Assessing Signals of Mesopredator Release under Two Apex Predator Regimes in Tswalu Kalahari Reserve, South Africa

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DECLARATION

I declare that, “ASSESSING SIGNALS OF MESOPREDATOR RELEASE UNDER TWO APEX PREDATOR REGIMES ON TSWALU KALAHARI RESERVE” is my own work and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references.

I further declare that I have not previously submitted this work, or part of it, for examination at UNISA for another qualification or at any other higher education institution.

31 October 2023

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Summary

Predators have a considerable influence on the structure and functioning of the ecosystems in which they occur, however their numbers have declined, and continue to do so. The eradication of top-level predators has been shown to cause cascading effects on the dynamics of surrounding biological assemblages. Mesopredator Release (MR) is one of these effects. The mesopredator release hypothesis posits that in the absence of apex predators, mesopredators respond functionally by increasing their populations. The Tswalu Kalahari Reserve (TKR) provides a unique opportunity to test this hypothesis in the context of an arid savanna that is managed according to conservation principles and that supports a diversity of indigenous species. The TKR is composed of two component regions, Korannaberg and Lekgaba, separated by predator-proof fencing, under two different predator management regimes. At the time of my study Korannaberg supported wild dogs (*Lycaon pictus,* Temminck, 1820) and cheetah (*Acinonyx jubatus*, Schreber, 1775) as top predators (no lions), and Lekgaba supported lions (*Panthera leo,* Linnaeus, 1758) (no wild dogs or cheetah). For the purpose of my study, I considered both sections homogenous. Both areas support similar assemblages of mesopredators, and a variety of species that are considered prey for mesopredators. I used a quantitative field experiment to gather numerical and empirical data. In this comparative study, I selected sample sites within Korannaberg and Lekgaba using a randomisation process. I compared large predator, mesopredator, large and small ungulate, small mammal and avian relative abundance and diversity, between regions and among seasons. I assessed the relative abundance and diversity of mesopredators and ungulates using camera traps, small mammals using live-trapping transects and birds using modified point counts. I detected no significant differences in relative abundances of the mesopredators ($p > 0.05$), ungulates ($p > 0.05$), small mammals ($p > 0.05$), or birds ($p > 0.05$), between the two sections, nor differences in diversity either. My results support an increasing body of knowledge suggesting that mesopredator release may not be a ubiquitous phenomenon. Previous studies have evaluated evidence (or lack thereof) for the mesopredator relase hypothesis in the African and more specifically South African context, however, none have been conducted in the Kalahari. In the Eastern Cape of South Africa, mesopredator release was demonstrated through a higher relative abundance of black-backed jackals (*Lupulella mesomelas* (Schreber, 1775) in the absence of apex predators, however in the Karoo, the control of top predators did not have an effect on the rate of predation of ground-dwelling bird nests. I did, however, find substantial seasonal differences in abundance and diversity of small mammals, with a higher number of captures in the cold-dry season. Additionally, avian populations had a higher relative abundance in the cold-dry season, and a higher diversity in the hot-wet season. There were no seasonal differences in relative abundance or diversity of predators or ungulates (large or small).

Keywords

Mesopredator Release Hypothesis

Kalahari

Relative Abundance

Small Mammals

Capture-Mark-Recapture

Bird Point Counts

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CHAPTER 1 - INTRODUCTION

"You don't see sick animals in the wild. You don't see lame animals in the wild, and it's all because of the predator…"

-Tippi Hedren, American Actress

1.1 Study background

Predators have a considerable influence on the structure and functioning of the ecosystems in which they occur, yet globally predator numbers have declined, and continue to do so (Ripple *et al.* 2014; Wallach *et al.* 2015). The primary cause of this phenomenon is anthropogenic in origin, through land use changes and the resultant habitat modification or through extirpation in response to direct conflicts with humans (Beasley *et al.* 2013; Conner and Morris 2015; Wang *et al.* 2015). With an ever-expanding human population, the rate of loss of suitable habitats for top-predators increases and accelerates and this has raised awareness of the necessity for the conservation and protection of predators as a top environmental priority (Gese 2001). The extirpation of top-level predators has been shown to cause cascading effects on the dynamics of ecosystems and their associated food webs (Crooks and Soulé 1999; Berger *et al.* 2010; Beasley *et al.* 2013; Swanson *et al.* 2014). Past studies have shown that fragmented ecosystems often display shifts in predator prey compositions (Crooks and Soulé 1999; Prugh *et al.* 2009; Beasley *et al.* 2013). One of these shifts is known as the mesopredator release and has resulted in the proposal of the mesopredator release hypothesis. The mesopredator release hypothesis is defined as, *"in the absence of large predators, dominant predators, smaller omnivores and predators undergo population explosions, sometimes becoming four to 10 times more abundant than normal."* (Soulé *et al.* 1988). In broader terms, it can be defined as, the explosion of community populations of mesopredators, with repercussions of deleterious effect on their prey populations, once apex predators have been removed from a system (Prugh *et al.* 2009). Predator suppression with cascading trophic level effects has been recorded in over 60 different studies worldwide (Crooks and Soulé 1999; Glen and Dickman 2005; Gehrt and Prange 2007; Johnson *et al.* 2007; Lloyd 2007; Rayner *et al.* 2007; Ritchie and Johnson 2009; Brook *et al.* 2012; Beasley *et* *al.* 2013; Yarnell *et al.* 2013; Swanson *et al.* 2014; Newsome and Ripple 2015; Wang *et al.* 2015; Taylor *et al.* 2016; Gordon *et al.* 2017; Newsome *et al.* 2017; Tambling *et al.* 2018; Jachowski *et al.* 2020; Takimoto and Nishijima 2022) across terrestrial and aquatic systems.

The primary cause of mesopredator release is thought to be the alteration of land use for anthropogenic reasons, with the subsequent removal of large predators. (Beasley *et al.* 2011,2013). Therefore, human-altered ecosystems, like livestock and agricultural farming lands, have been shown to have a higher incidence of mesopredator release (Beasley *et al.* 2013) because the top predators have been removed from these systems. Top predators occupy the highest trophic level and influence all trophic levels beneath them through several direct and indirect means. These means can include direct predation, or indirect influence on spatial behaviour through fear (Palacios *et al*. 2016; O'Regan 2019). Humans are known to disrupt natural predator prey systems in many ways and have been known to inadvertently step into the apex predator role through the persecution of mesopredators i.e. farmers that persecute caracal (*Caracal caracal,*Schreber, 1776) or black-backed jackal (*Lupulella mesomelas* Schreber, 1775) for preying on their livestock (Berger *et al.* 2010; Tambling *et al.* 2018). The suppression caused by apex predators on mesopredators has competition at its core, and is displayed either through (1) direct aggressive interactions (Caro and Stoner 2003; Prugh *et al.* 2009) or (2) direct avoidance by mesopredators to reduce the possibility of encountering an apex predator (Durant 1998; Glen and Dickman 2005; Berger et al. 2010). The combination of the extinction of apex predators and the release of mesopredators has been identified as a cause for the decline and extinction of wild birds and small mammal populations across several systems worldwide (Crooks and Soulé 1999; Medina *et al.* 2011; Ripple *et al.* 2013; Taylor 2015; Wang *et al.* 2015).

Mesopredator feeding habits are important drivers of ecosystem functioning as they have been shown to regulate invertebrates, and smaller vertebrate species that apex predators do not regulate (Bagniewska and Kamler 2013; van de Ven *et al.* 2013). Additionally, mesopredators play other important roles in ecosystems, by protecting human health through regulating rodent populations and the diseases they carry (Ostfeld and Holt 2004), by controlling several types of pest species (Newsome 1990), seed dispersal (Jordano *et al.*2007), and by cleaning ecosystems through waste management by facultative scavenging (Ćirović *et al.* 2016).

1.2 The role top predators play

In the early 1900s British ecologist, Charles Elton, acknowledged the importance of apex predators in the mechanism of “top down” control of animals at lower trophic levels (Elton 1927). The Green World Hypothesis was built on Elton’s observation and it suggested that predators maintain global plant biomass by controlling herbivore numbers (Hairston *et al.* 1960; Mougi and Iwasa 2011). Predators are often misunderstood and they inspire a variety of attitudes and responses in humans which can range from fascination to irritation/ annoyance. Predators are seen by some to be problematic, costly, and dangerous to humans, i.e. affecting the livelihoods of surrounding humans through preying on livestock, pets or even children (Andersen *et al*. 2005; Bothma 2012; Katel *et al.* 2014; Humphries *et al.* 2015). However, they are also considered potentially valuable in the tourism industry (Macdonald and Sillero-Zubiri 2002) and play an influential role in regulating ecosystem functioning (Macdonald and Sillero-Zubiri 2002; Ripple *et al.* 2014). Over time, humans have caused the drastic decline of terrestrial mammalian predators (Ripple *et al.* 2014). More predators are threatened (26.9%) with extinction and declining in species (48.3%) than any other mammal group worldwide (Fernández-Sepúlveda and Martin 2022). Human-carnivore conflicts are a concern and can surface for several reasons:

- Carnivores and humans have similar needs, a protein-rich diet and large home ranges (Treves and Karanth 2003; Khan 2004; Andersen *et al.* 2005).
- Occasionally opportunity arises for a predator to hunt domesticated species like sheep, cows and goats, which is considered a global challenge to humans who live in or near rural areas (Treves and Karanth 2003; Katel *et al.* 2014; Du Plessis *et al.* 2015; Humphries *et al.* 2015).
- Individual carnivores have been known to attack and kill humans (Treves and Karanth 2003; Khan 2004; Andersen *et al.* 2005).

Due to increasing demographic, economic and social pressures these challenges are growing in frequency and financial costs. These challenges are often combated using lethal methods of predator control (Treves and Karanth 2003; Bothma 2012; Du Plessis *et al.* 2015).

Past and present extinction rates of top predators from of outside of protected areas (i.e. reserves, parks, or sanctuaries) have caused dramatic shifts in ecological communities worldwide (Berger 1999). Examples of this include; the snow leopard, *Uncia uncia* (Schreber, 1775), which is found in the mountainous ranges of South and Central Asia (Fox 1989). Despite snow leopard's broad geographical distribution, this species is categorised as vulnerable and decreasing (IUCN 2017). The decline of this species is due mostly to retaliatory killings by farmers, hunting of snow leopards for meat and fur, and a decline in prey species populations as a result of subsistence hunting (Mishra *et al.* 2003). The Bengal tiger, *Panthera tigris tigris* (Linnaeus, 1758), is the largest of the wild felids occurring in India, Bangladesh, Nepal and Bhutan (Seidensticker *et al.* 2010). They are globally categorised as endangered (IUCN 2017) and still decreasing. Globally there are more tigers found in captivity than in the wild. The two primary reasons for killing Bengal tigers are retaliatory killings by local human inhabitants for attacks on humans or cattle and poaching for their pelts and body parts (Khan 2004). During the 1800's and early 1900's broad scale trapping of native predators including jaguars (*Panthera onca* Linnaeus, 1758), cougars (*Puma concolor* Linnaeus, 1771), grey wolves (*Canis lupus* Linnaeus, 1758) and grizzly bears (*Ursus arctos* Linnaeus, 1758) took place in North America (Beschta and Ripple 2009). Much of this killing can be attributed to Euro-Americans colonising the interior of North America. This resulted in the elimination of between 95 and 99% of the populations of these predators in North America (Berger 1999; Berger *et al.* 2008). The disappearance of predatory species is thought to have resulted in an increase in ungulate density and consequently overgrazing of endemic plant species (Berger 1999; Berger *et al.* 2008). The high density of ungulates has been destructive to native plant communities and has resulted in ecosystem changes (Beschta and Ripple 2009). In southern Greater Yellowstone National Park in North America, the removal of wolves resulted in the release of coyotes (*Canis latrans* Say, 1823). This precipitated an increase in coyote predation on pronghorn antelope (*Antilocapra americana* Ord, 1815) fawns, (Berger *et al.* 2008). African lions are considered the kingpin tourist attraction on the African continent, yet they are listed as vulnerable and decreasing on the IUCN Red Data list and are becoming increasingly rare outside of protected areas (Macdonald and Sillero-Zubiri 2002). The persecution of apex predators by humans alters the structure of animal communities and perturbs the ecological functioning thereof (Ripple *et al.* 2014). Futhermore, this has been identified as a major contributor to the loss of biodiversity globally (Gordon *et al.* 2017).

1.3 Suppression, competition and fear

Trophic interactions between predators are complex and often lead to complicated system dynamics (Thompson and Geese 2007). Predators that occur in sympatry regularly encounter interspecific competition, i.e. mesopredators might suffer a reduction in fitness due to competition for resources from larger top predators (Creel and Creel 1996; Brook *et al.* 2012). The overlapping of niches escalates interactions between predators and intraguild predation, competitive killing and exploitative or interference competition become common (Ritchie and Johnson 2009; Haswell *et al*. 2018). Interference competition takes place in the form of harassment, killing and occasionally consuming of smaller predators by the apex predators. Exploitative competition is where predatory species compete for the same prey species (Polis and Holt 1992; Brook *et al.* 2012). The suppression of mesopredators by apex predators is a common occurrence and these killings can take place in two manners. The first is where the top predator kills the mesopredator but does not eat it (interspecific killing), and the second is where apex predators kill and consume the mesopredator (intraguild predation) (Ritchie and Johnson 2009; Newsome *et al.* 2017; O'Regan 2019). Intraguild predation can reduce competition between predators while also providing energetic gain for the top predator (Polis and Holt 1992; O'Regan 2019). For example; tigers restrict wolves (Miquelle *et al.* 2007), wolves keep out coyotes (Ripple *et al.* 2013), coyotes regulate gray foxes (*Urocyon cinereoargenteus* Schreber, 1775) (Crooks and Soulé 1999), red foxes (*Vulpes vulpes* Linnaeus, 1758) kill feral cats (*Felis catus* Linnaeus, 1758) (Glen and Dickman 2005), cats control rats

(*Rattus exulans* Peale, 1848) (Rayner *et al.* 2007), and rats displace mice (*Mus musculus* Linnaeus, 1758) (Wanless *et al.* 2007).

Competition, between predators, plays an important role in the balance of predator to mesopredator to prey, as well as the homeostasis of ecosystems through top predators exerting non-consumptive effects on mesopredators which mitigates their impact on prey (Polis and Holt 1992; Wallach *et al.* 2015; Palacios *et al.* 2016). It also directly affects mesopredators by reducing their abundance, and altering their behaviour to allow them to avoid encounters with top predators (Polis and Holt 1992; Brook *et al.* 2012). The landscape of fear is defined as the subsequent behaviour displayed by species, in this case the mesopredators, which perceive a possible risk of predation (Laundré *et al*. 2010). This behaviour changes the way the mesopredators utilize their environments in order for them to minimize their risk of possible predation (O'Regan 2019; Curveira-Santos *et al.* 2021). This means that their perspective is one of a dangerous habitat scattered with patches of safety resulting in a restriction of habitat use by the mesopredators (Ritchie and Johnson 2009). This behaviour ultimately translates to impacts on a larger demographic scale not only for the mesopredators but also to their prey species (Ritchie and Johnson 2009; Palacios *et al.* 2016). One of the most well-known examples is that of the avoidance of lions and spotted hyaenas (*Crocuta crocuta* Erxleben, 1777) by cheetahs (Durant 1998) and wild dogs (Creel and Creel 1996). Both species actively avoid areas occupied by lion and spotted hyaena with the intention of avoiding competition for prey (their prey being stolen) and even direct predation (Creel and Creel 1996; Durant 1998; Ritchie and Johnson 2009). Where lions occur, they are thought to have such an impact in relation to the entire mesopredator assemblage in South Africa (Curveira-Santos *et al.* 2021).

