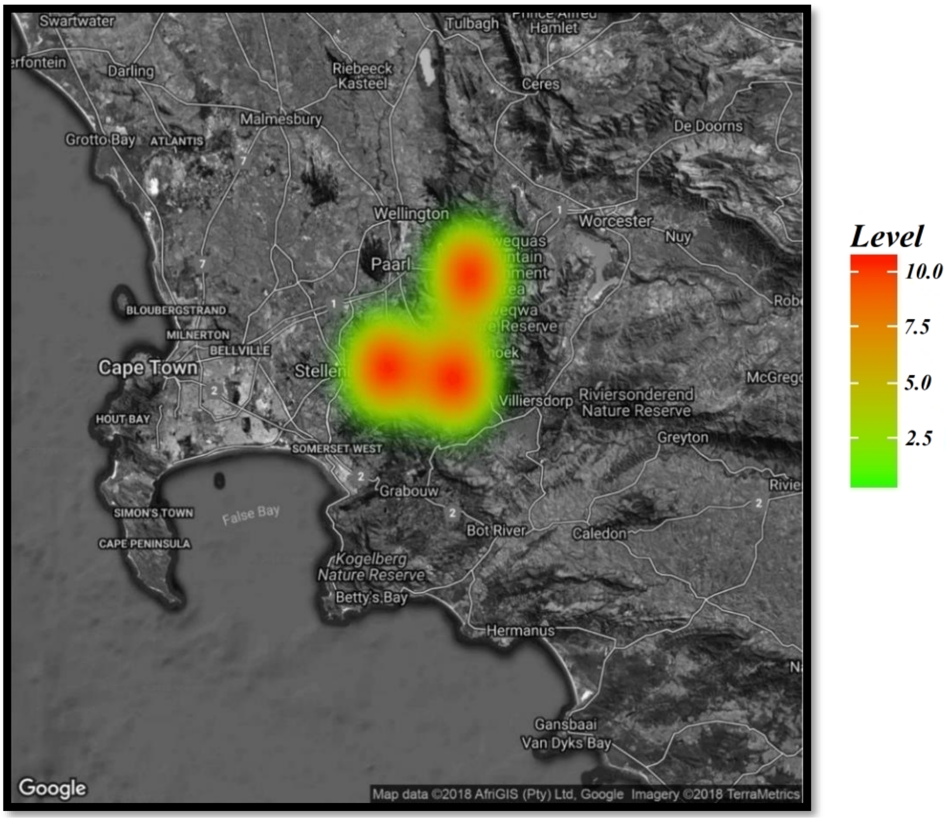


Figure 3.4. Human-wildlife conflict hotspots in the Boland Region, specified by species (a) Cape porcupine, (b) honey badger, (c) caracal, (d) chacma baboon, (e) rock hyrax, (f) grey duiker, (g), feral dogs, (h) eland, (i) genet spp., (j) helmeted guineafowl, (k) Cape clawless otter, (l) Indian peafowl, (m) feral pigs, (n) rodents spp., and (o) marsh mongoose. The intensity of the hotspot is indicated by a colour change between shades of green (least concern) and red (highest concern).

3.4.5 Characteristics of properties susceptible to species-specific conflict

The logistic regression models showed that, for each of the nine species (i.e. nine independent models), there were significant environmental and agricultural variables determining the likelihood of experiencing conflict with the species on a property (Table 3.3).

Table 3.3. Characteristics of properties susceptible to species-specific conflict, with corresponding regression coefficient (β), standard error (SE), z-ratio, and significance level (*P*-value).

Species	Variable	β	SE	z-ratio	*P*-value	Model χ^2	Pseudo R^2
Feral dog	Vineyards	1.53	1.64	-2.04	< 0.05		
	Livestock	1.19	0.80	0.03	< 0.05		
	Poultry	1.47	0.87	0.03	< 0.05		
	Natural land	0.02	0.12	0.05	< 0.05		
	Roadways	< 0.01	< 0.01	0.05	< 0.05	14.0	
	Housing areas	< 0.01	< 0.01	0.03	< 0.05	$P < 0.05$	0.27
Baboon	Vineyards	1.01	0.58	1.78	< 0.05		
	Orchards	0.33	0.51	0.64	< 0.05		
	Field crops	1.44	0.77	1.86	< 0.05		
	Proteas	0.16	0.91	0.18	< 0.05	23.1	
	Season: Summer	0.38	0.48	0.66	< 0.05	$P < 0.05$	0.22
Badger						19.5	
	Apiaries	22.47	0.75	0.01	< 0.05	$P < 0.05$	0.26
Duiker	Vineyards	1.86	0.77	2.41	< 0.001		
	Roadways	1.89	0.08	2.17	< 0.01		
	Housing areas	-3.39	0.01	-2.21	< 0.05	20.2	
	Season: Spring	1.11	0.16	1.88	< 0.05	$P < 0.05$	0.21
Caracal	Game	3.73	1.48	2.52	< 0.01		
	Poultry	3.88	1.12	3.46	< 0.0001		
	Roadways	< 0.01	< 0.01	2.33	< 0.05		
	Elevation	< 0.01	< 0.01	0.86	< 0.05	15.9	
	Season: Summer	1.16	0.48	1.76	< 0.05	$P < 0.05$	0.44
Porcupine	Orchards	0.59	0.57	1.10	< 0.05		
	Field crop	0.82	0.83	0.98	< 0.05		
	Livestock	-1.12	0.83	-1.35	< 0.05	10.3	
	Housing areas	-0.02	< 0.01	-2.57	< 0.01	$P < 0.05$	0.21

Feral pig	Water bodies	0.03	0.02	1.91	< 0.05	20.8	
	Game	3.46	1.04	3.32	< 0.0001	P < 0.05	0.31
Otter	Trout	3.38	1.02	3.12	< 0.0001		
	Water bodies	1.14	0.69	1.91	< 0.01		
	Rivers	1.50	1.34	2.99	< 0.05	11.1	
	Elevation	< 0.01	< 0.01	1.77	< 0.05	*P* < 0.05	0.22
Genet						12.9	
	Poultry	0.56	0.32	1.01	< 0.05	*P* < 0.05	0.28

Conflict with feral dogs was significantly predicted by seven variables (χ^2 = 14.0; P < 0.05), and was found to be highest on properties with vineyards, livestock and poultry farming, as well as properties with greater proportions of natural vegetation, properties close to major roadways, and properties close to residential areas. The likelihood of baboon conflict was significantly determined (χ^2 = 23.1; P < 0.05) by the presence of vineyards, orchards, field crops and protea farming, with conflict increasing where these agricultural productions occurred. Baboon conflicts also increased in summer months. Conflicts with honey badgers were significantly higher (χ^2 = 19.5; P < 0.05) on properties with apiaries present, while conflicts with grey duiker were significantly higher (χ^2 = 20.2; P < 0.05) in the months of spring, with the presence of vineyards, closer to roadways, and further away from residential areas. Conflict with caracal was highest during summer months, on properties with game and poultry farming, as well as properties that were closer to major roadways, and properties at higher elevation (χ^2 = 15.9; P < 0.05). Cape porcupine-conflict was highest on properties with orchard, field crop and livestock farming, as well as properties further away from residential areas (χ^2 = 10.3; P < 0.05). Conflict with feral pigs was greatest on game farms, and properties with high proportions of permanent water bodies (χ^2 = 20.8; P < 0.05), while genet-conflict was intensified on poultry farms (χ^2 = 12.9; P < 0.05). Conflicts with Cape clawless otter (χ^2 = 11.1; P < 0.05) was highest on trout farms, as well as properties with a high proportion of permanent water bodies, properties with permanent rivers, and properties at high elevations.

3.4.6 Predicted risk zones

The highest AUC values were recorded for porcupine (AUC = 0.991), grey duiker (AUC = 0.990) and feral dogs (AUC = 0.988), while the lowest values were recorded for caracal (AUC = 0.819) and Cape clawless otter (AUC = 0.832). Nonetheless, all AUC values (Table 3.4) were high enough to conclude that models were adequate for analysis (> 0.7).

Table 3.4. AUC values of the MaxEnt model, shown for all species.

Species	AUC
Cape porcupine	0.991
Grey duiker	0.990
Domestic/feral dogs	0.988
Chacma baboon	0.960
Honey badger	0.890
Cape clawless otter	0.832
Feral pigs	0.829
Caracal	0.819

For grey duiker, the predicted zones of highest conflict risk were on farms surrounding Paardenberg Nature Reserve, Paarl Mountain Local Nature Reserve, the Northern side of Stellenbosch towards Simonsberg Nature reserve, the north-eastern side of Groenlandberg Nature Reserve, and along the old du Toitskloof pass in the Hawequas Mountain Catchment area (Fig. 3.5a).

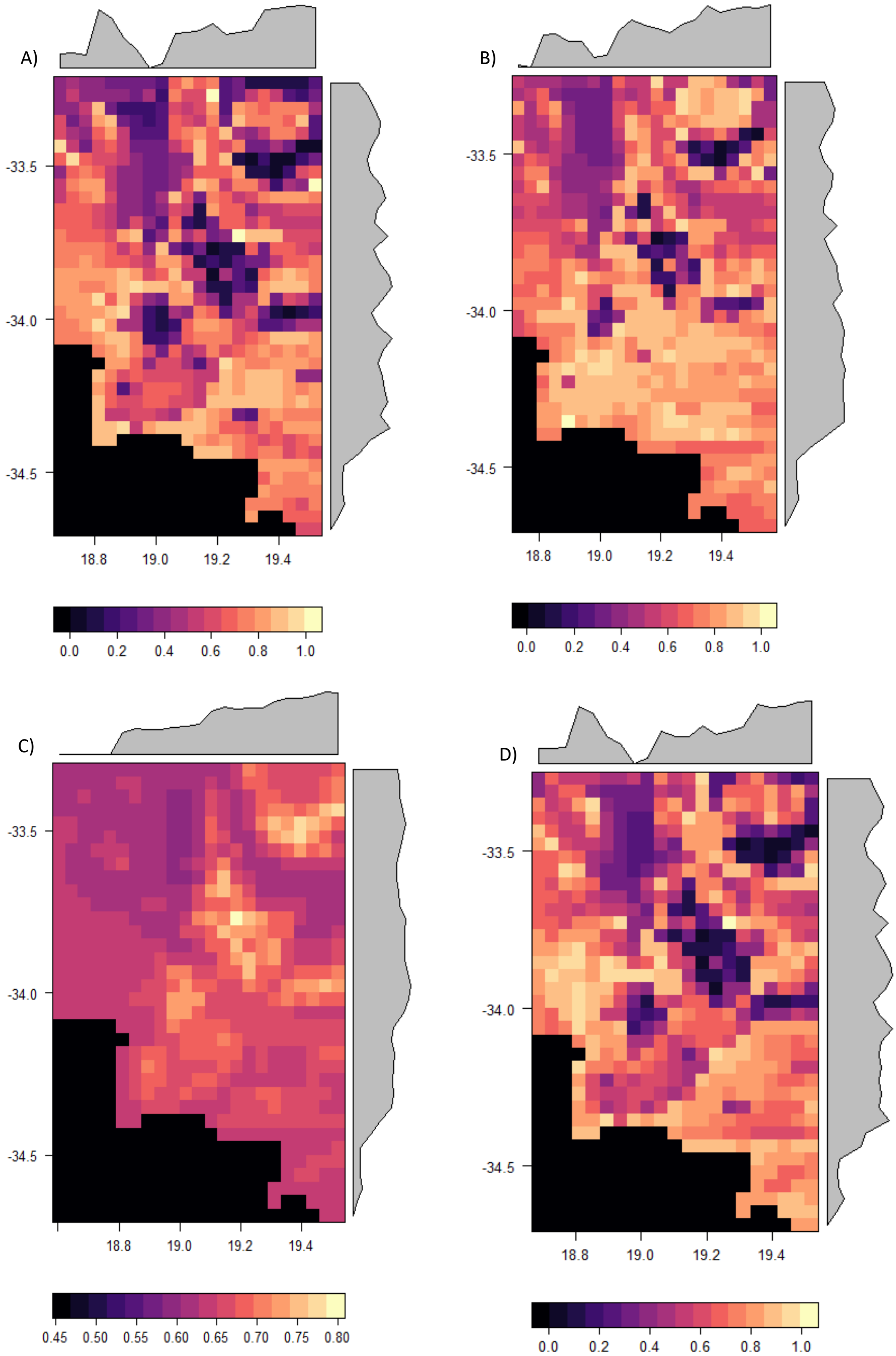
For chacma baboon, the predicted zones of highest conflict risk was widespread, encapsulating most of the greater Boland Region. Conflict risk was however particularly high on farms South of the Matroosberg mountain range near Worcester, in the Slanghoek and Franschhoek farming valleys, near the Jonkershoek Nature Reserve, in the EGVV farming community (Elgin, Grabouw, Vyeboom and Villiersdorp), near Bot River southwards towards Hermanus, and along the coastline at Gansbaai, Kleinmond, Betty's Bay, Pringle Bay and Rooi-els (Fig. 3.5b).

For feral pigs, the predicted zones were less widespread, concentrating around Bain's Kloof pass near Wellington and Southwards along the Hawequas mountain range towards Franschhoek. To a lesser extent feral pig conflict was also expected to spread to Fonteintjiesberg Nature Reserve, to Riversonderend Nature Reserve, and around the Berg River dam to Jonkershoek Nature Reserve and possibly the Banhoek Conservancy (Fig. 3.5c).

For Cape porcupine, the predicted zones of conflict risk were relatively widespread, intensifying on farms along the western slopes of Tierkloof, Witzenberg, Wittebrug, and Fonteintjiesberg Nature Reserves, near the Brandvlei Nature Reserve at Rawsonville, and on the farms surrounding Paarl, Franschhoek, Banhoek, Stellenbosch and Blaauwklippen. Porcupine conflict was also predicted for the Overberg regions, such as the Kogelberg Sonchem Link Nature Reserve, the Highlands farming community, and near Houwhoek Nature Reserve (Fig. 3.5d).

Predicted zones of conflict regarding Cape clawless otter were mainly confined to the Hawequas Mountain range, especially near the old du Toitskloof Pass and the Hawequas Nature Reserve. To a lesser extent conflict was also expected at the Fonteintjiesberg, Riviersonderend, and Theewaters Nature Reserves (Fig. 3.5e).

For feral dogs, the predicted zones of highest conflict risk were widespread, but intensified near human settlements. Most notably on the western side of the greater Boland Region, approaching the Cape metropole, i.e. farmed areas near Brackenfell, Kuils River, and Khayelitsha. The coastal towns of Strand, Gordon's Bay, Betty's Bay, Kleinmond and Hermanus is also predicted to experience elevated conflict risk. Inland hotspot areas include farms surrounding Franschhoek, Villiersdorp, the Elandskloof farming valley, Rawsonville, Worcester, and Bot River (Fig 3.5f).


A)
-33.5
-34.0
-34.5
18.8 19.0 19.2 19.4
0.0 0.2 0.4 0.6 0.8 1.0
B)
-33.5
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-34.5
18.8 19.0 19.2 19.4
0.0 0.2 0.4 0.6 0.8 1.0
C)
-33.5
-34.0
-34.5
18.8 19.0 19.2 19.4
0.45 0.50 0.55 0.60 0.65 0.70 0.75 0.80
D)
-33.5
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-34.5
18.8 19.0 19.2 19.4
0.0 0.2 0.4 0.6 0.8 1.0

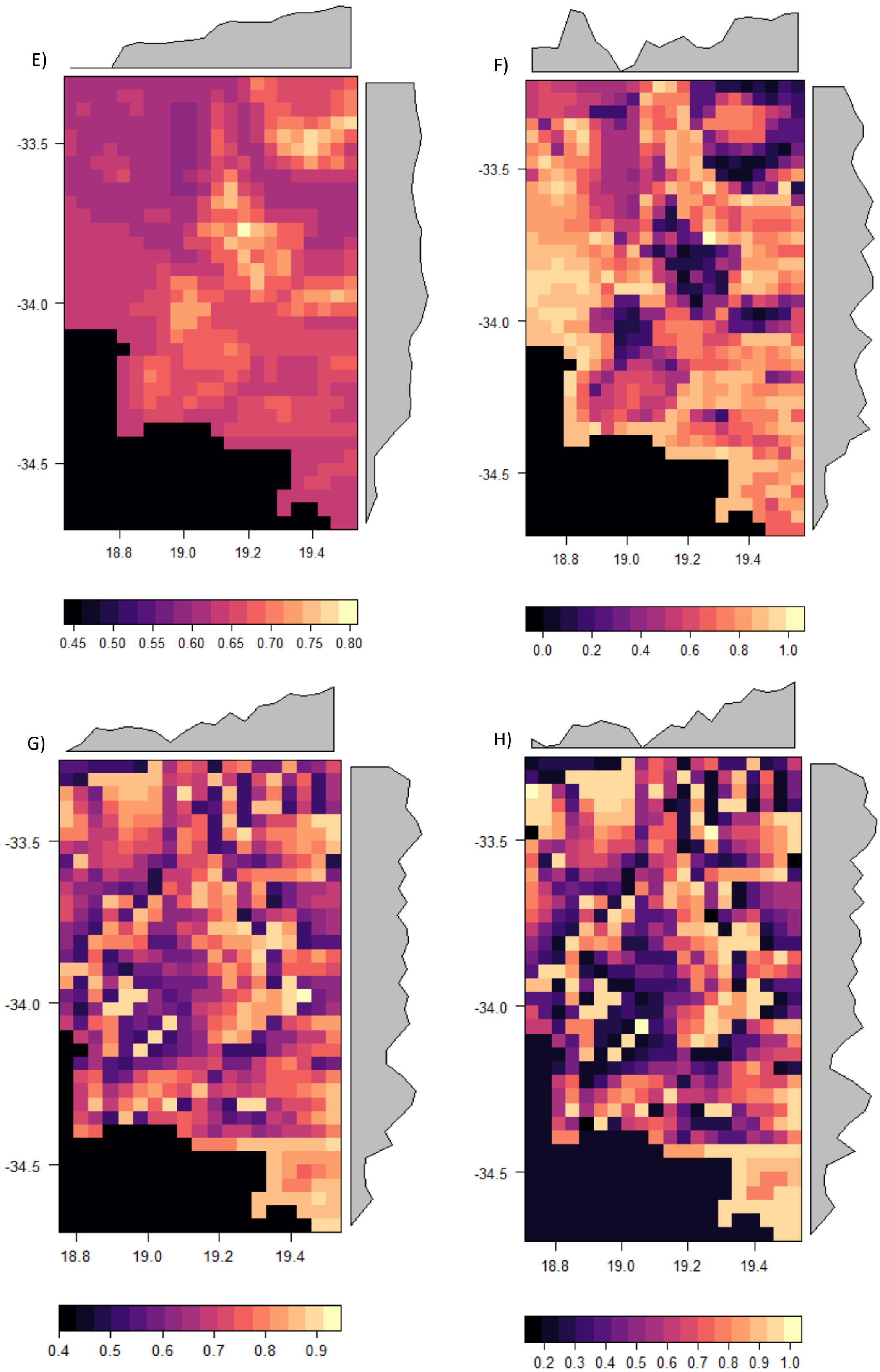

Figure 3.5. Maximum entropy (MaxEnt) model plots of predicted human-wildlife conflict risk in the greater Boland Region, for species (a) grey duiker, (b) chacma baboon, (c) feral pig, (d) Cape porcupine, (e) Cape clawless otter, (f) domestic/feral dog, (g) caracal, and (h) honey badger. Conflict intensities varied between predicted hotspots (white) and coldspots (purple).

Caracal conflict was predicted to intensify on farms near Blaauwklippen, the Jonkershoek, Groenberg, and Riviersonderend Nature Reserves, Villiersdorp, and in the Elandskloof farming valley. Caracal conflict risk was also expected to be high in the Grabouw valley between the Hottentots-Holland and Kogelberg Nature Reserves, at Kleinmond, and near Fernkloof Nature Reserve (Fig. 3.5g).

Predicted zones of honey badger risk was extremely widespread but confined to small, isolated areas. The highest risk was predicted for areas surrounding the Paardenberg, Mount Hebron and Kogelberg Nature Reserves, Blaauwklippen, Wedderwil, the Fairy Glen Private Nature Reserve, and the Slanghoek farming community. Badger conflict was also expected on farms near Klapmuts, Bot River, Hermanus, Gansbaai, Villiersdorp and Genadendal (Fig. 3.5h).

3.5 Discussion

Human-wildlife conflict in unprotected areas, particularly those bordering PA's (Stein et al. 2010), has resulted in reduced home ranges and population sizes for naturally occurring wildlife (Woodroffe & Ginsberg 2000). Simultaneously, a very tangible threat is posed to the livelihoods and agricultural security of people living close to PA's (Dublin & Hoare 2004; Hill 2004; Graham et al. 2005; Barua et al. 2013). In this study, the spatial location of conflict in the Boland Region aimed to provide a means for ameliorating conflict by developing preventative actions fine-tuned for specific areas, as well as to optimize conservation resources (Zarco-González et al. 2012). The characteristics and attitudes of farmers towards DCA's were expected to vary significantly, and thus influence the likelihood of an individual relying on the lethal control of DCA's (Schumann et al. 2012). The location of current conflict zones was highlighted, characteristics of properties and wildlife and people involved in HWC were identified, and the zones of future conflict risk beyond the sampled boundaries were spatially predicted.

3.5.1 Human-carnivore conflict

In this study, HCC was not as prevalent as conflict with crop-raiding species, opposing general research trends and focus (Woodroffe & Ginsberg 2000; Inskip & Zimmermann 2009). The cost associated with HCC was also relatively low, with certain obvious exceptions regarding caracal and feral dogs preying on valuable game animals. Even low predation of these animals can be extremely expensive, and is therefore likely one of the main drivers of the low overall tolerance shown by farmers towards carnivores, along with traditional attitudes (Macdonald et al. 2010).

Human-caracal conflict predominantly occurred on farms that included game or poultry farming systems, and it is therefore unsurprising that the majority of these conflicts were reported in the Northern region of the study area, particularly in the Agter-Groenberg farming community, where the majority of game farms were concentrated. The extremely low tolerance shown by farmers towards caracals (-0.42) was likely due to the expensive nature of game animals and even small livestock (Woodroffe & Frank 2005). For example, one respondent reportedly lost approximately R250 000 worth of springbok (*Antidorcas marsupialis*) in the past four years, while another reportedly introduced springbok to his property as a buffer between caracals and more expensive game animals, resulting in the deaths of 115 of the 129 introduced individuals. The majority of conflicts with caracal were however extremely rare, with most respondents reporting the loss of a single livestock specimen per decade. The models further showed caracal conflict was predicted to be high near the Jonkershoek- and Riviersonderend Nature Reserves, and in the Elandskloof farming valley, potentially due to the increase in caracal conflict observed at higher elevations during the study, as in previous studies on carnivores (Thorn et al. 2013; Constant et al. 2015). Caracal conflict also increased closer to major roadways, likely due to the high prevalence of roadkill and associated scavenging birds, and therefore potentially explains the high predicted incidence in Villiersdorp, the

Grabouw valley, Kleinmond and at Fernkloof Nature Reserve. Also, conflict with caracals peaked during summer months, potentially due to the limited cover available for this ambush predator during the dry season (Schiess-Meier et al. 2007), making the confined farmed animals an easier prey item. A peak in livestock and game births also extends from September to January (Constant et al. 2015), further explaining the peak in conflicts during summer months.

Conflicts with free-roaming domestic or feral dogs were the most widespread form of HCC in the study area, and were often reported to elicit extreme damage to livestock and natural wildlife. For example, wild dog hunting packs of up to 18 individuals were often reported, and were cited to have killed dairy cows, grey duiker, cape clawless otters, and entire dens of bat-eared foxes (*Otocyon megalotis*). Consequently, significant hostility towards any unidentifiable dog was shown by the majority of the respondents, as reflected by the tolerance index (-2.50), which was the lowest for any species in the study. It is assumed that the majority of feral dogs originated from human settlements, and it was therefore unsurprising that the majority of conflict with dogs was observed and predicted to be concentrated on farms near cities or towns, informal settlements and roadways. Conflicts with dogs also increased on farms with livestock or poultry production, and it was assumed that these items make up the bulk of their agricultural diet. The lack of dog-conflict on game farms is potentially explained by the high levels of interspecific competition in these areas, the overall low tolerance shown by game farmers, or the inability of previously-domesticated dogs to prey on large ungulates. The detrimental consequences of free-roaming dogs in the Cape Province were already acknowledged in the early 1900's, when farmers attributed an increase in stock losses to the large number of dogs that had been abandoned and turned wild, and warned that "*if there are more dogs than at present, they would prove as great a pest as the jackals themselves… sooner or later the dogs would not be properly fed, and they would take to killing the stock*" (Van Sittert 1998). More than a 100 years later, it is evident from the data that feral dog conflict remains a form of conflict that poses a great concern for animal husbandry and pastoralists in general, as well as for the safety of farm labourers and families. The importance of future research efforts on the topic needs to be emphasised to better understand the damage caused and the sources of the problem, as well as the cooperation of conservation and welfare agencies to curb the incidence of growing feral dog populations.

Conflicts with Cape clawless otters were commonly cited occurrences on fish farms (Lanszki & Molnar 2003; Freitas et al. 2007), and similarly this study found that otter conflict was confined to the Hawequas mountain range (from the old du Toitskloof pass to the Franschhoek valley) where all trout farms included in the study were located. Conflict with otters was also highest near permanent water bodies, rivers and at high elevations, which are all explained by their propensity to prey on commercial trout, as well as wild prey such as crabs, frogs and fish (Carnaby 2008). Low levels of tolerance were shown towards otters by farmers (-0.35). Marsh mongoose-conflict followed similar patterns to conflicts with otters, but received a much higher level of tolerance from farmers (0.05), perhaps due to its smaller body size (3.5 kg compared to 19 kg) (Stuart & Stuart 2015).

Genet conflict mostly occurred on poultry farms, and genets were especially prone to preying on chickens and geese in small-scale, subsistence pens that are generally not well enclosed. Nonetheless, low levels of tolerance were shown towards genets (-0.21) where they came into conflict with people, particularly in the Stellenbosch and Franschhoek regions. Conflicts with Cape grey mongoose (*Galerella pulverulenta*) and Egyptian mongoose (*Herpestes ichneumon*) over poultry were cited less frequently, and were also tolerated more (0.04).

Leopard (*Panthera pardus*) predation and resulting conflict is a commonly cited occurrence across Southern Africa (Schiess-Meier et al. 2007; Thorn et al. 2013; Boast 2014; Constant et al. 2015), the rest of Africa (Mizutani 1999; Inskip & Zimmermann

2009) and Asia (Wang & Macdonald 2006; Sangay & Vernes 2008; Dar et al. 2009). However, conflict with leopards was only reported on two properties in this study, and were thus not included in any analyses. Further research is needed to determine whether the low rate of leopard predation is due to underreporting for fear of prosecution, low leopard densities, or successful non-lethal mitigation measures.

3.5.2 Conflict with crop-raiding species

Rates of conflict with crop-raiding species were comparatively much higher in the study than that of HCC. There was also a higher overall tolerance level towards crop-raiding species than was shown for carnivores, despite financial losses from crop-raiding often far exceeding that of carnivore predation.

Despite baboons being the most highly persecuted species in this study, they received the highest level of tolerance from farmers (0.17). Only 36 farms (35%) included in this study never experienced noticeable damage to crops or infrastructure due to baboons, and the majority experienced extensive damages. For example, one respondent calculated annual loss of wine grapes to baboons (February 2017 to February 2018) to be roughly R150 000. Apart from destroying crops, baboons were also often cited to destroy infrastructure such as thatch roofs, fences, irrigation lines and livestock feeding troughs, as well as prey on poultry such as chickens and their eggs. Attitudes shown towards baboons were however not always negative. For example, one respondent reportedly tolerated the damage caused by baboons because of their tendency to clear orchards of fallen, unharvested fruit, thereby reducing the incidence of crop pests such as codling moth (*Cydia pomonella*). Human-baboon conflict was most severe in the Franschhoek farming valley, where extremely large troop sizes were reported by respondents. Conflicts with baboons however occurred in every area of the study, with the exception of farms on the slopes of Simonsberg Mountain. Previous studies have identified the human-baboon conflict problem in the Cape peninsula (Kansky & Gaynor 2000; Hoffman & O'Riain 2012; Fehlmann et al. 2017) where they often cause damage to urban properties, and extensive research and conservation efforts are currently being devoted to ameliorate conflict incidences in these areas (Hoffman & O'Riain 2012; Kaplan & O'Riain 2015). However, conflict mitigation is needed in the Cape Winelands and Overberg regions to protect the livelihoods of the societies co-existing with baboons in the area. Conflicts with baboons were most severe on properties consisting of all types of vineyards, orchards and field crops, as well as protea farms. An increase in baboon conflict was also observed during dry summer months, likely due to decreased food and water availability in their natural home ranges and the peak availability of crops during this time. Future conflicts can be expected in every major district of the Boland, but mainly in the Franschhoek valley, in the EGVV area, and in coastal towns such as Rooi-els, Pringle Bay, Betty's Bay and Kleinmond.

The incidence of conflict with Cape porcupines noticeably increased with distance away from cities, and occurred mostly on properties with large expanses of orchards and field crops. On these properties, porcupines are mostly found to unearth tuber vegetables such as onions and sweet potatoes, as well as ringbark fruit trees, eventually resulting in the death of the tree (Jacobs & Biggs 2002). Porcupines were however mostly reported to damage polyvinyl chloride (PVC) pipes that transfer water to crops. This thus further explains why human-porcupine conflict was intense on livestock farms as well. Tolerance shown towards porcupines by farmers was negative but proportional to the extent of damage incurred (-0.01). Human-porcupine conflict was most severe on the farms between Stellenbosch, Paarl and Franschhoek.

Conflicts with grey duiker were mostly concentrated around Stellenbosch and in the Elgin and Vyeboom farming communities. Duikers occurred in high densities in vineyards, and to a lesser extent in orchards across the Boland Region, where they often indulge on the young sprouting vines. This thus results in a peak in human-duiker conflict during the months of spring, when new vines start to form. The damage caused by duikers is however largely described as '*minimal*' and '*sufferable*', and many farmers reportedly enjoy having them around. As a result, we recorded a high level of tolerance shown towards duikers (0.04). Duiker conflict was further found to increase with distance away from residential areas, and closer to roadways. The former is explained by their skittish nature (Stuart & Stuart 2015), while the latter trend is potentially caused by the presumed increase in food availability near roadways due to water run-off from the compacted road surface. To a lesser extent, klipspringer (*Oreotragus oreotragus*) and grey rhebok (*Pelea capreolus*) were also reported to elicit damage to vineyards.

Conflicts with honey badgers were only reported on properties with apiaries, and mixed reports were received on the attitudes shown towards them; presumably due to the variations in the reliance on apiaries as a source of income among farmers. The damage caused to beehives were all reported to be high, but many practice beekeeping solely as a hobby, and were therefore not severely bothered by the occasional destruction of a beehive. This, coupled with the elusive and nocturnal nature of the honey badger, resulted in the second highest tolerance value in the study (0.08). Nonetheless, conflict with badgers was widespread, and therefore raises concerns for the species' continued survival. Higher levels of conflicts were observed on the farms near Paarl, Stellenbosch and Franschhoek, and in the Highlands farming community. The models similarly predicted risk zones to be widespread, but confined to small, isolated areas.

Feral pigs, or European boars, are alien invasive species in many parts of the world (Hone 2002) that were similarly introduced into south-western South Africa where they escaped captivity and eventually spread far beyond their intended boundaries (Lowe et al. 2000; Kelt 2004; Skead et al. 2011). They were subsequently included in the selection of the 100 worst invasive species globally (Lowe et al. 2000). Feral pigs are largely unwanted on agricultural properties due to their detrimental ability to till large areas of soil in search for roots, stems and macroinvertebrates (Kotanen 1995), as well as their negative impact on species diversity (Hone 2002), croplands (Caley 1993; Schley & Roper 2003), and their association with disease transmission to people (Briones et al. 2000) and livestock (de la Fuente et al. 2004). Additionally, we found feral pigs to cause extreme damage to fences and irrigation pipes in the Boland Region. Therefore their eradication is a top priority for many farmers, as reflected in the extremely low level of tolerance shown towards these animals (-1.65). In one instance, a group of farmers reportedly shot roughly 130 pigs in the year preceding the interview. Conflict with feral pigs was mostly found on game farms and near permanent water bodies, and confined to areas near Wellington, Franschhoek, du Toitskloof, and Paarl. The predictive risk models however showed that conflict could be expected to increase in the Hawequas mountain range from Groenberg Nature Reserve to Franschhoek valley. The range of conflict with feral pigs was also predicted to spread to other areas such as to Fonteintjiesberg Nature Reserve, to Riversonderend Nature Reserve, and around the Berg River dam to Jonkershoek Nature Reserve and possibly the Banhoek Conservancy. Feral pigs distribution is aided through illegal stocking by hunters (Spencer & Hampton 2005) as well as the expansion of agriculture (O'Brien 1987). Therefore, conservation actions should be dually focused on preventing the further spread of these invasive animals while eradication initiatives are being implemented (Morrison et al. 2007).

Conflicts with rock hyrax and free-roaming eland were less common, and confined to small areas. Hyraxes tend to chew on vines or feed on low-hanging fruit, and experienced conflicts in only a few select areas of the Franschhoek and Slanghoek farming valleys. The level of tolerance shown towards them was however very low (-0.55), while eland were far more tolerated (0.04). This is however understandable given the size and implied ownership of eland. Conflict with eland mainly originated from their propensity to run through wire fences and push over fruit trees, and was confined to the plains between Villiersdorp and the Theewaterskloof dam.

Among birds, helmeted guineafowl and Indian peafowl were the greatest cause of conflict with farmers, and were rarely tolerated (-1.17 and -0.97, respectively). The slightly higher tolerance for peafowl is potentially explained by their attractive appearance, which was often cited to attract tourists to wine farms. Both species occurred in great quantities in the Vyeboom and Elgin farming communities, while peafowl was additionally abundant in the Stellenbosch and Bot River surrounds, and conflict with guineafowl also peaked in Franschhoek and Grabouw. Reasons cited for the removal of these birds, as well as for Egyptian geese (*Alopochen aegyptiaca*), includes their noisiness and the destruction of crops such as oats and grapes. Guineafowl were however cited by one respondent to control pest insects in vineyards, and were therefore tolerated.

