



THE NAMIBIAN LEOPARD: NATIONAL CENSUS AND SUSTAINABLE HUNTING PRACTICES



In Cooperation with
The Ministry of Environment and Tourism

Study Report

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1. Executive Summary

The African Leopard (*Panthera pardus pardus*) is one of Africa's most distinguishable big cats. As the leopard has such a broad geographical range combined with its cryptic activities there is a limited amount of empirical evidence that exists which in turn can be applied to adaptive management strategies, through practical conservation methods and monitoring across Namibia and Southern Africa.

As a result of the limited coverage, lack of empirical evidence regarding the Namibian leopard populations, distribution and population dynamics the ability to determine long-term conservation strategies and effective monitoring has been limited. The leopard is highly adaptable and can utilise human dominated environments successfully compared to other large carnivores. However, leopards are under pressure across their range from habitat loss and fragmentation, reduced wild prey availability, and conflict with farmers due to livestock predation and retribution killing. A global human-wildlife conflict study found that the leopard is the leading carnivore conflict species as it featured in the greatest number of human-wildlife conflict case studies. This pattern can be found across Namibia with both freehold and communal farms reporting losses of livestock and game to leopards. In addition, freehold farmers claim that they have noticed a continual increase in leopard numbers on their farms in tandem to an increase in conflict cases. This situation has been exacerbated by the severe drought that occurred in Namibia from 2015 to 2017 with vast areas of the country yet to fully recover from the effects. As conflict increases so does the number of leopards labelled as problem animals, indiscriminate of age or sex, being removed from the farmland and in turn the national population. However, a high proportion of the leopards removed are not reported to the authorities, therefore the level of removal is currently unknown. The long-term sustainability of the leopard population in Namibia relies upon the understanding of all the highly complex dynamic pressures placed on the species and in turn creating viable and effective monitoring systems.

The last comprehensive leopard census undertaken in Namibia was in 2010/11 and was conducted in partnership with the Ministry of Environment and Tourism. This study assessed

the national population status of leopards in Namibia and recommended that the trophy hunting quota of 250 remain unchanged. The study also put forward permit distribution and monitoring methods.

The recent IUCN Red List (2016) change in status for the leopard across its entire range highlights the importance of having rigorous scientific data from individual countries to put forward towards international assessments. As each country, including Namibia, has its own challenges, pressures and legislation that will impact the species it is critical that each country has its own dataset. It is therefore critically important that rigorous and nationally approved scientific evidence is obtained regularly to be able to drive policy and direction. This ensures that the decision making process is transparent as it is clearly based on the empirical evidence provided.

To conserve large carnivores, it is necessary to understand their abundance in human dominated landscapes, which is where the real conservation action is needed through an interdisciplinary and adaptive approach (Winterbach *et al.*, 2012). Balme *et al.*, (2013) in agreement with Winterbach *et al.*, (2012) also states that research projects should not only be multi-disciplined but also based outside protected areas and not just focused on one dimension. As such this study takes a multi-disciplinary approach, inside and outside national parks, by combining ecological methodologies and social science to understand the pressures on, and status of, the leopard population across Namibia.

“The Namibian Leopard: National Census and Sustainable Hunting Practices study in partnership with the Namibia Professional Hunting Association and the Ministry of Environment and Tourism” ran from September 2017 to March 2019. The study undertook three field work phases, two camera trapping surveys and one questionnaire survey. However, respondents were given the opportunity to complete and return questionnaires throughout the study period but attendance at farm meetings was during a specific timeframe. In conjunction with the field work the study collated additional leopard presence and density data from multiple organisations across Namibia. Sustainable use and human-leopard conflict data, collected in partnership with the Ministry of Environment and Tourism, was also collected

throughout the study. The results of this study were then compared to the 2010/11 project to determine if there had been any change to the leopard population during that time.

The study results identified that leopard densities across Namibia varied significantly. The central and northern camera trap surveys revealed a 40% and 16% a higher density than the 2011 study. However, in the south of Namibia, the density estimate from this study is 38% lower. The highest leopard densities were found in areas that had the highest density of game compared with livestock. Based upon current known leopard densities the study has highlighted that leopard densities in Namibia are higher in some areas of the freehold farmland when compared to National Parks. However, in comparison to other leopard densities recorded in South Africa, both inside and outside National Parks, Namibia's densities overall are still very low. Density data collected has shown that the 'high density' category assigned to the north-east of Namibia was miscategorised, the densities now recorded in the region are the lowest in Namibia to date. As part of this study the number of areas categorised as 'No known occurrence' in 2011 have significantly decreased by increasing the presence records, particularly in the east, south and south-east of Namibia. A proportion of these new presence records for the south east are also outside the current IUCN Red List distribution for leopard in Namibia (Stein *et al.*, 2016).

As a result of this study's findings and other additional density studies the 2011 density categories have now been updated and re-defined to reflect the changes to the leopard population captured post 2011 to present day. In 2011 the national leopard population was estimated to be 14,154 (Stein *et al.*, 2011b). It is important to recognise that the leopard population is not declining country wide, in the centre and north of Namibia across freehold farms between 2011 and 2019 there has been an increase in leopard density by up to 40%. However, due to a combination of the re-classification of the density categories based upon new data and lower leopard density in some areas of Namibia this study has determined that the leopard population figure is now at a lower estimate of 11,733.

In relation to human-leopard conflict the study confirms that problem leopard removal and the subsequent lack of reporting to the Ministry of Environment and Tourism is one of the greatest

threats to the Namibian leopard population. Over the duration of the study respondents reported removing 342 leopards compared to 196 leopards recorded by the Ministry of Environment and Tourism and the 183 reported in 2010/11. In the communal conservancies an average of 336 leopard conflict incidents were logged per year. Since 2011 the reporting rate of problem leopard removal by freehold farmers has declined by 5% to just 45%. Ensuring that livestock and game losses were off-set by economic incentives such as, tourism and trophy hunting, was shown to have a direct link to increased tolerance to leopard presence and lower conflict levels.

In 2017 a total of 650 problem leopard incidents were recorded from freehold farms (152) and communal conservancies (498), which would rise to 846 if it is assumed there is no overlap between the study's respondents and the Ministry of Environment and Tourism reporters. In comparison, 161 leopards were removed through trophy hunting in the same year. The highest number of trophy hunts took place in the freehold farmland, followed by communal conservancies, and National Parks. The areas shown to have higher leopard density, suitable habitat and prey availability had the greatest hunting success rates. On average 27% of the trophy hunts undertaken were successful across Namibia. Since the implementation of the new TAG system in 2011 the quota of 250 leopards has never been reached, 2017 was the highest at 161 (-35.6%). This study recommends that the trophy hunting quota for leopard should remain at 250 as long as hunting success rates remain unchanged. Even with the reduction in the leopard population figure, the actual number of trophy leopards hunted per year, up to 161, is still considered to be sustainable. By keeping the 250 quota it maximises the opportunity for farmers to off-set economic losses through trophy hunting which in turn leads to a reduction in problem leopard removals. By taking into account these key relationships and undertaking effective management action when needed the leopard population's sustainability can be ensured. However, where specific leopard data is available for a given area this information must be taken into consideration when allocating the quota across regions. Quantifying and managing both the reported and unreported removal of problem leopards must be made a priority across Namibia. Undertaking trophy hunting at a landscape scale in tandem with economic incentives may be one way to reduce removals in the freehold farmland. However, the issue of removals in the communal conservancies is also cause for concern, especially as few specific leopard studies have been carried out in these areas to inform decision making.

The information captured in this study will be presented to the CITES Secretariat, Animals and Standing Committees as part of a review of Namibia's leopard quota. The study's recommendations will also feed into the Ministry of Environment and Tourism's national management strategy plan for leopard as well as other national and international studies to ensure the long-term survival of leopard, not only in Namibia but across Southern Africa.

2. Introduction

2.1. Background to the Study

Global biodiversity is changing at an unprecedented rate (Sala *et al.*, 2000; Magurran and Dornelas, 2010; Pereira *et al.*, 2012; Ripple *et al.*, 2016), it is predicted that 37 % of terrestrial species will be lost by 2050 (Bradford and Warren, 2014). The rate of biodiversity loss become very clear when the conservation status of taxonomic groups is evaluated. For example, 25 % of mammals are categorised as 'Critically Endangered' to 'Vulnerable' on the IUCN Red List of Threatened Species, to be known here on as IUCN Red List (IUCN, 2017). Namibia has a total of 1,894 animals across 115 species on the Red List across the taxonomic groups of which 0.5 % are 'Critically Endangered', 1.3 % 'Endangered', 2.9 % 'Vulnerable' and 5.9 % are 'Data Deficient' (IUCN, 2017). To reduce the rate of biodiversity loss, it is important that conservationists have a clear picture of the status and distribution of species. Therefore, the first step in the conservation process is determining levels of biodiversity and whether species are threatened. Once that information is acquired, informed management decisions about how best to manage biodiversity in a sustainable manner can be delivered (Foresman and Pearson, 1998; Pettoirelli *et al.*, 2010).

2.1.1. Carnivore Conservation

Carnivores comprise 287 extant species in 123 genera belonging to 16 families (Karanth and Chellam, 2009) of which 59% of the world's largest carnivores (more than or equal to 15 kilograms) are classified as threatened with extinction (Ripple *et al.*, 2016). The decline in the conservation status of carnivores and ungulates was underway 40 years ago and has since accelerated (Di Marco *et al.*, 2014). Seoraj-Pillai and Pillay (2016) provides the first global

assessment of human-wildlife conflict in relation to the most high-scale conflict species which has led to them becoming severely persecuted. The assessment of the conservation status in relation to conflict species yielded several high to moderate scale conflict species listed on the IUCN Red List (Seoraj-Pillai and Pillay, 2016). The African carnivores that featured prominently, posing high-scale conflict consisted of the African lion (*Panthera leo*) (Vulnerable), African wild dog (*Lycaon pictus*) (Endangered), brown hyaena (*Parahyaena brunnea*) (Near Threatened), cheetah (*Acinonyx jubatus*) (Vulnerable), and leopard (*Panthera pardus*) (Vulnerable) (Seoraj-Pillai and Pillay, 2016).

Large carnivores are currently facing severe threats and are experiencing substantial declines in their populations and geographical ranges around the world (Ripple *et al.*, 2014). Out of 295 carnivore species that have been investigated by the IUCN, 1.4 % are listed as ‘Critically Endangered’, 10.8 % ‘Endangered’, 14.2 % ‘Vulnerable’ and 2 % are ‘Data Deficient’ (IUCN, 2017). Large carnivores are particularly vulnerable to extinction and are often the focus of conservation studies due to four common factors: 1) they are sensitive indicators of ecosystem integrity, 2) they are wide ranging, 3) they live in small isolated populations and are therefore prone to extinctions and 4) they suffer directly from human interference. Multiple factors have been found to influence the risks to large carnivore extinction ranging from; ecological (interspecific competition, ranging behaviour, prey availability, livestock predation); socio-economic (people’s attitudes and behaviours); and political (policy development and implantation, land use) (Winterbach *et al.*, 2012; Ripple *et al.*, 2014).

Habitat loss and degradation is currently one of the greatest threats to the survival of large carnivore species worldwide (Inskip and Zimmermann, 2009). Consequently, many carnivores exist in semi degraded or human dominated landscapes, which increases levels of human carnivore conflict. Therefore, conservation efforts which includes controlled hunting in mixed-use landscapes are crucial for sustaining viable carnivore populations (Schuette *et al.*, 2013). Di Minin *et al.*, (2016)’s study on the effects of global land use change determined that change will potentially lead to important range losses, particularly amongst already threatened carnivore species and that innovative interventions are required to conserve carnivores outside protected areas.

As the human population continues to grow the rate at which it consumes resources also increases leading to habitat loss. This inevitably brings people into close proximity with wildlife, leading to a rise in human-wildlife conflict (Inskip and Zimmermann, 2009). Human-wildlife conflict is a risk to 31% of carnivore species that are listed as either ‘Threatened’ or ‘Data Deficient’ by the IUCN Red List (IUCN, 2017). People’s attitudes towards wildlife can be determined by multiple factors including household wealth, residency status and type and extent of an outreach programme, if any (Holmes, 2003). Anthropogenic threats or human interference can impact carnivore populations either directly or indirectly (Burton *et al.*, 2012).

Carnivores are highly susceptible to human-wildlife conflict as they are wide ranging and their protein rich diet often negatively impacts on people’s livelihoods (Treves and Karanth, 2003; Inskip and Zimmermann, 2009). Livestock predation by large carnivores is the widest spread cause of conflict and retaliatory killing by people is one of the most serious threats to carnivore survival (Woodroffe *et al.*, 2005). Although all carnivores are affected by human-wildlife conflict, conflict has been shown to be most severe in relation to large cats; caracal (*Caracal caracal*), cheetah, leopard and lions as, apart from caracal, the other species have either a moderate or large body mass which is a significant factor affecting severity of conflict (Inskip and Zimmermann, 2009).

Often carnivores are subjected to indirect and direct effects of human conflict at the same time, usually exacerbating risks of extinction. For example, jaguar (*Panthera onca*) populations have been declining not only due to direct persecution by being hunted for their skin but indirectly by their prey base collapsing due to bushmeat hunting pressures (Wallace *et al.*, 2003). The Siberian tiger (*Panthera tigris altaica*) is also directly poached for medicinal purposes and their wild prey base is also in decline which combined has resulted in less than 400 individuals remaining (Carroll and Miquelle, 2006; Miquelle *et al.*, 2011).

2.1.2. Carnivore Ecology

Carnivores play a critical role in the ecosystem as they maintain biodiversity and function as well as often acting as keystone species (Linnell and Strand, 2000; Ripple *et al.*, 2014).

However, even for key apex predators such as the lion and leopard, relatively little is known about their ecological effects (Ripple *et al.*, 2014). The current research suggests that top predators promote species richness or are associated with it in relation to: dependence on ecosystem productivity; trophic cascades; resource facilitation; sensitivity to dysfunctions; selection of heterogeneous sites and links to multiple ecosystem components (Sergio *et al.*, 2008). Furthermore, carnivores may function as structuring agents and biodiversity indicators in certain ecosystems (Sergio *et al.*, 2008; Estes *et al.*, 2011). The removal of apex predators from the system may result in unknown fluctuations of other mesopredator and prey species, altering the dynamics of the ecosystem, for example ‘mesopredator release’ (Treves and Karanth, 2003; Blaum *et al.*, 2009).

Carnivores often exist at low density due to their relatively high position in food webs. Existing at low densities makes them more susceptible to extinction caused by demographic and environmental stochasticity, which can lead to local extinctions (Karanth and Chellam, 2009; Pettorelli *et al.*, 2009). Carnivore density has been shown to be significantly influenced by factors such as habitat fragmentation (Creel, 2001), prey availability (Macdonald, 1983; Kaunda and Skinner, 2003; Hayward *et al.*, 2007; Hayward and Kerley, 2008; Burton *et al.*, 2012) and inter-species competition (Linnell and Strand, 2000; Rich *et al.*, 2012).

One of the key questions in carnivore ecology is whether their numbers are regulated by their prey or whether they regulate their prey density? Prey density is a fundamental determinant of carnivore density both within and across species, therefore consistent prey density is critically important to ensure stable carnivore populations in the future (Carbone and Gittleman, 2002). Carnivore densities are closely tied not only to prey size but also to prey biomass in their preferred weight range (Carbone and Gittleman, 2002; Hayward *et al.*, 2007). When a carnivore reaches 21.5 kg or more it cannot be sustained with small prey items such as invertebrates as the necessary energy intake requirements exceed that obtained from the food, therefore larger prey items are required (Carbone *et al.*, 1999).

Carnivore densities are also influenced by competitive interactions with other carnivores in the community. Competitive predator interactions can be based on exploitation or interference

(Linnell and Strand, 2000). Interspecific competition can have strong influences on the distribution and abundance of carnivores and should be an essential consideration in their conservation (Creel, 2001).

Certain carnivores alter their activity patterns to avoid direct inter-specific competition (Hayward and Hayward, 2006). Behavioural factors leading to differential use of space can facilitate predator co-existence within an area (Creel and Creel, 1996). Hyaenas, leopards and lions are predominately nocturnal and so the cheetahs and wild dogs avoid the large predators by being crepuscular (Mills and Briggs, 1993). Large bodied carnivores present in a human dominated landscape may utilise the same instinctive behaviour towards the human population. For example, wolves (*Canis Lupus*) were influenced by intraspecific competition and availability of resource as well as anthropogenic threats (lethal control) (Rich *et al.*, 2012).

2.2. African Leopard Conservation and Sustainable Use

2.2.1. Overview

Leopards historically lived across approximately 35,000,000km² globally and 20,000,000km² in Africa, overall 63% - 75% of the leopard's range has been lost (Jacobson *et al.*, 2016). Leopard distribution now covers 8,515,935km² in 173 extant patches from sub-Saharan and north Africa to the Middle East and Asia (Stein *et al.*, 2016; Jacobson *et al.*, 2016) (Figure 2.1). Suitable leopard range has been reduced by >30% worldwide in the last three generations (22.3 years) (Stein *et al.*, 2016) and has been removed from nine countries, the highest of all sub-species (Jacobson *et al.*, 2016). The leopard now occupies 25% to 37% of its historic range with approximately 17% of that extant range being protected (Jacobson *et al.*, 2016). The leopard has the widest distribution of all the cats in sub-Saharan Africa (Henschel *et al.*, 2005; 2008; Stein *et al.*, 2016). Yet, the estimated regional range loss for leopards across Africa was 48% - 67% with regional variations being; Southern Africa: 21% - 51%, North Africa: up to 99%, West Africa: 86% - 95% (Stein *et al.*, 2016; Jacobson *et al.*, 2016). The 2016 leopard review determined that there is a limited amount of reliable data on changes in the leopard status (distribution or abundance) throughout Africa over the last three generations as well as

there being no robust estimates of the total number of mature individuals across their entire range (Stein *et al.*, 2016).

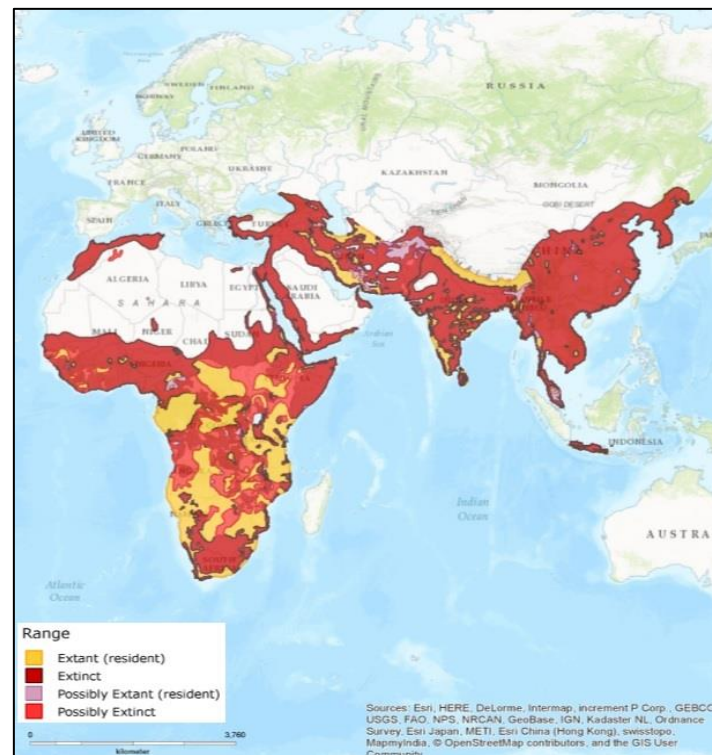


Figure 2.1. IUCN Red List global distribution of leopard showing historical and current range extent (Stein *et al.*, 2016).

Based on genetic analysis, nine leopard subspecies are recognised which includes the African leopard *Panthera pardus pardus* (Miththapala *et al.*, 1996; Uphyrkina *et al.*, 2001) and form part of a panmictic population (Stein and Hayssen, 2013). African leopards show the broadest range of genetic variation of all the leopard subspecies (Uphyrkina *et al.*, 2001, Castro-Prieto *et al.*, 2011). The leopard has been listed on CITES Appendix 1 since 1975 and is included under Appendix II of the Bern Convention. The leopard is also protected under the U.S. Endangered Species Act 16 United States Code, Section 1538 (Stein *et al.*, 2016) and from 2017 has been included in Appendix II of the Convention on the Conservation of Migratory Species of Wild Animals (CMS and UNEP 2017).

The African subspecies *Panthera pardus pardus* seems to be the least fragmented across its range showing an apparent healthy connectivity between the different populations (Jacobson

et al., 2016). However, populations in the western and northern range are highly threatened in comparison to populations in the eastern, central and southern areas of Africa (Jacobson *et al.*, 2016). Angola, Zambia, Zimbabwe and Mozambique have declining, but viable, leopard populations outside of human dominated areas while Botswana has had a continuous leopard population in the north and west (Stein *et al.*, 2016).

In 2008 the leopard was classified as ‘Near Threatened’ on the IUCN Red List (Henschel *et al.*, 2008). However, due to decline of the leopard populations by >30% over the last three leopard generations and was re-classified as ‘Vulnerable’ in 2016 (Stein *et al.*, 2016). This decline in population is the result of widespread habitat loss (21% in sub-Saharan Africa in 25 years) and prey loss inside African protected areas (Stein *et al.*, 2016). As stated, one main factor for the decline in the African population is related to declines in prey availability as leopard population density across Africa tracks the biomass of their principle prey species, medium and large-sized wild herbivores (Marker and Dickman 2005, Hayward *et al.*, 2007). Between 1970 and 2005 there has been an average decline of 59% in prey species abundance across 78 protected areas throughout West, East and Southern Africa which include key prey species for leopard (Stein *et al.*, 2016). Leopards are also under threat from habitat loss and fragmentation of their range as well as being hunted for the illegal wildlife trade, trophies and pest control (Henschel *et al.*, 2008; Stein *et al.*, 2016). Preliminary data suggest that 4,500 - 7,000 leopards are harvested annually as part of the illegal trade in leopard skins for cultural regalia, a practice that is extensive throughout Southern Africa (Stein *et al.*, 2016).

2.2.1.1. Leopard Ecology

An adult leopard weighs between 20 – 90 kg and as such the meat requirement ranges from 1.6 kg to 4.9 kg per day, which can lead to up to 60 prey items being killed per year depending on the geographical location (Hayward *et al.*, 2006). The leopard is an opportunistic ambush hunter with a catholic diet and the broadest diet of all the large predators of 92 prey species, they are non-selective predators but do show preferences in selecting prey (Hayward *et al.*, 2006). Leopard hunting success varies from 5 % success in the Serengeti (Tanzania) (Bertram, 1979) to 16 % in Kruger National Park (South Africa) (Bailey, 1993) and 38 % in Kaudom National Park (Namibia) (Stander *et al.*, 1997). Their prey base normally consists of medium-

sized ungulates species which range in size from 20 kg - 80 kg, in Gabon this size of prey accounted for 59 % of the total biomass consumed (Henschel *et al.*, 2005). Whereas on the African savannahs their prey included smaller species such as impala (*Aepyceros melampus*), bushbuck (*Tragelaphus scriptus*) and common duiker (*Sylvicapra grimmia*) (10 kg - 40 kg) (Hayward *et al.*, 2007). Their preferred prey weight is 25 kg or a mean body mass of 23 kg of individuals that are in small herds in dense habitat (Hayward *et al.*, 2006). In South Africa, out of a total 217 kills 185 were made up of six species and the mean number of kills per leopard was 7.37 ± 2.20 (Hayward *et al.*, 2006). Leopards regularly kill smaller carnivore competitors such as; cheetah, African civet (*Civettictis civetta*), black-backed jackal (*Canis mesomelas*) and occasionally brown hyaena, they also prey on baboons when larger prey is scarce (Hayward *et al.*, 2006). The majority of leopards hunt alone at night (Bailey, 1993) by stalking their prey and can sprint up to 120 m at speeds up to 60 kmh^{-1} (Bertram, 1979). Bothma and Le Riche (1984) determined that vegetation as low as 200 mm was sufficient to effectively conceal a leopard whilst stalking prey. On average leopards lose 5 – 10 % of their kills predominately to; lions, spotted hyaena (*Crocuta crocuta*), bush pig (*Potamochoerus larvatus*) and other leopards (Hayward *et al.*, 2006; Balme *et al.*, 2007) but this is compensated for by similar levels of scavenging (Bertram, 1979). However, leopards minimise kleptoparasitism by caching carcasses in trees or by putting it into thick bushes (Bertram, 1999; Stein *et al.*, 2015).

In Phinda Private Game Reserve, South Africa, leopards preferred hunting in habitats where prey was easier to catch rather than in areas with higher prey abundance (Balme *et al.*, 2007). Balme *et al.*, (2007) determined that leopards avoid grassland due to competition with lions and that beyond a certain threshold of vegetation density prey abundance alone will fail to accurately reflect its value for leopards. The leopard is highly adaptable and can survive across numerous landscape types as well as not being constrained by boundary fences and therefore freely moves across the landscape (Balme and Hunter, 2004; Balme *et al.*, 2007; Swanepoel *et al.*, 2013). Leopards are one of the few apex predators that occurs both within and outside protected areas and are the least affected by competition from lions and spotted hyaenas (Mills and Briggs, 1993).

Female leopards become sexually mature at 2.5 to 3 years old, whilst males reach sexual maturity between 2.5 and 4 years old (Bailey, 1993; Balme and Hunter, 2004). The sex ratio at

birth is assumed to be 50:50 (Clutton-Brock, 2016) however, males seem to have a higher mortality rate than females once reaching adulthood therefore, in the adult population, there are usually more females than males (Nowell and Jackson, 1996). Cubs are dependent from birth to 1.5-2 years (Bailey, 1993). In central Namibia there was found to be a two-year breeding cycle, this resulted in a temporary increase in the leopard density in the area every two to three years for a short duration.

2.2.1.2. Leopard Distribution

Leopards occurred historically throughout Namibia and their distribution remained similar from the 1930's into the 1980's (Shortridge 1934, Stuart and Wilson 1988). Currently, leopards inhabit most of the country except for the highly populated north-central region, the arid southeast farmlands and the desert coast and are absent from 30% of their historic range in Namibia (Hanssen and Stander 2004, Stein *et al.*, 2011b; 2016). The current IUCN Red List distribution for leopard can be seen in Figure 2.2. Leopard inhabit a wide range of habitats and climatic conditions, including; mountains, rocks, bushveld, woodlands, desert and semi-desert, forest, from sea-level to 2000m above sea-level, in areas of less than 100mm of rain to areas receiving above 1200mm of rain. They also occur in the Namib Desert where vegetation on banks of watercourses provides cover (Mills and Hes, 1997).

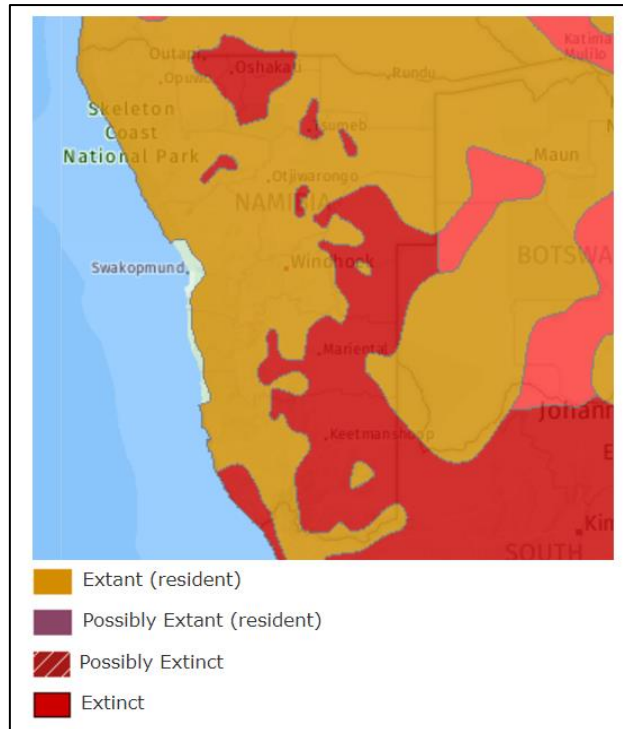


Figure 2.2. Outlines the IUCN Red List distribution of leopard in Namibia (Stein *et al.*, 2016).

Leopard home range sizes and densities of leopard vary with prey availability, habitat and degree of threat ranging from one per 100 km² to over 30 per 100 km² (Henschel *et al.*, 2008). Leopards are almost solitary, female territories are overlapped by larger territories of solitary males (Hayward *et al.*, 2006). Leopard home ranges fluctuate between 2,182 km² in the Central Kalahari (Bothma *et al.*, 1997) to 8.8 km² for a female in the rainforest habitat of Thailand (Grassman, 1999). In India, a human dominated landscape, the home range sizes were between 8 to 15 km² (Odden *et al.*, 2014). Leopards live in a complex land tenure system that is highly dependent on the stability of long-term relationships between individuals (Bailey, 2005). It has been determined that leopard resource use is governed by three key factors: avoidance of anthropogenic disturbance (such as roads and people), selection of prey-rich areas (such as river beds, protected areas and patches of recent rainfall) and selection of rocky areas with adequate vegetative cover to increase hunting success and minimise kleptoparasitism (Pitman *et al.*, 2017).

Leopards are sexually dimorphic, solitary and territorial (Voigt *et al.*, 2018). Male territorial boundaries in Namibia have been shown to be defined by natural features such as rivers, hills, dams and man-made structures such as roads (N/a'an ku sê Foundation, 2018 *unpublished*).

During drought periods in Namibia leopards have been found to utilise mountainous areas more as prey species move into these areas due to the availability of grazing (N/a'an ku sê Foundation, 2018 *unpublished*). High leopard densities in private reserves were due to the high resource availability and protection status (Noack, 2016).

Male territories normally encompass two to five female territories (Mills and Hes 1997, Hayward *et al.*, 2006). The degree of range overlap both between and within sexes can vary substantial (Stander *et al.*, 1997; Menges and Melzheimer, 2014). A ten-year study using VHF/GPS collars inside a private game reserve in Namibia has shown that over the course of a male's lifespan its territory size expands and contracts from 29km² to 120km² and back to 29km² (de Woronin Britz *pers. comm.*, 2018). During this time of change the number of females within the male's territory will rise and fall, from one (male 5-6 years) to four (male 8-10 years) and then back to one (13 years) (de Woronin Britz *pers. comm.*, 2018). Dispersal males can move into the territory of young territorial males which are still establishing themselves, kill and eat their cubs (de Woronin Britz *pers. comm.*, 2018). Males aged between 11-13 years start to become displaced when they lose territory to younger, neighbouring, males and will then remain on the boundaries between territories (de Woronin Britz *pers. comm.*, 2018).

In Namibia an adult male home range varies between 18.5km² in a private reserve to 451.2km² in the north-east (Table 2.1). For an adult female the smallest home range was also in a private reserve at 9.2km² with the largest found in the Hobatere concession at 224km² (Table 2.1). As the home range results highlight leopard territories in Namibia vary considerably in size (Table 2.1) and are directly related to prey abundance (Stander *et al.*, 1997, Marker and Dickman, 2005). The N/a'an ku sê Foundation (2018 *unpublished*) recorded their smallest female territory of 21km² around Windhoek, with their largest being recorded in the semi-arid area of the Tsauchab river system at 200km². The largest male territory of 240km² was found in the Okakarara area with the smallest 70km² again being recorded in the Windhoek area (N/a'an ku sê Foundation, 2018 *unpublished*). The results for the small female territory was corroborated with high density results from a camera trap survey in the same area. Overall the results showed that home ranges in the arid and semi-arid areas of Namibia's western-central region were substantially larger compared to those in the central and eastern regions of Namibia

as both prey and leopard density influenced male and female home ranges sizes (N/a'an ku sê Foundation, 2018 *unpublished*).

Table 2.1. Results of Namibian leopard home range studies; location, method, male and female home range sizes conducted between 2000 and 2018.

Location	Survey Method	Home Range Adult Male	Home Range Adult Female	Reference
Okonjima Nature Reserve – closed private reserve	Radio Collars	100.2 km ² (range 71.4 - 221.5 km ²) <i>n</i> = 6	72 km ² (range 70.8 - 73.2 km ²) <i>n</i> = 2	Stander and Hanssen (2000)
Hobatere concession	Radio Collars	131 km ² (range 94.9-166.9 km ²), <i>n</i> = 1	224 km ² (range 84.5-339.8 km ²), <i>n</i> = 5	Stander <i>et al.</i> , (2001)
Waterberg Conservancy	VHF Collars	229 km ² (SD ±95), <i>n</i> = 3	179 km ² (SD ±148), <i>n</i> = 4	Marker and Dickman (2005)
Central Namibia	VHF/GPS Collars	108 km ² , <i>n</i> = 1	40 km ² , 66 km ² , <i>n</i> = 2	Stein <i>et al.</i> , (2011a)
Okonjima Nature Reserve – closed private reserve	VHF Collars/ Camera Traps	18.3 km ² (SD ±10.1 km ²), <i>n</i> = 11	9.2 km ² (SD ±4.3 km ²), <i>n</i> = 13	Noack (2016)
Hardap, Khomas, Erongo, Otjozondjupa and Oshikoto	GPS Collars	150 km ² (range 70 - 240 km ²), <i>n</i> = 25	110 km ² (range 21 – 200 km ²), <i>n</i> = 17	N/a'an ku sê Foundation (2018 <i>unpublished</i>)
North-east of Namibia	GPS Collars	451.2 km ² (range 210-1,1164 km ²), <i>n</i> = 6	188.4 km ² (range 183-194 km ²), <i>n</i> = 3	Portas <i>pers. comm.</i> (2018)
Ongava Game Reserve	GPS Collars	190.6 km ² , <i>n</i> = 1	96.9 km ² , <i>n</i> = 1	Stratford <i>et al.</i> , (2018 <i>unpublished</i>)

2.2.1.3. Leopard Density

At the local scale, estimates of leopard population densities vary 300-fold. This broad spread makes reliably estimating population numbers from known geographic ranges particularly difficult (Jacobson *et al.*, 2016). Hanssen and Stander (2004) in the Namibia Large Carnivore Atlas estimated the leopard population to range between 5,469 and 10,610 animals. The Atlas's aim was to estimate distribution using data from questionnaires and estimate the population size. The Namibian leopard survey in 2011, over an area of 808,503km², resulted in a national

population estimate of 14,154 (13,356 - 22,706), and provided three density categories across the country (Stein *et al.*, 2011b). These were:

- northern Namibia: high density with 3.1 leopards/100km²,
- central Namibia: medium density with 2.0/100km², and
- southern Namibia: low density with 1.2/100km²

In the last 20 years several studies have provided leopard density estimates in Namibia using three methods: collars, spoor surveys and camera trap surveys (Table 2.2). The lowest leopard density in Namibia (0.25 leopards/100km²) was recently recorded in the Mudumu Landscape (Hanssen and Singwangwa, 2019 *unpublished*) (Table 2.2).

Table 2.2. Results of Namibian leopard density studies; location, survey method, density estimates conducted between 1997 and 2019.

Location	Survey Method	Density Estimate	Reference
Khaudum National Park and Nyae Nyae Conservancy	Spoor Survey	1.5 leopards/100km ²	Stander <i>et al.</i> , (1997)
Okonjima Nature Reserve - closed private reserve	GPS Collars	5.56 leopards/100km ²	Stander and Hanssen (2000)
Hobatere Concession and West Etosha National Park	Spoor Survey, GPS Collars	3.85 leopards/100km ²	Stander <i>et al.</i> , (2001)
Waterberg National Park	Camera Trap Survey	1.0 leopards/100km ² (SE \pm 0.7, 95% CI 0.8-1.5)	Stein <i>et al.</i> , (2011a)
Central Namibia	Camera Trap Survey	3.6 leopards/100km ² (SE \pm 3.6, 95% CI = 3-8)	Stein <i>et al.</i> , (2011a)
Bwabwata National Park	Spoor Survey	1.18 leopards/100km ² (sand ridges), 2.40 leopards/100km ² (omurambas)	Funston <i>et al.</i> , (2014)
Five freehold farms bordering the Tsau//Khaeb (Sperrgebiet) and Namib-Naukluft National Parks	Camera Trap Survey – Northern Area	0.9 leopards/100km ² (SD \pm 0.41)	Edwards <i>et al.</i> , (2015)

Location	Survey Method	Density Estimate	Reference
Five freehold farms bordering the Tsau//Khaeb (Sperrgebiet) and Namib-Naukluft National Parks	Camera Trap Survey – Southern Area	0.59 leopards/100km ² (SD ±1.15)	Edwards <i>et al.</i> , (2015)
Mudumu-North Complex	Camera Trap Survey	0.6 leopards/100km ² (SD ±0.54)	Hanssen <i>et al.</i> , (2015)
Okonjima Nature Reserve - closed private reserve	Camera Trap Survey	13.5 leopards/100km ²	Noack (2016)
Bwabwata National Park	Spoor Survey	1.27 leopards/100km ²	Hanssen <i>et al.</i> , (2017)
Southern section of Khaudum National Park	Camera Trap Survey	1.8 leopards/100km ² (SD ±0.40, 95% CI 1.11 - 2.50)	Portas <i>et al.</i> , (2018)
Hoanib River	Camera Trap Survey	1 leopard detected (density could not be determined)	Portas <i>et al.</i> , <i>in prep.</i> (2018)
Ongava Game Reserve	Camera Trap Survey	2.6-4.6 leopards/100km ²	Stratford <i>et al.</i> , (2018 <i>unpublished</i>)
Gondwana Canyon Park	Camera Trap Survey	0.64 leopards/100 km ² (SE ±0.36)	Edwards <i>et al.</i> , (2018)
Mudumu Landscape (Mudumu National Park and 3 conservancies)	Camera Trap Survey	0.25 (SD ±0.06) leopards/100km ²	Hanssen and Singwangwa (2019 <i>unpublished</i>)
Bwabwata National Park (Kwando Core Area)	Camera Trap Survey	0.85 (SD ±0.18) leopards/100km ²	Hanssen <i>et al.</i> , (2019 <i>unpublished</i>)
Bwabwata National Park (Multiple use area)	Camera Trap Survey	0.58 (SD ±0.21) leopards/100 km ²	Hanssen <i>et al.</i> , (2019 <i>unpublished</i>)

2.2.2. Human-Leopard Conflict

In a Seoraj-Pillai and Pillay (2016) study on global human-wildlife conflict patterns the leopard was the leading carnivore conflict species, as it featured in the greatest number of human–carnivore conflict case studies. This was due to a variety of reasons, first that leopards exhibit an array of biological and behavioural traits that render it a high-impact conflict species (Kissui, 2008). It is also a highly adaptable species which occupies the broadest geographic range and

is better equipped to utilise human-dominated environments, such as farms, than other large predators (Kissui, 2008).

In Namibia, 80% of the wildlife occurs on farmland, including leopard. Therefore, conflict exists between carnivores and humans due to predation on livestock and/or valuable game species (Stander *et al.*, 1997; Stein *et al.*, 2010; Menges and Melzheimer, 2014). This conflict has been ongoing over many years with Skinner *et al.*, in 1977 stating that “destruction following predation on domestic livestock and the leopard’s incompatibility with animal husbandry was the reason for the declining South African population.”

A study between 2008 and 2017 recorded 262 human-carnivore conflict cases through their farmer programme of which 52% were leopard related (N/a’an ku sê Foundation, 2018 *unpublished*). The average number of cases per year involving leopard was 38% with the highest being 74% in 2016 (N/a’an ku sê Foundation, 2018 *unpublished*). The programme has seen conflict increase since 2008 with the proportion of leopard associated conflict also rising, post 2015 more than 50% of all carnivore conflict cases reported (leopard, cheetah, brown and spotted hyaena, wild dog, caracal, lion), have been attributed to leopard (N/a’an ku sê Foundation, 2018 *unpublished*).

The legal killing of carnivores, including leopards, as a result of human wildlife conflict is permitted in Namibia. No. 4 Ordinance of 1975 states that if the species is deemed to be causing damage to stock or pose a threat to human life a permit can be obtained to dispose of the individual (Para (a) substituted by sec 8(b) of Act 27 of 1986, Ministry of Environment and Tourism, 2007). Problem animal removal can be undertaken by capturing or killing the animal with either chemical, mechanical and biological means, such as the application of the poison via the coyote getter, shooting by jackal cannon or gun trap (Section 61, 62, No. 4 of Ordinance 1975).

Stander *et al.*, (1997) reported a high mortality due to human-wildlife conflict in the north-east of Namibia and detected that 11 out of 15 leopards were killed by humans. In the Waterberg

Conservancy, north-central of Namibia, Stein *et al.*, (2010) found that farmers who employed at least one out of six livestock husbandry techniques reported 85% less conflict with carnivores. Leopards that are responsible for livestock depredation are generally specific individuals, often sub-adult males, that may at times prey on juvenile large stock, small stock and poultry (Hanssen *pers. comm.*, 2018). Individual leopards can enter night time enclosures designed to keep livestock safe due to their agility, climbing capability and ability to get through small gaps in mesh fencing (Hanssen *pers. comm.*, 2018). This can make it difficult to secure and protect livestock that are targeted (Hanssen *pers. comm.*, 2018).

Interviewed South African landowners felt that they lack control over the official process of dealing with livestock losses and that this frequently drove them to retaliatory killing to sort out the problem as quickly as possible (Grey *et al.*, 2017). This issue of not reporting is highlighted in a recent study on the use of poison in Namibia which revealed that out of the 412 freehold farmers interviewed they estimated that 67% ($n = 276$) of their peers purposefully killed a predator without the required permit over the past year (Santangeli *et al.*, 2016). In Namibia translocation has been utilised to alleviate conflict between landowners and leopards, six confirmed conflict leopards were relocated, all six established home ranges and four did not predate on livestock and reproduced successfully (Weise *et al.*, 2015). However, this is not a long-term solution as the number of suitable translocations sites are limited as well as information on the local population resident at those sites.

2.2.2. Sustainable Use

While the conservation value of regulated trophy hunting is recognised, it is important to note that there is a fine balance between sustainable and unsustainable offtake of leopards. For example, trophy hunting may selectively harvest large individuals with fitness-enhancing traits (Balme *et al.*, 2010; Ripple *et al.*, 2016). Poor management such as over-harvesting, corruption, or lack of reinvestment in conservation and development of local communities, could undermine the sustainability of trophy hunting and in turn threaten the species (Lindsey *et al.*, 2007). However, Grey *et al.*, (2017)'s study highlighted that a sense of economic value of the leopard is critically important across the range of stakeholders in a multi-use landscape. Legal consumptive use of leopards through trophy hunting is a means of generating revenue in remote

areas (Balme *et al.*, 2010).

Following on from Balme *et al.*, (2009)'s study it is vital to understand the population demographics of the leopard population in Namibia by looking at both sex and age ratios. Leopard trophy hunting targets adult male trophies and the sex ratio of problem animals removed is currently unknown. If the number of males decreases in the targeted population, adult sex ratios might become more biased towards females, potentially altering the social structure of the population (Loveridge *et al.*, 2007). An eleven-year case study (2002-2018) in a central Namibian private reserve showed that during the hunting period (6 years) the male to female ratio was 1:1 changing to 1:4 in the post hunting period (5 years) along with a decline in the number of dispersal males created (de Woronin Britz *pers. comm.* 2018). When a territorial male is removed from the territory by either trophy hunting or illegal activities it creates a 'vacuum' which is immediately occupied by the dispersal males in the area (Davidson *et al.*, 2011). As a male loses territory a female may then be sharing her territory with two males. This can result in infanticide and an unnatural ratio of males to females causing females to mate with the new neighbouring dispersal male (de Woronin Britz *pers. comm.* 2018). As such it is important to regularly monitor populations so changes in their structure can be determined and compensated for through adaptive management strategies.

2.2.2.1. Legislation

Namibia's Environmental Management Act No. 7 of 2007 states that Namibia's cultural and natural heritage including, its biological diversity, must be protected and respected for the benefit of present and future generations (Ministry of Environment and Tourism, 2007). Namibian trophy hunters must submit information on the age and sex of the leopard trophy to the Ministry of Environment and Tourism through the trophy hunting permit. On the 27th January 2016 a new record sheet (Schedule G) with the insertion of 114D regulations was implemented. To ensure that only males are being taken the 114D conditions regarding skin and skull of the hunted predator are as follows;

- scrotum of the hunted predator must be left attached to the skin to confirm the sex of the animal.
- skin found without an obvious scrotum attached will be treated as female and will not be allowed to be exported.

- skin of the hunted predator must be brought to the Ministry of Environment and Tourism office (CITES Office) for tagging and inspection before the export permit can be issued.
- skull of the hunted predator must be brought to the Ministry of Environment and Tourism office (CITES Office) for SCI measurements before the export permit can be issued.

2.2.2.2. Current Hunting Quotas

Leopards are included in CITES Appendix I. Trade of Leopard Skins and Products (CITES resolution 10:14) which is restricted to 2,483 individuals in 11 countries across sub-Saharan Africa (CITES, 2018a) (Table 2.3). Namibia has the 4th highest leopard quota within sub-Saharan Africa (CITES, 2018a) (Table 2.3). It is important to note that these are the maximum quotas that can be taken annually, the actual number of individuals removed annually will vary. For example, in 2013 Zambia placed a moratorium on leopard hunting while in 2016 South Africa suspended leopard trophy hunting for two years (Stein *et al.*, 2016). Late 2018 saw Tanzania lift its three year ban on trophy hunting including leopards. In 2014, sport hunting (including leopards) was banned in Botswana (Stein *et al.*, 2016) which is why Botswana is not listed as a country with a quota (Table 2.3). Prior to the sport hunting Botswana's quota was 130. Trophy hunting of leopard has been banned in Kenya since 1977.

Table 2.3. Outlines the assigned 2018 CITES leopard quota number and their distributed across the 11 sun-Saharan Africa countries (CITES, 2018a).

Country	Quotas	Specimens
Democratic Republic of the Congo	5	Skins
Uganda	28	Trophies / Skins
Malawi	50	Trophies / Skins
Kenya	80	Trophies / Skins
Mozambique	120	Trophies / Skins
South Africa	150	Trophies / Skins
Namibia	250	Trophies / Skins
Zambia	300	Trophies / Skins
Tanzania	500	Trophies / Skins
Ethiopia	500	Trophies / Skins
Zimbabwe	500	Trophies / Skins
Total	2,483	

In 1997 the CITES export quota for Namibia was set at 100 individuals which in 2004 was increased by 150% to 250 (CITES Resolution Conf. 10.14 (Rev.CoP13)). The quota was increased in 2004 as a result of the 7,745 population estimation by Martin and de Meulenaer (1988) from which an annual harvest of 332 animals or 4.2% of the population was calculated and determined to be a safe offtake level. Before Stein *et al.*, (2011b)'s report the annual hunting quota was still set at 250 individuals which represented 3 - 4% of the total adult male population. Stein *et al.*, (2011b) recommended that the quota of 250 (CITES Resolution Conf. 10.14 (Rev. CoP16)) was to remain with the introduction of an intensive monitoring programme to ensure that permits are distributed evenly across Namibia in accordance to the variation of the leopard density. Stein *et al.*, (2011b) suggested that in;

- High density areas, 0.5 adult male leopards / 100km² / year = 5.5 permits / 10,000km²
- Medium density areas, 0.35 adult male leopards / 100km² / year = 3.5 permits / 10,000km²
- Low density areas 0.21 male adult leopards / 100km² / year = 2.1 permits / 10,000km²

2.2.2.3. Current Hunting Regulations

The conditions applied to all trophy hunting of leopards (cheetah and lion) are set out in the Nature Conservation Ordinance (No. 4 of 1975: Section 83). Under the Nature Conservation Ordinance No. 4 of 1975 (Schedule 4 of 1975) the leopard is listed as a protected game species which means that illegal killing of a leopard, in or out of a national park, can result in a fine and imprisonment (Ministry of Environment and Tourism, 2007).

However, there is a caveat to No. 4 Ordinance of 1975;

1. The lawful holder of a permit granted by the Minister shall at any time hunt any specially protected game (Subsec (1) amended by sec 12 of Act 5 of 1996, Ministry of Environment and Tourism, 2007).

“Any person who contravenes or fails to comply with any provision of subsection (1) or any condition, requirement or restriction of a permit granted in terms of this section, shall be guilty of an offence, and liable on conviction to a fine not exceeding [R4 000] N\$500 000 or to imprisonment for a period not exceeding [four] years, or to both such fine and such imprisonment.” (Subsec (4a) amendment of section 27 of Ordinance No. 4 of 1975, as amended by section 9 of Act No. 27 of 1986 and section 12 of Act No. 5 of 1996, Ministry of Environment and Tourism, 2017).

2.2.2.4. Activity Patterns

Camera traps are frequently used to describe activity patterns from the date and time contained in the image metadata. In this survey camera traps allow the monitoring of multiple locations, 24 hours a day, for two months per survey area. Independent observations, usually taken 30 minutes between subsequent photos of the same species at the same camera location (O’Brien *et al.*, 2003), can be grouped by hour or by period of the day to describe activity. Individual identification is not necessary, and activity can therefore be described for all species photographed during camera trap surveys. As well as from time of day, some digital cameras are capable of recording environmental data such as temperature and moon phase.

The removal of problem animals and trophy hunts can take place between ‘the period from half an hour after sunset on any day to half an hour before sunrise on the following day’ (Problem

Animals: Section 38, No. 4 of Ordinance 1975) as such it is important to determine how leopard activity patterns fit into this structure in relation to hunting a specific individual of known sex.