1.4 Mesopredators and their roles

Mesopredators are mid-ranking mammalian predators. This classification can be based on size (mesopredators weigh 15 kg on average but weigh no more than 30 kg) (Prugh *et al.* 2009; van de Ven *et al.* 2013; Ripple *et al*. 2014), or they can be classified as species that are subject to top-down pressures from larger predators

(Prugh *et al.* 2009), but ultimately the classification of a mesopredator is context dependent (Crooks and Soulé 1999). A mesopredator can fill the top predator tier, in the context of an ecosystem, if no higher order predators remain in that system (Prugh *et al.* 2009; Roemer *et al.* 2009; Tambling *et al.* 2018). This is likely only to take place in disturbed systems, and because of the size of mesopredators their impact of prey species does not cover the relatively large prey species (Tambling *et al.* 2018). For example, in the Yellowstone ecosystem, coyotes are below wolves as mesopredators in the trophic system. However, in other areas of the United States, where larger predators have been extirpated, coyotes are considered the top predators (Crooks and Soulé 1999; Roemer *et al.* 2009). Similarly, feral cats function as mesopredators in many continental ecosystems, however they are top predators on several islands (Crooks and Soulé 1999). Mesopredators are "opportunistic feeders" with varied adaptable diets. Therefore, mesopredators are capable of living in an array of different habitats, including those that larger predators cannot occupy, in some cases within close proximity to human establishments (Roemer *et al.* 2009; van de Ven *et al.* 2013; Tambling *et al.* 2018). Their diets may consist of small to medium-sized mammals like hares and rodents, as well as small ungulates, reptiles, birds, bird eggs, carrion, invertebrates, certain plants, and even fruits and seeds (Sillero-Zubiri *et al.* 2004; Mills and Bester 2005; Tambling *et al.* 2018). Their variable and adaptable diets result in mesopredators influencing a wide diversity of smaller species (Bagniewska and Kamler 2013) which apex predators seldom, if ever, prey on and therefore do not regulate or impact directly. The ecological effect of mesopredators may appear relatively minor if one considers individual animals' physical size, however, mesopredators influence ecosystems in ways that apex predators cannot; i.e. like dispersing seeds (Jordano *et al.* 2007; Roemer *et al.* 2009).Other ways mesopredators ecologically regulate ecosystems by assisting in the removal of biological waste, as many are facultative scavengers (Ćirović *et al.* 2016), and they can even indirectly promote human health by decreasing the number of disease carrying rodents (Otsfeld and Holt 2004).

1.5 Trophic cascading

Trophic cascading is defined as "*an ecological phenomenon triggered by the addition or removal of top predators and involving reciprocal changes in the relative populations of predator and prey through a food chain, which often results in dramatic changes in ecosystem structure and nutrient cycling*" (Carpenter 2020). The concept behind the mesopredator release hypothesis can be traced back to Hairston *et al.* (1960), who developed the tri-trophic Green World Hypothesis. This three-tiered system placed predators at the top of the trophic hierarchy, followed by herbivores and then plants. It was only later labelled by Paine (1969) as trophic cascading when he observed that the species composition and physical appearance of ecosystems could be substantially altered by the presence or absence of a single species. He classified these species as the keystones of the community, as the stability of the communities are dependent on the activities and abundances of such species (Paine 1969). Conceptually, the building blocks of an ecosystem can be presented as a three-tier pyramid with the plants at the bottom, herbivorous species that consume the plants in the middle tier, and the predators that consume the herbivores, on the top tier of the food pyramid (Prugh *et al.* 2009). However, natural trophic systems are not this simple and linear. They can be relatively complex, and this often makes the predicting the responses of wildlife to human interventions very difficult (Thompson and Gese 2007; Prugh *et al.* 2009). Additionally, humans have evolved into the omnipotent apex predator on the planet with no enemies and free of any top-down influences (Wallach *et al.* 2015). Apex predators are carnivorous and are found at the top levels of food chains, and therefore are in direct competition with humans (Macdonald and Sillero-Zubiri 2002).

Trophic cascading has crudely been described as the indirect effects of predators on plants mediated by herbivores (Schmitz 2007; Ripple *et al.* 2016). Trophic cascades can be weakened or amplified by behavioural responses of prey and predators. Species at intermediate trophic levels may avoid top predators, while top predators' behaviour affects the foraging sites of prey or their preferences which alter trophic cascade relationships (Polis and Holt 1992; Carpenter *et al.* 2010; Palacios *et al.* 2016). For example cascading effects are taking place in Europe where the grey wolf, as an apex predator, supresses golden jackal (*Canis* *aureus* Linnaeus, 1758.) and red fox, however the persecution of the wolves has triggered a range expansion in golden jackals and red foxes (Krofel *et al.* 2017; Torretta *et al.* 2021; Tsunoda 2022). Regardless of extent of the imbalance of predator prey relationships, and whether that imbalance is of anthropogenic origin or not, when predators are removed from a community there is an obvious shift in the behaviour of species within that community, particularly the prey species (Berger 1999; Berger *et al.* 2008; Berger *et al* 2010). Understanding these relationships is scientifically important for several reasons:

- By understanding the importance of guild dynamics among various predators and by understanding that certain carnivores can be adversely influenced by other guild members is important in the conservation of these predators (Ritchie and Johnson 2009).
- The management, and protection or restoration of natural, undisturbed communities requires knowledge of the ecological dynamics of the system (Berger 1999; Treves and Karanth 2003).
- Programmes that aim to restore biological communities, with the introduction of predators, will be improved by understanding how these predators interact with other community members (Berger 1999).

Therefore, by understanding the indirect effects of predators on food webs dynamics, the ecological predictions of impacts surrounding predator loss, can be determined (Ripple *et al.* 2014). Apex predators play multiple roles in ecosystems. These roles include the controlling of large herbivores through predation and the ecology of fear (Roemer *et al.* 2009; Ripple *et al.* 2014), and influencing mesopredators through intraguild interactions. In this manner ecosystems are structured through numerous food-web pathways (Ripple *et al.* 2014). Predator populations are influenced by both top-down and bottom-up factors like habitat structure and complexity, and food availability (Woodgate 2014). All of which are known to alter the interactions among predators as well (Ritchie and Johnson 2009; Woodgate 2014).

1.6 The mesopredator release hypothesis

The term, “mesopredator release” was first coined by Crooks and Soulé (1999) who described how the disappearance of carnivores lead to an increased number of smaller carnivores which are the principle predators of smaller mammals, reptiles, invertebrates and birds. These smaller (middle-tiered) predators were called mesopredators because, unlike apex predators, they are not safe from predation. Mesopredators experience bottom-up influences from herbivores and top-down influences from apex predators and human persecution, compared with apex predators that are subject to bottom-up influences and only have human persecution as a top-down influence. Therefore, apex predator abundances are “self-regulated” whereas mesopredators are considered to be “extrinsically-regulated” (Wallach *et al.* 2015).

The mesopredator release hypothesis predicts that at decreased number of apex predators has as an outcome the increase in abundance and impact of mesopredators (Prugh *et al.* 2009). This hypothesis has been observed on all continents except Antarctica (Prugh *et al.* 2009). A study carried out in southern Australia found that presence of dingoes, *Canis familiaris* (Jackson *et al.* 2017) benefited small prey species including the little button-quail, *Turnix velox* (Gould 1841) by reducing the abundance of mesopredators, red fox and feral cats. The study also suggested that the active management of dingoes could be included in broad-scale biodiversity conservation programs as a strategy to alleviate the predatory impacts of mesopredators (Gordon *et al.* 2017). Across sub-Saharan Africa another mesopredator, the olive baboon (*Papio anubis* Lesson, 1827), has increased in abundance in areas where lion and leopard, *Panthera pardus* (Linnaeus, 1758), populations have been removed (Berger *et al.* 2010). A study conducted in South Africa by Yarnell *et al.* (2013) showed that the presence of apex/top predators, like lion and wild dog, altered the feeding ecology of mesopredators like black-backed jackal (*Lupulella mesomelas* Schreber, 1775) and brown hyaena (*Hyaena brunnea* Thunberg, 1820) by reducing levels of interspecific competition for food between mesopredators and by providing additional scavenging opportunities. Even though the mesopredator release may result in negative cascading effects on surrounding communities, this phenomenon is fundamentally an intraguild interaction within the community (Berger *et al.* 2008).

When comparing the impact of mesopredator release between intact versus modified environments, it can be noted that mesopredator release has been posited in relation to several predator guilds across several ecosystems, both intact (Prugh *et al*. 2009; Brashares *et al.* 2010; Newsome and Ripple 2015) and modified/ human-dominated (Elmhagen *et al.* 2007; Taylor *et al.* 2016; Krofel *et al.* 2017).

1.7 South african context

In Africa, lions are the apex predators of any animal community in which they occur. They are often the focus of human carnivore conflict as domestic livestock may comprise up to one third of their diet (Macdonald and Sillero-Zubiri 2002; Dickman *et al.* 2013). In South Africa, approximately 69% (839 281 km^2) of the total land area is used for domestic livestock farming and game ranching (Thorn *et al.* 2012). The current distribution of lions is approximately 17% of its historical range or they can be found in about 25% of the savanna biome in Africa (Ripple *et al*. 2014; Bauer *et al.* 2016). This is because of land use changes that have taken place during agriculture expansion and commercialised livestock farming and game ranching (James 2014; Ripple *et al*. 2014). The decline of lions both in terms of numbers and occupancy is thought to have led to an increase in mesopredators like cheetah, wild dog, black-backed jackal and caracal, *Caracal caracal* (Schreber, 1776) (Creel and Creel 1996; Vanak *et al.* 2013; Swanson *et al.* 2014; Swanson *et al.* 2016). Black-backed jackals are presently thought to be responsible for the cause of most livestock predation events in South Africa (Thorn *et al.* 2013; Minnie *et al.* 2016). In southern Africa and in the Serengeti Swanson *et al*. (2014) showed that lion contributed to the local extinction and continued absence of wild dogs in both of these areas, whereas the cheetah populations remained stable despite the increased lion population. Between wild dogs and cheetahs exists an interspecific competition relationship whereby both are found in open savanna ecosystems and both live in areas supporting high prey densities, which may intensify competition over carcasses (Caro and Stoner 2003).

Human-wildlife conflict is so ubiquitous in South Africa that legislation has been drafted to address it (Thorn *et al.* 2013). In 2010 an amendment was made to the National Environmental Management Biodiversity Act, Act 10 of 2004, for the management of "damage-causing" animals. The aim of the amended legislation is to prevent or mitigate recurring damage by these species. The remedial actions that may be taken include live capture (for permanent removal from the system by euthanasia, trans/relocation in the wild, or relocation to a scientific/ conservation facility for the purpose of research), or the animal/s may be killed (only as a last resort if no human error or negligence has been found) (Department of Environmental Affairs 2010).

In the Kalahari context, the predator mesopredator ranking is as follows; the lion is the apex predator and other predators are likely subject to top-down pressure as a result. Although spotted hyaena were not on TKR at the time of my study they are present now and are also ranked as an apex predator at the top of the food chain (Periquet *et al.* 2015). In the absence of lions, wild dogs, cheetahs, leopards and brown hyaena may be considered top predators. The mesopredator guild comprises: black-backed jackals, caracal, Cape fox (*Vulpes chama* A. Smith, 1833), bat-eared fox *(Otocyon megalotis* Desmarest, 1822), and African wild cat (*Felis silvestris cafra* Forster, 1780), with the additional small predators below them (MacDonald and Sillero-Zubiri 2002; Roxburgh 2008; Swanson *et al.* 2014; Wallach *et al.* 2015).

1.8 Research problem

The leading worldwide threat to carnivores is anthropogenic in origin (Blaum *et al.* 2009; Bergman *et al.* 2013; Thorn *et al.* 2013). Past research has shown that the conservation of predators is important because their presence may influence entire ecosystems through top-down mechanisms (Blaum *et al.* 2009). Consequences to changes in predator community structure, or losses of specific carnivores, are known to alter community and intraguild relationships resulting in changes in mesopredator assemblages and trophic cascading (Blaum *et al.* 2009). Cascading effects because of changes in top and mesopredator guilds alter small mammal and bird community dynamics (Crooks and Soulé 1999; Lloyd 2007;

Brook *et al.* 2012; Gordon *et al.* 2017). The TKR provides a unique opportunity to test the predictions of the mesopredator release hypothesis as two distinct large (includes apex and top) predator regimes are maintained there.

1.9 Motivation

Trophic cascading plays a vital role in the intra-guild structure of communities and populations within ecosystems, and the extirpation of predators, on any level, can cause habitat degradation and species loss across several trophic levels (Berger *et al.* 2010). The Green World Hypothesis predicts that carnivores are paramount in maintaining green ecosystems, with an abundance of vegetation, through the limitation of herbivores (Hairston *et al.* 1960; Mougi and Iwasa 2011). Theoretically, a simplified food chain comprises predators at the top, followed by their prey and the plants that they consume at the bottom (Hairston *et al.* 1960; Berger *et al.* 2010; Ripple *et al.* 2016). However, this is an oversimplification and many food chains are complex and have more than three levels (Thompson and Gese 2007). Larger predators are known to capture and consume larger prey than smaller predators (Lesmeister *et al.* 2015). The classification of apex, top, meso- and small predators is centred on several factors with the fundamental factor being size. Apex predators are broadly defined as carnivorous species that occupy the top trophic position and are generally large-bodied and specialized hunters (Ritchie and Johnson 2009). The larger the predator, the safer from predation, the higher up on the food web (Wallach *et al.* 2015). Apex predators, on the topof the food chain, generally have no natural predators, however, have a much higher risk of being hunted or affected by anthropogenic pressures (Schutte *et al.* 2013; Wallach *et al.* 2015).

The predators on top of the trophic pyramids maintan their populations, generally giving birth to fewer offspring, regardless of bottom-up pressures i.e. they can self-regulate (Wallach *et al.* 2015). For example, a female brown bear (*Ursus arctos* Linnaeus, 1758) is less likely to produce cubs if her nearest female brown bear neighbour already has cubs (Ordiz *et al.* 2008) and pregnant lionesses (*Panthera leo* Linnaeus, 1758) will either have a miscarriage or lose their young shortly after birth when a new male takes over a pride (Mills and Bester 2005). Predators at the top of the trophic pyramid can include apex and top predators (which are one-tier down from apex predators, but are still strong drivers of top-down pressures on their surroundings and can self-regulate) (Ritchie and Johnson 2009). Whereas mesopredators occupy the trophic level below apex predators, are generally smaller in stature and have a higher reproduction rate that is extrinsically regulated (Ritchie and Johnson 2009; Wallach *et al.* 2015). There are instances where in the absence of apex/ top predators, mesopredators step into the apex predator trophic position, however they will ultimately not functionally replace an apex predator (Mills and Bester 2005; Medina *et al.* 2011; Wallach *et al.* 2015). Ultimately, ecologically ranking predators is context dependent (Wallach *et al.* 2015). If we are to better conserve natural community dynamics, it is critical that we understand the relationships that exist among and between these species (Prugh *et al.* 2009). Therefore, by investigating the response of mesopredators to different apex predator regimes, information can be garnered in relation to many drivers of an ecological assemblage. This includes what the repercussions may be if apex predators are removed from a community or what were to unfold if apex predators were included into a system where they previously did not occur. Information regarding community structure and population dynamics is important in conservation biology and wildlife management. Systems can be better conserved when we understand the relationships between these different groups; humans and predator, predators and predators, predators and small mammals, predators and birds and small mammals and birds.

The aim of my study is to determine if the response of mesopredators (e.g. black-backed jackal, caracal), are the same irrespective of the presence of different apex/ top predators like lion, cheetah (*Acinonyx jubatus* Schreber, 1775) and wild dog (*Lycaon pictus* Temminck, 1820) in a Kalahari system, on Tswalu Kalahari Reserve (TKR).

The main objectives of my study were to:

1. Assess whether I could detect the cascading effects or signals that are traditionally associated with mesopredator release (increase in mesopredators, decrease in small mammals and birds) on TKR in the presence of different apex predators.

2. To investigate whether the relative abundance and diversity of mesopredator populations are altered based on the presence of different apex predators.
3. Additionally, I wanted to determine whether small mammal relative abundance and diversity differed in the presence of different apex predators.
4. Lastly, I wanted to establish whether relative abundance and diversity of avian populations differed in the presence of different apex predators.