Conflict with various rodent species (excl. porcupine) were also highly cited, as these animals tend to disrupt soils and ringbark fruit trees. Likewise, the invasive Eastern grey squirrel (*Sciurus carolinensis*) was often cited to destroy electrical wires, resulting in replacement costs of up to R20 000.

3.5.3 Mitigation

Among others, successful mitigation of human-wildlife conflict will require a better understanding of the people involved in conflicts (Treves et al. 2006; Anthony et al. 2010). The characteristics of farmers most likely to rely on the lethal control of DCA's were therefore isolated. Firstly, this study determined that South African citizens were more likely to rely on lethal control methods as opposed to immigrants from other countries. It might therefore be useful to consider and incorporate paradigm-shifting strategies employed successfully in other countries to promote co-existence with wildlife in South Africa. However, many immigrants to South Africa are presumably more financially able to tolerate revenue loss to wildlife, and therefore this result should be interpreted with caution. The results also showed that managers of farms affiliated with regional conservancies were more inclined to lethal control methods. This may simply be as a result of higher conflict occurrence in areas with high densities of co-occurring wildlife and thus areas where conservancies are predominantly founded. It does however nonetheless raise some concern about the efficacy to which these conservancies are managed and the principles portrayed to local farmers. It was further found that managers of large commercial properties were more likely to rely on lethal control methods, despite HWC generally impacting small subsistence farmers to a greater extent (Treves & Karanth 2003; Namgail et al. 2007; Baker et al. 2008; Dar et al. 2009). Finally, male farmers were more inclined to rely on lethal control methods compared to their female counterparts. Contrasting to previous studies, no significant influence of age or ethnicity of farmers and their reliance on lethal control methods was found (Thorn et al. 2012).

Despite lethal control methods still being the norm on many agricultural properties in the Boland Region, a change in the attitudes of many farmers away from the traditional pest eradication outlook (Macdonald et al. 2010) towards a more environmentally-friendly, holistic approach to farming was observed. Landowners are increasingly favouring non-lethal control methods that deter rather than kill DCA's, and a great number of such measures were being implemented in the study area (Fig. 3.2). The vast range and lack of coherence in non-lethal control measures however also indicates a lack of trusted methods of proven efficacy, and it is therefore important that research be implemented to find new non-lethal control methods, as well as to improve the existing methods, to further sway landowners away from lethal control methods. The majority of respondents who employ non-lethal control methods rely on acoustic deterrents aimed at eliciting neophobia in wildlife (Mason 1998), which included predominantly propane exploders (wind cannons) and pyrotechnics (fire-crackers and live ammunition). It was however reported that mammalian wildlife, especially baboons, quickly habituate, and acoustic deterrents were therefore not effective beyond provoking fear in birds. Physical barriers (variations of fencing and kraaling) were also popular, and were cited as effective for reducing conflict with many species, especially feral dogs and small antelope (Mason 1998). However, many caveats are associated with the erection of fences, such as the high labour and monetary cost of electrical fences (Shelton 1984; Angst 2001), the repurposing of fence wire by poachers for wire-snares (van Rooyen et al. 2016), the restriction of wildlife movement (Thouless & Sakwa 1995), and the reported propensity of small antelope, particularly grey duikers, to run into these fences and fatally injure themselves. Some farmers (< 20%) additionally employed up to six shepherds or '*chasers*', specifically for keeping baboons at bay. While this, alongside guard-dogs, was broadly cited as "*the only effective solution for reducing conflict with baboons*", the added labour costs are not feasible for most small-scale and even commercial farmers (Barua et al. 2013). The use of guard-animals as shepherds was particularly sparse (n = 4), compared to other regions of South Africa (Thorn et al. 2012). This form of control should be promoted as it is widely endorsed with shown efficacy (Ogada et al. 2003; Marker et al. 2005; Gehring et al. 2010; McManus et al. 2015; Potgieter et al. 2016), regardless of a few potential disadvantages, such as off-take of non-target species and dogs acting as reservoir hosts for diseases (e.g. rabies) that may spread to wildlife (Haydon et al. 2002; Ogada et al. 2003; Potgieter et al. 2016). Chemical deterrents designed to repel DCA's by irritating their sensory organs (Norman et al. 1992) or by mimicking specific semiochemical signals (Mason 1998) were also popular among respondents, but the specific methods employed varied substantially between farmers. The majority however consisted of semiochemical deterrents such as dog hair, human hair, and lion faeces, obtained from dog groomers, barbers, and lion parks, respectively. Chilli products (e.g. '*hot sauce*') containing capsaicin from chilli peppers (e.g. *Capsicum annuum*) were also applied to the leaves, stems or vines of crops to deter particularly antelope species. The planting of buffer crops was also cited as an effective control method for porcupine, and stockpiles of old, fallen fruit were cited as effective in reducing crop losses to baboons.

3.6 Conclusion

The observed and expected zones of conflict risk produced in this study offer visual guides to conservation agencies, policy makers and stakeholders (Rambaldi et al. 2006; Brown & Raymond 2007) to prioritize areas for intervention, accurately allocate financial and human resources (Mishra 1997), and improve land-use planning. Due to the success in using ENM's in the realm of HWC, the continual gathering of information and updating of the maps to track changes in conflict hotspots using long-term data is suggested (Miller et al. 2015). Future studies could also benefit from developing ensemble models based on multiple algorithms to identify areas of consistent conflict (Anderson et al. 2003). Additionally, the predictions of future conflict distribution in this study is largely predicated on environmental variables, and future predictions of this nature may therefore benefit from including socio-ecological, economic, and managerial layers as determinants of conflict hotspots. A further need exists to quantitatively assess the severity of the conflicts. Environmental or socio-economic variables determining conflict may differ regionally, and these results should therefore not readily be extrapolated to other regions in South Africa or elsewhere. For all species, tolerance levels may also be biased towards less abundant species, nocturnal species, or species that are more evasive and thus more difficult to hunt. The TDI values presented in this document should thus be interpreted with caution. The results from this study may also be somewhat subject to bias due to the reliance on reported information. Respondents may have, for example, deliberately or unintentionally inflated losses (Rasmussen 1999), or understated their use of lethal control methods in fear of persecution or criticism (St John et al. 2012). Nonetheless, the approximation of variables revealed here provides essential contextual data that are not available elsewhere, and can be used to provide insights on HWC in untried regions. These results can minimize conflicts, optimize agricultural yield, and reduce wildlife off-take if coupled with adequate intervention actions.

3.7 Reference list

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Chapter 4
The Use of Animals and Animal-Derived Constituents in African Traditional Medicine and Other Cultural Applications: Townships in the Western Cape Province

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4.1 Abstract

The use of animals and animal-derived materials in traditional medicine constitutes an important part of the belief-systems of indigenous African cultures, and is believed to be rapidly expanding in South Africa, where traditional healers are estimated to outnumber western doctors by 2000:1 in some areas, with an overall clientele consisting of 60 – 80% of South African citizens. Despite concerns about the impact of the trade in traditional medicine on biodiversity there has been only limited research on this topic in South Africa. Traditional healers operating from impoverished, rural communities in the Western Cape Province were consulted to provide a comprehensive inventory of the number and frequency of animals used and sold. Species richness estimators, diversity indices, and relative cultural importance (RCI) indices were used to highlight species of concern and assess market dynamics. A total of 26 broad use-categories for 12 types of animal parts or products from 71 species or morphospecies were recorded. The most commonly sold items were skin pieces, oil or fat, and bones. Results showed that leopard, chacma baboon, Cape porcupine, monitor lizard species, puff adder, African rock python and black-backed jackal were the species most used in the traditional medicinal trade. This study thus extends existing knowledge on the trade of animals in South African healing practices, and provides the first attempt in the Western Cape to quantify wildlife use for cultural traditions. The results have relevance for setting conservation priorities and may assist

in effective policy development inclusive of ecological sustainability priorities, as well as cultural demands.

Keywords: Ethnozoology, ethnopharmacology, informal settlements, South Africa, species accumulation curves, Xhosa medicine, zootherapy

4.2 Introduction

Zootherapy, the treating of human ailments through the use of animal-derived constituents (Costa-Neto 1999), has existed in traditional folk pharmacopoeias throughout history (Lev 2003; Betlu 2013), and remains an integral component in traditional medicinal practices and other cultural applications in contemporary landscapes (Cocks & Dold 2000; Whiting et al. 2011; Williams & Whiting 2016). Likewise in South Africa, the trade of, and dependence on, natural resources as traditional medicine amongst primarily indigenous African cultures is deemed to be pervasive (Williams et al. 2007; Shackleton 2009; Whiting et al. 2011).

African cultures and associated traditional healers in South Africa subscribe to a resolute belief that health and welfare issues are intimately connected with supernatural forces, social relationships and ancestral relationships (Berglund 1976; Bye & Dutton 1991; Simelane 1996). Consequently, traditional healers are highly esteemed members of the community (Hutchings 1989) whose consultation are often preferred to those of Western doctors. Furthermore, the relatively few per capita Western doctors available in South Africa (Williams et al. 2007) have resulted in a large proportion of the country's population being more dependent on traditional medicine (Cunningham & Zondi 1991; Mander et al. 2007). Estimations by several authors (Cunningham 1991; Mander 1998; Philander 2011) suggest that between 60% and 80% of South African citizens have at some point either purchased traditional medicine or consulted with a traditional healer (Mander et al. 2007). This is particularly relevant to communities existing in poor, rural areas (> 50% of the South African population – Statistics South Africa 2015) where little opportunity exists to consult with university-educated doctors, while traditional healers in comparison are far more accessible (Bye & Dutton 1991).

Traditional healers in South Africa can be broadly divided into two categories, namely *diviners* and *herbalists* (classification and terminology provided during interviews). Diviners (known throughout African cultures by the Zulu-word *isangoma* and the Xhosa-word *amgqirha*) employ a supernatural approach to make diagnoses and treat ailments, while herbalists (referred to as either *izinyanga* (Z) or *amaxwhele* (X)) dispenses traditional medicines (*umuthi*) made from natural substances derived from plant, animal, and mineral materials (Ngubane 1977). Diviners are often also trained as herbalists, and can practice both vocations either simultaneously or separately (Hutchings 1989; Krige 2009). In many instances no distinction is made between the two vocations, as in both instances a '*divine calling*' is received (Mtshali 2004). In all instances however, a strong emphasis is placed on animal-derived constituents, either incorporated alongside herbal remedies or used separately (Cocks & Dold 2000).

In the Western Cape, the use of animals in traditional medicine or cultural practice is largely dominated by Xhosa-speaking people (Petersen et al. 2014), who constitute the largest proportion of African ethnic groups in the province (24.7%, with Sotho-speaking people as their closest rival at 1.1% - Statistics South Africa 2011). Animal-use in Xhosa traditional medicine (*amazeya esiXhosa*) was documented as early as the 1930's (Cawston 1933), but the practice is certainly much older, since Xhosa communities had no contact with Western doctors and associated medical procedure prior to the 19th century (Simon & Lamia 1991). Similar to other indigenous African cultures in South Africa, Xhosa and Sotho healing practices place an equal or greater value on the use of animal constituents and derivatives for the curing of non-medical ailments, such as protection against bad luck and witches (Cunningham & Zondi 1991;

Anyinam 1995). Other '*symbolic magical*' or '*magico-medical*' purposes include the protection against physical and spiritual enemies and entities, love charms and aphrodisiacs, increased intelligence, acquiring wealth and prosperity, and aiding pastoral enterprises (Simelane 1996; Cocks & Dold 2000; White et al. 2004; Mander et al. 2007).

Despite its importance to indigenous communities in South Africa being widely acknowledged (Williams & Whiting 2016), ethnozoological research has been largely subjected to paucity, especially compared to ethnobotanical research (Herbert et al. 2003; Whiting et al. 2011; Betlu 2013). The lack of ethnozoological studies in South Africa is likely due to its small claim on the greater *Materia Medica* of indigenous cultures (Betlu 2013), as well as the popular association of ethnozoology with '*spiritual*' or '*magical*' components (Cocks & Dold 2000; White et al. 2004; Mander et al. 2007) and the Doctrine of Signatures (Lev 2002), withdrawing credibility from zootherapeutics as a realistic scientific pursuit (Williams & Whiting 2016). Despite research on ethnozoology in South Africa however being largely sporadic and subject to neglect (Betlu 2013), there has been an recent upsurge in available information during the past few decades originating from Kwazulu-Natal (Cunningham & Zondi 1991; McKean 1995; Derwent & Mander 1997; Ngwenya 2001), the Faraday market in Johannesburg (Whiting et al. 2011; Williams & Whiting 2016) and the Eastern Cape Province (Simelane 1996; Simelane & Kerley 1998) – subsequently greatly improving our understanding of the topic. A noticeable gap however still remains in the Western Cape amid the extensive and expanding demand for animal products for traditional uses (Hutchings 1989, 1996; Cunningham & Zondi 1991; La Cock & Briers 1992), exacerbated by the growing human population (Wittemyer et al. 2008), migratory influxes to the province (Jacobs 2014), and high levels of unemployment (Cunningham 1991; Statistics South Africa 2016).

The aim of this study was thus to acquire information, inventory, and document the use of animal-derived materials by diviners, herbalists and general animal-parts traders (hereafter collectively referred to as traditional healers) operating from impoverished, rural communities in the Boland Region of the Western Cape, South Africa. Ethnoecological research has transformed during the past few decades from predominantly qualitative descriptions to analyses that incorporate quantitative models and methods, allowing for the emergence of testable hypotheses and reproducible statistical comparisons (Begossi 1996; Cunningham 2001; Wong et al. 2001; Hoffman & Gallaher 2007). Quantitative analyses, although not always necessary, further allows for a better understanding of the research subject, the ability to evaluate sampling effort, and the ability to arrive at clear and concise conclusions (Begossi 1996; Williams et al. 2005). The incorporation of ecological models in the analysis of socio-ecological relationships have however not been without criticism (Balée 1989), but the popularity of these applications has nonetheless grown considerably.

In this study a variety of quantitative approaches were employed to specifically explore: (1) vertebrate species incidence, richness and diversity, and (2) the species most valued by local African communities. This will provide novel insights into the extent and dynamics of the traditional healing enterprise, as well as the demand for vertebrate taxa, thus determining conservation priorities and enabling effective policy development inclusive of both ecological sustainability priorities as well as social demands.

4.3 Materials and Methods

4.3.1 Study area

Research was undertaken in 17 townships and informal settlements in rural or peri-urban landscapes in the Boland Region (~4 000 km^2), part of the Western Cape

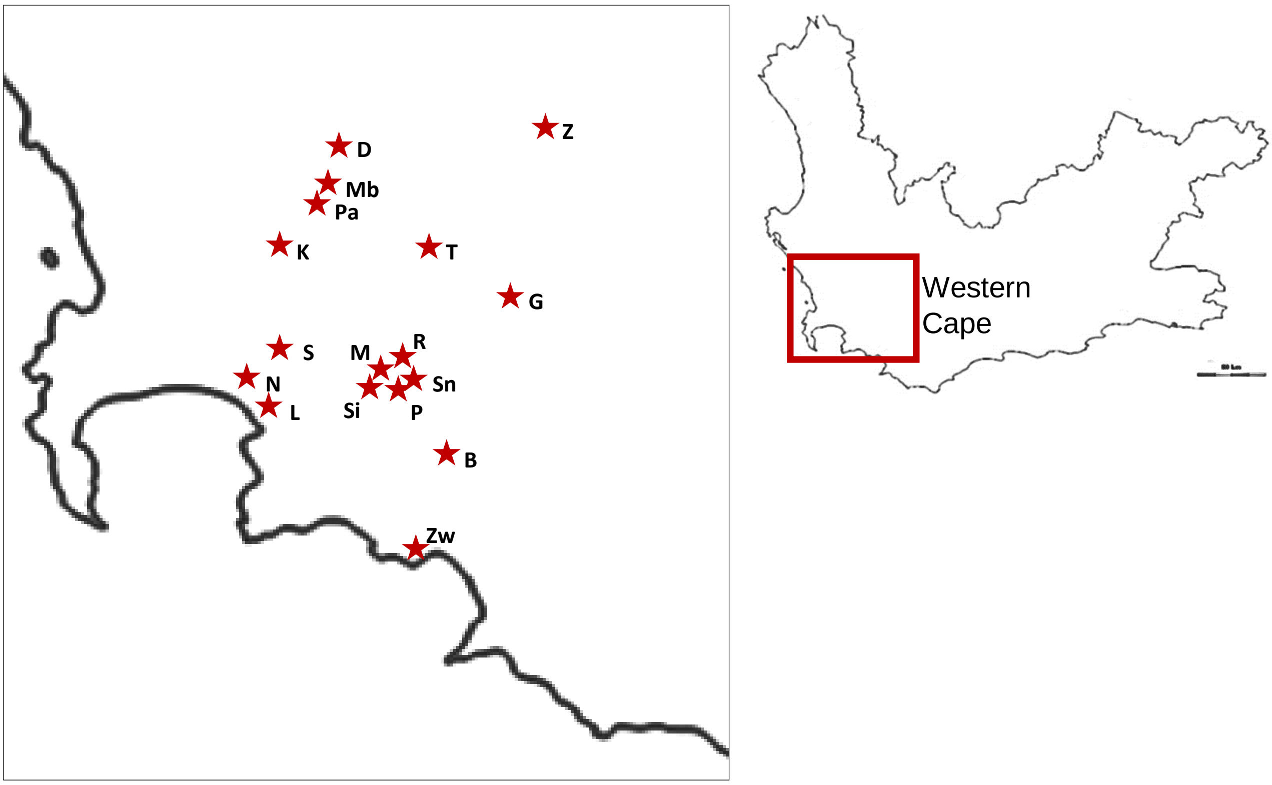

Figure 4.1. Sampled communities in the Western Cape Province of South Africa. The communities were: Zweletemba (Z), Drommedaris (D), Mbekweni (Mb), Paarl SP (Pa), Kayamandi (K), Tjotjombeni (T), Goniwe Park (G), Sir Lowry's pass village (S), Nomzamo (N), Lwandle (L), Marikana (M), Rooidakkies (R), Siyanyazela (Si), Pineview (P), Snake Park (Sn), Botrivier SP (B), and Zwelihle (Zw).

Province of South Africa (Fig. 4.1). The sampled sites were purposively chosen to be inclusive of all such residential communities in the study area with indigenous African people contributing to > 20% of the overall population demographics. Large racial and ethnic diversity however still remained among sites, enabling the comparison of demographically heterogeneous and homogeneous settlements. The Boland Region is divided into the inland Cape Winelands and the coastal Overstrand areas, consisting of various open and closed protected areas enveloped by vast expansions of agriculturally transformed lands. The remaining natural habitats support low animal biomass due to the largely unpalatable and nutrient-deficient fynbos vegetation (Coetzee 2016). As a result, leopard (*Panthera pardus)*, Cape mountain zebra (*Equus zebra zebra)* and bontebok (*Damaliscus pygargus*) are the only large mammal species that are able to persist without human intervention (Coetzee 2016). The Western Cape however boasts high levels of endemism of mammal, amphibian, reptile and avifauna taxa (11 – 54% endemism), and supports approximately 50% of all terrestrial vertebrate species found in South Africa (Turner 2012). Data collection took place in autumn (April to May) of 2018.

4.3.2 Data collection

To gather information on the use of vertebrate species by traditional healers, semi-structured interviews were conducted with 36 respondents in townships and informal settlements (n = 17). Unlike the more obvious and prominent *umuthi* markets found in Johannesburg and Durban (Williams et al. 1997, 2007; Mander 1998; Whiting et al. 2011; Williams & Whiting 2016), traditional healing in the Western Cape is much more discreet (Petersen et al. 2014). Consequently, identifying potential informants was achieved using a non-probability snowball-sampling approach i.e. community members were asked to locate neighbours fitting the criteria. Snowball-sampling is an efficient manner of gathering information during purposive sampling in situations where there is no obvious information on the whereabouts of the population of interest (Bernard 2002; Morgan 2008). All interviews (Appendix 6.3) were conducted in the home languages of informants, either isiXhosa (81%) or Sesotho (19%). Respondents ranged between ages 25 and 62 years (median = 37.5 years), and comprised mostly males (78%). All respondents reportedly migrated from the Eastern Cape Province. Information was recorded on the basic demographics of the respondent (e.g. age, level of education and ethnicity), the species part and products used, the purpose for their use, and product prices. Interviews ranged between 50 minutes and two hours. The Western Cape Province hosts a great variety of species lineages, as well as many cryptic species (i.e. species that are hard to discern morphologically) (Lindner 2002). Animal identification cards were therefore used to facilitate memory recall, eliminate the possibility of disparity in species names, or the grouping of species as ethnospecies (i.e. a folk or common name liberally assigned to a number of closely-related species) (Hanazaki et al. 2000). Where possible, species parts or products were identified on site.

All interviews were conducted anonymously and information was kept confidential. At least a certain degree of reluctance was expected in providing information on the use of animal species, especially those relating to the treatment of magico-medicinal ailments, as reported by Cunningham & Zondi (1991), Herbert et al. (2003) and Williams & Whiting (2016). However, respondents were almost exclusively extremely forthcoming, inviting, and even excited to have their knowledge formally documented. Only one respondent refused to participate due to recent run-ins with law enforcement (nonresponse = 2.7%), and uses for four species were not recorded (5.6%). Consent was obtained from each respondent for the use of their accounts prior to every interview (Appendix 6.5), and ethical clearance was obtained from the Stellenbosch University ethics committee: Humanities (Reference: CEE-2018-6251) preceding the data collection phase of this study.

4.3.3 Statistical analysis

4.3.3.1 Species inventory and use prevalence

Basic descriptive statistics were used to analyse the highest incidence of vertebrate species sold by traditional healers, as well as species sold that were of conservation concern. Species with the most uses as cited by traditional healers, the most prevalent animal parts or products, and their corresponding market values were also described.

4.3.3.2 Sampling performance

Sampling performance was evaluated by constructing rarefaction curves. Rarefaction curves (also known as species accumulation curves, species effort curves, and collector's curves) plot the cumulative number of species observed against the effort it required to obtain the species sample (Colwell and Coddington 1994; Hayek & Buzas 1997). Rarefying the sampled data thus produced a smoothed curve that estimates how many species would be found if sampling continued indefinitely (Colwell 2005) based on the rate at which new species were discovered during the study period. It is also an efficient manner in which to evaluate sampling effort and species richness when sample sizes vary, as comparing raw taxon counts may be incorrectly interpreted (Begossi 1996; Gotelli & Colwell 2001). In theory, sampling size is deemed sufficient when the expected number of species, $E(S_n)$, does not increase with the addition of more individuals to the sample (Begossi 1996). This is indicated by the 'levelling-off' of $E(S_n)$ on a rarefaction curve. For example, if more traditional healers were interviewed in more informal settlements, fewer new species are expected to be discovered as time progresses, and hence the curve will start to level off. The following formula computes the rarefaction curve:

$$E(S_n) = \sum\{1 - [\frac{N_n - pi}{N}]\}$$

Where, E(S) = expected number of species in the rarefied sample, n = standardized sample size, N = total number of individuals recorded in the sample to be rarefied, and pi = the number of individuals in the ith species in the sample to be rarefied (Magurran 1988).

4.3.3.3 Species richness, diversity and evenness

Ecological diversity indices were used to quantify the richness, diversity and evenness or equitability of animals used and sold by traditional healers in the Boland Region, to answer the following questions: (1) what was the observed richness of species sold and used by traditional healers within sampled communities, and how many new species can be expected if sampling continued? (2) Were the same species sold among sampled communities, and was the trade dominated by one or a few species? (3) How heterogeneous was the sampled community in terms of species sold/used, and how difficult would it be to correctly predict the identity of the next species found if sampling continued? Information regarding the number and frequency of species occurrence in the area was pivotal to achieving these calculations (Begossi 1996; Williams et al. 2005).

Species richness (S) is a simple numerical value describing the number of species recorded per unit area, and instantly expresses species diversity (Magurran 2004). Species richness typically depends on some measure of sampling effort (Begon et al. 1986; Hayek & Buzas 1997), i.e. it is likely that more species were recorded as more traditional healers were approached. Species richness indices were computed because it is not feasible to enumerate all species in the sampled population (Colwell & Coddington 1994). Observed species richness (S) may however be heavily dependent on sampling effort. Therefore, species richness estimation curves were used to measure and compare various estimators of species richness by adding unseen species to the observed species richness (Colwell & Coddington 1994; Gotelli & Colwell 2001; Colwell et al. 2004), facilitating improved interpretation of species richness outcomes, especially since samples varied in size (Williams et al. 2005). The performance of six non-parametric species richness estimators appropriate for incidence-based data (i.e. information on species frequencies), namely Chao 2, first-order jackknife (Jack 1), second-order jackknife (Jack 2), incidence-based coverage estimator (ICE), bootstrap (Boot), and Michaelis-Menten Means (MMMeans) were calculated and compared. The Chao 2 estimator (Chao 1987) considers rare species and the total number of species observed in the sample to calculate its richness (Basualdo 2011). Jack 1 and Jack 2 uses counts of singletons, and singletons and doubletons, respectively (Burnham & Overton 1978; Heltshe & Forrestor 1983; Smith & van Belle 1984). ICE is based on species found in 10 or fewer sampling units, and Bootstrap is based on the proportion of the samples containing each sample (Colwell & Coddington 1994; Colwell 2005;

Williams et al. 2007). For each of the calculations, the sample order was randomized 100 times to compute mean statistics at each sample order, thereby generating smooth accumulation curves (Palmer 1991; Colwell & Coddington 1994).

Species diversity indices are often used to quantify the use intensity of resources utilised by people in different communities to allow for the comparison between different communities (Begossi 1996). The application of species diversity indices in this study gave an approximation of how heterogeneous the sample was, i.e. how difficult it would be to correctly predict the species of the next animal recorded if sampling continued (Magurran 2004; Magurran & McGill 2011). If diversity is low, the probability of correctly predicting the species of the next animal recorded will be high, and vice versa. Both species richness (S) and evenness or equitability (i.e. how uniformly abundant species are in a sample) are incorporated to produce a single value based on the proportional abundance of a species in a sample (Ludwig & Reynolds 1988; Magurran 2004; Magurran & McGill 2011). Ecological diversity indices are typically divided into four categories (Williams et al. 2005), namely indices derived from information theory, such as the Shannon-Wiener diversity index and the Brillouin index, dominance indices, for example Simpson, McIntosh and Berger-Parker indices, Hill's diversity numbers, and Fisher's alpha. In this study the (i) Shannon-Wiener index, (ii) Simpson's index, (iii) a variety of Hill's numbers, and (iv) Fisher's alpha were calculated. These indices have previously been successfully applied in ethnobotanical studies (Begossi 1996; Williams et al. 2005, 2007), as well as in at least one ethnozoological study (Whiting et al. 2011).

(i) The Shannon-Wiener index (H') is widely applied in ecological studies and is regarded as having moderate sensitivity to sample size (Magurran 1988). Since uncertainty is synonymous with diversity (Krebs 1989), the Shannon-Wiener index essentially measures the average degree of uncertainty in a sample (Ludwig & Reynolds 1988). A high output value (H') would thus imply a high level of diversity, and subsequently also a high degree of uncertainty i.e. greater difficulty in correctly predicting the species found in a subsequent sample. The value H′ also tends to increase as species richness (S) increases, because more species results in a more even spread off individuals across species, thus increasing uncertainty levels. Maximum uncertainty ($H'_{max} = \ln(S)$) occurs when each species in a sample is equally represented (Hayek & Buzas 1997). Where the Shannon-Wiener's index has been applied in ethnoecological studies (Begossi 1996; Williams et al. 2005, 2007; Whiting et al. 2011), the resulting values were typically high (range: 2.99 – 5.46). The Shannon-Wiener index formula is given as:

$$H' = -\sum_{i=1}^{R} p_i \ln p_i$$

Where, P_i = the proportion of individuals belonging to the *i*th species.

(ii) Simpson's diversity index (λ), a dominance index, proposes that diversity is inversely related to the probability that two species picked at random from a sample belong to the same species (Simpson 1949; Williams et al. 2005). When there is a low probability (λ approaching 0), the sample has a high level of diversity (Pielou 1976; Ludwig & Reynolds 1988; Krebs 1989). Conversely, when λ approaches 1, most individuals in the sample are concentrated in a single species, implying a high level of dominance by one or a few species (Pielou 1976). Because λ decreases as diversity increases, Simpson's index is usually expressed as 1-λ, or 1/λ. In ethnoecological studies (Williams et al. 2005; Whiting et al. 2011), Simpson's diversity index is usually low, indicating high levels of diversity and evenly distributed individuals across species. Simpson's inverse index (1/λ) is given by the following formula:

$$1/\lambda = 1/\sum_{i=1}^{R} p_i^2$$

(iii) Hill's diversity numbers, developed by Hill (1973), gives a measure of the effective number of species in a sample (Hill 1973; Ludwig & Reynolds 1988). The numerical value for N_1 indicates the number of abundant species in a sample, and N_2 indicates the number of very abundant species in a sample. Low values imply low frequency of species occurrence, and thus greater dominance by a few species. Conversely, high values imply a large range of species in the sample with relatively equal occurrences. Hill's numbers are mathematically related to the Simpson and Shannon-Wiener indices.

$$N_1 = e^{H'} \qquad\qquad N_2 = \frac{1}{\lambda}$$

(iv) Fisher's alpha (α) is generated from a species abundance model that is used to fit logarithmic series distribution models once the parameter χ has been solved for iteratively (Williams et al. 2005). A low alpha value would thus be expected if our sampled community contained few species (Williams et al. 2005). Fisher's alpha is mostly affected by intermediate species, as opposed to rare or abundant species, therefore providing a value close to the observed number of species found in one sample (Hayek & Buzas 1997). Fisher's alpha is also affected by sample size (Whiting et al. 2011).

Measures of evenness (equitability or homogeneity) will attempt to quantify the abundance of species in the sampled community against a theoretical sample wherein all species are equally represented (Pielou 1976; Krebs 1989), using both species richness (S) and species diversity values (Magurran 1988). Values approaching 1 imply high evenness, and thus low dominance by any species in the sample. Conversely, values approaching 0 imply low evenness and thus high dominance in the use of a few species by traditional healers (Begossi 1996). In this study, evenness was expressed with the Shannon index of evenness (J'), and per recommendation of Ludwig & Reynolds (1988), also Hill's E5.

All values for indices and species-richness estimators were calculated using EstimateS software v9.1.0.

4.3.3.4 Relative cultural importance (RCI) indices

Quantitative analysis in ethnoecology has become increasingly popular (Begossi 1996; Cunningham 2001; Wong et al. 2001; Hoffman & Gallaher 2007). For example, the suite of relative cultural importance (RCI) indices has gained popularity since their inception in the late 1980's (Boom 1990), and are increasingly being used to enable the transformation of complex and multidimensional ideas of '*importance*' into standardized and comparable numerical scales and values (Turner 1988; Kvist et al. 1995; Alexiades & Sheldon 1996; Begossi 1996; Martin 2004; Reyes-García et al. 2006; Hoffman & Gallaher 2007).

4.3.3.4.1 Informant Consensus Factor (F_{ic})

The informant consensus factor (F_{ic}) method was first developed in the 1980's for calculating consensus and variation in the use of plants in traditional pharmacopoeias (Trotter and Logan 1986). The method rests on the assumption that the greater the degree of group consensus regarding the use of ethnomedicinal species for treating certain conditions are, the greater the probability that the specific treatment is physiologically active or effective (Trotter and Logan 1986). Heinrich et al. (1998) later adapted the method, followed by a further modification by Alves & Rosa (2006) to incorporate animals in zootherapeutics. Values are scored between 0 and 1, with high values relating to a higher degree of informant consensus or homogeny on which animals are considered effective in the treating of a certain ailment. Conversely, a low value approaching 0 indicates a high degree of variation in the number of different animals used to treat a particular ailment. Using this method, it is thus possible to calculate the degree of socio-cultural coherence regarding animals being used within and among certain communities with respect to similar ailments. According to the work of Alves & Rosa (2006), the F_{ic} formula is:

$$F_{ic} = (n_{ur} - n_t)/(n_{ur} - 1)$$

In this formula, n_{ur} equals the number of use citations in each use category, and n_t equals the number of species used per use category. F_{ic} values were only calculated for use categories with > 4 independent citations.