2.2.2.5. Leopard Trophy Sizes

Evolutionary biology has shown that for wolves larger males had a predatory advantage and that larger size improved performance of a strength-related task (MacNulty *et al.*, 2009). In turn, sexual dimorphism in wolf size also explained why males were outperforming females, overall the study determined that bigger predators are overall better hunters (MacNulty *et al.*, 2009). Selective poaching of large tusks in the African elephant (*Loxodonta africana*) population has led to the tusk-less females passing on the genetic tusk-less trait to their offspring (McDonald, 2016) and this pattern can be found in other hunted species as well. Therefore, it is important to understand if hunting of leopards is causing any change to the size of male trophies, which firstly affects trophy size for the hunter and secondly a reduction in male size could be impacting the population dynamics between males, such as increased territorial disputes.

2.3. Study Scope

To conserve large carnivores, it is necessary to understand their abundance in human dominated landscapes, which is where the real conservation action is needed through an interdisciplinary and adaptive approach (Winterbach *et al.*, 2012). Balme *et al.*, (2014) in agreement with Winterbach *et al.*, (2012) also notes that research studies should not only be multi-disciplined but also based outside protected areas and not just one dimensional i.e. ecology or diet. Therefore, this study takes a multi-disciplinary approach inside and outside national parks by combining ecological methodologies and social science to understand the pressures on and status of the leopard population across Namibia. Updated information on the status of the Namibian leopard was urgently required as the last national census was conducted nine years ago.

2.4. Study Objectives

2.4.1. Objective 1 - Leopard Population

- i) To determine whether there is an overall difference of leopard density between three leopard density zones (low: south, medium: central, high: north (Stein *et al.*, 2011)) across Namibia utilising remote camera traps.
- ii) To determine a national population density of leopard based on the results from the camera trapping study and data provided by collaborating partners.
- iii) To ascertain if there is a variation in leopard activity patterns across the three leopard density zones using remote camera trap data.
- iv) To utilise local respondent knowledge to map the distribution of leopard using the presence-absence data gathered from the national questionnaire and *in-situ* Non-Governmental Organisations across Namibia.
- v) To contribute information to a Ministry of Environment and Tourism national leopard management strategy plan.
- vi) To set up long-term leopard monitoring of leopard populations using rotating camera trap surveys across key areas in Namibia.
- vii) To produce a report for Namibia Professional Hunting Association and the Ministry of Environment and Tourism and share data with relevant partners to benefit the long-term conservation of leopard in both Namibia and Southern Africa.
- viii) To set up partnerships to ensure that the long-term camera trap stations are monitored, maintained and the data is fed into the leopard management strategy plan.

2.4.2. Objective 2 - Human-Leopard Conflict

- i) To determine the level of human-leopard conflict across Namibia in conjunction with the influential factors that drive the attitudes, perceptions and actions of Namibian stakeholder groups towards leopard using a national questionnaire.

2.4.3. Objective 3 - Sustainable Hunting of Leopard

- i) To establish if leopard trophy sizes have changed based upon Ministry of Environment and Tourism permit data.
- ii) Leopard hunting tags based on Ministry of Environment and Tourism permit data;
 - a) To map the distribution of the permits
 - b) To compare issued tags against successful and unsuccessful hunts.
- iii) To provide science based recommendations to the Ministry of Environment and Tourism and Namibia Professional Hunting Association regarding leopard hunting methods and quota allocations as well as distribution of quotas, age classes and sex permitted to promote sustainably utilisation of the Namibian leopard population long-term.
- iv) To provide recommendations on alternative and complimentary monitoring methods for leopards which have the potential to become part of future permitting conditions.

2.5. Study Outcomes

A comprehensive report for Namibia Professional Hunting Association (NAPHA) and the Ministry of Environment and Tourism (MET) containing the analytical results based upon the study objectives listed in 2.4.

2.5.1. Leopard Population and Human-Leopard Conflict

- i) To enable the Ministry of Environment and Tourism National to develop a National Leopard Strategy Management Plan based on current data from this study and information drawn from other Namibian collaborators.
- ii) To set up long-term monitoring of the Namibian leopard population using the remote cameras traps donated to this study by being deploying and rotating the cameras across key areas. The long-term monitoring can be facilitated through a partnership of NAPHA, Ministry of Environment and Tourism and Namibia's academic institutions¹. The information produced from the survey sites can provide

¹ This partnership is currently a concept idea

long-term trend data on changes in leopard population that can be integrated into MET's adaptive management plan for this species.

- iii) To produce a report for NAPHA and MET, scientific papers with collaborating partners, and share data with relevant partners to benefit the long-term conservation of leopard in both Namibia and Southern Africa.
- iv) To utilise the data to inform wider scientific leopard studies such as the Southern Africa Leopard density project. The study's aim is to determine the impact of anthropogenic factors as drivers of changes in leopard density and population trends across Southern Africa.

2.5.2. Sustainable Hunting of Leopard

- i) A review by the Ministry of Environment and Tourism of the number of yearly trophy hunting permits based upon the study's findings.
- ii) To provide science based recommendations to MET and NAPHA regarding leopard hunting methods, quota allocations as well as distribution of quotas, age classes and sex permitted in order to promote sustainably utilisation of the Namibian leopard population long-term.
- iii) To provide recommendations on alternative and complimentary monitoring methods of leopards which have the potential to become part of future permitting conditions.

3. Methodology

3.1. Survey Areas

3.1.1. National Survey Area

The study aims to collect leopard data (presence, density, conflict, trophy hunting) from across Namibia. This will be achieved by firstly utilising the data captured by the study itself which worked at the local, regional and national level, secondly by collaborating with multiple

stakeholder groups from farming and hunting, to conservation research and government and, thirdly by acquiring information through published scientific articles on the Namibian leopard.

3.1.2. Camera Trap Survey Areas

The survey took place across two different survey areas; the Auas Mountains and an area north-east of Omaruru (Blue and red box highlighted in Figure 3.1). These areas were chosen as they were previously surveyed as part of the National Leopard Project in 2009-2011 (Stein *et al.*, 2001b). The repetition of these areas allows for a direct comparison to determine potential leopard density change over time. A third survey area was undertaken in 2009-2011 (Black box highlighted in Figure 3.1), the re-surveying of this area was not undertaken by this study as a survey recently published by Edwards *et al.*, (2015) in the specific survey area can be used as the comparative data source. The details of each of the three survey areas can be seen in Table 3.1.

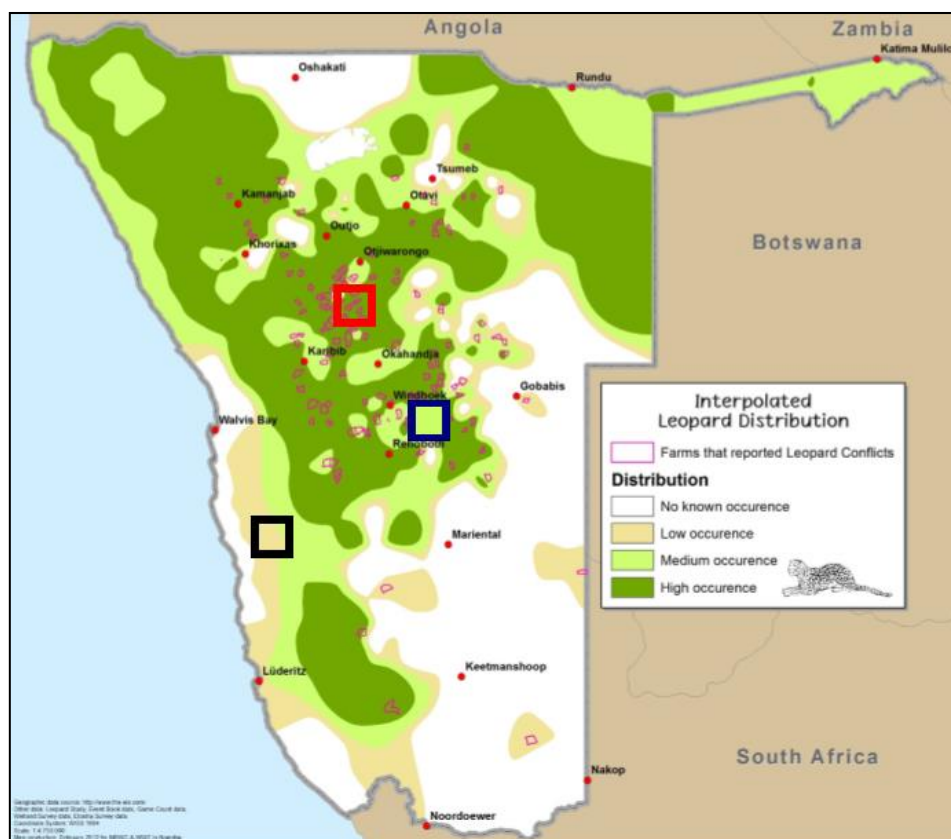


Figure 3.1. A map to show the 2011 leopard distribution, density and location of the three camera trap survey areas for Stein *et al.*, (2011b) and the 2017/18 survey.

Table 3.1. Overview of camera trap survey 1 undertaken by Edwards *et al.*, (2015) and surveys 2 and 3 completed as part of this study.

	Survey 1	Survey 2	Survey 3
Location	Namib-Naukluft/Tsauu//Khaeb National	Auas Mountains	Omaruru
Date	May - July 2013	September - November 2017	July - October 2018
Season	Dry	Dry	Dry
Survey Area Size	1281 km ²	1226 km ²	1200 km ²
Number of Farms	5	11	10
Land use type	Livestock and Game	Livestock, Game, Hunting, Tourism, Commercial Conservancy	Livestock, Tourism, Hunting and Game
Ownership	Freehold	Freehold	Freehold
Habitat Type	Southern desert / Dwarf shrub transition	Highland Shrubland	Thornbush Shrubland
Number of Sites	51	50	50
Number of Nights	60	60	60
Leopard Density (Stein <i>et al.</i>, 2011b)	1.2 leopards/100km ²	2.8 leopards/100km ²	3.1 leopards/100km ²

3.2. Characteristics of the Leopard Population

3.2.1. Density and Population Structure

Wildlife surveys have been greatly enhanced by the development of remote camera traps (O'Connell *et al.*, 2011). The benefits of using remote cameras are numerous; a key factor is the capture, confirmation and monitoring of rare and elusive species (Karanth and Nichols, 1998; Cuttler and Swann, 1999; Carbone *et al.*, 2001; Swann *et al.*, 2004; Kittle *et al.*, 2017) particularly when the species is located across large remote areas (Culter and Swann, 1999; Parker *et al.*, 2008). The method is non-invasive and produces little disturbance to the survey area or individual target animals (Maffei *et al.*, 2004).

Remote camera traps have become a preferred tool for sampling animal populations (Wemmer *et al.*, 1996). They can be left unattended in the field for extended periods of time, and thus are ideally suited for studying rare, elusive, and nocturnal animals that avoid humans (Tobler *et al.*, 2008). The use of camera traps in the study of wild animals has improved our understanding

of their ecological relationships and population dynamics (Rowcliffe and Carbone, 2008). The advantage of camera trapping in comparison to other methods used to record medium-sized to large terrestrial mammals is that photographs provide objective records of an animal's presence and identity (Rovero *et al.*, 2010). Moreover, camera trapping provides information on activity patterns, behaviour, and pelage characteristics that enable individual identification (Rovero *et al.*, 2010). Using remote camera traps to determine leopard density is a recognised biomonitoring methodology, multiple studies in Namibia have successfully utilised this type of survey design to ascertain leopard density in specific areas (Stein *et al.*, 2011ab; Edwards *et al.*, 2015, Portas *et al.*, 2018, Hansen *et al.*, 2019 *unpublished*).

Population density is a key ecological variable, and it has recently been shown how captures on an array of traps over several closely-spaced time intervals may be modelled to provide estimates of population density (Efford *et al.*, 2009). Camera trapping photographs are a common source of capture–recapture data as they ‘trap’ an individual animal in a photograph. Therefore, trapping period means ‘sampling an animal population with camera traps set for a known time, at known points in the habitat’. Time is usually divided into discrete intervals, and new animals may be captured, marked and released on each occasion within a photograph. Closed-population encounter histories are coded in binary form: on each occasion, an individual is either captured (1) or not captured (0) (Otis *et al.*, 1978). A spatial encounter history also records the location of each capture.

The first camera trap survey was undertaken in the Auas Mountains between 13th of September and 22nd of November 2017 (Table 3.3). The second survey was completed in an area north-east of Omaruru between 30th July and 7th October 2018 (Table 3.3). Both surveys were undertaken across freehold farms after obtaining permission from each landowner. Information on the farms such as farm size, fence type, land use activities, livestock and game numbers and problem animal issues were acquired from each landowner involved in the surveys to provide a baseline of information about the two survey areas (Auas Mountains and Omaruru). The surveys were conducted during the dry season. Camera trap surveys should include areas much greater than the home range of a single leopard as one cannot estimate population density by sampling at the scale of one animal. As such the camera sites were placed a maximum of 3km apart across the survey areas to create a trap matrix. This meant that at least one camera trap site was present per mean female leopard home range of between 30km² (home range = 30km²:

radius 3,090 m, diameter 6,160 m, $\frac{1}{2}$ home range = 3km) (Braczkowski *et al.*, 2016) to 40km² (Stein *et al.*, 2011a). Clusters of traps increase the expected number of spatial recaptures of individuals while the large spatial extent increases the expected number of unique individuals detected (Sun *et al.*, 2014). Therefore, the survey utilised a clustered trap configuration as it generally yields the most accurate estimators of abundance (total number of re-captures) (Sun *et al.*, 2014).

Each farm was visited and pre-surveyed, a week prior to commencement, accompanied by the landowner or manager to gain their insights into known leopard hotspots across their property. During these weeks over 200 potential camera trap sites were logged per survey area. Based upon the expert knowledge acquired during the pre-survey and the constraints of the survey design 50 camera trap sites were chosen for both areas (Figure 3.2 and 3.2).

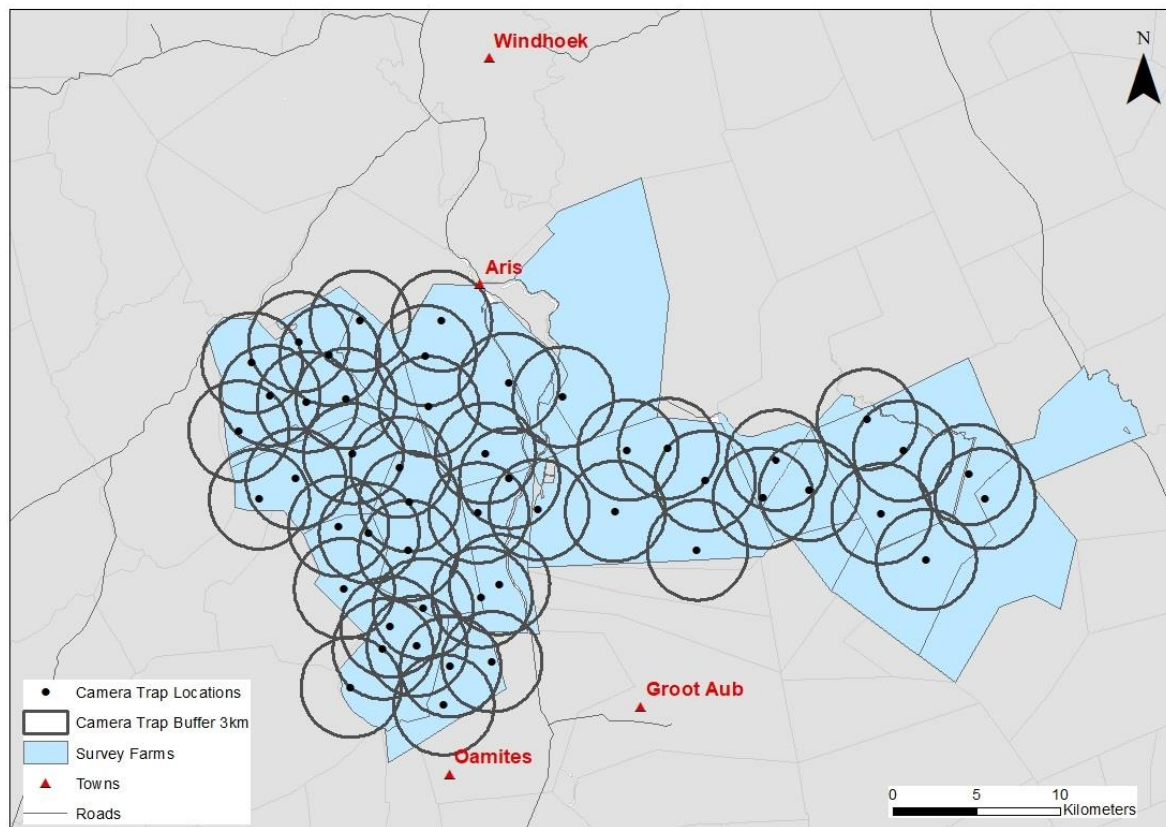


Figure 3.2. A map identifying the locations of the camera trapping sites in the Auas Mountains ($n = 50$).

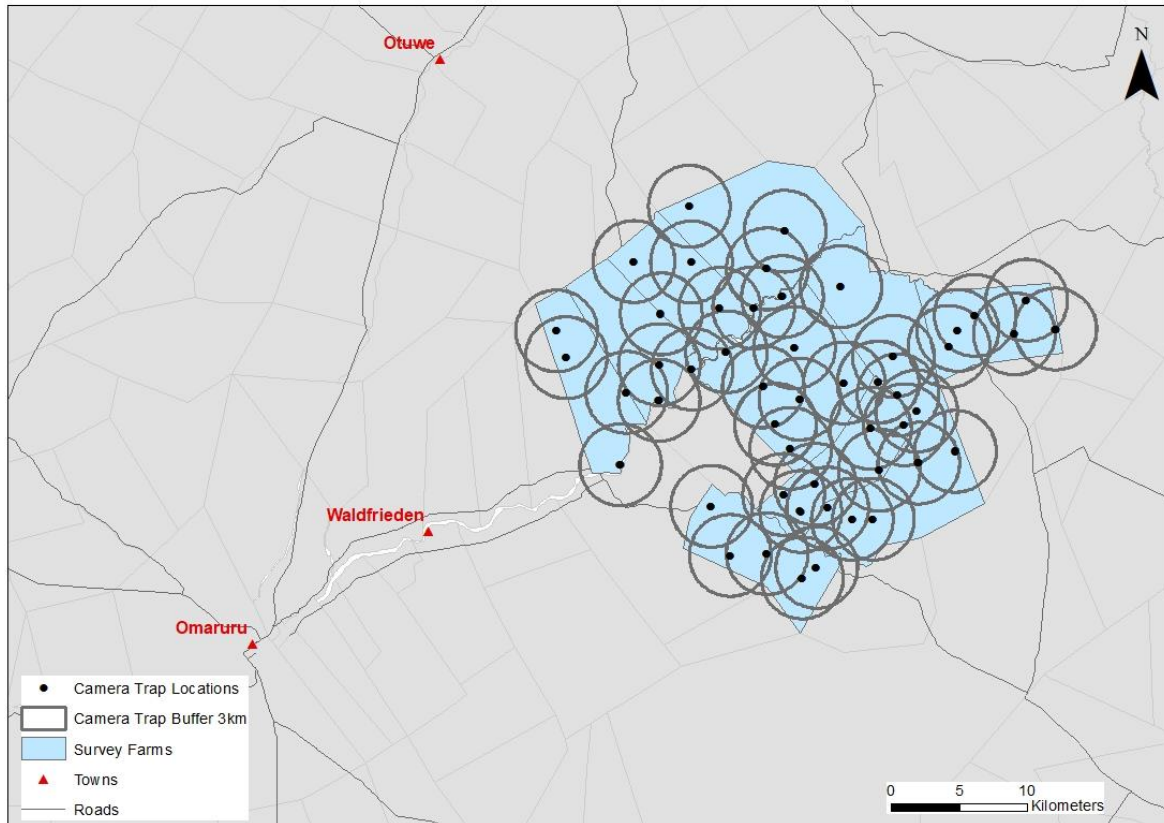


Figure 3.3. A map identifying the locations of the camera trapping sites in the area north-east of Omaruru ($n = 50$).

A paired camera set up (100 cameras) was utilised to maximise the opportunity to capture both the right and left flank of the leopard simultaneously. The cameras were offset three to four metres on opposite side of roads, game paths, water troughs, and dry riverbeds that are known high use areas for leopards (Figure 3.4).

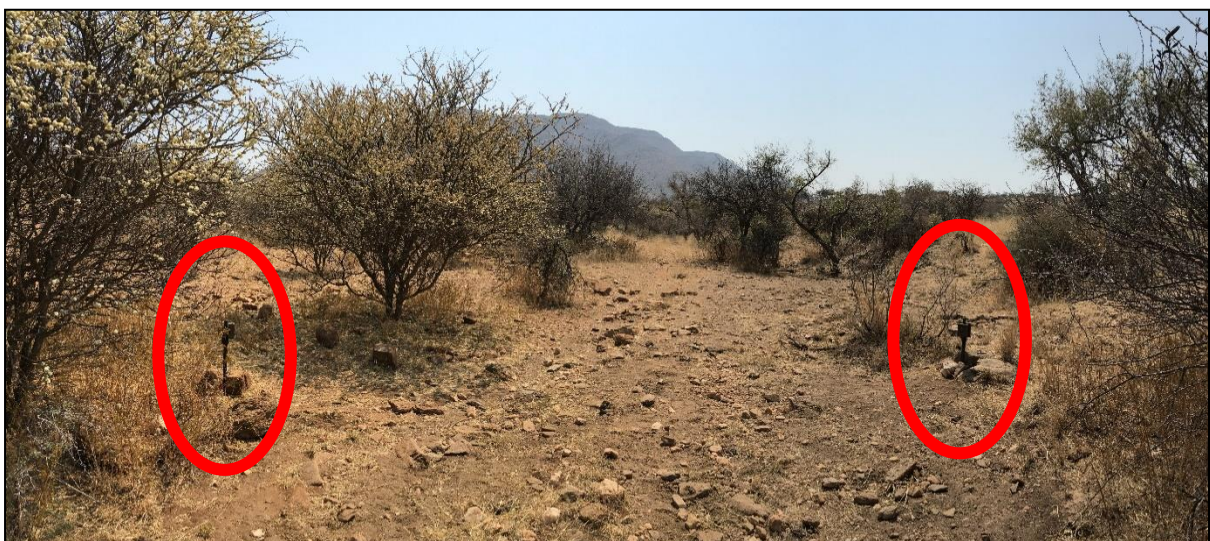


Figure 3.4. An example of the offset camera trap placement (red circles) along a small dry river bed and game trail in the Auas Mountains survey area.

No bait or lure was used in this study as this ensures standardisation of data collection with other leopard density studies in Namibia. The camera traps were set up at each survey area for 60 nights. The passive remote camera trap used for this study was a Minox DTC 550 which has an Invisible IR-flash with a range of 15 m / 49 ft. Every camera was fitted with a protective metal case to avoid damage by wildlife and placed on a purpose-built metal camera trap pole to keep the camera positioning secure. All the cameras were monitored throughout the survey periods, batteries and SD cards were regularly checked and changed to ensure continuous sampling over the 60 nights.

In the Auas Mountains 66% ($n = 33$) of the sites had leopard present with Omaruru having a higher proportion at 86% ($n = 43$). The leopard images for both surveys were sorted into useable and unusable images. Images where leopards had walked too close to the camera or the leopard image had blurred due to its movement were discarded. The remaining images were then used to individual identify the leopards through their unique rosette patterns on both their left and right side (Figure 3.5) and other distinctive features such as torn ears or scars.



Figure 3.5. Shows an adult male leopard right and left flank being captured almost simultaneously in Omaruru as a result of the paired camera design.

The abundance of elusive animals with individually distinct fur patterns, like the leopard (Figure 3.5), living at low densities is best estimated in the conceptual framework of capture-recapture models, which can provide estimates of encounter probabilities and abundance along with associated statistical errors.

Survey area density estimates were calculated using SPACECAP (Gopalaswamy *et al.*, 2012). SPACECAP is a Bayesian spatially explicit capture-recapture (SECR) model that uses several inputs from camera traps to model the variation in density across a study site. Inputs include the day, trap location and ID of each animal that was identified; trap locations (and the days that they were operational) and ‘home range centres’, a regular grid of points across the study space (S) which represent any points that an animal could originate from. In this case, S was created by taking a convex hull around camera trap locations and adding a 5km buffer (comparable to average half maximum mean distances moved identified within other studies) in order to ensure that all leopards captured originated within this area (meeting the requirements of a closed population). Home range centres were spaced at 2km intervals within S such that SPACECAP would provide leopard density predictions per 4km², where home range centres can be thought of as the centroid of 2km*2km pixels in this case. For each model, 60,000 Monte Carlo iterations were run with a burn-in rate of 20,000 (equivalent to discarding this many iterations from subsequent statistics), a thinning rate of 10 and a data augmentation value equal to 5 times the number of individuals observed (Ramesh *et al.*, 2017). Models were trained assuming no response to traps from leopards (i.e. behaviour remains unchanged after encountering a trap), in spatially explicit capture format, using the Bernoulli distribution to model the underlying distribution and a half-normal detection function to estimate detection rate at each of the home range centres.

3.2.2. Habitat Suitability

Species distribution models are a class of methods that use occurrence data in conjunction with environmental data to make a correlative model of the environmental conditions that meet a species’ ecological requirements and predict the relative suitability of habitat. The study applied maximum entropy modelling (MaxEnt) to build distribution models and identify environmental predictors for leopard across Namibia. MaxEnt software uses the principle of maximum entropy on presence-only data to estimate a set of functions that relate environmental variables and habitat suitability in order to approximate the species’ niche and potential geographic distribution (Phillips *et al.*, 2006). From a set of environmental (e.g., climatic) grids and georeferenced occurrence localities, the model expresses a probability distribution where each grid cell has a predicted suitability of conditions for the species. MaxEnt software is one of the most commonly used methods for inferring species distributions, including carnivores, and environmental tolerances from occurrence data (Kalle *et al.*, 2013; Abade *et al.*, 2014;

Kabir *et al.*, 2017). As such this study has utilised MaxEnt in order to understand the association between leopard occurrence and specific environmental factors.

To undertake the habitat suitability modelling using MaxEnt software this study collaborated with Dr. Vera De Cauwer, Senior Lecturer of Agriculture and Natural Resources Sciences at the Namibian University of Science & Technology. Occurrence (presence) data, land use type, livestock and game density were supplied by this study and additional occurrence data was added from the Atlasing project (EIS, 2018). Environmental variables were chosen based upon their known impact on leopard presence such as habitat type, altitude and climate e.g. rainfall. Dr. Vera De Cauwer then undertook the analysis of the data and supplied the results (Section 5.4.3.).

When running the model 666 presence records were used to train the model and 10,659 records were used to determine the MaxEnt distribution. The AUC (area under the curve) test was 0.870, an AUC value above 0.7 shows that model was accurate. The model started off with 25 different environmental variables, based on the results variables, where removed when they were found to be either uninfluential or highly correlated to one another. The final model was run with 19 environmental variables, the list of variables and estimates of their relative contributions of the environmental variables to the model can be seen in Appendix 3 Table 8.3. The output of the MaxEnt model was a probability distribution that sums to one, this gives the relative probability of observing leopard in each cell. Cells with environmental variables close to the means of the presence locations have high probabilities. The white dots show the presence locations used (Figure 5.7). The image uses colours to indicate predicted probability that conditions are suitable, red indicates high probability of suitable conditions for the species, green indicates condition typical of those where the species is found, and the lighter shades of blue indicates low predicted probability of suitable conditions (Figure 5.7).

3.2.3. National Population Size

This study's contribution to the updating of the leopard presence map was from the following data sources;

- National questionnaire (leopard presence points)
- Individual landowner camera trap photographs of leopard (year and farm information)
- The Ministry of Environment and Tourism problem leopard records 2005 to 2018 (year, farm name, farm number)
- The Ministry of Environment and Tourism successful trophy hunt records 2001 to 2018 (year, farm name, farm number)

In addition to the data collected by this survey leopard presence data for the map was obtained from the Atlasing project in Namibia (<http://www.the-eis.com/atlas/>). The Atlasing project contains data from various sources including records submitted by members of the public, individual scientists, research projects and other projects. The minimum information required is date, location and species. Locations can be coordinates, or one of the grid systems used in Namibia (monad 1 x 1km; pentad 5 x 5km; quarter degree square 15' x 15'). Record types for mammals are: anecdotal evidence, camera trap, capture, heard, sighting, specimen, spoor/scat, telemetry reading, and tracking devices. Leopard data in the system ranged from 1993 to present.

To derive the leopard presence map, data was downloaded from the Atlasing system and the spatial reference of each record was converted into a quarter degree square (QDS) reference. Records were disaggregated into historical (pre-1960); recent (1960-2007) and current (2008 onwards). Shapefiles were created out of these layers and mapped using Adobe Illustrator. The IUCN base map layers were obtained from the IUCN Red List (<http://maps.iucnredlist.org/map.html?id=15954>).

To scale up estimates to a national figure (and density map), several environmental variables were used alongside SPACECAP produced density estimates to train a Random Forest (RF; Breiman, 2001) model (Baines *et al.*, 2019 *in prep.*). RF, part of the Classification and Regression Trees (CART) family, is a machine-learning method that has proved popular for a wide variety of applications, including species distribution modelling (e.g. Howard *et al.*,

2014). A total of 17 environmental variables spanning the entirety of Namibia were derived (Table 3.2), and 16 chosen for input into the RF model after correlation analysis (removing highly correlated variable pairs, with a Pearson correlation > 0.8). All environmental variables were derived using Google Earth Engine (GEE) and were aggregated to a 2km resolution (4km^2) to match the resolution of SPACECAP density outputs. The RF model utilised the default number of variables to try per split (*mtry*) of $n/3$ (i.e. a third of the total number) and a minimum node size of 1, whilst 1000 trees were used to train the model as a higher number of trees has been shown to produce better results.

Table 3.2. Environmental variables derived used for training a Random Forest model examining variation in Leopard Densities. Variables in italics are those not included within the final model.

Environmental variables	Source (original resolution)
Mean aspect per 2km pixel	GMTED2010 (~225m)
Range in aspect values per 2km pixel	GMTED2010 (~225m)
Cattle density per 2km pixel	Ministry of Land Reform (shapefile)
Distance to nearest river	Ministry of Land Reform (shapefile)
Distance to nearest road	Ministry of Land Reform (shapefile)
Mean elevation per 2km pixel	GMTED2010 (~225m)
Range in elevation values per 2km pixel	GMTED2010 (~225m)
Land ownership per 2km pixel	Ministry of Land Reform (shapefile)
Modal land cover per 2km pixel	GlobCover 2009 (300m)
Number of unique land covers per 2km pixel	GlobCover 2009 (300m)
<i>Mean NDVI per 2km pixel</i>	<i>MODIS Aqua/Terra (250m)</i>
Range in NDVI values per 2km pixel	MODIS Aqua/Terra (250m)
Mean annual precipitation per 2km pixel	WORLDCLIM v1 (~1km)
Mean summer temperature per 2km pixel	WORLDCLIM v1 (~1km)
Mean slope per 2km pixel	GMTED2010 (120m)
Range in slope values per 2km pixel	GMTED2010 (120m)
Mean topographic diversity per 2km pixel	ALOS Topographic Diversity (270m)

After training the model, SPACECAP derived density figures were found to account for only a subset of key environmental variables (Table 3.2) (e.g. rainfall; Martin and Meulenaer, 1988). This is problematic as RF, unlike regression, cannot predict values lying outside the range of

the training dataset. To improve the representativeness of inputs therefore, density figures derived from published studies were also incorporated into estimates. These density figures, unlike those from SPACECAP, are much more spatially coarse, recording a single density value per study site. To establish a relationship with environmental variables (and to subsequently incorporate into the RF model), the mean value of environmental variables within each study site were taken and used as an input. As this resulted in just a single value per each of these study sites, significantly fewer than the number of inputs per SPACECAP site, each of these areas was oversampled to ensure inclusion into the RF model. Inclusion of these values was achieved by repeating the entries of each non-SPACECAP site within the model to inflate the total number of entries. The inclusion of this data was found to increase the total population estimate compared with SPACECAP figures alone. Inclusion of these values continued, repeating the number of times each figure appeared in the input dataset until further increasing the presence of these mean density estimates into the model no longer increased the total population estimate (i.e. total population plateaued). The data used for this model was acquired from the following sources; Stander *et al.*, (2001), Stein *et al.*, (2011a), Edwards *et al.*, (2015), Portas *et al.*, (2018), Stratford *et al.*, (2018 *unpublished*), Hanssen and Singwangwa, (2019 *unpublished*), and Hanssen *et al.*, (2019 *unpublished*), Auas Mountains and Omaruru camera trap areas, the site locations are shown in Figure 3.6.

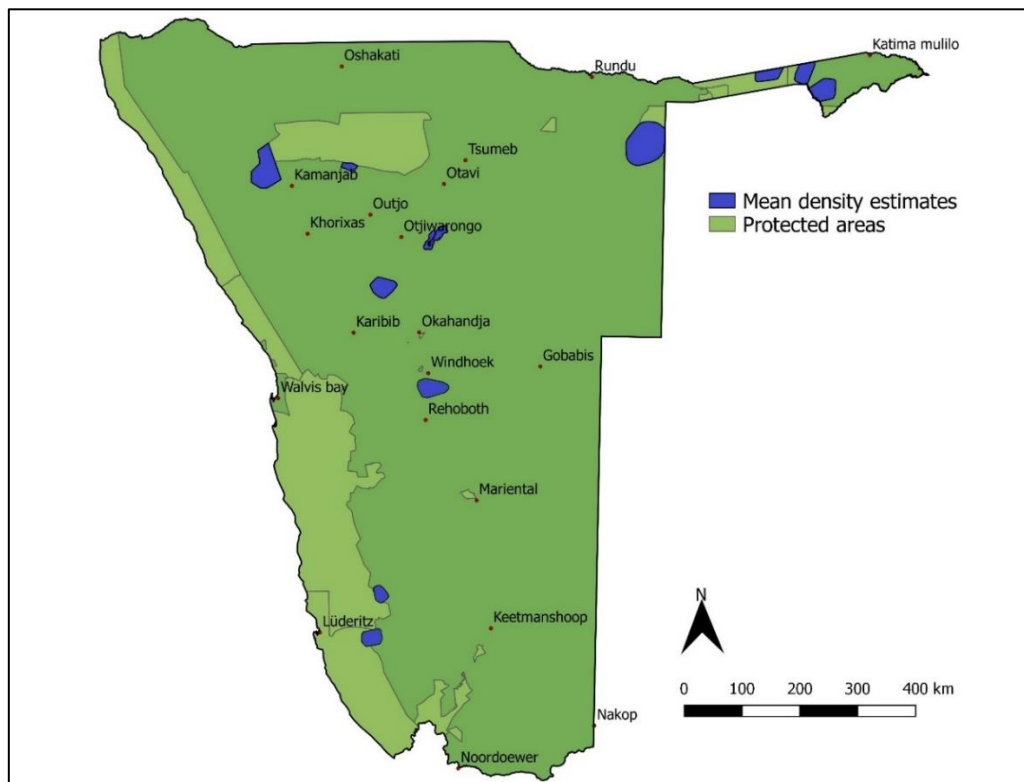


Figure 3.6. Locations of site for which density values were utilised within this study.

3.2.4. Activity Patterns

To determine the activity patterns for leopard across Namibia data from this study's two camera trap surveys was utilised as well as data from the Wiesel and Edwards (2014) human-carnivore-conflict study on farms bordering the Namib Naukluft and Tsau//Khaeb National Parks. Each camera trap photograph was categorised by hour. As one leopard could create three photographs and another 20 due to the time they spent in front of the camera only the first image of each time event was used to avoid any bias. The hourly categories were then defined across four groups; twilight pre sunrise, day, twilight post sunset and night. These groups reflect the trophy hunting time regulations as twilight pre sunrise is the 30 minute period pre sunrise and twilight post sunset is the period of 30 minutes after sunset. As the twilight times can vary by day, month and year, every date was individually categorised. This is why overlap between the twilight categories and the day/night categories has occurred (Figure 6.12).

3.3. Human-Leopard Conflict

3.3.1. National Questionnaire

The management of natural resources and conservation of threatened species often relies on the successful management of people's behaviour (Romanach *et al.*, 2007; St John *et al.*, 2012), as people's perceptions of human-wildlife conflict are critical to managing the conflict (Sillero-Zubiri *et al.*, 2007). Therefore, identifying the key stakeholder groups ensures that interventions are group-specific and effective (St John *et al.*, 2012). Equally, knowing who to target is imperative to maximise results within a limited budget. Thus, it is necessary for conservationists to understand how landowners react to the presence of wildlife such as carnivores on their land as this information can be used to develop mitigation plans that may reduce human-carnivore conflict (Inskip and Zimmermann, 2009).

Questionnaire based attitudinal surveys have led to several conclusions with direct relevance to this study; most findings indicate that, compared to areas of livestock farming, conservancies are more positive towards the presence of carnivores (Thorn *et al.*, 2009). Thorn *et al.*, (2013) determined that the major influencing factors in human-wildlife conflict were high elevation,

mixed purpose farming (i.e., stocking both game and livestock), dense vegetation cover and high perceived financial losses. For example, in Namibia, carnivore presence was tolerated in areas where income from wildlife was higher, income from livestock was lower, and financial losses from livestock depredation were lower (Lindsey *et al.*, 2013). Tolerance for losses is strongly influenced by socio-economic factors. For example, financial loss is a determinant of lethal control being undertaken in retaliation for livestock killings (Sillero-Zubiri *et al.*, 2007; Thorn *et al.*, 2012; 2013). Furthermore, farmers have been shown to undertake a range of predator control methods to protect their livestock and game from perceived predation events (Lindsey *et al.*, 2005; Blaum *et al.*, 2009), with target species varying between geographical locations (Lindsey *et al.*, 2005; Blaum *et al.*, 2009). The consensus is that, for attitudes towards wildlife to be positive, landowners need to achieve economic benefit in the form of ecotourism, benefit from a compensation scheme if livestock is lost, or be provided with financial incentives for predator conservation (Romanach *et al.*, 2007).

The distribution of the respondents surveyed must be representative of the study area otherwise geographical bias can skew results (Groves 1988; Groves and Peterson 1992). In a similar observation Lovell *et al.*, (1998) conceived that the questionnaire did provide a randomly sampled data set and, with the information provided, informed management decisions could be completed. Other forms of bias that are little explored in relation to natural resource management are the false consensus effect and a person's knowledge of the rules; as people's perceptions of the law vary so will their responses (St John *et al.*, 2012). Giving respondent's anonymity assists with the discussion of sensitive topics such as predator control methods (St John *et al.*, 2012). There is a need to be aware that biased sampling may result if the response rate is not the same for the different categories of interest on the questionnaire (Rushton *et al.*, 2004). For example, respondents who may be involved in harmful behaviours may be unwilling to discuss that specific topic, most particularly if the activity is illegal (St John *et al.*, 2012; Nuno and St John, 2015).

Evidence from recent, related investigations shows that questionnaires have become a valuable tool for investigating human-wildlife conflict (Newmark *et al.*, 1994) and landowner attitudes towards carnivores (Zimmermann *et al.*, 2001; Lindsey *et al.*, 2005; Arjunan *et al.*, 2006; Balme *et al.*, 2009; Stein *et al.*, 2011b; St John *et al.*, 2012; Thorn *et al.*, 2012). For the purposes

of an investigation into the conflict between humans and carnivores the assessment of people's attitudes to carnivores is clearly valuable. Reflecting the objectives of this study, White *et al.*, (2005) emphasised that questionnaires enable researchers to 'quantify human behaviour, for example perceptions or attitudes towards conservation strategies and/or the implantation of environment conservation directives.' A questionnaire can capture three types of attitudes; affective (feelings and emotions), cognitive (beliefs) and behavioural (Winter *et al.*, 2005; 2007), all of which are necessary to understand the mechanisms and thought processes behind human-wildlife interactions and conflict. As Santangeli *et al.*, (2016) recently showed that by utilising questionnaires it is possible to determine the relationship between freehold farmers, their use of poison and the carnivores that occupy their land.

Questionnaire data can also reveal the distribution of species efficiently (Groves and Peterson, 1992; Lovell *et al.*, 1998; Lariviere *et al.*, 2000; Nunez-Quiros, 2009; Karanth *et al.*, 2009). Rushton *et al.*, (2004) state that the use of questionnaires for collecting distribution data has considerable potential. For example, the distribution of six apex predators across the Namibian farmlands was successfully determined through questionnaires (Lindsey *et al.*, 2013), and in Mozambique, structured interviews of local people were used to indicate lion presence and areas of human-lion conflict (Jacobson *et al.*, 2013). Carnivores, which occur at low densities, are secretive and difficult to observe and identify in the field. The acquisition of data from questionnaires, utilising different sources such as hunters and park managers, may therefore prove beneficial (Lovell *et al.*, 1998; Nunez-Quiros, 2009). Survey work where species are rare can also be very expensive, and this has provided a strong financial incentive for analysing data derived from casual and non-systematic surveys (Rushton *et al.*, 2004).

It is critical to note that there are further limitations with questionnaire surveys, such as misidentified species, geographical bias towards populated areas, and the translation of terminology (Groves and Peterson 1992; Nunez-Quiros 2009). However, data obtained from questionnaires has been compared to data obtained through more traditional survey methods such as indirect survey signs and radio tracking and has been found to provide accurate distribution data (Blaum *et al.*, 2009; Nunez-Quiros, 2009; Thorn *et al.*, 2009). Due to multiple successful applications of questionnaires to the topics of human-wildlife conflict, species

presence and determination of attitudes, as discussed above, this study will therefore utilise this method of data capture.

A questionnaire survey was developed to gather information on the distribution of seventeen carnivore species, including leopard, and attitudes of landowners towards these carnivores across Namibia. The questionnaire was piloted on fifteen landowners at a freehold conservancy meeting prior to its release. As a result of the feedback minor changes were made to the questionnaire. The research was conducted between 1st September 2017 and 31st December 2018. The questionnaire targeted all major farming groups across Namibia that may be affected by human-wildlife conflict. In order to ensure that the questionnaire sample size and geographical distribution would be as broad as possible two distribution methods were utilised. Firstly, the questionnaire was made into an editable pdf which was then distributed via email and newsletters by multiple stakeholder groups (hunting, game ranchers, freehold conservancies, tourism, national agricultural union, farmer associations) to their members. Secondly through attendance of; farmer association meetings, freehold conservancy meeting, cattle and game auctions, and surveying customers outside of AGRA stores.

In February 2018 AGRA, the Windhoek Livestock Auctioneers (WLA) and Namboer Auctioneers agreed that attendance at their auctions and surveying at their stores would be allowed. During the same period a list of all the Namibian Farmer Associations with names and contact details was acquired from the Namibian Agriculture Union. A total of 72 associations were emailed individually to introduce the study and request attendance at their next meeting. A 30 minute overview presentation on the study was offered to each association. The same request and offer were made to all the freehold conservancy committees. Simultaneously all the farmer association's geographical locations were assessed in terms of; proximity to one another, leopard presence and logistics, they were then ranked high, medium and low priority. This ensured that when multiple invitations came in to request attendance on the same date a determination as to which meeting should be prioritised could be made. Overall 45 meeting invitations were received and 23 meetings (farmer associations, freehold conservancies, AGMs) were attended across Namibia in strategic locations, the full list can be seen in Appendix 2.

Both AGRA and WLA placed restrictions on activities during the auctions and it quickly became clear that farmers would arrive close to the start time and leave during the auction once their lots of interest were complete. This meant the time to interact with people was low and it was rapidly determined that attending auctions would be an ineffective way to get questionnaires completed. At the regional AGRA stores e.g. Otjiwarongo and Rehoboth (Appendix 2, Table 8.2.1) it was found that the number of target farmers visiting the stores were low, one every hour or two and that the decline rate by potential participants was high. However, the time spent at the AGRA Windhoek store was highly productive, in two days 44 questionnaires were completed. The study attended the NLU AgriBraai and secured a booth at the Outjo Wildsfees which is one of the largest game auctions and agricultural fairs in Namibia. The aim of attending these events was to raise awareness of the study and to get questionnaires completed.

Participants were anonymous in order to protect the confidentiality of respondents. Where necessary, to indicate level of agreement and disagreement, a five-point Likert scale was used where three was deemed neutral (Drinkwater, 1965). The majority of questions were closed-format with selected options. There were also open-ended questions that attempted to learn more about the respondents' opinions and feelings on certain topics. The questionnaire comprised of 21 questions in three areas of interest: (1) 'Respondents and their properties' regarding background information on farmers and farm characteristics; (2) 'Predators and predator control' to determine whether predator control is practised, methods used, frequency of control, and husbandry techniques utilised; (3) 'Questions relating specifically to leopard' including leopard presence, tools used to monitor leopards, trophy hunting undertaken, reasons for their attitudes, and whether they agreed or disagreed with a number of statements relating to leopard. All treatment variables were tested for normality using Kolmogorov-Smirnov tests and where appropriate parametric tests were used. When the data were found to be not normal, equivalent non-parametric tests were used.

3.3.2. Ministry of Environment and Tourism Records

MET maintains records of telephone reports made of the removal of problem leopards across freehold farms nationally. This study utilised these records from the period of 2001 to 2018.

The information collected in the telephone report includes; year, permit number, farmer name, farm name and number, farm size (ha), and species removed. Farmers reporting the removal of a problem animal are required to follow the telephone report with a motivational letter detailing the reasons why the animal was removed. For this study the information contained within the motivational letters was extracted including; age, sex, health, reason/s for removal, removal method/s, and impact of the problem animal e.g. number of calves killed.

3.3.3. Communal Conservancies

Across Namibia's communal conservancies human-wildlife conflict incidents are recorded by game guards through The Namibian Association of Community Based Natural Resource Management (CBNRM) Support Organisations (NACSO) Event Book system. An incident is defined as 'when a carnivore(s) kills or injures livestock (cattle, sheep, goat), however, it is unknown if the carnivore involved was killed in retaliation. These records will be used to calculate the number of leopard incidents recorded between 2001 and 2017 as well as their geographical spread.

3.4. Leopard Hunting in Namibia

MET holds annual records on the number of hunts undertaken and how many leopards were hunted per year. As such it is possible to determine how many of the annual 250 TAGs given out by MET were used and where the hunt occurred. This data set quantifies the number of successful hunts and can then be compared against the data gathered from the questionnaires regarding illegal off take of problem animals (reported/unreported). From 27th January 2016 a new Schedule G record sheet was introduced, this data sheet provides detailed information from individual trophy hunts, this information has also been included in this study. The new Schedule G record sheet also allows the reasons for successful and unsuccessful hunts to be determined as this information is required from every trophy hunt. All treatment variables were tested for normality using Kolmogorov-Smirnov tests and where appropriate parametric tests were used. When the data was found to be not normal, equivalent non-parametric tests were used.

3.5. Assumptions and Limitations

The camera trapping survey design was standardised to ensure data sets could be compared across the different locations. As the detection rates for leopard are known to be higher in the dry season it was determined that surveying should only be undertaken during this period. Therefore, only two camera trap surveys could take place within the timeframe of the study. The study recognised this limitation in the planning stage and sourced additional data sets from collaboration partners to ensure that the sample size and coverage of the data was adequate to answer the study's objectives. Leopard density data in the freehold farmland across Namibia is severely limited. Therefore, in order to utilise the comparable data that was available (Stein *et al.*, 2011b) the study took the approach to repeat surveys across the three areas as this was a clear way to determine if there had been any change to the leopard density between 2011 and 2018. This approach was advocated by Williams *et al.*, (2017)'s study which stated that conducting multiple surveys over several years was essential as it mitigated the problem of variation in estimates as well as enabling a determination of population trends with a high degree of confidence compared to a single point estimate. To ensure that the two data sets from the high, medium, low densities areas were comparable and standardised the study re-surveyed the named farms from Stein *et al.*, (2011b)'s study. The study recognises that the specific camera trap survey areas represent a proportion of the habitat and land use types where leopards are resident in Namibia. However, by including additional leopard density data into the national population model this ensured that a greater proportion of habitat type and land use variability was included in the analysis. The study also recognises that the national population density model analysis was compiled of leopard densities from multiple years which again is a direct reflection of the limited availability of data across Namibia.

Based upon the sample size and geographical distribution of the questionnaire respondents this study assumes that the results are representative of the wider freehold farming community. The questionnaire was designed to collect sensitive information from respondents on illegal activities. However, even when measures were put into place, such as anonymity for the respondent, the study recognises that not all respondents will, one answer, two answer honestly, when providing sensitive information on illegal activities.

In relation to the trophy hunting data it is important to note that the Schedule G record sheet is only returned to within the Ministry of Environment and Tourism once the leopard has been processed by the taxidermist and exported. The timeframes of this process can vary significantly from months to years. The study has captured all the records that were available within the Ministry of Environment and Tourism as of 20th of January 2019 and is reporting based upon this information. However, the study recognises that there are Schedule G records sheets locked within the export process and the data within them has not been captured.

4. Results

4.1. Results Summary Table

A comparison between this study's key results and Stein *et al.*, (2011b) study was undertaken, the results of which can be seen in Table 4.1.