1.10 Dissertation outline

- Chapter 1 introduces how the extirpation of top predators affects predator and prey assemblages below them on the food chain. This includes the mesopredator release hypothesis and how mesopredators and their prey species are influenced by different large predator regimes. This chapter also includes the aims, objectives and problem my study is addressing.
- Chapter 2 describes where my study took place and the biotic and abiotic factors associated with the Kalahari and more specifically the TKR.
- Chapter 3 is a description of the materials and methods that I used to achieve this study which took place using three approaches: 1) camera trapping, 2) small-mammal live-trapping and 3) avian point counts.
- Chapter 4 presents the results that I obtained during this study.
- Chapter 5 provides a discussion and explanation of my interpretation of the results and concluding remarks about the study.
- References – provides an alphabetical list of the references that I cited in my dissertation.
- Appendices includes all additional tables of data from the result chapter and examples of sampling sheets that I used in the field for my study.

CHAPTER 2 – STUDY SITE

“Mother Nature is a great healer. All she needs is time and space. Fortunately there are plenty of both on Tswalu.”

- Gus van Dyk, former General Manager, Tswalu Kalahari Reserve

2.1 Tswalu Kalahari Reserve

The TKR is situated in the north-eastern Northern Cape Province (27°2031’ S 22°4673’ E) in South Africa. It is a 114 268 ha privately owned reserve (Figure 1). The reserve is situated between the two small towns; Hotazel and Van Zylsrus (van Rooyen and van Rooyen 2017). The origin of TKR began in 1994 when Stephen Boler began the process of converting what were historically Kalahari livestock farms into a conservation area through a process of ecological restoration management (Roxburgh 2008). After Stephen Boler passed away the Oppenheimer family purchased the nascent reserve and continued to promote his vision and amalgamated more than 40 historic farming properties to create the conservation area that is the TKR. This has been achieved through the removal of all redundant infrastructure, internal fencing and livestock pens, and by restocking the property with indigenous endemic large mammals (Davis *et al.* 2010).

The landscape of TKR is characterized by the Korannaberg Mountains surrounded by sandy plains and dunes with dune valleys (van Rooyen and van Rooyen 2017). The Korannaberg Mountains extend from north to south, through the middle of the reserve. These mountains are high and often very steeply sloped (van Rooyen and van Rooyen 2017). The altitude ranges from 1020 m (Blou Krans) to 1580 m above sea level at the highest peak (Tokura 2016).

The TKR is divided into three distinctive wildlife management sections: the high value species breeding camp in the north-western corner where roan (*Hippotragus equinus* E. Geoffroy Sanit-Hilaire 1803) and sable antelope (*Hippotragus niger* Harris 1838) are bred (2 268 ha): the north-eastern section is known as Lekgaba, where a population of 20 lions, *Panthera leo* (Linnaeus, 1758) are present (18 386 ha): and the remainder is known as Korannaberg where lions are absent but other large predators, like cheetah, *Acinonyx jubatus* (Schreber, 1775) (N ~ 10) and wild dogs, *Lycaon pictus* (Temminck, 1820) (N ~ 14), are present (93 614 ha).

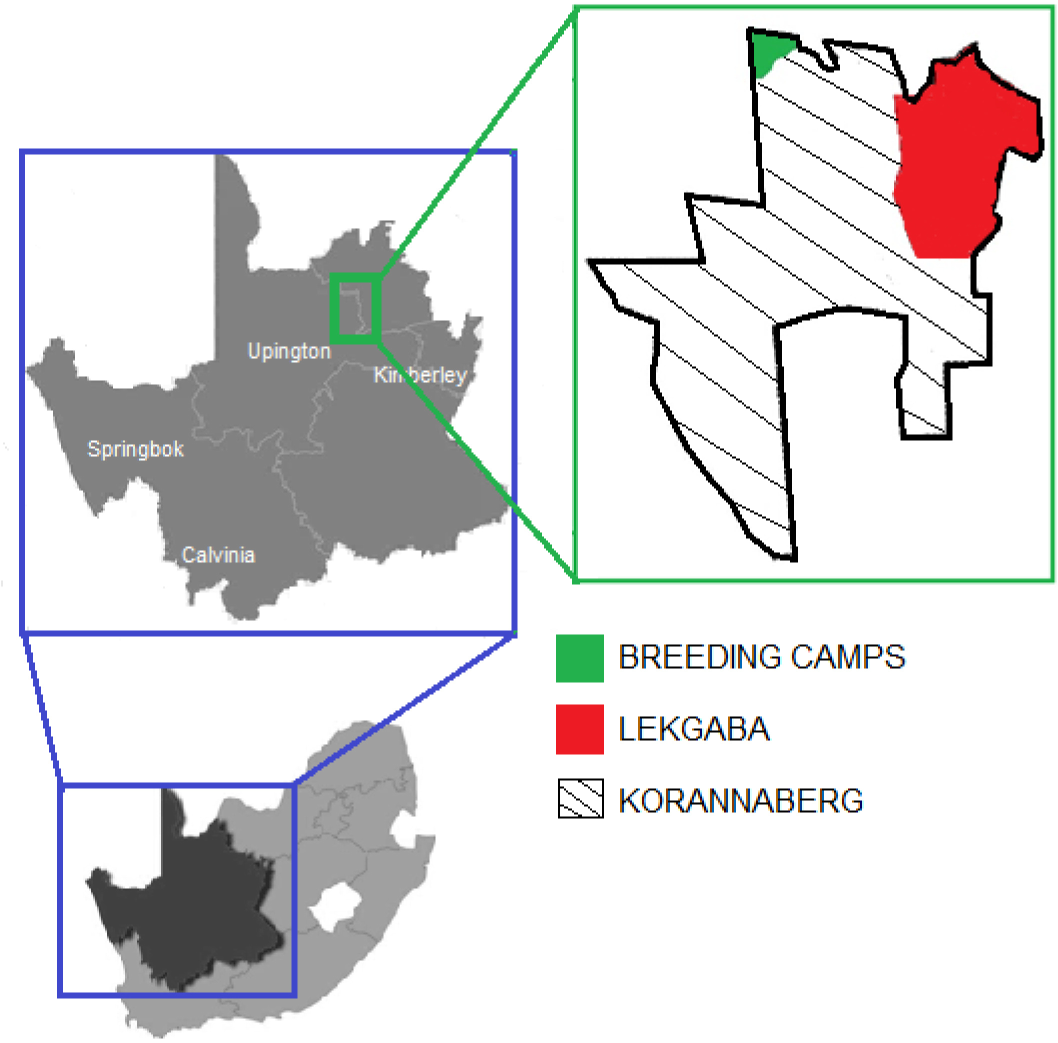

Figure 1: Location of Tswalu Kalahari Reserve in the Northern Cape of South Africa and how the reserve is divided between 3 distinctive wildlife management areas.

2.2 Climate

The TKR falls within the savanna biome in the north of the Northern Cape (Rutherford *et al.* 2006).The savanna biome encompasses ca. 33 % of South Africa’s vegetation and occupies most of the Northern Cape. The Savanna biomes is characterised by its wet summers and dry winters. This area has a relatively low incidence of frost during the winter periods. Due to the relatively low altitudes of the South African savanna biome, temperatures tend to be higher than those of the surrounding grasslands. The mean summer temperature in TKR is ~26°C, with the range from 16-32°C. In winter temperatures are more variable with maxima generally above 10°C and minima lower than 0°C (Figure 2). It annually receives approximately 318 mm of rainfall, ranging between 175-325 mm (Davis *et al.* 2010) (Figure 3). The Kalahari is subject to three distinct seasons; the hot-wet season (January to April), the cold-dry season (May to August) and the hot-dry season (September to December) (Leistner 1967; van Rooyen and van Rooyen 1998). Each season lasts approximately four months (Leistner 1967; van Rooyen and van Rooyen 1998). The mean annual rainfall on TKR is 325mm, which usually falls between December and April, i.e. the hot-wet season (van Rooyen and van Rooyen 2017). The driest months are from June to August during which ≤ 5mm of rain per month is recorded (van Rooyen and van Rooyen 2017).

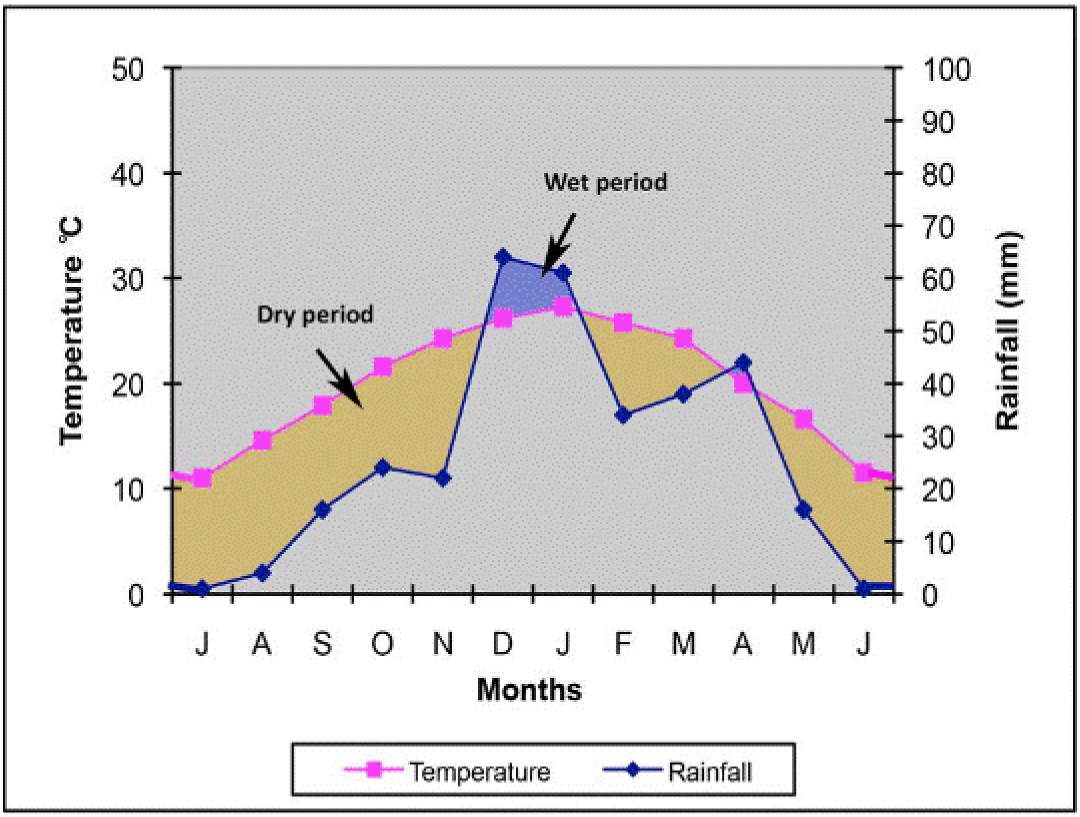

Figure 2: Climatograph for the Tswalu Kalahari Reserve region (Tswalu Kalahari Reserve rainfall data, Van Zylsrus temperature data; Weather Bureau 1988) (van Rooyen and van Rooyen 2017).

2.3 Geology

The savanna biomes' geology is characterized as Kaapvaal Craton which incorporates an array of sedimentary basins and igneous intrusions (Rutherford *et* *al.* 2006). The geology of TKR is comprised of the uppermost layer of the Kalahari Group sediments – Aeolian soils of the Gordonia Formation. The other principle rock types in the area include quartzite, slate, dolomite, jasper, subgraywacke, gravel, limestone and silcrete (van Rooyen and van Rooyen 2017). The soil found within the area is characterised as well-drained, red, sandy soil with a high base status (Rutherford *et al.* 2006). However, the southern Kalahari sands have a deficit in important nutrients including nitrogen, phosphorus, calcium, sodium, copper and zinc which is why TKR management annually provides 25 kg blocks of mineral licks, evenly distributed across the reserve to supplement wildlife diet (Abraham *et al.* 2021).

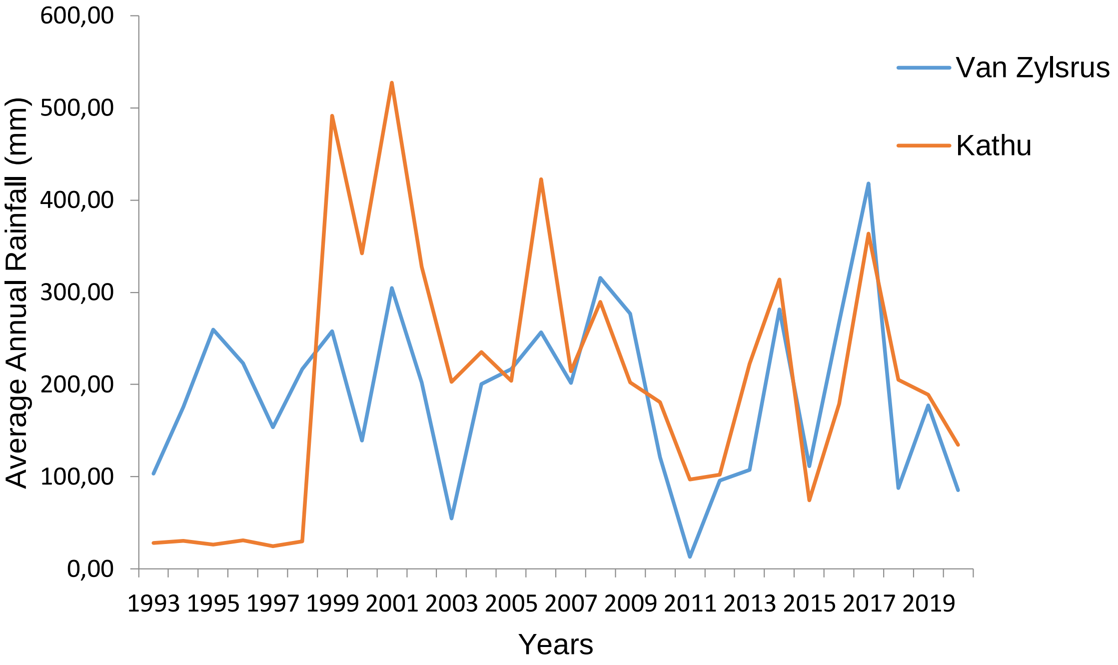

Figure 3: Average annual rainfall (mm) that fell in the proximity of the Tswalu Kalahari Reserve and was logged at Van Zylsrus to the north and Kathu to the south, from 1993 to 2020.

2.4 Flora and fauna

Savanna vegetation consists of an herbaceous layer dominated by grass species with a dispersed open tree layer. This vegetation type is more commonly known as the “Bushveld” in South Africa. Due to its naturally rich mammalian fauna, much of the “Bushveld” is used for wildlife ranching (Rutherford *et al.* 2006*a*). The vegetation on TKR can be classified into 5 different bioregions (Figure 4) namely the Koranna-Langeberg Mountain Bushveld, Gordonia Duneveld, Gordonia Plains Shrubveld, Olifantshoek Plains Thornveld and Kathu Bushveld (Tokura *et al.* 2018).

The position of TKR is within the Eastern Kalahari Bioregion, with the northern region lying on a small outlier patch of the Kalahari Duneveld Bioregion. Mountain Bushveld with Gordonia Duneveld to the north, Gordonia Plains Shrubland to the west, Olifantshoek Plains Thornveld close to the mountains both to the east and west, and Kathu Bushveld to the east (Rutherford *et al.* 2006; Davis et al. 2010; van Rooyen and van Rooyen 2017).