4.3.3.4.2 Fidelity Level (FL) and Rank Order Priority (ROP)

The fidelity level (FL) reveals the proportion of respondents using specific animal species for the same purpose (Alexiades & Sheldon 1996), thus enabling the quantification of the '*importance*' of a species for a given purpose. The fidelity level is calculated using the following formula:

$$FL\ (\%) = \frac{I_p}{I_u} \times 100$$

In this formula, I_p represents the number of respondents who cited the use of a species for a particular purpose, and I_u represents the total number of respondents that cited the species for any purpose. This method is however largely inhibited by the presence of inflated values seen in species with only a few citations. To adjust for this limitation, Friedman et al. (1986) suggested multiplying the fidelity level value with a relative popularity level (RPL) value to arrive at a corrected FL value, or rank order priority (ROP) value. RPL values are assigned based on the number of times a species is cited, and ranges from unpopular (0) to popular (1). The formula given by Friedman et al. (1986) is:

$$ROP = FL \times RPL$$

To assign a RPL value to each species and associated purpose of use, a coordinate system was created to analyse the association between the number of informants citing the use of a particular animal species and the number of uses cited for the animal species. The resulting graph (Fig. 4.2) showed a linear relationship between the two variables, i.e. the greater the number of informants citing the use of an animal for any purpose, the greater the amount of information, and hence the greater the average number of uses per species. However, since the number of uses for an animal is finite, the average number of uses would not indefinitely increase as more informants cite the use of an animal. To this end, all animal species were divided into '*popular*' and

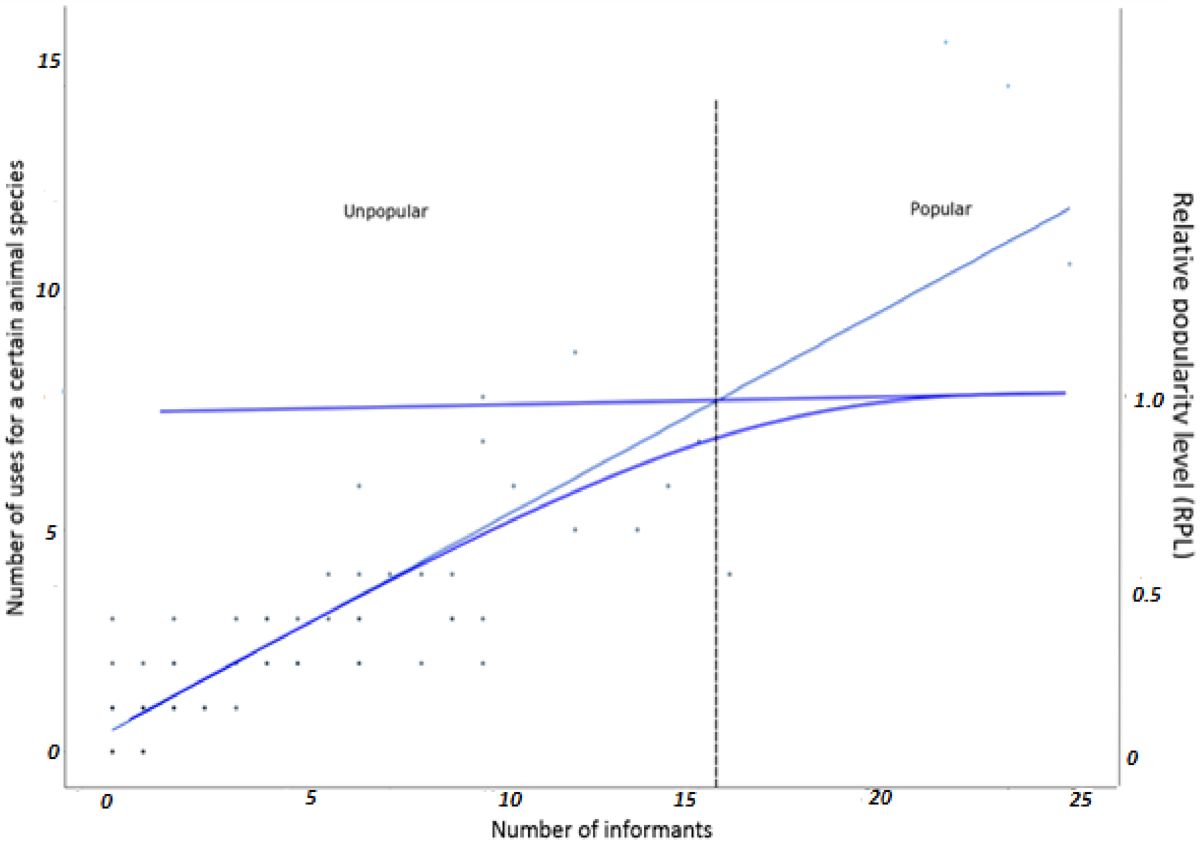

Figure 4.2. The relationship between the number of respondents who cited a particular animal species and the number of uses for that animal species cited by all respondents. Animal species were divided into '*unpopular*' and '*popular*' groups based on the number of times they were cited.

'*unpopular*' groups based on the number of informants that cited their use. The division was drawn perpendicular to the intersection of the linear regression line and the horizontal line representing the 'levelling-off' of new uses for the same species. Animal species assigned to the '*popular*' group (≥ 16 informants) received RPL values of 1. Animal species in the '*unpopular*' group (< 16 informants) received RPL values <1 in accordance with its position on the graph. For example, dolphin spp. were cited by 9 informants for use in 6 purposes, and consequently received a RPL score of 0.75, while the Egyptian mongoose (*Herpestes ichneumon*) was cited by 4 people for a single use, resulting in a RPL score of 0.125.

4.3.3.4.3 Cultural Significance Index (CSI)

The cultural significance index (CSI) was designed to calculate '*importance*' through researcher-determined weighted ranking of multiple factors (Hoffman & Gallaher 2007). The CSI was first proposed by Turner (1988) to record the cultural significance of various plant species in a Canadian cultural group. In Turner's (1988) original version of the CSI, scores were assigned on a five-point scale to variables describing the quality (q) and intensity (i) of the use of plant species by cultural groups, while a score of either 0.5, 1 or 2 was assigned to describe the preference or exclusivity (e) of use for the plant species. A major limitation of this method was the high emphasis placed on edible plants compared to other culturally significant plant species, for example plants used in ritual ceremonies (Da Silva et al. 2006). Turner's (1988) CSI formula is as follows:

$$ICS = \sum_{i=1}^{n} (q * i * e)$$

This formula was later adapted by Stoffle et al. (1990) and further modified by Lajones & Lemas (2001) by adding the ethnobotanical importance value index. The aim was to decrease subjectivity by adding the p/u variable, calculated as the sum of the total number of uses of plants used for a specific purpose (Hoffman & Gallaher 2007). This method however still valued the researchers' perspective more than the informant's (Da Silva et al. 2006).

$$EICS = \sum_{i=1}^{n} (\frac{p}{u} * i * e * c)$$

In 2006, Da Silva et al. (2006) further modified Turner's (1988) original CSI methodology by adding new elements, reformulated CSI variables, and the use-value (UV) calculation originally proposed by Phillips & Gentry (1993). Their applied study in North-eastern Brazil reduced the subjectivity of the CSI methodology by revising the calculations to a two-point scale, as well as by incorporating the use-value as a correction factor (CF) to further reduce the sensitivity of the CSI method to sampling intensity (Da Silva et al. 2006; Hoffman & Gallaher 2007). The revised CSI method given by Da Silva et al. (2006) for plant species was used in the current study for vertebrate animal species as given below:

$$CSI = \sum_{i=1}^{n} (i * e * c) * CF$$

Species management (i) quantifies the impact of an animal species on the daily life of the traditional healers in terms of resource allocation (Turner 1988). A value of 2 was given to a species that is reared, managed, or manipulated in any way. For example tortoise species that were kept to harvest scat or blood. A value of 1 was given to species found liberally in the Western Cape.

The use preference (e) value is based on the preference given to a particular animal species for a given purpose, compared to other species that are used for the same purpose. A value of 2 was given to species that were the preferred choice for a given purpose, while a value of 1 was given to all other species that were not the preferred choice for that given purpose.

The use frequency (c) value is based on the effectivity to which an animal species is used for a given purpose. In this study, a value of 2 was given to species that were cited > 2 times independently for a given purpose, while a value of 1 was assigned to species cited ≤ 2 times for a given purpose. The dividing value (2 citations) is the median number of all citations given for all species and all purposes, thus presenting a 50:50 split.

The correction factor (CF) is based on the degree of consensus presented by informants, calculated using the formula:

$$CF = \frac{Number\ of\ citations\ for\ a\ species}{Number\ of\ citations\ for\ the\ most\ cited\ species}$$

An example, using the caracal (*Caracal caracal*), of the CSI calculations for each species in the study, is given below (Table 4.1).

Table 4.1. An example of the cultural significance index (CSI) methodology as revised by Da Silva et al. (2006). In this example, weights were assigned to each variable (i, e, c) for a maximum of three specific uses (SU). CF = correction factor.

Species	**Citations**		**SU 1**	**SU 2**	**SU 3**	**Sum (i*e*c)**	**CF**	**CSI**
Caracal	13	Management (i)	1	1	1	**6**	0.41	**2.46**
		Preference (e)	1	1	1			
		Frequency (c)	2	2	2			
		(i*e*c)	**2**	**2**	**2**			

This revised method is however still limited by its assumption that the number of citations can be directly translated to '*importance*', while this may not be the case. Actual use can by influenced by numerous factors, including age, gender, seasonality, traditions, and knowledge and cultural degradation (Hoffman & Gallaher 2007).

4.4 Results

4.4.1 Species incidence and use prevalence

The 71 vertebrate species or morphospecies cited by traditional healers in the sampled communities belonged to four classes and 20 orders (Appendix 4.1). The main orders were Carnivora (20 species), Artiodactyla (17 species) and Squamata (10 species).

A total of 17 species and morphospecies were enumerated in > 30% of sampled communities (n = 17, Table 4.2). These were chacma baboon (*Papio ursinus*, 82.4%), leopard (*P. pardus*, 82.4%), Cape porcupine (*Hystrix africaeaustralis*, 76.5%), puff adder (*Bitis arietans*, 76.5%), genet spp. (*Genetta* spp., 58.8%), black-backed jackal (*Canis mesomelas*, 52.9%), monitor lizard spp. (*Varanus* spp., 52.9%), honey badger (*Mellivora capensis*, 47.1%), Hare spp. (*Lepus* spp., 47.1%), Cape clawless otter (*Aonyx capensis*, 41.2%), African buffalo (*Syncerus caffer*, 41.2%), African rock python (*Python sebae*, 41.2%), Owl spp. (41.2%), Cape fox (*Vulpes chama*, 35.3%), Spiral-horned antelope spp. (*Taurotragus* spp. & *Tragelaphus* spp., 35.3%), and Vulture spp. (*Gyps* spp., 35.3%). Of all vertebrate animal species listed, mammal taxa (73.2%) were far more prevalent than either reptile (18.3%) or bird (7.0%) taxa. Only one vertebrate fish species was recorded. No amphibian or invertebrate (marine or terrestrial) species were identified.

Table 4.2. High incidence species in sampled communities (n = 17). Species recorded in > 40% of the sampled communities are highlighted in grey shading.

Mammals		Reptiles		Birds	
Common name	**Samples (>20%)**	**Common name**	**Samples (>10%)**	**Common name**	**Samples (>10%)**
Chacma baboon	82.4	Puff adder	76.5	Owl spp.[a]	41.2
Leopard	82.4	Monitor lizard spp.[a]	52.9	Vulture spp.[a]	35.3
Cape porcupine	76.5	African rock python	41.2	Ostrich	17.7
Genet spp.[a]	58.8	Nile crocodile	29.4	Swallow spp.[a]	11.8
Black-backed jackal	52.9	Cape cobra	29.4	Cattle egret	11.8
Honey badger	47.1	Mamba spp.[a]	23.4		
Hare spp.[a]	47.1	Snake spp.[a]	17.7		
Cape clawless otter	41.2	Angulate tortoise	17.7		
African buffalo	41.2	Rinkhals	11.8		
Cape fox	35.3	Southern rock agama	11.8		
Spiral-horned antelope spp.[a]	35.3	Girdled lizard spp.[a]	11.8		
Caracal	35.3				
Aardvark	29.4				
Cape grysbok	29.4				
Springbok	29.4				
Rock hyrax	29.4				
Bat-eared fox	29.4				
Common duiker	29.4				
Aardwolf	29.4				
Grey rhebok	29.4				
Lion	23.5				
African wild cat	23.5				
Striped weasel	23.5				
Striped polecat	23.5				
Dolphin spp.[a]	23.5				
Cape fur seal	23.5				
Blue wildebeest	23.5				
Warthog	23.5				
Greater kudu	23.5				
Bushpig	23.5				
Brown hyena	23.5				

[a]Individuals not identifiable up to species-level

Twelve species recorded (Table 4.3) are listed of conservation concern by the IUCN Red List of threatened species (version 2017-3). Critically Endangered (CR) and Endangered (EN) species (highlighted in grey) included several vulture species of the genus *Gyps*, tiger (*Panthera tigris*), and an unidentified rhinoceros species (either of the genus *Diceros* or *Ceratotherium*). Additionally, several *Cordylus* spp. (girdled lizard spp.) are Near Threatened (NT). Honey badger and Southern African hedgehog (*Atelerix frontalis*) were listed as Near Threatened in the preceding version of the IUCN Red List of threatened species (version 3.1).

Table 4.3. Vertebrate species of conservation concern according to the IUCN Red List of Threatened Species (2001 categories, version 2017-3, Global assessment) that were sold by traditional healers in the sampled market.

Species	Common name	IUCN Category	Population trend
Acinonyx jubatus	Cheetah	VU	Decreasing
Aonyx capensis	Cape clawless otter	NT	Decreasing
Equus zebra zebra	Cape mountain zebra	VU	Unknown
Gyps spp.[a]	Cape vulture	EN/CR	Decreasing
Hippopotamus amphibius	Hippopotamus	VU	Stable
Hyaena brunnea	Brown hyena	NT	Unknown
Loxodonta africana	African elephant	VU	Increasing
Panthera leo	Lion	VU	Decreasing
Panthera pardus	Leopard	VU	Decreasing
Panthera tigris[b]	Tiger	EN	Decreasing
Pelea capreolus	Grey rhebok	NT	Decreasing
Diceros/Ceratotherium[c]	Rhinoceros spp.	CR	Increasing

[a]Conservation status varies between species
[b]Exotic species
[c]Species unknown

The number of uses for each animal varied from one to 16 (median = 2; Fig. 4.3), with the most uses attributed to Cape porcupine (16 uses), leopard (15 uses) and chacma baboon (11 uses). Uses were not recorded for four species, namely Southern African hedgehog, whale spp. (Cetaceae), tiger, and girdled lizard spp.

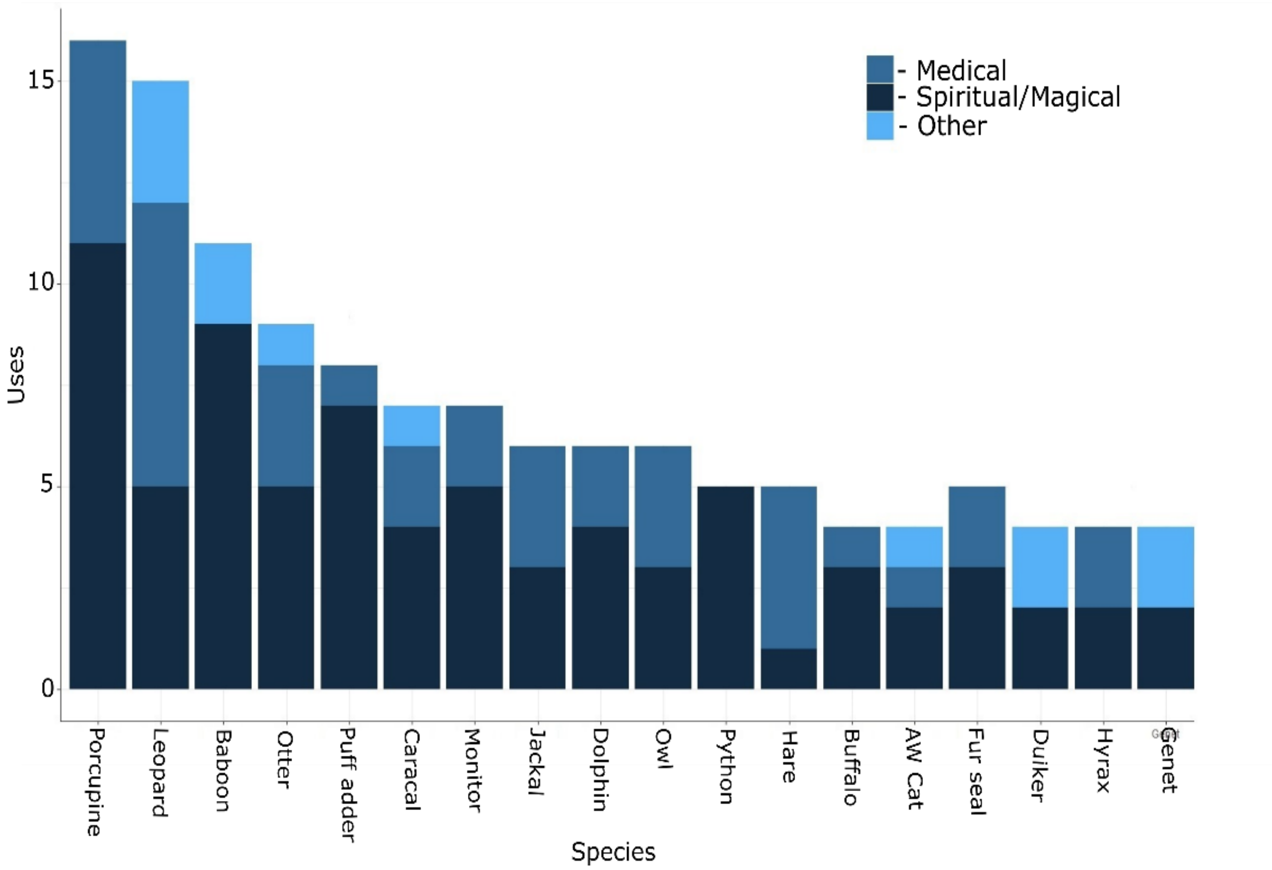

Figure 4.3. The animal species with the most use categories attributed to them by traditional healers. Bars were divided to display uses relating to medical ailments, spiritual or magical purposes, and other uses (clothing, jewellery, status symbols etc.).

A total of 716 vertebrate species parts or products were listed as being used in traditional medicinal practices in sampled communities (median = 23, range = 7 – 34). These were subsequently grouped into 12 categories for analyses (Fig. 4.4). Skin pieces and entire skins were the most prevalent form of animal constituents used or sold by traditional healers (258 items), followed by animal oil and subcutaneous fat (120 items). Animal bones (62 items), entire carcasses (56 items) and internal organs (46 items) were also highly prevalent, as well as assorted hooves, paws and talons (44 items), and quills, feathers, fur and scales (44 items).

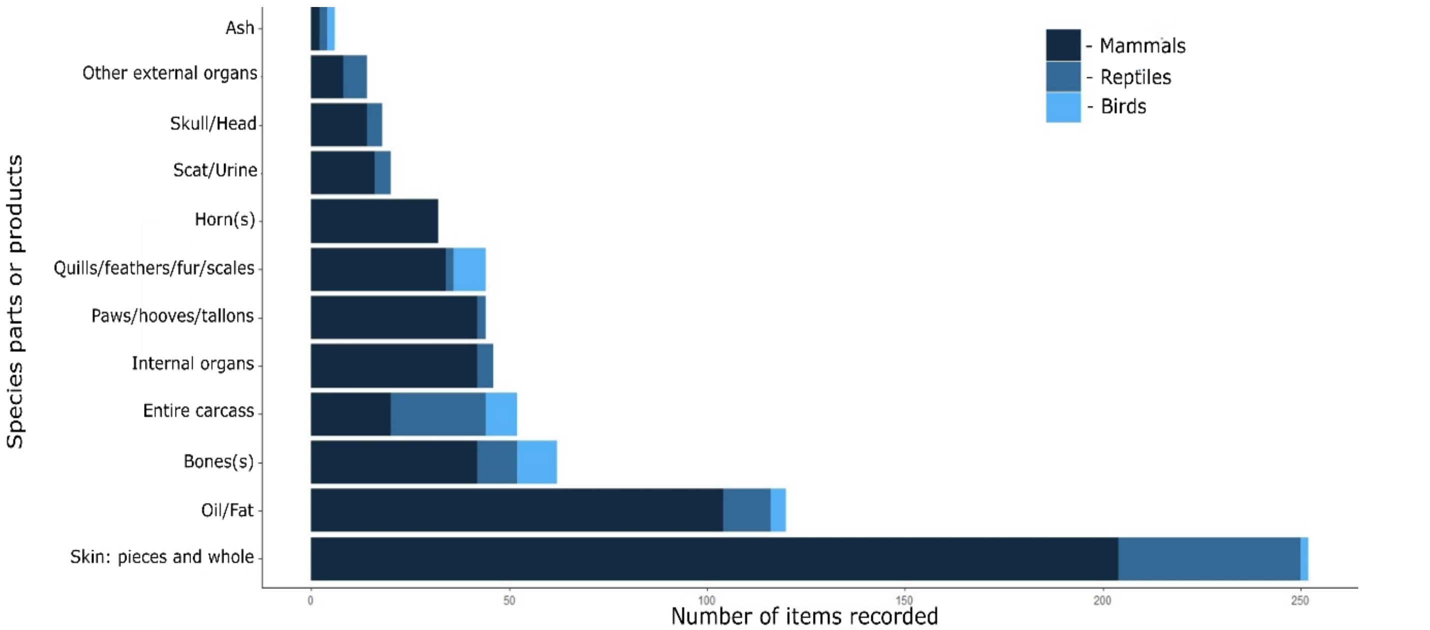

Figure 4.4. Species parts and products most frequently sold by traditional healers in the sampled community. Bars were divided to present proportional incidence of mammal, reptile and bird taxa.

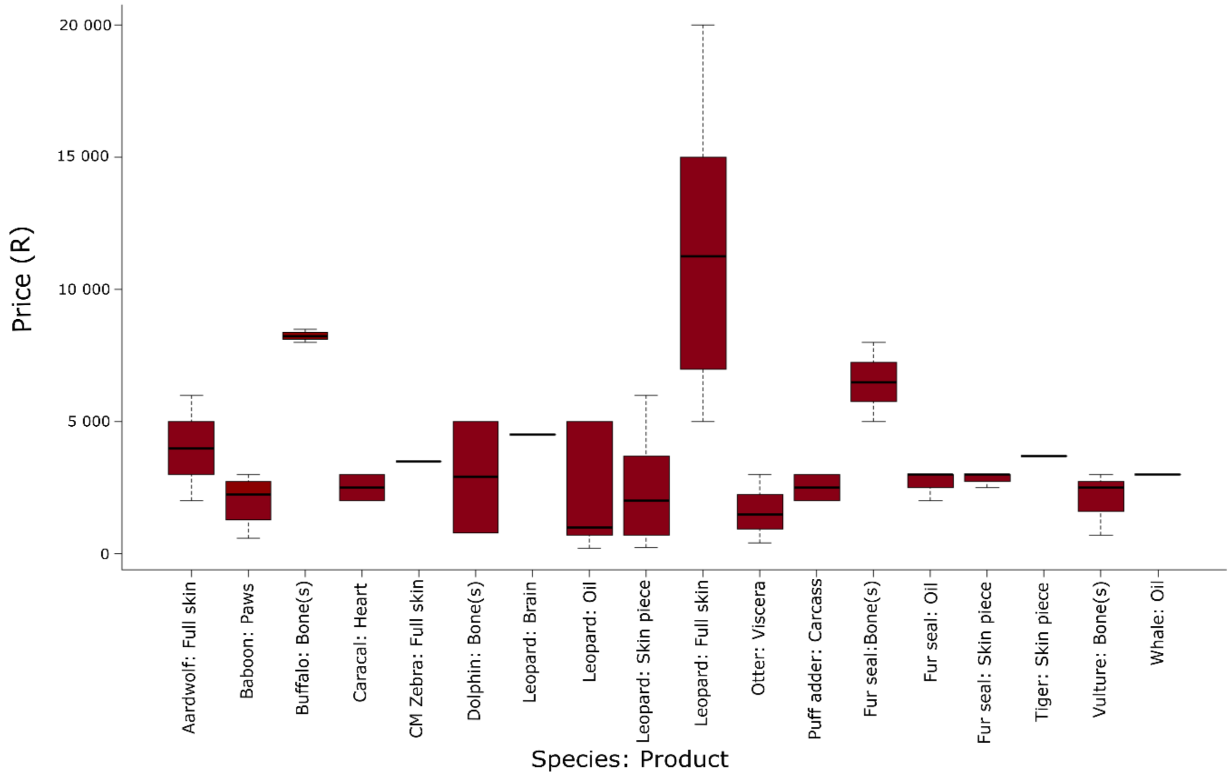

Figure 4.5. Comparative prices (R) of the most expensive animal parts or products recorded in traditional healing practices in the Western Cape Province. The minimum, first quartile, median, third quartile, and maximum value is shown. Entire carcasses of large carnivore species were excluded because it is rarely affordable for a single person.

High market values were recorded for wildlife items sold by traditional healers in the Boland (Fig. 4.5; Appendix 4.1). Prices were however highly variable across items as well as among the same items sold by different healers; even for the same purpose (range: R15 – R20 000). The most expensive animal items were entire skins of leopard (median = R11 250, range = R5 000 – R20 000), bones of African buffalo (median = R8 250, range = R8 000 – 8 500), bones of Cape fur seals (*Arctocephalus pusillus*, median = R6 500, range = R5 000 – R8 000).

4.4.2 Sampling performance

Sample-based rarefaction curves were plotted for vertebrate animal taxa sold by traditional healers in the Boland Region, along with corresponding 95% confidence intervals (Fig. 4.6). None of the samples truly reached an asymptote. However, they were all clearly approaching an asymptote as the rate at which new species were found gradually decreased. Therefore, sampling size was said to be sufficient (Heck Jr et al. 1975; Williams et al. 2007). Further sampling of traditional healers would thus not yield a significant number of novel, undiscovered species for any of the individual vertebrate classes. The rapid initial accumulation shown by reptiles and bird groups indicated that species were accumulating faster for comparatively small sample sizes. Species richness was correlated with the number of settlements in which the vertebrate class was found.

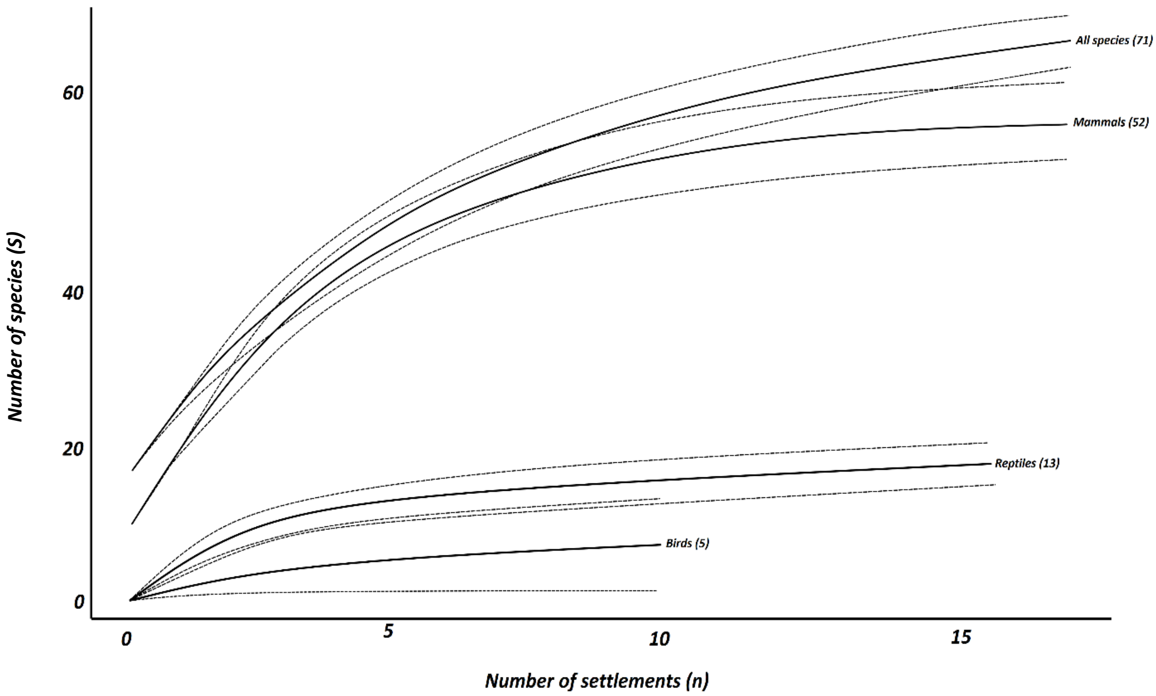

Figure 4.6. Rarefaction curves (solid lines) for vertebrate animals used by traditional healers in informal settlements in the Western Cape Province (n = 17), with 95% confidence intervals (dashed lines). Parentheses indicate sample sizes.

4.4.3 Species richness, diversity and evenness

Species richness estimates and other summary values were calculated for three sets of data; the first containing information on all species recorded in the sampled communities, the second only mammal species and the third only reptile species (Table 4.4). Bird taxa were excluded from analysis due to their relatively low incidence in the sampled community.

Table 4.4. Species richness estimates and other values characterising the use of animals by traditional healers in the Western Cape Province. Bird taxa were excluded due to their relatively low incidence in the market.

	All species	**Mammals**	**Reptiles**
Samples (n)	17	17	16
Individuals (N)	450	365	67
Observed species richness	71	52	13
Estimated species richness			
ICE	74.7	54.5	13.6
Chao 2	73.3 ± 3.0	55.4 ± 3.2	13.0 ± 1.8
Jack 1	79.4 ± 2.1	58.5 ± 1.5	14.9 ± 0.8
Jack 2	78.3	61.5	14.2
Bootstrap	75.5	55.8	14.2
MMMean	85.8	61.1	14.5
Singletons[1]	9	7	1
Doubletons[2]	13	6	3

[1]Number of species occurring only once across all samples
[2]Number of species occurring only twice across all samples

For the dataset containing records of all species, all estimators appeared to approach an asymptote sooner than the observed species accumulation curve (Fig. 4.7), indicating their usefulness as adequate species richness estimators (Toti et al. 2000). The MMMeans estimator understandably reached an asymptote first, given its asymptotic nature (Magurran 2004). The remaining estimators approached asymptotes in relative parallel to the observed species curve. The difference between the highest estimator (MMMeans) and lowest estimator (Chao 2) was 12.5 species. The observed species richness was 83% of the highest richness estimator. Plots of singletons and doubletons rose rapidly at the first samples and then leveled off as sample size increased.

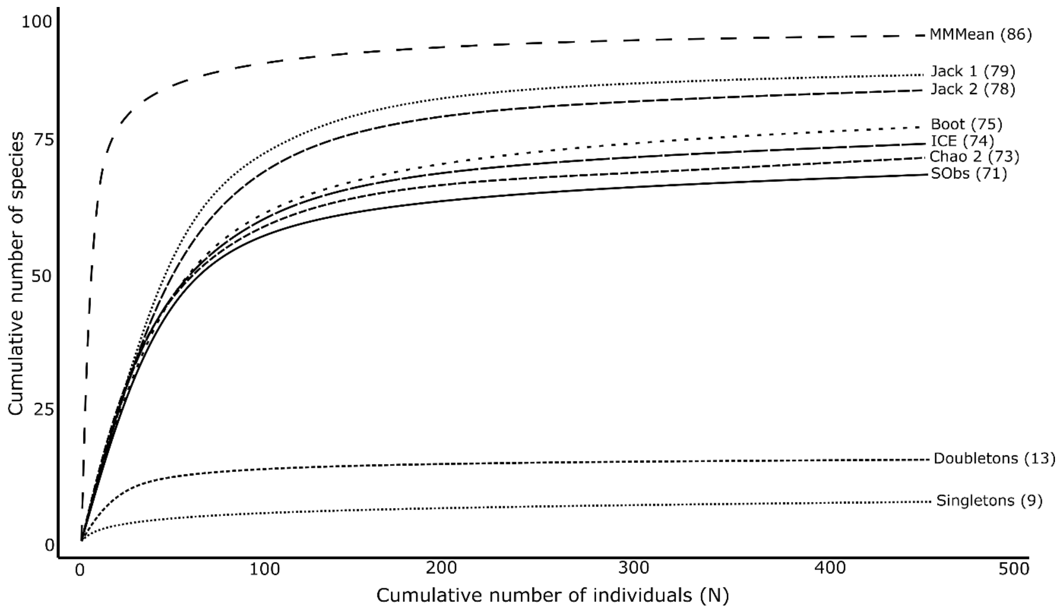

Figure 4.7. The comparative performance of six incidence-based species richness estimators (MMMeans, ICE, Jack 1, Jack 2, Bootstrap, and Chao 2), as well as the observed species accumulation curve, for all vertebrate species recorded (n = 71). The cumulative number of singletons and doubletons were also plotted. Estimated species richness is indicated in brackets. Samples were randomized 100 times.

The dataset containing records of only mammal species (Fig. 4.8) showed all estimators approaching an asymptote sooner than the observed species accumulation curve, indicating their usefulness as adequate species richness estimators (Toti et al. 2000). The Jack 1 and Jack 2 richness estimators reached an asymptote first. The remaining estimators approached asymptotes in relative parallel to the observed species curve. The difference between the highest estimator (Jack 2) and lowest estimator (ICE) was 7.0 species. The observed species richness was 85% of the highest richness estimator. Plots of singletons and doubletons rise rapidly with the first samples and then leveled off as sample size increased.

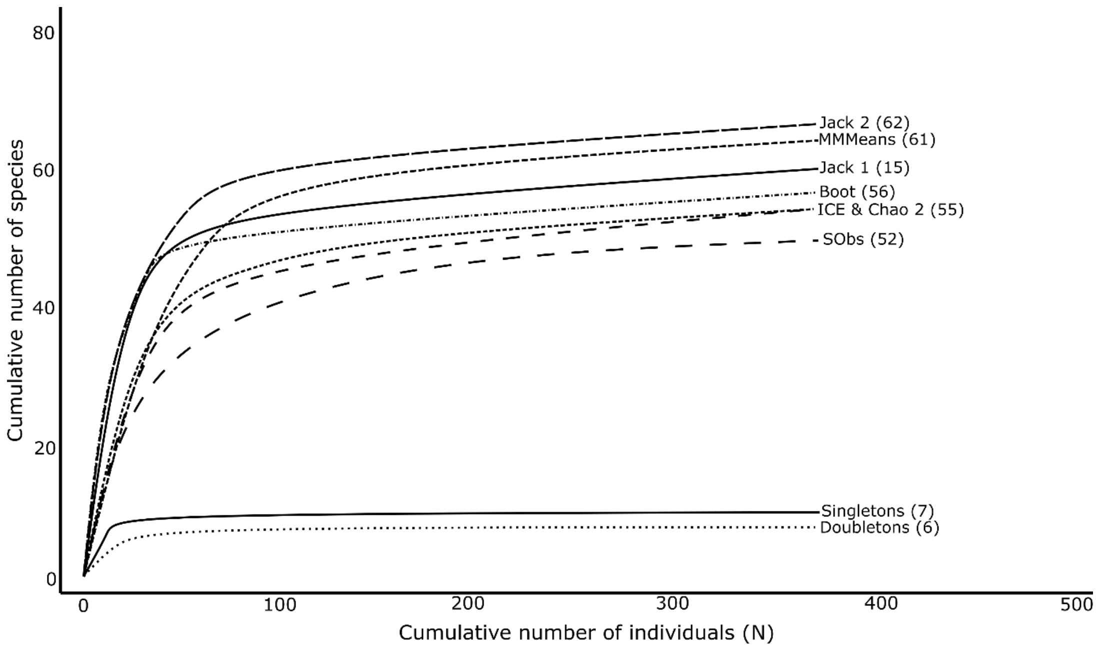

Figure 4.8. The comparative performance of six incidence-based species richness estimators (MMMeans, ICE, Jack 1, Jack 2, Bootstrap, and Chao 2), as well as the observed species accumulation curve, for all vertebrate mammal species recorded (n = 52). The cumulative number of singletons and doubletons were also plotted. Estimated species richness is indicated in brackets. Samples were randomized 100 times.