Table 4.1. Comparison of key results between Stein *et al.*, (2011b) study and the National Leopard Study 2019.

Results	Stein <i>et al.</i> , (2011b)	National Leopard Study 2019	Differences
Namib-Naukluft/Tsauu//Khaeb National Parks Camera Trap Survey 1	1.2 leopards/100km ²	0.59 – 0.9 leopards/100km ² (Edwards <i>et al.</i> , 2015)	-51% to -25%
Auas Mountains Camera Trap Survey 2	2.0 leopards/100km ²	2.8 leopards/100km ²	+40%
Omaruru Camera Trap Survey 3	3.1 leopards/100km ²	3.6 leopards/100km ²	+16%
Problem Leopard Removals (total)	183	342	+47%
Applications (%) for Problem Leopard Removal MET Permits	50%	45%	-5%
Leopard Presence Records	Baseline	Increase	
National Population Estimate	14,154	11,733	

4.2. Characteristics of the Leopard Population

4.2.1. Density and Population Structure

The Auas Mountains survey area covered 1,226km² with Omaruru covering 1,200km², all 100 cameras were deployed across the 50 sites in a paired design. A total of 11,516 camera trap nights were surveyed between August - November 2017 and July - October 2018 across the two survey areas, Auas Mountains (5,674) and Omaruru (5,842). A total of 484 trap nights were lost over the two surveys due to faulty SD cards or broken equipment following environmental or animal interactions. The Auas Mountains survey produced 3,992,194 images with Omaruru yielding 2,966,938 creating a combined total of 6,959,132 images. Due to the sensitivity of the cameras the movement of insects and vegetation moving in the wind were also recorded and included in the total image count. Both surveys were undertaken during the main dry season to ensure the detection probability of leopard was as high as possible. As such, a proportion of the cameras per survey area were placed directly at water points which caused a substantial increase in the overall number of images captured due to the constant movement of both wildlife and cattle. Of the 6,959,132 images collected, 0.12% (3,805) were of leopard, in the Auas Mountains 0.03% (1,237) of the images were of leopard with Omaruru being slightly higher at 0.09% (2,568).

4.2.1.1. Survey Farm Information

Overall 21 freehold farms were surveyed, 11 in the Auas Mountains and 10 in Omaruru, a list of all the farms survey can be seen in Appendix 1. The average farm size in Auas Mountains was 70km² \pm 18.5km² (SD) and Omaruru was larger at 81.9km² \pm 74.7km² (SD). The landowners stated that rainfall over the past five years in the Auas Mountains was between 201mm and 500mm with Omaruru seeing lower rainfall levels at 101mm to 400mm. All the farms surveyed had multiple natural and man-made water points that were all open to both cattle and game. The farms in the Auas Mountains generated the majority of their income from cattle (69%) followed by tourism (20%), hunting (7%), sheep and goats (1%) and other (1%). The sources of income reflect the fencing type found on the 11 farms with 73% having cattle only fencing, followed by high game fencing only (18%) and a combination of cattle and high game fencing (9%). In Omaruru the farms generated their income from a combination of cattle

(39%) and hunting (38%) followed by game (11%), sheep and goats (6%), and tourism (6%). The fencing type again reflects the variation in income generation with 37.5% have cattle only fencing but 25% had both cattle and high game fencing, 25% had only high game fencing and 12.5% had a combination of cattle and predator proof electrified high fencing. The large carnivores captured during the survey in the Auas Mountains was leopard, cheetah and brown hyaena. The Omaruru survey also captured leopard, cheetah and brown hyaena in addition spotted hyaena were captured across multiple camera trap sites.

Between October 2016 and October 2017, the landowners in the Auas Mountains reported a loss of 50 livestock; 46 cattle and 4 sheep to leopard. As a consequence of this loss during the same time period 11 problem leopards were removed using cage traps and shooting. The 11 were made up of 7 males, 3 females and 1 unknown sex. Between October 2017 and October 2018, the landowners in Omaruru reported a loss of seven cattle and 35 game individuals. As a result, two male problem animals were removed by shooting during that period.

4.2.1.2. Available Biomass

The herbivore stocking density was calculated for both study areas using game count data and livestock figures provided by the landowner and then divided by the total study area (km²). The available annual live biomass for both areas was calculated based on the average biomass calculated from 3/4 mean adult female body mass (Coe *et al.*, 1976), which assumes equal sex ratio and size distribution. The total game stock density for Auas Mountains was 1797 kg/km² and Omaruru 1714 kg/km² as a combination of the species in Figure 3.5. The total livestock density for Auas Mountains was 884 kg/km² and Omaruru 240 kg/km² as a combination of the species listed in Figure 4.1.

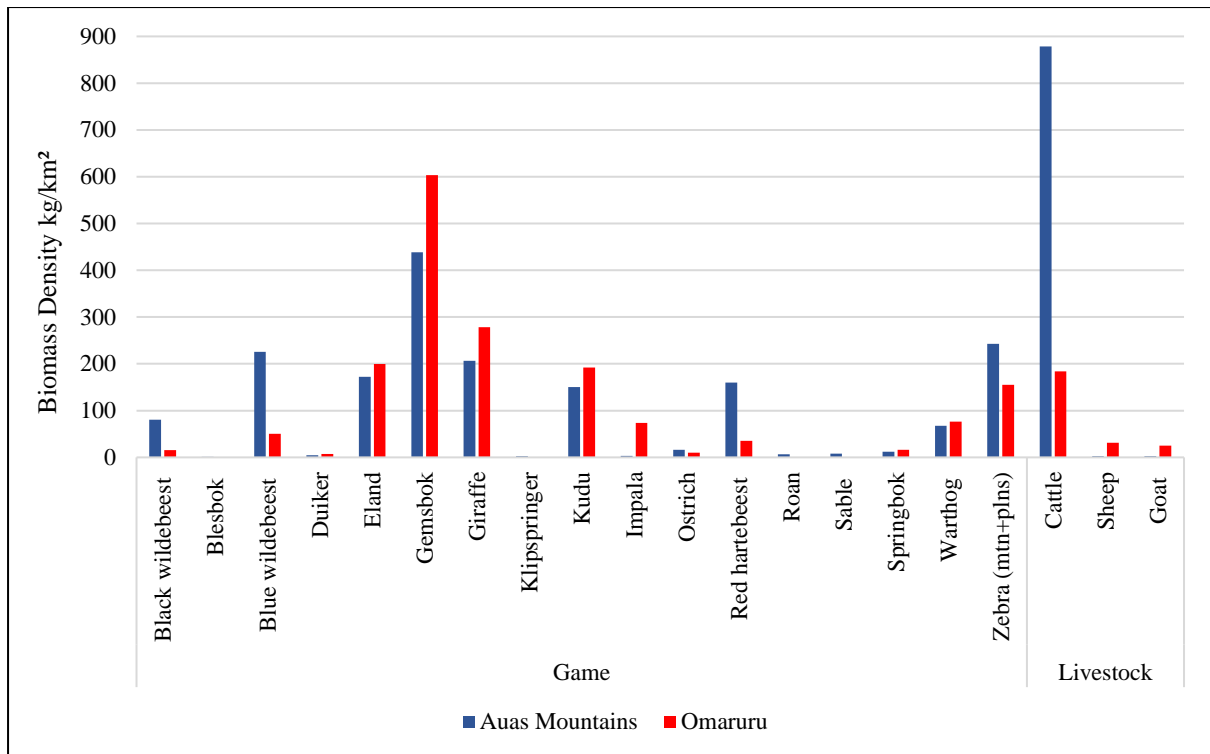


Figure 4.1. Compares the variation in live biomass density (game species and livestock) in 2017/18 between the Auas Mountains and Omaruru survey areas.

4.2.1.3. Namib-Naukluft/Tsauu//Khaeb National Parks Camera Trap Study

Edwards *et al.*, (2015) study (29th May - 28th July 2013) captured 28 usable leopard images from 51 camera sites split between the north ($n = 21$) and south ($n = 30$) study area (Figure 4.2) across 60 nights. Eight individual leopards were identified from the images and that information was used to calculate a density estimate for the two areas (Table 4.2). A detailed description of the methodology and analysis can be found in Edwards *et al.*, (2015).

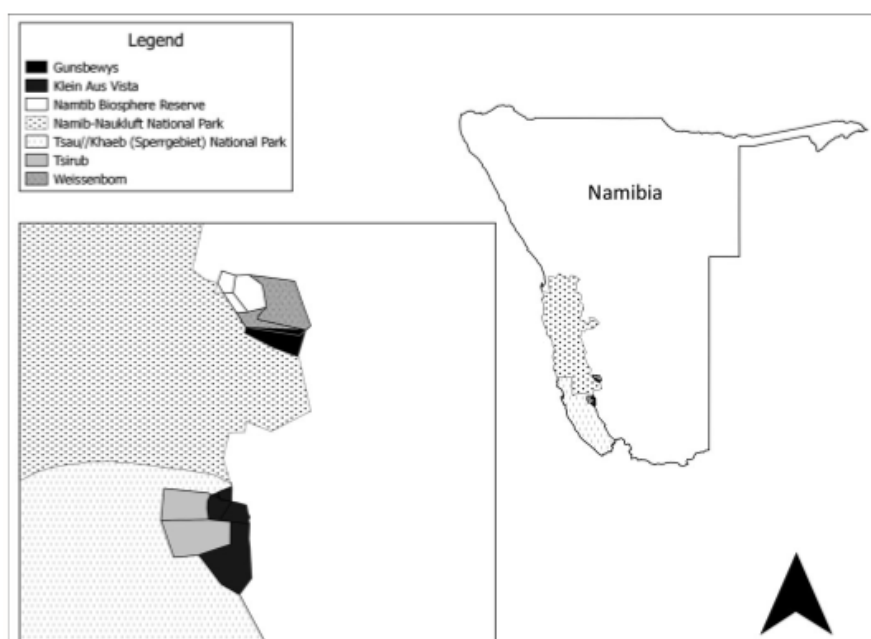


Figure 4.2. The location of the Namib-Naukluft/Tsauu//Khaeb National Parks camera trap study camera trap survey area undertaken by Edwards *et al.*, (2015).

The density estimates derived from Edwards *et al.*, (2015)’s study was lower than Stein *et al.*, (2011b) finding of 1.2 leopards/100km² which shows that there has been an average lowering of the leopard density by 38% in that area.

Table 4.2. The leopard density results from the southern camera trap survey area undertaken by Edwards *et al.*, (2015).

Site	Total area km ²	Number of leopards captured	Density/100km ² (95% CI)
North	428.92	3	0.9 (4 - 11)
South	852.01	5	0.59 (5 - 5)

4.2.1.4. Central and North Camera Trap Study

Overall 131 individuals were identified across the two study areas, a total of 48 were identified in the Auas Mountains and 83 in Omaruru. Once the individuals had all been identified their age and sex were then determined using indicators such as male genitalia, skull size, body size and shape, poor quality images were labelled as ‘unknown’. Of the 49 individuals in the Auas Mountains 28 were adults of which 15 were male and 14 female, 11 subadults (sex unknown), 2 cubs (sex unknown) and 20 were classified as unknown. In Omaruru the 83 individuals were classified as 67 adults of which 35 were males and 32 females, 8 subadults (sex unknown), 4

cubs (sex unknown) and 16 were labelled as unknown. In both survey areas the sex ratio was found to be approximately 1:1 (adult males:adults females).

The leopard density in the Auas Mountains determined by Stein *et al.*, (2011b) study was 2.0 leopards/100km². The density estimate derived by this study in the Auas Mountains was 2.8 leopards/100km² (Table 4.3), which shows that there has been a 40% increase in the leopard density for that area between 2011 and 2017. Stein *et al.*, (2011b)'s study determined that the density in the Omaruru area was 3.1 leopards/100km². The density estimate found by this study in the Omaruru area was 3.6 leopards/100km² (Table 4.3), which shows a 16% increase in the leopard density for that area between 2011 and 2018.

Table 4.3. The leopard density results from the Auas Mountains and Omaruru survey area.

Location	Number of leopards (95% CI)	Density/100km ² (95% CI)
Auas Mountains	143 (101-188)	2.8 (1.97 - 3.68)
Omaruru	163 (134-190)	3.6 (3.03 - 4.25)

4.2.2. Habitat Suitability

The output result of the MaxEnt habitat suitability analysis can be seen in Figure 4.3. Based upon the jack knife tests the environmental variable with highest gain when used in isolation was altitude, which therefore appears to have the most useful information by itself. Altitude was then followed in importance by temperature seasonality. The environmental variable that decreases the gain the most when it was omitted was land cover, which therefore appears to have the most information that isn't present in the other variables. Therefore, removing layers such as land cover, altitude and temperature seasonality did seriously reduce the accuracy of the model. The percentage contribution reflected the jack knife results with the addition of land ownership (freehold farm, communal conservancy, national park) Appendix 4 Table 8.3.

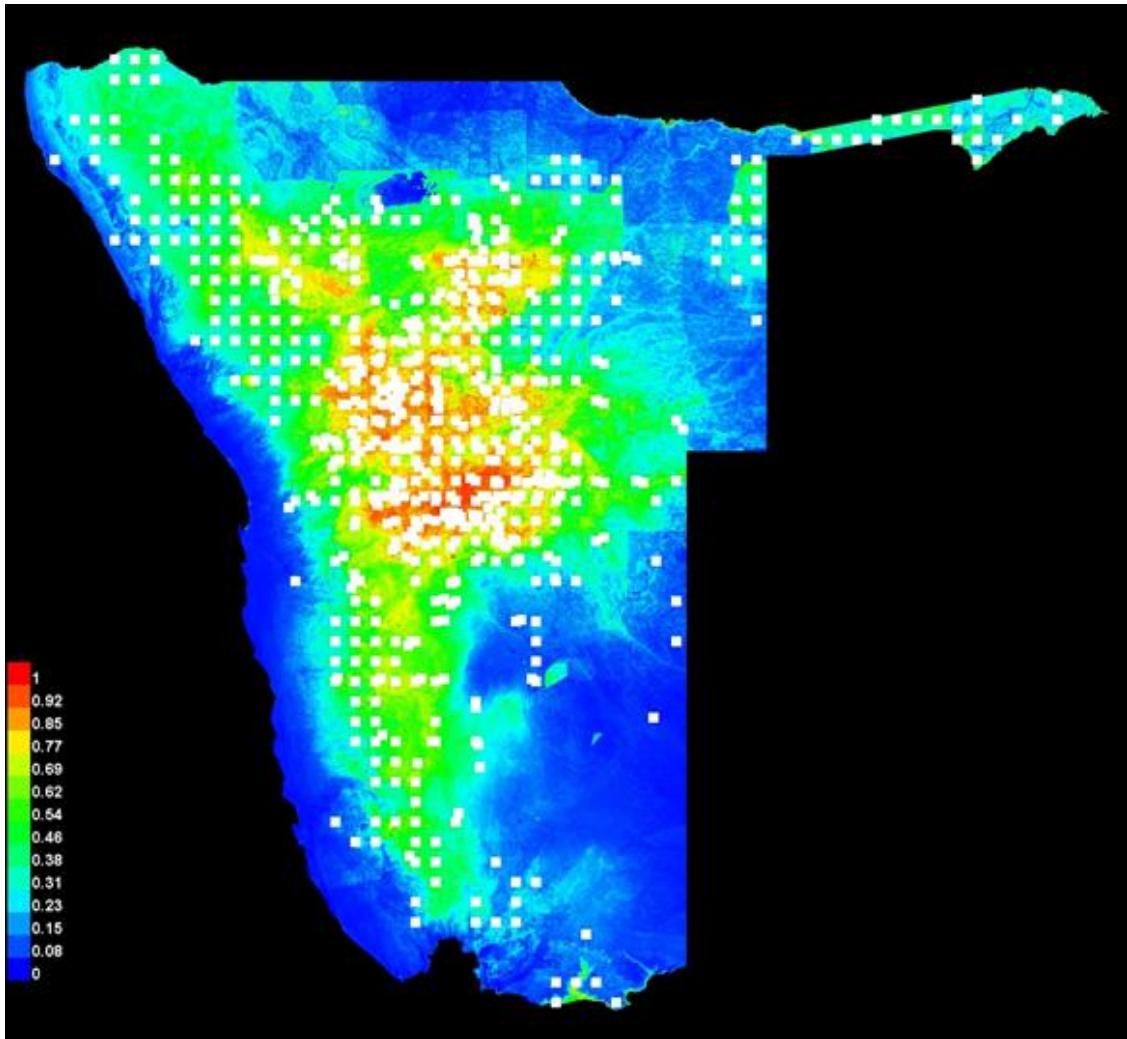


Figure 4.3. A representation of the MaxEnt habitat suitability model for leopard (Source: Dr. Vera De Cauwer).

4.2.3. National Population

The questionnaire respondents utilised three methods to determine if leopard was present on their properties. Of the three methods spoor tracks was the most utilised (43%) followed by direct sightings (33%) and lastly remote camera traps (24%) (Figure 4.4).

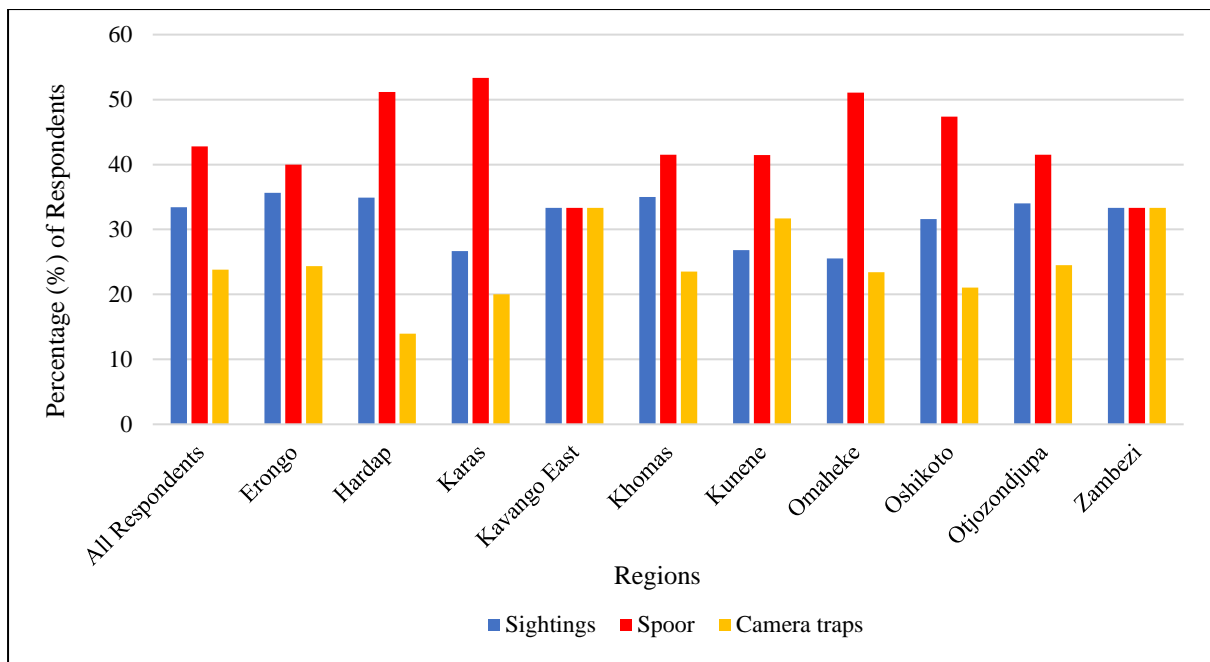


Figure 4.4. Questionnaire respondents' methods to determine leopard presence on their properties.

As previously discussed, 83% stated that leopard had been present on their property between October 2016 and December 2018, 18% stated that leopard was absent on their property (Figure 4.5).

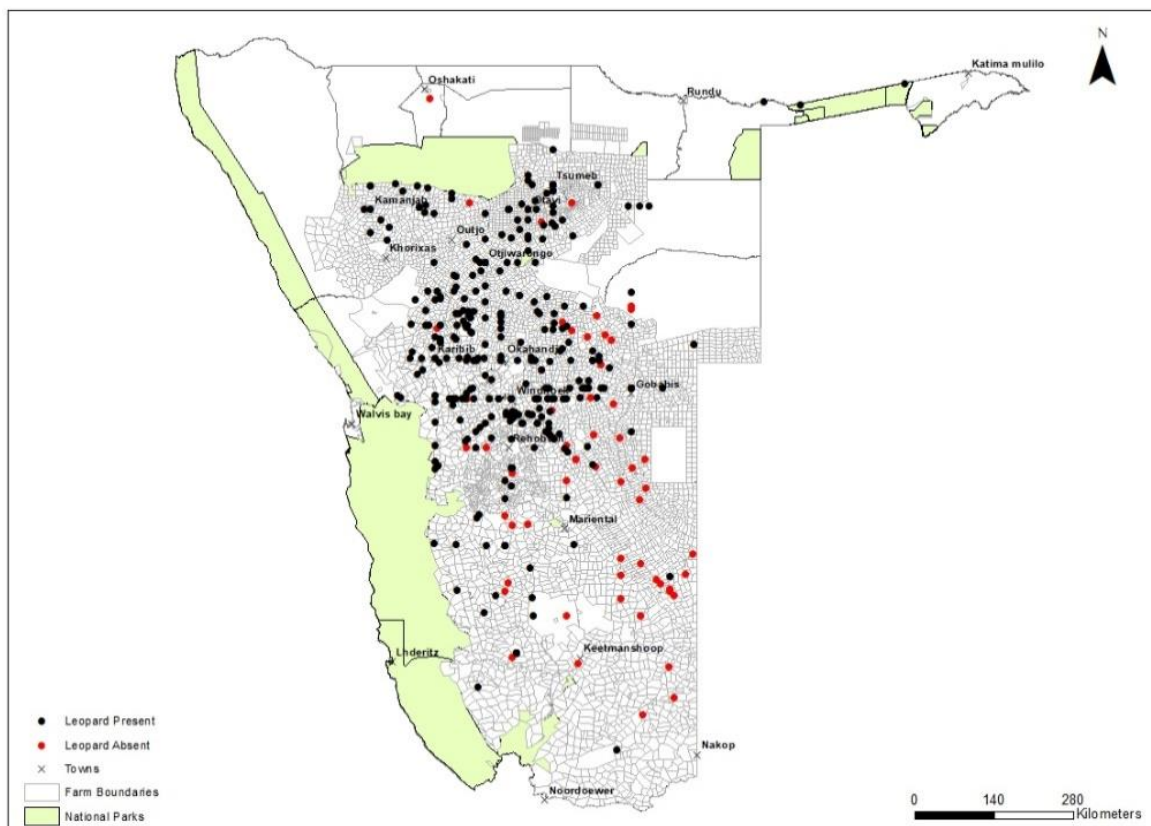


Figure 4.5. Shows the questionnaire respondents with and without leopard present on their property between October 2016 and December 2018 across Namibia.

4.2.3.1. Leopard Presence

The leopard presence map (Figure 4.6) data is derived from 6,529 EIS records from 49 contributors including this study, the full list of contributors can be seen in Appendix 2 Table 8.2.2. The map only contains information from presence records only, it outlines the collective known presence of leopard not the distribution of leopard across Namibia (Figure 4.6). The green background of the presence map shows the current IUCN Red List distribution for leopard and the cream areas show where leopard is labelled as extinct in Namibia (Stein *et al.*, 2016) (Figure 4.6). Therefore, the presence grids that are situated outside of the green boundary are outside the current recognised IUCN distribution and inside the area labelled as extinct for leopard in Namibia (Figure 4.6). This collaborative data set is therefore critically important as it can be used to update leopard distribution for Namibia both nationally and internationally.

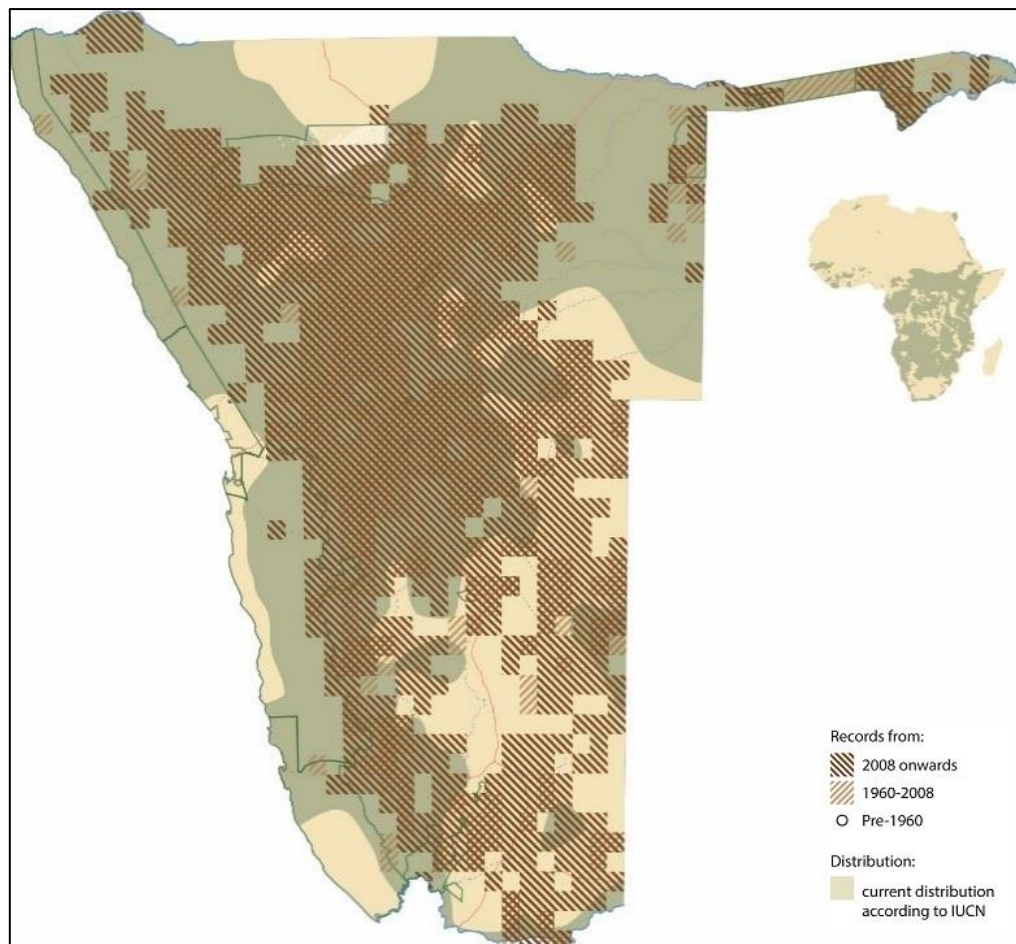


Figure 4.6. Presence of leopard across Namibia based on all known records (data source: EIS 2019).

The aim of illustrating the known presence data for leopard is to highlight the differences between the 2011b Stein *et al.*, leopard distribution map and known leopard presence map as

of March 2019 (Figure 4.7). In 2011 the south-east of Namibia was predominantly labelled as ‘No Known Occurrence’ (white areas) (Figure 4.7). As the purple circles shows a significant effort was made to collect presence data in the east, south-east and south of Namibia which has led to substantial proportions of the ‘No Known Occurrence’ areas now being defined as leopard present (Figure 4.7). The area surrounding Gobabis and to the north in 2011 was also labelled as ‘No Known Occurrence’, presence data for this area has also been increased (Figure 4.7).

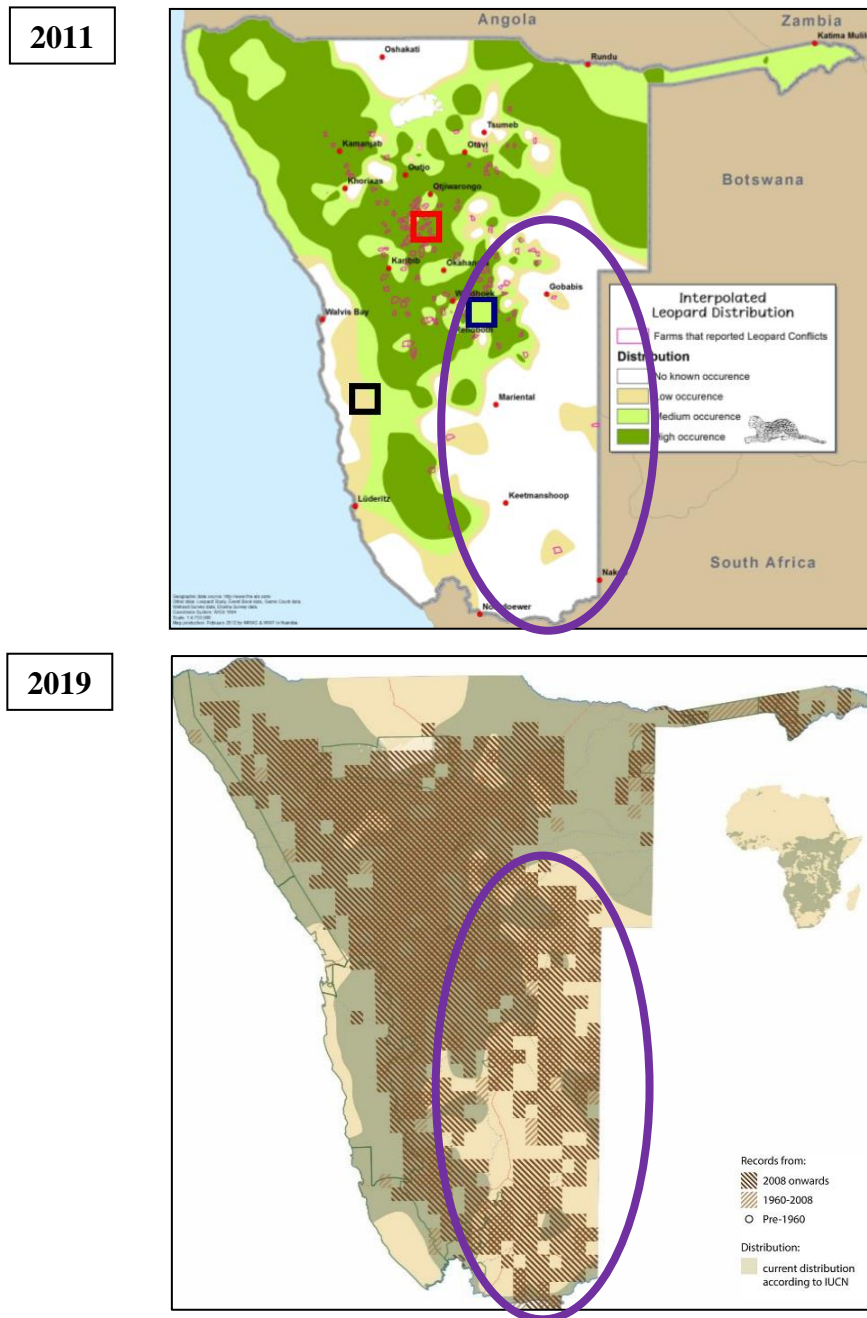


Figure 4.7. A comparison of all known leopard presence records between 2011 (Stein *et al.*, 2011b) and the collective information provided by the EIS (2019).

4.2.3.2. National Population Size

The national leopard population for Namibia was determined from the SPACECAP derived density figures. As such Namibia was broken into 207,707 individual 2km² pixels each with an associated leopard density linked to its environmental variables. The density data was then split into intervals of 0.5 leopards/100km², which created seven density intervals (Table 4.4). Within each category the number of leopards was summed along with the root mean square errors (Table 4.4). The leopard population was determined to be 11,733 with a root mean square error of 5,949 (Table 4.4). The total area covered by each category was also calculated and the percentage that area represents in relation to the total area of Namibia (Table 4.4). The approach taken to determining the national population figure provides significantly greater detail than the approach taken by Stein *et al.*, (2011b). As such when looking to apply subjective demarcations such as high, medium, low density categories assumptions would have to be made as to what values these would include. Instead by creating density intervals the detail of data is not lost from the process.

Table 4.4. A breakdown of the national leopard population by seven density intervals, number of leopards per interval, root mean square error, total area coverage (km²) and percentage of area coverage.

Density intervals (leopards per 100km²)	Number of leopards	Root mean square error	Area (km²)	Percentage of area (%)
0-0.5	253	730.1	101,352	12.2
0.5-1	1347	1233.4	172,400	20.8
1-1.5	1770	1026.5	142,482	17.2
1.5-2	3955	1623.6	227,253	27.4
2-2.5	2908	947.2	133,144	16.0
2.5-3	1413	368.5	51,981	6.2
3+	87	19.7	2,783	0.2
Total	11,733	5,949	831,395	100

The distribution of the leopard population (density) across Namibia can be seen in Figure 4.8. The darker green represents the lowest density areas, followed by yellow and then into red (Figure 4.8). The leopard density map clearly shows that five out of the thirteen regions of Namibia are where the majority of the leopard population resides (Figure 4.8). This result reflects the influence of the environmental variables that were firstly identified in the habitat suitability model and therefore subsequently used within the Random Forest model to create the density estimates. The highest leopard density in the Kunene region was in the Kaokoveld up on the Kamanjab plateau and the escarpment that runs up to the Angolan border. The regions

of Omusati, Ohangwena, Oshana, Oshikoto, Kavango, and Zambezi were all predominantly in the lower half of leopard density intervals. In the Erongo region the highest leopard density was found in the north east of the region around Omaruru, the Erongo Mountains and Mount Etjo. A significant proportion of the Khomas region had a higher leopard density in comparison to all the other regions due to the Khomas Hochland Plateau, Auas Mountains and the rugged landscape out to the Gamsberg Nature Reserve. The Otjozondjupa region has two distinct density areas. The higher density areas covered the freehold farms of the region whereas the eastern communal conservancies, N#u-a-Jaqua conservancy, Nyae Nyae Conservancy, and Ondjoi Conservancy on the Botswana border have a lower density than the central and western area of the region. In Omaheke, again, the highest density of leopard was found across the freehold farms in the centre and south of the region with the lowest density areas found in the communal conservancies. The majority of both Karas and Hardap were categorised in the lower density intervals (0-1.5 leopards/100km²) (Figure 4.8). The only variations outside of these densities in Hardap was 1) on the border with Khomas due to the Rehoboth Plateau and the Namib-Naukluft Mountain Zebra National Park and 2) the Weissrand Plateau on the border with Karas. In Karas, the Karasberg Mountains were identified as suitable areas for leopard in both the habitat suitability model and density model for the region. In the far south the orange river and its mountain areas was identified as suitable habitat for example the Sandfontein Nature and Game Reserve and Tantalite Valley (Figure 4.3). This area was broken into two different density intervals, 0.5-1 and 1-1.5 leopards/100km² to reflect the differences in habitat type. The model identified significantly sized dry river beds as areas of increased density as they are important habitat for leopard (Mills and Hes, 1997). In Otjozondjupa and Kavango the Omatako dry river can clearly be seen. Similarly, in Omaheke the Daneib, Otjozondjoi, Eiseb, Epukiro, and Rietfontein can all be individually identified. Further south the Nossob and Twee rivers are identifiable in the Hardap region.

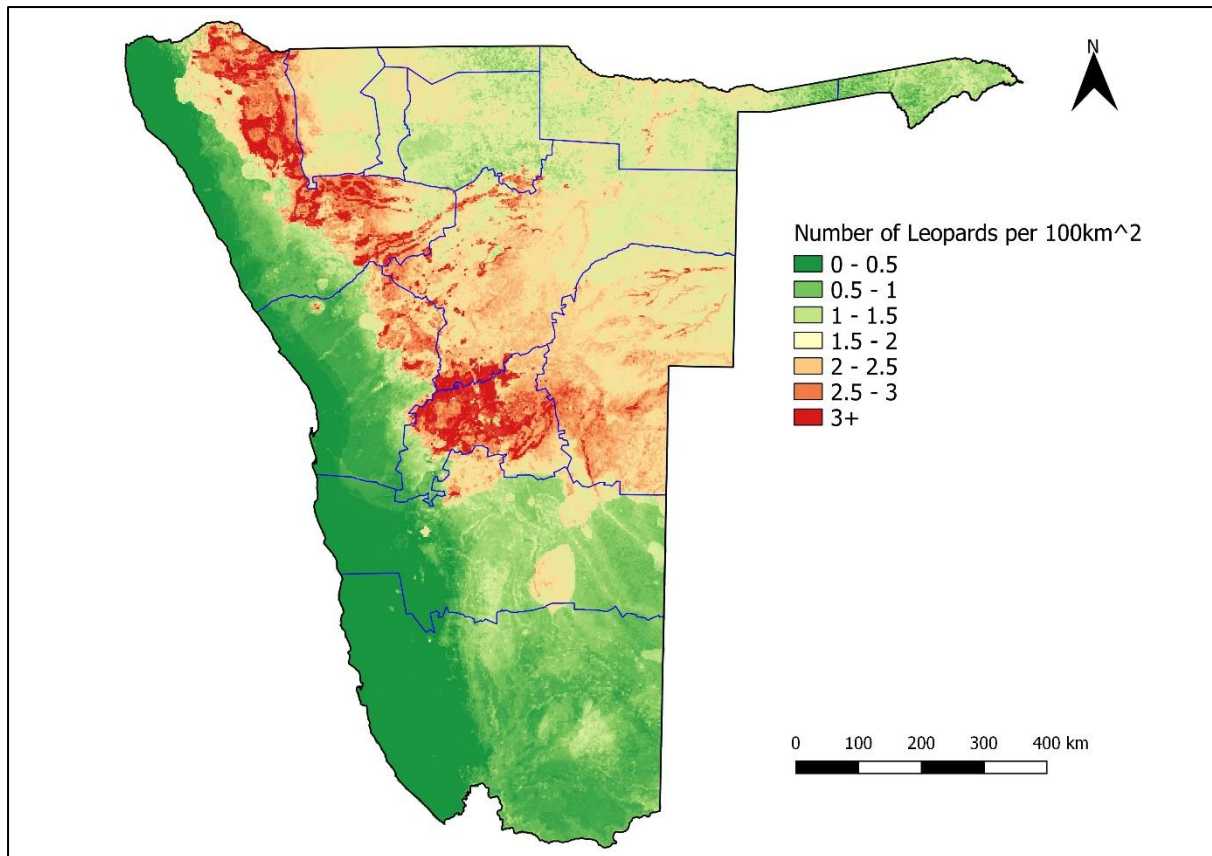


Figure 4.8. The variation in leopard density using seven density intervals across Namibia based upon the distribution model.

The seven density intervals were then broken down into discrete polygons to aid in the visualisation of where these intervals were located geographically (Figure 4.9). However, it is important to note that the fluidity and detail of the interactions between the density intervals shown in Figure 4.8 is in places lost in Figure 4.9.

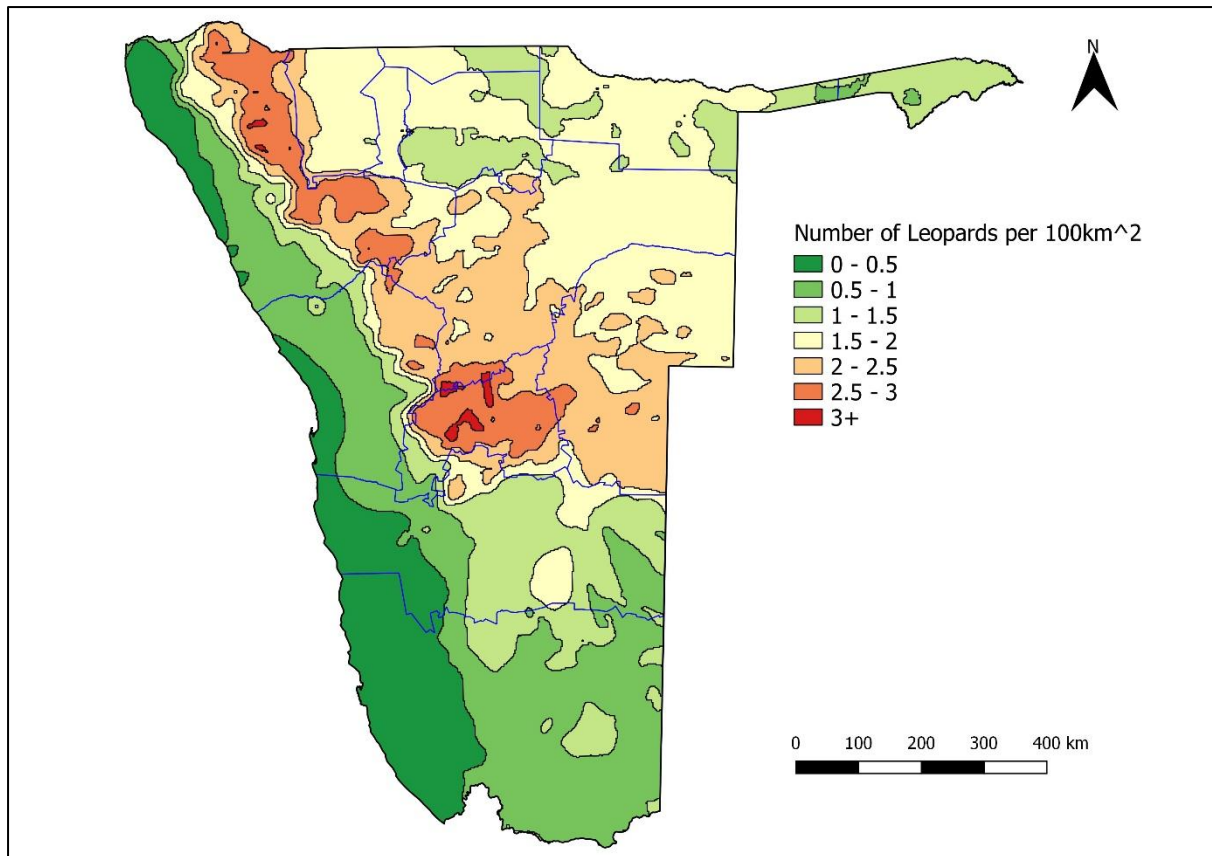


Figure 4.9. The variation in leopard density using seven density intervals derived into discrete contours across Namibia.

4.3. Human-Leopard Conflict

4.3.1. National Questionnaire

A total of 392 questionnaires were returned, the geographical distribution of the respondents can be seen in Figure 4.10.

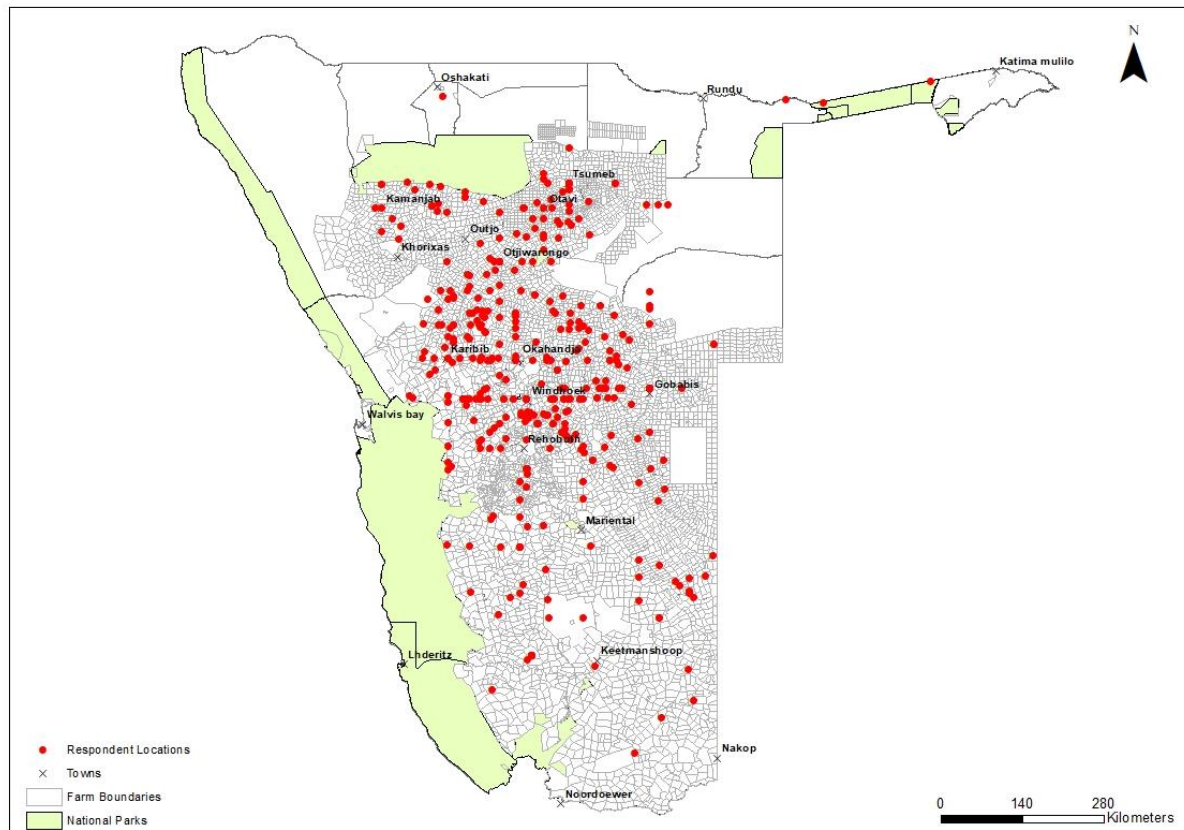


Figure 4.10. The geographical distribution of questionnaire respondents ($n = 392$) across Namibia.

A significant proportion of respondents were male (90%) compared to females (7%) and 3% did not answer. The age groups of respondents ranged from under 20 to over 60 with the age group 51 to 60 (29%) being the largest group, followed by over 60 (24%), 41 to 50 (20%), 31 to 40 (18%), 21 to 30 (4%), under 20 (1%), and unknown (5%). The dominant position held by the respondents was the owner of the property (79%) followed by the manager (9%) lease holder (8%) and other (3%), which consisted of sons of landowner, hunting guides, communal and re-settled farmers. The majority of the respondents came from the Otjozondjupa, and Khomas regions which reflects the individuals' level of engagement rather than any farming biases. The average farm size was $103.93 \pm 241.68\text{km}^2$ (SD) with a minimum of 0.2km^2 and a

maximum of 3,800km². The perimeter fencing type varied from cattle fencing (47.6%) and high game proof fencing (24.8%) to short game proof fencing (13.6%), predator proof fencing (11.8%), no fencing (2%) and other (0.2%). The predator proof fencing was made up of buried fencing (52%) and electrified fencing (48%). Cattle farming provided the highest average percentage of income, followed by sheep and goats, hunting, tourism, agriculture, other, game and mining (Table 4.5). Other income sources included; charcoal, horses, poultry, and businesses (hay, taxidermy, butchery, hunting school).

Table 4.5. Outlines the average percentage income derived per sector by respondents across ten regions in Namibia.

	Sources of income							
Regions	Tourism	Mining	Cattle	Sheep/Goat	Game	Hunting	Agriculture	Other
Erongo	33.5		63.2	36.3	30.8	52.9	10	38.3
Hardap	33.5		34.4	60	15	30.2	25.0	40
Karas	15		33.4	78.7	13.3	8.3	41.5	20
Kavango East						100		
Khomas	40.7	10	73.1	24.3	25.5	32.4		8.1
Kunene	25.7		62.5	24.3	12.5	70.7	5	35
Omaheke	15		76.7	29.3	22.0	29.2	25	23.3
Oshikoto	36.7		72.3	28.4	31.7		35	17.5
Otjozondjupa	30.1	12	69.1	16.8	28.4	34.3	27.7	29.7
Zambezi						100		
Average Percentage (%)	33.1	11.3	65.1	43.1	25.8	37.8	28.4	26.2

Overall the respondents had a greater game density (10.2 individuals per 100km²) in comparison to livestock (5.6 individuals per 100km²) (Figure 4.11). Two regions, Hardap and Karas had substantially higher sheep and goat density than cattle. Khomas, Kunene, Erongo, Zambezi, Kavango East and Otjozondjupa all showed high game densities (Figure 4.11). Only Omaheke and Oshikoto showed an equal spread between the game and cattle density across their properties.

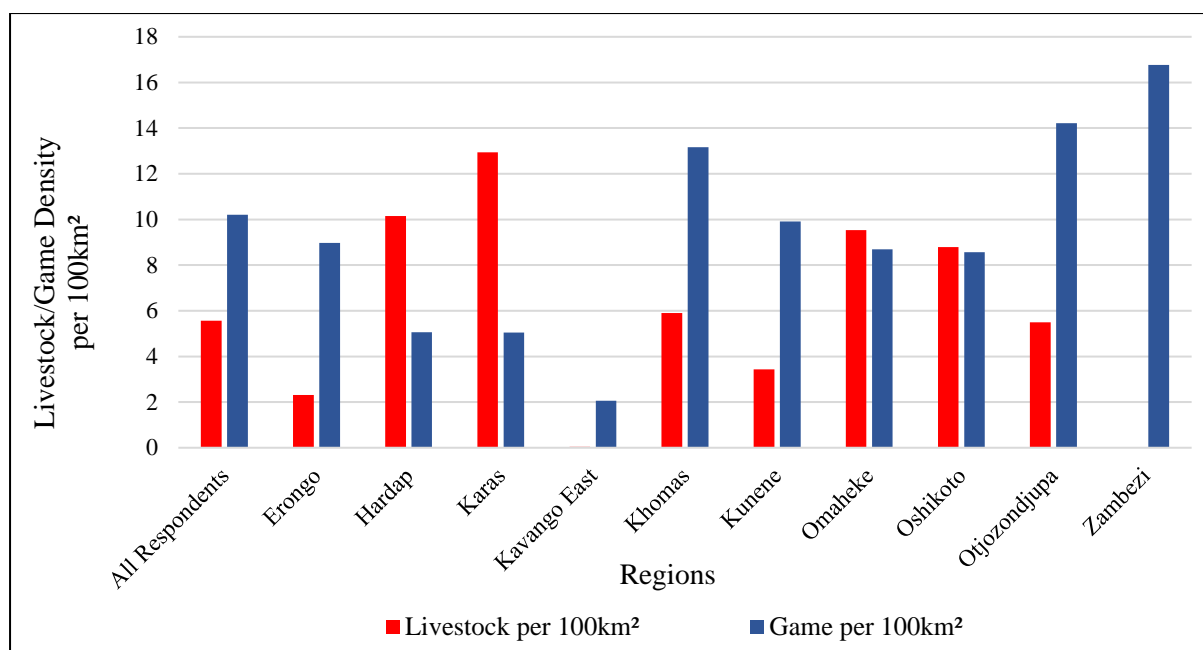


Figure 4.11. Overall livestock and game density (individuals per 100km²) on respondent's properties across the regions.