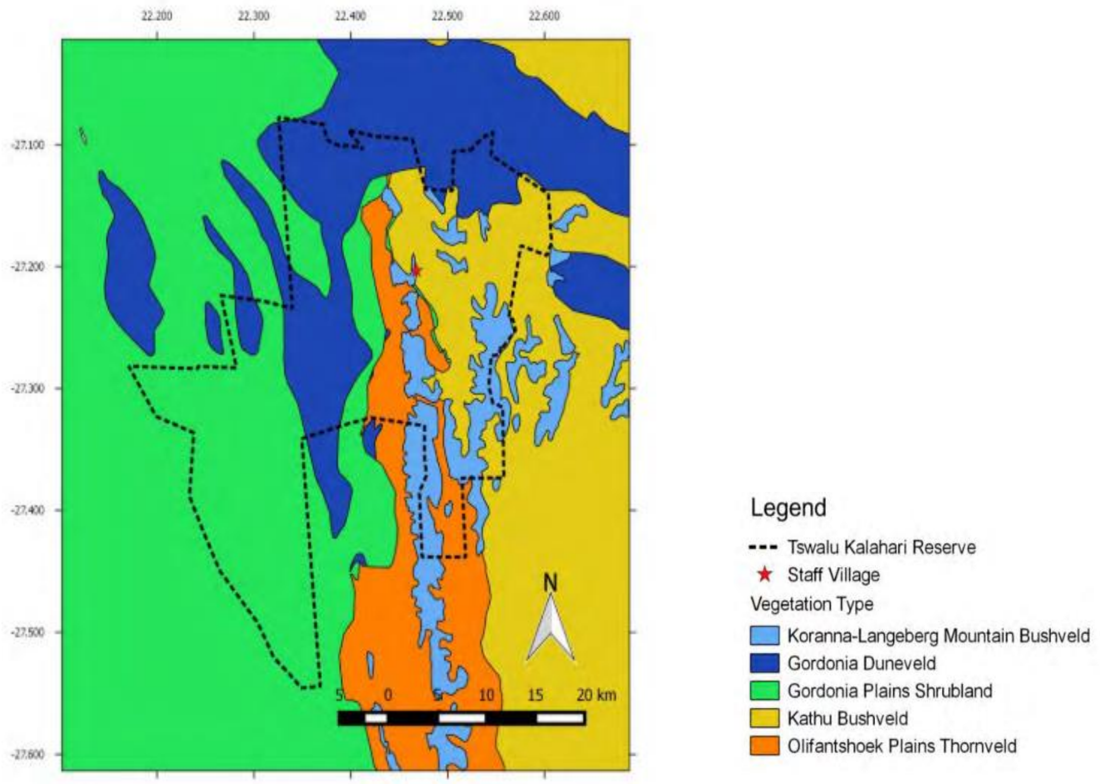

Figure 4: Vegetation bioregions of Tswalu Kalahari Reserve (Tokura 2016).

Three hundred and ninety-one species of indigenous plant species and 17 alien invasive plant species have been recorded on TKR. The plants contribute to 10 different plant communities that have been identified on the TKR (van Rooyen and van Rooyen 2017). The veld condition at TKR is highly dependent on rainfall and grazing pressure. Rainfall in this region is extremely variable (Figure 3). In 2017, 88% of the veld on the TKR was classified as being in poor condition. At that time the vegetation on Lekgaba had a higher percentage grass cover and veld condition score than Korannaberg

The TKR supports populations of approximately 80 different mammal species on the two separate wildlife management sections; the eastern part (Lekgaba – with lions present) and the western and central parts (Korannaberg – lions absent with wild dogs and cheetah as top predators) (Davis *et al.* 2010). Both sections support similar assemblages of ungulates (gemsbok, *Oryx gazella* (Linnaeus, 1758), springbok, *Antidorcas marsupialis* (Zimmermann, 1780), eland, *Taurotragus oryx* (Pallas, 1766), red hartebeest, *Alcelaphus caama* (É. Geoffroy Saint-Hilaire, 1803), zebra, *Equus burchellii* (Gray, 1824), and giraffe, *Giraffa camelopardalis* (Linnaeus, 1758)), predators (black-backed jackal, *Lupulella mesomelas* (Schreber, 1775) African wild cat, *Felis lybica* (Forster, 1780) bat-eared fox, *Otocyon megalotis* (Desmarest, 1822) aardwolf, *Proteles cristatus* (Sparrman, 1783), caracal, *Caracal caracal* (Schreber, 1776) honey badger *Mellivora capensis* (Schreber, 1776), and small-spotted genet *Genetta genetta* (Linnaeus, 1758)), and over 240 species of birds (Wetu 2013). Even though both sections contain similar ungulate assemblages, certain species that are present on Korannaberg are absent from Lekgaba, and these include buffalo *Syncerus caffer* (Sparrman, 1779), mountain reedbuck, *Redunca fulvorufula (*Afzelius, 1815), roan, and sable antelope, tsessebe, *Damaliscus lunatus (*Burchell, 1823), and waterbuck, *Kobus ellipsiprymnus (*Ogilby, 1833).

For my study I ranked the predators (Figure 5), based on past research (Macdonald and Sillero-Zubiri 2002; Roxburgh 2008; Swanson *et al.* 2014; Wallach *et al.* 2015) on TKR as follows. The lion is the apex predator on the first tier and in their presence all other predators are considered lower level or mesopredators. However, in their absence (as on Korannaberg) the remaining large predators like, wild dog, cheetah, leopard, *Panthera pardus* (Linnaeus, 1758) and brown hyaena, *Hyaena brunnea* (Linnaeus, 1758) fall into the position of top predator. Note that although spotted hyaenas are now extant on Korannaberg, at the time of my study they were absent. Consequently, I have not included them in my hierarchical predator ranking. However, given that they have now been introduced on Korannaberg, it seems that they are likely to supplant the wild dogs and cheetahs as the apex predator on this site, but based on past research they will challenge lions, in certain circumstances for position of apex predator (Periquet *et al.* 2015). On the third tier are species that are typically considered to be mesopredators like black-backed jackals, caracals, bat-eared foxes, Cape foxes, *Vulpes chama* (A. Smith, 1833) African wild cats and honey badgers. The bottom tier is comprised of the small predators like yellow, *Cynictis penicillata* (G.[Baron] Cuvier, 1829) and slender mongooses, *Galerella sanguinea* (Rüppell, 1835), meerkats, *Suricata suricatta* (Schreber, 1776) and genets.

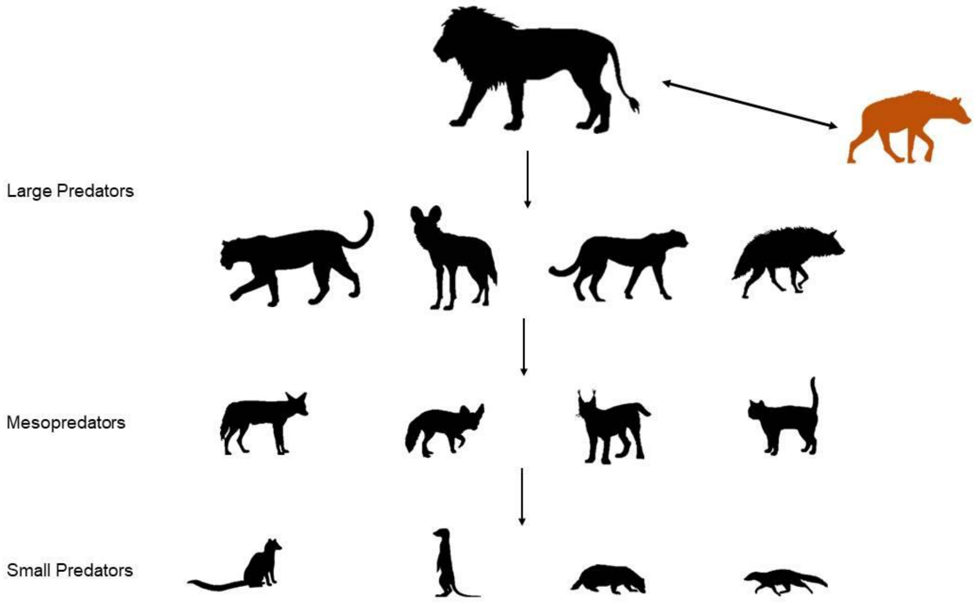

Figure 5: Hierarchical ranking of predators on Tswalu Kalahari Reserve during my study from June 2017 to May 2018. (Designed by R.Ungerer)

CHAPTER 3 – METHODS AND MATERIALS

> *“When we try to pick out anything by itself, we find it hitched to everything else in the Universe”*
>
> *- John Muir*

3.1 Research design

My study followed a similar design to that of Crooks & Soulé (1999), who investigated how the decline of coyotes, in conjunction with habitat fragmentation, affected the abundance and distribution of smaller carnivores and their avian prey species. My study aims to investigate whether I could detect any signals of mesopredator release on TKR. I anticipated detecting responses in mesopredators, and their prey species, small ungulates, small mammals and birds (Soulé *et al.* 1988; Crooks and Soulé 1999; Sillero-Zubiri *et al.* 2004; Glen and Dickman 2005; Lloyd 2007; Berger *et al.* 2008; Ritchie and Johnson 2009; Roemer *et al.* 2009; Bagniewska and Kamler 2013; Ripple *et al.* 2014; Woodgate 2014; Lesmeister *et al.* 2015; Wallach *et al.* 2015; Gordon *et al.* 2017; Tambling *et al.* 2018) Due to the management strategy on TKR I developed my study to detect differences between two top predator management strategies: lions, *Panther leo* (Linnaeus, 1758) absent (Korannaberg) versus lions present (Lekgaba). Therefore, I decided to use a quantitative field experiment to collect empirical data. My study was comparative in nature. I selected sample sites within the two separate sections (Korannaberg and Lekgaba) of the TKR using a process of stratified random sampling.

I used three sampling approaches to gather data: (1) camera traps (hereafter cameras) for the study of predator and ungulate relative abundance (Barea-Azcon *et al.* 2007; Kays *et al.* 2020; Gilbert *et al.* 2021): (2) small-mammal live-trapping to estimate population abundance, relative abundance and diversity (Van Deventer and Nel 2006; Rodas *et al.* 2009; Wijesinghe 2010) and (3) avian point counts to assess population relative abundance and diversity (Fuller and Langslow 1984; Skowno and Bond 2002; Tischler *et al.* 2013). I collected data on Korannaberg and Lekgaba using in precisely the same design to ensure that I could directly compare results between regions. My study was designed to ensure that the sample sites on Lekgaba and Korannaberg were spatially independent of each other (despite there being a predator proof fence separating the two regions) by imposing a buffer of four kilometres (the diameter of a black-backed jackal, *Lupulella mesomelas* (Schreber, 1775) home range in a similar system) (Ferguson *et al.* 1983), thereby spacing sampling locations at a distance larger than the diameter of the home range of the most common mesopredator in the area.

The homogeneity in the landscapes between the Korannaberg and Lekgaba section is what allowed me to be able to do a comparison in the data I collected: both sections contain a combination of rocky/ mountainous areas with limited soils: areas around and between the mountains with poorly-structured, high-base status red soils: and dunes covered in well-drained, high-base status, red soils (Van Rooyen and Van Rooyen 2017). According to the Global Humidity Index the entire region of TKR is classified as an arid zone (UNEP 2004; http://www.grid.unep.ch). Due to the proximity of the sections to one another, both Korannaberg and Lekgaba experience a regional climate range which is subject to extreme temperatures, low rainfall, high evaporation and frost in the winter (Van Rooyen and Van Rooyen 2017). The similarities in land-types and regional climate mean that both sections contain similar vegetation types (Van Rooyen and Van Rooyen 2017).

I considered not only spatial independence, but also temporal independence (seasonal variability) in my target populations. I collected seasonal data across a series of randomly located sample sites on Korannaberg and Lekgaba. I applied a temporal buffer of two months between each consecutive sampling season.

3.2 Assessment of mesopredator and ungulate prey relative abundance using camera traps

To isolate the responses of mesopredators, such as black-backed jackal, and caracal, *Caracal caracal* (Schreber, 1776), I assessed two alternate apex predator management strategies, using cameras. I used stratified random sampling to identify my camera sampling sites. I randomly selected camera locations by superimposing a 4 km^2 (2 km x 2 km) grid over a map of the TKR. I then assigned numbers to each grid square and then, using a random number generator (Microsoft Excel version 365 (2109, Build 16.0.14430.20154, 32-bit)), I identified 60 random grid squares (30 grid squares on Korannaberg and 30 grid squares on

Lekgaba) (Figure 6) as locations for camera deployment. I used the 15 cameras available to me for this study of which (four were Bushnell Natureview Essential HD® and the remaining eleven were Bushnell Essential E3 HD®).

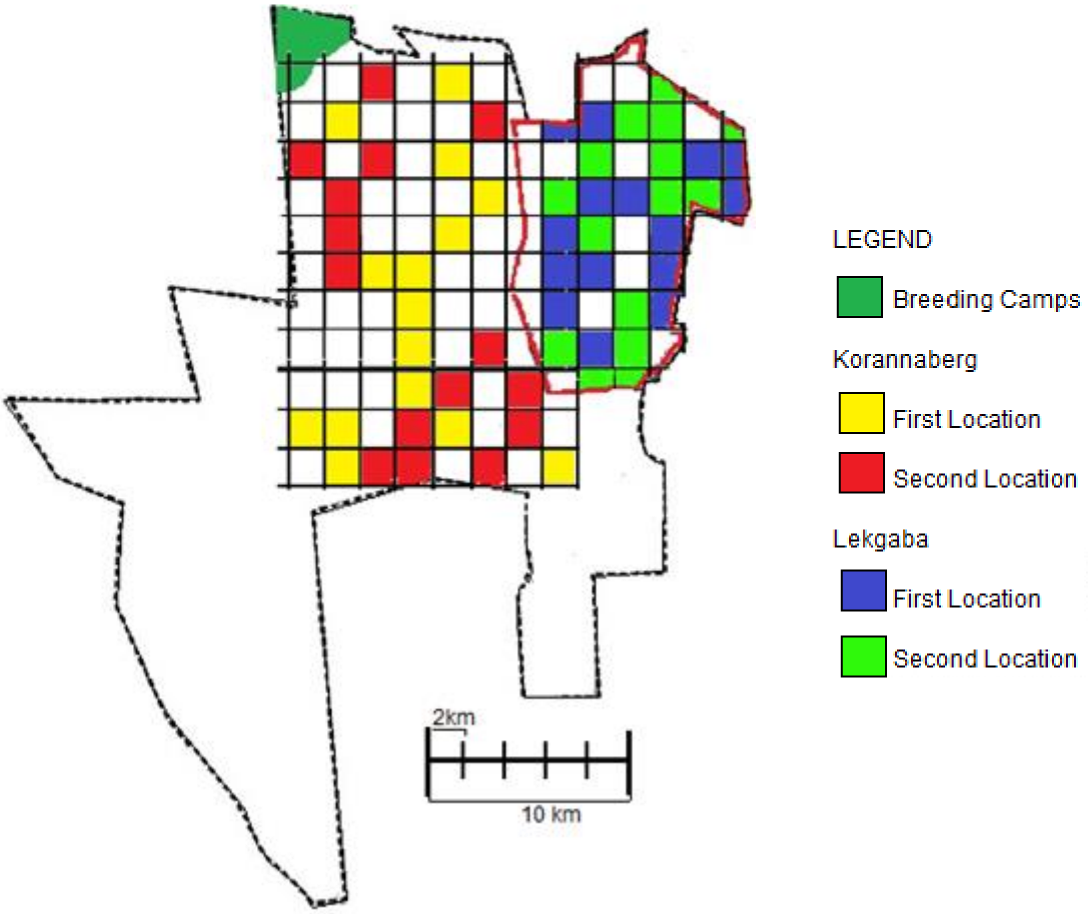

Figure 6: Randomly selected camera trap deployment locations for this study, on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Yellow and blue grid squares indicate where I deployed the first set of cameras each season; Red and light green grid squares indicate where I deployed the second set of cameras each season;

I based the grid square sizes on the home ranges of black-backed jackals, the most abundant mesopredator in the region (Ferguson 1980; Mills and Bester 2005). Past research in South Africa has showed that the home ranges of jackals are highly dependent on their habitat, dispersion and abundance of resources, as well as the intensity and frequency of predator management (Rowe-Rowe 1982; Minnie *et al.* 2016; Tambling *et al.* 2018). Jackal population densities are influenced by the density and composition of the carnivore community. Therefore, a high degree of variation in black-backed jackal population densities is to be anticipated among study sites (Rowe-Rowe 1982; Hiscocks and Perrin 1988; Minnie *et al.* 2016). Black-backed jackal home ranges are highly variable and can range between 4 and 33 km^2 (Minnie *et al.* 2016), while the home range of an adult black-backed jackal on Kalahari Gemsbok National Park ranged between 2.6 and 8.8 km^2 with a mean of 4.1 km^2 (Ferguson 1980). My inter-camera distances ranged from 2 km to 22km.