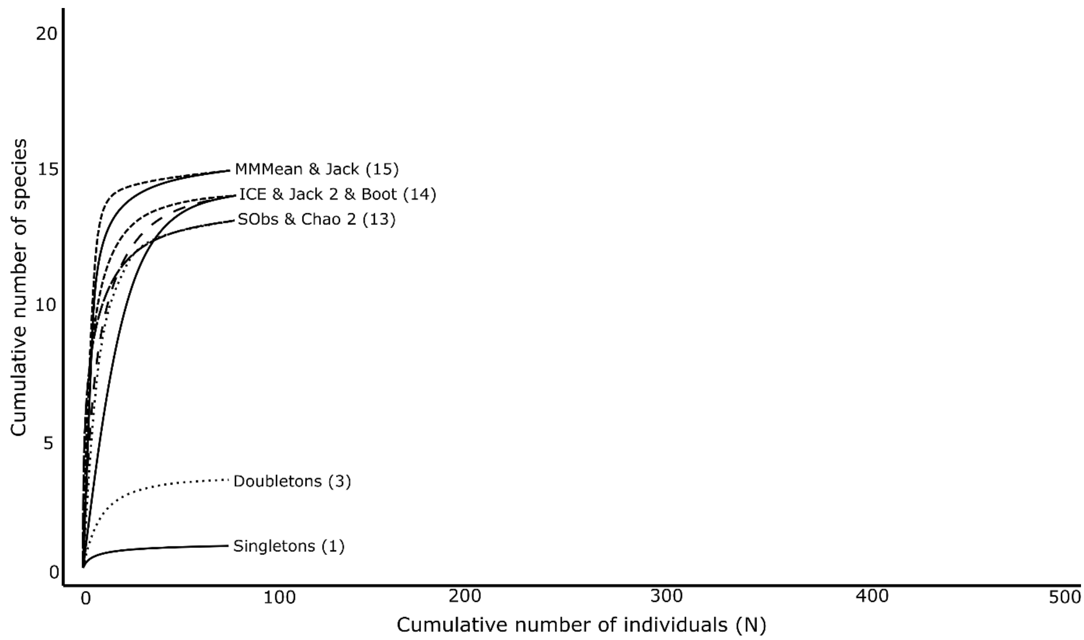

Figure 4.9. The comparative performance of six incidence-based species richness estimators (MMMeans, ICE, Jack 1, Jack 2, Bootstrap, and Chao 2), as well as the observed species accumulation curve, for all vertebrate reptile species recorded (n = 13). The cumulative number of singletons and doubletons were also plotted. Estimated species richness is indicated in brackets. Samples were randomized 100 times.

For the dataset containing records of only reptiles (Fig. 4.9), the MMMeans, Jack 1, and Bootstrap estimators appeared to reach an asymptote before the observed species richness. The difference between the highest estimator (Jack 2) and lowest estimator (ICE) was 1.3 species. The observed species richness was 87% of the highest richness estimator. Plots of singletons and doubletons rose rapidly at the first samples and then leveled off as sample size increased.

Overall, the species richness estimators predicted that a number of species have not yet been documented in the sampled communities. This was especially true for mammal species, where a greater number of singletons and doubletons were recorded compared to reptile species. The presence of singletons can be directly correlated to the number of species that remain 'unseen' (Williams et al. 2007). Based on the rate at which richness estimators reached asymptotes, the MMMeans, Jack 1 and Jack 2 estimators were the best estimators of species richness for the three sub-groups of samples. The success of a species richness estimator can further be measured by its consensus with other estimators (Toti et al. 2000). In this regard the aforementioned estimators remained the best suited for this study, as they all had relatively good consensus in the final estimated values. These estimators also provided relatively reasonable estimates close to what would be expected if the asymptote of the observed species richness curve would be extrapolated (Toti et al. 2000).

The overall species diversity recorded (Table 4.5) was relatively high (Shannon H' = 3.79; Simpson's $1/\lambda$ = 28.55), indicating high uncertainty and a relatively even distribution of individuals across species i.e. no single species dominated the market. The high diversity values also related to high levels of uncertainty, i.e. it would be difficult to correctly predict the identification of the next species found if sampling were to continue. Among taxonomic groups, mammals had comparatively higher diversity values (Shannon H' = 3.47; Simpson's $1/\lambda$ = 20.96) than reptiles (Shannon H' = 2.02; Simpson's $1/\lambda$ = 7.59) and birds (Shannon H' = 1.47; Simpson's $1/\lambda$ = 3.92), which was correlated with species richness between these groups. The low value for Fisher's alpha was expected due to the small number of species recorded compared to other ethnoecological studies (Whiting et al. 2011), especially ethnobotanical surveys (Williams et al. 2005, 2007).

Shannon J′ evenness values were high overall (range = 0.79 – 0.91), indicating that most species were evenly dispersed throughout the sampled communities, and that relatively few species were dominant. The low value for reptiles was likely due to the dominance of puff adder, monitor lizard spp., and African rock python. By comparison, E5 evenness values for all species and mammals were smaller (0.64 each), potentially caused by the dominance of leopard and baboon in the samples. Overall, the high evenness values indicated that traditional healers in the Boland Region did not necessarily narrowly favour a few specific animal species, even though some species were more popular than others. Evenness values should be interpreted with caution since scarcity of popular species during the time of sampling may have led to species being underreported.

Table 4.5. Selected measures of diversity, calculated for vertebrate taxa (together and separately) sold by traditional healers in informal settlements in the Western Cape Province.

Index/Measure	**Animals (n = 17)**	**Mammals (n = 17)**	**Reptiles (n = 16)**	**Birds (n = 10)**
Individuals	450	365	67	17
Species richness (S or N_0)	71	52	13	5
Mean S per community ± SD	18.2 ± 5.7	14.4 ± 5.0	4.12 ± 1.76	1.29 ± 1.16
Shannon-Wiener index (H′)	3.79	3.47	2.02	1.47
Simpson's ($1/\lambda$) = Hill's N_2	28.55	20.96	7.32	3.92
Hill's N1 ($e^{H'}$)	44.26	32.37	7.54	4.35
Fisher's α	23.22	16.12	4.25	2.13
E1 (Shannon J′) = (H'/H'_{max})[a]	0.89	0.88	0.79	0.91
Evenness E5 (N2-1/N1-1)	0.64	0.64	0.97	0.87
Mean number shared species	6.58 ± 3.28	5.28 ± 2.85	1.40 ± 1.02	0.22 ± 0.45
	Range: 1-14	Range: 1-13	Range: 0-4	Range: 0-2

[a]H'_{max} = ln(S)

4.4.4 Relative cultural importance (RCI) indices

4.4.4.1 Informant Consensus Factor (F_{ic})

The F_{ic} analysis identified definite variations in the degree of heterogeneity and homogeneity in the selection of different animals to treat certain conditions by traditional healers operating in informal communities (Table 4.6). The highest degree of informant consensus (i.e. high socio-cultural coherence) was for resolving court cases or reducing prison sentences (F_{ic} = 0.88) and for treating mental illnesses or improving cognitive abilities (F_{ic} = 0.85). Informants thus strongly agreed that black-backed jackal and Cape porcupine were superior animals for resolving court cases or reducing prison sentences, while mental illness or cognitive vigour was best treated with black-backed jackal, Cape fox (*Vulpes chama*), bat-eared fox (*O. megalotis*) and leopard. The lowest degree of informant consensus (F_{ic} = 0.00) was for treatments of cold or flu, headaches, kidney stones, and shingles, as well as for love charms and for protection against crime. All species mentioned by traditional healers for the treatment of these ailments were different from each other. Respondents exhibited comparatively lower consensus in the treatment of medical ailments (AverageF_{ic} = 0.31) compared to the treatment of spiritual- or magical-ailments (AverageF_{ic} = 0.36). Overall consensus for treating any ailment was however low (AverageF_{ic} = 0.34).

Table 4.6. Socio-cultural coherence in the treatment of commonly cited medical ailments within and between sampled communities, measured with the informant consensus factor (F_{ic}). N_t = number of species per use category, N_{ur} = number of citations per use category. F_{ic} values were only calculated for use categories with > 3 citations. Ailments: M = medicinal, S = spiritual/magical.

Ailment	N_t	N_{ur}	F_{ic}
Resolve court cases/reduce prison sentence (S)	2	9	0.88
Mental illness treatment/improved cognitive ability (M)	4	21	0.85
Improved muscle development/physical appearance (M)	5	14	0.69
Stroke (M)	3	7	0.67
Protect against black magic/bad muti/idliso[1]/harm (S)	6	12	0.55
Rituals/summon spirits (S)	5	8	0.50
Spiritual protection: infants (S)	3	5	0.50
Spiritual protection: individual (S)	33	62	0.48
Spiritual protection: homestead (S)	15	26	0.44

Predict future/enlightenment (S)	5	8	0.43
Acquire wealth or good fortune (S)	10	16	0.40
Fertility (M)	4	6	0.40
Protect livestock/crops (S)	4	6	0.40
Epilepsy (M)	8	12	0.36
Elicit black magic/harm/idliso[1] to others (S)	11	16	0.33
Ibekelo[2] (S)	5	7	0.33
Increased agricultural yield (S)	5	7	0.33
Paralysis (M)	4	5	0.25
Exorcism (S)	9	11	0.20
Arthritis (M)	7	8	0.14
Cold/flu (M)	4	4	0.00
Headaches (M)	4	4	0.00
Kidney stones (M)	4	4	0.00
Shingles (M)	5	5	0.00
Love charms (S)	4	4	0.00
Protect against crime (S)	4	4	0.00
Average	**6.65**	**11.19**	**0.34**

[1]Idliso (translates to '*African poison*') refers to a cultural phenomenon in which black magic is used to poison the food of others, with the intention to induce serious illness, misfortune or death.
[2]Ibekelo is a form of black magic poison that is placed on the walking path of the victim. It is believed that when the victim steps over it, he/she will become seriously ill. Illness is said to begin in the limbs, gradually moving upward.

4.4.4.2 Fidelity Level (FL) and Rank Order Priority (ROP)

The analysis revealed 17 species and morphospecies, relating to 21 uses, with high ROP values (ROP > 30; Table 4.7). Corresponding fidelity levels were highly variable, confirming the importance of including the RPL score. The majority of uses with ROP values > 30 were for individual protection against evil spirits (52.4%), followed by protecting homesteads from evil spirits (19.0%) and the use of animals for clothing or decorative purposes (19%). No medical ailments were listed, highlighting the superior importance of the spiritual component of traditional healing. The highest ROP value was assigned to monitor lizard spp. (ROP = 90.00) for individual protection against evil spirits, indicating the high level of importance placed on its use by traditional healers.

Table 4.7. Animal species and their associated uses with rank order priority (ROP) values > 30. RPL values between 0 and 1 were assigned, based on the number of times a species was cited. FL (%) = fidelity levels, I_p = number of respondents who cited the use of a species for a particular ailment, I_u = number of respondents that cited a species for any ailment.

Species	Purpose	Ip	Iu	FL (%)	RPL	ROP
Monitor lizard spp.	Spiritual protection: individual	18	20	90.00	1.00	90.00
Puff adder	Spiritual protection: individual	8	13	61.54	1.00	61.54
African rock python	Spiritual protection: individual	11	18	61.11	1.00	61.11
Cape porcupine	Spiritual protection: individual	16	28	57.14	1.00	57.14
Chacma baboon	Spiritual protection: individual	18	32	56.25	1.00	56.25
Cape clawless otter	Spiritual protection: individual	8	16	50.00	1.00	50.00
Leopard	Spiritual protection: individual	15	30	50.00	1.00	50.00
Dolphin spp.	Attract good fortune	5	9	55.56	0.75	41.67
African rock python	Spiritual protection: homestead	7	18	38.89	1.00	38.89
Genet spp.	Clothing and decoration	8	21	38.10	1.00	38.10
Genet spp.	Spiritual protection: individual	8	21	38.10	1.00	38.10
Owl spp.	Spiritual protection: homestead	7	14	50.00	0.75	37.50
Cape mountain zebra	Protection: lightning strikes	1	1	100.00	0.38	37.50
Mamba spp.	Spiritual protection: individual	6	6	100.00	0.38	37.50
Springbok	Clothing and decoration	10	11	90.91	0.38	34.09
Caracal	Spiritual protection: homestead	5	13	38.46	0.88	33.65
Leopard	Clothing and decoration	10	30	33.33	1.0	33.33
Aardvark	Spiritual protection: individual	8	9	88.89	0.38	33.33
Black-backed jackal	Spiritual protection: individual	6	19	31.58	1.00	31.58
Puff adder	Spiritual protection: homestead	4	13	30.77	1.00	30.77
Lion	Clothing and decoration	4	5	80.00	0.38	30.00

4.4.4.3 Cultural Significance Index (CSI)

The cultural significance index (CSI) values varied from 0.03 to 15 (Table 4.8). There were 22 species with CSI values > 2 (Table 4.7). Of these, 16 species were mammals (73%), four species were reptiles (18%) and two species were birds (9%); again confirming the dominance of mammal species in traditional medicine. The four species with the highest CSI values (shaded in grey) were thus ascribed to be the most valuable species to the African cultures in the Western Cape Province (Xhosa and Sotho). These were leopard (CSI = 15.00), monitor lizard spp. (CSI = 12.50), chacma baboon (CSI = 12.00), and African rock python (CSI = 11.25). Almost 44% of species (31 spp., 43.7%) had CSI values less than 1.00.

Table 4.8. The cultural significance index (CSI) values for species > 2, calculated according to the revisions by Da Silva et al. (2006). Sum = the added total scores (management x preference x frequency) for all specific uses (SU), CF = correction factor.

Genus and species	Common name	Sum (i*e*c)	CF	CSI
Panthera pardus	Leopard	16	0.94	15.00
Varanus spp.	Monitor lizard spp.	20	0.63	12.50
Papio ursinus	Chacma baboon	12	1.00	12.00
Python sebae	African rock python	20	0.56	11.25
Histrix africaeaustralis	Cape porcupine	8	0.88	7.00
Genetta spp.	Genet spp.	10	0.66	6.56
Canis mesomelas	Black-backed jackal	10	0.59	5.94
?	Owl spp.	13	0.44	5.69
Crocodylus niloticus	Nile crocodile	16	0.28	4.50
Syncerus caffer	African buffalo	12	0.38	4.50
Mellivora capensis	Honey badger	10	0.41	4.06
Lepus spp.	Hare spp.	8	0.50	4.00
Aonyx capensis	Cape clawless otter	8	0.50	4.00
Bitis arietans	Puff adder	8	0.41	3.25
Procavia capensis	Rock hyrax	8	0.34	2.75
Gyps spp.	Vulture spp.	7	0.38	2.63
Felis lybica	African wild cat	10	0.25	2.50
Caracal caracal	Caracal	6	0.41	2.44
Antidorcas marsupialis	Springbok	7	0.34	2.40
Poecilogale albinucha	African striped weasel	9	0.25	2.25
Vulpes chama	Cape fox	6	0.38	2.25
Connochaetes taurinus	Blue wildebeest	10	0.22	2.19

4.5 Discussion

The use of vertebrate species in traditional medicine is a controversial subject that has raised many concerns regarding its impact on natural wildlife (Simelane & Kerley 1998), and several statements have been made on the widespread abundance of traditional healers in South Africa (Williams et al. 2007; Petersen et al. 2012; Williams & Whiting 2016). This study not only substantiates those claims, but emphasises the previously undocumented relevance of the traditional medicinal trade in household businesses in the Boland Region of the Western Cape Province. Traditionally, ethnoecological research methodology relied on the simple compilation of species inventories, but the incorporation of quantitative analyses has the potential to enhance these studies (Höft et al. 1999). To this end, the use of a range of popular, novel and emerging quantitative techniques were employed to assess the dynamics of the traditional medicinal trade in the Boland.

Figure 4.10. (a) Skin of Cape clawless otter (*A. capensis*). (b) Fat deposits from chacma baboon (*P. ursinus*) are kept in leather armlets to prevent the carrier from being shot. (c) A travel case with assorted animal skins, most visibly skins of black-backed jackal (*C. mesomelas*), Cape hare (*L. capensis*), Nile monitor lizard (*V. niloticus*), and African rock python (*P. sebae*). (d) Cape porcupine (*H. africaeaustralis*) quills and paw. (e) Assortment of animal skins for sale, including leopard (*P. pardus*), blue wildebeest (*C. taurinus*), kudu (*T. strepsiceros*), brown hyena (*H. brunnea*), springbok (*A. marsupialis*), aardwolf (*P. cristata*), and lion (*P. leo*). Photo credits: C. Kühn (a, c-e); A. Wilkinson (b).

This study recorded 71 vertebrate species and morphospecies being used in traditional medicinal practices, constituting *c.* 4.5% of all mammal, reptile and bird fauna found in South Africa (1 488+ species, not including marine mammals). In accordance with previous ethnozoological inventories (Cunningham & Zondi 1991; Simelane & Kerley 1998; Ngwenya 2001; Whiting et al. 2011), mammals were the most prominent taxonomic group (73%, 11 orders). To a lesser extent reptiles (18%, three orders), birds (7%, five orders) and a single shark morphospecies were recorded. No amphibian or invertebrate (marine or terrestrial) species were recorded. Domestic animals are generally viewed as an unimportant source of medicine (Whiting et al. 2011), and consequently only three domestic species were recorded. The species most cited during ethnoecological inventories may however provide a false representation of the actual species abundance in the markets, especially when sampling only occurs in one season (Hoffman & Gallaher 2007). Calculating CSI values thus provided us with a means for assessing the relative importance of species to the Xhosa and Sotho cultural groups in the sampled communities, thereby shedding light on the species most at risk of overexploitation. The CSI results highlighted four species that were pivotal in Xhosa and Sotho traditional medicine, namely leopard, monitor lizard spp., chacma baboon, and African rock python.

To evaluate sampling performance and completeness of the dataset, rarefaction curves were constructed, as well as other species accumulation curves based on a variety of species richness estimators. The rarefaction curves never truly reached an asymptote, as required to deduce sufficient sampling effort (Gotelli & Colwell 2001). However, given the large area from which animals can be sourced, the relatively large study area, and the diversity of species sold in traditional medicine markets (Williams et al. 2005), it is unlikely that a true asymptote will ever be reached (Williams et al. 2007). Therefore, species richness estimators predicted that a number of species were not recorded in the study area, especially mammals, which had a greater number of singletons present (Williams et al. 2007). Based on the rate at which estimators reached an asymptote and their consensus with each other and other estimators (Toti et al. 2000), the Jack 1, Jack 2 and MMMeans estimators were considered the best estimators. The MMMeans estimator predicted that 86 species could be identified by further surveying in the sampled communities (i.e. 15 new species), while the Jack 2 and Jack 1 estimators predicted a further seven and eight species could be found, respectively (Fig. 4.7). Ethnoecological studies by Williams et al. (2007) and Whiting et al. (2011) also preferred the Jack 1 and Jack 2 estimators. Based on these estimators, the current study thus recorded 83 – 91% of the possible species in traditional medicine in the Boland Region's communities, which is more than adequate (Heck Jr et al. 1975). However, since many species were classified as morphospecies where no distinction was made between species by traditional healers, species richness is likely to be higher than reported in this study. Furthermore, since data collection occurred over a relatively short time period, some homogeneity is expected in the species recorded, and more species could therefore likely be found in subsequent studies spanning different seasons. Nevertheless, the species accumulation curves were all approaching asymptotes or had levelled-off, from which can be concluded, in conjunction with the rarefaction curves, that sampling size was sufficient and further sampling would likely not yield significantly more species.

Understanding the specific uses of species and the factors affecting medicine choices can shed light on whether or not alternative therapies can be realistically suggested (Starr et al. 2010). In this study, specific uses attributed to individual species greatly varied between species, as well as between traditional healers within and between communities. The Cape porcupine, leopard and chacma baboon were the most diversely utilised, with 16, 15, and 11 distinct uses being attributed to them, respectively. Concerns are therefore expressed towards the sustainability of these species by virtue of the number of uses attributed to them. Additionally, definite variations in the degree of heterogeneity and homogeneity in the selection of certain animals to treat certain conditions were observed. The ailments listed by traditional healers were grouped into 26 categories (Table 4.5), and the majority of these categories were found to relate to spiritual or magical ailments (58%), displaying a greater degree of socio-cultural coherence in the species used for treating these ailments as opposed to medical ailments. Overall, informant consensus for treating any ailment was however relatively low (0.34). It is unclear why these large discrepancies exist, but it undoubtedly decreases the validity of these treatments as viable solutions to medical afflictions. Beyond species inconsistencies, a plethora of animal parts and products are furthermore used for the treatment of various ailments. These include predominantly skin pieces, oils and fats, and animal bones. The low degree of socio-cultural coherence identified in this study is therefore actually much lower, and it would be of great conservation benefit to increase culturally sensitive educational programmes to promote the use of western medicine in the treatment of purely medical ailments. The majority of informants in this study were certified by traditional healer and herbalist associations, which urges them not to discourage the use of western medicine, but rather supplement its use with natural products. A dislike or antagonism towards western medicine is thus not prevalent in these communities as popularly believed. Although the demand for traditional medicine is frequently reported as growing (Hutchings 1989, 1996; Cunningham & Zondi 1991; La Cock & Briers 1992), it was evident in this study that many considered the use of traditional medicine as '*outdated*' and '*not part of their modern belief systems*'. As observed on more than one occasion, consumers of the trade travelled substantial distances to seek the help of traditional healers due to the fear of being judged or ridiculed in their respective communities. However, the majority of listed treatments are related to spiritual ailments, especially for the protection against evil spirits and demons, and can thus not realistically be scientifically screened or tested, inhibiting the ability to prescribe alternative medicines.

It would benefit conservation and educational programmes to understand the psychology underlying the selection of species for traditional medicine. For example, the highest degree of informant consensus in this study was observed for the treatment of mental health issues and improved cognitive function through the use of jackal and fox species. The use of these species is likely founded on the general cross-cultural recognition of the cunningness of these animals, and their popular portrayal as sly and clever characters in folk tales and fables. The selection of species for use in traditional medicine however depends on various anthropological, behavioural and phenotypic factors (Williams et al. 2014), and it is not always easy to deduce the thinking pattern underlying the use of certain species. Nonetheless, morphology or '*signatures*' have been suggested on numerous occasions as having a substantial influence on the species chosen for medicinal purposes (Pujol 1990; Pandita et al. 2016; Williams & Whiting 2016). This study similarly found further evidence for these trends. For example, porcupine quills were used in preparing and administering medicine because the porcupine is known to largely feed on bulbs and tubers which are often employed in ethnobotanical medicine, cattle egret (*Bubulcus ibis*) were used to guard homesteads because they are seen '*guarding*' livestock as they characteristically walk alongside them, and rinkhals (*Hemachatus haemachatus*) were used to protect crops against hail since it is believed that the characteristic hood of the cobra can cover the crops. Furthermore, since traditional medicinal practitioners believe animal traits can be transferred from animals to people, antelopes and hares were often avoided because they were viewed as scared and weak animals, while apex predators such as leopard, lion and crocodile are highly sought after to transfer properties of strength and dominance (Williams & Whiting 2016).

Identifying the prices given to products may elucidate factors relating to consumer behaviour and preferences (Dutton et al. 2011; Liu et al. 2016), as well as local purchasing power (Lindsey et al. 2011). The market value of items used and sold in the sampled community was highly variable, ranging from R15 (baboon oil) to R20 000 (leopard skin). Leopard products were highly popular, and were consistently the highest priced. Items from Cape fur seals, dolphin spp., vulture spp., aardwolf (*Proteles cristata*) and African buffalo were also abnormally expensive, likely due to the difficulty in obtaining these items. It is however worth noting that few items could be acquired for less than R500, and the majority exceeded R1 000. In some instances, traditional healers explained that expensive products and treatments are acquired in dividends, primarily divided into '*deposit*' payments and '*finalising*' payments. Others welcomed the outmoded form of bartering, for instance releasing a product in exchange for livestock (e.g. poultry, goats or cattle) or tools (e.g. grinding stones). Interestingly, one informant explains the considerable variation in prices by stating: "*…many of them [traditional healers] exploit customers to get money… for example in asking R2 000 for a genet skin… while others, like me, don't care about the money, but only the acknowledgement… gratification from a higher deity is far greater than money"* (translated from Xhosa). This sheds light on the extent to which traditional healers are motivated by financial rewards, potentially also explaining why traditional healers in the sampled communities varied in their involvement in the industry, and their reliance on the income generated. For example, a number of informants had full-time jobs in a diversity of vocations, and only practiced healing on weekends, while others practiced healing full-time. There was also great disparity in living conditions and dependency by other family members, all potentially influencing market prices. Although the motivations stated for becoming traditional healers were relatively coherent across informants (mostly through divine revelation), it is undeniable that some healers are at least partially driven by monetary requirements, fulfilled by the growing and lucrative traditional healing industry (Petersen et al. 2012). It is therefore not so farfetched to consider the possibility of '*fake traditional healers*', intent on generating income through deceitful practice. In fact, one informant stated "*we [the community] do not visit the hoaxes [fake healers], but rather travel to the Eastern Cape to see the real diviners"* (translated from Xhosa). Future policy interventions need to take these motivations into consideration, and adjust their approaches to these separate groups accordingly (Petersen et al. 2014). For many traditional healers or general traders, selling traditional medicine is not their primary choice of occupation, but necessitated by a lack of broader education and skills, as well as job scarcity (Williams 2003). Therefore, a real need exists to invest in the education of these communities to equip and enable them to pursue careers beyond their current prospects.

Overall, the species diversity in the sampled communities was relatively high, as observed in most ethnoecological studies (Begossi 1996; Williams et al. 2005, 2007; Whiting et al. 2011). Within taxonomic groups, diversity was consistent with the species richness of those groups and the number of singletons recorded in those groups. Likewise, overall evenness values were relatively high, substantiating that most species were evenly dispersed throughout our sampled communities. Reptile taxa was however dominated by a few species, namely puff adder, monitor lizard spp., and African rock python, resulting in lower evenness values than other taxonomic groups. Traditional healers therefore did not narrowly favour a few specific animal species, but utilised a plethora of different species. However, despite high evenness values, it was apparent that the species sold varied between communities. The reasons for these discrepancies remain unclear, however it is worth noting that the species sold in communities consisting of Sotho natives differed significantly from the species sold in Xhosa communities.

Twelve species of conservation concern were recorded during this study (11 mammals, 1 bird), raising concerns about their continued existence alongside the rapidly expanding traditional healing market. The majority of species enumerated that were of conservation concern however occurred in small quantities, understandably considering their presumed scarcity. Leopard, lion (*Panthera leo*), cape clawless otter, vulture species and brown hyena (*Hyaena brunnea*) were however found in high densities. Less attention is usually given to species not of immediate conservation concern. However, given the regional scarcity of many species abundant in other regions of Southern Africa (e.g. black-backed jackal, aardwolf, aardvark (*Orycteropus afer*), vulture spp. and monitor lizard spp., the ecological sustainability of the use of these animals is also questionable. Indiscriminate harvesting of these species from likely standardized locals and protected areas, coupled with increased migration patterns into the Western Cape Province (Statistics South Africa 2011), threatens conservation efforts and ultimately species survival. This study took the first step in improving the overall understanding of the requirements and demands associated with the traditional healer-belief system, but failed to identify and map the sources providing the animal material for traditional healers. Informants were generally reluctant in revealing their sources, as reported previously (Whiting et al. 2011), only admitting that material is sourced locally as far as possible, and only imported from areas such as Johannesburg, Durban and the Eastern Cape when they cannot be found locally. At least one informant reported that materials are sourced exclusively from outside of the Western Cape, and four respondents mentioned that local farm workers supply them with animal carcasses in exchange for monetary compensation. Another informant claimed to be part of a group of traditional healers that together employ the services of a local, licensed professional hunter. It seems therefore, that although large quantities of animal material is sourced from outside of the Western Cape, the market inarguably places some measure of pressure on local wildlife populations. Furthermore, this study is limited by a lack of knowledge on the actual turnover of species, since the rate at which products are sold and replaced was not obtainable. The lack of knowledge on the source populations or localities for animal harvesting, and the specific demand for products, will hinder conservation efforts and effective mitigation development, such as the identification of target areas for restoration initiatives. It would thus be of great future benefit to identify sources of animal material to incorporate these into effective policing and development frameworks that promote wildlife conservation and welfare, while still ensuring access to wildlife and wildlife uses that are deemed culturally acceptable.

4.6 Conclusion

This study extends existing knowledge on the trade of whole animals and animal-derived products in South African traditional healing practices, and provides the first quantification for the practices in the Western Cape Province. The species inventory provided here is comprehensive, and will therefore provide valuable information to inform management decision making. Considering all analyses employed in this study, we recommend future investigations and subsequent conservation interventions assign special attention to the following species: leopard, chacma baboon, Cape porcupine, monitor lizard spp., genet spp., puff adder, African rock python, and black-backed jackal. The pressures exerted on these species by traditional healing, as well as their importance in the trade, were highlighted through numerous quantitative analyses. Of these species, those that are range-restricted, threatened, habitat-specific or occurring in small population sizes (Williams & Whiting 2016) are most at risk of localised extinctions. The newly described and significant threat that traditional medicine poses to leopard populations in the Western Cape cannot be overly emphasized. In contrast to similar studies in other parts of South Africa (Simelane & Kerley 1998; Whiting et al. 2011), leopards in this study had the highest incidence among communities (82%), had the second most uses attributed to them (15 uses), were considered important in the treatment of various ailments, and were ranked first among the species most important to traditional medicine in the Xhosa and Sotho cultures. These results, coupled with the severe reduction and isolation of leopard populations during the past few decades due to habitat fragmentation, reduced prey bases and human-wildlife conflict (Nowell & Jackson 1996; Ray et al. 2005), present a real threat to their continued existence. Policymakers, conservationists and managers are now thus faced with an immense challenge, namely to further assess the dynamic trade of animals for traditional medicine, and subsequently generate policies and interventions that balances socio-economic development and cultural requirements with the diverse demands of biodiversity.

4.7 Reference list

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4.8 Appendices

Appendix 4.1. All vertebrate species (n = 71) documented in the sampled communities (n = 17), with corresponding parts and products utilised, the purpose or use of those animal constituents, and the market value of each component. NIA = No information available.