Of the 392 respondents 82.4% stated that leopard had been present on their property between October 2016 and December 2018, 17.6% stated that leopard was absent on their property. Respondents from ten regions stated that leopard presence on their property was substantially greater than leopard absence (Table 4.6). However, the respondents in the Hardap and Karas regions stated that leopard absence was greater than leopard presence (Table 4.6).

Table 4.6. Variation in leopard presence and absence across ten regions by respondents.

Region	Leopard	
	Present (%)	Absent (%)
Erongo	100	0
Hardap	46	54
Karas	42	58
Kavango East	100	0
Khomas	90	10
Kunene	95	5
Omaheke	61	39
Oshikoto	77	23
Otjozondjupa	98	2
Zambezi	100	0

When asked if the respondents felt that leopard numbers had changed over the past five years 64% stated that the leopard population had increased, with 22% stating that the population had

remained constant, 6% believed the population had declined and 8% were unsure (Figure 4.12). Only in Kunene did respondents feel that leopard numbers had remained constant overall rather than increasing (Figure 4.12).

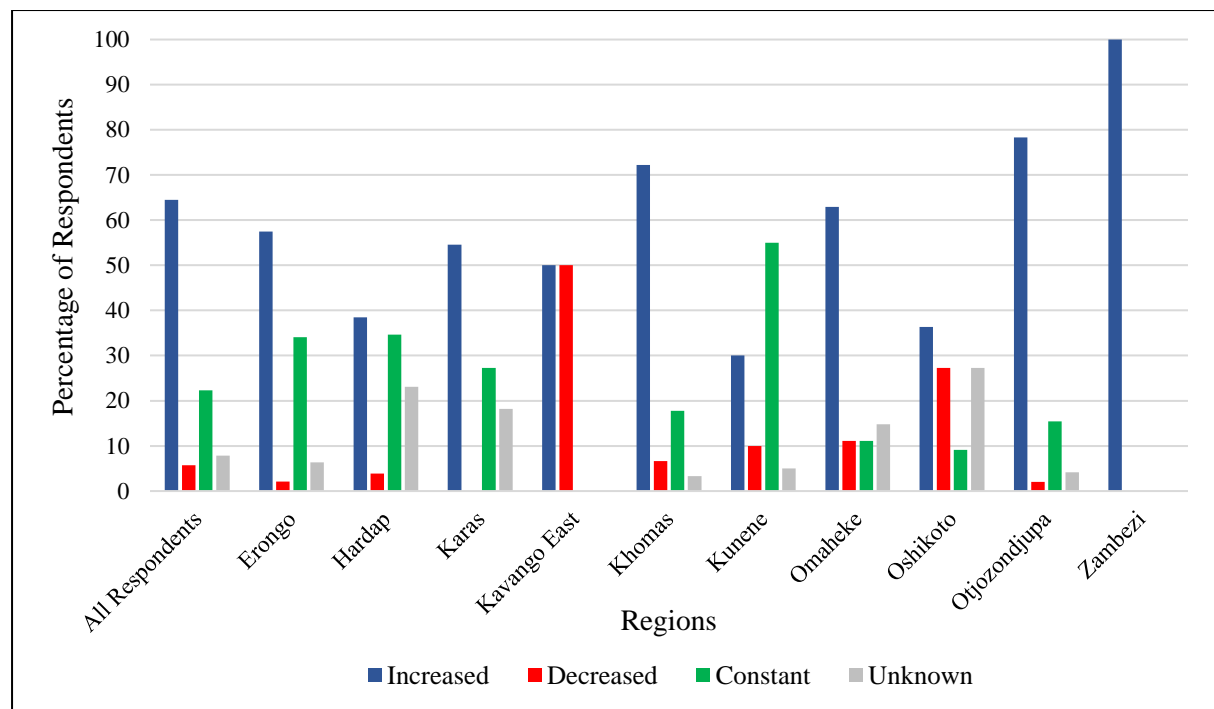


Figure 4.12. The views of respondents, by region, as to whether they felt that the leopard population had increased, decreased, remained constant or unknown on their property in the last five years.

The questionnaire measured respondents' attitude to having leopard present on their property. The median attitudinal score of all the respondents to leopard was 2.5, 3 being neutral and 2 being unfavourable. However, there was significant variation in the attitudinal score across the regions (Kruskal Wallis test, $W = 59.51$, d.f. = 9, $P < 0.001$). Only one region, Kavango East, had a positive score towards leopard while Erongo, Kunene, Otjozondjupa and Zambezi showed a neutral attitude (Figure 4.13). The remaining five regions all had an unfavourable attitude towards leopard with the Karas region having the lowest (Figure 4.13).

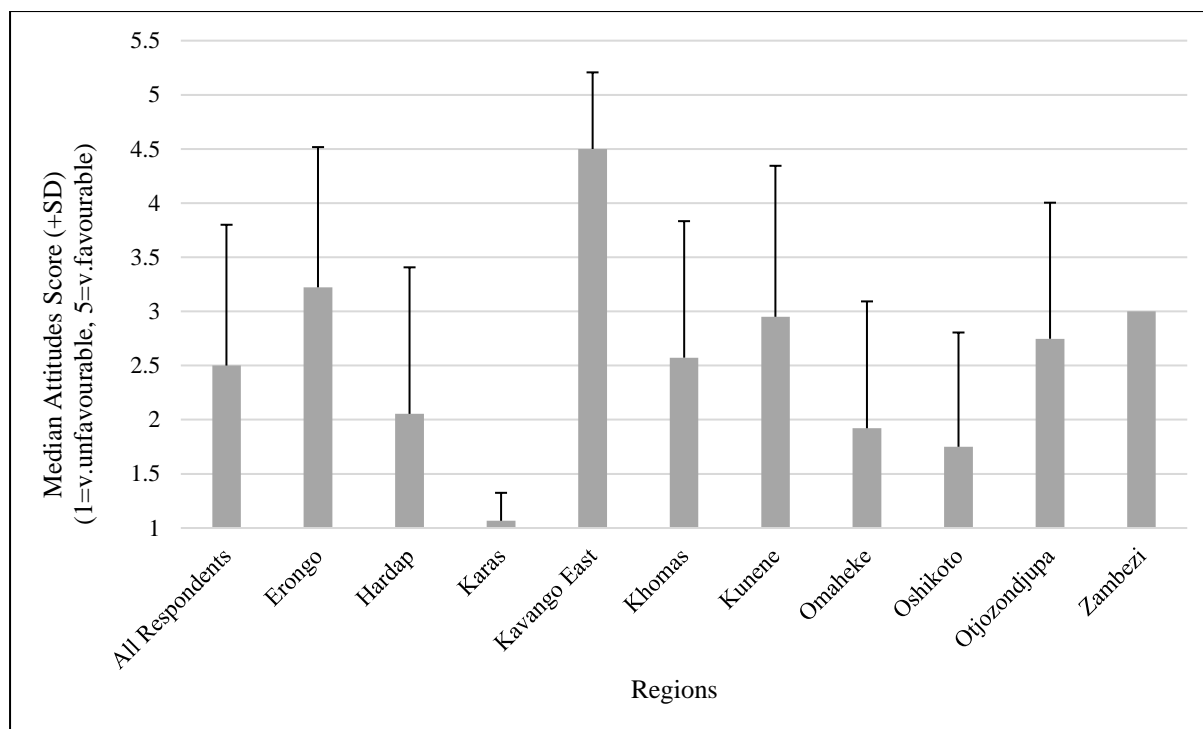


Figure 4.13. The median attitudinal scores of respondents to leopard (1=v.unfavourable, 5=v.favourable) (+SD) by region.

Between October 2016 and December 2018 respondents reported a total loss of 3,977 (livestock and game) due to leopard predation (Table 4.7). The loss of cattle was the highest during this period, with the Khomas region showing the greatest regional cattle loss (Table 4.7). The second greatest loss was of game, particularly in the Otjozondjupa region (Table 4.7). The Karas region reported the largest combined sheep and goat losses (Table 4.7). The Zambezi region respondents did not have livestock, only game. However, in Karas the loss per respondent was substantially higher than any of the other regions (Table 4.7).

Table 4.7. Outlines the total number of cattle, sheep, goats, horses, and game that respondents have lost and the loss per respondent due to leopard predation between October 2016 and December 2018.

	Number of individuals lost						
Regions	Cattle	Sheep	Goats	Horses	Game	Total	Loss per respondent
Erongo	165	6	49		86	306	10
Hardap	87	14	14	2	356	473	39
Karas	30	251	94		20	395	99
Kavango East						0	0
Khomas	1242	22	12	1	109	1386	18
Kunene	43	1	11		1	56	5
Omaheke	119	7	9		33	168	10
Oshikoto	20				15	35	8
Otjozondjupa	588	6	33		531	1158	14
Zambezi	N/A	N/A	N/A	N/A	0	0	0
Total	2294	307	222	3	1151	3977	17

The Khomas region had the highest levels of livestock loss to leopard predation as a proportion of the total number of livestock (cattle/sheep/goats) recorded (Table 4.8). Of the regions that had recorded losses Omaheke and Oshikoto had the lowest lost as a proportion of the total livestock numbers (Table 4.8).

Table 4.8. Outlines the total number of livestock (cattle/sheep/goats) owned by respondents per region, the number of livestock lost to leopard predation and the proportion of the total lost to predation.

Regions	Total livestock number	Number of livestock lost to predation	Percentage (%) of total loss to predation
Khomas	41922	1386	3.3
Erongo	9987	306	3.01
Otjozondjupa	52047	1158	2.2
Karas	26288	395	1.5
Kunene	5105	56	1.1
Hardap	46606	473	1
Omaheke	32667	168	0.5
Oshikoto	6300	35	0.5
Kavango East	160	0	0
Zambezi	0	0	0

When asked if respondents undertook predator control for any carnivore species (wild dog, cheetah, black-backed jackal, lion, leopard, brown hyaena, spotted hyaena, caracal) 78% stated they did, compared to 22% who did not (Table 4.9). However, of the 22% who did not

undertake predator control 65% had suffered losses (cattle/game) through predation (Table 4.9). In contrast 4% of the 78% who did undertake predator control did not have any actual loss. In these cases, the carnivore was removed due to a perceived threat to the respondent's livelihood (Table 4.9). When leopard was listed as a problem animal 100% of respondents undertook control methods. Of that 100%, only 3% were responding to a perceived threat (Table 4.9) rather than actual loss.

Table 4.9. Outlines whether the respondents undertook predator control methods for all listed carnivore species and specifically for leopard.

All carnivores (n = 350)	Undertake predator control	Predator control with/without loss
Yes	78% (n = 272)	4% no losses (n = 11)
No	22% (n = 78)	65% losses (n = 51)
Leopard only (n = 157)		
Yes	100%	2% no losses (n = 3)
No	0	

A total of 5,646 individual carnivores (nine carnivore species) were removed from respondent's properties between October 2016 and December 2018 (Table 4.6). Black-backed Jackal (*Canis mesomelas*) made up 80.6% of the total number of individuals removed, followed by leopard at 6.1%, cheetah (*Acinonyx jubatus*) 4.2%, caracal (*Caracal caracal*) 4.1%, brown hyaena (*Parahyaena brunnea*) 3.8%, spotted hyaena (*Crocuta crocuta*) 0.9%, lion (*Panthera leo*) 0.3%, wild dog (*Lycaon pictus*) 0.1% and honey badger (*Mellivora capensis*) 0.04% (Table 4.10). The proportion of male to female removals was 30.6% to 25.6% however, just under half were listed as unknown sex (43.7%) (Table 4.10).

Table 4.10. The total number of carnivores removed from respondent's properties between October 2016 and December 2018.

Species	Male	Female	Unknown sex	Total	Proportion of total removed (%)
Wild dog			4	4	0.1
Cheetah	118	89	28	235	4.2
Lion	9	6		15	0.3
Black-backed Jackal	1286	1082	2183	4551	80.6
Leopard	218	97	27	342	6.1
Brown hyaena	78	34	100	212	3.8
Spotted hyaena	19	5	29	53	0.9
Caracal		133	99	232	4.1
Honey badger	2			2	0.04
Total	1730	1446	2470	5646	
<i>Proportion of total removed (%)</i>	30.6	25.6	43.7		

Between October 2016 and December 2018, the number of leopards removed from respondents' properties ($n = 157$) was 342 of which 64% were male ($n = 218$), 28% female ($n = 97$) and 8% were listed as unknown sex ($n = 27$). The Karas region had the highest average problem animal (leopard) removal rate (5 per respondent) while Kunene had the lowest (1.25 per respondent) (Figure 4.14).

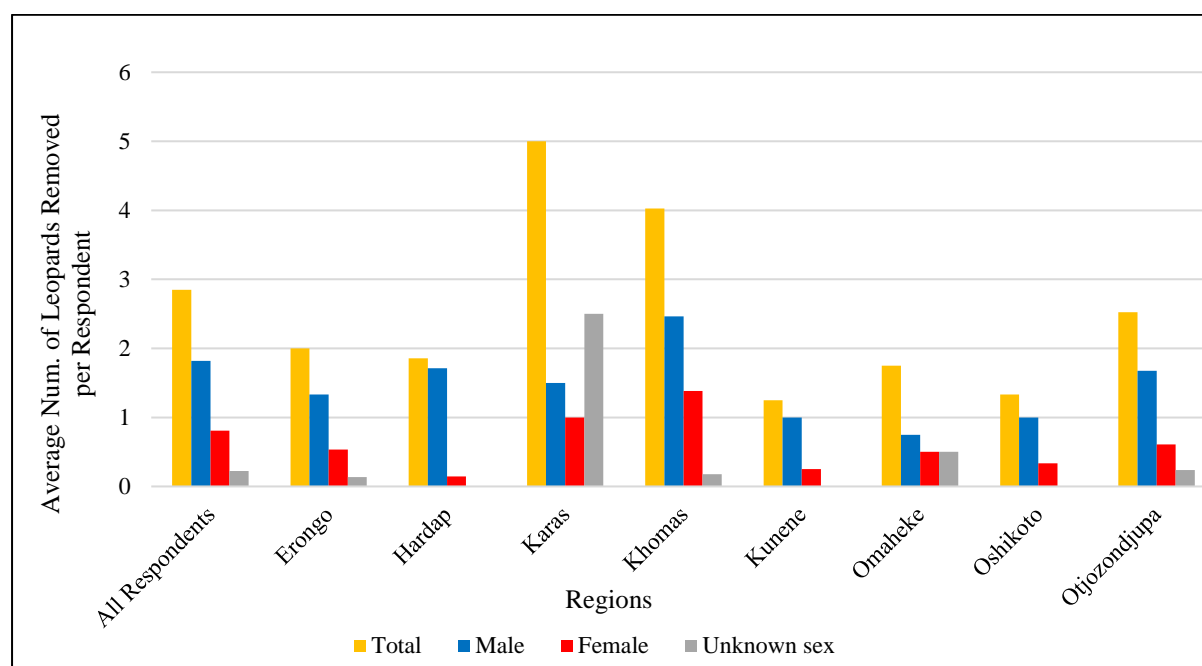


Figure 4.14. The average number of leopards (male, female, unknown sex) removed per respondent between October 2016 and December 2018 across all respondents and the ten regions.

The majority of respondents utilised shooting and cage traps (82%) as their primary methods of removing leopard from their property (Table 4.11). Respondents also stated that they utilised the opportunity to trophy hunt (12%) a leopard in response to loss of livestock and/or game (Table 4.11). A very low number of respondents used hunting with dogs, gin traps and poison as a removal method (Table 4.11).

Table 4.11. Summarises the removal methods and the number of times they were utilised on problem animals (leopard) between October 2016 and December 2018.

Removal method (<i>n</i> = 133)	Number of times utilised
Shot	63
Cage trap	57
Trophy hunt	17
Hunting	6
Dogs	1
Gin trap	1
Poison	1

Of the respondents who stated the number of problem animal (leopard) removed 50% did not apply to MET for a problem animal permit (Figure 4.15), 45% did apply for a permit and 5% did not answer the question (Figure 4.15).

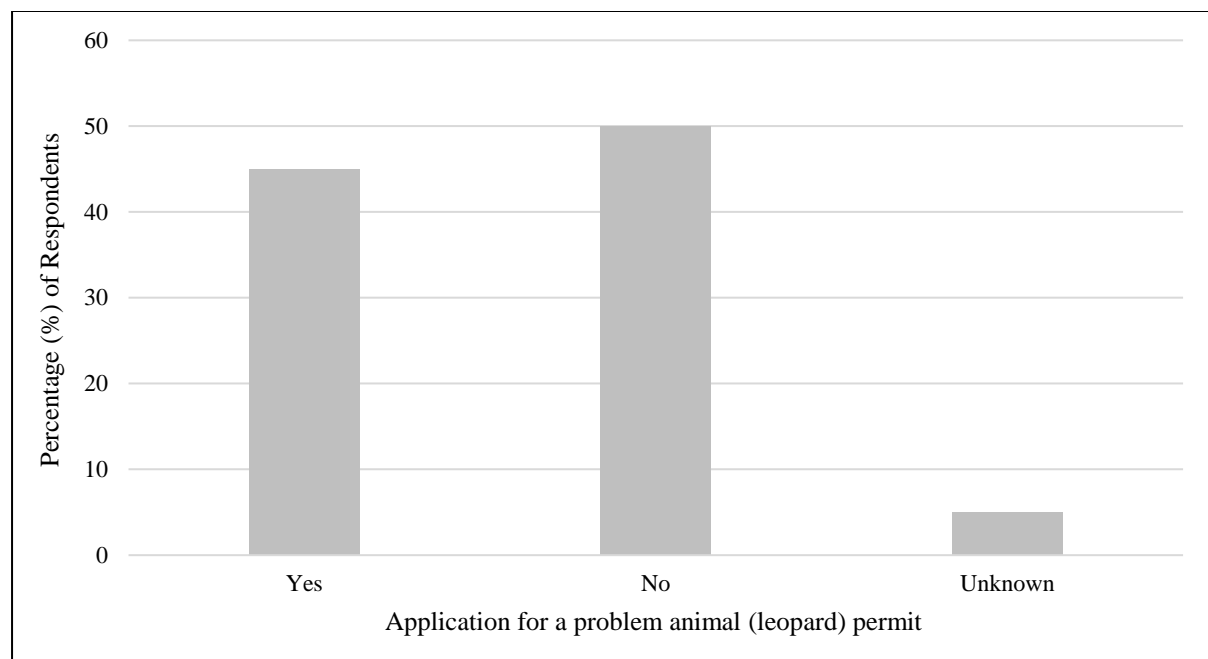


Figure 4.15. The percentage of respondents who did, did not, and did not answer if a Ministry of Environment and Tourism problem animal permit was applied for pre or post removal.

Overall 53% of respondents reported that they want leopard on their property, 33% did not want leopard present on their properties, 8% did not know and 6% did not answer. The questionnaire found that of the 53% two key reasons for wanting leopard on their property; 1) leopards are part of the ecosystem (74%) and 2) the opportunity to trophy hunt a leopard to offset the economic loss of livestock and/or game (17%). The category of 'leopards are part of the ecosystem' covered a broad range of aspects from leopards' aesthetic value, its right to belong and the ecosystem services it provides by controlling other species such as baboons. Another reason for wanting leopard present was for tourism (6%) purposes. The main reason for not wanting leopards present was the loss of livestock (82%) that resulted from their presence in conjunction with loss of game (5%) and loss of income (12%). Examples of respondents statements that were included into the Other (11%) category are; that leopards were 'not a serious problem', 'they do less harm than cheetahs', 'Namibia has enough wildlife', 'they are dangerous', 'not hampering farming'.

4.5.2. The Ministry of Environment and Tourism Problem Animal Records

Between 2005 and 2018 MET recorded a total of 1543 permit records for the removal of problem leopards from freehold farms across Namibia (Table 4.12). The average number of permits recorded per year was 110 ± 25 (SD) with the highest number of permits being recorded in 2010 (Table 4.12).

Additional information was acquired from the motivational letters that landowners submit after reporting the problem animal issue via the telephone. If the sex of the leopard was specified in the letter it was also captured in Table 4.12. Between 2005 and 2018 a total of 1,589 individuals were removed of which 449 were males and 176 were females, this creates an average yearly removal rate of 32 ± 14 (SD) males, 13 ± 8 (SD) females and overall 112 ± 24 (SD) (Table 4.8). However, for 942 individuals the sex was not stated in the motivational letter (Table 4.12). The total number of individuals (1,567) is higher than the total number of records (1,543) as information captured from several motivational letters stated the number of leopards removed was greater than one (Table 4.12). When this information was unavailable it was assumed that one record equated to one leopard removal with no assigned sex.

Table 4.12. The number of Ministry of Environment and Tourism problem animal and the total number of individual leopards (male, female, unknown sex) removed from freehold farms between 2005 and 2018 across Namibia.

	Number of problem animal records	Number of individual leopards removed			
Year		Male	Female	Unknown Sex	Total
2005	72	8	8	56	72
2006	105	13	8	84	105
2007	147	30	11	107	148
2008	109	25	5	80	110
2009	97	18	4	76	98
2010	149	22	1	126	149
2011	134	33	11	92	136
2012	130	32	15	84	131
2013	128	39	16	73	128
2014	122	41	14	69	124
2015	100	48	14	38	100
2016	79	56	18	19	93
2017	77	43	20	15	78
2018	94	41	31	23	95
Total	1543	449	176	942	1567^a
Average per year (±SD)	110 (±25)	32 (±14)	13 (±8)	67 (±33)	112 (±24)
^a Multiple leopards were reported under the same problem animal record					

Of the 1,543 permit records additional information within the motivational letters on the reasons for removal could be collected for 996 records. The predominant reason for landowners requesting a permit and removing leopard from their property was actual loss of domestic animals (cattle, sheep, goats, horses, donkeys) (Table 4.13). Second to that was the loss of game followed by the combined loss of domestic and game animals. Removal of leopard also occurred due to a perceived threat to both domestic and game animals as landowners wanted to mitigate the potential risk of loss (Table 4.13). Human safety was also a concern and another reason for removal (Table 4.13). The ‘Other’ category covers leopards that had drowned in water bodies and involved in vehicle accidents (Table 4.13).

Table 4.13. The reasons reported by landowners to MET for the removal of a problem animal (leopard) on their property.

Reasons for Removal	Number of Records
Actual Loss - Domestic	878
Actual Loss - Domestic and Game	40
Actual Loss - Game	48
Actual Loss - Unknown	1
Perceived Threat - Domestic	11
Perceived Threat - Game	1
Human Safety	10
Other	7
Unknown	92
No Record	477
Total	1565

The overwhelming majority of landowners shot (60%) leopards as the primary method of removal either on its own or in conjunction with cage traps (67%) (Table 4.14). A small proportion utilised gin traps, hunting with dogs and snares (Table 4.14).

Table 4.14. The removal methods utilised by landowners for leopard.

Removal method ^a	Number of times utilised
Shot	940
Cage trap	109
Gin trap	5
Dogs	2
Snare	2
Unknown method	136
No Record	477
Total	1565
^a Do not include leopards captured, released or translocated	

4.5.3. Communal Conservancies

Over 16 years (2001 – 2017), in ten regions across 75 communal conservancies, 5,718 incidents of human-wildlife conflict involving leopard were catalogued (NACSO, 2018). The average number of incidents logged per year was 336. The number of incidents increase across the ten regions was as follows; Karas (1), Oshikoto (2), Kavango (9), Ohangwena (21), Erongo (108), Omusati (118), Omaheke (176), Otjozondjupa (241), Zambezi (329) and Kunene (4,713) (NACSO, 2018).

Leopard incidents were logged in Kunene and Zambezi from 2001, with Erongo following on in 2002, however most of the other regions did not log incidents until post 2008 (NACSO, 2018). This regional pattern reflects the findings of the N/a'an ku sê Foundation (2018 *unpublished*) which determined that 35% of the leopard cases on freehold farms were located in the Khomas region followed by Otjozondjupa (29%) and Erongo (15%). Three communal conservancies Ondjou 103 (Otjozondjupa), Kwandu 119 (Zambezi) and Orupupa 614 (Kunene) showed significant incidents across all the regions compared to all other conservancies (NACSO, 2018). It is important to recognise that survey effort does vary across the communal conservancies which can influence the number of incidents logged. However, the geographical distribution and number of incidents recorded does reflect the reach and growth in recorded human-leopard conflict cases across Namibia.

4.4. Leopard Hunting in Namibia

4.4.1. Trophy Hunts

When asked what evidence was used to ascertain the number, age and sex of leopards visiting baited sites 48% of hunters used spoor tracks, followed by 42% utilising camera traps and 10% from direct sightings. Often several methods were used in combination, 33% of hunters stated that they used a combination of camera tracks and spoor tracks, 29% only used spoor tracks, followed by 23% only using camera traps, 7% used all three methods (camera traps, spoor tracks, sightings), 6% used spoor tracks and sightings and 2% used camera traps on their own. The blind and bait method was the most utilised hunting method (94%), followed by hunting on foot (4%), 2% stated no method and 1% used both bait and blind and on foot tracking. When hunters were asked to describe the health of the hunted leopard 96% stated it to be a healthy individual while 4% said unhealthy, 47% recorded that the leopard had a medium full stomach, 32% of the leopards had full stomachs and 21% had empty stomachs.

Between 2001 and 2018 a total of 2,601 leopards have been trophy hunted across Namibia (Figure 4.16). Figure 4.16 outlines the variation in the number of trophy leopard hunted between 2001 and 2018 in conjunction with the 250 TAG quota limit.

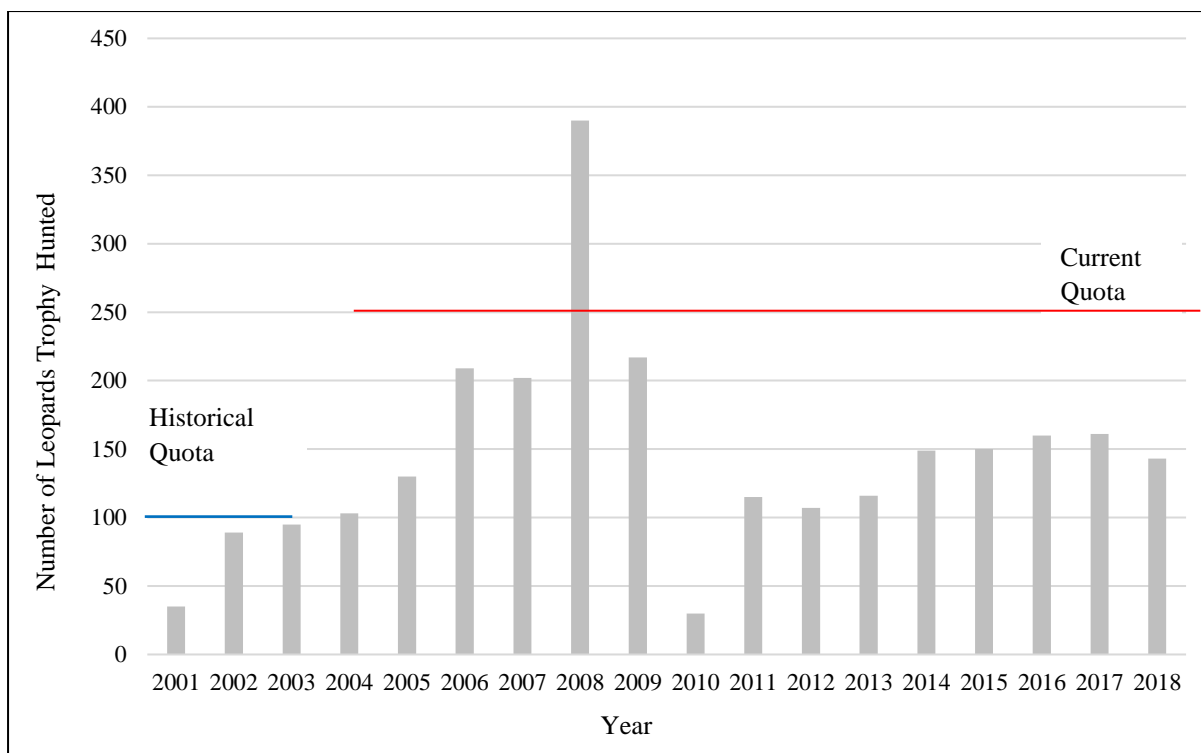


Figure 4.16. The number of trophy hunted leopards and quota limits between 2001 and 2018 across Namibia.

The current (2017-2019) leopard trophy hunting quota for communal conservancies is 33 which represents 13% of the total quota of 250 (Appendix 4 Table 8.4). The quota is spread across six regions and 29 communal conservancies. Of the 29 conservancies undertaking leopard trophy hunting, 20 are in the Kunene region. All the communal conservancies have a quota of one with two exceptions N#a-Jaqna and Nyae Nyae conservancies which both have three per year (Appendix 4 Table 8.4). However, nine of the 29 conservancies across the regions are allocated a single quota which covers the full three year period (Appendix 4 Table 8.4). In 2017 and 2018 the communal conservancies successfully hunted 16 and 17 leopards per year respectively which is, on average, 50% below the maximum yearly quota allocation of 33. Prior to the change in quota allocation in 2017, 2016 had 40 successful hunts, the highest number since 2001 which, in part, was due to five successful hunts in N#a-Jaqna Conservancy and six in Nyae Nyae Conservancy (Appendix 4 Table 8.4). Between 2001 and 2017 conservancies with success rates ranging from 7 to 16 trophy leopards were Otjozondjupa (2 conservancies), Kunene (5 conservancies) and Erongo (1 conservancy) making them the most successful communal conservancies overall.

Five national parks in Namibia have varying quota allocations; Bwabwata West (2/yr), Bwabwata East (2/yr), Namib Naukluft Park (3 per 5 yrs), Western Kavango and Mangetti National Park (2 per 5 yrs), and Waterberg National Park (1/yr) (Appendix 5 Table 8.5). In 2017 five leopards were successfully trophy hunted and in 2018 four leopards were hunted across all the national parks (Appendix 5 Table 8.5). Since 2012 Bwabwata East has had the highest success rate for leopard trophy hunting inside a national park (Appendix 5 Table 8.5). The complete breakdown for successful and unsuccessful hunts between 2010 and 2018 in relation to land use type, regions and hunt days can be found in Appendix 7 Tables 8.7.1 – 8.7.6.

The data for the number of unsuccessful hunts was only available from 2016 to 2018 therefore, only these three years can be compared (Table 4.15). In 2018 of the 143 successful trophy hunts undertaken 85% were in the freehold farmland, 11% were in communal conservancies with 4% inside national parks.

Table 4.15. The total and average number of hunting applications, successful and unsuccessful hunts, and conversion rates of permits to successful hunts recorded between 2016 and 2018.

Year	Number of applications	Successful hunts	Unsuccessful hunts	Total Number of Permits	Conversion Rate of Permits to Successful Hunts (%)
2016	247	160	462	622	26
2017	252	161	392	553	29
2018	286	143	386	529	27
Total	785	464	1,240	1,704	-
Average (±SD)	262 (±21)	155 (±10)	413 (±42)	568 (±48)	27

Over the three years a total of 464 successful hunts took place compared with 1,240 unsuccessful hunts with an average of 155 ± 10 (SD) successful and 413 ± 42 (SD) unsuccessful hunts (Table 14.5). The average conversion rate of leopard permits to successful hunts was 27% (Table 14.5). In terms of application numbers this information was only available from 2016 to 2018, during which time there has been a 15% increase in the number of leopard trophy

hunt applications (Table 4.15). Between 2001 and 2018, 247 successful trophy hunts were undertaken across communal conservancies, national parks, freehold conservancies and community associations with a yearly average of 15 ± 9 (SD) and ranged from three successful hunts per year in 2004 to 40 in 2016 (Appendix 6 Table 8.6).

Across all land use types the region of Otjozondjupa had the largest number of successful hunts with 890 (34%) followed by Khomas 534 (21%), Erongo 462 (18%) and Kunene 394 (15%) (Table 4.16). The regions that have the highest number of successful hunts also have the largest number of unsuccessful hunts (Table 4.16). The unknown category represents farms that did not have any district or regional information and so they were not able to be placed geographically.

Table 4.16. The number of successful and unsuccessful leopard trophy hunts per year and by regions from 2001 to 2018.

	Year	Erongo	Hardap	Karas	Kavango (Ukn)	Kavango East	Kavango West	Khomas	Kunene	Omaheke	Oshikoto	Otjozondjupa	Zambezi	Unknown
Successful hunts	2001	6				1	1	5	5	1		6	1	9
	2002	19	1	1	2	3		17	10	0		31	1	4
	2003	13	1	1		4		25	12	2		37		
	2004	28	1		1	1		19	17	3	2	29	2	
	2005	31	2					32	8	2		54	1	
	2006	32	4					48	32	11	4	77	1	
	2007	36	2					46	38	9	1	70		
	2008	75	5	1		1		90	57	19	7	132	1	2
	2009	37	7	3		1		66	26	8	1	64		4
	2010	4	1					1	2	1		5	1	15
	2011	21	2	1		2		15	23	4		36		11
	2012	23	1			1		14	22	1		35	2	8
	2013	22	2			3		23	16	5	1	41	1	2
	2014	23	2			2	2	26	28	6		53	1	6
	2015	30	3	1		2	1	22	19	9		57	3	3
	2016	20	4	1		3	1	31	33	6		51	4	6
	2017	22	5	1		3	1	31	26	8		55		9
	2018	20	5	2		4		23	20	8		57	2	2
	Total	462	48	12	3	31	6	534	394	103	16	890	21	81
Unsuccessful hunts	Year	Erongo	Hardap	Karas	Kavango (Ukn)	Kavango East	Kavango West	Khomas	Kunene	Omaheke	Oshikoto	Otjozondjupa	Zambezi	Unknown
	2016	90	11			9	2	106	71	23	3	121	4	22
	2017	48	8			10		79	55	22	5	155	4	6
	2018	57	11	1		4		119	50	10	2	121	1	9
	Total	195	30	1	0	23	2	304	176	55	10	397	9	37

The predominant land use on which leopard trophy hunting takes place was the freehold farms with a total of 2,300 successful hunts (Table 4.17). The communal conservancies were the second land use type followed by national parks, freehold conservancies and community associations (Table 4.17). This pattern is repeated for the number of unsuccessful hunts across the different land use types (Table 4.17).

Table 4.17. Successful and unsuccessful leopard trophy hunted between 2001 to 2018 across five land use types.

		Land use type					
	Year	Freehold Conservancy	Communal Conservancy	Community Association	National Park	Freehold Farm	Unknown
Successful hunts	2001	2			2	26	5
	2002	3	1		3	80	2
	2003	5	4		4	82	
	2004		2		1	100	
	2005	1	3			126	
	2006					209	
	2007		7			195	
	2008	1	17			370	2
	2009		6			208	3
	2010		9			21	
	2011		17		2	86	10
	2012		15	1	1	82	8
	2013		10	1	3	98	4
	2014		18		5	119	7
	2015		18		3	125	4
	2016	3	32		5	113	7
	2017		15		5	139	2
	2018		17		4	121	1
	Total	15	191	2	38	2300	55
		Land use type					
	Year	Freehold Conservancy	Communal Conservancy	Community Association	National Park	Freehold Farm	Unknown
Unsuccessful hunts	2016		39		8	410	5
	2017	3	45		2	342	
	2018		35		1	349	1
	Total	3	119	0	11	1101	6

4.4.2. Leopard Sex

Overall 2,091 male leopards have been trophy hunted compared to 484 females, 26 were labelled as unknown as the information on leopard sex was not available. The number of female leopards hunted starts to reduce from 2010, 2001 to 2010 476 females were trophy hunted

compared with 8 from 2011 onwards (Table 4.18). The average number of male leopards trophy hunted between 2011 and 2018 was 136 ± 23 (SD) and between 2016 and 2018 was 155 ± 10 (SD).

Table 4.18. The number of male and female leopards trophy hunted across Namibia between 2001 and 2018.

	Leopard sex			
Year	Male	Female	Unknown	Total
2001	23	10	2	35
2002	61	27	1	89
2003	61	34		95
2004	65	32	6	103
2005	86	43	1	130
2006	138	68	3	209
2007	126	72	4	202
2008	262	123	5	390
2009	151	64	2	217
2010	27	3		30
2011	111	4		115
2012	105	1	1	107
2013	114	2		116
2014	149			149
2015	148	1	1	150
2016	160			160
2017	161			161
2018	143			143
Total	2091	484	26	2601

As anticipated from the number of successful hunts, the regions of Erongo, Khomas, Kunene and Otjozondjupa had the highest number of trophy hunted male leopard (Table 4.19) totalling 1,808 which accounts for 86% of all male leopards trophy hunted in Namibia. Of all the regions Khomas trophy hunted the largest number of female leopards pre 2010 (Table 4.19). Khomas along with Erongo, Kunene and Otjozondjupa accounted for 93% of all trophy hunted female leopards. The freehold farms accounted for 87% of all male trophy hunted leopards and 95% of all female trophy hunted leopards (Table 4.20). The communal conservancies followed the freehold farms as the next largest for male leopards trophy hunted but not for female leopards (Table 4.20).

Table 4.19. Outlines the number of male, female and unknown sex trophy hunted leopards by year and by region from 2001 to 2018.

	Leopard sex																		
	Male																		
Regions	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Erongo	5	12	6	15	24	23	21	49	24	3	21	23	21	23	30	20	22	20	362
Hardap		1	1	1	2	2	2	3	7	1	2	1	2	2	3	4	5	5	44
Karas			1					1	2		1				1	1	1	2	10
Kavango (Ukn)		2																	2
Kavango East		2	4	1				1	1		2	1	3	2	2	3	3	4	29
Kavango West	1													2	1	1	1		6
Khomas	4	10	16	6	18	27	16	36	28	1	14	13	23	26	22	31	31	23	345
Kunene	4	6	7	11	5	19	17	34	23	2	23	22	16	28	19	33	26	20	315
Omaheke	1		2	3	1	7	6	18	7	1	4	1	5	6	9	6	8	8	93
Oshikoto				1		3	1	4	1				1						11
Otjozondjupa	2	23	24	27	35	57	63	116	55	4	33	35	40	53	56	51	55	57	786
Zambezi	1	1			1					1		1	1	1	3	4		2	16
Unknown	5	4							3	14	11	8	2	6	2	6	9	2	72
Total	23	61	61	65	86	138	126	262	151	27	111	105	114	149	148	160	161	143	2091
	Female																		
Regions	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Erongo	1	7	7	10	6	8	15	24	12	1			1						92
Hardap						2		2											4
Karas		1							1										2
Kavango (Ukn)																			0
Kavango East	1	1																	2
Kavango West																			0
Khomas	1	7	9	13	14	20	29	54	38		1	1							187
Kunene	1	4	5	6	3	13	20	22	3										77
Omaheke					1	4	3	1	1										10

Table 4.19. Outlines the number of male, female and unknown sex trophy hunted leopards by year and by region from 2001 to 2018 continued.

	Female																		
Regions	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Oshikoto				1		1		3											5
Otjozondjupa	4	7	13		19	19	5	15	8	1	3		1						95
Zambezi				2		1		1											4
Unknown	2							1	1	1					1				6
Total	10	27	34	32	43	68	72	123	64	3	4	1	2	0	1	0	0	0	484
	Unknown Sex																		
Regions	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Erongo				3	1	1		2	1										8
Hardap																			0
Karas																			0
Kavango (Ukn)				1															1
Kavango East																			0
Kavango West																			0
Khomas						1	1												2
Kunene							1	1											2
Omaheke																			0
Oshikoto																			0
Otjozondjupa		1		2		1	2	1	1						1				9
Zambezi												1							1
Unknown	2							1											3
Total	2	1	0	6	1	3	4	5	2	0	0	1	0	0	1	0	0	0	26

Table 4.20. Outlines the number of male, female and unknown sex trophy hunted leopards by year and by land use type from 2001 to 2018.

	Leopard sex																		
	Male																		
Land use type	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Freehold Conservancy	1	3	4		1											3			12
Communal Conservancy		1	4	1	1		4	9	5	8	17	15	9	18	18	32	15	17	174
Community Association													1						1
National Park	1	2	4	1							2	1	3	5	3	5	5	4	36
Freehold Farm	18	53	49	63	84	138	122	253	143	19	82	81	97	119	124	113	139	121	1818
Unknown	3	2							3		10	8	4	7	3	7	2	1	50
Total	23	61	61	65	86	138	126	262	151	27	111	105	114	149	148	160	161	143	2091
	Female																		
Land use type	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Freehold Conservancy	1		1					1											3
Communal Conservancy				1	2		3	7	1	1			1						16
Community Association																			0
National Park	1	1																	2
Freehold Farm	8	26	33	31	41	68	69	114	63	2	4	1	1						461
Unknown								1							1				2
Total	10	27	34	32	43	68	72	123	64	3	4	1	2	0	1	0	0	0	484

Table 4.20. Outlines the number of male, female and unknown sex trophy hunted leopards by year and by land use type from 2001 to 2018 continued.

	Unknown Sex																		
Land use type	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Total
Freehold Conservancy																			0
Communal Conservancy							1												1
Community Association												1							1
National Park																			
Freehold Farm		1	6	1	3	4	3	2							1				21
Unknown	2						1												3
Total	2	1	6	1	3	4	5	2	0	0	0	1	0	0	1	0	0	0	26

4.4.3. Leopard Age

The data on the age of the trophy hunted leopards from MET was available for the period of 2013 to 2018. Between 2013 and 2018 the average age of a trophy hunted leopard was 8.1 ± 2.6 (SD), the variation on the average age by year was minimal (Table 4.21). The average minimum age was 2.9 ± 0.7 (SD) with a maximum age of 17 ± 2.7 (SD). Across all of the trophy hunted leopards 1.52% were aged 2-3.5, 26.7% was aged 4-6.5 and the largest group was 71.8% aged 7 and above.

Table 4.21. The average age, minimum and maximum age of trophy hunted leopards between 2013 and 2018.

Year	Average age (\pm SD)	Minimum age	Maximum age
2013	7.8 (± 3.1)	3	20
2014	7.9 (± 2.4)	3	15
2015	8.0 (± 2.6)	2	15
2016	8.1 (± 2.4)	4	14
2017	8.6 (± 2.5)	3	18
2018	8.3 (± 2.9)	2.5	20

By region the average age ranged between 7.7 ± 2.5 (SD) in Otjozondjupa to 9.1 ± 5.6 (SD) in the Zambezi (Table 6.8). Both the Zambezi and Kavango East recorded the oldest leopards at 20 years old with Otjozondjupa recording the youngest leopard of two years old (Table 4.22). Karas recorded the smallest age gap, seven years (6-13), between the youngest and oldest leopard trophy hunted. The differences between average ages across the different land use types was minimal, the national parks had the largest age variation (Table 4.23). Data was not available for leopard ages for the Kavango (Ukn) and Oshikoto regions as well as for freehold conservancies and community associations.

Table 4.22. The average age, minimum and maximum age of trophy hunted leopards by region between 2013 and 2018.

Regions	Average age (\pmSD)	Minimum age	Maximum age
Erongo	8.8 (\pm 2.8)	3	15
Hardap	8.2 (\pm 2.7)	5	15
Karas	8.6 (\pm 2.9)	6	13
Kavango (Ukn)	N/A	N/A	N/A
Kavango East	8.5 (\pm 4.2)	2.5	20
Kavango West	8.0 (\pm 3.5)	5	13
Khomas	8.8 (\pm 2.8)	3	16
Kunene	7.7 (\pm 2.2)	3	15
Omaheke	7.8 (\pm 2.2)	3	12
Oshikoto	N/A	N/A	N/A
Otjozondjupa	7.7 (\pm 2.5)	2	18
Zambezi	9.1 (\pm 5.6)	5	20
Unknown	8.7 (\pm 2.3)	5	16

Table 4.23. The average age, minimum and maximum age of trophy hunted leopards by land use type between 2013 and 2018.

Land use type	Average age (\pmSD)	Minimum age	Maximum age
Freehold Conservancy	N/A	N/A	N/A
Communal Conservancy	8.1 (\pm 2.7)	4	20
Community Association	N/A	N/A	N/A
National Park	8.5 (\pm 3.8)	2.5	20
Freehold Farm	8.1 (\pm 2.6)	2	18
Unknown	8.6 (\pm 2.5)	5	16

4.4.4. Hunt Effort

Hunt days equals the number of days required to either achieve a successful hunt or cease hunting due to failure. The number of hunt days were calculated using the hunt start and end date provided for each trophy hunt for the period of 2010 to 2018. When this information was not available the permit start and end date was taken as the proxy measurement. Between 2010 and 2018, 6,880 hunt days were logged for successful trophy hunts with an average of 6.6 ± 4.4 (SD) hunt days per year per hunt (Appendix 7, Table 8.7.1). Across the regions Otjozondjupa logged the most hunt days totalling 2,303 which accounted for 85% of the total number of hunt days across Namibia (Appendix 7, Table 8.7.2). The lowest number of hunt days was recorded in Oshikoto with just three hunt days (Appendix 7, Table 8.7.2). However, there was no significant difference between the number of successful hunt days recorded across all the regions (Kruskal Wallis test, $W = 16.02$, d.f. = 10, $P = 0.099$). The majority of the hunt days

were recorded on the freehold farms which totalled 5,569 days this was followed by the communal conservancies with 873 days, national parks at 178 days and finally freehold conservancies with 6 days (Appendix 7, Table 8.7.3). Still, there was no significant difference between the number of successful hunt days recorded across the different land use types (Kruskal Wallis test, $W = 1.97$, d.f. = 3, $P = 0.578$).

As discussed previously data on unsuccessful hunts was only available from 2016 as such successful an unsuccessful hunt could only be compared over this timeframe. Over these three years a total of 2,619 hunt days were recorded with an average of 6 ± 4.1 (SD) for successful hunts in comparison to 16,506 hunt days with an average of 13.3 ± 17.7 (SD) for unsuccessful hunts (Appendix 7, Table 8.7.4). The majority of the unsuccessful hunt days were logged in 2016 with a total of 6,056, with 2017 having the highest average number of hunt days per unsuccessful hunt (15.3 ± 30.5 SD) (Appendix 7, Table 8.7.4). In relation to the regional variation in hunt days Karas had the highest average number of hunt days per successful hunt 8 ± 4.1 (SD) (Appendix 7, Table 8.7.5). Four regions; Erongo, Khomas, Kunene, Otjozondjupa accounted for 86% (14,229) of all the unsuccessful hunt days recorded. The average number of successful hunt days varied between 4.7 ± 3.3 (SD) in the national parks to 6.1 ± 4.1 (SD) in the freehold farms. The freehold farms recorded 14,493 unsuccessful hunt days, followed by the communal conservancies at 1,735, national parks with 155 and freehold conservancies with 39 (Appendix 7, Table 8.7.6).

Between 2016 and 2018 the month of August had the highest number of unsuccessful hunts closely followed by June and November (Figure 4.17). May recorded 66 successful hunts in comparison to two in February (Figure 4.17). However, there was no significant difference between the number of successful hunt and the month in which they were undertaken (Kruskal Wallis test, $W = 9.54$, d.f. = 9, $P = 0.389$).