In each grid square, I identified a tree with a long and bare stem with little overhanging material (branches and leaves) to minimize false triggers caused by the movement of branches (Meek *et al.* 2014; Kolowski and Forrester 2017; Egna *et al.* 2020). I chose trees situated such that the camera sensor and lens would have an unimpeded 'view'. Where possible, I deployed cameras on trees that 'overlooked' roads (paths cleared of vegetation for vehicle use which are not tarred or paved), management tracks or an animal paths as predators use linear features such as these at night when foraging (Forman and Alexander 1998; Gompper *et al.* 2006; Balme *et al.* 2009; Roemer *et al.* 2009; Hines *et al.* 2010). Where my grid squares did not incorporate roads or tracks, I fastened my cameras to the best-suited tree that was closest to the centre of the grid square. I deployed a single camera trap at each site. This method of camera deployment did not bias my results, as I did not intentionally target areas preferred or avoided by animals (Howe *et al.* 2017).

I fastened my cameras to tree trunks/ branches at a height of 0.75 m above the ground to target medium-sized mammals, including mesopredators like African wild cat, *Felis lybica* (Forster, 1780), bat-eared fox, *Otocyon melagotis* (Desmarest, 1822), Cape fox, *Vulpes chama* (A. Smith, 1833), caracal, and black-backed jackal (Ancrenaz *et al.* 2012; Meek *et al.* 2014). I did not bait the cameras as past studies have shown that baiting may result in altered animal movements and may influence movements of sympatric smaller/ prey species (Ancrenaz *et al.* 2012; Rocha *et al.* 2016). I set the cameras to record 30-second digital video clips when triggered. I set the delay between sequential recordings on the camera traps at three seconds; this ensured continuous sampling and increased the likelihood of recording groups of animals that were travelling together (Ancrenaz *et al.* 2012; Meek *et al.* 2014). Because my clips were 30 seconds long, I did not differentiate between multiple sequential detections of the same animal/ group of animals from the clips and each clip was counted as an individual detection (Caravaggi *et al.* 2016, 2020; Debata and Swain 2018). The trigger sensitivity that I set was based on habitat structure, as sensitivity settings have been shown to affect the detection rates of cameras, by either misfiring if set on “high” sensitivity because of moving vegetation or not being triggered when smaller, faster animals go past when set on “low” sensitvity (Meek *et al.* 2012; Rovero *et al.* 2016; Egna *et al.* 2020). I ensured equal effort sampling at each of the sample sites by repeating the same protocol on both sections of the reserve and by ensuring that the duration of my camera sampling was the same in each section (Vigle 2008). I achieved this by deploying the cameras for 10 consecutive nights, each season, at each set of site locations, after which I moved all of them to another set of locations. Each season I deployed the cameras in four sets of locations, the first month was spent sampling on Korannaberg followed by a month of sampling on Lekgaba (Figure 6). Once in situ, I did not check the cameras until it was time to move them to new locations (after 10 nights). I did this in an attempt to mitigate possible influences of my presence on animal movements and detections (Ancrenaz *et al.* 2012).

The purpose of the cameras, for my study, was to estimate the relative abundance of predators (mesopredators in particular) and ungulates on TKR (Moeller *et al.* 2018; Kays *et al.* 2020; Amburgey *et al.* 2021; Palencia *et al.* 2021). I selected the measurement of relative abundance (detections per 100 trap nights) and diversity (Shannon-Weiner Index) as my measurement units for the comparison of predators and ungulates. The use of relative abundance is considered to be the best method for comparisons (trends, changes or differences) within populations and to monitor these population trends (Allen and Engeman 2015; Rovero *et al.* 2016). The use of relative abundance methods are pivotal in establishing qualitative and quantitative insights about species populations and management effects of them (Macdonald and Rushton 2003; Allen and Engeman 2015).

3.3 Assessment of small mammal abundance and diversity through live trapping

I used a live-capture, mark-recapture method to estimate population density and variability, absolute and relative abundance of small mammals in the Kalahari environment (Nel *et al.* 1984; Kerley *et al.* 1990). I used Sherman ™ large folding aluminium (LFA) traps for my small mammal trapping. I randomly selected the transect locations by superimposing a 2 km x 2 km grid over a map of my sample area on TKR. I then assigned numbers to each grid square and using a random number generator in Microsoft Excel version 365 (2109, Build 16.0.14430.20154, 32-bit), 24 random grid square numbers, within the range of numbers on the map, were identified (12 grid squares on Korannaberg and 12 grid squares on Lekgaba and these were the locations for the transects (Figure 7). My inter-transect distances ranged from 2 km to 22 km. I based the grid square sizes on the home ranges of black-backed jackal, this deployment protocol also ensured spatial independence of the small mammal sampling as their home ranges are considerably smaller than those of black-backed jackal and consequently these transects could be considered to be spatially independent. The typical home range size of the four native gerbil (*Gerbillus* Desmarest, 1804) species found in the Kalahari range from 0.002 km^2 to 0.04 km^2 (Stuart and Stuart 2015). Therefore, the use of a 4 km^2 grid ensured I had no overlap in the small mammal populations I sampled.

I implemented the mark–recapture methodology with an open population model to estimate the species specific small mammal population sizes during my study (Jolly 1965; Seber 1965) because my sampling continued over an extended period, to detect any seasonal variation in small mammal populations. Open models incorporate temporal variability in capture probabilities because of changes in abundances caused by births, deaths, emigrations or immigrations (Hammond 1990).

I used 120 Sherman™ LFA traps available for my study. I deployed these traps, simultaneously in four 30 trap linear transects. I decided on transects rather than grids as transects tend to yield more total and individual captures, as well as more species across a larger sample area (Pearson and Ruggiero 2003). I deployed the 30 small mammal traps at 10 m intervals (Ramahlo *et al.* 2019) (total transect length was 290 m) along each transect. I sampled four separate 30 trap transects at four different locations concurrently. I deployed the traps in the field on each transect for three consecutive nights. I checked all of my traps first thing in the mornings (from approx. 30 min after sunrise). After checking my traps, I closed them to minimise the possibility that diurnal small mammals or birds might enter the traps during the day and potentially suffer due to exposure in the traps. I baited and set my traps in the early evenings, just before sunset (cold-dry season ~ 18:00; hot-dry season ~19:00; hot-wet season 19:00), with a bait-ball and a cotton wool ball and checked them at sunrise the following morning – the specific sunset/sunrise times were dependant on seasonal solar cycles.

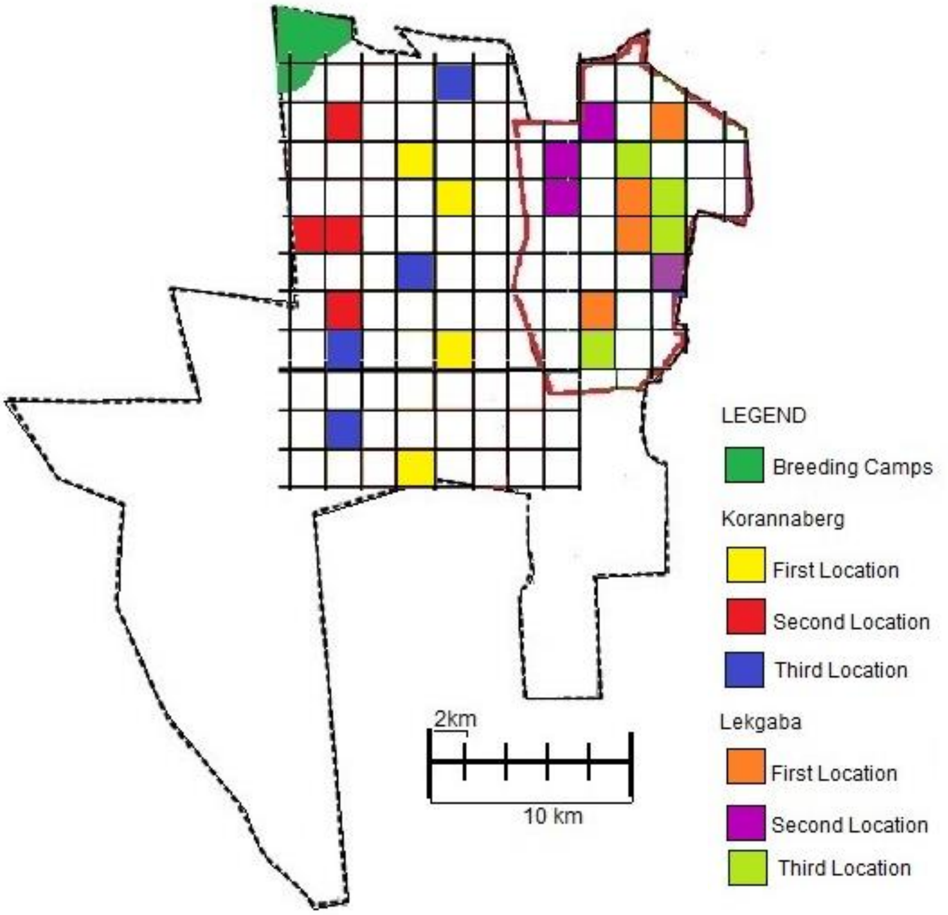

Figure 7: Randomly selected small mammal trapping locations for this study, on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Yellow, red and blue grid squares indicate where I deployed the first, second and thirdt set of small mammal transects each season on Korannaberg;; Orange, purple and light green grid squares indicate where I deployed the first, second and third set of small mammal transects each season on Lekgaba;.

I typically had all the traps checked, cleared and closed by 11 am, before the warmest time of the day (Jose and Packer 2020). This ensured that the animals spent as little time as possible in the traps and mitigated against hyper- and hypothermia in the summer and winter respectively. After three sampling nights, I moved the traps to the next set of sampling sites and deployed them in accordance with the same protocol. I sampled all my sites on Korannaberg first and then repeated the process on my sites at Lekgaba. I released any individuals from non-target taxa (birds, reptiles, and insects), that I captured in my traps, immediately upon discovery.

I used a standard bait of peanut butter, rolled oats and pilchards, hand-rolled into a little bait ball (Edalgo and Anderson 2007; Ramahlo *et al.* 2019; Jose and Packer 2020). When I opened the traps in the evenings, I rebaited all traps from which the bait had been removed or eaten. During each new deployment (when I moved trap transects from one location to another), I rebaited all of the traps with fresh bait. I did not pre-bait the traps as several studies have demonstrated that pre-baiting may alter and reduce trapping success (Edalgo and Anderson 2007; Rodas *et al.* 2009). Furthermore, Wijesinghe (2010) demonstrated that pre-baiting of small mammal live traps added to trapping time and prolonged trapping protocols, while increasing the mortality rate of the small mammals.

To process a small mammal that had been captured in a trap, I transferred the animal from the Sherman trap to a Ziploc™ plastic bag. I positioned the mouth of the bag over the opening of the trap, turning the trap upside down, opening the trap door with one finger and then carefully shaking the animal out into the bag. I then carefully manoeuvred the small mammal to the opening of the bag and allowed its head to poke through it while keeping the rest of the body secured in the bag. With the head exposed I could then safely and harmlessly mark the small mammal. I did this by removing a small patch of hair, using fine nail scissors, from the back of the head (the area least likely to be rubbed or scuffed (Twigg 1975; Boitani *et al.* 2000) of each small mammal that I captured (Figure 8). I used a non-toxic gel nail polish (Essence Cosmetics™) to apply a colour-coded semi-permanent mark to the 'shaved' area on each small mammal. I used six different coloured gel polishes (*"take me to heaven*" – blue, *"walking on sunshine"* – yellow, *"lilac vibes"* – purple, *"never say never"* – green, *"legally pink"* – pink, *"fame fatal"* – red) for these marks which I used in different colour combinations (from six colours I had 720 possible colour mark combinations) to identify the individuals when I recaptured them. My decision to use this method was informed by the necessity to minimise handling time for the small mammals (Twigg 1975; Boitani *et al.* 2000).

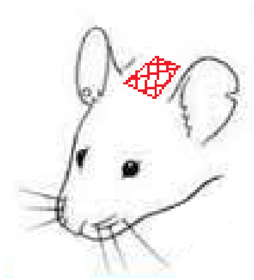

Figure 8: Diagrammatic representation of the area from which I clipped the fur for the purpose of marking animals during my study on Tswalu Kalahari Reserve from June 2017 to May 2018.

After I had applied the individual colour marks to the small mammals, I recorded the following for each small mammal that I captured: date, time, trap transect number and trap number, species, colour code, and a set of morphometric measurements (head length, body length, tail length, full body length including the tail, and head breadth) using a Vernier gauge. I used a combination of field guides and expert opinion (Dr D. McFadyen)* to identify small mammals to species level. I recorded these data on a field form (Appendix II). After the small mammals were captured, measured, and identified. I categorized them according to their dietary functional group based on Skinner and Chimimba (2005b), and, Stuart and Stuart (2015).

3.4 Assessment of avian populations occurrence and relative abundance

I used a point-count method along transects to determine the relative abundance of birds on TKR (Fuller and Langslow 1984; Ogada *et al.* 2008; Sirami and Monadjem 2012; Thompson 2014). Point-counts involve recording birds that are either seen or heard in a specific area, within a specific time, and they are commonly used in studies to determine avian-habitat relationships and the response of these populations to different management treatments (Thompson 2014). Point-count methods rely on the assumption that the numbers of individuals identified represent a constant portion of the actual numbers of the species present within that given area (Thompson 2014). I used transects to do my point-

* Dr. Duncan MacFadyen, Head Research and Conservation, Oppenheimer Generations, Personal communication by email, November 2017 – January 2018

counts which I sampled in the morning (at sunrise) and in the afternoon (at sunset) when birds are most active (Jain *et al.* 2005; Trnka *et al.* 2006). I randomly selected the transect locations by superimposing a 2 km x 2 km grid over a map of my sample area on TKR. I then assigned numbers to each grid square and using a random number generator in Microsoft Excel version 365 (2109, Build 16.0.14430.20154, 32-bit), 16 random grid square numbers, within the range of numbers on the map, were identified (8 grid squares on Korannaberg and 8 grid squares on Lekgaba and these were the locations for the transects (Figure 9). My inter-transect distances ranged from 2 km to 22km.

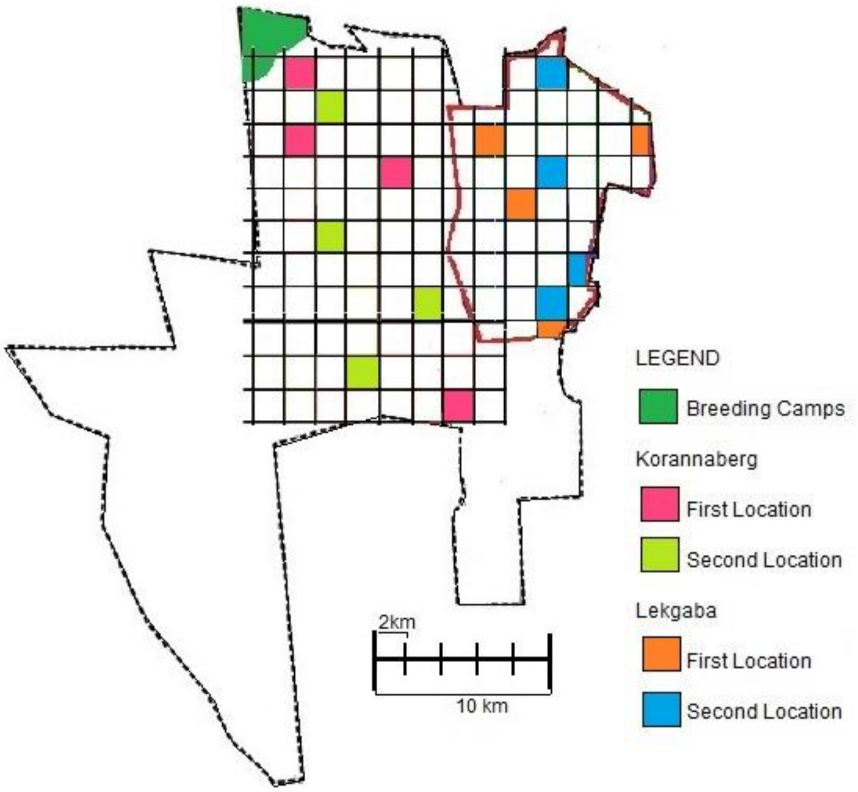

Figure 9: Randomly selected avian point-count site locations for this study, on Tswalu Kalahari Reserve, used during my study between June 2017 and May 2018.