Class	Order	Genus and species	Common name	Part/product used	Purpose/Use	Price (R, Median, range)
Aves	Pelecaniformes	*Bubulcus ibis*	Western cattle egret	Entire carcass[1]	Protect home from evil spirits[1]	*NIA*
Aves	Accipitriformes	*Gyps* spp.	Vulture spp.	Skin piece[1] Feathers[2] Bone(s)[3]	Protection against evil spirits[1,2] Arthritis treatment[2,3] Remove evil spirits from home[3]	[1]1350, 1000 – 1700 [2]1 000, 700 – 1 700 [3]2 500, 700 – 3 000
Aves	Passeriformes	*Hirundo* spp.	Swallow spp.	Entire carcass	Protection against lighting Induce lightning to kill others Protection against evil spirits	1 175, 1 000 – 1 350
Aves	Strigiformes	-	Owl spp.	Oil[1] Entire carcass[2] Ash[3] Feathers[4] Subcutaneous fat[5]	Arthritis treatment[1] Kidney stones treatment[1] Protection against evil spirits[2] Remove evil spirits from home[3,4] Stroke treatment[5] Prevent material theft[5]	[1]750, 700 – 800 [2]2 000 [3]1 500 [4]700, 700 – 700 [5]800, 800 – 800
Aves	Struthioniformes	*Struthio camelus*	Common ostrich	Bone(s)	Promote physical growth Protection against evil spirits	450, 450 – 500
Chondrichthyes	Selachimorpha	-	Shark sp.	Skin piece	Protection against evil spirits	250, 200 – 300

Mammalia	Carnivora	*Acinonyx jubatus*	Cheetah	Skin piece	Transfers strength	3 000
Mammalia	Artiodactyla	*Antidorcas marsupialis*	Springbok	Horn[1] Skin piece[2]	Clothing/decoration[1,2] Rituals[1,2] Community status[2]	[1]550, 500-600 [2]600, 500 – 800
Mammalia	Carnivora	*Aonyx capensis*	Cape clawless otter	Viscera[1] Skin piece[2] Scat[3] Skull[4] Entire carcass[5]	Epilepsy treatment[1] Clothing/decoration[2] Protection against evil spirits[2,5] Increase crop yield[2,3] Cold/Flu treatment[2] Shingles treatment[2] Predict future[4] Spiritual enlightenment[4] Cause misfortune[4]	[1]1 500, 400 – 3 000 [2]700, 400 – 2 000 [3]2000 [4]1 600, 1 600 – 1 600
Mammalia	Carnivora	*Arctocephalus pusillus*	Cape fur seal	Oil[1] Skin piece[2] Bones(s)[3]	Shingles treatment[1] Protection against evil spirits[1] Attract good fortune[2] Paralysis treatment/induce[3]	[1]3 000, 2 000 – 3 000 [2]3 000, 2 500 – 3 000 [3]6 500, 5 000 – 8 000
Mammalia	Eulipotyphla	*Atelerix frontalis*	Southern African hedgehog	Quills	*NIA*	*NIA*
Mammalia	Cetacea	-	Whale spp.	Oil	*NIA*	3 000
Mammalia	Carnivora	*Canis mesomelas*	Black-backed jackal	Subcutaneous fat[1] Fur[2] Skin piece[3] Bone(s)[4] Liver[5] Oil[6]	Reduce prison sentence[1,2] Protection against evil spirits[1,3,6] Improve physical appearance[3] Resolve court cases[3,4] Cold/flu treatment[3]	[1]600, 500 – 700 [2]700 [3,4,5]1 350, 680 –2000 [6]2000 [7,8]75, 50 – 100

				Brain[7] Urine[8]	Improved cognitive ability[7,8]	
Mammalia	Artiodactyla	*Capra hircus*	Domestic goat	Entire carcass[1]	Spiritual offerings[1] Spiritual rituals[1]	*NIA*
Mammalia	Carnivora	*Caracal caracal*	Caracal	Skin piece[1] Subcutaneous fat[2] Ears[3] Heart[4] Liver[5] Bone(s)[6] Oil[7]	Remove evil spirits from home[1] Clothing/decoration[1] Cold/Flu treatment[1,6] Protection against evil spirits[2,7] Predict future[3] Create conflict[4] Epilepsy treatment[5]	[1]1 000, 200 – 2 000 [2,5]700, 700 – 700 [3]200 [4]3 000 [6]1 350, 1 000 – 1 700 [7]400, 400 - 400
Mammalia	Primates	*Chlorocebus pygerythrus*	Vervet monkey	Oil[1] Skin piece[2]	Protection against evil spirits[1,2]	[1]210, 200 – 220 [2]300
Mammalia	Artiodactyla	*Connochaetes taurinus*	Blue wildebeest	Skin piece[1] Fur[2]	Clothing/decoration[1] Epilepsy treatment[1] Remove evil spirits from home[2]	[1]600, 500 – 700 [2]350
Mammalia	Cetacea	-	Dolphin spp.	Bone(s)	Attract good fortune Improved fertility Increase livestock numbers Alcoholism cure Shingles treatment Love charm	1 900, 800 – 5 000
Mammalia	Perissodactyla	*Equus* spp.	Equine spp.	Hoof	Sports betting/gambling	2 000
Mammalia	Perissodactyla	*Equus caballus*	Domestic horse	Afterbirth	Improved fertility	600

Mammalia	Perissodactyla	*Equus zebra zebra*	Cape mountain zebra	Subcutaneous fat[1] Entire skin[2]	Prevent lightning strikes[1] Improved fertility[1] Clothing/decoration[2]	[1]750, 700 – 800 [2]3 500
Mammalia	Carnivora	*Felis catus*	Domestic cats	Entire carcass	Protect home from evil spirits	400
Mammalia	Carnivora	*Felis libyca*	African wild cat	Subcutaneous fat[1] Scat[2] Skin piece[3] Entire skin[4] Paws[5] Head[6]	Stroke treatment[1,2] Rituals[3,5,6] Cause a relationship break-up[3,5,6] Clothing/decoration[4]	[1,2]700, 700 – 700 [3,5,6]750, 750 – 750 [4]1 000
Mammalia	Carnivora	*Galerella pulverulenta*	Cape grey mongoose	Skin piece	Cause misfortune	250
Mammalia	Carnivora	*Genetta* spp.	Genet spp.	Skin piece[1] Entire carcass[2] Oil[3]	Clothing/decoration[1] Community status[1] Good luck[1] Protection against evil spirits[2,3]	[1]600, 50 – 1 500 [2]1 050, 100 – 2 000 [3]400, 400 – 400
Mammalia	Carnivora	*Herpestes ichneumon*	Egyptian mongoose	Skin piece	Protection against evil spirits	525, 500 – 550
Mammalia	Artiodactyla	*Hippopotamus amphibius*	Common hippopotamus	Subcutaneous fat	Love charm	1 150, 800 – 1 500
Mammalia	Carnivora	*Hyaena brunnea*	Brown hyena	Skin piece[1] Subcutaneous fat[2]	Clothing/decoration[1] Attract good fortune[2]	[1]900, 700 – 1 100 [2]675, 550 – 800
Mammalia	Rodentia	*Hystrix africaeaustralis*	Cape porcupine	Quills[1] Oil[2] Muscle tissue[3] Viscera[4] Bone(s)[5]	Stir medicine[1] Headache treatment[1,4] Protection against evil spirits[1,2,4,5] Remove ibekelo[1]	[1]775, 200 – 1 700 [2]800, 800 – 800 [3]1 300, 900 – 1 700 [4]600, 400 – 2 000 [5]725, 700 – 750

				Scat[6] Entire carcass[7] Skin piece[8] Stomach contents[9] Head[10]	Acupuncture[1] Rid bad blood[1] Remove idliso[1,6] Protect infants from evil spirits[1,10] Attract good fortune[3] Epilepsy treatment[4] Arthritis treatment[4] Predict future[4] Reduce prison sentence[4,6] Repel tokoloshe[6] Protection against bad muti[7,8] Stroke treatment[9]	[6]650, 400 – 1 200 [7,8]1 350, 1 000 -1700 [9]500, 400 – 600 [10]200
Mammalia	Carnivora	*Ictonyx striatus*	Striped polecat	Skin piece	Create conflict Protect from being attacked Headache treatment	700, 600 – 700
Mammalia	Lagomorpha	*Lepus* spp.	Hare spp.	Urine[1] Muscle tissue[2] Head[3] Foot[4] Viscera[5] Entire carcass[6]	Return idliso[1] Improve physical appearance[2] Epilepsy treatment[3,4,5,6] Kidney stones treatment[6] Shingles treatment[6]	[1,2]700, 700 – 700 [3,4]1 500, 1 500- 1500 [5]200 [6]500, 400 – 600
Mammalia	Proboscidea	*Loxodonta africana*	African elephant	Oil	Protection against evil spirits	325, 200 – 450
Mammalia	Carnivora	*Mellivora capensis*	Honey badger	Skin piece[1] Paws[2] Bone(s)[3]	Protection against evil spirits[1] Protection against bad muti[1,2,3]	[1]850, 500 – 1 700 [2]1 150 [3]1 350, 1 000 – 1 700

				Subcutaneous fat[4] Oil[5]	Protect infants from evil spirits[4,5]	[4,5]700, 700 – 700
Mammalia	Artiodactyla	*Oreotragus oreotragus*	Klipspringer	Skin piece	Clothing/decoration	500, 400 – 600
Mammalia	Tubulidentata	*Orycteropus afer*	Aardvark	Skin piece[1] Bone(s)[2] Oil[3]	Protection against bad muti[1,2] Remove evil spirits from home[1,3] Protection against evil spirits[2,3]	[1]1 700, 1 000 – 2 000 [2]1 300, 1 000 – 1 700 [3]500, 400 – 1 300
Mammalia	Carnivora	*Otocyon megalotis*	Bat-eared fox	Brain[1] Urine[2] Skin piece[3] Entire skin[4]	Improved cognitive ability[1,2] Remove evil spirits from home[3] Clothing/decoration[4]	[1,2]75, 50 – 100 [3]800, 800 – 800 [4]500, 200 – 800
Mammalia	Carnivora	*Panthera leo*	Lion	Liver[1] Bone(s)[2] Skin piece[3]	Poison others (idliso)[1] Paralysis treatment[2] Clothing/decoration[3]	[1]850, 850 – 850 [2]8 000 [3]950, 800 – 1 100
Mammalia	Carnivora	*Panthera pardus*	Leopard	Skin piece[1] Oil[2] Entire skin[3] Subcutaneous fat[4] Brain[5] Vertebrae[6] Nails[7] Eye[8] Bone(s)[9] Entire carcass[10] Muscle tissue[11]	Clothing/decoration[1,3] Community status[1] Protection against evil spirits[1,2,4,9] Mental illness treatment[1,2,5,6,11] Cold/flu treatment[1] Epilepsy treatment[2,4] Increased business success[2] Attract good fortune[2] Good luck[2] Kidney stones treatment[2]	[1]1 100, 250 – 6 000 [2]650, 400 – 5000 [3]11250, 5000-20 000 [4]700, 700 – 700 [5,6]4 500, 4 500 -4500 [7]3 000 [8]1 000 [9]1 700, 1 000 – 2 000 [10]11 500 [11]500, 400 – 600

					Shingles treatment[4] Arthritis treatment[7] Predict future[8] Jewellery[9] Headache treatment[11]	
Mammalia	Carnivora	*Panthera tigris*	Tiger	Skin piece	*NIA*	3 700
Mammalia	Primates	*Papio ursinus*	Chacma baboon	Subcutaneous fat[1] Paws[2] Skin piece[3] Head[4] Palm skin[5] Oil[6] Entire carcass[7] Ash[8] Bone(s)[9] Testes[10]	Acquire wealth[1,2,9] Protection against bullets[1] Fertilize crop fields[1,2,6] Protection against evil spirits[1,2,3,4,6,8,9,10] Attract good fortune[1,2,3,10] Black magic[2,5] Community status[3] Protect livestock from evil spirits[3] Clothing/decoration[3] Remove idliso[3] Poison others (idliso)[3]	[1]415, 30 – 800 [2]500, 100 – 3 000 [3]625, 300 – 1 400 [4]1 150, 800 – 1 500 [5]650, 600 – 700 [6]300, 30 – 750 [7]500, 200 – 3 000 [8]625, 500 – 750 [9]500, 400 – 600 [10]800, 800 – 800
Mammalia	Artiodactyla	*Pelea capreolus*	Grey rhebok	Skin piece[1] Horn(s)[2]	Clothing/decoration[1,2] Used as casing for powders[2]	[1]550, 400 – 800 [2]750, 700 – 800
Mammalia	Artiodactyla	*Phacochoerus africanus*	Common warthog	Skin piece	Clothing/decoration	500, 400 – 600
Mammalia	Carnivora	*Poecilogale albinucha*	African striped weasel	Skin piece[1] Paws[2] Entire carcass[3]	Protect livestock from evil spirits[1] Unify livestock[2] Protection against evil spirits[3]	[1]3 000 [2]1 200 [3]1 050, 100 – 2 000

Mammalia	Artiodactyla	*Potamochoerus larvatus*	Bushpig	Skin piece	Protection against evil spirits	725, 700 - 750
Mammalia	Hyracoidea	*Procavia capensis*	Rock hyrax	Paws[1] Viscera[2] Skin piece[3] Fur[4]	Promote physical growth[1] Fertility treatment[2] Increase livestock fertility[3,4] Manipulate sex of stock offspring[3]	[1]675, 650 – 700 [2]800, 800 – 800 [3]2 000, 2 000 – 2 000 [4]700
Mammalia	Carnivora	*Proteles cristata*	Aardwolf	Entire skin[1] Skin piece[2] [3]Bone(s)	Clothing/decoration[1] Attract good fortune[2,3]	[1]4 000, 2 000 – 6 000 [2,3]1 350, 1 000 -1700
Mammalia	Artiodactyla	*Raphicerus melanotis*	Cape grysbok	Skin piece	Clothing/decoration Prevents road accidents	500, 400 – 600
Mammalia	Perissodactyla	*Ceratotherium/ Diceros sp.*	Rhinoceros spp.	Horn	Attract good fortune	*NIA*
Mammalia	Artiodactyla	*Sus scrofa*	Feral pig	Bone(s)	Arthritis treatment	200
Mammalia	Artiodactyla	*Sylvicapra grimmia*	Common duiker	Skull[1] Skin piece[2] Horn(s)[3] Hoof[4]	Invite spiritual presence (rituals)[1] Clothing/decoration[2] Jewellery[3] Protection against evil spirits[4]	[1]2000 [2]500, 400 – 600 [3,4]700, 700 – 700
Mammalia	Artiodactyla	*Syncerus caffer*	African buffalo	Subcutaneous fat[1] Bone(s)[2] Horn(s)[3]	Protect home against evil spirits[1,3] Protection against evil spirits[1] Paralysis treatment[2] Exorcism[3]	[1]750, 700 – 800 [2]8 250, 8 000 – 8 500 [3]1 750, 1 500 – 2 000
Mammalia	Artiodactyla	*Taurotragus oryx*	Common eland	Horn(s)	Exorcism	500, 400 – 600

Mammalia	Artiodactyla	*Taurotragus* spp. & *Tragelaphus* spp.	Spiral-horned antelope spp.	Horn(s)	Protect home against evil spirits Exorcism	600, 400 – 800
Mammalia	Artiodactyla	*Tragelaphus angasii*	Nyala	Horn(s)	Exorcism	500, 400 – 600
Mammalia	Artiodactyla	*Tragelaphus strepsiceros*	Greater kudu	Horn(s)	Exorcism Clothing/decoration	700, 400 – 800
Mammalia	Artiodactyla	*Tragelaphus sylvaticus*	Cape bushbuck	Skull[1] Horn(s)	Invite spiritual presence (rituals)[1] Exorcism[2]	[1]2000 [2]500, 400 – 600
Mammalia	Carnivora	*Vulpes chama*	Cape fox	Entire skin[1] Brain[2] Urine[3]	Clothing/decoration[1] Improved cognitive ability[2,3] Headache treatment[3]	[1]500, 500 – 600 [2,3]75, 50 – 100
Reptilia	Squamata	*Agama atra*	Southern rock agama	Entire carcass[1] Entire skin[2]	Protection against evil spirits[1,2]	[1]900 [2]700
Reptilia	Squamata	*Bitis arietans*	Puff adder	Bone(s)[1] Oil[2] Skin piece[3] Head[4] Tail[5] Entire carcass[6]	Exorcism[1] Protection against evil spirits[2,3,4,5] Arthritis treatment[3] Protect livestock from evil spirits[6] Send evil spirits to kill livestock[6] Protect home from evil spirits[6] Remove evil spirits from home[6]	[1]1 400, 800 – 2 000 [2]300, 200 – 400 [3]275, 150 – 400 [4,5]2000 [6]2 000, 400 – 3 000

					Remove ibekelo[6]	
Reptilia	Testudines	*Chersina angulata*	Angulate tortoise	Blood[1] Entire carcass[2] Carapace[3]	Epilepsy treatment[1] Protection against evil spirits[2]	[1]800 [2]1 000 [3]400
Reptilia	Squamata	*Cordylus* spp.	Girdled lizard spp.	Entire carcass	*NIA*	*NIA*
Reptilia	Crocodilia	*Crocodylus niloticus*	Nile crocodile	Bone(s)[1] Vertebrae[2] Subcutaneous fat[3]	Protect home from evil spirits[1,2] Protection against evil spirits[3]	[1]500, 500 – 500 [2,3]800, 800 – 800
Reptilia	Squamata	*Dendroaspis* spp.	Mamba spp.	Skin piece[1] Bone(s)[2] Subcutaneous fat[3] Entire carcass[4]	Protection against evil spirits[1,2] Remove evil spirits from home[1,3] Exorcism[4]	[1]825, 700 – 950 [2]1 050, 100 – 2 000 [3]700 [4]800
Reptilia	Squamata	*Hemachatus haemachatus*	Rinkhals	Entire carcass[1] Ash[2]	Protect crops from hail damage[1] Protection against evil spirits[2]	[1,2]2 000, 2 000- 2000
Reptilia	Squamata	*Naja nivea*	Cape cobra	Skin piece[1] Head[2] Tail[3]	Remove evil spirits from home[1] Protection against evil spirits[1,2,3]	[1]450, 150 – 750 [2,3]2 000, 2 000 -2000
Reptilia	Squamata	*Pseudaspis cana*	Mole snake	Skin piece	Protection against evil spirits	600
Reptilia	Squamata	*Python sebae*	African rock python	Skin piece[1] Bone(s)[2] Entire carcass[3]	Protection from ibekelo[1] Protection against evil spirits[1] Remove evil spirits from home[1]	[1]700, 600 – 700 [2]2 600 [3]800, 800 – 800

					Remove ibekelo[2] Exorcism[3]	
Reptilia	Squamata	-	Snake spp.	Entire carcass	Protection against evil spirits	1 000, 600 – 1 700
Reptilia	Testudines	*Stigmochelys pardalis*	Leopard tortoise	Scat	Epilepsy treatment	500
Reptilia	Squamata	*Varanus genus*	Monitor lizard spp.	Muscle tissue[1] Skin piece[2] Paws[3] Oil[4]	Stop bad dreams from occurring[1] Idliso treatment[1] Protection against evil spirits[1,2,3] Rid bad luck[2,4] Prevent a miscarriage[2] Arthritis treatment[2] Kidney stones treatment[2]	[1]400, 400 – 400 [2]700, 50 – 800 [3]800, 800 – 800 [4]15

Appendix 4.2. Animal species and their associated uses with rank order priority (ROP) values. RPL values were assigned between 0 and 1, based on the number of times a species was cited. FL (%) = fidelity levels, Ip = number of respondents who cited the use of a species for a particular ailment, Iu = number of respondents that cited a species for any ailment. Only uses that were cited > 4 times independently are listed.

Purpose	Species	Ip	Iu	FL (%)	RPL	ROP
Acquire wealth or good fortune (S)	Dolphin spp.	5	9	55.56	0.75	41.67
Acquire wealth or good fortune (S)	Chacma baboon	7	32	21.88	1.00	21.88
Acquire wealth or good fortune (S)	Cape porcupine	5	28	17.86	1.00	17.86
Acquire wealth or good fortune (S)	Cape fur seal	3	9	33.33	0.50	16.67
Acquire wealth or good fortune (S)	Genet spp.	3	21	14.29	1.00	14.29
Acquire wealth or good fortune (S)	Leopard	4	30	13.33	1.00	13.33
Acquire wealth or good fortune (S)	Rhinoceros sp.	1	1	100.00	0.13	12.50
Acquire wealth or good fortune (S)	Brown hyena	3	7	42.86	0.25	10.71
Acquire wealth or good fortune (S)	Aardwolf	2	7	28.57	0.25	7.14
Acquire wealth or good fortune (S)	Leopard	2	30	6.67	1.00	6.67
Arthritis treatment (M)	Puff adder	2	13	15.38	1.00	15.38
Arthritis treatment (M)	Monitor lizard spp.	3	20	15.00	1.00	15.00
Arthritis treatment (M)	Feral pig	1	1	100.00	0.13	12.50
Arthritis treatment (M)	Owl spp.	2	14	14.29	0.75	10.71
Arthritis treatment (M)	Cape porcupine	2	28	7.14	1.00	7.14
Arthritis treatment (M)	Vulture spp.	2	12	16.67	0.38	6.25
Arthritis treatment (M)	Leopard	1	30	3.33	1.00	3.33
Clothing/decoration/jewellery	Genet spp.	8	21	38.10	1.00	38.10
Clothing/decoration/jewellery	Cape mountain zebra	1	1	100.00	0.38	37.50
Clothing/decoration/jewellery	Springbok	10	11	90.91	0.38	34.09
Clothing/decoration/jewellery	Leopard	10	30	33.33	1.00	33.33
Clothing/decoration/jewellery	Lion	4	5	80.00	0.38	30.00
Clothing/decoration/jewellery	Caracal	4	13	30.77	0.88	26.92
Clothing/decoration/jewellery	Blue wildebeest	5	7	71.43	0.38	26.78
Clothing/decoration/jewellery	African wild cat	4	8	50.00	0.50	25.00
Clothing/decoration/jewellery	Bat-eared fox	5	9	55.56	0.38	20.83
Clothing/decoration/jewellery	Grey rhebok	5	6	83.33	0.25	20.83
Clothing/decoration/jewellery	Common duiker	4	10	40.00	0.50	20.00
Clothing/decoration/jewellery	Greater kudu	7	9	77.78	0.25	19.44
Clothing/decoration/jewellery	Brown hyena	5	7	71.43	0.25	17.86
Clothing/decoration/jewellery	Cape grysbok	4	6	66.67	0.25	16.67
Clothing/decoration/jewellery	Cape fox	5	12	41.67	0.38	15.63
Clothing/decoration/jewellery	Chacma baboon	5	32	15.63	1.00	15.63
Clothing/decoration/jewellery	Aardwolf	4	7	57.14	0.25	14.29

Clothing/decoration/jewellery	Klipspringer	1	1	100.00	0.13	12.50
Clothing/decoration/jewellery	Common warthog	3	5	60.00	0.13	7.50
Clothing/decoration/jewellery	Cape clawless otter	1	16	6.25	1.00	6.25
Clothing/decoration/jewellery	Common duiker	1	10	10.00	0.50	5.00
Clothing/decoration/jewellery	Leopard	1	30	3.33	1.00	3.33
Cold/flu treatment (M)	Caracal	2	13	15.38	0.88	13.46
Cold/flu treatment (M)	Cape clawless otter	1	16	6.25	1.00	6.25
Cold/flu treatment (M)	Black-backed jackal	1	19	5.26	1.00	5.26
Cold/flu treatment (M)	Leopard	1	30	3.33	1.00	3.33
Community status	Genet spp.	3	21	14.29	1.00	14.29
Community status	Leopard	2	30	6.67	1.00	6.67
Community status	Springbok	1	11	9.09	0.38	3.41
Community status	Chacma baboon	1	32	3.13	1.00	3.13
Elicit harm/idliso[1] to others (S)	African wild cat	3	8	37.50	0.50	18.75
Elicit harm/idliso[1] to others (S)	Striped polecat	2	6	33.33	0.38	12.50
Elicit harm/idliso[1] to others (S)	Swallow spp.	1	3	33.33	0.38	12.50
Elicit harm/idliso[1] to others (S)	Cape fur seal	2	9	22.22	0.50	11.11
Elicit harm/idliso[1] to others (S)	Puff adder	1	13	7.69	1.00	7.69
Elicit harm/idliso[1] to others (S)	Lion	1	5	20.00	0.38	7.50
Elicit harm/idliso[1] to others (S)	Caracal	1	13	7.69	0.88	6.73
Elicit harm/idliso[1] to others (S)	Cape clawless otter	1	16	6.25	1.00	6.25
Elicit harm/idliso[1] to others (S)	Chacma baboon	2	32	6.25	1.00	6.25
Elicit harm/idliso[1] to others (S)	Hare spp.	1	16	6.25	1.00	6.25
Elicit harm/idliso[1] to others (S)	Cape grey mongoose	1	3	33.33	0.13	4.17
Epilepsy treatment (M)	Hare spp.	4	16	25.00	1.00	25.00
Epilepsy treatment (M)	Cape clawless otter	2	16	12.50	1.00	12.50
Epilepsy treatment (M)	Leopard tortoise	1	1	100.00	0.13	12.50
Epilepsy treatment (M)	Angulate tortoise	2	5	40.00	0.25	10.00
Epilepsy treatment (M)	Cape porcupine	2	28	7.14	1.00	7.14
Epilepsy treatment (M)	Caracal	1	13	7.69	0.88	6.73
Epilepsy treatment (M)	Blue wildebeest	1	7	14.29	0.38	5.36
Epilepsy treatment (M)	Leopard	1	30	3.33	1.00	3.33
Exorcism (S)	African rock python	5	18	27.78	1.00	27.78
Exorcism (S)	Bushbuck	2	3	66.67	0.25	16.67
Exorcism (S)	Puff adder	2	13	15.38	1.00	15.38
Exorcism (S)	African buffalo	3	12	25.00	0.50	12.50
Exorcism (S)	Common eland	2	2	100.00	0.13	12.50

Exorcism (S)	Nyala	2	2	100.00	0.13	12.50
Exorcism (S)	Mamba spp.	2	6	33.33	0.38	12.50
Exorcism (S)	Spiral-horned antelope spp.	3	7	42.86	0.25	10.71
Exorcism (S)	Greater kudu	3	9	33.33	0.25	8.33
Fertility treatment (M)	Cape mountain zebra	1	1	100.00	0.38	37.50
Fertility treatment (M)	Rock hyrax	5	11	45.45	0.50	22.73
Fertility treatment (M)	Dolphin spp.	1	9	11.11	0.75	8.33
Fertility treatment (M)	Domestic horse	2	3	66.67	0.13	8.33
Headache treatment (M)	Striped polecat	2	6	33.33	0.38	12.50
Headache treatment (M)	Cape porcupine	2	28	7.14	1.00	7.14
Headache treatment (M)	Cape fox	2	12	16.67	0.38	6.25
Headache treatment (M)	Leopard	1	30	3.33	1.00	3.33
Ibekelo[2] (S)	Puff adder	1	13	7.69	1.00	7.69
Ibekelo[2] (S)	Cape porcupine	2	28	7.14	1.00	7.14
Ibekelo[2] (S)	Chacma baboon	2	32	6.25	1.00	6.25
Ibekelo[2] (S)	African rock python	1	18	5.56	1.00	5.56
Ibekelo[2] (S)	Cape porcupine	1	28	3.57	1.00	3.57
Improved physical appearance (M)	Cheetah	1	1	100.00	0.13	12.50
Improved physical appearance (M)	Hare spp.	2	16	12.50	1.00	12.50
Improved physical appearance (M)	Rock hyrax	2	11	18.18	0.50	9.09
Improved physical appearance (M)	Common ostrich	1	3	33.33	0.25	8.33
Improved physical appearance (M)	Black-backed jackal	1	19	5.26	1.00	5.26
Increased agricultural yield (S)	Cape clawless otter	2	16	12.50	1.00	12.50
Increased agricultural yield (S)	Chacma baboon	3	32	9.38	1.00	9.38
Increased agricultural yield (S)	Rock hyrax	2	11	18.18	0.50	9.09
Increased agricultural yield (S)	Dolphin spp.	1	9	11.11	0.75	8.33
Increased agricultural yield (S)	Striped weasel	1	8	12.50	0.38	4.69
Kidney stones treatment (M)	Hare spp.	2	16	12.50	1.00	12.50
Kidney stones treatment (M)	Owl spp.	1	14	7.14	0.75	5.36
Kidney stones treatment (M)	Monitor lizard spp.	1	20	5.00	1.00	5.00
Kidney stones treatment (M)	Leopard	1	30	3.33	1.00	3.33
Love charm (S)	Hippopotamus	2	2	100.00	0.13	12.50
Love charm (S)	Dolphin spp.	1	9	11.11	0.75	8.33
Love charm (S)	African buffalo	1	12	8.33	0.50	4.17

Love charm (S)	African elephant	1	3	33.33	0.13	4.17
Mental illness treatment/ improved cognitive ability (M)	Black-backed jackal	5	19	26.32	1.00	26.32
Mental illness treatment/ improved cognitive ability (M)	Leopard	4	30	13.33	1.00	13.33
Mental illness treatment/ improved cognitive ability (M)	Bat-eared fox	3	9	33.33	0.38	12.50
Mental illness treatment/ improved cognitive ability (M)	Cape fox	4	12	33.33	0.38	12.50
Paralysis treatment (M)	Lion	2	5	40.00	0.38	15.00
Paralysis treatment (M)	Cape fur seal	2	9	22.22	0.50	11.11
Paralysis treatment (M)	Puff adder	1	13	7.69	1.00	7.69
Paralysis treatment (M)	African buffalo	1	12	8.33	0.50	4.17
Predict future/enlightenment (S)	Cape clawless otter	2	16	12.50	1.00	12.50
Predict future/enlightenment (S)	Cape porcupine	2	28	7.14	1.00	7.14
Predict future/enlightenment (S)	Caracal	1	13	7.69	0.88	6.73
Predict future/enlightenment (S)	Leopard	2	30	6.67	1.00	6.67
Protect against black magic/bad muti/idliso[1]/harm (S)	Cape mountain zebra	1	1	100.00	0.38	37.50
Protect against black magic/bad muti/idliso[1]/harm (S)	Monitor lizard spp.	3	20	15.00	1.00	15.00
Protect against black magic/bad muti/idliso[1]/harm (S)	Swallow spp.	1	3	33.33	0.38	12.50
Protect against black magic/bad muti/idliso[1]/harm (S)	Cape porcupine	3	28	10.71	1.00	10.71
Protect against black magic/bad muti/idliso[1]/harm (S)	Honey badger	3	13	23.08	0.38	8.65
Protect against black magic/bad muti/idliso[1]/harm (S)	Aardvark	2	9	22.22	0.38	8.33
Protection against crime (S)	Striped polecat	3	6	50.00	0.38	18.75
Protection against crime (S)	Owl spp.	1	14	7.14	0.75	5.36
Protection against crime (S)	Cape grysbok	1	6	16.67	0.25	4.17
Protection against crime (S)	Chacma baboon	1	32	3.13	1.00	3.13
Resolve court cases/ reduce prison sentence (S)	Black-backed jackal	5	19	26.32	1.00	26.32
Resolve court cases/	Cape porcupine	1	28	3.57	1.00	3.57

reduce prison sentence (S)						
Rituals/summon spirits (S)	Domestic goat	1	1	100.00	0.25	25.00
Rituals/summon spirits (S)	African wild cat	3	8	37.50	0.50	18.75
Rituals/summon spirits (S)	Bushbuck	1	3	33.33	0.25	8.33
Rituals/summon spirits (S)	Springbok	2	11	18.18	0.38	6.82
Rituals/summon spirits (S)	Common duiker	1	10	10.00	0.50	5.00
Shingles treatment (M)	Dolphin spp.	1	9	11.11	0.75	8.33
Shingles treatment (M)	Cape clawless otter	1	16	6.25	1.00	6.25
Shingles treatment (M)	Hare spp.	1	16	6.25	1.00	6.25
Shingles treatment (M)	Cape fur seal	1	9	11.11	0.50	5.56
Shingles treatment (M)	Leopard	1	30	3.33	1.00	3.33
Spiritual protection: agriculture (S)	Puff adder	2	13	15.38	1.00	15.38
Spiritual protection: agriculture (S)	Striped weasel	3	8	37.50	0.38	14.06
Spiritual protection: agriculture (S)	Rinkhals	1	2	50.00	0.25	12.50
Spiritual protection: agriculture (S)	Chacma baboon	2	32	6.25	1.00	6.25
Spiritual protection: homestead (S)	African rock python	7	18	38.89	1.00	38.89
Spiritual protection: homestead (S)	Owl spp.	7	14	50.00	0.75	37.50
Spiritual protection: homestead (S)	Caracal	5	13	38.46	0.88	33.66
Spiritual protection: homestead (S)	Puff adder	4	13	30.77	1.00	30.77
Spiritual protection: homestead (S)	Blue wildebeest	4	7	57.14	0.38	21.43
Spiritual protection: homestead (S)	Mamba spp.	3	6	50.00	0.38	18.75
Spiritual protection: homestead (S)	Aardvark	4	9	44.44	0.38	16.67
Spiritual protection: homestead (S)	African buffalo	4	12	33.33	0.50	16.67
Spiritual protection: homestead (S)	Bat-eared fox	4	9	44.44	0.38	16.67
Spiritual protection: homestead (S)	Spiral-horned antelope spp.	4	7	57.14	0.25	14.29
Spiritual protection: homestead (S)	Nile crocodile	5	9	55.56	0.25	13.89
Spiritual protection: homestead (S)	Domestic cat	1	1	100.00	0.13	12.50
Spiritual protection: homestead (S)	Vulture spp.	4	12	33.33	0.38	12.50
Spiritual protection: homestead (S)	Cape cobra	6	13	46.15	0.25	11.54
Spiritual protection: homestead (S)	Western cattle egret	1	2	50.00	0.13	6.25
Spiritual protection: individual (S)	Monitor lizard spp.	18	20	90.00	1.00	90.00
Spiritual protection: individual (S)	Puff adder	8	13	61.54	1.00	61.54
Spiritual protection: individual (S)	African rock python	11	18	61.11	1.00	61.11
Spiritual protection: individual (S)	Cape porcupine	16	28	57.14	1.00	57.14
Spiritual protection: individual (S)	Chacma baboon	18	32	56.25	1.00	56.25
Spiritual protection: individual (S)	Cape clawless otter	8	16	50.00	1.00	50.00

Spiritual protection: individual (S)	Leopard	15	30	50.00	1.00	50.00
Spiritual protection: individual (S)	Genet spp.	8	21	38.10	1.00	38.10
Spiritual protection: individual (S)	Mamba spp.	6	6	100.00	0.38	37.50
Spiritual protection: individual (S)	Aardvark	8	9	88.89	0.38	33.33
Spiritual protection: individual (S)	Black-backed jackal	6	19	31.58	1.00	31.58
Spiritual protection: individual (S)	African buffalo	7	12	58.33	0.50	29.17
Spiritual protection: individual (S)	Striped weasel	6	8	75.00	0.38	28.13
Spiritual protection: individual (S)	Common duiker	5	10	50.00	0.50	25.00
Spiritual protection: individual (S)	Rinkhals	2	2	100.00	0.25	25.00
Spiritual protection: individual (S)	Swallow spp.	2	3	66.67	0.38	25.00
Spiritual protection: individual (S)	Vulture spp.	8	12	66.67	0.38	25.00
Spiritual protection: individual (S)	Cape fur seal	4	9	44.44	0.50	22.22
Spiritual protection: individual (S)	Owl spp.	4	14	28.57	0.75	21.43
Spiritual protection: individual (S)	Caracal	3	13	23.08	0.88	20.19
Spiritual protection: individual (S)	Honey badger	7	13	53.85	0.38	20.19
Spiritual protection: individual (S)	Nile crocodile	7	9	77.78	0.25	19.44
Spiritual protection: individual (S)	Common ostrich	2	3	66.67	0.25	16.67
Spiritual protection: individual (S)	Cape cobra	8	13	61.54	0.25	15.38
Spiritual protection: individual (S)	African elephant	3	3	100.00	0.13	12.50
Spiritual protection: individual (S)	Mole snake	1	1	100.00	0.13	12.50
Spiritual protection: individual (S)	Snake spp.	3	3	100.00	0.13	12.50
Spiritual protection: individual (S)	Angulate tortoise	2	5	40.00	0.25	10.00
Spiritual protection: individual (S)	Bushpig	3	4	75.00	0.13	9.38
Spiritual protection: individual (S)	Egyptian mongoose	3	4	75.00	0.13	9.38
Spiritual protection: individual (S)	Southern rock agama	2	3	66.67	0.13	8.33
Spiritual protection: individual (S)	Vervet monkey	2	3	66.67	0.13	8.33
Spiritual protection: individual (S)	Shark spp.	1	2	50.00	0.13	6.25
Spiritual protection: infants (S)	Honey badger	5	13	38.46	0.38	14.42
Spiritual protection: infants (S)	Cape porcupine	4	28	14.29	1.00	14.29
Spiritual protection: infants (S)	Puff adder	1	13	7.69	1.00	7.69
Stroke treatment (M)	African wild cat	2	8	25.00	0.50	12.50
Stroke treatment (M)	Owl spp.	1	14	7.14	0.75	5.36
Stroke treatment (M)	Cape porcupine	1	28	3.57	1.00	3.57

[1]Idliso (translates to '*African poison*') refers to a cultural phenomenon in which black magic is used to poison the food of others, with the intention to induce serious illness, misfortune or death.
[2]Ibekelo is a form of black magic poison that is placed on the walking path of the victim. It is believed that when the victim steps over it, he/she will become seriously ill. Illness is said to begin in the limbs, gradually moving upward.

Appendix 4.3. The relative importance of all species and morphospecies recorded in the sampled communities to the African cultures (Xhosa and Sotho) surveyed in the Western Cape Province, ranked according to the cultural significance index (CSI). Sum = sum of values assigned for species use intensity, exclusivity of use and quality of use, CF = correction factor.