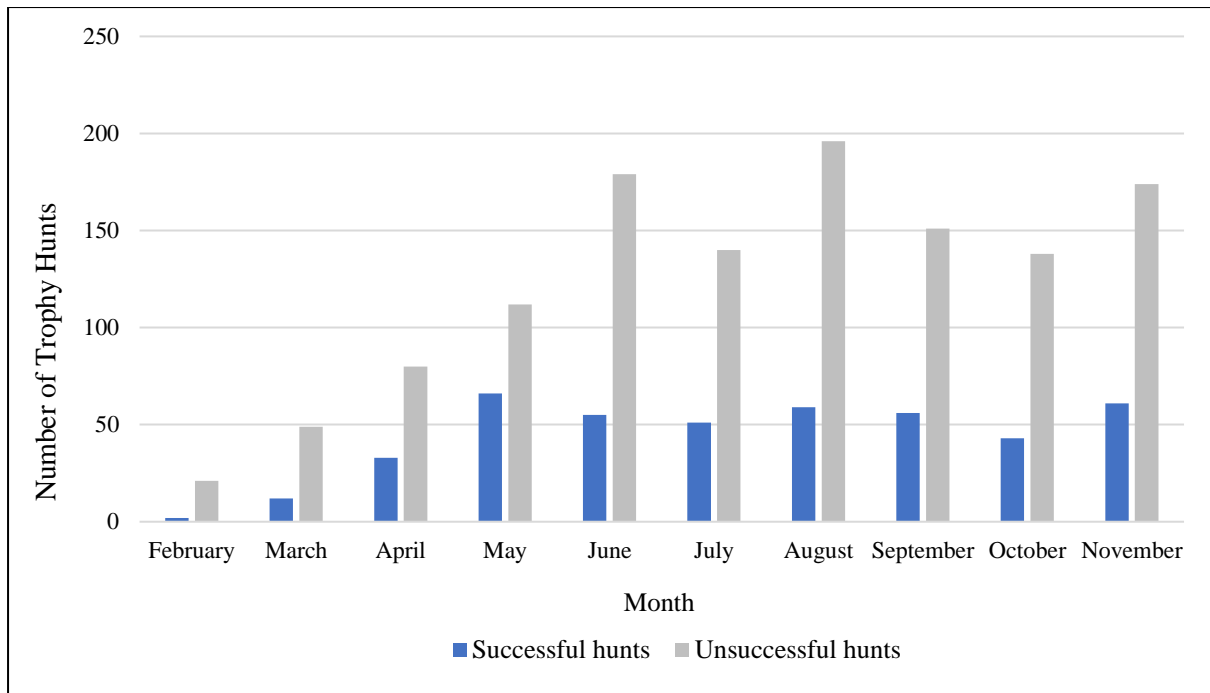


Figure 4.17. The number of successful and unsuccessful leopard trophy hunt recorded per month between 2016 and 2018.

Out of the 256 hunting records that stated if pre-baiting was used during the hunt, 96% of hunters did pre-bait their sites and 4% did not. The average number of baited sites per successful hunt used between 2016 and 2018 was 4.7 ± 3.3 (SD). The highest average number of baited sites used was in the freehold conservancy with 8 sites, followed by the communal conservancies 6.5 ± 3.7 (SD), national parks 5 ± 2 (SD), and the freehold farms 4.4 ± 3.1 (SD). Across all nine known regions the average number of bait nights per successful hunt was 29.1 ± 3.1 (SD), Kunene had the highest average number of bait nights per hunt with Karas having the least, overall Table 4.24 reflects the varying levels of trophy hunting activities across the regions. The freehold farms recorded the majority of the baited nights which resulted in successful trophy hunts, this is to be expected given the fact that 88% of hunts take place on freehold farms.

Table 4.24. The total and average number of bait night nights by region and land use type between 2016 and 2018.

Regions	Total number of bait nights	Average number of bait nights (\pmSD)
Erongo	520	17.3 (\pm 19.9)
Hardap	274	27.4 (\pm 36.0)
Karas	17	8.5 (\pm 2.1)
Kavango East	84	21.0 (\pm 26.7)
Khomas	1868	35.9 (\pm 49.9)
Kunene	1409	38.1 (\pm 47.0)
Omaheke	177	16.1 (\pm 15.3)
Otjozondjupa	2751	28.4 (\pm 34.6)
Unknown	202	25.3 (\pm 17.6)
Zambezi	32	32.0
Total	7334	29.1 (\pm38.1)
Land use type	Total number of bait nights	Average number of bait nights (\pmSD)
Freehold Conservancy	6	6.0
Communal Conservancy	577	20.6 (\pm 21.7)
National Park	163	27.2 (\pm 28.6)
Freehold Farm	6588	30.4 (\pm 39.9)
Total	7334	29.1 (\pm38.1)

When comparing the number of bait nights used against the number of hunt nights recorded for every successful hunt (Figure 4.18) there was a significant weak positive relationship between the number of bait nights and successful hunt days (Spearman's rank, $\rho = +0.730$, $n = 199$, $P < 0.001$).

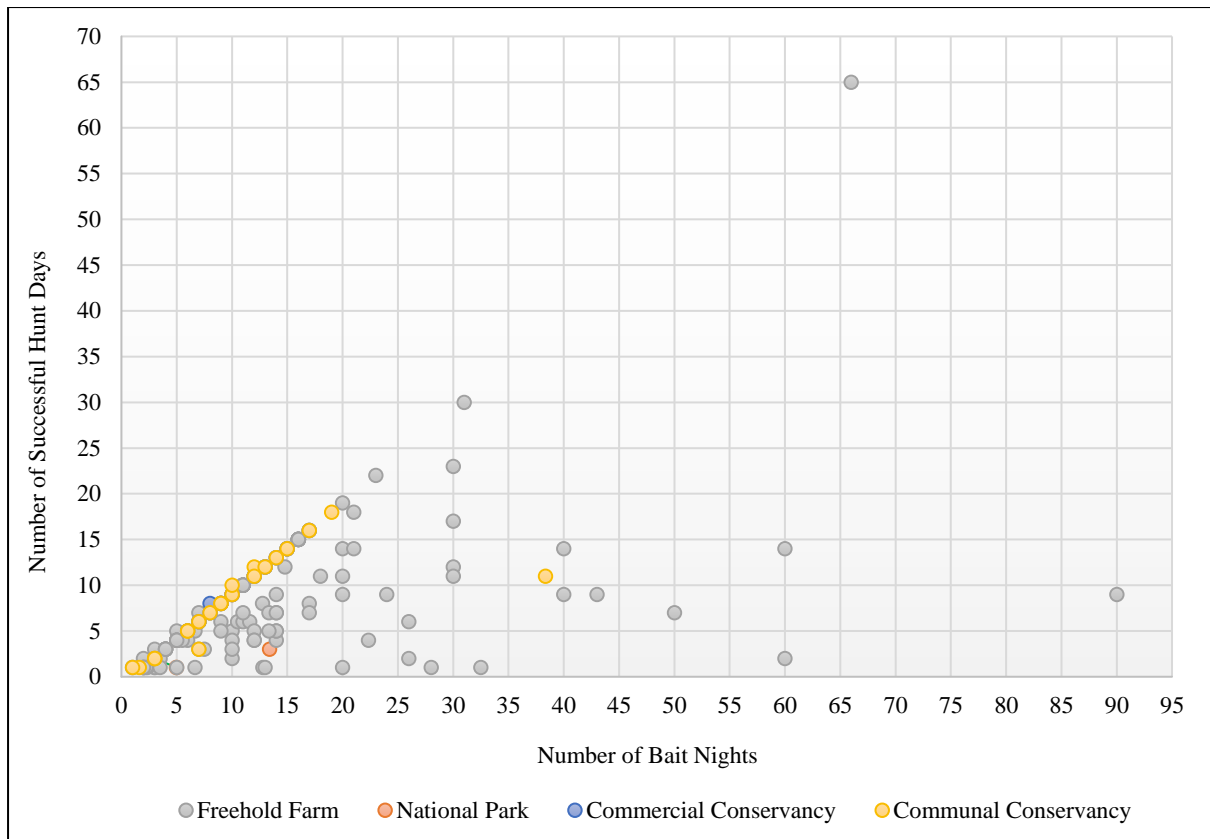


Figure 4.18. The relationship between number of bait nights and number of hunt days per successful hunt by land use type between 2016 and 2018.

Hunt effort was calculated using the data from the number of hunt days, number of bait sites, total number of baited nights, total number of baits used and total number of leopards identified during the hunt. The output equates to the number of leopards identified per hunt effort. The hunt effort was then used as an indicator to compare the input and output for each successful hunt by each region which can be seen in Figure 4.19. However, there was no significant difference between the eight regions and the hunt effort put in for each successful hunt (Kruskal Wallis test, $W = 9.79$, d.f. = 8, $P = 0.280$).

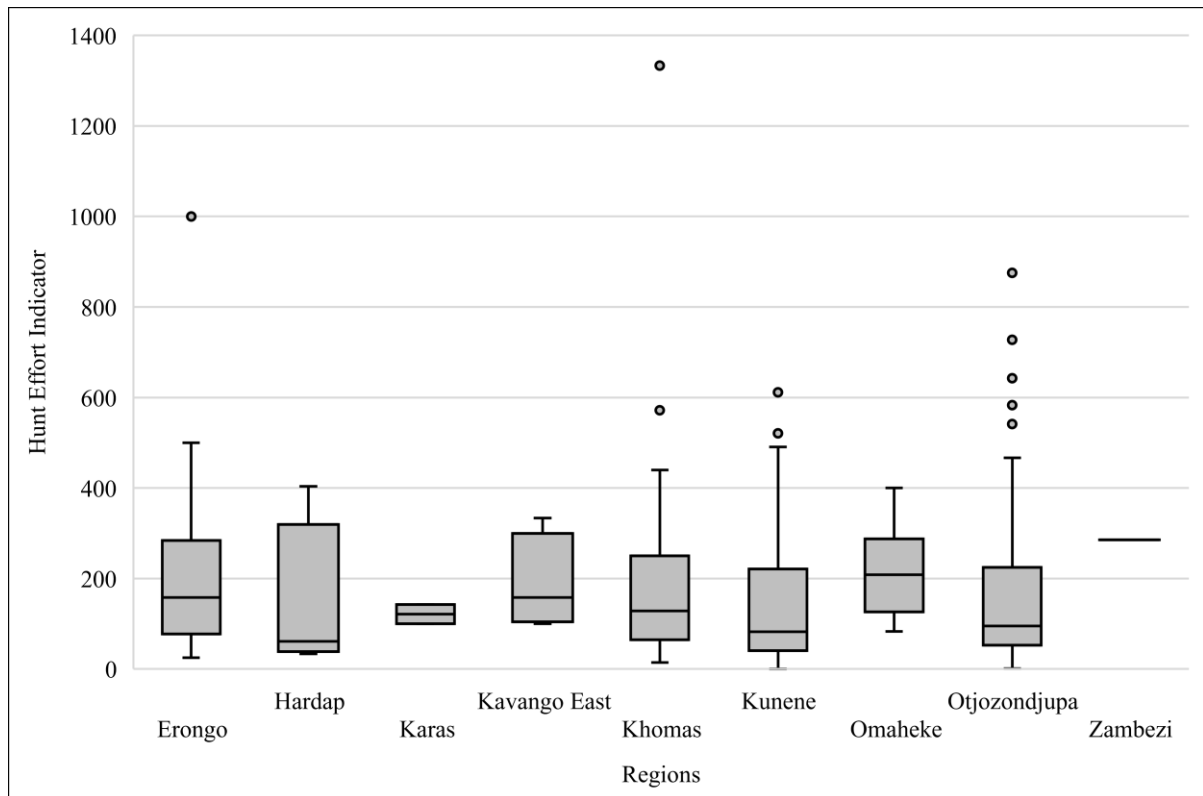


Figure 4.19. The hunt effort determined for successful trophy hunts for each region between 2016 and 2018.

4.4.5. Trophy Size

Leopard trophy size data was available for the following years:

- Weight (kg) and body length (cm): 2001 to 2018
- Neck circumference (cm) and shoulder height (cm): 2016 to 2018
- Skull width (cm) and length (cm): 2010 to 2018
- Skull total (sum of skull width and skull length): 2001 to 2018

As pre 2011 both males and females were trophy hunted but not post 2011 two comparisons of the trophy sizes were undertaken 1) with all the male and female leopards included and 2) with males only.

4.4.5.1. Weight

The highest weight of a hunted leopard was recorded in the Khomas region on a freehold farm at 95kg (2017) (Table 4.25). The lowest weight recorded (15kg) (Table 4.25) was shared

between three individuals, 2 females and 1 male, hunted in the Erongo and Khomas regions on freehold farms (2002, 2003, 2008). The average weight recorded of the leopard in each region ranged from 65kg in Kavango West to 36.05kg in the Zambezi (Figure 4.20).

Table 4.25. The average, minimum and maximum measurements recorded from a trophy hunted leopard.

Trophy measurement	Average (\pm SD)	Minimum	Maximum
Weight (kg)	54.8 (\pm 13.7)	15	95
Body length (cm)	194.7 (\pm 33.4)	32	290
Neck circumference (cm)	54.8 (\pm 9.7)	15	127
Shoulder height (cm)	72.4 (\pm 13)	10	213
Skull length (cm)	24.2 (\pm 2.9)	13.3	54.7
Skull width (cm)	14.9 (\pm 1.7)	11.5	27
Skull total (cm)	37 (\pm 5.7)	12	79

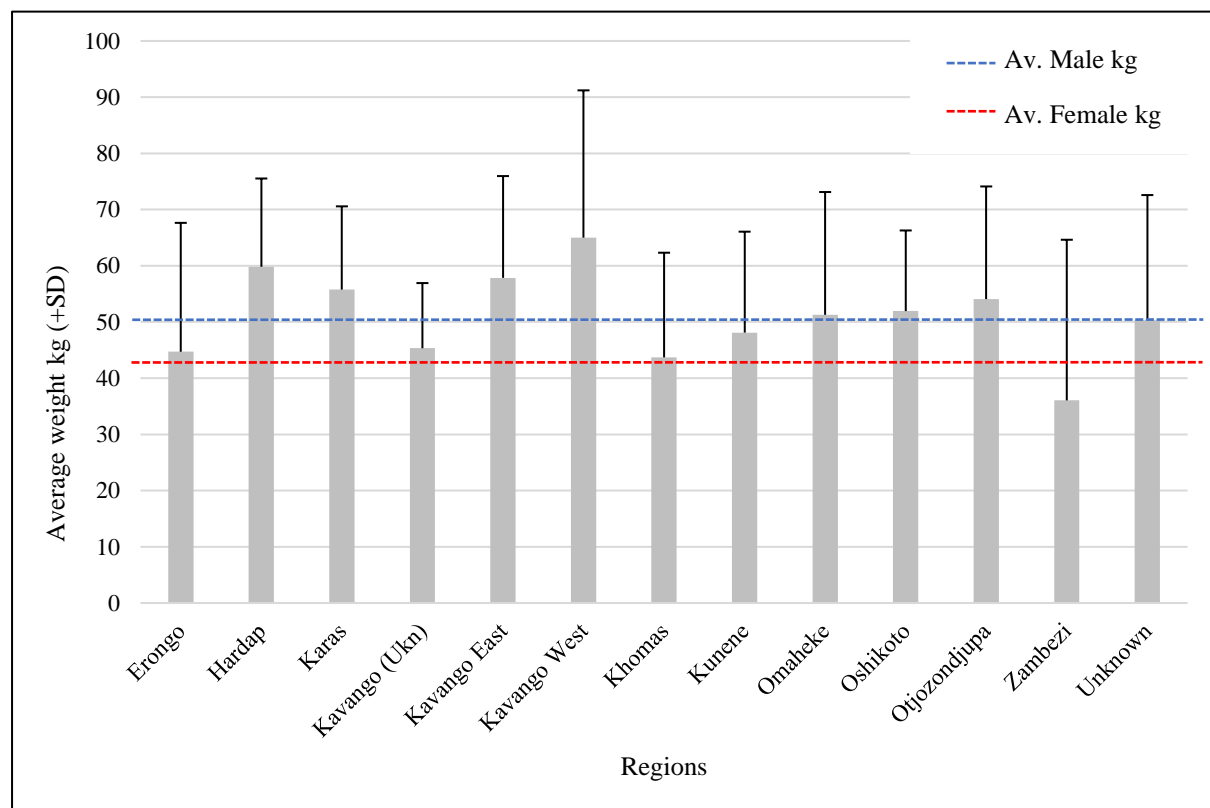


Figure 4.20. Average weight (kg) of trophy hunted leopard by region between 2001 and 2018 in Namibia and the average male/female leopard weight (Estes, 1991).

The weight of a trophy hunted leopard was significantly different across the seventeen years (Kruskal Wallis test, $W = 318.48$, d.f. = 17, $P < 0.001$). In 2001, the lowest average weight was $44\text{kg} \pm 15$ (SD), 2016 and 2017 both had the highest average weight of 63 ± 11 (SD) and 63 ± 9 (SD). Leopards averaging 60kg or more were only recorded from 2013 onwards. Looking at the variation of leopard's weight pre and post implementation of the TAG system (introduced 2011) there was a significant difference between the weight of the two groups (Mann-Whitney U test, $U = 732977.5$, $n = 2078$, $P < 0.001$). The 2011 to 2018 group had an average weight of $60\text{kg} \pm 3$ (SD) in comparison to 2010 to 2001 which averaged less at $50\text{kg} \pm 3$ (SD). However, the leopard's weight did not significantly vary between 2016 and 2018 (Anova test, $F = 1.417$, d.f. = 2, $P = 0.244$). There was a significant variation between a leopard's weight found across three different land use types (freehold, communal conservancy, national park) (Kruskal Wallis test, $W = 8.17$, d.f. = 2, $P = 0.017$). Significant differences were found between the freehold farms and the national parks as well as the communal conservancies and national parks.

Between 2001 and 2018 the average male weight was $59\text{kg} \pm 11.9$ (SD), without females the male average weight increased by 4.2kg. With male only trophy hunts there was still a significant variation between a leopard's weight recorded between 2001 and 2018 (Kruskal Wallis test, $W = 103.48$, d.f. = 17, $P < 0.001$) and pre and post TAG implementation with leopards weighing more post TAG (Mann-Whitney U test, $U = 445113.0$, $n = 1743$, $P < 0.001$).

4.4.5.2. Body Length

The smallest leopard body length of 32cm (2009) for two females was recorded in the Erongo region and the largest body length 290cm (2013) for a male was recorded in the Otjozondjupa region all on freehold farms. Between 2001 and 2018 the body length (cm) of a trophy hunted leopard was significantly different (Kruskal Wallis test, $W = 358.12$, d.f. = 17, $P < 0.001$) (Figure 4.21). The smallest average body lengths were recorded in 2002 (162 ± 48 SD) and the largest in 2015 (214 ± 21 SD). There was a significant variation of all trophy hunted leopard's body length pre (2001-2010) and post (2011-2018) implementation of the TAG system (Mann-Whitney U test, $U = 785072.0$, $n = 2146$, $P < 0.001$). Prior to the TAG system the average body length was 183 ± 9.6 (SD) and after its implementation was 207 ± 5.1 (SD). However, there was no significant variation of body length between 2016 and 2018 (Kruskal Wallis test, $W = 4.08$, d.f. = 2, $P = 0.130$) and across the three land use types (Kruskal Wallis test, $W = 5.62$, d.f. = 2,

$P = 0.060$). The largest average body length was found in the Kavango East region with the smallest being recorded in the Erongo region (Figure 4.22).

Between 2001 and 2018 the average male body length was $201\text{cm} \pm 31.0$ (SD), without females the male average body length increased by 6.3cm. With male only trophy hunts there was still a significant variation between a leopard's body length recorded between 2001 and 2018 (Kruskal Wallis test, $W = 156.71$, d.f. = 17, $P < 0.001$) and pre and post TAG implementation with body length being bigger post TAG (Mann-Whitney U test, $U = 501031.5$, $n = 1785$, $P < 0.001$).

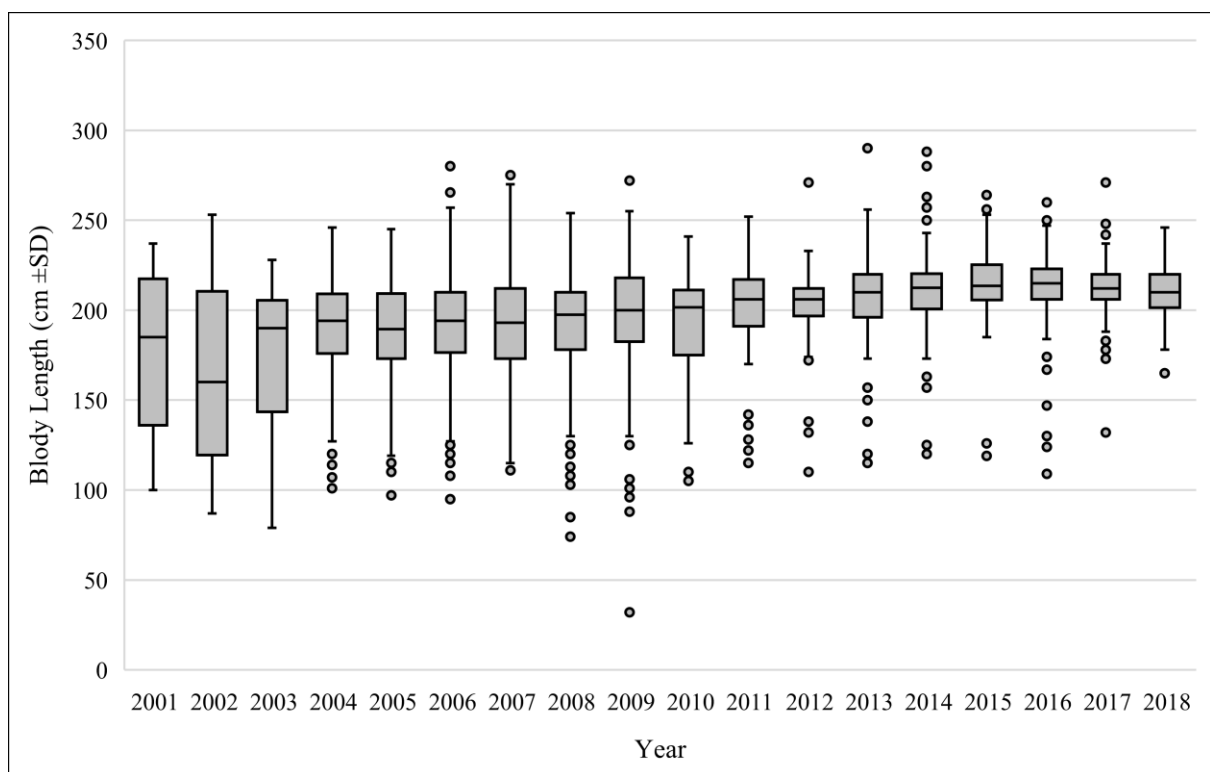


Figure 4.21. Variation in trophy hunted leopard body length (cm) between 2001 and 2018.

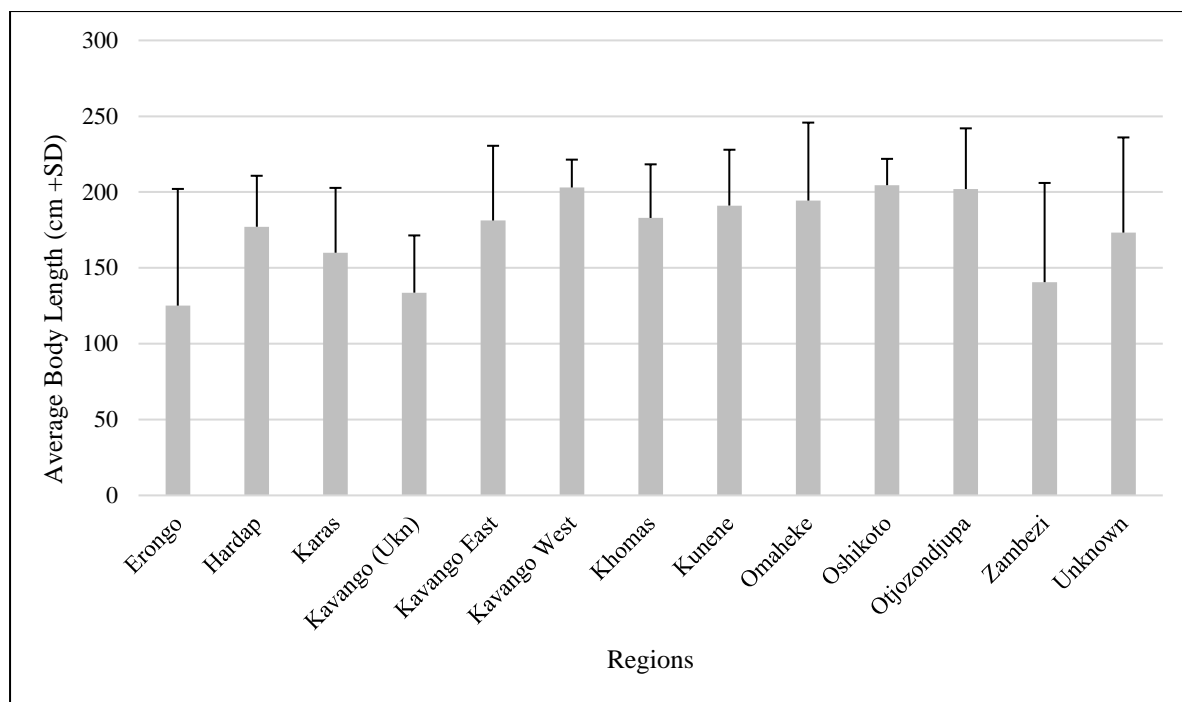


Figure 4.22. Average trophy hunted leopard body length across the regions between 2001 and 2018.

4.4.5.3. Shoulder and Neck

Between 2016 and 2018 there was no significant difference in the neck circumference (cm) recorded across the three land use types (Kruskal Wallis test, $W = 5.64$, d.f. = 2, $P = 0.060$) or across the years (Kruskal Wallis test, $W = 1.14$, d.f. = 2, $P = 0.493$). However, for the shoulder height (cm) there was a significant difference recorded over the three years (Kruskal Wallis test, $W = 6.44$, d.f. = 2, $P = 0.040$). There was also a significant difference recorded between the three land use types (Kruskal Wallis test, $W = 8.69$, d.f. = 2, $P = 0.013$). Significant differences were found between the communal conservancies and the national parks as well as the freehold farms and the national parks. The variation in neck and shoulder measurements across the regions can be seen in Figure 4.23.

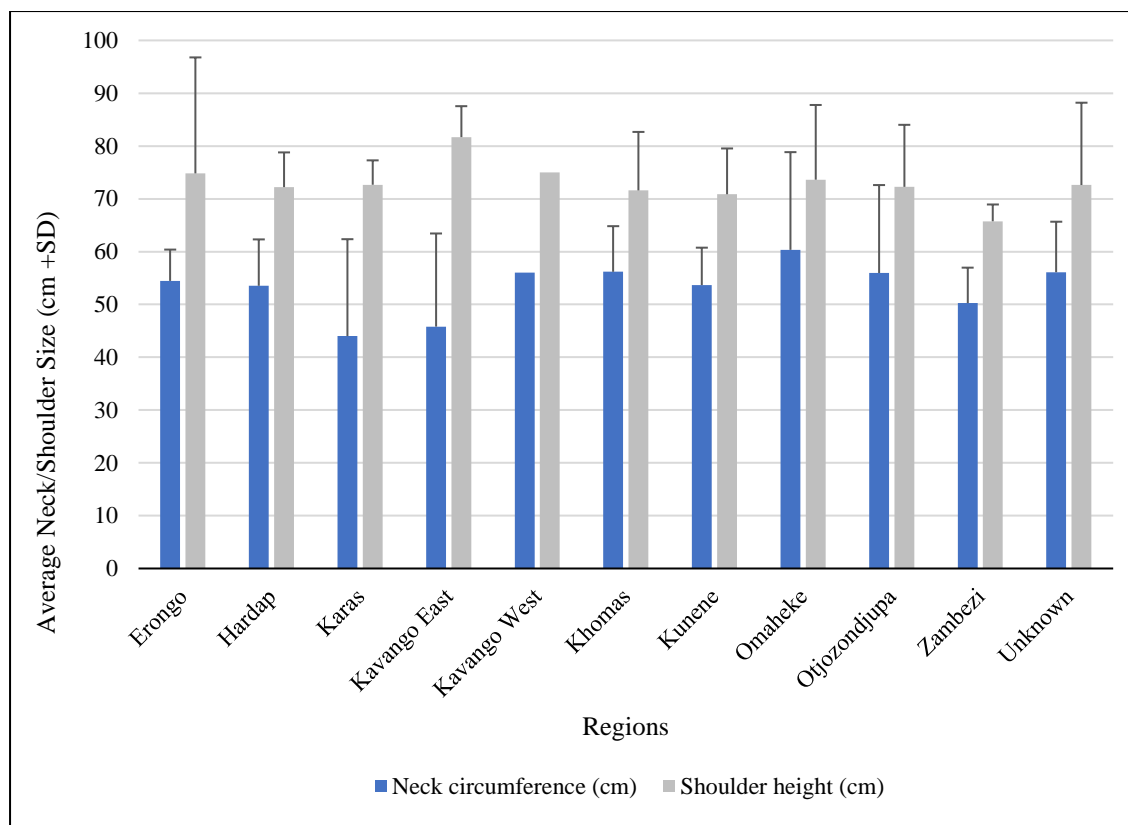


Figure 4.23. The average size of the leopard's neck circumference (cm) and shoulder height (cm) across the regions between 2016 and 2018.

4.4.5.4. Skull Size

Between 2010 and 2018 the skull length for all trophy hunted leopards showed a significant difference (Kruskal Wallis test, $W = 34.11$, d.f. = 8, $P < 0.001$). The skull width also showed significant variation over the eight years (Kruskal Wallis test, $W = 32.99$, d.f. = 8, $P < 0.001$). The significant difference in size was due to the 2012 results in comparison to the other years, particularly 2015 and 2017. The variations in skull sizes across the years can be seen in Figure 4.24.

Between 2010 and 2018 the average male skull width was $15\text{cm} \pm 1.7$ (SD) and skull length was $24\text{cm} \pm 2.9$ (SD). There were minimal differences between any of the average skull measurements with and without the inclusion of female leopards. With male only trophy hunts there was still a significant variation between a male skull length recorded between 2013 and 2018 (Kruskal Wallis test, $W = 32.36$, d.f. = 8, $P < 0.001$) and the skull width (Kruskal Wallis test, $W = 34.20$, d.f. = 8, $P < 0.001$).

The Nature Conservation Ordinance 4 of 1975 section 84 conditions of predator trophy hunting permits 114C. (5a) states that a minimum skull measurement of 32cm is required for leopard. Between 2001 and 2016, 7.1% ($n = 104$) skull total measurements for males only were below 32cm. However, post TAG implementation between 2011 and 2018 for males 0.7% ($n = 5$) were below the 32cm requirement.

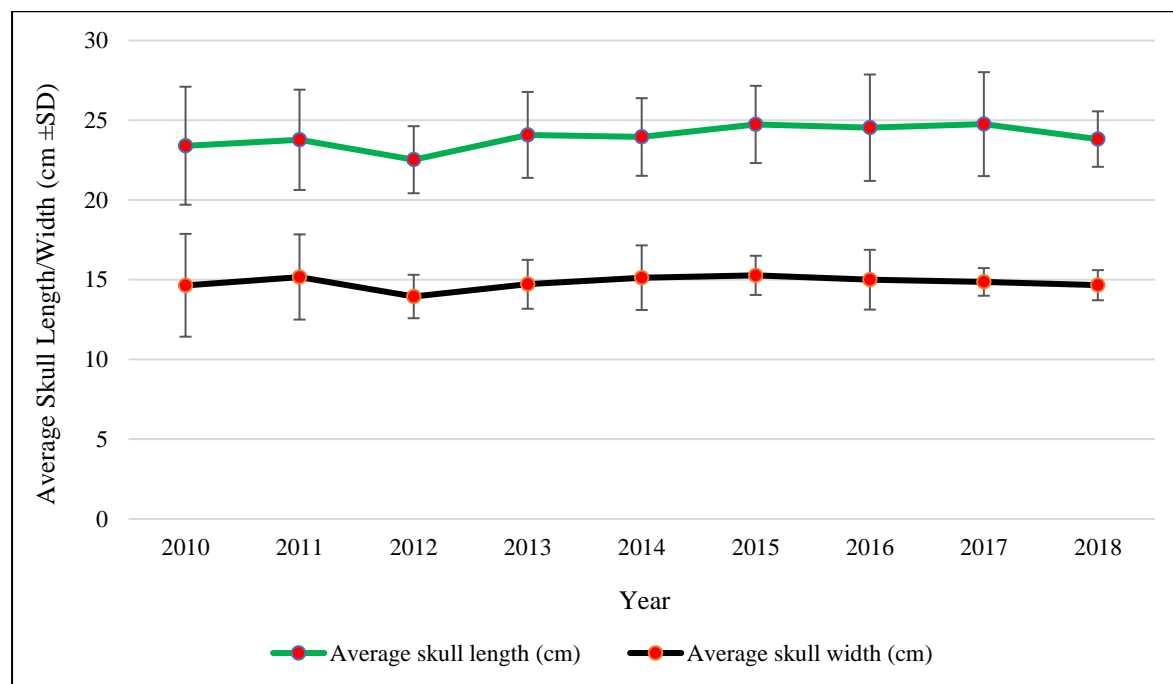


Figure 4.24. Average skull length (cm) and width (cm) measurements for all trophy hunted leopards between 2010 and 2018.

Between 2001 and 2018 the skull total (skull width plus length) (cm) showed significant variation across the eighteen years (Kruskal Wallis test, $W = 191.23$, d.f. = 17, $P < 0.001$) (Figure 4.25). The average skull total size across the 17 years was $37.7\text{cm} \pm 2.1$ (SD), 2001 had the smallest average skull size at $31.6\text{cm} \pm 8.5$ (SD) and 2015 on average had the largest skull size at $40.1\text{cm} \pm 3.17$ (SD) (Figure 4.25). In relation to skull size and the regions, Omaheke had the largest average skull size (total) followed by Kavango West then Hardap (Figure 4.26). All the variations in all the skull measurements across the regions can be seen in Figure 4.26.

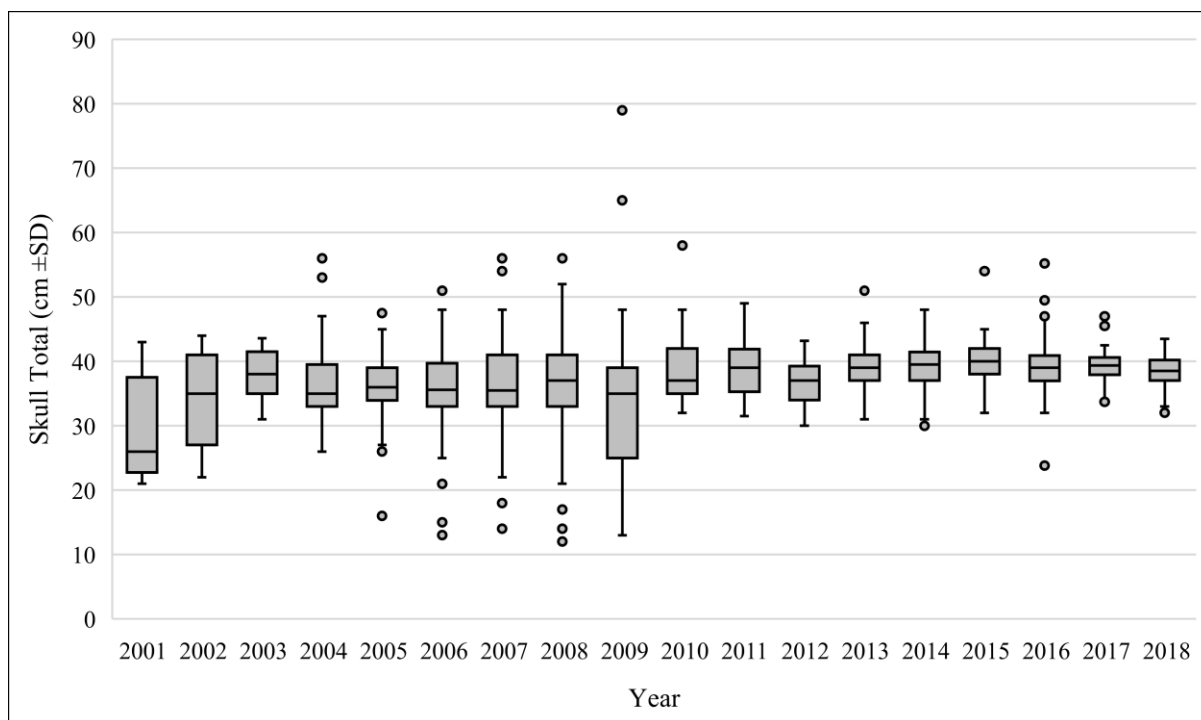


Figure 4.25. The skull total (cm) measurements for trophy hunted leopards between 2001 and 2018.

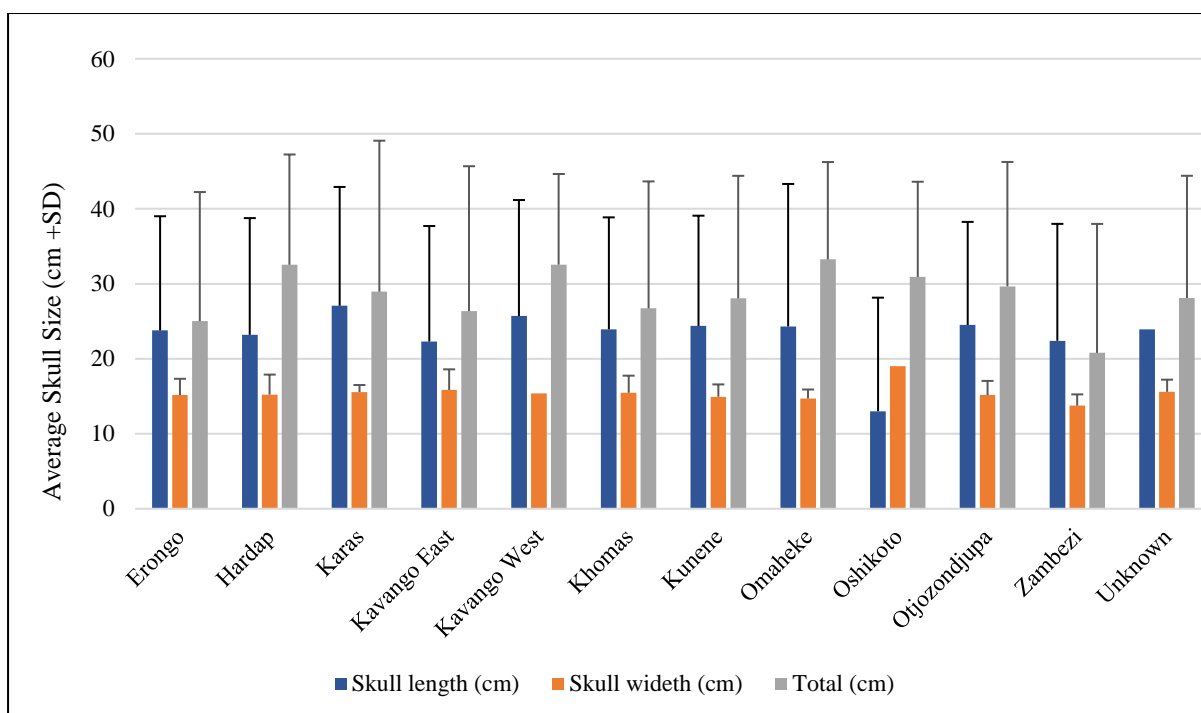


Figure 4.26. The average size of skull length, width and total (cm) across the regions between 2010 and 2018.

4.4.6. Activity Patterns

To determine the activity patterns for leopard across Namibia data from this study's two camera trap surveys was utilised as well as data from the Wiesel and Edwards (2014) human-carnivore-conflict study on farms bordering the Namib Naukluft and Tsau//Khaeb National Parks. Each camera trap photograph was categorised by hour. As one leopard could create three photographs and another 20 due to the time they spent in front of the camera only the first image of each time event was used to avoid any bias. The hourly categories were then defined across four groups; twilight pre sunrise, day, twilight post sunset and night. These groups reflect the trophy hunting time regulations as twilight pre sunrise is the 30 minute period pre sunrise and twilight post sunset is the period of 30 minutes after sunset. As the twilight times can vary by day, month and year, every date was individually categorised. This is why overlap between the twilight categories and the day/night categories has occurred (Figure 4.27).

A total of 552 time events were captured across the three study areas. Of that the majority were captured during the night 84.8%, followed by 8.9% in the day and 4% in the twilight post sunset and 2.4% during the twilight pre sunrise (Figure 4.27).

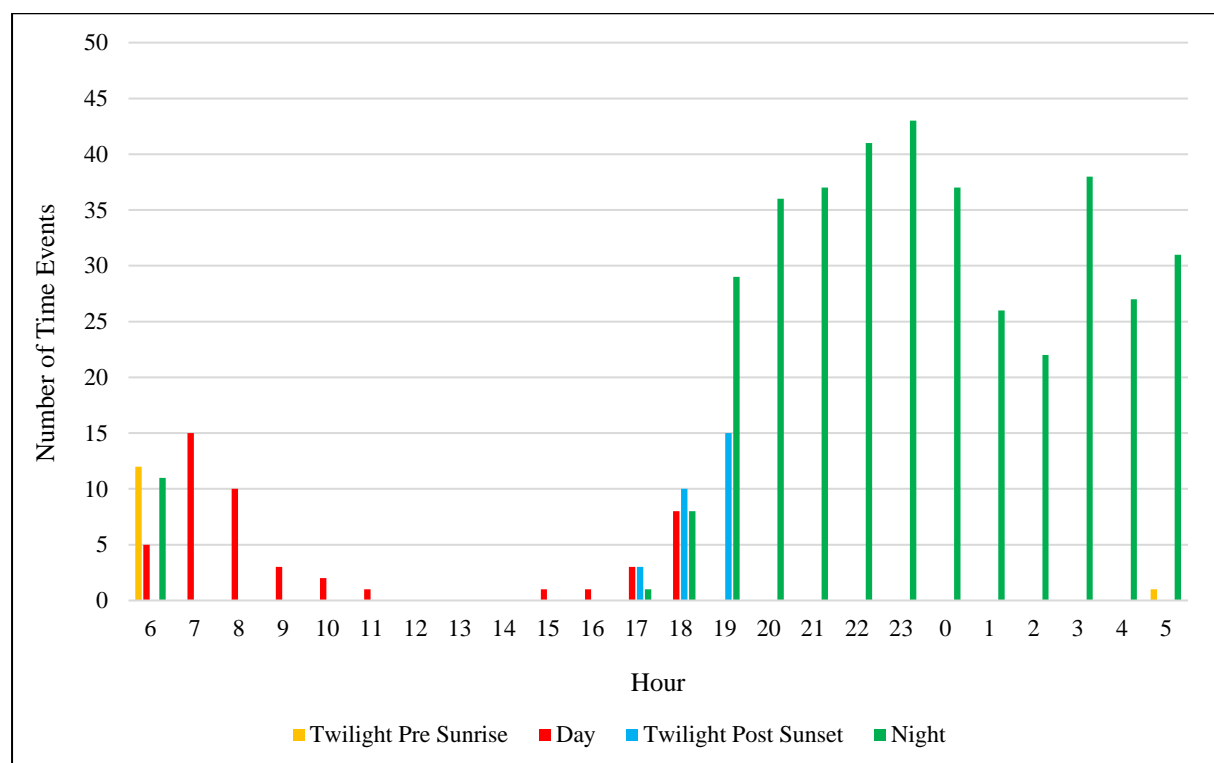


Figure 4.27. The number of time events captured for all leopard across the three camera trap survey areas broken down into four activity groups.

Activity patterns were also looked at in relation to moon phases. The category of ‘Total Full Moon’ is a combination of ‘Pre Full Moon’, ‘Full Moon’ and ‘Post Full Moon’. The ‘Pre Full Moon’ equates to the week prior to full moon, ‘Full Moon’ is the day of the full moon and ‘Post Full Moon’ is the week following the full moon. ‘Outside Full Moon’ is the two weeks which fall outside these times. Across the three survey areas 53% of time events were captured outside of the full moon period with 47% being recorded inside the full moon period (Figure 4.28). Of that 47% inside the full moon, 48% of time events were pre full moon, 6% during full moon and 46% post full moon (Figure 4.28).

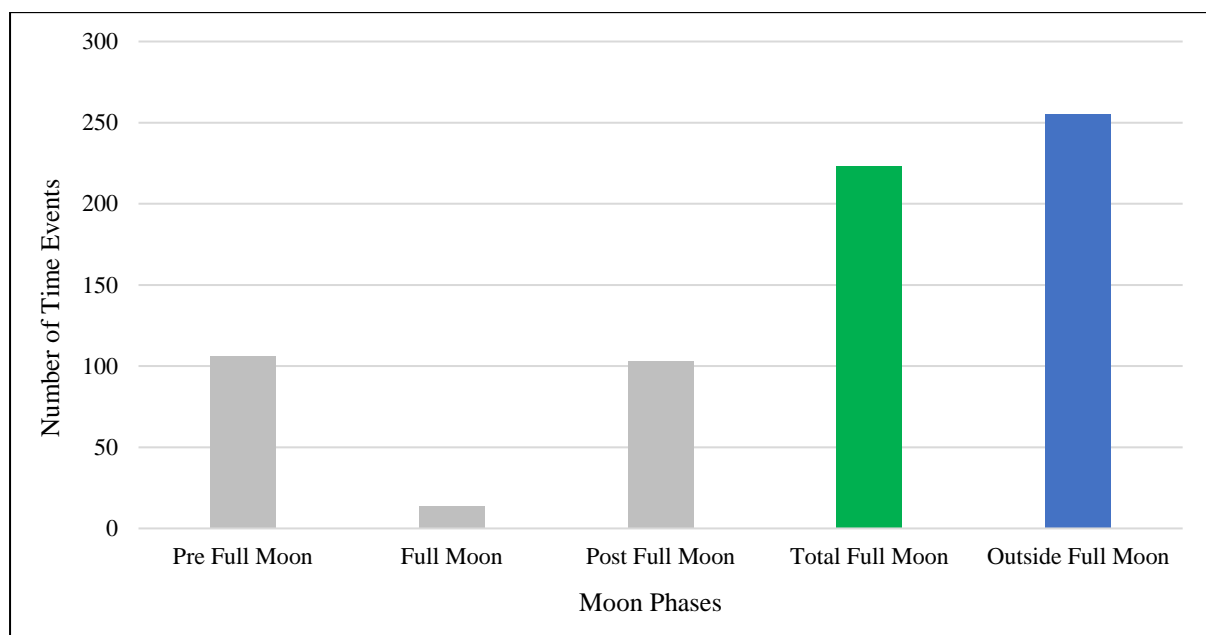


Figure 4.28. The total number of time events for the three survey areas captured across the moon phases.

For the two survey areas of the Auas Mountains and Omaruru the sex of each identified leopard was known. A total of 161 time events across the two survey areas were captured for male leopards only (Figure 4.29). The night category captured the majority of the time events (84%), followed by the day (9%), twilight post sunset (6%) and twilight pre sunrise (1%) (Figure 4.29). The time events for both male and female leopards was compared, and no significance difference was found (Mann-Whitney test, $U = 9960.0$, $n = 288$, $P = 0.706$).

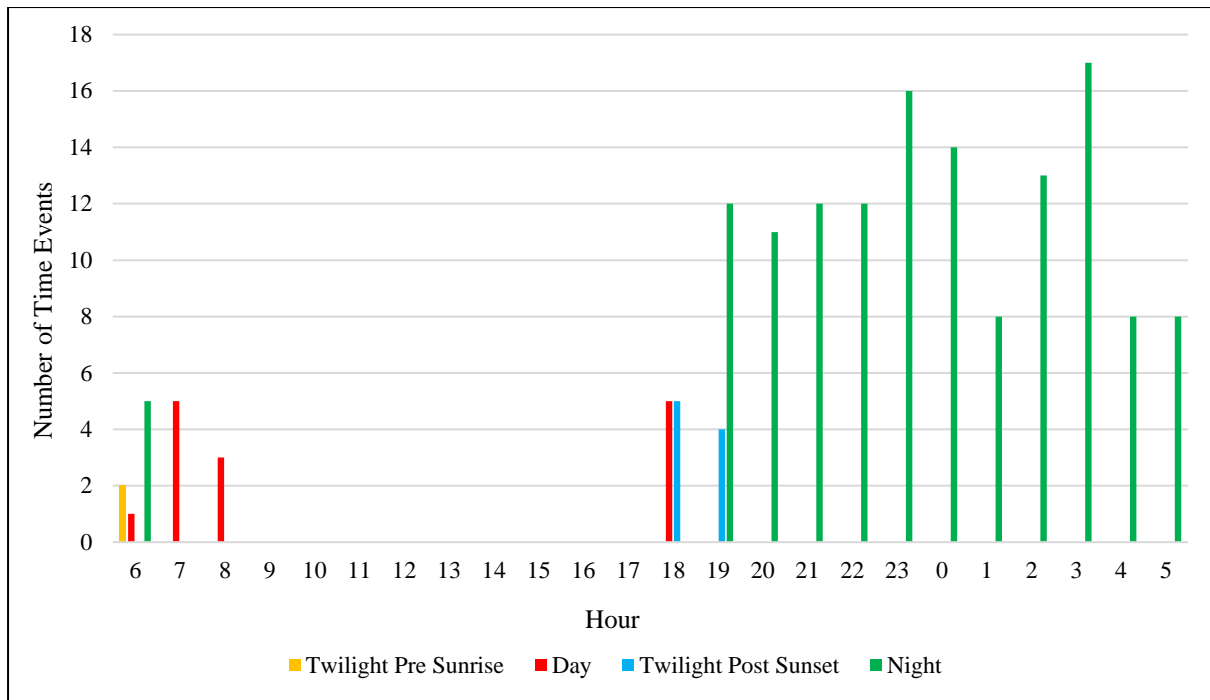


Figure 4.29. The total number of time events for male leopards only across two camera trap survey areas broken down into four activity groups.

For male leopards 52% of time events were captured outside of the full moon period. Within the full moon period 48% of time events were captured, of that 48%, 47.4% were captured post full moon, 46.2% pre full moon and 6.4% during full moon (Figure 4.30). Overall there was no significant difference between the time events captured across the moon phases (Mann-Whitney test, $U = 22762.0$, $n = 425$, $P = 0.766$).