Note: Pink and light green grid squares indicate where I conducted the first and second set of point counts on Korannaberg each season; Orange and blue grid squares indicate where I conducted the first and second set of point counts on Lekgaba each season;

I conducted my bird point counts along 400 m transects. When placing my transects I measured out 100 m lengths at a time and placed a marker at the point where I would need to conduct the sampling. I sampled each transect by stopping at the four markers (100 m apart) along the length of the 400 m transect. During each stop I spent 30 minutes recording all bird species that could be visually observed from that point. The one common practice used in all point-count methods is that the observer remains stationary at a predetermined location for a predetermined length of time and records all birds seen or heard are recorded (Fuller and Langslow 1984).

I recorded all birds that were visible (including birds that were flying above the transect, but were visible to the naked eye, and all those that were perched or were actively foraging along or across the transect). I did not record aerial feeders, like swallows and swifts, unless they were perched. I did not record birds that I heard as a detection unless I verified it with a sighting of that species. I attempted to avoid double counting birds by keeping track of the number of individuals within flocks and the sex ratio of the birds, between consecutive points.

I sampled two different transects each day, one at sunrise and one at sunset, when I was conducting my seasonal bird counts. I counted birds once per season along each bird transect. I alternated the observation time for each transect between sunrise and sunset each consecutive season (e.g. Lekgaba, Transect 1, cold-dry season sampled at sunrise, hot-dry season sampled at sunset). During each season of sampling, I first conducted my bird counts along transects on Korannaberg and then on Lekgaba.

The method I employed for this study was adapted from Skowno and Bond (2003), birds at each sample site were observed for first 120 min of light in the early morning and last 120 minutes of light in the late afternoon. Although this method was developed to be used in open grasslands in mesic African savannas (Skowno and Bond 2003; Ogada *et al.* 2008), it seemed that it would be appropriate for implementation in the sparse vegetation of the Kalahari. The benefit of using the point-count method is that it can be used in a wide variety of vegetation types (Fuller and Langslow 1984). I identified all visible birds using binoculars (*Nikon Prostaff 3S 10 x 42*). I recorded the following for each bird that I detected: species (Roberts *et al.* 2018), sex, number of individuals, and their location in relation to my location (approximate distance (in meters)), which I estimated based on the bird/s proximity to each 100 meter marker along the transect which I had measured out) and in which direction they were away from me (unless they were overhead, in which case I noted they were flying over and in which direction they were flying), as well as the date, time and transect number (Appendix II).

3.5 Data analyses

I recorded all field data on physical field forms (Appendix II) and then transcribed these data into MS excel (Microsoft Corporation, 2018) spreadsheets. I could sort my data based on site, season, species, or any category that I wanted to analyse. I calculated my various metrics – for my camera detections I multiplied the number of detections per camera by 10 to get detections per 100 trap nights. For my small mammal data, I calculated the estimated sample size (ESS) of each species per site using the Cormack-Jolly-Seber (CJS) model for live captures in Program MARK (White and Burnham 1999; Cooch 2018). For my bird detections I calculated my metric by multiplying my detections per transect by 2.5 to get detections per kilometre.

I used the Shannon-Wiener diversity index (Equation 1)* to analyse the species diversity of the predator, ungulate, small mammal and avian populations (Heip and Engels 1974; Smith and Wilson 1996; Heip *et al.* 1998) on Korannaberg and Lekgaba. To calculate a species diversity, two indicies are required, species richness and species evenness. Species richness is simply the number of species in the community being sampled, whereas species evennss requires calculating a species proportional/ relative abundance. The proportional abundance of a species is a measue of how common/rare a species is relative to other species within a defined community. I used Microsoft MS Excel to calculate the proportional abundance of the species evenness and diversity. As well as for plotting the graphs and tables from these results.

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

*Equation 1: Shannon-Wiener Diversity Index Equation:
H' = Diversity Index;
S = Number of Species
p_i = Proportion of individuals in the sample belonging to the ith species;
Where (p_i = ni/N); ni = number of individuals of species; N = total number of individuals of all species

All of the data that I collected from the cameras was quantitative because I gathered numeric values for the detections of predators and ungulates (Hernandez 2021). I compared detections and detections per 100 trap nights between sections, among camera traps sites, within seasons among sites, using t-tests and

Chi-squared tests (Field *et al.* 2012). The majority of the data I collected from the small mammal trapping was quantitative because I counted and compared number of species/ abundances and diversity and then compared measurements of those species. For my entire small mammal capture set and estimated sample size (ESS) I compared between sections, transects, within sections among seasons, body morphometries and dietary functional groups I used t-tests and Chi-squared tests (Field *et al.* 2012). I compared my avian point-count results between sections, among seasons, transect sites, sites within seasons, between time of the day (morning vs. evening) and I compared my metric index of detections per km by sections, season, transect sites, between sections within seasons and between the time of day (Hernandez 2021) using t-tests and Chi-squared tests (Field *et al.* 2012). My bird data were quantitative because I gathered numeric values for the detections of the avian populations on TKR.

Although my data were spatially and temporally independent, I used a Shapiro-Wilk (Royston 1992) test to assess the normality of my data (Thode 2002) if the resultant p-value was ≥0.05, I assumed that my data were normal. In the case of normal data I used appropriate parametric statistical tests. Where my data were non-normal, I used the non-parametric equivalents of the above tests, like the Mann-Whitney-Wilcox U-test (Field *et al.* 2012), or the Mann-Whitney-Wilcox U-test (Thode 2002; Field *et al.* 2012).

I used One-way ANOVA for all the seasonal differences in predators, ungulates, small mammal and avian data that were normal and if there was a significant difference present a post hoc Tukey Honestly Significant Different (HSD) test was used to determine where the significant differences were present between (Thode 2002).

I conducted all the statistical tests in Program R (R version 4.1.2 (2021-11-01) - "Bird Hippie") (Dalgaard 2008; Zuur *et al.* 2009; RStudio 2021). All statistical tests for camera, small mammal capture and avian detection data were evaluated at $\alpha = 0.05$.

All data collection procedures and handling of live animals was done following the ethics code set out by the College of Agricultural and Environmental Sciences Ethics Committee of the University of South Africa, through the approval of an ethics application (ERC Reference #: 2017/CAES/092).

CHAPTER 4 – RESULTS

“People need to look at wildlife conservation in its totality. As soon as you lose the apex predator, it has harmful consequences right down the food chain.”

-Thomas Kaplan, American Businessman

4.1 Camera trap survey

During my study I recorded 49 897 thirty-second video clips using 15 cameras on TKR ($\bar{X}$ = 831.6, SD = 571.1), Korannaberg (n = 25 894.0, $\bar{X}$ = 863.1, SD = 4 45.1), Lekgaba (n = 24 003.0, $\bar{X}$= 800.1, SD = 672.5) (Appendix I – Table A2). I deployed the cameras seasonally for 10 days at 60 sites (Figure 10) on both Korannaberg (n = 30) and Lekgaba (n = 30), a total of 180 trap stations for the study. From these I recorded the highest number of triggers in the hot-wet season (n = 21 522, $\bar{X}$ =1 434.8, SD = 820.2), followed by the hot-dry season (n = 16 852, $\bar{X}$ = 1 123.5, SD = 601.2), and the lowest in the cold-dry season (n = 11 523, $\bar{X}$= 768.2, SD = 707.5).

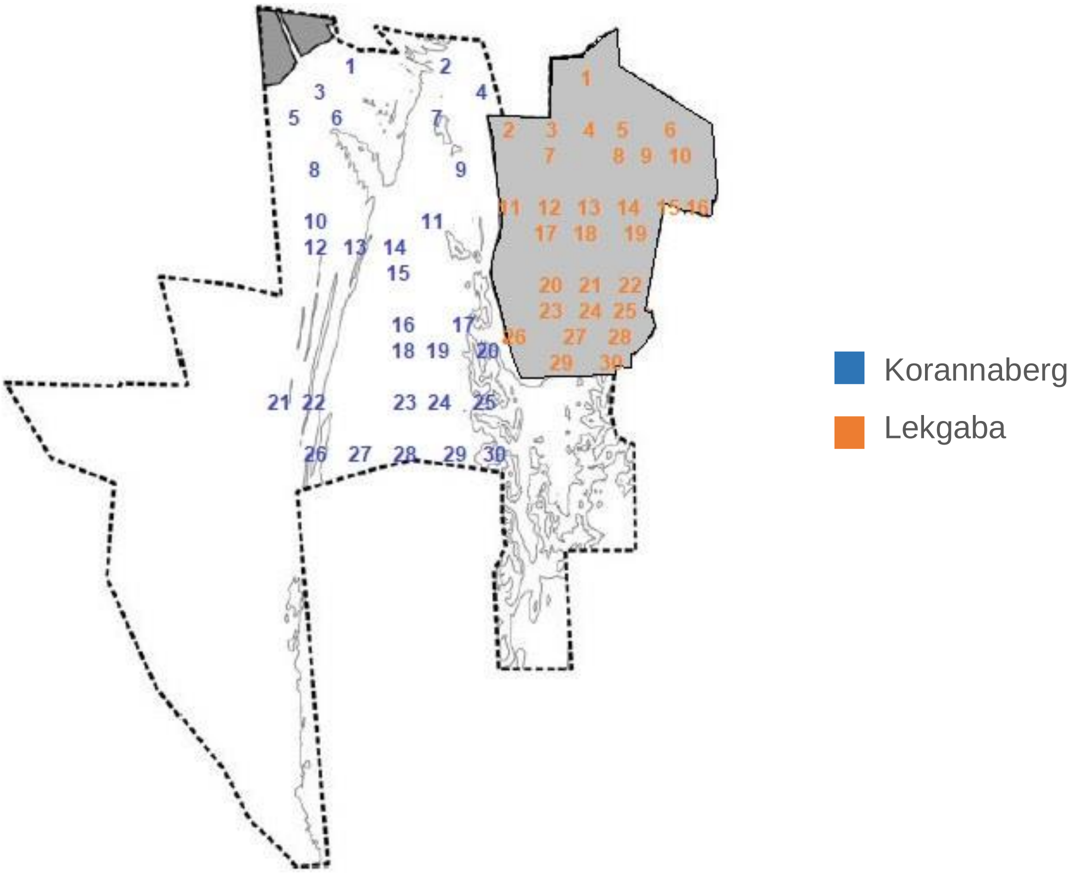

Figure 10: Locations of camera traps deployed during my study on Tswalu Kalahari Reserve between June 2017 and May 2018 (GPS co-ordinates of camera traps in Appendix I – Table A1).

Not every camera trigger resulted in a detection of an animal, these constituted false triggers. Of the total of 49 897 triggers only 8 441 (16.9%) constituted detections. On Korannaberg I recorded 25 894 triggers of which 4 436 (17.1%) were animal detections and on Lekgaba I recorded 24 003 triggers of which 4 005 (16.7%) were detections. I sampled for a total of 1800 camera nights and recorded 8 441 ($\bar{X}$= 140.7, SD = 215.3) detections of animals for the entire TKR, 4 436 ($\bar{X}$ = 147.9, SD = 162.8) for Korannaberg, and 4 005 ($\bar{X}$ = 133.5, SD = 257.1) for Lekgaba (Appendix I - Table A3 & A4). I found that neither the total number of detections per camera (χ^2 = 760.0, df = 728, p = 0.19), nor the mean number of detections per camera (χ^2 = 760.0, df = 728, p = 0.19) differed between Korannaberg and Lekgaba. I found no significant difference in the number of detections per camera between sections (t = 0.3, df = 49, p = 0.801).

I found that the trend in the number of detections per season was opposite to the trend in the number camera triggers per season (Figure 11). I recorded the highest number of triggers in the hot-wet season (n = 21 522, $\bar{X}$ = 1 434.8, SD = 820.2), followed by the hot-dry season (n = 16 852, $\bar{X}$ = 1 123.5, SD = 601.2) with the lowest number of triggers recorded in the cold-dry season (n = 11 523, $\bar{X}$ = 768.2, SD = 707.5).

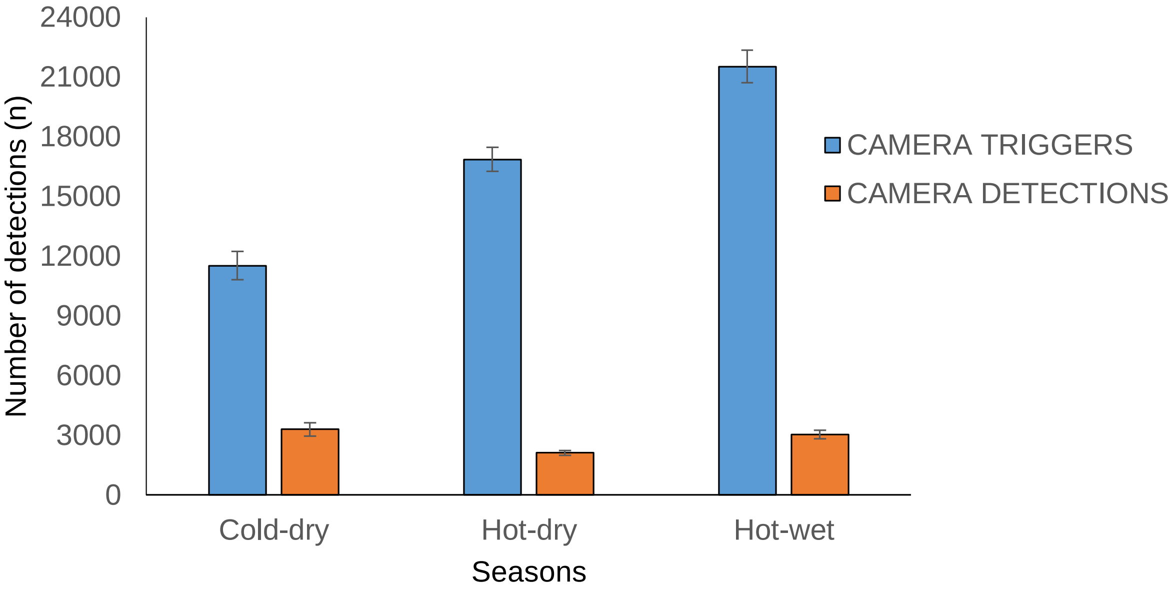

Figure 11: Seasonal camera trap triggers and camera trap detections of animals recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

I recorded the highest number of detections in the cold-dry season (n = 3 294, $\bar{X}$ = 219.6, SD = 328.1), followed by the hot-wet season (n = 3 029, $\bar{X}$ = 201.9, SD = 217.6), and I recorded the lowest number of detections in the hot-dry season (n = 2 118, $\bar{X}$ = 141.2, SD = 123.2) (Table 1). I found that number of detections differed seasonally (χ^2 = 4 164.0, df = 28, $p < 0.001$) and that in the cold-dry season I recorded significantly more detections thanduring both the hot-dry season ($p < 0.05$) and the hot-wet season ($p < 0.05$), but I found no significant difference in detections between the hot-dry and hot-wet seasons.