Genus and species	**Common name**	**Sum (i*e*c)**	**CF**	**CSI**
Panthera pardus	Leopard	16	0.94	15.00
Varanus spp.	Monitor lizard spp.	20	0.63	12.50
Papio ursinus	Chacma baboon	12	1.00	12.00
Python sebae	African rock python	20	0.56	11.25
Histrix africaeaustralis	Cape porcupine	8	0.88	7.00
Genetta spp.	Genet spp.	10	0.66	6.56
Canis mesomelas	Black-backed jackal	10	0.59	5.94
?	Owl spp.	13	0.44	5.69
Crocodylus niloticus	Nile crocodile	16	0.28	4.50
Syncerus caffer	African buffalo	12	0.38	4.50
Mellivora capensis	Honey badger	10	0.41	4.06
Lepus spp.	Hare spp.	8	0.50	4.00
Aonyx capensis	Cape clawless otter	8	0.50	4.00
Bitis arietans	Puff adder	8	0.41	3.25
Procavia capensis	Rock hyrax	8	0.34	2.75
Gyps spp.	Vulture spp.	7	0.38	2.63
Felis lybica	African wild cat	10	0.25	2.50
Caracal caracal	Caracal	6	0.41	2.44
Antidorcas marsupialis	Springbok	7	0.34	2.40
Poecilogale albinucha	African striped weasel	9	0.25	2.25
Vulpes chama	Cape fox	6	0.38	2.25
Connochaetes taurinus	Blue wildebeest	10	0.22	2.19
Arctocephalus pusillus	Cape fur seal	7	0.28	1.97
Sylvicapra grimmia	Common duiker	6	0.31	1.90
Panthera leo	Lion	12	0.16	1.88
Hyaena brunnea	Brown hyena	8	0.22	1.75
Otocyon megalotis	Bat-eared fox	6	0.28	1.69
?	Dolphin spp.	6	0.28	1.67
Tragelaphus sylvaticus	Greater kudu	6	0.28	1.67
Naja nivea	Cape cobra	4	0.41	1.63
Ictonyx striatus	Striped polecat	8	0.19	1.50
Orycteropus afer	Aardvark	5	0.28	1.40
Taurotragus/tragelaphus spp.	Spiral-horned antelope	6	0.22	1.31
Dendroaspis spp.	Mamba spp.	6	0.19	1.13
Pelea capreolus	Grey rhebok	4	0.19	0.75
Raphicerus melanotis	Cape grysbok	4	0.19	0.75

Proteles cristata	Aardwolf	3	0.22	0.66
Phacochoerus africanus	Warthog	4	0.16	0.63
Chersina angulata	Angulate tortoise	4	0.16	0.63
Tragelaphus sylvaticus	Bushbuck	6	0.09	0.56
Potamochoerus larvastus	Bushpig	4	0.13	0.50
Hirundo spp.	Swallow spp.	4	0.09	0.38
Loxodonta africana	Elephant	4	0.09	0.38
Herpestes ichneumon	Egyptian mongoose	2	0.13	0.25
Hippopotamus amphibius	Hippopotamus	4	0.06	0.25
Hemachatus haemachatus	Rinkhals	3	0.06	0.19
?	Shark spp.	3	0.06	0.19
Chlorocebus pygerythrus	Vervet monkey	2	0.09	0.19
Capra hircus	Domestic goat	6	0.03	0.19
Equus caballus	Domestic horse	2	0.09	0.19
Struthio camelus	Ostrich	2	0.09	0.19
Galerella pulverulenta	Cape grey mongoose	2	0.09	0.19
Equus spp.	Equine spp.	2	0.06	0.13
Tragelaphus angasii	Nyala	2	0.06	0.13
Equus zebra zebra	Cape mountain zebra	4	0.03	0.13
Acinonyx jubatus	Cheetah	4	0.03	0.13
Agama atra	Southern rock agama	1	0.09	0.09
Ceratotherium/Diceros sp.	Rhinoceros	2	0.03	0.06
Bubulcus ibis	Western cattle egret	1	0.06	0.06
Taurotragus oryx	Eland	1	0.06	0.06
Cordylus spp.	Girdled lizard spp.	1	0.06	0.06
Stigmochelys pardalis	Leopard tortoise	2	0.03	0.06
Felis catus	Domestic cat	1	0.03	0.03
Sus scrofa	Feral pig	1	0.03	0.03
Oreotragus oreotragus	Klipspringer	1	0.03	0.03
Pseudaspis cana	Mole snake	1	0.03	0.03

Chapter 5
A Synergistic Assessment of Animal Species' Vulnerability to Three Major Hunting Practices in the Western Cape Province

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5.1 Abstract

Three major forms of hunting are on the increase in the Western Cape Province of South Africa, and independently and synergistically pose some of the greatest threats to the continued survival of local vertebrate species. A newly devised Rapid Vulnerability Assessment (RVA) method that incorporates animal species, is proposed to prioritise species most at risk of being threatened or overexploited by a combination of subsistence poaching for bushmeat, retaliatory or preventative hunting stemming from human-wildlife conflict, and the harvesting of animals for use in traditional medicine. In this study, new elements and reformulated variables and scales were introduced to the existing RVA-method designated for botanical species. The results revealed eight species under high and very high potential endangerment, and a further 22 species under moderate potential endangerment. The reliability of the adapted RVA-test was statistically assessed. Variables were found to be independent and poorly correlated, suggesting that all variables contributed significantly to the final threat scores. The RVA-method proposed here is thus considered suitable for a rough prioritization of species endangerment in the Boland Region of the Western Cape.

Keywords: Bushmeat poaching, human-wildlife conflict, South Africa, traditional medicine, vulnerability assessment, Western Cape Province, wildlife off-take

5.2 Introduction

Growing human populations have an increasingly detrimental impact on natural resources and associated ecosystem dynamics (Pimentel et al. 1997). Specifically, the threat posed by anthropogenic activities to native wildlife populations are currently being acknowledged and documented more than ever before (Barnosky et al. 2012; Cardinale et al. 2012). These '*invisible threats*' (Phillips 1997) directly result in the decline and sometimes the localised extinction of animal species. Defaunation (i.e. the human-mediated extinction of medium and large-sized vertebrates) (Dirzo & Miranda 1990) is therefore now being recognized as one of the biggest threats to the continued existence of natural wildlife populations (Dirzo 2001). Defaunation is driven directly by hunting, poaching and the illegal trade of animals or animal parts, as well as a by-product of the synergy amongst these drivers (Galetti & Dirzo 2013).

Subsistence poaching for bushmeat, primarily with the use of wire-snares (Lindsey et al. 2013) has rapidly escalated in the past few decades (Becker et al. 2013), and has been described as one of the greatest contemporary threats to the continued existence of natural wildlife (Fa et al. 2002) due to its correlation with severe reductions or extirpations of target species, high rates of non-target off-take of threatened species, and the loss of entire wildlife communities (Hofer et al. 2000; Lindsey et al. 2011, 2013; Becker et al. 2013). Similarly, human-wildlife conflict (HWC) resulting in the persecution of both genuine and perceived damage-causing animals (DCA's) is now viewed among the leading world-wide threats to wildlife (Treves & Karanth 2003; Graham et al. 2005; Inskip & Zimmermann 2009; Stein et al. 2010), and has greatly contributed to population declines experienced by wildlife during the previous century, in particular carnivores (Woodroffe & Ginsberg 1998). Finally, the harvesting of non-timber forest products (NTFP's), such as wild animals, for use in traditional medicinal practices is highly prevalent in South Africa (Shackleton 2009; Williams & Whiting 2016). Species are thus increasingly at risk of being overexploited, especially in areas where a mixture of the abovementioned animal off-take occurs, such as in the Western Cape Province, since the commercial harvesting of animals is directly associated with high potential endangerment of the species involved (Salick et al. 2006). Given the lack of information on basic species ecology and population dynamics of species occurring in the province, assessing the direct impact presented by these various forms of species off-take on a species- or local-scale proves unfeasible.

Table 5.1. The placement of numerical values of vulnerability into a RVA-test, according to the evaluation system of Ghimire & Aumeeruddy-Thomas (2005), and modified by Wagner et al. (2008).

Indicator	Category	Threat value
Life form	Annual/biennial	1
	Perennial	2
	Woody	3
Parts used	Leaves	1
	Generative organs, whole above-ground plant parts,	2
	bark	3
	Whole plant, whole below-ground plant parts	
Distribution	Wider distribution	1
	Himalaya-endemic	2
	Nepal-endemic	3
Local frequency	Frequent	1
	Moderate frequent	2
	Rare	3
Intensity of use	Rare	1
	Occasional	2
	Frequent	3
Use value	Single	1
	Multiple	2

Several models have been developed to predict the species most at risk of being overexploited based on factors that affect the vulnerability of NTFP's (Cunningham 1991; Cunningham 1996). Cunningham (1996) developed one such model, namely the Rapid Vulnerability Approach (RVA) to identify plants that were vulnerable to overexploitation in Bwindi National Park, Uganda. In the early phase, the assessment of vulnerability was conducted descriptively. The RVA approach was later described (Watts et al. 1996; Wild & Mutebi 1996) and modified to allow the incorporation of numerical scales of vulnerability (Lama et al. 2001; Ghimire & Aumeeruddy-Thomas 2005). The modified RVA-test consisted of eight indicators (plant parts used, life form, habitat, distribution, local rarity, use value, trade status, and threat status) that each had an associated score ranging from one to four to indicate levels of vulnerability. Using this approach, each of the eight values are totalled to find a final vulnerability value for each species. Wagner et al. (2008) reduced the Ghimire & Aumeeruddy-Thomas (2005) RVA-test to only six indicators (plant part used, life form, local frequency, distribution, intensity of use, and use-value) (Table 5.1) with a potential final score ranging between two and 10. They reasoned that the earlier versions of the RVA-test were too complex due to the incorporation of detailed information on plant biology, population dynamics, and habitat (Wild & Mutebi 1996). Furthermore, due to the lack of replicable information in several categories, Wagner et al. (2008) further improved the accuracy of vulnerability scores by combining the indicators into two separate groups.

In this study, the application of a newly revised vulnerability assessment approach is proposed to highlight animal species most at risk of being overexploited in the Western Cape Province. Specifically, new elements and reformulated RVA variables and scales for the RVA-test proposed by Wagner et al. (2008) are introduced, to assess the potential endangerment posed by three major forms of hunting in the Western Cape Province, namely subsistence poaching for bushmeat, retaliatory or preventative hunting resulting from human-wildlife conflict, and animal harvesting for use in the local traditional medicines.

5.3 Materials and Methods

5.3.1 Study area

The Boland Region forms part of the Western Cape Province of South Africa. Within the core of this area are numerous open and closed protected areas (PA's) enveloped by transformed lands for agriculture and urban expansion. These PA's represent the last critical refuges for many charismatic species, and generally support low animal biomass due to the largely unpalatable and nutrient-deficient fynbos vegetation (Coetzee 2016). The area however boasts high levels of vertebrate endemism and supports approximately half of all terrestrial vertebrate species found in South Africa (Turner 2012). Target study sites were predominantly located on private properties bordering protected areas, and in rural communities. The area is approximately 4 000 km^2, and is typified by a Mediterranean climate.

5.3.2 Sampling

Information obtained in three separate previously conducted studies that occurred in the same geographic region and during the same relative time period (2017 – 2018) was used. Each study conducted structured and semi-structured interviews based on questionnaires to obtain information on three major hunting practices in the province. In the first study, interviews were conducted with permanently employed farm labourers (n = 307) on privately owned agricultural properties adjacent to protected areas, to obtain information on illegal poaching for bushmeat, predominantly with the use of wire-snares. The second study interviewed landowners and managers (n = 103) of agricultural properties adjacent to protected areas, to obtain information on human-wildlife conflict dynamics and the associated off-take of DCA's. In the final study, interviews were held with traditional healers (n = 36) of indigenous African cultures in rural or peri-urban townships and informal settlements to inventory the animals used and sold. All interviews were conducted in isolation from other respondents, and in the preferred language of the respondents (either Afrikaans, English, isiXhosa or Sesotho).

5.3.3 The new RVA-method calculations

The new approach presented here for an animal-orientated RVA-method replaced five of the six variables in the version presented by Wagner et al. (2008), and made a logical alteration to the remaining variable. The scoring system was also adapted to accommodate the new variables. Despite the alterations made to the previous version, the essence of the original variables remain captured in the new system, and ultimately a similar filtering-process will be incorporated. With the proposed modifications to Wagner et al. (2008), the RVA-test for animals was calculated using the following variables:

(i) Social structure. This variable replaced the '*parts used*' variable to assess the impact incurred by removing one individual of a species. Solitary species thus receive a greater value than gregarious species, since the removal of one individual would impact the community of the latter to a lesser extent. Gregarious species are also further divided into small (< 20) and larger groups (≥ 20).

(ii) Species size. This variable replaces the '*life form*' variable to determine the rate at which an animal removed from any population is replaced. The body size (in kilograms) of individual species is used as a proxy for reproduction rate, whereby larger species are assumed to have longer generation times, lower fecundity, and lower intrinsic growth rates (Cardillo et al. 2005). A larger impact is thus expected when larger species are removed from communities (Peres 2001; Ceballos & Ehrlich

2002). Species were grouped into small-bodied (< 5 kg), medium-bodied (5 ≤ 35 kg) and large-bodied (> 35 kg) categories.

(iii) Species distribution. This variable was modified to accommodate the geographic context of the species' distributions. Species are classified as 'widespread', 'endemic to Southern Africa' or 'endemic or near-endemic to the Western Cape Province'. Species with limited ranges are assumed to be more at risk of extinction (Williams & Whiting 2016), and thus receive higher threat scores.

(iv) IUCN Conservation status. This variable replaced the '*local frequency*' variable as little to no reliable information was available on the relative abundance of species within the confines of the study area. Species of conservation concern were assumed to be more at risk of regional extirpations, and therefore received a higher threat score. The conservation status of species was obtained from the IUCN Red List of threatened species database (http://www.iucnredlist.org). Species that have not been assessed were assumed to have healthy populations and were grouped in the category for species of Least Concern (LC). Species were further grouped as either Near Threatened (NT) or Vulnerable (VU), Endangered (EN), and Critically Endangered (CR).

(v) Use-value. The use-value (Phillips & Gentry 1993) quantifies the threat posed to species in terms of how many times they were cited. The value was calculated by the formula UV = U/N, where U represents the number of citations given for a species and N represents the total number of informants (Trotter and Logan 1986). The UV for all species was calculated separately for the three hunting practices since the sample sizes varied significantly. The resulting UV's were then averaged to obtain a final UV for analysis (Appendix 5.1). Species were classified into three categories (< 0.08; 0.08 ≥ 0.15; > 0.15), with low UV's corresponding to low threat scores.

Threat level A:

Species size \ Social structure	Gregarious, large (≥ 20)	Gregarious, small (2 < 20)	Solitary (≤ 2)
< 5 kg	1	2	3
5 ≤ 35 kg	1	3	4
> 35 kg	1	3	5
		Threat value A:	

Threat level B:

IUCN status \ Distribution	Widespread distribution	Endemic to Southern Africa	Endemic or near-endemic to WCP
DD/LC	1	2	3
NT	2	3	4
VU/EN/CR	3	4	5
		Threat value B:	

Threat level C:

Threat value B \ Use-value	< 0.08	0.08 ≥ 0.15	> 0.15
1	1	2	3
2	1	2	4
3	2	3	4
4	2	3	5
5	3	4	5
		Threat value C:	

Threat level D:

Threat value C \ Number of threats	Single threat	2x threats	3x threats
1	1	1	2
2	1	2	3
3	2	3	4
4	3	4	5
5	4	5	5
		Threat value D:	

Threat values A + D = Final threat value

Final threat scores

2 No potential endangerment

3-4 Low potential endangerment

5-6 Moderate potential endangerment

7-8 High potential endangerment

9-10 Very high potential endangerment

Figure 5.1. Flow diagram illustrating the process proposed in the new RVA-test for assessment of potential endangerment. The six indicators, the threat scores at levels A, B, C and D, and the final categories for potential endangerment are indicated.

(vi) Number of threats. This variable replaced the '*use value*' variable (not to be confused with the abovementioned use-value) to further assess the degree of pressure placed on the species across all hunting practices. Species affected by only one form of hunting received a lower value than species affected by two or all three hunting practices.

The steps for assessment of potential endangerment are illustrated in the flow chart (Fig. 5.1). Wagner et al. (2008) admitted that the classification of potential endangerment into only three categories is "*very rough, but suitable for practice*". Therefore, the final classification was changed to five categories to improve the accuracy of the system.

5.3.4 Statistical analysis

For a RVA-test to be implemented, the following conditions have to be met: (1) The used indicators show no correlation to each other, (2) the threat values A and D, from which the final threat value was calculated, should be independent from each other, and (3) every indicator should make a significant contribution to the assessment of potential endangerment (Wagner et al. 2008).

To assess the criteria set forth for a successful RVA-test, Spearman's rank correlation coefficient was used to analyse the correlation between the indicators used for assessing potential endangerment, as well as with the final threat score. Mean values for each indicator were calculated to determine which indicator had the strongest effect on the final threat value, and a Wilcoxon signed-rank test was used to determine whether threat values differed significantly from each other to help determine the validity of the changes made to the RVA-test. All statistical analyses were performed in RStudio v3.5.0.

5.4 Results and discussion

A total of 82 vertebrate species were influenced by at least one of the three major hunting practices assessed in this study. After elimination of domestic animals, the threat scores of the remaining 78 species were assessed to prioritise species most at risk. Of these, most were mammals (68.3%), followed by reptiles (15.9%) and birds (14.6%). One fish species and no amphibian species were documented. The species recorded belonged to 40 families (most notably *Bovidae* – 11 spp., and *Felidae* – seven spp.) and 65 genera. Where individuals were not distinguishable up to species-level, they were grouped together as '*morphospecies*'. For example, respondents were unable to distinguish between the two genet species found in the study area (i.e. *Genetta genetta and G. tigrina*), and they were therefore grouped together as a single species. This study thus underestimates the number of species affected by hunting practices in the Western Cape Province.

5.4.1 Validity of the new RVA-test

The analysis of correlation between the indicators used for this RVA-test did not show any strong correlation to each other, with the exception of the correlation between threat level C and threat level D (Fig. 5.2). The strong correlation between these two threat levels was partially explained by threat level C being nested within threat level D. Since use-values (threat level C) were averaged across all threats, they were further expected to yield similar results to threat level D, where species affected by a higher number of threats received greater threat scores. Nonetheless, threat level A was not strongly correlated with any of the other variables (C_{corr}range = 0.20 – 0.32), therefore it can be concluded that threat level A and D independently contributed to the final threat score, and the revised RVA-test was thus valid (Wagner et al. 2008).

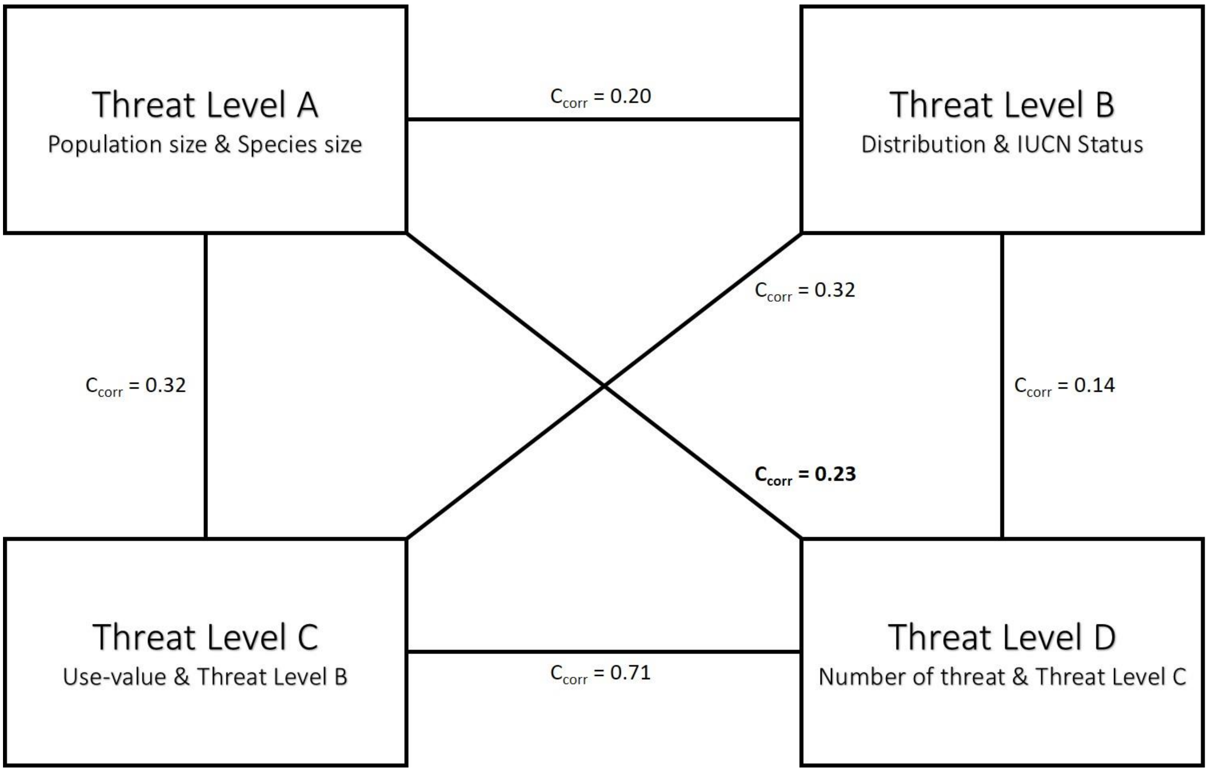

Figure 5.2. Spearman's rank correlation scores for the threat levels used in the modified RVA-test. Correlations between threat levels are shown on connecting lines.

All threat levels had positive correlations to the final threat score. Threat levels A and D had the highest correlations (C_{corr} = 0.84 and 0.67, respectively), as expected since these two values were summed to find the final threat score. The high correlation of threat level A to the final threat score highlights the importance of species' replacement rates following a removal event. The lowest correlation to the final threat score was shown by threat level B (C_{corr} = 0.26), indicating that the greater distribution of species and those listed of conservation concern by the IUCN does not have a substantial influence on the species vulnerability within the Western Cape Province.

Comparison of the means of threat levels A, B, C and D showed that the mean of threat level A (mean = 2.90) was higher than the mean of threat level D (mean = 1.55). Therefore, threat level A contributed the most to the final threat score, and it can be inferred that species with greater recovery potential were less threatened by hunting practices. Legalised hunting of these species can thus potentially occur sustainably.

When comparing the means of threat level B (mean = 1.61), C (mean = 1.73) and D, it was found that threat level C contributed most to the final threat score. Therefore, the use-value index, i.e. the proportional number of people hunting the species, contributed greatly and adversely to their survival potential.

The Wilcoxon signed-rank test showed significant differences between threat levels A and D ($P < 0.0001$), as well as between B and C ($P < 0.05$) and C and D ($P < 0.01$). It can therefore be concluded that each variable makes a significant contribution to the final threat score (Wagner et al. 2008). All the assumptions for a valid RVA-test are thus met.

5.4.2 RVA-results

The species included in the analysis showed different levels of threat towards hunting practices and thus in the level to which hunting of these species can be undertaken sustainably. Of all vertebrate species assessed in this study (n = 78), two species received a final threat score higher than nine (2.6%), and were thus listed under very high potential endangerment (Table 5.2). These were leopard (*Panthera pardus*) and grey duiker (*Sylvicapra grimmia*). A further six species (7.7%) received final threat scores of seven or eight, and were thus listed under high potential endangerment. These included chacma baboon (*Papio ursinus*), caracal (*Caracal caracal*), Cape porcupine (*Hystrix africaeaustralis*), aardvark (*Orycteropus afer*), genet spp. (*Genetta* spp.) and Cape clawless otter (*Aonyx capensis*). These species are thus highly threatened by current hunting practices in the Western Cape Province, and it is recommended that future investigations and conservation actions prioritise these species. The RVA-test also identified 21 species (26.9%) that were listed under moderate potential endangerment. These species thus do not appear to be under real immediate threat, but should be closely monitored. The remaining 49 species (62.8%) appeared to have low or no potential endangerment to their continued existence due to hunting practices. A full list of RVA-scores assigned to all species can be seen in Appendix 5.2.

Table 5.2. Species most vulnerable to the combined threat of the three major hunting practices in the Western Cape Province.

Species name	Threat level A	Threat level B	Threat level C	Threat Level D	Final threat score
Leopard	5	3	4	5	10
Grey duiker	4	2	4	5	9
Chacma baboon	3	2	4	5	8
Caracal	4	1	3	4	8
Cape porcupine	4	1	3	4	8
Aardvark	5	1	2	2	7
Genet spp.	3	2	3	4	7
Cape clawless otter	3	3	4	4	7

It is worth noting that these results are presented for threats originating solely in the Western Cape Province, and should therefore not be extrapolated and transferred to other geographic regions where the species may also occur. Further studies are needed to assess the impact of hunting on species in these areas, which may differ significantly due to a different species pool, different ethnic groups and varying levels of socio-economic development. Furthermore, vulnerability is solely dependent on what the authors considered the three major forms of animal off-take in the study area. Therefore, species listed as not under immediate threat should not be assumed to have healthy populations, as there are a great number of threats presented to native wildlife that were not included in this study.

5.5 Conclusion

The revised RVA-method proposed here is able to rapidly and effectively demonstrate the vulnerability of an animal species to several modes of species off-take, and subsequently roughly prioritise species demanding high levels of conservation and management consideration. It is therefore possible to apply this method for similar or variant hunting practices in other geographic regions, as well as for a range of completely different threats posed by anthropogenic activities. In this study, eight species were highlighted that are greatly impacted by the three dominant hunting practices in the study area, and close monitoring of population numbers is recommended to ensure their continued survival. Further comparisons with conspecifics in other geographic regions is however needed to draw a broader picture of conservation and sustainable management practices. Critical reviewing of this proposed method is also advised. A major limitation of this study was the lack of available information on the regional abundances of the species. Therefore, species that are rare in the study area may be classified as occurring in high densities due to their overall abundance across their entire distribution range. This study also emphasises the importance of more strongly regulated policing to curb the off-take of species, especially those that are range-restricted, threatened, habitat-specific, or occurring in low densities.

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5.7 Appendices

Appendix 5.1. The use-values (UV) for all species assessed in this study, calculated separately for each of the three major hunting pressures discussed, and averaged.

Species name	Bushmeat: wire-snares			Human-wildlife conflict			Traditional medicine			Average
	U	N	UV	U	N	UV	U	N	UV	
Acinonyx jubatus	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Agama atra	0	110	0.00	0	53	0.00	3	36	0.08	0.03
Alcelaphus caama	1	110	0.01	0	53	0.00	0	36	0.00	0.00
Alopochen aegyptiaca	1	110	0.01	6	53	0.11	0	36	0.00	0.04
Antidorcas marsupialis	0	110	0.00	0	53	0.00	11	36	0.31	0.10
Aonyx capensis	0	110	0.00	2	53	0.04	16	36	0.44	0.16
Arctocephalus pusillus	0	110	0.00	0	53	0.00	9	36	0.25	0.08
Atelerix frontalis	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Bitis arietans	0	110	0.00	0	53	0.00	13	36	0.36	0.12
Bos taurus	1	110	0.01	0	53	0.00	0	36	0.00	0.00
Bubulcus ibis	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Canis lupus familiaris	12	110	0.11	26	53	0.49	0	36	0.00	0.20
Canis mesomelas	0	110	0.00	0	53	0.00	19	36	0.53	0.18
Capra hircus	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Caracal caracal	3	110	0.03	6	53	0.11	13	36	0.36	0.17
Ceratotherium/Diceros spp.[1]	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Cetacea dolphin spp.[1]	0	110	0.00	0	53	0.00	9	36	0.25	0.08
Chersina angulata	0	110	0.00	0	53	0.00	5	36	0.14	0.05
Chlorocebus pygerythrus	0	110	0.00	0	53	0.00	3	36	0.08	0.03

Columbidae spp.[1]	1	110	0.01	0	53	0.00	0	36	0.00	0.00
Connochaetus taurinus	0	110	0.00	0	53	0.00	7	36	0.19	0.06
Cordylus spp.[1]	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Crocodylus niloticus	0	110	0.00	0	53	0.00	9	36	0.25	0.08
Dama dama	0	110	0.00	1	53	0.02	0	36	0.00	0.01
Dendroaspis spp.	0	110	0.00	0	53	0.00	6	36	0.17	0.06
Equine spp.[1]	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Equus caballus	0	110	0.00	0	53	0.00	3	36	0.08	0.03
Equus zebra zebra	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Felis catus	1	110	0.01	2	53	0.04	1	36	0.03	0.02
Felis lybica	0	110	0.00	0	53	0.00	8	36	0.22	0.07
Galarella pulverulenta	1	110	0.01	0	53	0.00	3	36	0.08	0.03
Genetta spp.[1]	2	110	0.02	1	53	0.02	21	36	0.58	0.21
Gyps spp.[1]	0	110	0.00	0	53	0.00	12	36	0.33	0.11
Bostrychia hagedash	0	110	0.00	1	53	0.02	0	36	0.00	0.01
Hemachatus haemachatus	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Herpestes ichneumon	0	110	0.00	0	53	0.00	4	36	0.11	0.04
Hippopotamus amphibius	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Hirundo spp.[1]	0	110	0.00	0	53	0.00	3	36	0.08	0.03
Hyaena brunnea	0	110	0.00	0	53	0.00	7	36	0.19	0.06
Hystrix africaeaustralis	39	110	0.35	5	53	0.09	28	36	0.78	0.41
Ictonyx striatus	0	110	0.00	0	53	0.00	6	36	0.17	0.06
Lepus spp.[1]	1	110	0.01	0	53	0.00	16	36	0.44	0.15
Loxodonta africana	0	110	0.00	0	53	0.00	3	36	0.08	0.03
Mellivora capensis	0	110	0.00	1	53	0.02	13	36	0.36	0.13
Naja nivea	0	110	0.00	0	53	0.00	13	36	0.36	0.12
Numida meleagris	22	110	0.20	14	53	0.26	0	36	0.00	0.15

Oreotragus oreotragus	7	110	0.06	0	53	0.00	1	36	0.03	0.03
Orycteropus afer	1	110	0.01	0	53	0.00	9	36	0.25	0.09
Otocyon megalotis	0	110	0.00	0	53	0.00	9	36	0.25	0.08
Owl spp.[1]	0	110	0.00	0	53	0.00	14	36	0.39	0.13
Panthera leo	0	110	0.00	0	53	0.00	5	36	0.14	0.05
Panthera pardus	1	110	0.01	1	53	0.02	30	36	0.83	0.29
Panthera tigris	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Papio ursinus	3	110	0.03	32	53	0.60	32	36	0.89	0.51
Passerine spp.[1]	1	110	0.01	1	53	0.02	0	36	0.00	0.01
Pavo cristatus	2	110	0.02	3	53	0.06	0	36	0.00	0.02
Pelea capreolus	4	110	0.04	0	53	0.00	6	36	0.17	0.07
Phacochoerus africanus	0	110	0.00	0	53	0.00	5	36	0.14	0.05
Poecilogale albinucha	0	110	0.00	0	53	0.00	8	36	0.22	0.07
Potamochoerus larvatus	0	110	0.00	0	53	0.00	4	36	0.11	0.04
Procavia capensis	4	110	0.04	3	53	0.06	11	36	0.31	0.13
Proteles cristata	0	110	0.00	0	53	0.00	7	36	0.19	0.06
Pseudaspis cana	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Python sebae	0	110	0.00	0	53	0.00	18	36	0.50	0.17
Raphicerus melanotis	36	110	0.33	0	53	0.00	6	36	0.17	0.16
Sciurus carolinensis	0	110	0.00	4	53	0.08	0	36	0.00	0.03
Shark spp.[1]	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Snake spp.[1]	0	110	0.00	2	53	0.04	3	36	0.08	0.04
Stigmochelys pardalis	0	110	0.00	0	53	0.00	1	36	0.03	0.01
Struthio camelus	0	110	0.00	0	53	0.00	3	36	0.08	0.03
Sturnus vulgaris	0	110	0.00	12	53	0.23	0	36	0.00	0.08
Sus scrofa	10	110	0.09	12	53	0.23	1	36	0.03	0.12
Sylvicapra grimmia	66	110	0.60	6	53	0.11	10	36	0.28	0.33

Syncerus caffer	0	110	0.00	0	53	0.00	12	36	0.33	0.11
Taurotragus oryx	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Taurotragus/Tragelaphus spp.[1]	0	110	0.00	0	53	0.00	7	36	0.19	0.06
Tragelaphus angasii	0	110	0.00	0	53	0.00	2	36	0.06	0.02
Tragelaphus strepsiceros	0	110	0.00	0	53	0.00	9	36	0.25	0.08
Tragelaphus sylvaticus	0	110	0.00	0	53	0.00	3	36	0.08	0.03
Varanus spp.[1]	0	110	0.00	0	53	0.00	20	36	0.56	0.19
Vulpes chama	2	110	0.02	0	53	0.00	12	36	0.33	0.12
Whale spp.[1]	0	110	0.00	0	53	0.00	2	36	0.06	0.02

[1]Species not identifiable up to species-level

Appendix 5.2. Rapid Vulnerability Assessment (RVA) scores for all species (final threat score), as well as scores for threat levels A, B, C and D.