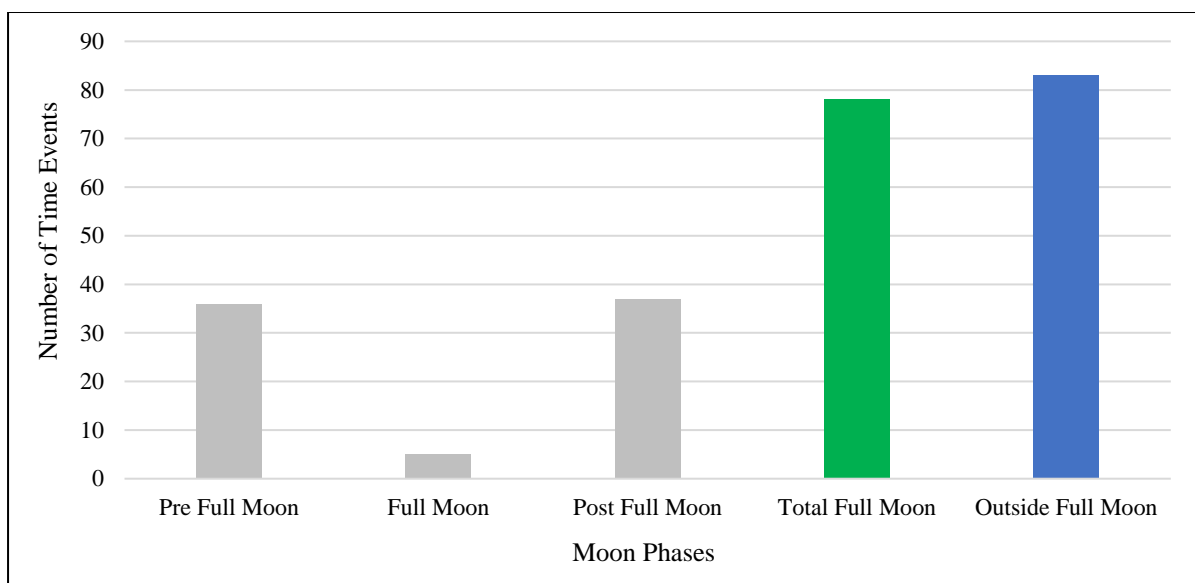


Figure 4.30. The total number of time events for just male leopard across two survey areas captured across the moon phases.

4.4.7. Trophy Hunting Quota Allocation

The quota allocations per interval were calculated from the national leopard population SPACECAP derived density figures. A 3% offtake is considered a sustainable level of harvesting for leopards (Caro *et al.*, 2009). The current off-take rate is 3-4% of male leopards, which assumes a sex ratio of 1:1 (adult male to adult female) (Stein *et al.*, 2011b). Density intervals 1 (0-0.5) and 2 (0.5-1) were combined and intervals 6 (2.5-3) and 7 (3+) were also combined to make the five quota zones (Table 4.25).

The quota system allows for 1.8% of the leopard population to be utilised for trophy hunting. Therefore the 2011 population estimate of 14,154 allows for a total of 250 TAGs to be issued annually. The revised population estimate of 11,733 gives a new annual TAG allocation of 211, as can be seen in Table 4.25.

Table 4.25. The five quota zones defined by leopard density intervals and their associated TAG quota.

Quota Zones	Density intervals (leopards per 100km ²)	Number of leopards	Root mean square error	Area (km ²)	Percentage of area (%)	Quota per interval
1	0-1	1600	1963.5	273752	33	29
2	1-1.5	1770	1026.5	142482	17.2	32
3	1.5-2	3955	1623.6	227253	27.4	71
4	2-2.5	2908	947.2	133144	16	52
5	2.5-3+	1500	388.2	54764	6.4	27
	Total	11,733	5,949	831,395	100	211

The geographical coverage of the five quota zones can be seen in Figure 4.31 and where the quota boundaries fall in relation to freehold farms, national parks and communal conservancies (Figure 4.32).

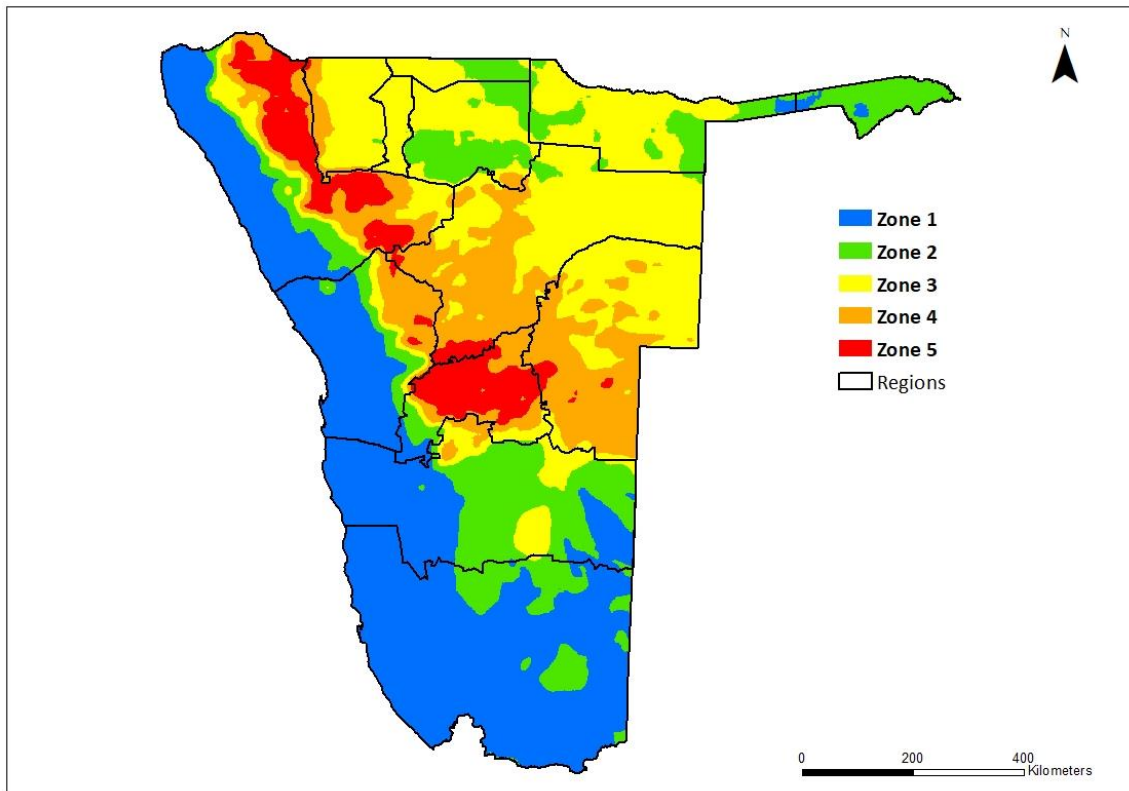


Figure 4.31. The geographical coverage of the five quota zones for leopard trophy hunting in Namibia.

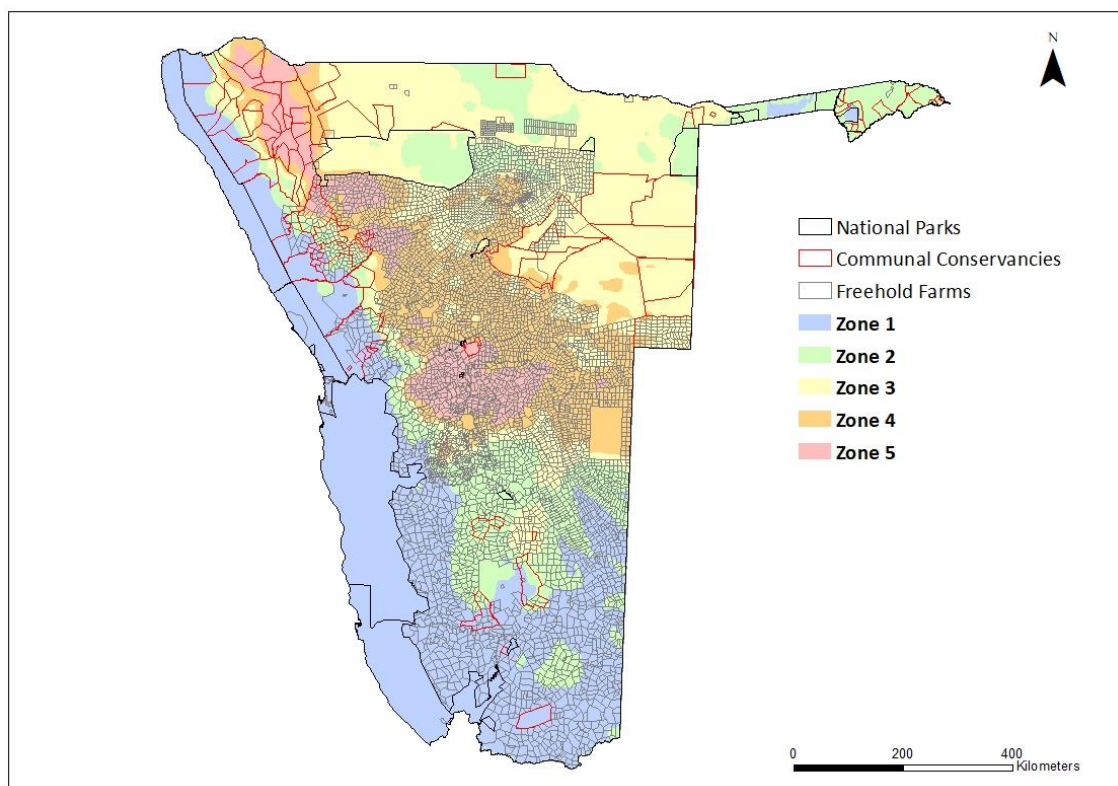


Figure 4.32. The geographical coverage of the five quota zones for leopard trophy hunting in Namibia in relation to freehold farms, national parks and communal conservancies.

5. Discussion and Conclusions

The National Leopard Census Study was commissioned to provide an update on the status of the Namibian leopard population across three key areas; Namibian leopard ecology, Conflict and Sustainable use.

18 months of ecological enquiry and the co-operation of stakeholders and research partners has brought together data and anecdotal evidence from across Namibia to build an objective view of Namibian leopard status on a landscape level. The outcomes of these investigations are provided in the Results sections of this report and were used to support the conclusions and recommendations outlined below.

5.1. Trends in Leopard Numbers

The current understanding is that leopards inhabit most of Namibia except for the highly populated north-central region, the arid southeast farmlands and the desert coast and are absent from 30% of their historic range (Hanssen and Stander 2004; Stein *et al.*, 2011b; 2016).

This study agrees with the previous studies that leopards inhabit most of Namibia except for the desert coast. However, it disagrees with the belief that leopards do not inhabit the north-central region or the arid south east farmland. Multiple sightings of leopard have been recorded in the suburban areas of Windhoek, such as Olympia and Avis. This follows the trend of leopards being found in other semi-urban and suburban environments across their geographical range, for example Mumbai and Johannesburg, due to their opportunistic nature (Moyer *et al.*, 2008; Murphy and Macdonald, 2010; Gehrt *et al.*, 2011; Kuhn, 2014; Brackowski *et al.*, 2018). In the 2011 national leopard study the south east region of Namibia was categorised as ‘No known occurrence’ due to a lack of records at the time of the study (Stein *et al.*, 2011b). With the collaborative effort of multiple stakeholder groups across Namibia this study has captured presence records for leopard in areas that were previously described as ‘No known occurrence’. This is particularly important for the east and south-east of Namibia as a more complete picture of leopard presence has been achieved which are in line with the

communications from the landowners who reside in these areas. A proportion of these new presence records for the south-east are also outside the current IUCN Red List distribution for leopard in Namibia (Stein *et al.*, 2016). By acquiring this new information, the study has achieved one of its primary objectives, to consolidate all leopard presence records to update the known areas of occurrence. This new information can be integrated into the long-term conservation and management strategy for leopard across Namibia.

Leopard inhabit a wide range of habitats and climatic zones, including; mountains, rocks, bushveld, woodlands, desert and semi-desert, forest, from sea-level to 2000m, and in areas which receives less than 100mm of rain to areas receiving above 1200mm (Stein *et al.*, 2016). An example of the leopard's adaptability can be seen in the Namib Desert, where vegetation on the banks of watercourses provides cover, which is a contributing factor to leopard presence in the area (Mills and Hes, 1997). Leopards within the Sperrgebiet and Namib-Naukluft National Park borders were found to prefer open plains, mountains and mountain transition habitats, but they moved away from the mountains in the winter (Edwards *et al.*, 2015). In southern Namibia there are several large private reserves that were previously areas of farmland, with limited carnivore presence but, due to the protection they afford, they are now a refuge for multiple carnivore species including leopard. In the NamibRand reserve leopards have been sighted in the dunes, a habitat considered to be marginal due to its limited resources (Tindall *pers. obs.*, 2018). Leopards have regularly been captured on camera traps along the western edge of the Nubib mountains. In Gondwana Canyon Park leopard sightings continue to increase particularly in the southern, western and northern side of the park, which is due to the area being more mountainous and further away from the eastern farmland (Hartung *pers. comm.*, 2018). Between 2003 and 2014 there was a threefold increase in game numbers, this increase in prey availability is considered to be one of the main factors for the increase in leopard presence inside the park (Hartung *pers. comm.*, 2018).

The habitat suitability model produced by this project agreed with other studies that altitude, land cover, temperature seasonality and land ownership had the greatest influence over suitable habitat for leopard. For example, in Hobatere and western Etosha, the leopards showed a strong preference towards koppies and mountainous habitat (Stander *et al.*, 2001). All these environmental variables alongside known leopard densities were included in the Random

Forest model in order to create the national leopard densities. This ensured that the leopard densities were geographically linked to the environmental variables that had the greatest influence over the leopard population across Namibia.

It has been determined that leopard resource use is governed by three key factors: avoidance of anthropogenic disturbance, selection of prey-rich areas and selection of rocky areas with adequate vegetative cover to increase hunting success and minimise kleptoparasitism (Pitman *et al.*, 2017). For example, in the Savé Valley Conservancy, Zimbabwe the leopard density on the private land was 7.6 leopards/100km² whereas in the resettlement areas leopard could not be detected (Williams *et al.*, 2016). One reason for this was the extensive increase in bushmeat poaching in the resettlement areas which reduced prey availability (Williams *et al.*, 2016). The leopard density in the Niassa National Reserve, Mozambique ranged from 2.18 - 12.65 leopards/100km² between 2008 and 2010 with riparian habitats having double the leopard density when compared to miombo forests (Jorge, 2012). The leopard density of 7.9 leopards/100km² in the Tarangire National Park, Tanzania was a result of the medium to low prey availability (Msuha, 2009). It is the variations in the available resources that have contributed to the substantial differences in leopard densities across Africa.

One significant factor in the decline of the African leopard population is decreasing prey numbers as leopard population density across Africa tracks the biomass of their principle prey species, medium and large-sized wild herbivores (Marker and Dickman 2005, Hayward *et al.*, 2007; Stein *et al.*, 2016). The relationship between prey availability and leopard density can be seen in the results of this study. The highest leopard densities were found in areas which, according to the questionnaire respondents, have the highest density of game per 100km² compared to livestock density. Prey availability and habitat type are of course linked but these results highlight the importance of having a healthy prey base in order to sustain a leopard population.

Carnivore densities are also influenced by competitive interactions with other carnivores in the community. Competitive predator interactions can be based on exploitation or interference (Linnell and Strand, 2000). Of the large carnivore species, the study found that across both the

Auas Mountains and Omaruru survey areas, leopard and brown hyaena were photographed regularly at sites throughout the survey area. Cheetah were captured at one camera site in the Auas Mountains and two cameras sites in Omaruru which could be attributed to the survey design rather than a reflection of cheetah presence in the area. In Omaruru, spotted hyaena were captured in groups of up to three individuals across multiple farms throughout the survey area. It was unclear if the spotted hyaenas were transient or resident but according to the landowners spotted hyaena had only recently been sighted in the area. Loss of cattle on one of the survey farms was attributed to the spotted hyaenas based upon spoor tracks and the landowner's own camera trap footage. As part of the survey the spotted hyaenas were photographed chasing a herd of cattle on the farm which had recorded the recent losses. Interspecific competition can have strong influences on the distribution and abundance of carnivores (Creel, 2001). Even so, the increases in the leopard density recorded for both the Auas Mountains and Omaruru survey areas shows that interspecific competition may not be adversely affecting the population growth to a significant degree. However, if the spotted hyaena numbers were to increase and territories became stable across the freehold farmland then interspecific competition could become an additional pressure on the leopard population due to increased kleptoparasitism, injuries and mortality (Stein *et al.*, 2015) and would need to be closely monitored. In the freehold farmland the leopard is an apex predator, however, inside some national parks and adjoining landscapes, such as those found in the north east of Namibia. Interspecific competition between the leopard and resident lion, wild dogs and spotted hyaenas in these areas could be a contributing factor to the low leopard densities found across this region of Namibia.

Leopard density in Namibia range from 0.25 leopards/100km² in the Mudumu Landscape north east Namibia (Hanssen and Singwangwa, 2019 *unpublished*) to 13.5 leopards/100km² in the private Okonjima Nature Reserve, central Namibia (Noack, 2016). Between 1997 and 2019 seven leopard studies recorded a density of 1.0 leopards/100km² or lower, particularly in the north-east and southern Namibia (Stein *et al.*, 2011a; Edwards *et al.*, 2015; Hanssen *et al.*, 2015; Portas *et al.*, 2018 *in prep.*; Edwards *et al.*, 2018; Hanssen and Singwangwa, 2019 *unpublished*; Hanssen *et al.*, 2019 *unpublished*). Similar leopard densities have also been recorded in the Central Kalahari Game Reserve, Botswana (0.4 leopards/100km² (CARACAL, no date)), the Kgalgadi Transfrontier Park, Botswana, across both tree-savannah (0.19 leopards/100km²) and dune-savannah (0.6 leopards/100km²) habitats (Funston *et al.*,

2001). In the Hoanib River (north-west Namibia) only one leopard was detected as such a density estimate could not be quantified (Portas *et al.*, 2018 *in prep.*). Four leopard studies in the north-east of Namibia recorded leopard densities between 1.1 leopards/100km² and 2.5 leopards/100km² (Stander *et al.*, 1997; Funston *et al.*, 2014; Hanssen *et al.*, 2017; Portas *et al.*, 2018). Comparative leopard densities were recorded in the farmland of the Ghanzi region, Botswana (1.08 leopards/100km² (Kent, 2011)), Xonghile Game Reserve, Mozambique (1.53 leopards/100km² (Strampelli, 2015)), Okavango Delta (Kwando area), Botswana (1.5 leopards/100km² (CARACAL, no date)), and the north eastern portion of Hwange National Park, Zimbabwe (1.46 leopards/100km² (Loveridge *et al.*, 2017)). A further four studies from the centre and north of Namibia recorded leopard densities between 2.6 leopards/100km² and 5.6 leopards/100km² (Stander and Hanssen, 2000; Stander *et al.*, 2001; Stein *et al.*, 2011a; Stratford *et al.*, 2018 *unpublished*) which, are comparable to densities found in the Buby Valley Conservancy, Zimbabwe (2.8 – 5.3 leopards/100km² (du Preez, 2014)), Luambe National Park, Zambia (3.36 leopards/100km² (Ray, 2011)), and the Game Management Area Chanjuzi, Zambia (4.79 leopards/100km² (Ray, 2011)).

The highest density recorded in Namibia was 13.5 leopards/100km² in the private Okonjima Nature Reserve (Noack, 2016) which is an outlier result for Namibia. A large proportion of the recorded leopard density across Africa falls between 5.6 leopards/100km² and 13.5 leopards/100km² and above. For example, the Mpala Conservancy, Kenya, which is a mixture of cattle and wildlife recorded an average leopard density of 12.03 leopards/100km² (O'Brien and Kinnaird, 2011). Overall, leopard densities recorded across South Africa were consistently higher than those recorded in Namibia for example; Kruger National Park at 12.7 leopards/100km² (Maputla *et al.*, 2013), Soutpansberg mountains found 10.7 leopards/100km² (Chase Grey *et al.*, 2013), and northern Kwazulu-Natal recorded 12.7 leopards/100km² (Maputla *et al.*, 2013). Furthermore, the Karongwe private reserve game reserve in South Africa has one of the highest recorded leopard densities at 18.8 leopards/100km² (Owen *et al.*, 2010) as the reserve has comparative conditions to those found in the Okonjima Nature Reserve, no persecution and high prey availability.

Some leopard densities have been influenced by their location within a protected area for example, a density of 11.11 leopards/100km² was recorded inside the core protected area in the

Phinda-Mkhuze Complex, South Africa (Balme *et al.*, 2010). In the buffer of the protected area the density reduced to 7.51 leopards/100km², and the transition into the non-protected areas saw density decrease again to 2.49 leopards/100km² (Balme *et al.*, 2010). The variation between densities inside national parks and those found on the borders was also recorded in the Luangwa Valley, Zambia where the density ranged from 8.50 – 5.08 leopards/100km² respectively (Rosenblatt *et al.*, 2016). In Gabon the maximum leopard density outside of Lopé and Ivindo National Parks was 2.7 leopards/100km². In comparison, the maximum density recorded inside the two national parks which was substantially higher at 13.16 leopards/100km² (Henschel, 2008).

When comparing Namibia to other African countries a leopard density range of 0.25 - 5.6 leopards/100km² would be considered as low. However, it is important to note that these findings are influenced by the variation in survey effort across the different the regions of Namibia. The Namibian studies also highlight that leopard density in the national parks and communal conservancies are lower than those found in the freehold farmland which contrasts with other countries. In South Africa the low density value for the non-protected areas (2.49 leopards/100km²) is still higher than those of Namibia's national parks such as Khaudum National Park (1.8 leopards/100km², Portas *et al.*, 2018) and Bwabwata National Park (0.58 – 0.85 leopards/100km², Hanssen *et al.*, 2019 *unpublished*). However, even the highest density recorded in the freehold farmland in Namibia (3.6 leopards/100km²) is still relatively low compared to other studies both inside and outside of protected areas.

The study found that the national leopard population estimation in 2019 (11,733) was lower than the estimated population in 2011 (14,154). However, it is essential to recognise that this lowering has been driven by significant reassessment of the density intervals across Namibia and therefore represents an increase in the accuracy of the population estimate rather than a potential change in the population. Advances in spatial modelling and statistical analysis allow for the use of sixteen environmental variables in conjunction with SPACECAP derived density figures to create seven graded density levels. By comparison, the 2011 report was only able to assign three density levels. As a result, the area defined as high density and holding 67% of the leopard population in the 2011 report has decreased from 308,091km² (Stein *et al.*, 2011b) to 2,783km². Multiple leopard density studies have been conducted across Namibia between 1997

and 2019 of which a large proportion were in the north-east. Studies undertaken between 2017 and 2019, in the north-east recorded leopard densities ranging from 0.25 - 1.27 leopards/100km² (Funston *et al.*, 2014; Hanssen *et al.*, 2015; Hanssen and Singwangwa, 2019 *unpublished*; Hanssen *et al.*, 2019 *unpublished*). Therefore, another factor contributing to the change in the national population figure is the increased survey effort since 2011, especially in the north-east of the country. The results of which have shown that the leopard density should be categorised as low instead of its previous categorisation of high at 2.0 - 3.1 leopards/100km² (Stein *et al.*, 2011b).

Overall, 64% of questionnaire respondents indicated that they believed the national leopard population had increased over the last five years. Of the three camera trap surveys areas incorporated into the study, two indicated a healthy and growing leopard population (Auas Mountains, 40% increase, and Omaruru, 16% increase), which is in line with the perception of questionnaire respondents in those regions. The remaining site, Namib-Naukluft/Tsauu//Khaeb National Parks, showed a lowering by 38%, which is at odds with the respondents from the region. The study does recognise that the camera trap survey for this area was completed in 2013 and as such changes to density may have occurred since that time. In recognition of this limitation the study has recommendations that this area should be re-survey to determine if the population density has altered since 2013. However, the results do still highlight the fact that leopard densities are not consistent across Namibia. Both this study and the N/a'an ku sê Foundation (2018 *unpublished*) found a high density of leopards in the Windhoek area which was also in line with the small home range sizes recorded there. The N/a'an ku sê Foundation (2018 *unpublished*) determined that leopard home ranges in the Okakarara area where much larger which reflected a low density, these results are reflected in density estimate produced by this study for the area.

During a ten year study of a leopard population that had been trophy hunted for a proportion of time before the practice was halted showed that, as the leopards were in a private reserve, trophy hunting was the only pressure placed on the population, the ratio of adult males to adult females varied depending on the trophy hunting status. During the trophy hunting period the ratio was 1:1 however, after trophy hunting stopped the ratio became 1:4 adult males to adult females (de Woronin Britz *pers. comm.*, 2018). It was identified that the ratio of 1:1 represented

a population in flux, whereas 1:4 indicated a stable population. The sex ratio in the Auas Mountains and Omaruru areas was approximately 1:1 (adult male:adult female) and would therefore be described as a population in flux. Even though Omaruru removed a low level of problem animals illegal activities, such as bushmeat poaching, was very high. In 2017, as a result of the illegal activity a leopard was killed in a wire snare on one of the survey farms. This study captured images of both brown hyaena and spotted hyaena with the remains of wire snares around their necks. All the landowners confirmed that poaching using wire snares across the study area was a major issue. Anti-poaching patrols were instigated which led to the monthly collection of hundreds of snares, despite the patrols efforts landowners felt that illegal activities, such as setting snares, was on the rise. Anti-poaching patrols are an ongoing additional economic cost to the landowner, a cost that could be offset by generating income through trophy hunting. In areas where there has been an increase in the level of illegal hunting of ungulates livestock predation by carnivores has the potential to be amplified (Soofi *et al.*, 2017). The impact of poaching on the ungulates, leopard and other carnivore species is poorly understood in this area and in others with a significant number of freehold farms. It is an additional pressure on the carnivore populations that needs to be better understood and quantified to inform long-term management strategies including sustainable use.

In conclusion, the study has found that leopards are widespread across Namibia, inhabiting a variety of habitats and land use types. As with leopard populations in other African countries Namibia's leopard densities vary significantly across the country. The highest densities were found in the freehold farmland and the lowest densities inside national parks and communal conservancies. The pressures on the leopard population from human-leopard conflict, illegal activities, prey availability and interspecific competition are linked to their geographical distribution. The limited number of leopard studies in Namibia, outside the north-east of the country, needs to be addressed to better understand the variability in the population and the pressures which create these differences. This information can then be used to inform the national leopard management plan and ensure their long-term conservation.

5.2. Hunting Sustainability

Trophy hunting provides an income for a large number of people across Namibia and therefore has an influence on the tolerance people and communities have to target species such as the leopard. Effectively assigning an economic value to leopard is critically important for stakeholders in a multi-use landscape, a point that has been raised multiple times by landowners in discussions regarding livestock loss, trophy hunting and retribution killing of leopards in Namibia. Landowners consistently indicated that if the leopard loses its economic value, particularly through trophy hunting, then the number of unreported removals of problem animals will rapidly increase to halt to their economic losses.

The questionnaire found that a proportion of the respondents (12%) used leopard trophy hunting as a tool to compensate for livestock loss. Furthermore, 17% of the respondents who held positive views to leopard presence on their land ascribed this to the opportunity to trophy hunt leopard to offset the negative economic impact as a result of livestock and/or game losses from leopard predation. The consensus is that, for attitudes towards wildlife to be positive across Namibia, landowners need to receive an economic benefit in the form of ecotourism, a livestock compensation scheme, or be provided with financial incentives for carnivore conservation such as trophy hunting (Romanach *et al.*, 2007; Funston *et al.*, 2013.). In Namibia's case financial incentives can be derived from the sustainable utilisation of leopard through trophy hunting. The recognition of the value of wildlife from trophy hunting across the Namibian communal conservancies has had a two-fold impact. One, a reduction in poaching and two, recovery of wildlife populations (Weaver and Petersen, 2008).

The trophy hunting of female leopards was prohibited in Namibia in 2011. Despite the change in policy there were records for females post 2011 due to hunter misidentification of sex. However, these records represent a very small proportion of successful trophy hunts overall and the policy has seen a significant decrease in female leopard offtake. Prior to 2011 female leopards represented 32% of the total successful hunts compared to 0.7% post implementation. The ratio of male to female leopards hunted and the ratio of adult to sub-adult leopards hunted in the absence of population information could provide an indicator as to the health of the population. A declining or over-harvested population will lead to an increase in the proportion

of female and sub-adult males being hunted as the primary target, the male, has been depleted (Brackowski *et al.*, 2015). Namibia no longer allows the hunting of females but the presence of females as well as males is captured on the remote camera traps used by hunters to monitor their baited hunting sites. The Schedule G record form captures this information as to the number, sex and age of leopards that are present in the hunting area. Using the ratios of the leopard's sex and age gathered through the record form, across the regions of Namibia, could provide a useful management tool in the absence of localised population data. A rapid assessment of this information could be made throughout the course of the hunting season, and between seasons, which could lead to an adaptive management strategy that would see the re-allocation of permits to different geographical areas to protect a potentially over stressed local population. Before applying this strategy as a management tool, the concept would need to be tested to ensure that the data provided by the hunter's camera trap photographs are acting as an indicator of the health of that local population. This could be achieved by simultaneously running independent camera trap surveys and hunter camera trap surveys in the same hunting area.

Another change which may be attributable to the introduction of the TAG system was a significant increase in body weight, length and skull size. This was the case when all trophy hunted leopards, males and females, were compared and when male only leopards pre 2011 were compared to males post 2011. In Tanzania a trophy male leopard must measure at least 1.30m from the tip of the nose to the base of tail (CITES, 2002) whereas Mozambique minimum trophy size is 1.20m (Annex 1 CITES, 2018b). In Zimbabwe the skull of the male leopard must be 35cm or larger in order to be exported (Lindsey and Chikerema-Mandisodza, 2012). In Namibia the minimum trophy size is based upon the minimum skull measurement of 32cm (Nature Conservation Ordinance 4 of 1975 section 84 conditions of predator trophy hunting permits 114C. (5a)). It was determined that the number of trophy hunted leopards with a total skull size under the required 32cm declined by 6.4% to 0.7% after the implementation of the TAG system. The average skull length found in Namibia was 37.72cm which is comparable to that of Mozambique (av. 40.00cm 2013-17, MITADER, 2018). Trophy hunting has the potential to reduce the genetic diversity as the fittest or largest individuals are targeted (Balme *et al.*, 2010; Ripple *et al.*, 2016) as well as causing the inheritance of undesirable traits (Coltman *et al.*, 2003) as such, it is important to include genetic monitor as part of the long-term management strategy for the Namibian leopard.

Currently the TAG system does not limit the age at which a male leopard may be trophy hunted. However, there are calls for this to be taken into consideration in the future to ensure that male leopards have the opportunity to reproduce and to help ensure that female leopards are not misidentified as sub-adult males (Balme *et al.*, 2012). The minimum age at which male leopards have reproduced is commonly accepted to be seven years of age while physical distinctiveness by sex is apparent from the age of four (Balme *et al.*, 2012). The ages recorded by MET for male leopard removal by trophy hunting since 2013 ranges from 2.5 to 20 years of age. The proportion of ages were; 2 - 3.5 was 1.52%, 4 - 6.5 was 26.7% and 7+ was 71.8%. In areas with low density taking into account the population dynamics, if known, is an important part of the long-term monitoring of the population. The recent camera trap survey in the Nyae Nyae Conservancy determined that the oldest leopards are between 3 and 4 years which is still considered to be a sub-adult (Hanssen and Singwangwa, 2019 *unpublished*). Under the current system removal of these ages is acceptable but for this specific area it would be highly detrimental to the long-term viability of the population. Zimbabwe are currently piloting a new awards points system that uses the age of the trophy hunted leopard to adjust an area's quota (ZPWMA, 2018). Hunting young leopard results in a reduced quota for the area whereas hunting an older leopard or no leopard at all results in the quota in the area either remaining the same or potentially increasing (ZPWMA, 2018). The ZPWMA will be monitoring the results of the new system through their hunt return forms, trophy photographs and other information they deem necessary. The results of this pilot project could help inform Namibian decision makers as to how, if required, one type of age-based hunting system could be implemented. The Stein *et al.*, (2011b) report also outlined an age-based leopard hunting points system that is similar to the one currently being piloted in Zimbabwe (ZPWMA, 2018).

Another area of interest pertaining to the TAG system was the period of the day permitted for trophy hunting to take place. The system allows hunting to occur from 30 minutes prior to sunrise until 30 minutes after sunset. The study utilised the time events captured in the camera trapping survey to determine the activity patterns for leopard and particularly for males. As anticipated the majority of time events were captured outside of these hours, although there was a marginal difference between the number of time events captured prior to sunrise and post sunset. The 30 minutes after sunset saw 1.6% more activity than pre-sunrise, however, when sex was taken into account, males were 5% more active after sunset than pre-sunrise. Another factor which was expected to have an effect on leopard activity patterns was moon phase (Prugh

and Golden, 2014). The study found that moon phase had almost no effect on general leopard activity or that of males specifically. Activity was also evenly spread throughout the pre and post full moon periods.

The majority of successful leopard trophy hunts took place in four regions, Erongo, Khomas, Kunene and Otjozondjupa. Again, this reflects the relatively high density of leopard across these regions. This is also where most unsuccessful hunts took place which is to be expected. There was a limited amount of information as to why a trophy hunt was unsuccessful however, when reasons were given, they fell into four groups 1) environmental conditions (rain and wind), 2) missed opportunities by a client, 3) no suitable males at the sites and, 4) no male leopards were at the sites during legal hunting hours. Understanding why a trophy hunt failed is just as important as knowing that it was successful, therefore, more information from the hunting community is required as it will assist in the decision making process in order to manage the leopard population appropriately.

5.2.1. Quota Allocations

MET allocates leopard trophy hunting quotas based on the size of land, 2,500ha and above, and any relevant scientific information which includes but not limited to;

- Leopard population estimate
- Leopard density
 - Farm location and district
- Habitat suitability
 - Farm location and district
- Trophy size and trends
 - Previous trophy quality of applicant/farm (if applicable)
- Hunting success rate
- Problem animal incidents
 - Farm location and district
- Amount of detail provided in application

The study aims to increase the transparency of the application process for trophy hunters by providing the information above. The study has also provided updated and enhanced detail on the population estimate and density, trophy quality, hunting success rates, problem animal removal and habitat suitability which will allow decision makers to make more informed and accurate TAG allocations in the future. For instance, the known presence of leopard has expanded beyond the boundaries identified in the 2011 Stein *et al.*, study, particularly in the east and south-east of the country. Additionally, finer detail is provided on the geographical density variation of the leopard population and therefore farms can be sorted into more specific quota zones than was previously possible. Crucially, the study produced a habitat suitability map, specifically for leopard, to be used as a tool to assist with the application process and as such the sustainable management of the population.

As the study has identified, the core leopard population of Namibia, trophy hunting of leopard and problem leopard removal predominantly occurs in the freehold farmland and the communal conservancies of Kunene. This indicates that conservation effort for leopard should be focused on these areas with long-term monitoring prioritised to ensure the viability of the national leopard population. This also highlights the role Namibia's freehold farms and communal conservancies in Kunene have to act as the custodians of the leopard population outside of national parks.

The current annual quota for leopard trophy hunting is set at 250 males and is based on the 2011 population estimate (Stein *et al.*, 2011b). This study has concluded that the national leopard population estimation is 17% lower than the previous estimation and using these figures alone it is reasonable to suggest that the quota should be reduced to 1.8% of the new estimate. However, using the population estimate alone to set the quota does not take into account other factors which may have a greater impact on the leopard population. The highest number of utilised TAGs recorded in a single year was 161 in 2017, which is 64.4% of the 250 total. This figure of 161 is comparable to other countries highest offtake, Mozambique highest annual offtake was 117 (Annex 1 CITES, 2018b) with Zimbabwe recording 186 (Annex 6 CITES, 2018b). Since the introduction of the current TAG system in Namibia in 2011, the quota of 250 has never been fully utilised. This finding is not unusual for leopard trophy hunting and reflects the situation in other African countries. Between 2011 and 2017 in

Mozambique 40 - 50% of the quota was utilised (Annex 1 CITES, 2018b), Tanzania only utilised 32.4% (Annex 4 CITES, 2018b) similarly, Zimbabwe utilised 33.1% (ZPWMA, 2018). In South Africa, between 2005 and 2016 hunters utilised just under half of their 150 quota (Annex 3 CITES, 2018b).

In Namibia over the past three years an average of 413 hunting permits have been allocated which has led to an average of 155 leopard trophies being hunted which produces an average conversion rate of leopard permits to successful hunts of 27%. In comparison 882 leopard hunting permits have been issued in Zimbabwe which resulted in 261 successful leopard trophy hunts, a conversion of 30%. Even though this is below the country's 300 quota allocation it is felt that these offtake figures are unsustainable (du Preez *et al.*, 2014). The TAG utilisation over the past five years in Namibia has shown a consistent trend of underutilisation of approximately a third; 59.6%, 60%, 64%, 64.4% and 57.2% with an average of 61%. Within just the communal conservancies the quota utilisation is even lower at 50%. These figures are well within the sustainable trophy hunting off-take limits of both the current and revised population estimate. Based on the average utilisation rate a reduction in quota from 250 to 211 would only result in a reduction of 23 leopards being removed annually while denying 39 landowners the opportunity to benefit from sustainable usage income.

When current data is available for specific areas and/or regions this must be taken into account when determining the TAG allocations. This adaptive management strategy has already been put into practice in some areas by MET. For example, the leopard density survey results in Mahangu Core Area in 2017 determined that the leopard population was too low for there to be off-take within the population, as such the leopard quota was removed. In 2014/15 the leopard density in the Mudumu landscape was found to be 0.4 leopards/100km² however, since then the density has dropped to 0.25 leopards/100km² (Hanssen and Singwangwa, 2019 *unpublished*). The leopard densities in the north-east of Namibia have been consistently studied and more recently leopard density studies have taken place in Nyae Nyae Conservancy and Mangetti National Park. The data for these two studies is still in the analysis phase and therefore results are absent from this study. The results of these surveys and this study should be used to guide MET's upcoming review of the trophy hunting quotas in national parks and communal conservancies at the end of 2019. This highlights the importance of an adaptive management

strategy for quota setting that constantly evolves as new information is captured from research projects.

From 2005 to 2009, the main method for hunting leopard was with dogs which accounted for the spike in the number of successful leopards' hunts in 2009. Leopard hunting with dogs was not limited to Namibia, both Zimbabwe and Zambia saw declines in their leopard population due to this hunting technique (Purchase and Mateke, 2008; Packer *et al.*, 2009). In 2010 hunting leopards with dogs was banned in Namibia which resulted in a substantial decline in successful hunts, leading to 2010 have the lowest number of successful hunts between 2001 and 2018. In contrast to the decline in the number of successful trophy hunts, 2010 saw the highest number of reported problem animal cases/removals to MET since record capture began in 2005. This highlights the need to ensure that farmers in Namibia can offset their losses through trophy hunting or the consequence will be an increase in problem animal removal. As this study has shown, the actual figures for both reported and unreported problem animal removal is very high and is therefore one of the major threats to the Namibian leopard population. If the economic incentives for farmers to co-exist with leopard are limited, as they were in 2010, the rise in removals would become highly detrimental to the leopard population.

When the sex of the leopard was recorded as part of the MET problem leopard records, 72% were male and 28% were female. The questionnaire showed a similar pattern with 64% of males being removed compared to 28% of females. Females are a key reproductive unit (Daly *et al.*, 2005) and are more difficult to replace than adult males, as such removals of females through either trophy hunting or as problem animals can directly impact the population viability. The geographical location of male removals can be controlled through the distribution of the quota allocations across Namibia. However, problem animal removal of both sexes happens indiscriminately across the freehold farms and the communal conservancies. When a territorial male is removed from the territory by either trophy hunting or illegal activities, it creates a 'vacuum' which is immediately occupied by the dispersal males in the area (Davidson *et al.*, 2011). As a male loses territory a female may then be sharing her territory with two males. This can result in infanticide and an unnatural ratio of males to females causing females to mate with the new neighbouring dispersal male. Infanticide can also lead to females not

raising young due to the incursion of new males (Balme *et al.*, 2009; Balme, 2010). All of these interactions will have a significant impact on the long-term viability of the leopard population.

Swanepoel *et al.*, (2014; 2015) identified that retaliatory killing of leopards could have significant negative impacts on leopard population demographics, whereas the demographic consequences of hunting is likely to be less impactful. Following on, it was determined that mitigating conflict rather than reducing leopard quotas may have more influence over maintaining a viable leopard population (Swanepoel *et al.*, 2014). However, poorly managed trophy hunting can have a similar impact on the leopard population as problem animal removal does (Balme *et al.*, 2010). Therefore, it is essential that trophy hunters provide all the required information, in accordance with the permit regulations, to enable government in this case MET, to make informed management decisions. This study found sizeable variations in the amount of information captured in the Schedule G trophy hunt record sheet for both successful and unsuccessful hunts. For example, since the implementation of the Schedule G record sheet in 2016 only 2.5% of hunters gave a response to the question ‘why was the hunt unsuccessful?’ However, the implementation of the Schedule G record sheet was a positive management decision as it has provided MET, this study and others to come, with a level of detail on leopard trophy hunting that has not been captured before in Namibia. As such this study is in full agreement with the IUCN (2018) statement that improving the management of trophy hunting must happen in tandem with reducing losses from other causes of mortality, particularly problem animal removal. Regular monitoring of populations is essential in order for changes in population dynamics to be determined rapidly and subsequently addressed through adaptive management strategies.

In conclusion, this study has shown the removal problem animals across Namibia is high, both in the freehold farms and communal conservancies, which will have a direct impact on the leopard populations social organisation and demographic patterns and consequently the species long-term survival. Problem animal removal has been shown to be the greatest pressure on the national leopard population. A reduction in the opportunity to undertake trophy hunting could have a detrimental impact as a result of landowners not having trophy hunting available to them as a way of mitigating loss and dealing with problem animals. However, landowners must recognise that if problem leopard removals continue to rise then the trophy hunting quota would

have to be adjusted to compensate for those removals in order to maintain a viable population. The study has also demonstrated that, landowners' tolerance towards leopards on their property was improved by the potential revenue which could be generated from trophy hunting and tourism. This was underlined by questionnaire respondents who indicated that if trophy hunting was banned, they would have no choice but to remove some or all leopards from their land. These findings echo other studies that have demonstrated the importance of the relationship between offsetting economic loss and tolerance levels of farmers towards leopards (Blame *et al.*, 2010; Chase-Grey, 2011; Di Minn *et al.*, 2016).

5.3. Human-Leopard Conflict

In a study on global human-wildlife conflict patterns, the leopard was the leading carnivore conflict species as it featured in the greatest number of human-carnivore conflict cases (Seoraj-Pillai and Pillay, 2016). This was due to a variety of reasons; firstly, the leopard exhibits an array of biological and behavioural traits such as; its opportunistic hunting behaviour, solitary living and varied diet which renders it a high-impact conflict species (Kissui, 2008). Conflict exists between carnivores and humans due to predation on livestock and/or valuable game species (Stein *et al.*, 2010; Menges and Melzheimer, 2014). This study found that the leopard was the second highest conflict species, after black-backed jackal, when looking at the total number of carnivore species removed from respondents' properties due to predation. Over the duration of the study, respondents reported removing 342 leopards compared to 196 leopards recorded by the Ministry of Environment and Tourism in the same timeframe and the 183 reported in 2008-10 (Stein *et al.*, 2011b). In the communal conservancies an average of 336 leopard conflict incidents were logged per year. In comparison, on average 155 leopards were trophy hunted annually (2016-2018), this contrast between the problem animal removal and trophy hunted leopard figures is not unusual (Swanepoel *et al.*, 2015). The problem animal removal results reflect the broad geographical spread that leopards have across Namibia, due to their adaptability across multi-use landscapes, and the fact that the majority of leopards reside in the freehold farms.

The vast geographical spread of leopards across Namibia means that there are resident leopard populations across all land use types. Neither protected area boundary fences or high game

fences, are acting as an impenetrable barrier for leopards. As has been documented, retaliatory killing of leopards occurs due to livestock predation in any area where the leopard is not within either a national park or private reserve i.e. the freehold farmland and communal conservancies. However, it is important to recognise that leopards are resident in those areas, they have not moved outside of a protected area boundary into the farmland which is the case for other species such as the lion (Trinkel *et al.*, 2017) and it is this movement that is the catalyst for the human-carnivore conflict. In Kenya's Tsavo Amboseli Ecosystem, an area that lies between two national parks, Amboseli and Tsavo West National Parks, 108 lions were killed over a five year period due to conflict from livestock predation which occurred once the lions had left the park boundaries (Frank *et al.*, 2006). The recorded leopard densities in the Namibian national parks are lower than that of the surrounding freehold farmland therefore the parks may not be acting as source (source-sink theory, Pulliam, 1988) for the leopard population as is the case in other countries (Balme *et al.*, 2010). However, the private reserves with their high densities and stable populations (Noack, 2016; de Woronin Britz *pers. comm.*, 2018) have the potential to act as source populations. The Omaruru camera trap survey area's southern point bordered onto the northern boundary of Erindi private game reserve. One collared male was photographed moving between two freehold farms next to Erinidi and was identified as one of their collared leopards from an ongoing study (de Woronin Britz *pers. comm.*, 2018). This shows that there is movement of leopards between Erindi and the adjacent freehold farms, as problem animal removal is higher outside of the reserve it is likely that leopards from the reserve will move into the territorial vacuums (Davidson *et al.*, 2011) that are created.

It is important to recognise that the leopard densities found across Namibia's national parks were substantially lower than those found in the freehold farmland. National parks aim to provide a sanctuary for wildlife and are an important management tool when looking to conserve a species nationally. Therefore, the expectation is that due to the protection the national parks affords a species, like the leopard, densities should be higher. However, in Namibia this is simply not the case as the majority of the leopard population is found outside of the national parks. Therefore, it is critical to recognise that the majority of the Namibian leopard population will be under significant anthropogenic pressures which in turn will directly impact the populations long-term viability. For example, the study found that regions which had substantially higher livestock than game density were where the incidence of leopard absence was most common. This was the case for the Hardap and Karas regions which

predominantly stocked sheep and goats and had the second and third smallest game densities. In contrast, the regions with a higher game density over livestock had a higher incidence of leopard presence. For two specific regions, Omaheke and Oshikoto, their game and livestock densities were equal which led to a 2:1 presence to absence relationship. As such, it is essential to recognise that if the leopard population outside of the national parks were to become unviable the populations inside the national park may not be enough to re-establish the national population.

In South Africa, it was determined that human-wildlife conflict was influenced by four key factors; high elevation, mixed purpose farming, dense vegetation, and high perceived financial loss (Thorn *et al.*, 2013). The findings of this study also show a convergence between these four factors which determine the level of conflict. The habitat suitability model results showed that altitude, land cover and land ownership were the most influential environmental variables for leopard and as such these are the areas with the highest leopard densities.

Data collected through a questionnaire must achieve a broad geographical spread to ensure that the data is representative of the study area and in turn that the results are not skewed. Based upon the respondent's location it was determined that this study did achieve the geographical spread required to infer results based upon location. It was also critical to ensure respondent's anonymity as it assisted with the discussion of sensitive topics such as carnivore control methods and unreported removals (St John *et al.*, 2012). This proved an effective method as the respondents provided information on problem animal removal, removal methods and their subsequent reporting to MET or lack thereof. The study recognises that even with anonymity not all respondents will be completely honest, however, the study still feels that the results are representative of the wider community.

Between October 2016 to December 2018 MET had a record of approximately 196 problem leopards being removed by 190 individuals across Namibia. In the same period a sample size of 157 questionnaire respondents claimed to have removed 342 problem leopards from freehold farms. In 2017 a total of 650 problem leopard incidents were recorded from freehold farms (152) and communal conservancies (498), which would rise to 846 if it is assumed there is no

overlap between the study's respondents and the MET reporters. The results showed that the majority of removals were males which means the pressure of problem animal removals on the population will be skewed. This could have a direct impact on the quantity and quality of male trophies for hunting. However, a genetics study in Botswana determined that more females than presumed were removed in response to livestock predation due to the misidentification of over half the leopard's sexed by the farmers (Kerth *et al.*, 2013). Therefore, the impact of problem animal removal could be greater than anticipated due to its negative affect on the population dynamics as female removal is potentially higher than the reporting figures have captured.

Both the questionnaire respondents and the MET records consistently showed that cage trapping and shooting were the primary methods of removal. These findings are in line with Williams *et al.*, (2017)'s study which also found shooting to be a primary tool for removal. It was encouraging to discover that the use of other methods such as poisoning, gin traps and snares was minimal which goes against the results of the Williams *et al.*, (2017)'s study.

Current legislation requires all problem leopard removals from freehold farms to have a permit issued by MET, either prior to the removal or retrospectively. Questionnaire data revealed that half of the respondents who had removed problem leopards did not apply for the permit. Since 2011 the reporting rate of problem leopard removal by freehold farmers to MET has declined by 5% to just 45%. The respondents who applied for a permit accounted for the removal of 60 leopards per year while those who did not apply accounted for 92 leopards per year. A small proportion of respondents who did remove problem leopards did not state if they had, or had not, applied for a permit and accounted for 6 leopards removed per year. The study's findings are lower than those captured in the Santangeli *et al.*, (2016) poison study which found that 67% of the respondents removed any carnivore species without the necessary permit.