On Korannaberg I recorded similar numbers of animal detections in both the cold-dry (n = 1 034, $\bar{X}$ = 68.9, SD = 95.4) and hot-dry seasons (n = 1 012, $\bar{X}$ = 67.5, SD = 95.4) I found the highest numbers of detections in the hot-wet season (n = 2390, $\bar{X}$ = 159.3, SD = 194.0) (Figure 12 and Table 1). On Lekgaba I recorded the most detections in the cold-dry season (n = 2 260, $\bar{X}$ = 150.7, SD = 332.2), followed by the hot-dry season (n = 1 106, $\bar{X}$ = 73.7, SD = 88.4) I found the lowest number of detections in the hot-wet season (n = 639, $\bar{X}$ = 42.6, SD = 43.5) (Table 1). I found significant seasonal differences in the total number of detections, (χ^2 = 270.5, df = 2, p <0.001) and the mean number of detections (χ^2 = 9.0, df = 2, p <0.001) (Appendix I – Table A5).

Table 1: Seasonal camera trap detections recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Tswalu			Korannaberg[1]			Lekgaba[2]		
Season	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Cold-Dry	3294	109.8	247.8	1034	68.9	95.4	2260	150.7	332.2
Hot-Dry	2118	70.6	82.1	1012	67.5	75.2	1106	73.7	88.4
Hot-Wet	3029	101.0	152.2	2390	159.3	194.0	639	42.6	43.5

[1]Lions absent; [2]Lions present

I compared detections per season per sites and found no significant difference between Korannaberg and Lekgaba in the cold-dry season (t = -0.9, df = 33, p = 0.388) and I recorded no significant difference in the hot-dry season between sites (t = -0.2, df = 35, p = 0.835). For the hot-wet season (χ^2 = 33.8, df = 35, p = 0.528) I recorded similar values for both sections (Table 1), however, I recorded significantly more detections on Korannaberg than on Lekgaba (t = 2.1, df = 32, p = 0.042) (Table 1).

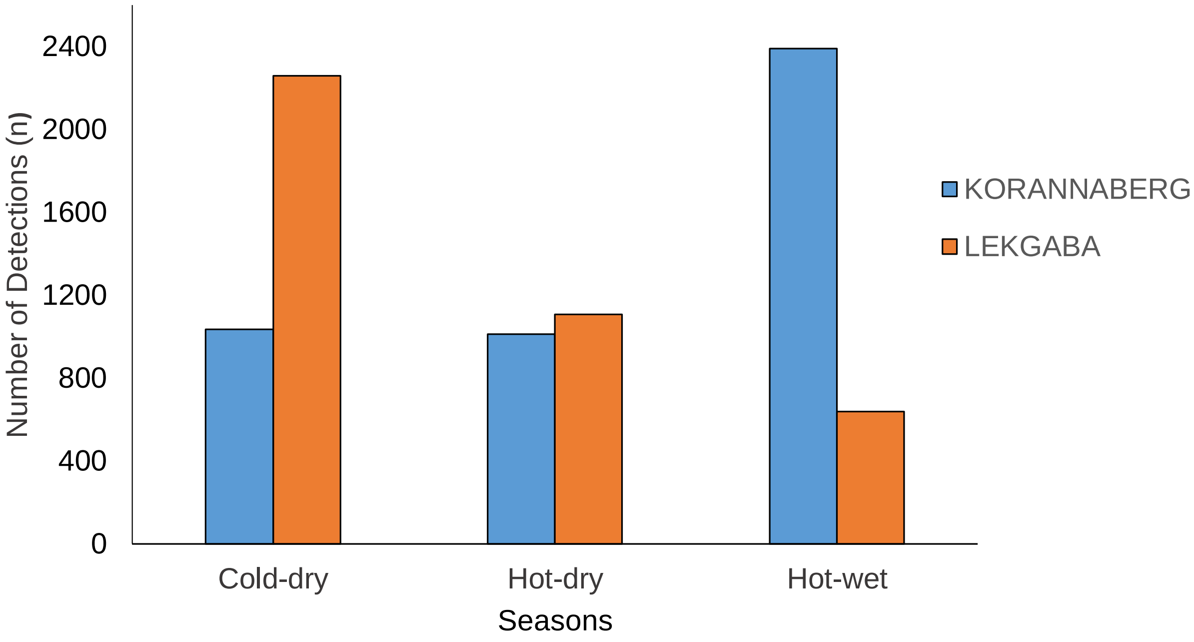

Figure 12: Seasonal camera trap detections recorded on Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

From the 8441 clips with detections I recorded sightings from five taxa; (birds = 3489, omnivores = 2402, predators = 666, reptiles = 8, ungulates = 26903) which I analysed by section (Appendix I - Table A3 & Figure A1), season (Appendix I – Table A3 & Figure A2) and seasonally between sections.

Table 2: Percentages of camera trap detections for vertebrate groups recorded on the Tswalu Kalahari Reserve from June 2017 to May 2018.

Taxa	Tswalu	Korannaberg[1]	Lekgaba[2]
Birds	10.4	7.3	12.7
Omnivores	7.2	10.9	4.4
Predators	2.0	2.4	1.7
Ungulates	80.4	79.4	81.1

[1] Lions absent; [2] Lions present

Note that reptile percentage detections were removed from this table as percentage amount was less than 0.1

4.1.1 Predators

4.1.1.1 Korannaberg vs. Lekgaba comparison

Korannaberg and Lekgaba support different assemblages of large predators, therefore detections of large predators differed between sites (Korannaberg; n = 29, $\bar{X}$ = 0.4, SD = 0.3, Lekgaba; n = 96, $\bar{X}$ = 0.4, SD = 1.3) (Table 3). I found that the mesopredator and small predator assemblages were similar on both sites. The only difference was that I only detected caracals, *Carcal caracal* (Schreber, 1776) on Lekgaba (Table 3). Total mesopredator detections were higher on Korannaberg (n = 214, $\bar{X}$ = 0.4, SD = 1.9) than Lekgaba (n = 191, $\bar{X}$ = 0.3, SD = 1.6). Total small predator detections were also higher on Korannaberg (n = 88, $\bar{X}$ = 0.3, SD = 2.4) than Lekgaba (n = 52, $\bar{X}$ = 0.2, SD = 1.5).

Table 3: Camera trap detections of predators on the Tswalu Kalahari Reserve from June 2017 to May 2018.

	Tswalu			Korannaberg[1]			Lekgaba[2]		
Predator Species	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Large Predators									
Acinonyx jubatus	2	0.0	0.1	2	0.0	0.1	0	0.0	0.0
Hyaena brunnea	32	0.2	0.8	10	0.1	0.4	22	0.2	1.0
Lycaon pictus	9	0.1	0.7	9	0.1	0.9	0	0.0	0.0
Panthera leo	73	0.4	3.8	0	0.0	0.0	73	0.8	5.4
Panthera pardus	9	1.5	0.0	8	2.0	0.0	1	1.0	0.0
Mesopredators									
Lupulella mesomelas	290	1.6	7.8	155	1.7	8.5	135	1.5	7.0
Caracal caracal	2	0.0	0.1	0	0.0	0.0	2	0.0	0.1
Felis silvestris lybica	41	0.2	0.9	22	0.2	1.0	19	0.2	0.7
Genetta genetta	2	0.0	0.1	1	0.0	0.1	1	0.0	0.1
Otocyon megalotis	50	0.3	1.2	28	0.3	1.3	22	0.2	1.2
Vulpes chama	20	0.1	0.4	8	0.1	0.3	12	0.1	0.5
Small Predators									
Cynictis penicillata	86	0.5	4.6	78	0.9	6.5	8	0.1	0.5
Ictonyx striatus	3	0.0	0.1	1	0.0	0.1	2	0.0	0.1
Suricata suricatta	51	0.9	2.9	9	0.1	0.7	42	0.5	4.0

[1]Lions absent; [2]Lions present

The ratio of detections of large predators to mesopredators on Korannaberg was 1:7 compared with 1:2 on Lekgaba, and the ratio of detections of small predators to mesopredators was 1:2 on Korannaberg and 1:4 on Lekgaba (Figure 13). I found that the total number of mesopredator detections was 5.7% higher on Korannaberg than on Lekgaba. I found that large predator detections were 53.6% higher on Lekgaba than on Korannaberg and that small predator detections were 25.7% higher on Korannaberg than Lekgaba. I recorded a significantly higher number of predator detections (χ^2 = 1780.5, df = 27, p < 0.001) on Lekgaba than Korannaberg. However, I found that the mean number of detections per site did not differ between Korannaberg and Lekgaba (χ^2 = 23.6, df = 27, p = 0.7)

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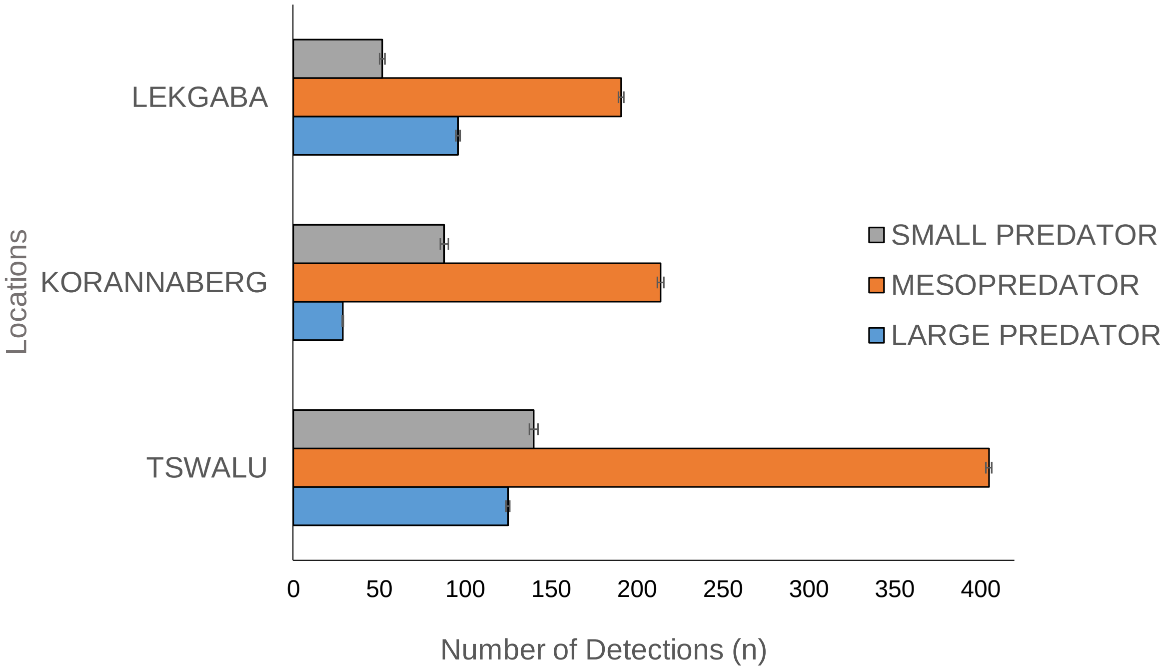

Figure 13: Predator detections recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

I found that neither the relative abundance (detections per 100 trap nights), (χ^2 = 108.5, df = 99, p = 0.2) nor the mean number of predator detections per camera (χ^2 = 86.3, df = 70, p = 0.1) differed between Korannaberg and Lekgaba (Appendix I – Table A7). I found no species-specific differences (p > 0.05) in detection rate between Korannaberg and Lekgaba (Table 4 and Table 5).

I found that mesopredators (61.0%) comprised the majority of predator detections on the TKR, followed by small predators (20.7%) with the large predators contributing the fewest detections (18.3%). My detections of predators on Korannaberg were dominated by mesopredators (64.3%), followed by small predators (26.9%) and large predators had the lowest detections (8.8%). On Lekgaba, I found that predator detections were dominated by mesopredators (57.6%), followed by large predator detections (27.8%), with the least number of detections made up of small predators (14.6%). My total number of camera detections for mesopredators (n = 406) on the TKR were approximately three times more than that of either small (n = 138) or large predators (n = 122) (Figure 15). I recorded more detections of large predators on Lekgaba (n = 93) than Korannaberg (n = 29). I recorded higher numbers of detections of both mesopredators (Korannaberg n = 213; Lekgaba n = 193) and small predators (Korannaberg n = 89: Lekgaba n = 49) on Korannaberg than Lekgaba (Figure 13).

Table 4: Chi-squared test comparing relative abundance (detections per 100 trap nights) of predators between Korannaberg and Lekgaba recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

Predator Species	χ^2	df	p
Large Predator			
Acinonyx jubatus	0.5	2	0.472
Hyaena brunnea	4.2	2	0.521
Lycaon pictus	0.0	2	1.000
Panthera leo	5.5	2	0.244
Panthera pardus	2.0	2	0.368
Mesopredator			
Lupulella mesomelas	12.2	2	0.431
Caracal caracal	0.5	2	0.472
Felis silvestris lybica	3.3	2	0.649
Genetta genetta	0.0	2	1.000
Otocyon megalotis	5.0	2	0.660
Vulpes chama	1.4	2	0.701
Small Predator			
Cynictis penicillata	5.6	2	0.353
Ictonyx striatus	0.0	2	1.000
Suricata suricatta	5.0	2	0.285

The predator (and mesopredator) that I detected most frequently was black-backed jackal, *Lupulella mesomelas* (Schreber, 1775) (TKR n = 290; Korannaberg n = 155; Lekgaba n = 135). The large predator that I detected most frequently was the brown hyaena, *Hyaena brunnea* (Thunberg, 1820), (TKR n = 32; Korannaberg n = 10; Lekgaba n = 22). The small predator for which I recorded the highest number of detections was the yellow mongoose, *Cynictis penicillata* (G. [Baron] Cuvier, 1829) (TKR n = 86; Korannaberg n = 79; Lekgaba n = 7) (Appendix I - Table A7). I found that both the total number of predator detections per camera ($\chi^2 = 2990.0$, df = 14, $p < 0.001$) and the mean number of predator detections per camera ($\chi^2 = 213.3$, df = 14, $p < 0.001$) were significantly higher on Lekgaba than Korannaberg (Table 5). My mesopredator data were normally distributed (W = 1.0, p = 0.715) (Appendix I - Table A8). I found no significant differences between total mesopredator detections (t = 0.1, df = 10, p = 0.908) nor the mean number of mesopredator detections between Korannaberg and Lekgaba (t = 0.1, df = 10, p = 0.902) (Table 5).

Table 5: Comparison of the number of predator detections (per camera trap) between Korannaberg and Lekgaba using a t-test on Tswalu Kalahari Reserve between June 2017 and May 2018.

Predator Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Large Predator					
Acinonyx jubatus	0.4	0.0	29	1.4	0.161
Hyaena brunnea	2.2	4.9	38	-1.2	0.232
Lycaon pictus	2.0	0.0	29	1.0	0.326
Panthera leo	0.0	16.2	29	-1.4	0.163
Panthera pardus	1.8	0.2	29	0.9	0.393
Mesopredator					
Lupulella mesomelas	34.4	29.6	56	0.2	0.834
Caracal caracal	0.0	0.4	29	-1.4	0.161
Felis silvestris lybica	4.9	4.2	51	0.3	0.789
Genetta genetta	0.2	0.2	58	0.0	1.000
Otocyon megalotis	6.0	4.9	58	0.3	0.760
Vulpes chama	1.8	2.7	40	-0.6	0.571
Small Predator					
Cynictis penicillata	17.6	1.6	29	1.1	0.261
Ictonyx striatus	0.2	0.4	53	-0.6	0.562
Suricata suricatta	2.0	9.3	31	-0.9	0.399

[1] Lions absent; [2] Lions present

I found that the cameras (3A, 5B, 15A) that recorded the highest number of predators (n = 5) detected 36% of the total number of predators that occur on the TKR. The highest number of predator detections that I recorded at a single camera location on Korannaberg (n = 90) was Camera 5 (Appendix I – Table A9), and on Lekgaba the camera location at which I recorded the highest number of predator detections (n = 134) on Camera 16 (Appendix I – Table A10). Of the 30 camera sites on TKR, four on Korannaberg, and nine on Lekgaba recorded no predator detections during my study (Appendix I – Table A9 & A10). I recorded an average of two predator species per camera on both Korannaberg and Lekgaba (Appendix I – Table A9 & A10). I found no significant difference in the relative abundance (detections/ 100 trap nights) between sites ($\chi^2 = 870.0$, df = 841, p = 0.237) (Table 6). Additionally, I found no significant differences between Korannaberg and Lekgaba for total and mean number of detections / 100 trap nights (t = -0.03, df = 56, p = 0.978) (Appendix I – Table A11 & A12).