Species	**Threat Level A**	**Threat Level B**	**Threat Level C**	**Threat Level D**	**Final Threat Score**
Acinonyx jubatus	5	4	2	1	6
Agama atra	2	2	1	1	3
Alcelaphus caama	1	2	1	1	2
Alopochen aegyptiaca	3	1	1	1	4
Antidorcas marsupialis	3	2	2	1	4
Aonyx capensis	3	3	4	4	7
Arctocephalus pusillus	1	1	2	1	2
Atelerix frontalis	3	2	1	1	4
Bitis arietans	3	1	2	1	4
Bos taurus	1	1	1	1	2
Bubulcus ibis	2	1	1	1	3
Canis lupus familiaris	3	1	3	3	6
Canis mesomelas	4	1	3	2	6

Capra hircus	1	1	1	1	2
Caracal caracal	4	1	3	4	8
Ceratotherium/Diceros spp.	5	4	2	1	6
Chersina angulata	3	2	1	1	4
Chlorocebus pygerythrus	2	1	1	1	3
Columbidae spp.	3	1	1	1	4
Connochaetus taurinus	1	2	1	1	2
Cordylus spp.	2	2	1	1	3
Crocodylus niloticus	3	1	2	1	4
Dama dama	1	1	1	1	2
Dendroaspis spp.	3	1	1	1	4
Dolphin spp.	1	1	2	1	2
Equine spp.	1	1	1	1	2
Equus caballus	3	1	1	1	4
Equus zebra zebra	3	5	3	2	5
Felis catus	2	1	1	2	4
Felis lybica	3	1	1	1	4
Galarella pulverulenta	3	2	1	1	4
Genetta spp.	3	2	3	4	7
Gyps spp.	4	4	3	2	6
Hadeda ibis	2	1	1	1	3
Hemachatus haemachatus	3	2	1	1	4
Herpestes ichneumon	2	1	1	1	3
Hippopotamus amphibius	3	3	2	1	4
Hirundo spp.	3	1	1	1	4
Hyaena brunnea	5	3	2	1	6
Hystrix africaeaustralis	4	1	3	4	8

Ictonyx striatus	3	1	1	1	4
Lepus spp.	3	1	3	3	6
Loxodonta africana	1	3	2	1	2
Mellivora capensis	4	1	2	2	6
Naja nivea	3	2	2	1	4
Numida meleagris	1	1	2	2	3
Oreotragus oreotragus	4	2	1	1	5
Orycteropus afer	5	1	2	2	7
Otocyon megalotis	2	2	2	1	3
Owl spp.	3	1	2	1	4
Panthera leo	3	3	2	1	4
Panthera pardus	5	3	4	5	10
Panthera tigris	5	3	2	1	6
Papio ursinus	3	2	4	5	8
Passerine spp.	3	1	1	1	4
Pavo cristatus	2	1	1	1	3
Pelea capreolus	3	3	2	2	5
Phacochoerus africanus	3	1	1	1	4
Poecilogale albinucha	3	1	1	1	4
Potamochoerus larvatus	3	1	1	1	4
Procavia capensis	2	1	2	3	5
Proteles cristata	4	2	1	1	5
Pseudaspis cana	3	2	1	1	4
Python sebae	5	1	3	1	6
Raphicerus melanotis	2	3	4	4	6
Sciurus carolinensis	3	1	1	1	4
Shark spp.	4	1	1	1	5

Snake spp.	3	1	1	1	4
Stigmochelys pardalis	4	1	1	1	5
Struthio camelus	3	1	1	1	4
Sturnus vulgaris	3	1	2	1	4
Sus scrofa	3	1	2	3	6
Sylvicapra grimmia	4	2	4	5	9
Syncerus caffer	1	1	2	1	2
Taurotragus oryx	1	1	1	1	2
Taurotragus/Tragelaphus spp.	3	1	1	1	4
Tragelaphus angasii	3	2	1	1	4
Tragelaphus strepsiceros	3	1	2	1	4
Tragelaphus sylvaticus	5	1	1	1	6
Varanus spp.	4	1	3	2	6
Vulpes chama	3	2	2	2	5
Whale spp.	3	1	1	1	4

Chapter 6

Project Summary with Implications for Management, Conservation and Policy Development

6.1 Project overview

Wildlife off-take in unprotected areas, especially those bordering reserves, has resulted in the global home range and population size reduction of naturally occurring wildlife (Treves & Karanth 2003; Inskip & Zimmerman 2009), and therefore potentially threatens the continued existence of wildlife in the Western Cape Province. Simultaneously, rural communities and commercial farmlands increasingly depend on natural resources to sustain their livelihoods, or face severe threats to their livelihood security as a direct result of co-existence with wildlife (Marker & Dickman 2005; Barua et al. 2013). As a result, an increasing number of people participate in, and depend on, the hunting of wildlife either for bushmeat, traditional medicine, or to protect their agricultural security. To investigate these largely undocumented forms of wildlife off-take, we relied on the local knowledge of people to elucidate the dynamics and interwoven social, economic and ecological factors underlying hunting patterns in the Boland Region. See page 3 for full project summary.

Key results of this study, as well as the implications thereof to management, conservation and policy development, are discussed in detail below.

6.2 Key research findings

Chapter 2: Socio-economic and biophysical determinants of wire-snare poaching incidence and behaviour in the Boland Region of South Africa

- The likelihood of a respondent being involved in wire-snare poaching activities was significantly determined by six variables. Firstly, local respondents and migrants from the Northern Cape Province were more likely to be involved in snaring activities than respondents that migrated from other regions across Southern Africa. Secondly, respondents with past historical influences (i.e. respondents who had parents or grandparents who practiced wire-snaring behaviour) were more likely to be involved in wire-snare activities themselves. Thirdly, respondents who were not regulated by anti-snaring regulations, or who were unaware of existing regulations, were more likely to be involved in wire-snare activities. Fourthly, respondents belonging to larger families (i.e. respondents who had more dependents), were more likely to set wire-snares. Finally, older respondents and respondents with longer job tenures were more likely to be involved in wire-snare activities.
- Labourers that admitted to involvement in regular snaring activity (n = 42) were personally motivated by the relative convenience of snares compared to other hunting methods (i.t.o. ease of use, obscurity, production cost, and the low risk of arrest it poses), food insecurity, and pest control.
- Agricultural properties that were more susceptible to wire-snare poaching activities were significantly described by three variables. Firstly, more snares were found on properties with a large number of labourer families living on the property. Secondly, farms with seasonal or contract workers in employment during any time of the year were more likely to have wire-snare activity. Lastly, farms that had no enforced punitive measures for wire-snare poaching were more likely to experience poaching activities.
- Wire-snare density was significantly influenced by four geographic, abiotic variables. Reported wire-snare density was high close to or on the borders of protected areas, and decreased with distance away from them. Wire-snare density was also highest further away from major residential areas and major roadways, as well as at high elevations.

- Significantly more wire-snares were found during summer months than during other times of the year.
- The species most desired by wire-snare poachers were, in order of popularity, grey duiker (*Sylvicapra grimmia*), Cape grysbok (*Raphicerus melanotis*), Cape porcupine (*Hystrix africaeaustralis*), landfowl (mostly helmeted guineafowl – *Numida meleagris*), feral pigs (*Sus scrofa*), grey rhebok (*Pelea capreolus*) and klipspringer (*Oreotragus oreotragus*). A total of 17 species or species groups were listed as desired by respondents to this study.
- The species most frequently caught in wire-snares by the study respondents were grey duiker, Cape porcupine, Cape grysbok, landfowl, feral pigs, grey rhebok and domestic or feral dogs (*Canis lupus familiaris*). Besides these seven species/species groups, an additional 13 species were caught in the three months preceding the interviews.
- Wire-snare activity was found in all major regions of the sampled area. Wire-snare activity hotspots were found, in order of intensity, in (1) the Agter-Groenberg farming community close to the Groenberg Nature Reserve near Wellington, (2) the Slanghoek farming community adjacent to the Hawequa Nature Reserve and close to Rawsonville, as well as opposite the N1 highway towards the Brandvlei Nature Reserve, (3) the Elandskloof farming community situated between the Theewaters Nature Reserve and the Southern end of the Hawequa Nature Reserve, and (4) near the North-eastern slopes of the Kogelberg Nature Reserve, close to Grabouw.

Chapter 3: Farmer attitudes and regional risk modelling of human-wildlife conflict on South African farmlands

- The most popular non-lethal control methods incorporated on farms in the Boland Region included acoustic deterrents, physical barriers, human shepherds, and chemical deterrents.

- The likelihood of farmers relying on lethal control methods was explained by five factors. Firstly, South African farmers were more likely to rely on lethal control methods than farmers who immigrated to South Africa. Secondly, male farmers were more likely to rely on lethal control methods than female farmers. Thirdly, farmers residing on their property on a full-time basis were more likely to rely on lethal control than farmers living on neighbouring properties or in a nearby town. Fourthly, farmers managing larger properties were more likely to rely on lethal control, and lastly, farmers from properties that were affiliated with regional conservancies had more reliance on control methods.
- The level of tolerance shown by farmers in the Boland Region towards damage causing species (DCA's) varied significantly among species. The highest level of tolerance was shown towards chacma baboons (*Papio ursinus*), honey badger (*Mellivora capensis*), mongoose spp., common eland (*Taurotragus oryx*), and grey duiker (*S. grimmia*). The lowest level of tolerance was shown for feral dogs (*C. familiaris*) and feral cats (*Felis catus*), snake spp., Egyptian geese (*Alopochen aegyptiaca*), feral pigs (*S. scrofa*), European starlings (*Sturnus vulgaris*), and helmeted guineafowl (*N. meleagris*). Low levels of tolerance were also shown for genet spp. (*Genetta* spp.), Cape clawless otter (*Aonyx capensis*), Caracal (*Caracal caracal*), leopard (*Panthera pardus*), and rock hyrax (*Procavia capensis*).
- Our study revealed clear patterns in the areas most vulnerable to human-wildlife conflict, regarding 15 species. Most notably:
  - Conflict with Cape porcupine was most prevalent on farms between Stellenbosch, Paarl and Franschhoek, increasing closer to Stellenbosch.
  - Conflict with honey badgers was highly prevalent in the central and southern divisions of the study area, most notably near the towns of Paarl, Stellenbosch, Franschhoek, and in the highlands farming area east of Kogelberg Nature reserve.

- Conflict with caracal was mostly confined to the Northern areas of the study, especially in the Agter-Groenberg areas near Wellington and Paarl.
- Conflicts with baboons were extremely widespread, occurring in all parts of the study area, but predominating in the Franschhoek valley, near Paarl, and in the Elgin and Vyeboom farming areas.
- Conflict with grey duikers predominated on the farms surrounding Stellenbosch and in the Vyeboom and Elgin farming communities.
- Conflict with feral dogs was the most widespread form of conflict in the area, occurring in all major study regions, especially between the towns of Paarl, Stellenbosch and Somerset-West.
- Conflict with Cape clawless otter was highly concentrated in the mountainous areas near Franschhoek and du Toitskloof pass.
- Conflict with feral pigs occurred mostly on farms near Wellington, and decreased towards Paarl and Franschhoek.

- Human-feral dog conflict was determined by the presence of vineyard-, livestock-, and poultry farming systems. Feral dog conflict also increased on properties with high amounts of natural vegetation, as well as properties closer to major roadways and residential areas.
- Human-baboon conflict was determined by the presence of vineyard-, field crop-, orchard-, and protea farming systems. Baboon-human conflict also increased in summer months.
- Human-honey badger conflict was determined by the presence of apiaries.
- Human-duiker conflict was determined by the presence of vineyards. Duiker conflict also increased during spring, on properties closer to major roadways, and on properties further away from residential areas.
- Human-caracal conflict was determined by the presence of game- and poultry farming systems. Caracal conflict was also higher during summer, on properties close to major roadways, and on properties at high elevation.

- Human-porcupine conflict was determined by the presence of orchard-, field crop- and livestock farming systems. Conflict with Cape porcupine also increased on properties far away from residential areas.
- Human-feral pig conflict was determined by the presence of game farming systems, and also increased on properties with large proportions of permanent water bodies.
- Conflict with Cape clawless otters was highest on trout farms, as well as properties with large proportions of permanent water bodies and river systems, and properties situated at high elevations.
- Human-genet conflict was determined by the presence of poultry farming systems.
- The risk maps revealed specific zones of predicted conflict risk with eight species in sampled and unsampled parts of the Boland Region based on several socio-economic and environmental factors:
    - Grey duiker: predicted zones of highest conflict risk were on farms surrounding Paardenberg Nature Reserve, Paarl Mountain Local Nature Reserve, the Northern side of Stellenbosch towards Simonsberg Nature reserve, the north-eastern side of Groenlandberg Nature Reserve, and along the old du Toitskloof pass in the Hawequas Mountain Catchment area.
    - Chacma baboon: predicted zones of highest conflict risk were widespread, encapsulating most of the greater Boland Region. Conflict risk was however particularly high on farms South of the Matroosberg mountain range near Worcester, in Slanghoek and Franschhoek farming valleys, near the Jonkershoek Nature Reserve, in the EGVV farming community, near Bot River southwards to Hermanus, and along the coastline at Gansbaai, Kleinmond, Betty's Bay, Pringle Bay and Rooi-els.

- Feral pigs: predicted zones were less widespread, concentrating around Bain's Kloof pass near Wellington and Southwards along the Hawequas mountain range towards Franschhoek. To a lesser extent feral pig conflict can also be expected to spread to Fonteintjiesberg Nature Reserve, to Riversonderend Nature Reserve, and around the Berg River dam to Jonkershoek Nature Reserve and possibly the Banhoek Conservancy.
- Cape porcupine: predicted zones of highest conflict risk were relatively widespread, intensifying on farms along the western slopes of Tierkloof, Witzenberg, Wittebrug, and Fonteintjiesberg Nature Reserve, near the Brandvlei Nature Reserve at Rawsonville, on the farms surrounding Paarl, Franschhoek, Banhoek, Stellenbosch and Blaauwklippen. Porcupine conflict is also predicted for the Overberg regions, such as the Kogelberg Sonchem Link Nature Reserve, the Highlands farming community, and near Houwhoek Nature Reserve.
- Cape clawless otter: predicted zones of conflict were mainly confined to the Hawequas Mountain range, especially near the old du Toitskloof Pass and the Hawequas Nature Reserve. To a lesser extent conflict was also expected at the Fonteintjiesberg, Riviersonderend, and Theewaters Nature Reserve.
- Feral dogs: predicted zones of highest conflict risk were widespread, but intensified near human settlements. Most notably on the western side of the greater Boland Region, approaching the Cape metropole, i.e. farmed areas near Brackenfell, Kuils River, and Khayelitsha. The coastal towns of Strand, Gordon's Bay, Betty's Bay, Kleinmond and Hermanus were also predicted to experience elevated conflict risk. Inland hotspot areas include farms surrounding Franschhoek, Villiersdorp, the Elandskloof farming valley, Rawsonville, Worcester, and Bot River.

- Caracal conflict: predicted to intensify on farms near Blaauwklippen, the Jonkershoek, Groenberg, and Riviersonderend Nature Reserves, Villiersdorp, and in the Elandskloof farming valley. Caracal conflict risk is also expected to be high near the in the Grabouw valley between the Hottentots-Holland- and Kogelberg Nature Reserve, at Kleinmond, and near Fernkloof Nature Reserve.
- Honey badger: predicted zones of conflict risk are extremely widespread but confined to small, isolated areas. The highest risk was predicted for areas surrounding the Paardenberg, Mount Hebron and Kogelberg Nature Reserves, Blaauwklippen, Wedderwil, the Fairy Glen Private Nature Reserve, and the Slanghoek farming community. Badger conflict was also expected on farms near Klapmuts, Bot River, Hermanus, Gansbaai, Villiersdorp and Genadendal.

Chapter 4: The use of animals and animal-derived constituents in African traditional medicine and other cultural applications: townships in the Western Cape Province

- A total of 71 species (or morphospecies) were cited by traditional healers in the sampled communities, belonging to four classes and 20 orders. The main orders were Carnivora (20 species), Artiodactyla (17 species) and Squamata (10 species).
- The most abundant species were chacma baboon, leopard, Cape porcupine, puff adder (*Bitis arietans*), monitor lizard spp. (*Varanus* spp.), honey badger, hare spp. (*Lepus* spp.), Cape clawless otter, African buffalo (*Syncerus caffer*), African rock python (*Python sebae*), and owl spp.
- Twelve species recorded during this study are listed of conservation concern by the IUCN Red List of threatened species (version 2017-3).

- The number of uses for each species varied from one to 16, with the most uses attributed to Cape porcupine (16 uses), leopard (15 uses), and chacma baboon (11 uses).
- A total of 716 vertebrate species parts or products were recorded during our study, and were subsequently grouped into 12 categories. The most prevalent was skin pieces or entire skins (258 items), oil and subcutaneous fat (120 items), and animal bones (62 items).
- Prices for animal items ranged from R15 to R20 000. The most expensive items were leopard skins (median = R11 250), African buffalo bones (median = R8 250), and Cape fur seal (*Arctocephalus pusillus*) bones (median = R6 500).
- Sample size was deemed sufficient through the construction of a rarefaction curve.
- Species richness estimators all approached asymptotes or near-asymptotes. The Jack 1, Jack 2 and MMMeans estimators were deemed the most efficient, and predicted that an additional 7-15 species could be recorded if our sampling continued.
- Species diversity for all species in the sampled community was relatively high (Shannon $H' = 3.79$; Simpson's $1/\lambda = 28.55$), as well as evenness values (Shannon $J' = 0.89$).
- Informant consensus for the use of particular species in treating similar ailments differed in their level of homogeneity and heterogeneity. The highest degree of informant consensus was observed for resolving court cases or reducing prison sentences (0.88), and for treating mental illnesses or improving cognitive abilities (0.85). Comparatively lower consensus was shown for the treatment of medical ailments (0.31) compared to the treatment of spiritual or magical ailments (0.36). Overall consensus for treating any ailment was exceptionally low (0.34).

- The most prioritised species and associated use in the sampled community were monitor lizard spp., puff adder and African rock python, all for the purpose of protecting individuals from evil spiritual entities.
- The species identified as the most significant to traditional healing for the African cultures in the sampled communities were leopard, monitor lizard spp., chacma baboon, and African rock python.

Chapter 5: An assessment of animal species' vulnerability to major hunting practices in the Western Cape Province.

- A total of 82 vertebrate species or morphospecies were documented to experience adverse effects from either of the three hunting methods assessed in this thesis, or a combination thereof. Domestic animals were eliminated from the analyses, resulting in a total of 78 species belonging to 40 families and 65 genera.
- No strong correlation was shown between the indicators of threat level A and threat level D, and all variables were found to make a significant contribution to the final threat score; the new RVA-test was therefore deemed valid.
- The most important factor contributing to a species' vulnerability to hunting pressures was shown as the species' ability to replace individuals following a removal event.
- Final threat scores revealed:
    - Leopard and grey duiker are experiencing very high potential endangerment from hunting practices.
    - Chacma baboon, caracal, Cape porcupine, aardvark (*Orycteropus afer*), genet spp., and Cape clawless otter are experiencing high potential endangerment from hunting practices.
    - A further 21 species were listed as receiving moderate potential endangerment from hunting practices, while the remaining 49 species

appeared to have little to no potential endangerment at the time of the study.

6.3 Management implications and recommendations

Finding solutions for alleviating hunting pressures is important to ensure the survival of an array of wildlife species, but a clear understanding of the extent and drivers of hunting activities is essential for interventions to be successful. Adequate mitigation strategies should therefore draw on accumulated empirical knowledge concerning wildlife behavioural ecology, context-specific local experiences, and public perceptions of wildlife management (Treves & Karanth 2003), thereby incorporating both human and biodiversity conservation interests. Furthermore, intervention tools aiming to balance these two aspects sustainably should be persistently efficient, include minimal unintended environmental consequences, be selective towards problematic animals or communities, incur a lower cost than that of the damage prevented, and be socially acceptable (McManus et al. 2015). Based on the findings of this study, as well as the relevant findings of previous studies conducted elsewhere, the following measures are recommended as mitigation tools to ameliorate the detrimental consequences of wildlife off-take.

6.3.1 Bushmeat hunting and the associated use of wire-snares

Alternative livelihoods

Alternative livelihood programmes are initiatives designed specifically for reducing poaching incidence by generating alternative financial dependencies for communities reliant on bushmeat (Becker et al. 2013). With regards to the respondents, who were all permanently employed farm labourers with fixed monthly incomes, the development of alternative livelihoods is not particularly relevant. Future research can however greatly benefit from identifying poachers beyond the scope of farm labourers to ultimately introduce and promote alternative livelihood strategies to the presumably, mostly unemployed people. This implies providing local communities with alternative ways to accrue income, such as job creation on a national scale, and teaching new skills on a local level to facilitate the onset of new job markets (Lewis 2007). Often, natural environments are used as the basis for creating new job opportunities, such as beekeeping, craft production and nurseries (Vliet 2011) through, for example, several well-known integrated conservation and development projects (ICDP's), and community-based wildlife management (CBM) schemes (Metcalfe 1993; Lewis & Phiri 1998). These projects provide impoverished rural communities with the opportunity to become involved in wildlife-based industries, and promote sustainable development options such as eco-tourism, agroforestry, and the sustainable utilisation of natural resources (Naughton-Treves et al. 2005). Ultimately, these schemes aim to foster a greater appreciation and kinship for wildlife, thus promoting sustainable use. Another example is the community markets for conservation (COMACO) scheme, implemented since 2003 in the Luangwa Valley, Zambia. This model uses an adaptive business style to teach alternative livelihood skills, and endorse sustainable agricultural systems, referred to as '*conservation farming*' (Lewis et al. 2011). There is however little reliable information available on the successes of previously implemented alternative livelihood projects (Lewis et al. 1990; Vliet 2011), while an inability of the schemes to improve livelihoods and confer conservation goals have been reported (Naughton-Treves et al. 2005). Increased income may also increase demand for bushmeat (Nielsen et al. 2012; Nuno et al. 2013), and therefore alternative livelihood schemes should be implemented in conjunction with new or updated legislation and policies. Economic theory does however suggest that provisioning rural communities with access to affordable and conventional substitutes to bushmeat has the potential to promote wildlife conservation by reducing wire-snare poaching (Wilkie et al. 2005).

Extending sufficient financial benefits to offset potential income from bushmeat poaching is however inherently difficult (Lindsey et al. 2011b), and may promote the incidence of bushmeat poaching due to individuals seeking unwarranted compensation. Despite the caveats and challenges associated with alternative livelihood development, suggested future research should be undertaken to identify and describe the reliance of unemployed people in the Boland Region on bushmeat, to subsequently enable welfare and conservation organisations to contact and appeal to local communities relying on bushmeat to promote alternative livelihoods. Ultimately, for CBM and ICDP schemes to be successful, they need to remain simple, inexpensive, and reliant on local leadership (Wright 1988).

Alternative food sources

Beyond alleviating poverty through promoting alternative livelihoods, providing an alternative, affordable source of protein to communities dependent on bushmeat may further help address poaching, and is equally applicable to farm labourers with fixed monthly incomes. Bushmeat is however often preferred due to cultural convictions (Wilkie et al. 2005), and is thus not always readily interchangeable with other protein sources, such as meat from livestock. An improved understanding of the social dynamics regarding bushmeat consumption is thus required. In the Savé Valley Conservancy, Zimbabwe, a meat distribution programme commenced in 2009 whereby meat from culled elephants is sold to local communities at a subsidized price (Lindsey et al. 2011b). We propose that a similar programme be designed and implemented in the Boland Region, specifically for the sale of feral pig (*S. scrofa*) meat. These invasive species are detrimental to agricultural infrastructure and crops (Hone 2002) and therefore their eradication is a top priority for many stakeholders in the Boland Region. It would therefore be of dual conservation benefit to implement increased actions to eradicate feral pigs, and subsequently distribute the meat amongst poorer communities. Given the large population densities, it is however unfeasible to supply everyone with these reduced meats, but focusing meat distribution can nonetheless realistically reduce poaching. Attention should be given to identify the ideal recipients and supply ratio, as well as implement adequate disease control measures (Barnett 1997; De Garine & de Garine-Wichatitsky 1999). Providing certain communities with this option will likely also result in an influx of migrants to the area (Wittemyer et al. 2008), and this therefore should also be monitored. Another viable solution to reduce the dependency of farm labourers on bushmeat in the Boland Region is by funding initiatives that teach communities to grow and maintain small crop gardens, particularly legumes that have a high protein content. Increased crop production may however lead to additional conservation challenges, such as an increase in human-wildlife conflict (Lewis et al. 2011). Therefore, education on multiple levels is required. A further addition to this initiative is to involve local schools. Many schools in South Africa have feeding schemes to support pupils without access to food sources. It can therefore be of great benefit to promote the establishment of a food garden in school terrains, whereby students are given the responsibility to grow a variety of crops for use in the feeding schemes, concurrently learning how to cultivate crops. The ultimate vision of any alternative food source program should however be to empower communities to overcome dependency on the programme, and not to foster reliance on this source of food security.

Education

In order to eradicate wire-snare poaching in the Boland and elsewhere, educational interventions should commence on multiple levels. Firstly, conservation bodies and agencies should educate and encourage landowners to become involved in conservation-orientated initiatives, so as to promote a holistic approach to farming, and thereby combat issues such as wire-snare poaching. Furthermore, our study showed that wire-snare poaching was more prevalent on properties with contract workers in employment, presumably at least partially due to the lack of repercussions faced when not in permanent employment at a specific farm, and it would therefore be of further benefit for conservation bodies to approach third party employment agencies issuing these contract workers. Simultaneously, the responsibility rests with landowners to develop or join numerous existing regional conservancies in their area, which urges them to adhere to various issues of conservation concern. These conservancies also provide a valuable platform to form communication networks between farmers, as well as conservation agencies. Disparity in the perceptions on conservation problems between stakeholder groups often impedes the application of conservation initiatives (Biggs et al. 2011), and it is therefore important to facilitate communication between landowners. Secondly, landowners should ensure that their labourers are educated with regards to the issue, and that they comply with the anti-snaring regulations set forth. On 51.4% of the properties included in our study and on which regular snaring activity occurs, farmers were unaware of any such activity. To ensure better communication and compliance, regular information sessions should be held by property owners. To further ensure compliance, landowners are encouraged to include labourers in anti-snaring initiatives, for example by giving them the responsibility to monitor fence lines on a regular basis, and thereby also removing all snares encountered. Finally, greater public awareness of the threat posed by bushmeat poaching is needed to generate involvement in anti-poaching initiatives, as well as to leverage more funding for these initiatives (Lindsey et al. 2013). Additionally, conservation outreach programs should focus on educating unemployed people, people working in non-agricultural sectors near natural habitat (such as forestry or mining companies), and the youth.

Adequate legal and policy frameworks

Developing countries, like South Africa, typically lack adequate structure, strategy and funding in terms of environmental law enforcement efforts, which allows illegal bushmeat poaching to proliferate (Anthony et al. 2010; Parr 2011). Furthermore, the lack of law enforcement essentially transforms PA's into open access resource areas, resulting in situations referred to as '*the tragedy of the commons*' (Hardin 1968); where natural resources used commonly by local communities are overexploited or degraded (Feeny et al. 1990). In the Western Cape Province, the current punitive measures implemented by both government and private landowners to govern wire-snaring seem to be largely inadequate and do not therefore act as sufficient deterrents to eradicate wire-snare poaching. On a governmental level, wire-snare poachers are rarely convicted as wire-snaring is considered a minor wildlife offence (Loibooki et al. 2002; Becker et al. 2013), and when fines are issued, they are lower than potential earnings from selling bushmeat, particularly given hyperinflation (Lewis et al. 2011; Lindsey et al. 2011b; Knapp 2012). At the level of private properties, landowners in this study often complained of having their hands tied i.e. they cannot justifiably dismiss a worker for setting wire-snares due to legislative loopholes (Pers. Obs.). It is therefore important that legislation be updated to reflect the true value of wildlife, and to give sufficient power to the court and landowners to adequately persecute poachers. Furthermore, judiciary and law enforcement agencies should be made aware of the value of wildlife and the threat posed by wire-snare poaching (Lindsey et al. 2013). Although we suggest more strongly regulated governing regulations be implemented, we acknowledge the existence of a paradox wherein a blatant dismissal or monetary fine is potentially counterproductive if it increases economic hardship for the individual or his family and friends, potentially exacerbating the issue by encouraging the need for additional resources obtainable through bushmeat poaching (Knapp 2012). Therefore, stronger law enforcement measures should be coupled with efforts to extend benefits in the form of alternative livelihoods or protein sources (Keane et al. 2008; Brashares et al. 2011).

Anti-snaring patrols

Increasing security on farms will likely reduce poaching incidence (Jachmann & Billiouw 1997; Stokes et al. 2010), but will present an additional and often unfeasible cost to farmers. On a regional level, governmental anti-poaching enforcement will likely reduce wildlife poaching (Wilfred 2010). One such particular method is the use of anti-snaring patrols led by communities or conservation bodies, coupled with on the ground intelligence gathering (Kenney et al. 1995; Becker et al. 2013). This creates the opportunity to further involve communities in conservation, thereby promoting wildlife appreciation and alternative livelihoods (Vongkhamheng et al. 2013), and ultimately reducing poaching pressures (Johannesen & Skonhoft 2005; Steinmetz et al. 2014). Similarly, well-trained rangers will be the primary deterrent that renders high-level policies effective (Rowcliffe et al. 2004). On-going training and the engaging of individuals as rangers will thus create additional employment opportunities to combat poverty. A good understanding of efficient patrolling techniques is however required to keep up with the increasing scale and sophistication of poaching (Phelps et al. 2012; Pimm et al. 2015). To this end, law enforcement monitoring (LEM) tools provide an immediate means to collect, analyse, and report data from field observations (Ripple et al. 2015). Some early technological advances in LEM tools have been applied widely across Southern Africa (Kruger & MacFadyen 2011), namely CyberTracker and MIST (Management Information System). These systems both integrate a basic database with geographic information systems (GIS) and a range of analytical functions. More recently however, an improved open-source monitoring tool called SMART (Spatial Monitoring and Reporting Tool) was developed by a broad consortium of conservation organizations (Pimm et al. 2015). The SMART technology provides a platform to measure law enforcement efforts and threats in a particular area to improve the protection of conservation target species that are threatened by poaching (Hoette et al. 2016), and has been adopted by governments in at least 30 countries in Asia, Africa and South America as the standard for protected area monitoring (Pimm et al. 2015). Integrating this platform in the Boland Region thus provides an exciting new opportunity to aid conservation agencies and management authorities in mitigating and extirpating wire-snare poaching through patrolling areas of known poaching intensity. Based on our results, we thus recommend that the areas of Agter-Groenberg-, Slanghoek-, and Elandskloof farming communities, as well as the Northern Kogelberg Nature Reserve be prioritised for monitoring.

Reducing wire-snare availability

Wire used for making snares in readily available on all agricultural properties through the disentanglement of fence and crop wires, and its availability is thus not likely to decrease in the near future. However, it has been suggested that alternative materials for fences can be used to reduce the incidence of wire-snares (Lindsey et al. 2011a). Fences made from kinked, mesh (*bonnox/veldspan™*) for example cannot be used for wire-snares like those made from steel or barbed wire (Lindsey et al. 2012; van Rooyen et al. 2016). The erection of these types of fences will thus reduce opportunity for making wire-snares. Additionally, landowners will further benefit financially through reduced theft of fence wires, especially expensive electrical fencing.

Sustainable use of bushmeat

Due to the widespread occurrence of bushmeat harvesting and the importance of bushmeat to local communities, it is unlikely that the reliance on bushmeat will disappear. Therefore, future policies may consider transforming the use of bushmeat into a legal, controlled framework that monitors species and habitat conditions to guide sustainable utilisation of bushmeat as a natural resource. With few exceptions, the current management approaches and policies regarding wire-snare poaching is not conducive to the sustainable use of bushmeat as a natural resource. Therefore updating the legislative and policy framework to sustainably govern bushmeat utilisation will require strenuous amounts of resources. Nonetheless, the spatial harvest theory developed by McCullough (1996) may serve as an initial guideline to guide the process. The theory advocates that management divide natural areas into sources (protected areas with no hunting allowed) and sinks (areas where controlled hunting is allowed). Animals will subsequently be allowed to move without restriction between the two areas to allow depleted areas to become continually replenished. Allowing bushmeat harvesting to occur in a legalised environment will further allow for more controlled harvesting, thus reducing non-target off-take and promoting the targeting of specific individuals. For example, hunters will mostly be encouraged to target males of a species to achieve higher meat yields (Lindsey et al. 2011b). Furthermore, conservation agencies and researchers will be better able to monitor species populations, as well as bushmeat markets. We failed to establish the presence of a bushmeat market in the Boland Region, but previously conducted studies have placed great emphasis on market demands fuelling bushmeat off-take (Bowen‐Jones et al. 2003). It is thus entirely possible that a bushmeat market potentially exists to some extent in the study region, however likely without the input of farm labourers, and may greatly contribute to wire-snare incidence. Future research is thus suggested to identify these markets, as addressing the policies that regulate them may be more effective in reducing wire-snare poaching incidence than directly addressing the poachers (Damania et al. 2005).

Land-use planning

Human populations and agricultural boundaries continue to increase and expand (Wittemyer et al. 2008; Becker et al. 2013), almost certainly increasing the prevalence of wire-snare poaching activities by broadening the interface between people and wildlife (Lindsey et al. 2011a). Therefore, land-use plans should be adopted with demarcated zones for conservation use that discourages agricultural development. Simultaneously, zones for urban expansion, including the establishment of informal settlements, should be adequately planned so as to not endanger natural wildlife habitat. At the centre of the land-use planning initiative is however the importance of providing continuous corridors for wildlife movement, thereby not isolating populations and thus exposing them to extirpation due to local stressors, such as wire-snare poaching (Becker et al. 2013; Berentsen et al. 2013). By promoting connectivity through conservatively delineating zones for development and settlement, the *in situ* conservation of the Boland Region's wildlife will thus be promoted by diminishing edge effects (Creel et al. 2013) and further degradation of buffer zones (Watson et al. 2013). Simultaneously, PA's should remain of sufficient size to retain wildlife diversity

(Newmark 2008). The movement of people through PA's, as well as private agricultural land, should also be controlled as far as possible, as this provides a gateway for potential poachers to easily access wildlife habitat. Caution should also be exerted towards people claiming to access the areas for medicinal plant collection (Lindsey & Bento 2012), as this is often associated with animal harvesting for medicinal purposes. Finally, land-uses occurring within PA's, such as forestry, require careful management to reduce unnecessary human influxes and to maintain habitat for wildlife (Clark et al. 2009; Poulsen et al. 2009). Land-use zoning will however require substantial cross-ministerial communication and cooperation (Lindsey et al. 2013), and will result in high expenses and time commitments (Naughton-Treves et al. 2005). Finances for land zoning can potentially be acquired from external investments, such as involving shareholders, the private sector, or NGO's in co-management agreements (Lindsey et al. 2013). Additionally, land reforming is increasingly becoming a political priority in South Africa (Fourie and Fourie 2016), and therefore some form of land-use redistribution will likely take place in the near future. It is therefore essential that conservation professionals be involved in the process to ensure adequate zoning of wildlife habitat.

Conclusion

A variety of social, economic, political and environmental factors contribute to wire-snare poaching incidence (Lindsey et al. 2011a, 2011b; Becker et al. 2013; Watson et al. 2013), and therefore no management scheme can be successfully implemented without a fundamental and empirically-based understanding of the area experiencing poaching problems. A robust, long-term quantification of wire-snare trends and patterns is thus the first step to achieving successful mitigation and eradication goals. Many of the recommendations made here will also require substantial time and monetary commitments. There is thus a need for the application of several of these initiatives simultaneously, with short-term applications occurring in the interim. It is also vital that a cooperative synergistic attitude between government, private landowners, tourism agencies, the invested public, and NGO's be created. Ultimately, employment seems to be the only long-term way to release poachers from their dependency on bushmeat (Knapp 2007). In the short-term, adequate punitive measures and enforcement, such as anti-snaring patrols, seem to be the most cost-effective.