In 2017, given the questionnaire respondents data on problem animal removal and permit application, it reasonable to assume that 25% of the MET problem animal records were from the questionnaire respondents. As the MET data should capture all problem animal removals on freehold farms it again highlights the issue of lack of reporting. In comparison, the

communal conservancies reported 498 problem leopard incidents in 2017 and on average they report annual incidences of 336 problem leopard. These figures suggest that the anticipated removal rate on freehold farms should be significantly higher than is currently captured in MET records. Both NACSO (2018) and the N/a'an ku sê Foundation (2018 *unpublished*) have seen a rise in the number of reported leopard conflict cases since 2008, including a surge in 2015. The MET problem leopard records do not reflect this pattern of increase. Clearly there is a disconnect between the picture generated by the reports provided to, and recorded by, MET and the actual scale of problem animal removal taking place. By way of comparison, interviewed South African landowners felt that they lack control over the official process of dealing with livestock losses and that this frequently drove them to retaliatory killing to sort out the problem as quickly as possible (Grey *et al.*, 2017). In Zimbabwe the level of reporting livestock predation was firstly based upon people's attitudes towards different predators and secondly on the perceptions of whether the report would be acted upon by the management authorities (Loveridge *et al.*, 2017). Anecdotal information gathered during the study confirms that the Namibian landowners' attitudes reflect those documented in South Africa which has resulted in the disconnect between the problem leopard removal figures collected by this study, MET and other organisations. The study's results are in agreement that problem leopard removal and the subsequent lack of reporting to MET is one of the greatest threats to the Namibian leopard population.

It is important to note that live leopard captures and translocations due to human-leopard conflict does occur in Namibia but on an ad hoc basis when a conflict situation arises. Under these circumstances it makes it difficult to track the artificial movement of leopards. During the camera trap surveys, the study was able to confirm that was no artificial movement of leopards either in or out of the survey areas. At a national level information on leopard movements undertaken by either MET or non-governmental organisations were not captured as part of this study. However, in the future this information should be captured from MET's translocation permits as it will aid in our understanding of the frequency at which translocations occur, their effectiveness and, geographical reach of one of Namibia's national conflict mitigation strategies.

Prey availability is also linked to human-wildlife conflict, the greater the prey base the less likely carnivores, including leopard, are to predate on livestock (Woodroffe *et al.*, 2005; Khorozyan *et al.*, 2015). Lack of sufficient prey has been proposed as the major driver of big cat depredation, including leopard, on livestock (Khorozyan *et al.*, 2015). It has been determined that carnivores, including leopard, predation rates on cattle significantly increase when prey biomass is less than 812.41 kg/km² and to kill sheep and goats when prey biomass is below 544.57 kg/km² (Khorozyan *et al.*, 2015). When prey biomass drops below 540 kg/km² cattle, sheep and goats will all be intensively predated upon to optimize energy intake (Khorozyan *et al.*, 2015). Overall Karas had the greatest number of livestock lost to leopard per respondent, followed by Hardap. These findings follow the relationship between prey biomass and livestock predation as Karas and Hardap are the two regions with one of the lowest game densities, the highest livestock density and consequently the highest predation rates.

In Namibia, carnivore presence was tolerated in areas where income from wildlife was higher, income from livestock was lower, and financial losses from livestock depredation were lower (Lindsey *et al.*, 2013). The predominant revenue generating activities of respondents were 1) cattle farming 2) sheep and goat farming 3) hunting activities 4) tourism. Attitudinal data derived from the questionnaire underlined the importance of a sustainable use policy to promote favourable attitudes towards having leopard present on respondents' properties. The questionnaire identified that farmers tolerance to the presence of leopard on their properties was in part financially motivated. If farmers felt that there was an opportunity to undertake trophy hunting to offset the economic impact of both livestock and game losses then they would tolerate the presence of leopard as is also the case in South Africa (Brackowski *et al.*, 2015). Respondents from Kavango East and Zambezi indicated that 100% of their income was generated through hunting activities and subsequently had the most favourable attitude toward leopard presence. However, the majority of farmers would not tolerate leopards due to the perceived risk of livestock and game loss which equated to a loss in income generation. Karas and Oshikoto respondents declared the lowest level of income generation from hunting and the highest revenue from livestock farming, and as could be expected, had the least amount of tolerance for leopards. Respondents that had an equal mix of income generation from cattle farming and hunting activities, such as those from Erongo and Otjozondjupa, returned a neutral attitudinal response towards leopard on their properties.

Tourism can also provide an economically viable, non-consumptive use of leopards (Lindsey *et al.*, 2007; Funston *et al.*, 2013; Mossaz *et al.*, 2015). Since most leopards in Namibia live outside of national parks the realisation of this economic value is critical to ensure the long-term conservation of the species. For example, land use in the broader Namib area is shifting away from farming and moving towards tourism which has led to a decline in human-leopard conflict (Tindall *pers. comm.*, 2018). Tourism is an important industry for Namibia, in 2017 it contributed to 13.8% of the national gross domestic product in part due to its ability to offer the ‘Big Five’ safaris (Goodwin and Leader-Williams, 2000; Williams *et al.*, 2000) which includes the leopard. In a survey out of all African wildlife the leopard came out as one of the highest ranked in terms of key species that tourist wanted to see (Di Minin *et al.*, 2012). For the questionnaire respondent’s tourism was the fourth highest revenue generating activity and it was also one of the specific reasons that landowners wanted to have leopard present on their property.

In 2017 the estimated Namibia agriculture production value of the cattle (N\$3,403,613,974), sheep (N\$ 705,464,949) and goat (N\$ 130,665,020) industries was N\$ 4,239,743,943 which equated to 56% of the total national production value generated that year (NAU-NLU, 2018 *unpublished*). These figures not only outline the importance of the livestock industry to individual farmers but the contribution this industry makes to the national economy as well. In the Blouberg Mountains Range, South Africa leopards were responsible for 89% of the reported game attacks and 60% of the reported cattle attacks (Constant *et al.*, 2015). The average annual loss of cattle income per household due to predation by leopards was ZAR 12,183 on commercial farms and ZAR 10,500 on communal land which, is the equivalent of 2.6 % and 58.3% of the estimated annual income for commercial and communal farmers respectively (Constant *et al.*, 2015). Due to these economic losses landowners have been shown to undertake a range of predator control methods to protect their livestock and game from perceived predation events (Lindsey *et al.*, 2005; Blaum *et al.*, 2009). In the Soutpansberg mountains, South Africa between 2008 and 2016 the leopard density declined by 66%, predominately due to human-leopard conflict in the form of retaliation to perceived livestock predation and bushmeat poaching (Williams *et al.*, 2017). The problem animal information collected in this study shows the link between actual loss, perceived loss and removal rates. A small proportion of landowners removed leopards due to a perceived risk of loss rather than actual loss. Actual financial loss has also been shown to be a determinate if lethal control is

undertaken in retaliation for livestock killings (Sillero-Zubiri *et al.*, 2007; Thorn *et al.*, 2012; 2013). However, half of the landowners in this study who experienced loss from any of the eight named carnivore species did not remove a problem animal as a consequence of the loss. Specifically, for leopard this was not the situation, all cases of livestock/game loss resulted in a minimum of one problem leopard being removed. MET data also demonstrated that problem animal removal was undertaken not only due to actual loss but perceived threat to game and livestock and risk to human safety.

The questionnaire portion of the study re-emphasised the relationship between livestock loss and problem animal removal. Leopards have been found to predate on cattle when they are the apex carnivore in the landscape but shift to sheep and goats when co-existing with other dominant carnivore such as lions (Khorozyan *et al.*, 2015). Over the duration of the study respondents stated that leopards preyed upon 2,294 cattle, followed by 1,151 game, 307 sheep, 222 goats and 3 horses equating to a total loss of 3,977 individuals. The high levels of cattle predation reflect the apex carnivore status of the leopard in the freehold farmland. It has been found that patterns of predation by the leopard reflects its wild prey preference for specific body sizes (Loveridge *et al.*, 2017). In Zimbabwe leopards were found to primarily predate on goats (all ages) and cattle calves (Loveridge *et al.*, 2017), in the Soutpansberg Mountains calves made up the majority of the reported depredation by leopards (Chase-Grey, 2011) which reflects the findings of this study as the majority of the cattle lost were described as calves.

Proportionally, Khomas, Erongo and Otjozondjupa had the highest percentage of livestock predation in relation to total livestock numbers across both the freehold farmland and communal conservancies. Livestock across the communal conservancies are an important source of protein, income, savings and social standing, as such, loss through predation can have a significant impact on these communities (Megaze *et al.*, 2017). The data collected from the questionnaire agrees with the findings of NACSO (2018) and the N/a'an ku sê Foundation (2018 *unpublished*) which highlighted Otjozondjupa and Khomas as conflict hotspots. Given that this study found these regions to hold significant leopard numbers this is to be expected. These regional results were also reflected in the two camera trap survey areas. Both areas, Auas Mountains and Omaruru, had similar game densities which for both areas were over the minimum threshold for prey biomass but the Auas Mountains had a higher livestock density.

This resulted in the removal of 11 leopards over a one year period in the Auas Mountains as a direct consequence of conflict and livestock predation compared to two in the Omaruru area in response to loss of game. As low prey biomass was not a contributing factor the livestock predation may have been a result of the opportunistic nature of hunting leopards (Hayward *et al.*, 2006). One farm in Omaruru accounted for a substantial proportion of the livestock density for the whole study area. The level of livestock loss on this farm was minimal which they attributed to their utilisation of multiple husbandry techniques, this is in line with the findings of Balme *et al.*, (2009) and Stein *et al.*, (2010)'s studies.

In conclusion, if the rate of removal suggested by the questionnaire is indicative of removals across freehold farms in Namibia in general then the true figure for leopards removed as problem animals would be far greater. In addition, when the problem animal removals figures from the freehold farmland and the communal conservancies are combined the scale of the situation is clear and unequivocally represents the most significant pressure on the Namibian leopard population.

6. Recommendations

6.1. Landscape Approach to Trophy Hunting Including Quotas

As discussed in section 5.2 and 5.3, the relationship between trophy hunting and problem animal removal means that the lowering of quotas may in fact have the opposite effect to that which is intended. It would make a minimal difference to the number of leopards removed by trophy hunting directly, while decreasing the tolerance for leopard presence across a much greater proportion of the population resulting in a disproportional increase in problem leopard removal. In terms of proactive population management removals from problem animal cases are unreported, uncontrolled and indiscriminate of age, sex and population density, whereas, trophy hunting is regulated and limited to areas which has a leopard population capable of sustaining controlled off-take.

In addition, this report has highlighted that the number of trophy hunted leopard is not equal to the quota limit and year on year figures demonstrate an average annual TAG utilisation rate of 61%. Taking these factors into account it should be possible to retain the existing quota on the condition that utilisation rates do not exceed 84%, in which case reassessment would be required.

Additional to this would be a policy of site-specific revision based on changes to local populations captured in ongoing leopard density monitoring. Assigning quotas based upon the narrower density intervals described below (Table 6.1) allows for a more considered approach to permit allocation in areas which straddle the existing high/medium/low boundaries (Stein *et al.*, 2011b) which mitigates the potential for over utilisation.

Table 6.1. The five quota zones defined by leopard density intervals and their associated TAG quota.

Quota Zones	Density intervals (leopards per 100km²)	Number of leopards	Root mean square error	Area (km²)	Percentage of area (%)	Quota per interval
1	0-1	1600	1963.5	273752	33	34
2	1-1.5	1770	1026.5	142482	17.2	38
3	1.5-2	3955	1623.6	227253	27.4	84
4	2-2.5	2908	947.2	133144	16	62
5	2.5-3+	1500	388.2	54764	6.4	32
	Total	11,733	5,949	831,395	100	250

Effective conservation management is of paramount importance to wide-ranging carnivores living in human-dominated landscapes outside of protected areas (Muntiferi *et al.*, 2006), especially when existing at low densities as it makes them more susceptible to extinction caused by demographic and environmental stochasticity (Karanth and Chellam, 2009; Pettorelli *et al.*, 2009). It has been found that in the event of widespread human-caused declines in cougar (*Puma concolor*) populations will reside in low-quality habitats on the range margins which makes them susceptible to decline (Stoner *et al.*, 2013). Leopards extensive distribution in combination with their wide-ranging movements mean that management of leopards needs to occur at large spatial scales (Pitman *et al.*, 2015). Therefore, it is important that a meta-population management approach to leopard is taken. In KwaZulu-Natal, South Africa, leopard hunting zones adjacent to protected areas have been established, each with a population considered extensive and robust enough to sustain hunting (Balme *et al.*, 2010). The aim was

to increase the geographical area over which trophy hunting could take place in order to spread the impact of the off-take in conjunction with the protected area acting as a source for the hunting zone (sink) (Pulliam, 1988).

In Namibia a landscape scale approach to resource management has already been instigated through the establishment of communal conservancies. Leopard trophy hunting quotas are assigned to individual communal conservancies, the majority of which cover large geographical areas and offer increased connectivity due to their proximity to other conservancies. This conservancy model was mirrored in the freehold farmland and saw the establishment of 21 freehold conservancies across Namibia between 1991 and 2018. This study has shown that a substantial proportion of the Namibian leopard population resides in the freehold farmland and more specifically in the central and north-west regions. As such, the importance of managing the leopard population at a landscape scale across the freehold farmland needs to be recognised. The freehold conservancies highlighted how this can be achieved as they were monitoring and co-managing their resources which included trophy hunting. The leopard home ranges in both high- and low-density areas cover multiple freehold farms and, as 99% of the questionnaire respondents attested to, leopards can move both in and out of their properties' perimeter fence. Over the course of this study the freehold conservancy system has undergone several changes the most important being the removal of the granting of hunting permits at a landscape scale. Landowners must now apply individually by farm, effectively moving away from a landscape level management approach to a single unit (one farm). For the long-term management of the leopard population this would appear to be a backwards step. The idea of allocating Namibia's annual leopard quota through a landscape approach is not a new one, the 2011 report advocated that the national quota should be divided up into farming units (Stein *et al.*, 2011b). The report went on to suggest that the farming units could be either freehold conservancies or farmer associations through which monitoring of the leopard population could occur (Stein *et al.*, 2011b).

A landscape approach to leopard trophy hunting could be created through the leopard management zones across the freehold farms. The freehold conservancies have demonstrated that it is possible to establish landscape management zones of mixed farm types and the study highly recommends that these zones are re-established as part of a stratified monitoring system

for Namibia's leopards as called for by the IUCN SSC Cat Specialist Group in 2018 (IUCN, 2018). These management zones would be responsible for monitoring their natural resources, including leopard which would include; population density and structure, environmental variables and problem animal removals. As these management zones would be spread across the known areas of leopard presence in Namibia they have the potential to acquire a high volume of information on the local leopard population which would, in turn, be fed back to MET. By acquiring localised data sets MET would be better informed on the status of the local leopard population, thereby enabling improved decision making on trophy hunting quota distribution. Problem animal removal is one of the major threats to leopard in Namibia so it is important to find solutions to this issue. As such it is important that landowners are aware that problem animal removal data is considered as part of the MET trophy hunting permit application (Section 5.2.1.) decision making process. This would help to encourage landowners to report all their problem animal removals, compared to less than half as is currently reported. The underreporting described in this study is cause for real concern and needs to be thoroughly addressed. Therefore, not only is encouraging landowners to report removals critically important but it must occur in tandem with MET employing more rigorous monitoring processes of problem animal removals in order to better manage the national population. MET must incorporate the collection of problem animal removal data into their existing farm visits to conduct fence checks and/or game counts. MET also regularly attends leopard trophy hunts as part of their law enforcement activities, problem animal data could be collected off the landowner at that time. In addition, problem animal removal numbers should be specifically requested on the leopard trophy hunting application form. By acquiring problem animal removal data from applicants MET could increase their number of records. It should be recognised however, that collecting problem animal removal data from multiple sources could lead to duplication of records which would need to be addressed in MET's data management process.

Another possible solution to human-leopard conflict could be taking a percentage of the management zone's profits generated from leopard trophy hunting and distributing the funds to landowners within the same zone who have incurred livestock losses due to leopard predation. This would minimise problem leopard removals in the zones as tolerance levels to livestock predation would be increased by the financial remuneration. The importance of offsetting economic loss for farmers was also recognised in the 2011 report which suggested

that 10% of the profits generated from leopard trophy hunting should be used to compensate farmers for the loss of livestock (Stein *et al.*, 2011b). This study advocated that the compensation should not only be applied in the communal conservancies but also in the freehold farmland through the re-establishment of management zones. In one freehold conservancy the management committee re-distributed a proportion of their elephant trophy hunting profits to help their members re-build fences and water points that had been damaged by the elephants. This in turn increased the farmer's tolerance to having elephant present on their land (Veldsman *pers comm.*, 2018). This example demonstrates that utilising trophy hunting profits from a specific species to offset economic losses and thereby increasing tolerance is viable within a management zone and could be replicated.

6.2. Further Camera Trap Survey Areas

As the study has outlined, the density, territory size and distribution of leopard varies greatly across Namibia due to variations in rainfall, habitat, persecution levels and prey availability. The continued understanding of the impact these variables have on a national scale is critically important to understanding the leopard in Namibia as a whole. Surveying across the freehold farmland in particular needs to be made a priority as data is currently limited for this important leopard area.

The study strongly recommends the continuation of the camera trap surveys with a focus on the freehold farms and the Kunene communal conservancies alongside the re-surveying of the freehold farmland bordering the Namib-Naukluft/Tsauu//Khaeb National Parks. Leopard presence records have now been established in the east and south-east of the country and therefore this area warrants further investigation into leopard density with a suggested focus in:

- Karasberg Mountains
- Weissrand Plateau
- The Namibian border linked to the Kgalagadi Transfrontier Park

These areas will complement the already ongoing leopard research (density, home ranges, population structure) in the Sandfontein Nature Reserve and Oana Nature Reserve. The landowners on the south-east Namibian and Botswana border have noted the transboundary movement of leopard and other carnivores onto their properties. Further research into the relationship between the Transfrontier Park and freehold farms is needed in order to understand leopard population dynamics in this area.

To the north the study suggests camera trap surveys to be undertaken in the Kaokoveld around Opuwo and further north to the Angolan border within the communal conservancies. These areas can be linked to carnivore research already being undertaken in Etosha Heights and just north of the Namibian border in Angola's Parque Nacional do Iona.

Further areas that require investigation are; 1) the southern portion of Omaheke below Gobabis as this is a data deficient area for leopards and 2) the area east of Grootfontein prior to the communal conservancies. This area requires further investigation due to a disparity in the reported leopard densities detected in two connected areas. Anecdotal evidence suggests a very high leopard density on a freehold farm adjacent to both the N#u-Jaqua Conservancy and Otjituu Conservancy. However, the leopard density in N#u-Jaqua and Nyae Nyae conservancies have been shown to be very low. 3) The Erongo Mountains where a 120,000ha freehold conservancy has been created by taking down all internal fences. This area contains highly suitable habitat for leopards which in conjunction with its land use status, makes for an interesting case study.

The leopard presence data was not only collected through the national questionnaire and camera traps surveys but through a citizen science driven requests to the public for their personal camera trap photos. As the study has shown, hunters utilise remote camera traps in order to monitor their baited sites for leopard and so many hunters donated their photographs. In addition, camera trap photographs were also collected from gold mines, private reserves and farms. The willing participants provided large volumes of leopard data from across Namibia alongside attribute data about their land/farm including latitude and longitude. The information from the photographs were then deposited in the EIS which went onto contribute to the

updating of the Namibian leopard presence map. The study found that participants were eager to share their information and therefore, would recommend the continuation of the citizen science driven project in order to collect more leopard photographs and continue to increase the EIS's database of leopard presence across Namibia.

The study recommends, if applicable, the use of spatially explicit capture-recapture (SECR) models for the camera trap data analysis. One of the benefits of the analysis is that it produces a detailed output of the variation in density across the study area which enables a greater understanding of the detail as to how the leopards are spread across the study area. For example, the SPACECAP pixel density outputs for the Auas mountains (Figure 6.1) shows that the leopard density varied west to east, with the highest density in the west, decreasing in the central area and increasing again in the east.

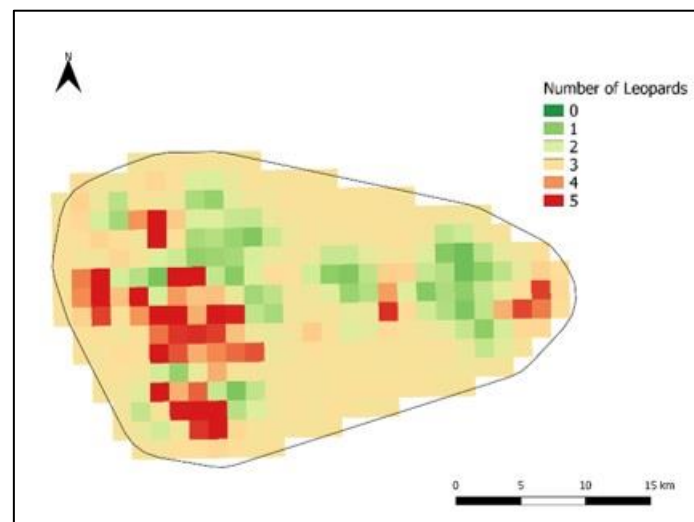


Figure 6.1. The SPACECAP Pixel density outputs for the Auas mountains camera trap survey area.

6.2.1. Complimentary Monitoring Methods

Genetic analyses have long provided important information on species biology to complement traditional taxonomic, demographic and behavioural data collection (Sunnucks, 2000). Building a Namibian leopard DNA database would provide multiple benefits for leopard conservation both nationally and internationally. DNA can provide useful data for answering questions on conservation and population genetics of wide ranging species such as the leopard. Advancements in non-invasive genetics have now made it possible to analyse large samples

from a variety of sources including hair. Furthermore, the ability to investigate patterns of genome-wide variation, even on the population level, using next-generation sequencing technologies will soon be feasible. This will enable a greater understanding into the genetic profiles of the Namibian leopard. DNA can also be used for DNA-based assignment tests, from which it is possible to infer geographic origins of DNA samples from seized illegal leopard products such as skins which in turn can help to identify trade routes and poaching hotspots for leopards at a subcontinent scale, as has been the case in India (Mondol *et al.*, 2015).

Obtaining multiple samples over time and a broad geographical spread can be difficult. However, trophy hunted leopards provide a consistent opportunity for the annual collection of multiple genetic samples from across Namibia. The collection of hair samples stored in paper envelopes under relatively dry climatic conditions will be sufficient to preserve the DNA. This simple data collection method allows for the capture and storage of a sample that will still have the quality required for the genetic analysis. Therefore, it is the recommendation of this study that the collection of a genetic sample, post hunt, becomes a permit requirement. Having worked with MET on a potential protocol it is the understanding of this study that this will be applicable from the start of the 2019 hunting season. This will enable MET to develop a genetic database for the Namibian leopard which can act as a central repository for leopard DNA. Having an understanding of the genetic makeup of the leopard population will be another valuable tool for decision makers in relation to long-term management and sustainable use policies. This information can then be linked with the wider Southern Africa leopard genetics programme which is already underway. The implementation of DNA collection as part of the trophy hunting permit requirements could be seen as phase 1. Phase 2 therefore, would be the inclusion of DNA collection as part of the problem animal removal permit requirements. This would substantially increase the sample size and geographical spread of leopard DNA collected on an annual basis.

6.3. Data Management

The study would recommend that for the collection of data regarding trophy hunting permits and problem removal reports, MET develops or brings in third party software which could be completed by landowners online. This would have several benefits;

- The standardisation of data collected
- Automated filing and storage of data
- Accessible and flexible format allowing for dynamic reporting
- Decreasing the labour and time required would likely result in an increase in reporting

To be able to manage a species on a landscape scale it is important to understand how the pressures such as problem animal removal and trophy hunting activities are spread geographically. However, due to inaccuracies and inconsistencies in the location data i.e. farm name, this study was only able to link 20% per dataset to the spatial land use data acquired from the Ministry of Land Reform, this was deemed to be insufficient for publication. However, the study does recommend that a spatial database is created. This will allow for the mapping of key variables such as; trophy success, trophy sizes, hunt effort and problem animal hotspots. In turn, this will provide a valuable tool which can be used as part of the trophy hunting permit application process, ongoing reporting as well as the long-term management of the leopard nationally.

The study found variation in the quality of information captured in the Schedule G trophy hunt record sheet for both successful and unsuccessful hunts. When an unsuccessful hunt occurred the record sheet requires a reason as to why success was not achieved. Since the implementation of the Schedule G sheet in 2016 only 2.5% of hunters gave a response to this question. This information can assist decision makers as it gives them an insight into the situation across the freehold farms on a yearly basis. In order to increase the reporting rate, changes have already been put into place by moving from a paper form to an electronic editable document that can be completed and emailed back to a dedicated MET trophy hunting email. The study further recommends that, as with the application process, the level of detail put into the Schedule G record sheet is taken into account in the following years application process. As discussed previously, the Schedule G record sheet is only returned to MET once the trophy is exported which can take months or years. This means that valuable information on the most recent hunting season is locked up in the process and decision makers can only act on historical data. As the Schedule G record sheet can now be emailed to MET, the study recommends setting a time limit by which this action should take place after a hunt. This will ensure that all the

information from both successful and unsuccessful trophy hunts is with MET by the end of the hunting season. This will ensure that all of the necessary data is available in order to inform management decisions for the following year.

7. References

- Abade, L., Macdonald, D. W. and Dickman, A. J., (2014) Using landscape and bioclimatic features to predict the distribution of lions, leopards and spotted hyaenas in Tanzania's Ruaha landscape. *PLoS One*, **9**(5): p.e96261.
- Araujo, M. B. and Pearson, R. G., (2005) Equilibrium of species' distributions with climate. *Ecography*, **28**: 693-695.
- Arjunan, M., Holmes, C., Puyravaud, J-P. and Priya, D. P., (2006) Do developmental initiatives influence local attitudes toward conservation? A case study from the Kalakad–Mundanthurai Tiger Reserve, India. *Journal of Environmental Management*, **79**(2) 188-197.
- Bailey, T. N., (1993) The African leopard: ecology and behavior of a solitary felid, New York: Columbia University Press.
- Bailey, T. N., (2005) The African Leopard: Ecology and Behaviour of a Solitary Felid, 2nd edn. Caldwell, New Jersey: Blackburn Press.
- Balme, G. and Hunter, L., (2004) Mortality in a protected leopard population, Phinda Private Game Reserve, South Africa: a population in decline. *Ecological Journal*, **6**: 1-6.
- Balme, G., Hunter, L. and Slotow, R., (2007) Feeding habitat selection by hunting leopards *Panthera pardus* in a woodland savanna: prey catchability versus abundance. *Animal Behaviour*, **74**(3): 589-598.
- Balme, G. A., Slotow, R. and Hunter, L. T., (2009) Impact of conservation interventions on the dynamics and persistence of a persecuted leopard (*Panthera pardus*) population. *Biological Conservation*, **142**(11): 2681-2690.

- Balme, G.A., Hunter, L.T., Goodman, P., Ferguson, H., Craigie, J. and Slotow, R., (2010) An adaptive management approach to trophy hunting of leopards *Panthera pardus*: a case study from KwaZulu-Natal, South Africa. *Biology and conservation of wild felids*, Oxford University Press, Oxford, pp.341-352.
- Balme, G. A., Hunter, L. and Brackowski, A. R., (2012) Applicability of age-based hunting regulations for African leopards. *PloS one*, **7**(4): p.e35209.
- Balme, G. A. and Hunter, L. T., (2013) Why leopards commit infanticide. *Animal behaviour*, **86**(4): 791-799.
- Balme, G.A., Lindsey, P.A., Swanepoel, L.H. and Hunter, L.T., (2014) Failure of research to address the rangewide conservation needs of large carnivores: leopards in South Africa as a case study. *Conservation Letters*, **7**(1): 3-11.
- Berg, R. L., McDonald, L. L. and Stickland, M. D., (1983) Distribution of mountain lions in Wyoming as determined by mail survey questionnaire. *Wildlife Society Bulletin*, **11**: 265-268.
- Bertram, B. C. B., (1979) Serengeti predators and their social systems. In Serengeti: dynamics of an ecosystem: 221–285. Sinclair, A.R.E. & Norton-Griffiths, M. (Eds). Chicago: University of Chicago Press.
- Bertram, B. C. B., (1999) Leopard. In The encyclopedia of mammals: 44–48. Macdonald, D. W. (Ed.). Oxford: Andromeda Oxford Limited.
- Blaum, N., Tietjen, B. and Rossmann, E., (2009) Impact of Livestock Husbandry on Small – and Medium – Sized Carnivores in Kalahari Savannah Rangelands. *The Journal of Wildlife Management*, **73**(1): 60-67.
- Bothma, J. D. P. and le Riche, E. A. N., (1984) Aspects of the ecology and the behaviour of the leopard *Panthera pardus* in the Kalahari desert. *Koedoe*, **84**: 259–279.
- Bothma, J. d. P., van Rooyen, N. and le Riche, E. A. N., (1997) Multivariate analysis of the hunting tactics of Kalahari leopards. *Koedoe*, **40**: 41–56.

- Braczkowski, A. R., Balme, G. A., Dickman, A., Macdonald, D. W., Fattebert, J., Dickerson, T., Johnson, P. and Hunter, L., (2015) Who bites the bullet first? the susceptibility of Leopards *Panthera Pardus* to trophy hunting. *PloS one*, **10**(4): p.e0123100.
- Braczkowski, A. R., Balme, G. A., Dickman, A., Fattebert, J., Johnson, P., Dickerson, T., Macdonald, D. W. and Hunter, L., (2016) Scent lure effect on camera-trap based leopard density estimates. *PLoS One*, **11**(4): p.e0151033.
- Braczkowski, A. R., O'Bryan, C. J., Stringer, M. J., Watson, J. E., Possingham, H. P. and Beyer, H. L., (2018) Leopards provide public health benefits in Mumbai, India. *Frontiers in Ecology and the Environment*, **16**(3): 176-182.
- Bradford, M. A. and Warren, R. J., (2014) Terrestrial biodiversity and climate change. *Global Environmental Change*, pp.355-361.
- Breiman, L., (2001) Random forests. *Machine learning*, **45**(1): 5-32.
- Burnham, K. P. and Overton, W. S., (1978) Estimation of the size of a closed population when capture probabilities vary among animals. *Biometrika*, **65**(3): 625-633.
- Burton, A. C., Sam, M. K., Balangtaa, C. and Brashares, J. S., (2012) Hierarchical Multi-Species Modeling of Carnivore Responses to Hunting, Habitat and Prey in a West African Protected Area. *PloS one*, **7**(5): e38007.
- CARACAL., (no date) Centre for Conservation of African Resources: Animals Communities and Land use, ODMP Wildlife Management Component. Leopard and cheetah baseline inventory in the Okavango Delta particularly in relation to areas of human wildlife conflict. Final Report. Kasane, Botswana.
- Carbone, C. and Gittleman, J. L., (2002) A common rule for the scaling of carnivore density. *Science*, **295**(5563): 2273-2276.
- Carbone, C., Mace, G. M., Roberts, S. C. and Macdonald, D. W., (1999) Energetic constraints on the diet of terrestrial carnivores. *Nature*, **402**: 286–288.

- Carbone, C., Christie, S., Conforti, K., Coulson, N., Franklin, N., Ginsberg, J. R., Griffiths, M., Holden, J., Kinnaird, M., Laidlaw, R., Lynam, A., MacDonald, D. W., Martyr, D., McDougal, C., Nath, L., O'Brien, T., Seidensticker, J., Smith, J. L. D., Tilson, R. and Wan Shahrudin, W. N., (2001) The use of photographic rates to estimate densities of tigers and other cryptic mammals. *Animal Conservation*, **4**(1): 75-79.
- Caro, T. M., Young, C. R., Cauldwell, A. E. and Brown, D. D. E., (2009) Animal breeding systems and big game hunting: models and applications. *Biological Conservation*, **142**(4): 909-929.
- Carroll, C. and Miquelle, D. G., (2006) Spatial viability analysis of Amur tiger *Panthera tigris altaica* in the Russian Far East: the role of protected areas and landscape matrix in population persistence. *Journal of Applied Ecology*, **43**(6): 1056-1068.
- Castro-Prieto, A., Wachter, B., Melzheimer, J., Thalwitzer, S. and Sommer, S., (2011) Diversity and evolutionary patterns of immune genes in free-ranging Namibian leopards (*Panthera pardus pardus*). *Journal of Heredity*, **102**(6): 653-665.
- Chapin, F. S., Zavaleta, E. S., Eviner, V. T., Naylor, R. L., Vitousek, P. M., Reynolds, H. L., Hooper, D. U., Lavorel, S., Osvaldo, E. S., Hobbie, S. E., Mack, M. C. and Díaz, S., (2000) Consequences of changing biodiversity. *Nature*, **405**(6783): 234-242.
- Chase-Grey, J., (2011) *Leopard population dynamics, trophy hunting and conservation in the Soutpansberg Mountains, South Africa* (Doctoral dissertation, Durham University).
- Chase-Grey, J. N. C., Kent, V. T. and Hill, R. A., (2013) Evidence of a high density population of harvested leopards in a montane environment. *PloS one*, **8**(12): p.e82832.
- CITES., (2002) Interpretation and implementation of the Convention. Regular and special reports. Appendix-I species subject to export quotas. Leopard. Amendment to the quota of the United Republic of Tanzania. Twelfth meeting of the Conference of the Parties, Santiago (Chile), 3–15 November 2002. CoP12 Doc. 23.1.2., 7 pp.

- CITES., (2018a) The CITES export quotas.
Available at: <https://www.cites.org/eng/resources/quotas/index.php>
(accessed on 01.03.2019).
- CITES., (2018b) Quotas for leopard hunting trophies. Interpretation and implementation matters. General compliance and enforcement. AC30 Doc. 15 Annex 1-4. Thirtieth meeting of the Animals Committee Geneva, Switzerland, 16-21 July.
- Clutton-Brock, T., (2016) Mammal societies, John Wiley & Sons.
- CMS and UNEP (2017) Proposal for the inclusion of the leopard (*Panthera pardus*) on Appendix II of the Convention. 12th Meeting of the Conference of the Parties, Manila, Philippines, 23–28 October 2017. UNEP/CMS/COP12 Doc. 25.31.4. 35 pp.
- Coe, M. J., Cumming, D. H. and Phillipson, J., (1976) Biomass and production of large African herbivores in relation to rainfall and primary production. *Oecologia*, **22**(4): 341-354.
- Coltman, D. W., Pilkington, J. G. and Pemberton, J. M., (2003) Fine-scale genetic structure in a free-living ungulate population. *Molecular Ecology*, **12**(3): 733-742.
- Constant, N. L., Bell, S. and Hill, R. A., (2015) The impacts, characterisation and management of human–leopard conflict in a multi-use land system in South Africa. *Biodiversity and Conservation*, **24**(12): 2967-2989.
- Creel, S. and Creel, N. M., (1996) Limitation of African wild dogs by competition with larger carnivores. *Conservation Biology*, **10**(2): 526-538.
- Creel, S., (2001) Four factors modifying the effect of competition on carnivore population dynamics as illustrated by African wild dogs, *Conservation Biology*, **15**: 271–274.
- Cutler, T. L. and Swann, D. E., (1999) Using remote photography in wildlife ecology: a review. *Wildlife Society*, **27**(3): 571–581.

- Daly, B., Power, J., Camacho, G., Traylor-Holzer, K., Barber, S., Catterall, S., Fletcher, P., Martins, Q., Martins, N., Owen, C. and Thal, T., (2005) Leopard (*Panthera pardus*) population and habitat viability assessment. In *Proceedings of a workshop of the Conservation Breeding Specialist Group World Conservation Union IUCN Species Survival Commission. Endangered Wildlife Trust, Johannesburg.*
- Davidson, Z., Valeix, M., Loveridge, A. J., Madzikanda, H. and Macdonald, D. W., (2011) Socio-spatial behaviour of an African lion population following perturbation by sport hunting. *Biological Conservation*, **144**(1): 114-121.
- Di Marco, M., Boitani, L., Mallon, D., Hoffmann, M., Iacucci, A., Meijaard, E., Visconti, P., Schipper, J. and Rondinini, C., (2014) A retrospective evaluation of the global decline of carnivores and ungulates. *Conservation Biology*, **28**(4): 1109-1118.
- Di Minin, E., Fraser, I., Slotow, R. and MacMillan, D. C., (2012) Understanding heterogeneous preference of tourists for big game species: implications for conservation and management. *Animal Conservation*, **16**(3): 249-258.
- Di Minin, E., Slotow, R., Hunter, L.T., Pouzols, F. M., Toivonen, T., Verburg, P. H., Leader-Williams, N., Petracca, L. and Moilanen, A., (2016) Global priorities for national carnivore conservation under land use change. *Scientific Reports*, **6**.
- du Preez, B. D., Loveridge, A. J. and Macdonald, D.W., (2014) To bait or not to bait: a comparison of camera-trapping methods for estimating leopard *Panthera pardus* density. *Biological Conservation*, **176**: 153-161.
- Drinkwater, B. L., (1965) A comparison of the direction-of-perception technique with the Likert method in the measurement of attitudes. *The Journal of Social Psychology*, **67**(2): 189-196.
- Edwards, S., Aschenborn, O., Gange, A.C. and Wiesel, I., (2015) Leopard density estimates from semi-desert commercial farmlands, south-west Namibia. *African Journal of Ecology*, **54**(1): 103-106.

- Edwards, S., Gange, A. C. and Wiesel, I., (2016) An oasis in the desert: the potential of water sources as camera trap sites in arid environments for surveying a carnivore guild. *Journal of Arid Environments*, **124**: 304-309.
- Edwards, S., Cooper, S., Uiseb, K., Hayward, M., Wachter, B. and Melzheimer, J., (2018) Making the most of by-catch data: Assessing the feasibility of utilising non-target camera trap data for occupancy modelling of a large felid. *African Journal of Ecology*, **56**(4): 885-894.
- Efford, M. G., (2004) Density estimation in live-trapping studies. *Oikos*, **106**: 598–610.
- Efford, M. G., Dawson, D. K., Robbins, C. S., (2004) Density: software for analyzing capture–recapture data from passive detector arrays. *Animal Biodiversity and Conservation*, **27**: 217–228.
- Efford, M. G., Borchers, D. L. and Byrom, A.E., (2009) Density estimation by spatially explicit capture–recapture: likelihood-based methods. In *Modeling demographic processes in marked populations* (pp. 255-269), Springer US.
- Efford, M. G., (2011) Estimation of population density by spatially explicit capture–recapture analysis of data from area searches. *Ecology*, **92**(12): 2202-2207.
- EIS., (2018) Atlasing in Namibia | Mammal Records. <http://www.the-eis.com/atlas/?q=atlas-of-carnivores>. Downloaded on 1st September 2018.
- EIS., (2019) Atlasing in Namibia | Mammal Records. <http://www.the-eis.com/atlas/?q=atlas-of-carnivores>. Downloaded on 1st March 2019.
- Estes, R. D., (1991) The behavior guide to African mammals, University of California Press, Berkeley.
- Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., Carpenter, S. R., Essington, T. E., Holt, R. D., Jackson, J. B. and Marquis, R. J., (2011) Trophic downgrading of planet Earth. *Science*, **333**(6040): 301-306.

- Foresman, K. R. and Pearson, D. E., (1998) Comparison of proposed survey procedures for detection of forest carnivores. *The Journal of Wildlife Management*, **62**(4): 1217-1226.
- Frank, L., MacLennan, S., Hazzah, L., Bonham, R. and Hill, T., (2006) Lion killing in the Amboseli-Tsavo ecosystem, 2001-2006, and its implications for Kenya's lion population. *Living with Lions, Nairobi, Kenya*, 9.
- Funston, P. J., Herrmann, E., Babupi, P., Kruiper, A., Kruiper, H., Jagers, H., Masule, K. and Kruiper, K., (2001) Spoor Frequency Estimates as a Method of Determining Lion and Other Large Mammal Densities in the Kgalgadi Transfrontier Park.
- Funston, P. J., Groom, R.J. and Lindsey, P. A., (2013) Insights into the management of large carnivores for profitable wildlife-based land uses in African savannas. *PloS One*, **8**(3): p.e59044.
- Funston, P. J., Hanssen, L. C. and Moeller, M. L., (2014) Large carnivore survey in Bwabwata National Park, (2014) Report submitted to the Namibian Ministry of Environment and Tourism.
- Gehrt, S. D., Brown, J. L. and C. Anchor, C., (2011) Is the urban coyote a misanthropic synanthrope? The case from Chicago. *Cites and Environment*, **4**: 1–23.
- Geldmann, J., Barnes, M., Coad, L., Craigie, I. D., Hockings, M. and Burgess, N. D., (2013) Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation*, **161**: 230-238.
- Goodwin, H. and Leader-Williams, N., (2000) Tourism and protected areas - distorting conservation towards charismatic megafauna? In *Priorities for the conservation of mammalian diversity: has the panda had its day?*: 257– 275. A. Entwistle & N. Dunstone (Eds). Cambridge: Cambridge University Press.
- Gopaldaswamy A. M., Royle J. A., Hines J. E., Singh P., Jathanna D., Jumar N. S., Karanth K. U., (2012) 'Program SPACECAP: software for estimating animal density using spatially explicit capture-recapture models'. *Methods in Ecology and Evolution*, **3**: 1067-1072.

- Grassman, L. I. J., (1999) Ecology and behavior of the Indochinese leopard in Kaeng Krachan National Park, Thailand. *The Natural History Bulletin of the Siam Society*, **47**: 77–93.
- Grey, J. N. C., Bell, S. and Hill, R. A., (2017) Leopard diets and landowner perceptions of human wildlife conflict in the Soutpansberg Mountains, South Africa. *Journal for Nature Conservation*, **37**: 56-65.
- Grimbeek, A. M., (1992) The ecology of the leopard (*Panthera pardus*) in the Waterberg, MSc Thesis, University of Pretoria, Pretoria.
- Groves, C. R., (1988) Distribution of the wolverine in Idaho as determined by mail questionnaire. *Northwest Science*, **62**(4): 181-185.
- Groves, C. R. and Peterson, C., (1992) Distribution and population trends of Idaho amphibians as determined by mail questionnaire. Unpublished report on file at the Idaho Department of Fish and Game, Nongame and Endangered Wildlife Program, Boise.
- Hanssen, L. and Stander, P., (2004) Namibia large carnivore atlas. Predator Conservation Trust, Windhoek.
- Hanssen, L., Alfred, B., Maseliso, M., Jaga, N., (2015). Status of African Wild dogs in Bwabwata National Park, Zambezi Region, Namibia. Kwando Carnivore Project. Report December 2015.
- Hanssen, L., Funston, P., Alfred, B., Alfred, S., (2017) Large Carnivore Survey Bwabwata National Park, Namibia, August 2017. Report submitted to the Namibian Ministry of Environment and Tourism.
- Hanssen, L. and Matambo Singwangwa, M., (2019 *unpublished data*) Mudumu National Park carnivore surveys, Kwando Carnivore Project report, Namibia.
- Hanssen, L., Alfred, B., and Sem Alfred, M., (2019 *unpublished data*) Bwabwata National Park carnivore surveys, Kwando Carnivore Project report, Namibia.

- Hayward, M. W. and Hayward, G. J., (2006) Activity patterns of reintroduced Lion *Panthera leo* and spotted hyaena *Crocuta crocuta* in the Addo Elephant National Park, South Africa. *African Journal of Ecology*, **45**(2): 135-141.
- Hayward, M. W. and Kerley, G. I., (2008) Prey preferences and dietary overlap amongst Africa's large predators. *South African Journal of Wildlife Research*, **38**(2): 93-108.
- Hayward, M. W., Henschel, P., O'Brien, J., Hofmeyr, M., Balme, G. and Kerley, G. I. H., (2006) Prey preferences of the leopard (*Panthera pardus*). *Journal of Zoology*, **270**(2): 298-313.
- Hayward, M. W., O'Brien, J. and Kerley, G. I., (2007) Carrying capacity of large African predators: predictions and tests. *Biological Conservation*, **139**(1): 219-229.
- Henschel, P., Abernethy, K.A. and White, L. J. T., (2005) Leopard food habits in the Lope National Park, Gabon, Central Africa. *Africa Journal of Ecology*, **43**: 21-28.
- Henschel, P., (2008) *The conservation biology of the leopard (Panthera pardus) in Gabon* (Doctoral dissertation, Niedersächsische Staats-und Universitätsbibliothek Göttingen).
- Henschel, P., Hunter, L., Breitenmoser, U., Purchase, N., Packer, C., Khorozyan, I., Bauer, H., Marker, L., Sogbohossou, E. and Breitenmoser-Wursten, C., (2008) *Panthera pardus*. In: IUCN 2013. IUCN Red List of Threatened Species. (Version 2013.2.); <http://www.iucnredlist.org>, Gland.
- Holmes, C. M., (2003) The influence of protected area outreach on conservation attitudes and resource use patterns: a case study from western Tanzania. *Oryx*, **37**(3): 305-315.
- Howard, C., Stephens, P. A., Pearce-Higgins, J. W., Gregory, R. D. and Willis, S. G., (2014) 'Improving species distribution models: the value of data on abundance'. *Methods in Ecology and Evolution*, **5**: 506-513.

- Inskip, C. and Zimmermann, A., (2009) Human-felid conflict: a review of patterns and priorities worldwide. *Oryx*, **43**(1): 18-34.
- IUCN (2017) The IUCN Red List of Threatened Species, Version 2017-1. <<http://www.iucnredlist.org>>. Downloaded on 12 May 2017.
- IUCN, (2018) IUCN SSC CAT SPECIALIST GROUP POSITION STATEMENT ON LEOPARD QUOTAS AND NON-DETRIMENT FINDINGS, Seventieth meeting of the Standing Committee Rosa Khutor, Sochi (Russian Federation), 1-5 October 2018.
- Jacobson, A. P., Cattau, M. E., Riggio, J. S., Petracca, L. S. and Fedak, D. A., (2013) Distribution and abundance of lions in northwest tete Province, Mozambique. *Tropical Conservation Science*, **6**(1): 87-107.
- Jacobson, A. P., Gerngross, J., Lemeris, R., Schoonover, R., Anco, C., Breitenmoser, C., Durant, S.D., Farhadinia, M. S., Henschel, P., Kamler, J. F., Laguardia, A., Rostro-Garcia, S., Stein, A. B. and Dollar, L., (2016) Leopard (*Panthera pardus*) status, distribution and the research efforts across its range. *PeerJ*, **4**: e1974.
- Jorge, A. A., (2012) The sustainability of leopard *Panthera pardus* sport hunting in Niassa National Reserve, Mozambique (Doctoral dissertation, Thesis: University of KwaZulu-Natal Westville, South Africa).
- Kabir, M., Hameed, S., Ali, H., Bosso, L., Din, J.U., Bischof, R., Redpath, S. and Nawaz, M. A., (2017) Habitat suitability and movement corridors of grey wolf (*Canis lupus*) in Northern Pakistan. *PloS one*, **12**(11): p.e0187027.
- Kalle, R., Ramesh, T., Qureshi, Q. and Sankar, K., (2013) Predicting the distribution pattern of small carnivores in response to environmental factors in the Western Ghats. *PloS one*, **8**(11): p.e79295.
- Kaunda, S. K. K. and Skinner, J. D., (2003) Black-backed jackal diet at Mokolodi Nature Reserve, Botswana. *African Journal of Ecology*, **41**(1): 39-46.

- Karanth, K. U. and Chellam, R., (2009) Carnivore conservation at the crossroads. *Oryx*, **43**(1): 1.
- Karanth, K. U. and Nichols, J. D., (1998) Estimation of tiger densities in India using photographic captures and recaptures. *Ecology*, **79**(8): 2852-2862.
- Karanth, K. K., Nichols, J. D., Hines, J. E., Karanth, K. U. and Christensen, N. L., (2009) Patterns and determinants of mammal species occurrence in India, *Journal of Applied Ecology*. **46**(6): 1189-1200.
- Kent, V. T., (2011) *The status and conservation potential of carnivores in semi-arid rangelands, Botswana the Ghanzi Farmlands: A case study* (Doctoral dissertation, Durham University).
- Kerth, G., Gusset, M., Garbely, J., König, B., Gabanapelo, T. and Schiess-Meier, M., (2013) Genetic sexing of stock-raiding leopards: not only males to blame. *Conservation genetics resources*, **5**(4): 1101-1105.
- Khorozyan, I., Ghoddousi, A., Soofi, M. and Waltert, M., (2015) Big cats kill more livestock when wild prey reaches a minimum threshold. *Biological Conservation*, **192**: 268-275.
- Kissui, B. M., (2008) Livestock predation by lions, leopards, spotted hyenas, and their vulnerability to retaliatory killing in the Maasai steppe, Tanzania. *Animal Conservation*, **11**(5): 422-432.
- Kittle, A. M., Watson, A. C. and Fernando, T. S. P., (2017) The ecology and behaviour of a protected area Sri Lankan leopard (*Panthera pardus kotiya*) population, *Tropical Ecology*. **58**(1): 71-86.
- Kuhn B. F., (2014) A preliminary assessment of the carnivore community outside Johannesburg, South Africa. *South African Journal of Wildlife Research*, **44**(1): 95–98.

- Larivière, S., Jolicoeur, H., Crête, M., (2000) Status and conservation of the gray wolf (*Canis lupus*) in wildlife reserves of Québec. *Biological Conservation*, **94**(2): 143-151.
- Lindsey, P. A., du Toit, J. T. and Mills, M. G. L., (2005) Attitudes of ranchers towards African wild dogs *Lycaon pictus*: Conservation implications on private land, *Biological Conservation*. **125**(1): 113-121.
- Lindsey, P. A., Roulet, P. A. and Romanach, S. S., (2007) Economic and conservation significance of the trophy hunting industry in sub-Saharan Africa. *Biological conservation*, **134**(4): 455-469.
- Lindsey P. A. and Chikerema-Mandisodza R., (2012) Preliminary report for leopards in Zimbabwe. *TRAFFIC Southern Africa*, 27 pp.
- Lindsey, P. A., Havemann, C. P., Lines, R., Palazy, L., Price, A. E., Retief, T. A., Rhebergen, T. and Van der Waal, C., (2013) Determinants of persistence and tolerance of carnivores on Namibian ranches: implications for conservation on Southern African private lands. *Plos One*, **8**(1): p.e52458.
- Linnell, J. D. and Strand, O., (2000) Interference interactions, co-existence and conservation of mammalian carnivores. *Diversity and Distributions*, **6**(4): 169-176.
- Lovell, C. D., Leopold, B. D., Shropshire, C. C., (1998) Trends in Mississippi Predator Populations, 1980-1995. *Wildlife Society Bulletin*, **26**(3): 552-556.
- Loveridge, A. J., Searle, A. W., Murindagomo, F., Macdonald, D. W., (2007) The impact of sport-hunting on the lion population in a protected area. *Biological Conservation*, **134**, 548–558.
- Loveridge, A. J., Kuiper, T., Parry, R. H., Sibanda, L., Hunt, J. H., Stapelkamp, B., Sebele, L. and Macdonald, D. W., (2017) Bells, bomas and beefsteak: complex patterns of human-predator conflict at the wildlife-agropastoral interface in Zimbabwe. *PeerJ*, **5**: p.e2898.
- Macdonald, D. W., (1983) The ecology of carnivore social behaviour. *Nature*, **301**: 379-384.

- MacNulty, D. R., Smith, D. W., Mech, L. D. and Eberly, L. E., (2009) Body size and predatory performance in wolves: is bigger better?. *Journal of Animal Ecology*, **78**(3): 532-539.
- Maffei, L., Cuéllar, E. and Noss, A., (2004) One thousand jaguars (*Panthera onca*) in Bolivia's Chaco? camera trapping in the Kaa-Iya National Park. *Journal of Zoology*, **262**(3): 295-304.
- Magurran, A. E. and Dornelas, M., (2010) Biological diversity in a changing world. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **365**(1558): 3593-3597.
- Maputla, N. W., Chimimba, C. T. and Ferreira, S. M., (2013) Calibrating a camera trap-based biased mark-recapture sampling design to survey the leopard population in the N'wanetsi concession, Kruger National Park, South Africa. *African Journal of Ecology*, **51**(3): 422-430.
- Marker, L. L. and Dickman, A. J., (2005) Factors affecting leopard (*Panthera pardus*) spatial ecology, with particular reference to Namibian farmlands. *South African Journal of Wildlife Research*, **35** (2): 105-115.
- Martin, R. B. and De Meulenaer, T., (1988) 'Survey of the Status of the Leopard (*Panthera pardus*) in Sub-Saharan Africa' *CITES*: Lausanne, Switzerland.
- McDonald, E., (2016) Too Big to Fail: Rescuing the African Elephant. *The Fletcher Forum of World Affairs*, **40**(1): 113-131.
- Megaze, A., Balakrishnan, M. and Belay, G., (2017) Human-wildlife conflict and attitude of local people towards conservation of wildlife in Chebera Churchura National Park, Ethiopia. *African zoology*, **52**(1): 1-8.
- Menges, V. and Melzheimer, J., (2014) Roan News, Windhoek.
- Mills, M. G. L. and Hes, L., eds., (1997) The complete book of southern African mammals. Struik.

- Mills, M. G. L. and Briggs, H. C., (1993) Prey apportionment and related ecological relationships between large carnivores in Kruger National Park. *Zoological Symposium*, **65**: 253-268.
- Ministry of Environment and Tourism., (2007) Environmental Management Act, 2007 (Act No. 7 of 2007), Government Gazette 27 December 2007 No. 3966.
- Ministry of Environment and Tourism., (2017) Promulgation of Nature Conservation Amendment Act, 2017 (Act No. 3 of 2017), Government Gazette 28 June 2017 No. 6344.
- Miquelle, D., Darman, Y. and Seryodkin, I., (2011) *Panthera tigris ssp. Altaica*. In: IUCN 2013. IUCN Red List of Threatened Species, (Version 2013.2.); <http://www.iucnredlist.org>, Gland.
- MITADER., (2018) Ministry of Land, Environment and Rural Development, Review of the Leopard (*Panthera pardus*) quota of Mozambique, established per Resolution Conf. 10.14 (Rev. CoP16) and non-detriment determinations, in accordance with CITES Decision 17.114.
- Miththapala, S., Seidensticker, J. and O'Brien, S. J., (1996) Phylogeographic subspecies recognition in leopards (*Panthera pardus*): molecular genetic variation. *Conservation Biology*, **10**(4): 1115-1132.
- Mondol, S., Sridhar, V., Yadav, P., Gubbi, S. and Ramakrishnan, U., (2015) Tracing the geographic origin of traded leopard body parts in the Indian subcontinent with DNA-based assignment tests. *Conservation Biology*, **29**(2): 556-564.
- Moyer, M. A., McCown, J. W. and Oli, M. K., (2008) Scale-dependent habitat selection by female Florida black bears in Ocala National Forest, Florida. *Southeastern Naturalist*, **7**: 111–124.
- Mossaz, A., Buckley, R. C. and Castley, J. G., (2015) Ecotourism contributions to conservation of African big cats. *Journal for Nature Conservation*, **28**: 112-118.

- Msuha, M. J., (2009) *Human impacts on carnivore biodiversity inside and outside protected areas in Tanzania* (Doctoral dissertation, UCL (University College London)).
- Muntifering, J. R., Dickman, A. J., Perlow, L. M., Hruska, T., Ryan, P. G., Marker, L. L. and Jeo, R. M., (2006) Managing the matrix for large carnivores: a novel approach and perspective from cheetah (*Acinonyx jubatus*) habitat suitability modelling. *Animal Conservation*, **9**(1): 103-112.
- Murphy T. and Macdonald, D. W., (2010) Pumas and people: lessons in the landscape of tolerance from a widely distributed felid. Pages 431–451 in D. W. Macdonald and A. J. Loveridge, editors. *The biology and conservation of wild felids*. Oxford University Press, Oxford, United Kingdom.
- N/a'an ku sê Foundation, (2018 *unpublished*) Research Programme Report: Conflict and population status of Leopard (*Panthera pardus*), Hyaena and Cheetah (*Acinonyx jubatus*) on commercial farmlands in Namibia between 2007 and 2018.
- NACSO, (2018) The Namibian Association of Community Based Natural Resource Management (CBNRM) Support Organisations (NACSO) annual communal conservancy report, downloaded on the 1st January 2019, www.nacso.org.na/.
- Newmark, W. D., Manyanza, D. N., Gamassa, D. G. M. and Sariko, H. I., (1994) The conflict between wildlife and local people living adjacent to protected areas in Tanzania: human density as a predictor. *Conservation Biology*, **8**(1): 249-255.
- Noack, J., (2016) The assessment of leopard (*Panthera pardus*) density and population size via a capture – recapture framework in an island bound conservation area in Namibia. Retrieved from: <http://www.africat.org/projects/the-africat-predator-population-density-study-in-the-okonjima-nature-reserve>.
- Noss, A.J., Gardner, B., Maffei, L., Cuéllar, E., Montaña, R., Romero-Muñoz, A., Sollman, R. and O'Connell, A. F., (2012) Comparison of density estimation methods for mammal populations with camera traps in the Kaa-Iya del Gran Chaco landscape. *Animal Conservation*, **15**(5): 527-535.

- Nowell, K. and Jackson, P., (1996): Wild Cats: Status Survey and Conservation Action Plan. 421 pp. IUCN/SSC Cat Specialist Group, International Union for Conservation of Nature and Natural Resources (IUCN), Gland, Switzerland.
- Núñez-Quirós, L., (2009) Distribution of the Iberian Wolf (*Canis lupus signatus*) in Galicia (NW Spain): Concordance Between Field Sampling and Questionnaires, *Wildlife Biology in Practice*, **1**(5): 23-32.
- Nuno, A. and St John, F. A. V., (2015) How to ask sensitive questions in conservation: A review of specialized questioning techniques. *Biological Conservation*, **189**: 5-15.
- O'Brien, T. G., Kinnaird, M. F. and Wibisono, H. T., (2003) Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape, In *Animal Conservation forum* (Vol. 6, No. 2, pp. 131-139), Cambridge University Press.
- O'Brien, T. G. and Kinnaird, M. F., (2011) Density estimation of sympatric carnivores using spatially explicit capture–recapture methods and standard trapping grid. *Ecological Applications*, **21**(8): 2908-2916.
- O'Connell, A. F., Nichols, J. D. and Karanth, K. U., (2011) *Camera traps in animal ecology: methods and analyses*, Tokyo: Springer.
- Odden, M., Athreya, V., Rattan, S. and Linnell, J. D., (2014) Adaptable neighbours: movement patterns of GPS-collared leopards in human dominated landscapes in India. *PLoS One*, **9**(11): p.e112044.
- Olson, L. J., (2006) The economics of terrestrial invasive species: a review of the literature. *Agricultural and Resource Economics Review*, **35**(1): 178-194.
- Otis, D. L., Burnham, K. P., White, G. C., Anderson D. R., (1978) Statistical inference from capture data on closed animal populations. *Wildlife Monographs*, **62**: 1–135.
- Owen, C., Niemann, S. and Slotow, R., (2010) Copulatory parameters and reproductive success of wild leopards in South Africa. *Journal of Mammalogy*, **91**(5): 1178-1187.

- Packer, C., Kosmala, M., Cooley, H.S., Brink, H., Pintea, L., Garshelis, D., Purchase, G., Strauss, M., Swanson, A., Balme, G. and Hunter, L., (2009) Sport hunting, predator control and conservation of large carnivores. *PloS one*, **4**(6): p.e5941.
- Parker, I. D., Watts, D. E., McCleery, R. A., Lopez, R. R., Silvy, N. J. and Davis, D. S., (2008) Digital versus film-based remote camera systems in the Florida Keys. *Wildlife Biology in Practice*, **4**(1): 1-7.
- Pereira, H. M., Navarro, L. M. and Martins, I. S., (2012) Global biodiversity change: the bad, the good, and the unknown. *Annual Review of Environment and Resources*, **37**: 25-50.
- Pettorelli, N., Bro-Jørgensen, J., Durant, S. M., Blackburn, T. and Carbone, C., (2009) Energy availability and density estimates in African ungulates. *The American Naturalist*, **173**(5): 698-704.
- Pettorelli, N., Lobora, A. L., Msuha, M. J., Foley, C. and Durant, S. M., (2010) Carnivore biodiversity in Tanzania: revealing the distribution patterns of secretive mammals using camera traps. *Animal Conservation*, **13**(2): 131-139.
- Phillips, S. J., Anderson, R. P. and Schapire, R. E., (2006) Maximum entropy modeling of species geographic distributions. *Ecological Modeling*, **190**: 231–259.
- Portas, R., Edwards, S., Beytell, P., Wachter, B. and Melzheimer, J. (2018). Leopard survey in Khaudum National park and Kaokoland. Final report. Ministry of Environment and Tourism of the Republic of Namibia. Windhoek.
- Pitman, R. T., Swanepoel, L. H., Hunter, L., Slotow, R. and Balme, G. A., (2015) The importance of refugia, ecological traps and scale for large carnivore management. *Biodiversity and Conservation*, **24**(8): 1975-1987.
- Pitman, R. T., Fattbert, J., Williams, S. T., Williams, K. S., Hill, R. A., Hunter, L. T., Robinson, H., Power, J., Swanepoel, L., Slotow, R. and Balme, G. A., (2017) Cats, connectivity and conservation: incorporating data sets and integrating scales for wildlife management. *Journal of applied ecology*, **54**(6): 1687-1698.

- Prugh, L. R., Stoner, C. J., Epps, C. W., Bean, W. T., Ripple, W. J., Laliberte, A. S. and Brashares, J. S., (2009) The rise of the mesopredator. *Bioscience*, **59**(9): 779-791.
- Prugh, L. R. and Golden, C. D., (2014) Does moonlight increase predation risk? Meta-analysis reveals divergent responses of nocturnal mammals to lunar cycles. *Journal of Animal Ecology*, **83**(2): 504-514.
- Pulliam, H. R., (1988) Sources, sinks, and population regulation. *The American Naturalist*, **132**(5): 652-661.
- Purchase, G. and Mateke, C., (2008) The state of knowledge regarding leopard (*Panthera pardus*) in three range states (Mozambique, Zambia and Zimbabwe) in the context of improving management of trophy hunting, *Report for the CAMPFIRE Association of Zimbabwe*.
- Ramesh, T., Kalle, R., Rosenlund, H. and Downs, C. T., (2017) Low leopard populations in protected areas of Maputaland: a consequence of poaching, habitat condition, abundance of prey, and a top predator. *Ecology and evolution*, **7**(6): 1964-1973.
- Ray, R. R., (2011) *Ecology and population status and the impact of trophy hunting of the leopard Panthera pardus (LINNAEUS, 1758 in the Luambe National Park and surrounding Game Management Areas in Zambia* (Doctoral dissertation, PhD thesis, Universität Bonn).
- Rich, L. N., Mitchell, M. S., Gude, J. A. and Sime, C. A., (2012) Anthropogenic mortality, intraspecific competition, and prey availability influence territory sizes of wolves in Montana. *Journal of Mammalogy*, **93**(3): 722-731.
- Ripple, W. J., Estes, J. A., Beschta, R. L., Wilmers, C. C., Ritchie, E. G., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M. P., Schmitz, O. J., Smith, D. W., Wallach, A., D. and Wirsing, A. J., (2014) Status and Ecological Effects of the World's Largest Carnivores. *Science*, **343**(6167): 12414841-10.

- Ripple, W. J., Chapron, G., López-Bao, J. V., Durant, S. M., Macdonald, D. W., Lindsey, P. A., Bennett, E. L., Beschta, R. L., Bruskotter, J. T., Campos-Arceiz, A. and Corlett, R. T., (2016) Saving the world's terrestrial megafauna. *BioScience*, **66**(10): 807-812.
- Romanach, S. S., Lindsey, P. A. and Woodroffe, R., (2007) Determinants of attitudes towards predators in central Kenya and suggestions for increasing tolerance in livestock dominated landscapes. *Oryx*, **41**(2): 185-195.
- Rosenblatt, E., Creel, S., Becker, M. S., Merkle, J., Mwape, H., Schuette, P. and Simpamba, T., (2016) Effects of a protection gradient on carnivore density and survival: an example with leopards in the Luangwa valley, Zambia. *Ecology and evolution*, **6**(11): 3772-3785.
- Rowcliffe, J. M. and Carbone, C., (2008) Surveys using camera traps: are we looking to a brighter future? *Animal Conservation*, **11**: 185-186.
- Rovero, F., Tobler, M. and Sanderson, J., (2010) Camera trapping for inventorying terrestrial vertebrates, *Manual on field recording techniques and protocols for All Taxa Biodiversity Inventories and Monitoring. The Belgian National Focal Point to the Global Taxonomy Initiative*, pp.100-128.
- Rushton, S. P., Ormerod, S. J. and Kerby, G., (2004) New paradigms for modelling species distributions? *Journal of Applied Ecology*, **41**(2): 193-200.
- Sala, O. E., Chapin, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R. and Wall, D. H., (2000) Global biodiversity scenarios for the year 2100. *Science*, **287**(5459): 1770-1774.
- Santangeli, A., Arkumarev, V., Rust, N. and Girardello, M., (2016) Understanding, quantifying and mapping the use of poison by commercial farmers in Namibia—Implications for scavengers' conservation and ecosystem health. *Biological Conservation*, **204**: 205-211.

- Schuetz, P., Creel, S. and Christianson, D., (2013) Coexistence of African lions, livestock, and people in a landscape with variable human land use and seasonal movements. *Biological Conservation*, **157**: 148-154.
- Seoraj-Pillai, N. and Pillay, N., (2016) A meta-analysis of human–wildlife conflict: South African and global perspectives. *Sustainability*, **9**(1): 34.
- Sergio, F., Caro, T., Brown, D., Clucas, B., Hunter, J., Ketchum, J. and Hiraldo, F., (2008) Top predators as conservation tools: ecological rationale, assumptions, and efficacy. *Annual review of ecology, evolution, and systematics*, **39**: 1-19.
- Shortridge, G. C., (1934) The Mammals of South West Africa. Vols 1 & 2. London, Heinemann.
- Sillero-Zubiri, C., Sukumar, R. and Treves, A., (2007) Living with wildlife: the roots of conflict and the solutions. *Key topics in conservation biology*, 266-272.
- Skinner, J. D., Fairall, N. and Bothma, J. du P., (1977) South African red data book – large mammals. South African National Scientific Programmes Report no.18, Pretoria: Council for Scientific and Industrial Research.
- St John, F. A., Keane, A. M., Edwards-Jones, G., Jones, L., Yarnell, R. W. and Jones, J. P., (2012) Identifying indicators of illegal behaviour: carnivore killing in human-managed landscapes. *Proceedings of the Royal Society B: Biological Sciences*, **279**(1729): 804-812.
- Stander, P. E., Haden, P. J., Kagece, I. I. and Ghau, I. I., (1997) The ecology of asociality in Namibian leopards. *Journal of Zoology*, **242**(2): 343-364.
- Stander, P. E., (1998) Spoor counts as indices of large carnivore populations: the relationship between spoor frequency, sampling effort and true density. *Journal of Applied Ecology*, **35**(3): 378-385.

- Stander, P. and Hanessen, L., (2000) Population ecology and long-term monitoring of free-ranging populations in Namibia's marginal and arid environments, Predator Research Programme October Progress Report.
- Stander, P., Hanssen, L., Maclellan, S., Junker, J., Moller, W., Isak, W., Dougab, W., Ariseb, S., (2001) Population ecology and long-term monitoring of free-ranging populations in Namibia's marginal and arid environments, Predator Research Programme May Progress Report.
- Stein, A. B., Fuller, T. K. and Marker, L. L., (2008) Opportunistic use of camera traps to assess habitat-specific mammal and bird diversity in northcentral Namibia. *Biodiversity and Conservation*, **17**(14): 3579-3587.
- Stein, A. B., Fuller, T. K., Damery, D. T., Sievert, L. and Marker, L. L., (2010) Farm management and economic analyses of leopard conservation in north-central Namibia. *Animal Conservation*, **13**(4): 419-427.
- Stein, A. B., Fuller, T. K., DeStefano, S. and Marker, L. L., (2011a) Leopard population and home range estimates in north-central Namibia. *African Journal of Ecology*, **49**(3): 383-387.
- Stein, A., Andreas, A., Aschenborn, O., Kastern, M., Andreas, A. and Thompson, S., (2011b) Namibian National Leopard Survey 2011 Final Report. Ministry of Environment and Tourism Internal Report, Windhoek, Namibia.
- Stein, A. B., and Hayssen, V., (2013) *Panthera pardus* (Carnivora: Felidae). *Mammalian Species*, **45**(900): 30-48.
- Stein, A. B., Bourquin, S. L. and McNutt, J. W., (2015) Avoiding intraguild competition: Leopard feeding ecology and prey caching in northern Botswana. *African Journal of Wildlife Research*, **45**(2): 247-257.
- Stein, A. B., Athreya, V., Gerngross, P., Blame, G., Henschel, P., Karanth, U., Miquelle, D., Rostro-Garcia, S., Kamler, J. F., Laguardia, A., Khorozyan, I. and Ghoddousi, A., (2016) *Panthera pardus* (errata version published in 2016), The IUCN Red List of Threatened Species 2016: e.T15954A102421779.

- Stoner D. C., Wolfe, M. L., Rieth, W. R., Bunnell, K. D., Durham, S. L., Stoner, L.L., Jeschke, J., (2013) De facto refugia, ecological traps and the biogeography of anthropogenic cougar mortality in Utah. *Divers Distribution*, **19**: 1114–1124.
- Stuart, C., and Wilson, V., (1988) The cats of Southern Africa. Chipangali Wildlife Trust, Bulawayo, Zimbabwe.
- Strampelli, P., (2015) *Status and habitat use responses of leopard (Panthera pardus) in a human impacted region of rural Mozambique* (Doctoral dissertation, Department of Life Sciences, Silwood Park, Imperial College London).
- Stratford, K., Naholo, S., Crawford, S., Stratford, S., (2018 *unpublished*) Ongava's Leopards.
- Sun, C. C., Fuller, A. K. and Royle, J. A., (2014) Trap configuration and spacing influences parameter estimates in spatial capture-recapture models. *PloS one*, **9**(2): p.e88025.
- Sunnucks, P., (2000) Efficient genetic markers for population biology. *Trends in Ecology and Evolution*, **15**: 199-203.
- Swanepoel, L. H., Lindsey, P., Somers, M. J., Hoven, W. V. and Dalerum, F., (2013) Extent and fragmentation of suitable leopard habitat in South Africa. *Animal Conservation*, **16**(1): 41-50.
- Swanepoel, L. H., Lindsey, P., Somers, M. J., Van Hoven, W. and Dalerum, F., (2014) The relative importance of trophy harvest and retaliatory killing of large carnivores: South African leopards as a case study. *South African Journal of Wildlife Research*, **44**(2): 115-134.
- Swanepoel, L. H., Somers, M. J. and Dalerum, F., (2015) Functional responses of retaliatory killing versus recreational sport hunting of leopards in South Africa. *PLoS one*, **10**(4): p.e0125539.

- Swann, D. E., Hass, C. C., Dalton, D. C. and Wolf, S. A., (2004) Infrared-triggered cameras for detecting wildlife: an evaluation and review. *Wildlife Society Bulletin*, **32**(2): 357-365.
- Thorn, M., Scott, D. M., Green, M., Bateman, P. W., Cameron, E. Z., (2009) Estimating brown hyaena occupancy using baited camera traps. *African Journal of Wildlife Research*, **39**(1): 1-10.
- Thorn, M., Green, M., Dalerum, F., Bateman, P. W. and Scott, D. M., (2012) What drives human–carnivore conflict in the North West Province of South Africa? *Biological Conservation*, **150**(1): 23-32.
- Thorn, M., Green, M., Scott, D. and Marnewick, K., (2013) Characteristics and determinants of human-carnivore conflict in South African farmland. *Biodiversity and Conservation*, **22**(8): 1715-1730.
- Tobler, M. W., Carrilo-Percastegui, S. E., Pitman, R. L., Mares, R. and Powell, G., (2008) An evaluation of camera traps for inventorying large and medium sized terrestrial rainforest mammals. *Animal Conservation*, **11**: 169–178.
- Treves, A. and Karanth, K. U., (2003) Human-carnivore conflict and perspectives on carnivore management worldwide. *Conservation Biology*, **17**: 1491-1499.
- Trinkel, M., Fleischmann, P. H. and Slotow, R., (2017) Electrifying the fence or living with consequences? Problem animal control threatens the long-term viability of a free-ranging lion population. *Journal of Zoology*, **301**(1): 41-50.
- Uphyrkina, O., Johnson, W. E., Quigley, H., Moquette, D., Marker, L., Bush, M. and O'Brien, S. J., (2001) Phylogenetics, genome diversity and origin of modern leopard, *Panthera pardus*. *Molecular Ecology*, **10**(11): 2617–2633.
- Voigt, C. C., Krofel, M., Menges, V., Wachter, B. and Melzheimer, J., (2018) Sex-specific dietary specialization in a terrestrial apex predator, the leopard, revealed by stable isotope analysis. *Journal of Zoology*. doi:10.1111/jzo.12566.

- Wallace, R. B., Gomez, H., Ayala, G. and Espinoza, F., (2003) Camera Trapping for Jaguar (*Panthera Onca*) in the Tuichi Valley, Bolivia. *Journal of Neotropical Mammalogy*, **10**(1): 133-139.
- Weaver, L. C. and Petersen, T., (2008) Namibia communal area conservancies. In: R.D. Baldus, G.R. Damm and K. Wollscheid, eds., *Best Practices in Sustainable Hunting - a guide to best practices from around the world*. CIC - International Council for Game and Wildlife Conservation, pp.48– 52.
- Weise, F. J., Lemeris, J., Stratford, K. J., van Vuuren, R. J., Munro, S. J., Crawford, S. J., Marker, L. L. and Stein, A. B., (2015) A home away from home: insights from successful leopard (*Panthera pardus*) translocations. *Biodiversity and conservation*, **24**(7): 1755-1774.
- Wemmer, C, Kunz, T, Lundie-Jekins, G, McShea, W., (1996) Mammalian Sign. In: Wilson DE, Cole FR, Nichols JD, Rudran R, Foster MS (eds) Measuring and monitoring biological diversity, Standard methods for mammals. Smithsonian Institution, USA, pp 157–176.
- White, P. C. L., Jennings, N. V., Renwick, R. R. and Barker, N. H. L., (2005) Questionnaires in ecology: a review of past use and recommendations for best practice. *Journal of Applied Ecology*, **42**(3): 421-430.
- Wiesel, I. and Edwards, S., (2014) Brown Hyaena Research Project, Human-carnivore-conflict study on farms bordering the Namib Naukluft and Tsau//Khaeb NPs. Raw data.
- Williams, P. H., Burgess, N. D. and Rahbek, C., (2000) Flagship species, ecological complementarity and conserving the diversity of mammals and birds in sub-Saharan Africa. *Animal Conservation*, **3**: 249-260.
- Williams, S. T., Williams, K. S., Joubert, C. J. and Hill, R. A., (2016) The impact of land reform on the status of large carnivores in Zimbabwe. *PeerJ*, **4**: p.e1537.
- Williams, S. T., Williams, K. S., Lewis, B. P. and Hill, R. A., (2017) Population dynamics and threats to an apex predator outside protected areas: implications for carnivore management. *Royal Society open science*, **4**(4): 161090.

- Winter, S. J., Esler, K. J. and Kidd, M., (2005) An index to measure the conservation attitudes of landowners towards Overberg Coastal Renosterveld, a critically endangered vegetation type in the Cape Floral Kingdom, South Africa. *Biological Conservation*, **126**(3): 383-394.
- Winter, S. J., Prozesky, H., Esler, K. J., (2007) A Case Study of Landholder Attitudes and Behaviour Toward the Conservation of Renosterveld, a Critically Endangered Vegetation Type in Cape Floral Kingdom, South Africa. *Environmental Management*, **40**: 46–61.
- Winterbach, H. E. K., Winterbach, C. W., Somers, M. J. and Hayward, M. W., (2012) Key factors and related principles in the conservation of large African carnivores. *Mammal Review*, **43**(2): 89-110.
- Woodroffe, R., Thirgood, S. and Rabinowitz, A. (Eds.), (2005) *People and wildlife, conflict or co-existence?*, (No. 9), Cambridge University Press.
- Zimmermann, B., Wabakken, P. and Dötterer, M., (2001) Human-carnivore interactions in Norway: How does the re-appearance of large carnivores affect people's attitudes and levels of fear. *Forest Snow and Landscape Research*, **76**(1): 1-17.
- ZPWMA., (2018) Zimbabwe Parks and Wildlife Management Authority, Zimbabwe's Review of the Convention on International Trade in Endangered Species (CITES) Leopard (*Panthera pardus*) Quota, AC30 Doc. 15 Annex 6.

8. Appendices

Appendix 1: Table 8.1. Outlines the individual freehold farms involved in the two camera trap survey, Auas Mountains and Omaruru.

Survey area 1 - Auas Mountains Farms
Aris
Auas Safari Lodge
Binenheim
Gocheganas Nature Reserve
Haigamas
Krumhuk
Lichtenstein East
Lichtenstein North
Lichtenstein South
Lichtenstein West
Neu-Brack
Survey area 2 - Omaruru Farms
Haidehof
Klein Okosombuka
Okatjerute
Okaturua
Omburo Nord East
Omburo Nord West
Osera Omewa
Otjikoko
Otjikoko South
Ozondjisse

Appendix 2:

Table 8.2.1. Outlines all the meeting invitations received and whether those meetings were attended.

Date	Meetings Attended	Invited	Attended
28 th November 2017	NAPHA AGM	Yes	Yes
2018			
February	Waterberg Conservancy	Yes	No
February	Keinab BV	Yes	No
20 th February	Rietfontein FA	Yes	No

Date	Meetings Attended	Invited	Attended
1 st March	Kalkplato BV	Yes	No
6 th March	Summerdown FA	Yes	Yes
8 th March	WLA Auction - Windhoek	Yes	Yes
10 th March	Karibib FA	Yes	Yes
12 th March	Outjo FA	Yes	Yes
12 th March	Seeis FA	Yes	No
12 th March	Windhoek SLU	Yes	No
13 th March	Gobabis FA	Yes	No
14 th March	Maltahöhe BV	Yes	Yes
15 th March	Steinhausen FA	Yes	No
15 th March	Northern Khomas FA	Yes	No
15 th March	Keetmanshoop SLU	Yes	Yes
16 th March	Abenab FA	Yes	No
16 th March	Bethanie FA	Yes	Yes
16 th March	Omitara FA	Yes	No
20 th March	Hochfeld FA	No (full schedule)	No
20 th March	Epukiro FA	Yes	No
21 st March	Tsumeb FA	Yes	Yes
23 rd March	AGRA Otjiwarongo		Yes
24 th March	Namatanga Conservancy	Yes	Yes
27 th March	SONOP FA	Yes	Yes
28 th March	Platvelt FA	Yes	Yes
5 th April	Namboer Auction - Windhoek		Yes
7 th April	Seeis Conservancy	Yes	Yes
11 th April	Omaruru FA	Yes	Yes
11 th April	Nina FA	Yes	No
11 th April	Otjikondo	Yes	No
12 th April	Karasburg FA	Yes	No
13 th April	Aroab FA	Yes	No
13 th April	Excelsior FA (near Etosha)	No (full schedule)	No
13 th April	AGRA Windhoek		Yes
16 th April	AGRA Windhoek		Yes
17 th April	Osire / Waterberg FA	Yes	Yes
18 th April	Otavi FA	Yes	Yes
19 th April	Wilhelmstal-Okasise FA	Yes	No

Date	Meetings Attended	Invited	Attended
19 th April	Dorabsis FA	Yes	Yes
20 th April	Okahandja FA	Yes	Yes
22 nd April	Witvlei FA	Yes	No
23 rd April	AGRA Rehoboth		Yes
25 th April	Hochfeld FA	Yes	Yes
26 th April	CANAM AGM	Yes	Yes
30 th April	AGRA Grootfontein	Yes	Yes
4 th May	Outjo Wildsfees	Booth	Yes
5 th May	Outjo Wildsfees	Booth	Yes
7 th May	AGRA Otjiwarongo	Yes	Yes
8 th May	Kalkfeld Conservancy	Yes	Yes
24 th May	Helmeringhausen FA	Yes	No
26 th May	2018 NLU AgriBraai		Yes
21 st June	Windhoek Farmerverein	Yes	No
14 th August	Otjiwarongo FA	Yes	Yes
22 nd August	Khomas Hochland FA	Yes	Yes
11 th October	NAU Congress	Yes	Yes
17 th October	Otjikondo FA	Yes	No
19 th October	Excelsior BV	Yes	Cancelled
	Kalahari Oss BV	No (full schedule)	No
	Koes	Yes	No
	Leonardville	No (full schedule all year)	No
	Mariental	No (full schedule)	No
	Noordgrens BV	Not currently holding meetings	No
	Klein Karas BV	Been dissolved	No
	Keetmanshoop BV	Don't hold official meetings anymore	No

Table 8.2.2. Outlines the contributors and the number of leopard presence records provided to the Atlasing project.

Recorder	Number of records
Event Book, HWC data	1780
IZW, Berlin	935
Cheetah Conservation Fund	846
Naankuse, Group	843
Brown Hyena Research, Project	502
Ongava, Research Centre	410
Richmond-Coggan, Louisa	354
Game Count, Zambezi	247
Old carnivore atlas	120
Tindall, Murray	76
Game Count, Kavango	53
Carnivore Tracker, App	52
AfriCat, Namibia	37
Cooper, Sue	31
Ward, David	27
Hauptfleisch, Morgan	24
Versfeld, Wilferd	24
Gondwana, Collection	22
Weise, Florian	21
Kwando Carnivore, Project	19
Guides, NamibRand	15
Jo Tagg, Vera Neuhaus	13
Otjandaue, Farm#70	12
Game Count, North West	8
Navachab, Farm#54	7
Rudman, Duane	6
Nesticky, Viktor	6
Barthorp, Ed	4
L'Estrange, Piers	4
Periquet, Stephanie	3
Heger, Gudrun	3
Game Count, Greater Fish River	3
Tarr, Peter	2
Peters, Raymond	2
Recorder	2
Kayser, Conny	2
Bétrissey, Sophie	2
Farm#70, Abbabis	2
Dantu, Sandra	1
Briers-Louw, Willem	1
Walters, Matthew	1
Mannheimer, Coleen	1

Recorder	Number of records
Dietz, Horst	1
Kelly, Mel	1
Fels, Manuela	1
Tony Robertson, Alice Jarvis	1
Moeller, Michelle & Carl-Heinz	1
Game Count, GSN	1
Brown, Chris	1
Total	6529

Appendix 3: Table 8.3. Estimates of relative contributions of the environmental variables to the Maxent model (Source: Dr. Vera De Cauwer).

Variable	Percent contribution	Permutation importance
Temperature seasonality	21.3	40.5
Land cover (GLC2006 1km)	20.3	1.2
Land ownership	18.1	3.1
Precipitation of the wettest month	11.5	7.8
Mean temperature in the wettest month	5.7	6.8
Sand% in topsoil (1km)	3.7	5.1
Cattle density	3.4	1.2
Soil depth (up to bedrock) 1km	3.4	1.5
Small stock density	2.5	4.3
Human population density (1km)	1.6	2.7
Carrying capacity (1km)	1.5	3.6
Slope	1.4	3
Altitude	1.3	0.5
Distance to road	0.9	2.2
Potential evapotranspiration in December	0.9	2.6
Precipitation in the warmest quarter	0.8	4.5
Average Enhanced vegetation index over whole year for the time period (2000 – 2018)	0.7	3.9

Variable	Percent contribution	Permutation importance
Average Enhanced vegetation index in December for the time period (2000 – 2018)	0.7	3.9
Average Enhanced vegetation index in December for the time period (2000 – 2018)	0.6	3.6

Appendix 4: Table 8.4. The communal conservancies leopard trophy hunting quota allocation, distribution and quota conditions (2017 to 2019).

Conservancy by regions	2017-2019 (per year)	Quota conditions
Erongo Region		
Tsiseb	1	One over three years
Ohungu	1	One over three years
# Gaingu	1	One over three years
Zambezi Region		
Mayuni	1	One over three years
Kavango Region		
George Mukoya & Muduva Nyangana	1	One over three years
Kunene Region		
Anabeb	1	
!Khoru !goreb	1	
Orupembe	1	
Puros	1	
Sesfontein	1	
Otuzemba	1	
Orupupa	1	
Okangundumba	1	One over three years
//Huab	1	
Otjambangu	1	One over three years
Otjombande	1	One over three years
Ombujokanguindi	1	One over three years
Kunene river	1	
Epupa	1	
Sorri sorris	1	
Torra	1	
Ozondundu	1	
Omatendeka	1	
#Khoadi-//Hoas	1	

Conservancy by regions	2017-2019 (per year)	Quota conditions
Ehi-Rovipuka	1	
Otjozondjupa Region		
Nyae Nyae	3	
N#a-Jaqna	3	
Ondjou	1	
Omaheke Region		
Eiseb	1	
Total	33	

Appendix 5: Table 8.5. The national parks leopard trophy hunting quota allocation as of March 2019.

Location	Quota allocation
Bwabwata West	2 per year
Bwabwata East	2 per year
Namib Naukluft Park	3 over a period of 5 years
Western Kavango and Mangetti National Park	2 over a period of 5 years
Waterberg National Park	1 per year

Appendix 6: Table 8.6. A breakdown by year of the successful leopard trophy hunts across communal conservancies, national parks, freehold conservancies and community associations between 2001 and 2018.

Year	Communal Conservancy / National Park / Freehold Conservancy / Community Association	Number of successful trophy hunts
2001	Mahangu National Park	1
	Mangetti National Park	1
	Seeis Conservancy	1
	Waterberg Conservancy	1
	Sub-total	8
2002	Bwabwata (Ukn)	1
	Mahangu National Park	2
	Naye-Naye Conservancy	1
	Ngarangombe Conservancy	1
	Okawi Conservancy	1
	Ombotozu Conservancy	1
	Sub-total	7
2003	≠Khoadi-/Hôas Conservancy	1

Year	Communal Conservancy / National Park / Freehold Conservancy / Community Association	Number of successful trophy hunts
	Bwabwata (Ukn)	4
	Etosha Conservancy	5
	Naye-Naye Conservancy	3
	Sub-total	13
2004	Kwandu Conservancy	1
	Mahangu National Park	1
	Naye-Naye Conservancy	1
	Sub-total	3
2005	Ehi-Rovipuka Conservancy	1
	Kwandu Conservancy	1
	Naye-Naye Conservancy	1
	Ombotozu Conservancy	1
	Sub-total	4
2007	Anabeb Conservancy	1
	N#a-Jaqna Conservancy	1
	Naye-Naye Conservancy	1
	Sesfontein Conservancy	1
	Sorris Sorris Conservancy	1
	Torra Conservancy	2
	Sub-total	7
2008	#Khoadi-//Hôas Conservancy	1
	Anabeb Conservancy	2
	Ehi-Rovipuka Conservancy	1
	Etosha Conservancy	1
	N#a-Jaqna Conservancy	4
	Naye-Naye Conservancy	4
	Sesfontein Conservancy	3
	Sorris Sorris Conservancy	2
	Sub-total	18
2009	Ehi-Rovipuka Conservancy	2
	N#a-Jaqna Conservancy	3
	Naye-Naye Conservancy	1
	Sub-total	6
2010	#Gaingu Conservancy	2
	Ehi-Rovipuka Conservancy	1
	Mashi Conservancy	1
	N#a-Jaqna Conservancy	1
	Naye-Naye Conservancy	2
	Omatendeka Conservancy	1
	Otjimboyo Conservancy	1
	Sub-total	9
2011	#Gaingu Conservancy	1
	#Khoadi-//Hôas Conservancy	2
	Bwabwata (Ukn)	1
	Ehi-Rovipuka Conservancy	3

Year	Communal Conservancy / National Park / Freehold Conservancy / Community Association	Number of successful trophy hunts
	Khaudum North Complex	1
	N#a-Jaqna Conservancy	3
	Naye-Naye Conservancy	2
	Omatendeka Conservancy	2
	Otjimboyo Conservancy	1
	Puros Conservancy	1
	Torra Conservancy	2
	Sub-total	19
2012	#Gaingu Conservancy	1
	//Huab Conservancy	2
	≠Khoadi-//Hôas Conservancy	2
	Anabeb Conservancy	1
	Bwabwata West	1
	Kwandu Conservancy	1
	Kyaramacan Association	1
	N#a-Jaqna Conservancy	1
	Naye-Naye Conservancy	1
	Omatendeka Conservancy	2
	Orupembe Conservancy	1
	Otjimboyo Conservancy	1
	Ozondundu Conservancy	1
	Sorris Sorris Conservancy	1
	Sub-total	17
2013	#Gaingu Conservancy	2
	≠Khoadi-//Hôas Conservancy	2
	Bwabwata East	1
	Bwabwata West	2
	Ehi-Rovipuka Conservancy	1
	Kyaramacan Association	1
	N#a-Jaqna Conservancy	2
	Naye-Naye Conservancy	2
	Sesfontein Conservancy	1
	Sub-total	14
2014	≠Khoadi-//Hôas Conservancy	2
	Anabeb Conservancy	1
	Bwabwata East	2
	Ehi-Rovipuka Conservancy	1
	Kwandu Conservancy	1
	Mangetti National Park	2
	N#a-Jaqna Conservancy	1
	Naye-Naye Conservancy	2
	Omatendeka Conservancy	1
	Orupembe Conservancy	1
	Orupupa Conservancy	1
	Otjimboyo Conservancy	1

Year	Communal Conservancy / National Park / Freehold Conservancy / Community Association	Number of successful trophy hunts
	Otuzemba Conservancy	1
	Puros Conservancy	2
	Sesfontein Conservancy	1
	Torra Conservancy	2
	Waterberg Plateau Park	1
	Sub-total	23
2015	#Gaingu Conservancy	1
	//Huab Conservancy	1
	≠Khoadi-//Hôas Conservancy	2
	Anabeb Conservancy	1
	Bwabwata East	1
	Ehi-Rovipuka Conservancy	1
	Mahangu National Park	1
	Mangetti National Park	1
	Mayuni Conservancy	1
	N#a-Jaqna Conservancy	2
	Naye-Naye Conservancy	4
	Sesfontein Conservancy	2
	Torra Conservancy	2
	Wuparo Conservancy	1
	Sub-total	21
2016	#Gaingu Conservancy	1
	//Huab Conservancy	1
	Anabeb Conservancy	1
	Bwabwata East	2
	Etosha Conservancy	2
	Kunene River Conservancy	1
	Kwandu Conservancy	1
	Mahangu National Park	1
	Mangetti National Park	1
	Mashi Conservancy	2
	N#a-Jaqna Conservancy	5
	Naye-Naye Conservancy	6
	Okangundumba Conservancy	1
	Okongoro Conservancy	1
	Omatendeka Conservancy	2
	Orupupa Conservancy	1
	Otjitanda Conservancy	1
	Otuzemba Conservancy	1
	Ozondundu Conservancy	1
	Puros Conservancy	1
	Sesfontein Conservancy	2
	Sorris Sorris Conservancy	2
	Tsiseb Conservancy	2
	Waterberg Plateau Park	1

Year	Communal Conservancy / National Park / Freehold Conservancy / Community Association	Number of successful trophy hunts
	Sub-total	40
2017	//Huab Conservancy	1
	#Khoadi-//Hôas Conservancy	1
	Anabeb Conservancy	1
	Bwabwata (Ukn)	1
	Bwabwata East	1
	Bwabwata West	1
	Ehi-Rovipuka Conservancy	1
	Eiseb Conservancy	1
	Mangetti National Park	1
	N#a-Jaqna Conservancy	3
	Naye-Naye Conservancy	3
	Omatendeka Conservancy	1
	Ozondundu Conservancy	1
	Puros Conservancy	1
	Sesfontein Conservancy	1
	Tsiseb Conservancy	1
	Waterberg Plateau Park	1
	Sub-total	21
2018	#Gaingu Conservancy	1
	Anabeb Conservancy	1
	Bwabwata East	2
	Bwabwata West	2
	Ehi-Rovipuka Conservancy	1
	Eiseb Conservancy	1
	Kunene River Conservancy	1
	Mashi Conservancy	1
	Mayuni Conservancy	1
	N#a-Jaqna Conservancy	3
	Naye-Naye Conservancy	3
	Omatendeka Conservancy	1
	Ondjou Conservancy	1
	Otuzemba Conservancy	1
	Ozondundu Conservancy	1
	Sub-total	21
	Average per year (\pmSD)	15 \pm9
	Total	247

Appendix 7:

Table 8.7.1. The average number of hunts per year for successful hunts between 2010 and 2018.

	Year	Number of hunt days	Average number of hunt days (\pmSD)
Successful hunts	2010	76	6.9 (\pm 4.1)
	2011	718	7.0 (\pm 5.2)
	2012	730	7.4 (\pm 4.7)
	2013	804	7.2 (\pm 4.5)
	2014	960	6.9 (\pm 4.2)
	2015	973	6.9 (\pm 4.7)
	2016	844	5.7 (\pm 3.8)
	2017	890	5.8 (\pm 3.8)
	2018	885	6.4 (\pm 4.6)
	Total	6880	6.6 (\pm4.4)

Table 8.7.2. The average number of hunt days per region for successful hunts between 2010 and 2018.

	Regions	Number of hunt days	Average number of hunt days (\pmSD)
Successful hunts (2010-2018)	Erongo	1181	6.7 (\pm 4.9)
	Hardap	203	8.8 (\pm 6.2)
	Karas	53	8.8 (\pm 3.4)
	Kavango (Ukn)	0	0
	Kavango East	139	7.0 (\pm 3.9)
	Kavango West	21	5.3 (\pm 3.1)
	Khomas	1146	6.4 (\pm 4.4)
	Kunene	1223	7.3 (\pm 4.4)
	Omaheke	279	5.9 (\pm 4.1)
	Oshikoto	3	3.0
	Otjozondjupa	2303	6.3 (\pm 4.2)
	Zambezi	59	6.6 (\pm 5.1)
	Unknown	270	6.3 (\pm 3.9)
	Total	6880	6.6 (\pm4.4)

Table 8.7.3. The average number of hunt days per land use type for successful hunts between 2010 and 2018.

	Land use type	Number of hunt days	Average number of hunt days (\pm SD)
Successful hunts (2010-2018)	Freehold Conservancy	6	6.0
	Communal Conservancy	873	7.1 (\pm 4.5)
	Community Association	0	0.0
	National Park	178	6.6 (\pm 3.7)
	Freehold Farm	5569	6.5 (\pm 4.4)
	Unknown	254	7.5 (\pm 5.6)
	Total	6880	6.6 (\pm4.4)

Table 8.7.4. The average number of hunt days per year for successful and unsuccessful hunts between 2016 and 2018.

	Year	Number of hunt days	Average number of hunt days (\pm SD)
Successful Hunts (2016-2018)	2016	844	5.7 (\pm 3.8)
	2017	890	5.8 (\pm 3.8)
	2018	885	6.4 (\pm 4.6)
	Total	2619	6.0 (\pm4.1)
	Year	Number of hunt days	Average number of hunt days (\pm SD)
Unsuccessful Hunts (2016-2018)	2016	6056	13.1 (\pm 5.1)
	2017	6017	15.3 (\pm 30.5)
	2018	4433	11.5 (\pm 5.3)
	Total	16506	13.3 (\pm17.7)

Table 8.7.5. The average number of hunt days per region for successful and unsuccessful hunts between 2016 and 2018.

	Regions	Number of hunt days	Average number of hunt days (\pm SD)
Successful hunts (2016-2018)	Erongo	383	6.5 (\pm 4.2)
	Hardap	96	6.9 (\pm 4.3)
	Karas	32	8.0 (\pm 4.1)
	Kavango (Ukn)	0	0
	Kavango East	50	5.0 (\pm 3.7)
	Kavango West	8	4.0 (\pm 4.2)
	Khomas	483	5.8 (\pm 4.0)
	Kunene	457	6.3 (\pm 3.8)
	Omaheke	118	5.4 (\pm 4.2)
	Oshikoto	0	0

	Regions	Number of hunt days	Average number of hunt days (\pm SD)
	Otjozondjupa	927	5.9 (\pm 4.3)
	Zambezi	22	5.5 (\pm 3.8)
	Unknown	43	4.3 (\pm 2.8)
	Total	2619	6.0 (\pm4.1)
	Regions	Number of hunt days	Average number of hunt days (\pm SD)
Unsuccessful hunts (2016-2018)	Erongo	2548	13.1 (\pm 6.0)
	Hardap	382	12.7 (\pm 4.6)
	Karas	15	15.0
	Kavango East	380	16.5 (\pm 4.1)
	Kavango West	30	15.0 (\pm 1.4)
	Khomas	3659	12.0 (\pm 4.7)
	Kunene	3460	19.7 (\pm 44.8)
	Omaheke	672	12.2 (\pm 5.2)
	Oshikoto	152	16.5 (\pm 6.0)
	Otjozondjupa	4562	11.5 (\pm 5.2)
	Zambezi	139	15.4 (\pm 4.2)
	Unknown	507	13.3 (\pm 5.3)
	Total	16506	13.3 (\pm17.7)

Table 8.7.6. The average number of hunt days per land use type for successful and unsuccessful hunts between 2016 and 2018.

	Land use type	Number of hunt days	Average number of hunt days (\pm SD)
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Successful Hunts (2016-2018)	Freehold Conservancy	6	6
	Communal Conservancy	326	5.9 (± 4.0)
	Community Association	0	0.0
	National Park	66	4.7 (± 3.3)
	Freehold Farm	2199	6.1 (± 4.1)
	Unknown	22	3.7 (± 4.2)
	Total	2619	6.0 (± 4.1)
Unsuccessful Hunts (2016-2018)	Land use type	Number of hunt days	Average number of hunt days (\pmSD)
	Freehold Conservancy	39	13.0 (± 5.3)
	Communal Conservancy	1735	14.6 (± 4.3)
	Community Association	0	0
	National Park	155	14.1 (± 5.2)
	Freehold Farm	14493	13.2 (± 18.7)
	Unknown	84	14.0 (± 7.7)
	Total	16506	13.3 (± 17.7)