6.3.2 Off-take due to human-wildlife conflict

Lethal control

Given the large economic impact of HWC on the livelihoods of farmers in the Boland Region, and elsewhere (Thorn et al. 2012; Barua et al. 2013), the use of lethal control methods is usually justified along economic lines. In a South African context, the use of lethal control methods such as gin-traps, gun-traps, poison, and hunting with or without dogs, was traditionally deemed acceptable and the majority of farmers readily relied on these methods to control damage-causing animals (DCA's) (Macdonald et al. 2010). In modern times, opponents of wildlife rehabilitation and conservation continue to exert strong political and pro-active pressures through the intentional killing of DCA's (Woodroffe & Ginsberg 1998; Landa et al. 1999), but an improved understanding of ecological function has given rise to new social constituencies promoting wildlife conservation in support of ecosystem and animal wellbeing (Treves & Karanth 2003). Nonetheless, the common misconception that lethal control is the cheapest and most effective method for managing conflict species is still widely accepted (Conover 2001; Mitchell et al. 2004), despite growing scientific evidence illuminating the limitations associated with its use (Prugh et al. 2009; Conradie & Piesse 2013; Nattrass & Conradie 2013). Most importantly, lethal control is suggested to have little benefit over a prolonged period of time because intense off-take triggers compensatory demographic responses such as an influx of replacement individuals and increased recruitment to areas under hunting pressures, as well as reduced emigration and natural mortality (Prugh et al. 2009). These trends have previously been observed in red foxes (*Vulpes vulpes)* (Baker & Harris 2006), as well as leopard and caracal (Conradie & Piesse 2013). Black-backed jackal (*Canis mesomelas*) have also been shown to adapt to increased persecution by adapting their reproductive strategy i.e. by increasing their litter sizes and by breeding at a younger age (Bingham & Purchase 2002; Beinart 2008; Nattrass & Conradie 2013). Demographic responses of this nature is especially present in territorial carnivores, and would thus result in an increased risk of depredation due to the increase in local predator population size (Crooks & Soulé 1999; Knowlton et al. 1999). Other problems associated with the use of lethal control methods include the ongoing time and monetary commitment it requires (Conover 2001; Mitchell et al. 2004), as well as its non-selectivity; meaning that the individuals hunted are often not the true DCA's and its removal thus has no effect on conflict prevalence (Avenant & du Plessis 2008). It is however worth adding that, although highly controversial, certain selective lethal control methods (e.g. euthanasia following cage trapping by certified professionals) have been shown to not only reduce livestock depredation, but also increase the overall tolerance shown by farmers towards carnivores (Treves & Naughton-Treves 2005; Ripple et al. 2014). It is however essential that the selection of target individuals for these control methods follow a highly selective regime (Sacks et al. 1999; Treves et al. 2002, 2004), as the majority of perceived DCA's have no proven involvement in conflict situations (Gipson 1975; Horstman & Gunson 1982; Sacks et al. 1999). Furthermore, the lethal control of invasive and feral species engaging in HWC is socially accepted and similarly we encourage farmers to participate in the eradication of these species, thereby supporting native wildlife and ecosystem survival (Lowe et al. 2000; Young et al. 2011; Barrios-Garcia & Ballari 2012). In the Boland Region, these are thus feral dogs (*C. familiaris*), feral pigs (*S. scrofa*), European starlings (*S. vulgaris*), Eastern grey squirrels (*Sciurus carolinensis*), and Indian peafowl (*Pavo cristatus*).

Physical barriers

Variations of fencing, kraaling (bomas or corralling), and livestock collaring to exclude DCA's is not a new concept, and are methods applied widely in areas where people experience conflict with co-existing wildlife (Taylor 1999; Schiess‐Meier et al. 2007; Dickman 2010). Similarly, respondents to the study widely reported the use of physical barriers, most notably electrified fences, for the exclusion of DCA's. Electrified fences are particularly effective in deterring antelope species (Mason 1998), and provide an effective means to reduce conflict with feral dogs in the Boland Region. The erection of electrical fencing does however have several disadvantages. Firstly, constructing and maintaining electrical fences is laborious and expensive (Shelton 1984; Angst 2001), and is thus not a feasible mitigation solution for most small-scale commercial and subsistence farmers. Secondly, they can restrict the movement of wild animals (Thouless & Sakwa 1995), thereby decreasing the size of their natural home ranges and potentially isolating populations. Thirdly, respondents reported grey duiker (*S. grimmia*) to frequently run into fences and fatally injure themselves. Finally, wire from fences are often dismantled to manufacture snares (van Rooyen et al. 2016), thus fuelling another aspect of conservation concern. The use of traditional barriers made from natural materials were not reported in this study, but have previously been reported to work successfully and are generally cheaper to maintain than modern fencing (Jackson & Wangchuk 2001; Ogada et al. 2003). These thus offer a potential solution for subsistence farmers and farm labourers in the Boland Region to protect their livestock and crops from DCA's. The use of '*jakkalsdraad*' (Afrikaans for 'jackal-wire') is also deemed to be sufficient in preventing unwanted animals from entering properties, especially species that have a propensity for tunnelling passageways underneath fences. Given sufficient time, individuals however learn to penetrate any barrier (Shelton 1984; Thouless & Sakwa 1995). This was especially true for baboons in the Boland Region, who were frequently reported to climb over electrified fences with relative ease. The use of kraaling (or bomas) is also used in some areas, and has been reported as successful for keeping predators from accessing livestock (Treves &

Karanth 2003). However, kraaling has simultaneously been abandoned in many areas because it tends to promote erosion rates through overgrazing and land degradation (Beinart 2008). Bomas have also been suggested to promote surplus killings of livestock (Nowell & Jackson 1996) and increase rates of disease transmission among livestock (Van Sittert 1998). Other physical barriers with reported successes in this study, included the use of netting to exclude birds from orchards and vineyards and the use of stilts to prevent honey badgers from accessing beehives. Additionally, enclosing young trees with *jakkalsdraad* to prevent conflict with small antelope, and constructing tactical fences to allow certain wildlife to enter farms unharmed, were recorded as being used successfully in this study. The use of cages to protect beehives from badgers were reported to be largely unsuccessful. Overall, we acknowledge the potential merit of physical barriers in keeping certain species out, but also concede that physical barriers on their own will likely not be successful deterrents for the majority of DCA-species in the Boland Region. Therefore, they should be coupled with other non-lethal control methods (Treves & Karanth 2003). We further urge landowners to consider the ecological implications of isolating their properties, particularly large properties that provide the only corridor for wildlife movement between natural habitats, before extensive physical barriers are erected.

Chemical deterrents

Chemical deterrents aim to repel DCA's by irritating their sensory organs (Norman et al. 1992), by mimicking specific semiochemical signals, or by causing gastrointestinal malaise (Mason 1998). Chemical deterrents aimed at irritating the sensory organs of DCA's are typically viewed as superior compared to the latter two categories, because they elicit immediate avoidance with no prior learning period required (Mason 1998). Overall, chemical repellents are predominantly directed towards herbivores (few products exist for deterring carnivores) and are found to be most effective when applied directly to food items (Mason 1998). To reduce consumption of agricultural crops, these deterrents should thus be directly applied to the vineyards or orchards in areas of predation. In this study, the most prevalent form of sensory irritants were chilli products or '*hot sauce*' containing capsaicin from Capsicum peppers (e.g. *Capsicum annuum*) that are applied directly to the leaves, stems or vines of crops. Sensory irritants including ingredients such as allyl isothiocyanate, ammonia, carbon dioxide, and formaldehyde are also vastly popular (Mason & Otis 1990; Parker & Osborn 2006; Sitati & Walpole 2006). Many however agree, that taste is rarely an effective repellent, especially regarding crop-raiding species (Nolte et al. 1994), and similar observations were recorded in this study. Sensory irritants have further not been shown to be effective deterrents with specific taxa (Norman et al. 1992). On the other hand, olfactory cues resembling the presence of predators, such as semiochemical odours resembling predator urine, or odour resulting from protein degradation (Mason 1998), are generally more effective in deterring herbivore species (Nolte et al. 1994). This study recorded the use of dog hair, human hair, and lion faeces (obtainable from pet groomers, barber shops, and felid parks, respectively) with mixed reports on effectiveness. Overall however, chemical deterrents are limited by short-term effectiveness, their unpredictability, and often their unintended effects on non-target species (Ratnaswamy et al. 1997). Further research is therefore needed to empirically test the effectiveness of the wide range of chemical repellents available to farmers. One respondent in this study reported magnesium to be an effective deterrent of small antelope, particularly duiker. In order for chemical repellents to be effective, they should however be used in conjunction with other control methods, and ideally form part of a wider range of strategies of integrated pest management (IPM).

Acoustic deterrents

Acoustic deterrents act as fear-provoking stimuli aimed at eliciting neophobia in wildlife (Mason 1998), consequently keeping them temporarily or permanently away from predominantly croplands. Acoustic deterrents include the use of distress calls, propane exploders, and pyrotechnics (live ammunition, firecrackers etc.), and are mainly used for repelling damage-causing bird species, although some success has been reported in mammal species (Bomford & O'Brien 1990). In the current study the use of wind cannons, bangers (firecrackers), and gunfire to deter baboons were widely applied, but general consensus was that these animals eventually learn not to fear acoustic deterrents. As a result, many farmers used gunfire as both a lethal and non-lethal deterrent, mainly issuing warning shots but occasionally fatally wounding an individual. Overall, acoustic deterrents are not recommended for deterring mammalian DCA's, but may prove effective in deterring unwanted bird species. Another promising emerging technology is the use of virtual fences to keep DCA's from entering farmlands (Richardson et al. 2016), mainly owing to the unpredictability of these devices for DCA's (Shivik et al. 2003), but the efficacy and ecological consequences of this method remains largely unknown.

Visual deterrents

Like acoustic deterrents, visual deterrents such as eyespots, predator effigies (e.g. scarecrows) and mylar are most effective in deterring birds, primarily because birds possess colour vision and the ability to see ultraviolet light (Hunt et al. 1997). Some level of efficacy is however expected in mammal species. Only two properties employed some form of visual deterrents. The first displayed broken cd's to deter birds from vineyards, with little success. The second employed fladry lines with unknown success. Fladry lines are commonly used visual deterrents, and have been found to reduce wolf predation (Davidson-Nelson & Gehring 2010). The same study however found no significant effect of fladry lines on coyote visitation rates. Although its efficacy remains largely unknown, fladry may provide some temporary relief for farmers from certain depredating species, bearing in mind that the labour and equipment costs of fladry, as well as other visual deterrents, can be substantial (Davidson-Nelson & Gehring 2010).

Human shepherds

The use of traditional shepherding or herding techniques are widely promoted to mitigate HWC (Ogada et al. 2003; Shivik 2006; Dickman 2010), especially in East African countries where pastoralists co-exist with predators (Kruuk 1980), compared to the more segregated approach seen in Southern Africa (Ogada et al. 2003). The employment of people as shepherds or '*chasers*' were similarly highly popular across the sampled community, and were vastly cited as the only effective solution for reducing conflict with baboons. Additionally, unemployment is South Africa is high, and this creates much-needed job opportunities. However, employing labourers as shepherds can amount to significant additional costs (Barua et al. 2013), which are generally not achievable for small-scale and even large-scale, commercial farmers in the Boland Region. Respondents employed up to six additional labourers for the sole purpose of keeping baboons at bay. Further concerns are raised when shepherding amounts to a loss of sleep, the inability of children to attend school, and increased disease contraction (Barua et al. 2013).

Guard-animals

A variety of animals are used to guard livestock and croplands from unwanted DCA's, including donkeys (*Equus africanus asinus*), alpacas (*Lama pacos*), llamas (*Lama glama*) and dogs (Conover 2001; Crawshaw 2004). In particular, livestock-guarding dogs, especially Anatolian shepherd dogs, are progressively promoted as an effective non-lethal control mechanism in traditional animal husbandry (Dickman & Marker 2005; Graham et al. 2005; Gehring et al. 2010; Rigg et al. 2011; McManus et al. 2015; Potgieter et al. 2016). These dogs (also known as Kangal dogs) have a large body size (25 – 50 kg) and have the potential to act as introduced carnivores (Potgieter et al. 2016). The results of livestock-guarding dogs have been mainly positive. One of the biggest agents of livestock guarding dogs, the Cheetah Conservation Fund, have reported reduced livestock losses on Namibian rangelands (Marker et al. 2005). In another study, Ogada et al. (2003) showed lions are less likely to take cattle from bomas where dogs are present, but simultaneously the presence of dogs had no effect on leopard depredation. In this study, surprisingly few farms used any form of livestock-guarding animal on their property. The reason for which remains unclear. A few caveats have however been listed for the use of dogs as non-lethal deterrents. For example, guard dogs may act as reservoir hosts for diseases (e.g. rabies) that may spread to wildlife (Cleaveland & Dye 1995; Rhodes et al. 1998; Haydon et al. 2002), particularly other canid species such as the bat-eared fox (*Vulpes chama*), Cape fox (*Otycyon megalotis*), and black-backed jackal (*C. mesomelas*). Guard dogs may also on occasion kill non-target species (Potgieter et al. 2016).

Buffer crops

Although planting buffer crops will likely not prove to be the solution for deterring DCA's, these crops can be used in conjunction with other non-lethal control methods, as respondents reported some success in using this method, for example: onion plantations were relatively sufficient for keeping porcupine and baboons away from field crops. This is likely because selective breeding has made some crops such as tea and sisal less palatable to wildlife, and therefore these species can be used in buffer crop plantations (Thouless 1994). Reported success were also recorded for fruit stock-piles in keeping baboons out of orchards i.e. fallen and rotten fruit is gathered from orchards and dumped at the interface of farmland and natural habitat. One respondent further applied this method to game farming, and used springbok as a buffer for more expensive game animals.

Translocations

In some extreme cases (Treves & Karanth 2003), known problem species are removed from areas where they engage in conflict situations and relocated to other areas (Stander 1990; Bradley et al. 2005; Athreya et al. 2011). However, translocation interventions for felids rarely turn out as planned (Athreya et al. 2011). Translocations often result in high mortality rates, presumably due to capture-related stress, injuries, infanticide and intraspecific aggression within new ranges, or excessive post-release movement (Miller et al. 1999; Treves & Karanth 2003; Athreya 2006; Letty et al. 2007). Additionally, translocated animals often attempt to return to their original home ranges post-release (Rogers 1988), or continue to engage in conflict with people in their novel ranges (Bradley et al. 2005). Felids such as leopards (*P. pardus*) (Athreya 2006), jaguars (*Panthera onca*) (Rabinowitz 1986) and cougars (*Puma concolor*) (Ruth et al. 1998) have all been documented to return to their home ranges following translocation releases. More research is needed on this topic, as interventions in HWC, as well as other fields of conservation, will likely rely more on translocations in the future.

Compensation schemes

Globally, many governments issue financial compensation to landowners whenever they experience loss as a direct result of HWC in an attempt to mitigate the conflict (Maclennan et al. 2009; Treves et al. 2009). For example, compensation schemes have been implemented extensively in the Americas and Europe (Montag & Patterson 2001). The efficacy of these schemes have however been widely criticised, mainly because many farmers have abused the scheme for their own benefit (Nyhus et al. 2003). Furthermore, it is argued that the compensation scheme encourages lax livestock husbandry, such as farmers letting livestock stray or failing to adequately protect livestock (Bulte & Rondeau 2005). For compensation schemes to be implemented, a good understanding is also required of the underlying mechanisms and dynamics of HWC, as well as a robust and ongoing commitment from governmental and private sectors (Constant 2014). In South Africa, like most other developing nations, the implementation of compensation schemes is thus not currently feasible (Dickman et al. 2011; Thorn et al. 2013). Hence we expect a clear lack of support from government to landowners if a compensation system should be advocated for the Boland Region. Other developing nations in Africa have however attempted to implement such schemes, such as Malawi, Kenya, Zimbabwe and Botswana, but little to no data exist to evaluate the ultimate success or failure of these schemes (Nyhus et al. 2003).

Education

Central to any conservation issue is ensuring the involved community is continually educated on the relevant issue. Similarly therefore, interventions and developments that received substantial empirical backing should be continuously communicated to landowners to raise awareness of the importance of protecting wildlife, as well as the advantages and caveats of available mitigation methods. The importance of increasing landowner involvement in regional conservancies, to provide a valuable platform for the joint communication of issues of conservation concern needs to be emphasised. The implementation of biodiversity stewardship programmes (see for example the World Wildlife Fund (WWF) Conservation Champions; http://www.wwf.org.za/conservation_champions_list.cfm) should also be promoted in the Boland to ensure that secure conservation statuses are awarded to farmlands maintaining high biodiversity value, and subsequently to ensure that landowners receive tangible benefits for their conservation efforts (Paterson 2009).

Prey abundance

Many researchers believe that depredation only occurs when natural prey items become depleted (Bagchi & Mishra 2006). Crop farmers should be mindful that the eradication of crop-raiding species will likely lead to increased livestock depredation rates on their own or neighbouring farms. Similarly, herbivore numbers will likely boom when predation pressures are removed (Redford 1992), thus leading to increased conflict with crop-raiding species when depredating predators are eradicated. As a result, some farmers have implemented strategies to improve habitats on or near their farm, thereby increasing prey numbers and subsequently diverging predators away from livestock (Constant et al. 2015).

Conclusion

In the Boland Region, it is evident that concerted efforts have been made to balance the needs of people and wildlife, resulting in a considerable proportion of farmers abandoning traditional dogmas promoting the use of lethal control methods. Similar trends globally have likewise resulted in increased research aimed at finding alternative solutions to manage DCA's in a non-lethal manner (Treves et al. 2009). It is furthermore evident that a single panacea to resolving conflict rarely exists, and we therefore advise the implementation of a combination of the control methods discussed above (Distefano 2005), tailored for the specific needs of different farm types and conflict species. Paradoxically however, it seems that the diversity of available tools for the non-lethal control of DCA's has decreased as demand has increased (Clark 1998), potentially indicating increased consensus and thus increased effectivity for a few control methods. Nonetheless, data supporting the effectivity of several non-lethal control methods in real world applications remain incredibly sparse (Mason 1998). Further research is thus required before trusted recommendations for control methods can be made to farmers. Furthermore, considering the novel socio-political context in South Africa, many existing strategies may also need to be re-evaluated to include our improved understanding of wildlife ecology and management.

6.3.3 Animal harvesting for use in traditional medicine

Poverty reduction

For some traditional healers or general traders, selling traditional medicine is not their primary choice of occupation, but necessitated by a lack of broader education and skills, as well as job scarcity (Williams 2003). Therefore, a real need exists to invest in the education of these communities to equip and enable them to pursue careers beyond their current prospects. We believe that in doing so the number of traditional healers will greatly decrease, because many reportedly enter the profession as purely financially driven '*fake sangomas*', masquerading under the same titles as certified traditional healers. Given the status-conscious mindsets of African cultures and the enhanced reputation traditional healers enjoy within their communities, convincing traditional healers to leave their profession will not be an easy task (Drury 2011).

Persecution

The first response offered by the general public to the real or perceived misappropriation of natural resources is usually to issue fines or arrest trespassers. However, given the sensitivity and nature of traditional medicinal use, we believe increased law enforcement efforts will result in antagonistic responses from the individuals involved in the trade. A culture-sensitive approach is therefore required, coupled with concise and realistic rules and guidelines for the harvesting of NTFP's.

Permit-monitoring systems – certification systems

It has previously been suggested by various conservation proponents that the harvesting of natural resources should be incorporated into a legalised framework operating under a permit-monitoring system (Williams 2003). In the Western Cape Province, the Traditional Health Practitioners Act (no. 22 of 2007) is currently the standard for determining traditional healers' general code of conduct. Additionally, many respondents to the study freely provided documentation illustrating their affiliation with local traditional healer and herbalist associations. We therefore recommend that developments of this kind, as well as contractual relationships, be further promoted to allow for the widespread formalization of the medicinal trade in wild animals, thereby increasing the ability to regulate and police the trade (Williams 2003). Ultimately, we believe this will contribute to the improved sustainability of the trade (Petersen et al. 2012), as well as allow for a more rigorous understanding of the trade. For example, if source populations are made known to conservationists, or actively controlled by government, targeted restoration, comprehensive risk assessments, and enhanced surveillance all become possible. The formalised platform will furthermore provide opportunities to potentially alter the perceptions of traditional healers and raise awareness for conservation benefit. For example, the harvesting of species of conservation concern can potentially be reduced, and selective harvesting can be promoted. Finally, this creates the possibility of introducing a quota system (bag-limit) similar to the one issued to private agricultural landowners experiencing conflict with wildlife. Permits may however act as gateways for indiscriminate resource extraction if not subject to sufficient policing efforts (Petersen et al. 2014), and therefore we acknowledge that a permit-monitoring system in a formalised environment, although hopeful, will not succeed without the full support of the South African government and local authorities (Williams 2003). The success of sustainable utilisation programs will furthermore depend on wealth distribution, the state of environmental controls, the adequacy of governance and legislation, and local community cooperation (Loveridge et al. 2010). Extensive research on habitat quality and population viability of species would also be required, and may amount to the use of vast resources. The sustainable fur trade in North America (Nowell & Jackson 1996) provides evidence that sustainable resource use programs are indeed possible, however challenging they may seem.

Demand reduction

For conservation efforts to succeed, the demand for wildlife as traditional medicine will have to be drastically reduced. Therefore, it will be equally necessary to appeal to consumers of the trade as opposed to solely targeting traditional healers. Firstly, further research is required to better understand the psychology and general motivations for people relying on the consultations of traditional healers. Secondly, relationships will have to be cultivated with communities to convince people that their health is not dependent on animal parts (Sumrall 2009). It goes without saying that this will not be an easy feat, as age-old religious, spiritual and cultural convictions generally underpin consumer behaviour (Lee et al. 2014). It has however been shown on numerous occasions that consumer behaviour can be altered through education (Yang et al. 2007; Wasser & Jiao 2010), as well as by raising awareness among communities (Wasser & Jiao 2010).

Alternatives

Raising conservation consciousness may further provide a gateway to promote the use of alternative synthetic materials, animals, or sources among traditional healers (Liu et al. 2016). For example, arrangements can be made to donate the carcasses of animals culled on private agricultural properties to traditional healers. Similarly, game reserves often cull individuals of populations exceeding the maximum carrying capacity of their habitat, and these too could be distributed. Arrangements of this nature can however fuel the rate of off-take by landowners if they receive compensation for their efforts or if the sense of purpose removes their previous hesitations in killing DCA's.

The use of roadkill in traditional healing presented a viable solution to ameliorate the impact of medicinal animal harvesting, but after discussing this with traditional healers in the sampled community it became clear that roadkill is not considered adequate for spiritual healing. General traders of animal parts will however likely not object to the distribution of roadkill, and this idea should therefore be promoted among traders. Finally, replacing wild harvested materials with synthetic substitutes may provide the ultimate solution for reducing wildlife off-take. Animal skins and bones worn as clothing or jewellery in particular may be successfully substituted with synthetic replicates. For example, great success has been achieved in supplying Zulu and Shembe tribes with synthetic leopard pelts for use in their traditional gatherings (Pieterse 2016), and it would be worthwhile to distribute those incentives among local African tribes. The reduced price of synthetic products compared to wild harvested products will likely also aid in the effectiveness of similar programs, as consumers will be more likely to pay for cheap synthetic substitutes (Liu et al. 2016); provided of course that they accurately resemble genuine materials.

Conclusion

The mitigation of sustainability issues originating from spiritual, religious and cultural convictions is arguably one of the most difficult challenges faced by conservationists today. For wildlife to survive the growing pressures posed by the traditional healing industry, policymakers, managers and conservationists will however have to band together to find effective solutions, whilst at the same time remaining culturally sensitive and empathetic. Based on the findings of this study, we recommend that efforts be focused on introducing the synthetic leopard skins developed for the Zulu tribes, since leopards were the most widely harvested species in the sampled community. Chacma baboon were similarly widely harvested, and therefore we further request the cooperation of local farmers to supply carcasses of conflict species to traditional healers, especially given the high numbers of baboon off-take recorded in Chapter 3. Overall, there seems to be no reliable short-term solution for reducing traditional medicinal demand.

6.4 Project conclusion

Wildlife off-take practices, coupled with growing human populations, have led to the depression of most wildlife habitats (Woodroffe et al. 2005), particularly those bordering reserves. This document thus provides essential contextual data to incorporate into effective mitigation interventions to ensure the continued survival of wildlife alongside people, and took a vital step in promoting wildlife conservation by elucidating the extent and dynamics associated with hunting practices. It is however not only wildlife who will benefit, as it can be assured that the benefits people derive from co-existing with wildlife (e.g. food security and traditional medicine) will eventually falter (Bennett et al. 2002). It is therefore essential that social interventions be implemented to ensure that communities can successfully exist without depending on wildlife. Our reliance on communities for data for this study was warranted by their roles as immediate custodians of their natural resources, thus enabling them to supply us with information not obtainable elsewhere. It is however also true that some information provided may be subject to bias, for example, when participants deliberately or unintentionally inflate answers due to ulterior motives, or underestimate values due to the fear of persecution or due to antagonistic attitudes towards conservation (Rasmussen 1999). Nonetheless, the current study leaves little doubt that wildlife off-take in the Boland Region is pervasive, and the underlying dynamics remains complex and multifaceted. It is therefore unlikely that its occurrence will be resolved in the near future. The approach to this study opened a dialogue between rural communities and conservationists, managers, policy-makers and researchers on issues that have received no formal attention to date. We therefore suggest the application of unilateral policies be implemented in conjunction with further studies to empirically validate the findings and broaden our understanding of the heterogeneity in local scale socio-ecological dynamics. The novel information provided in this study can be used to develop effective management and eradication frameworks focused on an ever-increasing threat to wildlife populations in the Boland Region of South Africa.

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6.6 Appendices

Appendix 6.1. Questionnaire used for conducting interviews with permanently employed labourers on agricultural properties with regards to the use of wire-snares to capture bushmeat.

Labourer interview questionnaire

Location: ………………………………………………………… **Time:** ……………………………

Interviewer: …………………………………………………………

		Other:		
1.2	Gender	Male	Female	
1.3	Age			
1.4	Home language	Afrikaans	English	isiXhosa
		Other:		
1.5	Job title			
1.6	Family size			
1.7	Period of employment			
1.8	Migratory status			
1.9	Place of residency	On property	Neighbouring property	Nearby town
		Informal settlement	Other:	

Section A: Respondent information

Section B: Wire-snares

2.1	Have you ever seen wire-snares on the property?	Yes		No	
2.2	How many snares have you seen in the last month?				
2.3	Snares sightings in past year	Average per month:			
		Maximum per month:			
		Minimum per month:			
2.4	Date when first snare was seen on property				
2.5	What time of the year do you find more snares?	Summer		Autumn	
		Winter		Spring	
2.6	To your knowledge, how many people set snares on this property?				
2.7	Have you ever set wire-snares?	Yes		No	
2.8	What is your main reason(s) for setting snares?				
2.9	Is using snares easier than other hunting methods?	Yes		No	
2.10	Do you ever set snares for:	The convenience thereof?		Yes	No
		Food?		Yes	No
		Cultural reasons?		Yes	No
		Selling meat/parts?		Yes	No
		Sport/hobby/comradery?		Yes	No

		Pest control?	Yes	No
2.11	What time of the year do you set most snares?			
2.12	How many snares have you set in the last month?			
2.13	What is the max/min number of snares you've set in a month?			
2.14	Date when first started snaring			
2.15	If you sell caught animals, to who?	Other labourers	Markets	
		Other:		
2.16	What are the species most desired to catch?			
2.17	List species caught in the past month			
2.18	Where did you learn the skill of snaring?	Peers	Older family members	
		Other:		
2.19	What are the laws that apply to snaring on this property?			

Appendix 6.2. Questionnaire used for conducting interviews with landowners or managers of private agricultural properties with regards to damage-causing animals (DCA's) on the property.

Landowner/manager interview questionnaire

Location: .... **Time:** ....

Interviewer: ....

1.3	Deed number		
1.4	Property size (ha)		
1.5	Land-use distribution (% or ha)	Livestock:	Orchards:
		Game:	Vineyards:
		Field crops:	Forestry:
		Natural:	Infrastructure:
		Other (apiaries, trout, fynbos, poultry etc.):	
1.6	Conservancy affiliation?	Yes	No
		Name:	

Section A: Property description

Section B: Respondent information

2.1	Population group	Asian	Black
		Coloured	White
		Other:	

2.2	Gender	Male		Female
2.3	Age			
2.4	Home language	Afrikaans	English	isiXhosa
		Other:		
2.5	Job title			
2.6	Job tenure			
2.7	Migratory status			
2.8	Place of residency	On the property	Neighbouring property	Nearby town

Section C: Labourers

3.1	Number of workers & families resident on property						
3.2	Are seasonal/contract workers employed?	Yes			No		
3.2.1	When are seasonal/contract workers employed?	Jan	Feb	Mar	Apr	May	Jun
		Jul	Aug	Sept	Oct	Nov	Dec
3.2.2	Where do seasonal/contract workers stay during employment?	On the property			Neighbouring property		
		Informal settlement			Nearby town		
3.3	Are labourers allowed to hunt on the property?	Yes			No		
3.3.1	What repercussions are faced if they do hunt?						

3.4	Are you aware of wire-snare activities on the property?	Yes	No

Section D: Damage-causing animals

4.1	Have any wildlife been hunted on this property?	Yes	No
4.1.1	List all animals hunted (facilitate with flashcards)		

Template (repeat for all species hunted):

Species hunted	*Reason (e.g. crop/infrastructure damage, stock predation, hobbyist hunting etc.)*
Time of year when the species are hunted (list months)	*How many individuals are shot annually?*
Maximum number of individuals shot in a monthly period?	*Minimum number of individuals shot in a monthly period?*
Method used for hunting (e.g. firearms, trapping, dogs, poison etc.)	*Date (month/year) when species was first hunted*

4.2	Are there any non-lethal methods used?	None
		Acoustic deterrents:
		Physical barriers:
		Human shepherds:
		Guard-animals:
		Chemical deterrents:
		Visual deterrents:
		Other:
4.3	What laws apply to hunting on the property?	
4.4	Are the number of animals hunted on the property reported? Where?	

4.5	Rank the species that cause the most damage financially	

Appendix 6.3. Questionnaire used for conducting interviews with traditional healers in informal settlements and townships in rural or peri-urban landscapes with regards to the use of wildlife in traditional medicine.

Traditional medicine interview questionnaire

Location: .... **Time:** ....

Interviewer: ....

Section A: Socio-economic information

Population group	Asian	Black	Coloured	White
Gender	Male		Female	
Age				
Migratory status				
Family size				
Home language	isiXhosa	Sesotho	Other:	
Level of schooling				
Identification (e.g. diviner, herbalist, trader etc.)				

Section B: Market details

Where do you get you stock from?	
How are animals caught?	

Where are animals caught?						
When is most produce sold?	Jan	Feb	Mar	Apr	May	Jun
	Jul	Aug	Sept	Oct	Nov	Dec
How did you start?	Family business		Entrepreneurship		Spiritual revelation	
# Employees?						
Established period						
How many traders in the area?						

Section C: Species information

Species sold	**Part(s) used**	**Purpose/use**	**Price (R)**	**Quantity sold/month**	**Demand trend (incr/decr/sta)**	**Shelf-age**	**Animal life stage**

Appendix 6.4. Letter of consent given to potential respondents on agricultural properties prior to interviews. Consent letters were also made available in Afrikaans and isiXhosa.

Letter of consent to be interviewed

Dear participant, this is a letter of consent stating that you agree to participate in this interview as part of a study investigating wildlife off-take on agricultural properties bordering protected areas. Specifically, we will be enquiring about the use of wire-snares to capture wildlife as bushmeat, and the hunting of damage-causing animals on the property. Your contribution to this study is invaluable as the knowledge you have on these topics are unattainable elsewhere. This study will further allow for the identification of threats to local wildlife, as well as the human communities, so that both wildlife populations and human livelihoods can be protected.

The interview will take place in a mutually agreed upon location on the property. Your identity shall remain anonymous, and all accounts given will be kept confidential. Your participation is entirely voluntary, and if at any point you wish to stop the interview and have your answers removed from the records, you may withdraw without any consequences. We only ask that you provide honest accounts, and if you are uncomfortable answering any question, you simply decline to answer.

We ask your permission to use the answers given by you in further analysis and potential publications. Please also feel free to ask any questions should you have any, or if any of our questions are unclear to you. There are no known or anticipated risks to you as a participant to this study.

Date: ………………………… ID: …………………………… Consent: Yes/No

If you have any further questions or want more information on this project, please feel free to contact:

Wian Nieman (PI, Stellenbosch University): 17688132@sun.ac.za OR 083 [redacted]

Anita Wilkinson (SA, The Cape Leopard Trust): [redacted] OR 082 [redacted]

Appendix 6.5. Letter of consent given to potential respondents (traditional medicine) prior to interviews. Consent letters were also made available in Afrikaans and isiXhosa.

Letter of consent to be interviewed

Dear participant, this is a letter of consent stating that you agree to participate in this interview as part of a study aiming to identify and quantify the commercialized trade of bushmeat and ethnotherapeutic animal parts/products in informal settlements in the Boland. It will be conducted by two researchers from Stellenbosch University (Wian Nieman, MSc candidate) and the Cape Leopard Trust (Ismail Wambi, Community Outreach Officer).

Your contribution to the study is invaluable as the knowledge and information you possess is unattainable elsewhere. This study will assess whether any threats exist, so that measures for both wildlife and stakeholder conservation can be implemented.

The interview will take place in mutually agreed upon location. You shall remain anonymous and all information provided will be kept confidential. Your participation in entirely voluntary, and if at any point you wish to stop the interview, you may withdraw without any consequences. We only ask that you provide honest accounts, and decline to answer when a specific question makes you uncomfortable.

We ask your permission to record this interview with a tape-recorder to ensure we attain all the information provided, and to use the information in further analyses and potential publications. Please feel free to ask any questions of us if they should arise. There are no anticipated risks to you as a participant in the study.

Date: ………………………… **ID: ……………………………** **Consent: Yes/No**

If you have any further questions or want more information on the project, please feel free to contact:

Wian Nieman (PI, Stellenbosch University): 17688132@sun.ac.za OR 083 [redacted]

Anita Wilkinson (SA, The Cape Leopard Trust): anita@capeleopard OR 082 [redacted]