Table 6: Comparison of the relative abundance (detections per 100 trap nights) of predators (per camera trap) between Korannaberg and Lekgaba, using a t-test, on Tswalu Kalahari Reserve between June 2017 and May 2018.

Predator Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Large Predator					
Acinonyx jubatus	0.1	0.0	89	1.4	0.159
Hyaena brunnea	11.1	24.4	119	-1.2	0.239
Lycaon pictus	0.7	0.0	89	1.0	0.320
Panthera leo	0.0	5.4	89	-1.4	0.161
Panthera pardus	8.9	1.1	92	0.9	0.388
Mesopredator					
Lupulella mesomelas	17.2	15.0	172	0.2	0.849
Caracal caracal	0.0	2.2	89	-1.4	0.159
Felis silvestris lybica	2.4	2.1	163	0.3	0.797
Genetta genetta	0.1	0.1	178	0.0	1.000
Otocyon megalotis	2.1	1.6	177	0.4	0.714
Vulpes chama	0.6	0.9	145	-0.7	0.507
Small Predator					
Cynictis penicillata	5.8	0.6	90	1.1	0.264
Ictonyx striatus	0.1	0.1	161	-0.6	0.563
Suricata suricatta	0.7	3.1	95	-0.9	0.395

[1] Lions absent; [2] Lions present

When I compared predator diversity, I found that Korannaberg and Lekgaba had equal species richness (S = 12.0), but the species diversity (H') was slightly lower on Korannaberg (H' = 0.12) than on Lekgaba (H' = 0.13) (Figure 14). For mesopredators (Figure 16), I found that Lekgaba had a higher species richness (S = 6.0) than Korannaberg (S = 5.0). Yet, species diversity was slightly different for Korannaberg (H' = 0.15) and Lekgaba (H' = 0.16). Additionally, I compared predator diversity by camera location (Appendix I – Table 13) and found that richness was higher on Korannaberg (S = 26.0), than Lekgaba (S = 21.0). I found that species diversity per camera was also marginally higher on Korannaberg (H' = 0.08) than Lekgaba (H' = 0.07) (Figure 15).

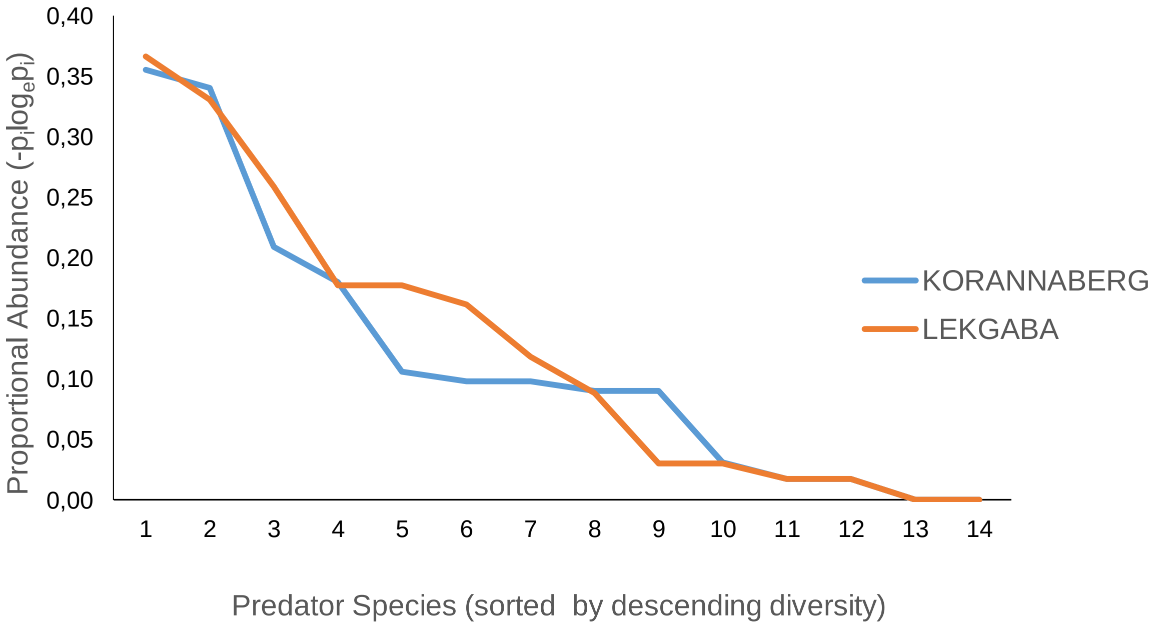

Figure 14: Relative diversity of predators detected using camera traps on Tswalu Kalahari Reserve between June 2017 and May 2018.

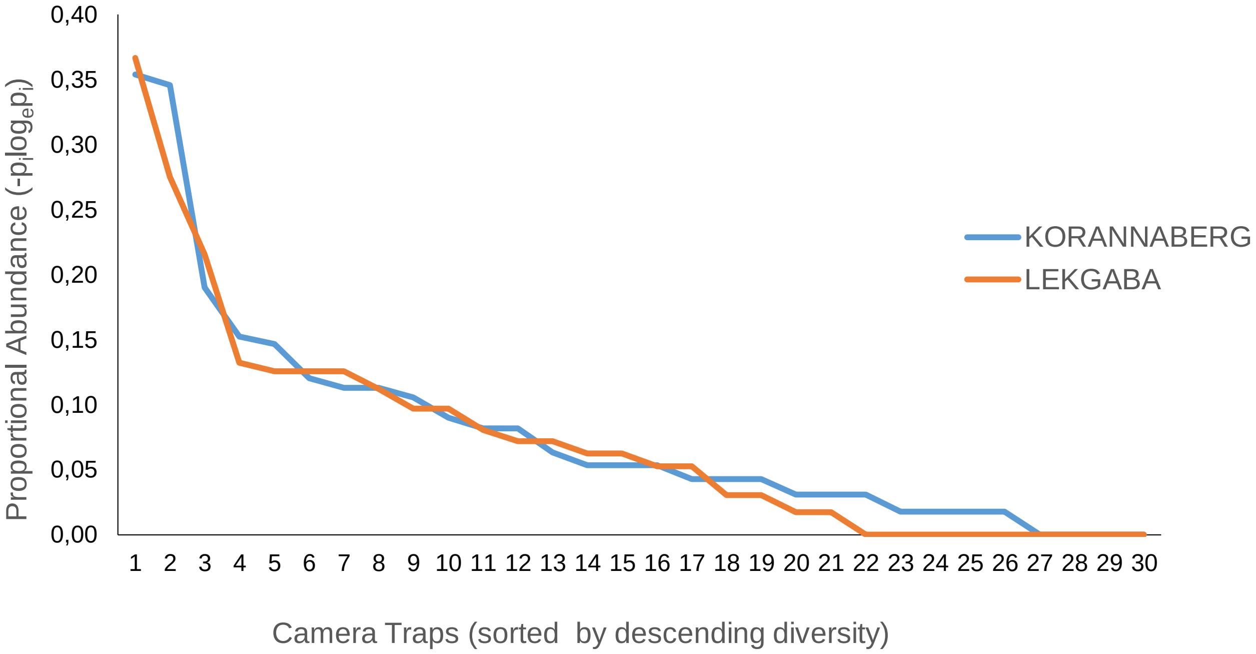

Figure 15: Relative diversity of the predators detected per camera site on Tswalu Kalahari Reserve between June 2017 and May 2018.

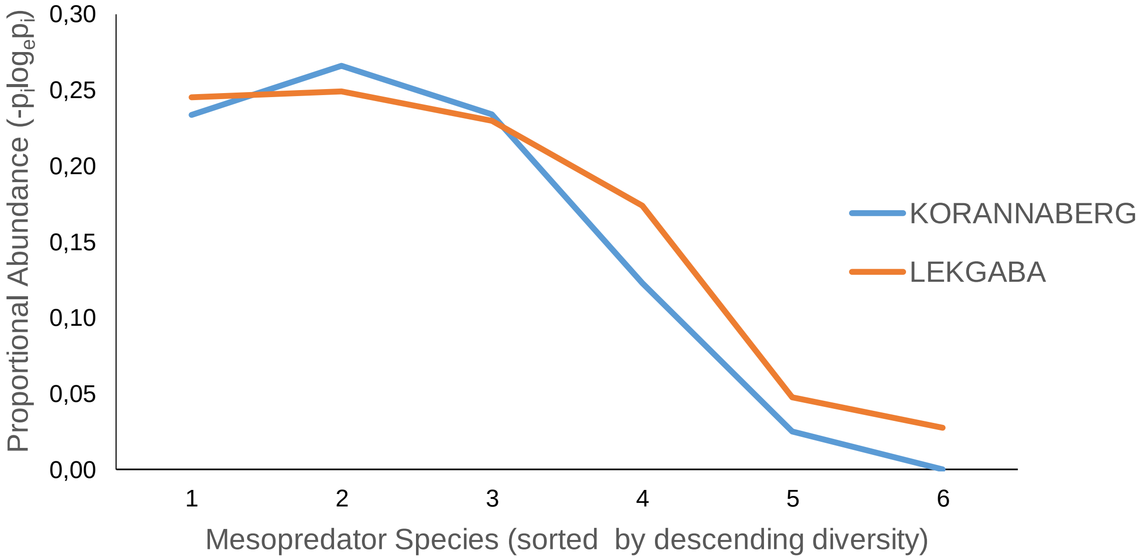

Figure 16: Relative diversity of mesopredators detected using camera traps on Tswalu Kalahari Reserve between June 2017 and May 2018.

4.1.1.2 Seasonal comparison

I recorded the highest number of predator detections in the cold-dry season (n = 297, $\bar{X}$ = 0.3, SD = 2.7), followed by the hot-wet season (n = 266, $\bar{X}$ = 0.3, SD = 3.2), with the lowest detections in the hot-dry season (n = 107, $\bar{X}$ = 0.1, SD = 0.8). I recorded a similar trend for the large predators (Figure 19) (Appendix I – Table A14) with the highest detections in the cold-dry season (n = 90, $\bar{X}$ = 0.3, SD = 1.8), followed by the hot -wet season (n = 23, $\bar{X}$ = 0.1, SD = 0.5), and I recorded the lowest large predator detections in the hot-dry season (n = 12, $\bar{X}$ = 0.04, SD = 0.2). I recorded the most mesopredator detections (Figure 17) in the hot-wet season (n = 166, $\bar{X}$ = 0.5, SD = 2.2), followed by the cold-dry season (n = 154, $\bar{X}$ = 0.4, SD = 1.8), and the lowest in the hot-dry season (n = 85, $\bar{X}$ = 0.2, SD = 0.9). I noted the same trend in the small predators (Figure 17) where I recorded the highest number of small predator detections in the hot-wet season (n = 77, $\bar{X}$ = 0.4, SD = 3.0), followed by the cold-dry season (n = 53, $\bar{X}$ = 0.3, SD = 1.9), and the lowest in the hot-dry season (n = 10,= 0.1, SD = 0.2).

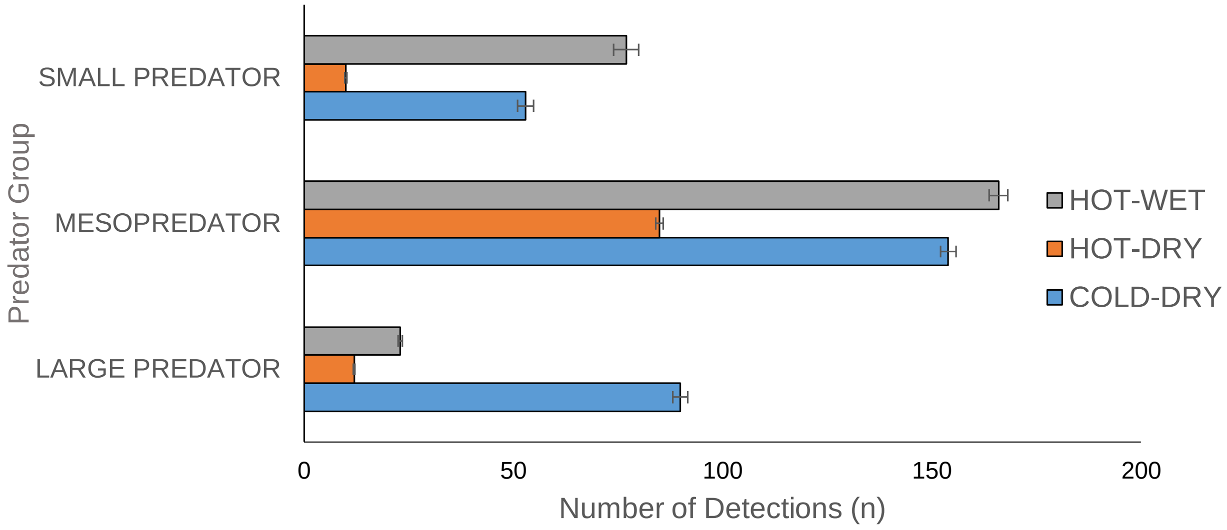

Figure 17: Total seasonal detections of predators recorded using camera traps, on Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

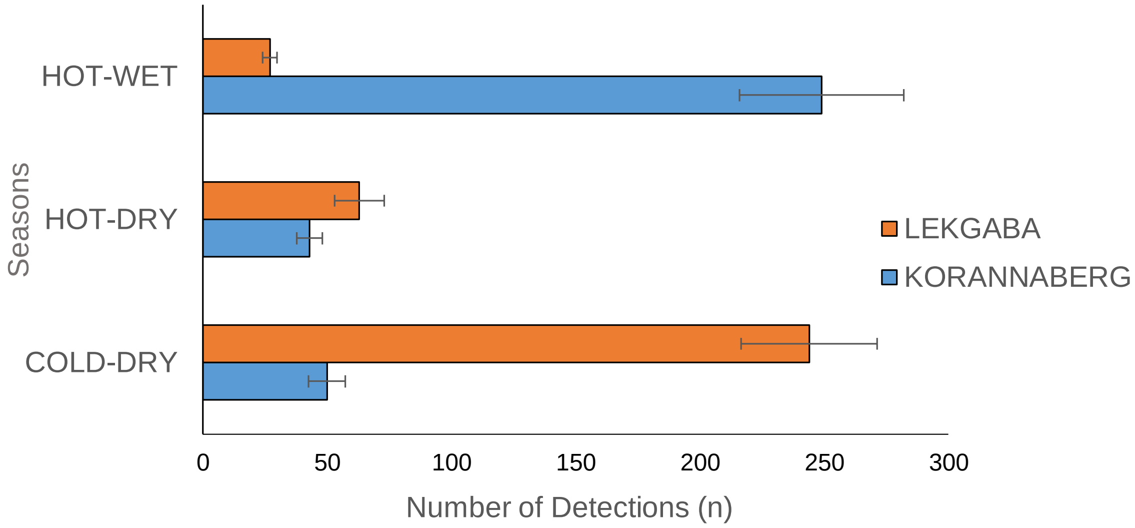

Figure 18: Seasonal predator detections recorded per section on the Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

For seasonal detections of predators by site (Figure 18) I recorded the highest number of detections for Korannaberg in the hot-wet season (n = 249, $\bar{X}$ = 17.4, SD = 33.1), followed by the cold-dry season (n = 50, $\bar{X}$ = 3.6, SD = 7.4), with the lowest number detections in the hot-dry season (n = 43, $\bar{X}$ = 3.1, SD = 5.1). On Lekgaba I recorded the highest number of detections in the cold-dry season (n = 244, $\bar{X}$ = 17.4, SD = 27.4), followed by the hot-dry season (n = 63, $\bar{X}$ = 4.5, SD = 10.0), with the lowest detections in the hot-wet season (n = 27, $\bar{X}$ = 1.9, SD = 2.8). I found that seasonal predator detections were normally distributed (W = 0.9, p = 0.418), and there was no significant difference ($p > 0.05$) in detections of predators by camera per season on Tswalu (Table 7).

Table 7: ANOVA comparing seasonal camera trap detections collected on Tswalu Kalahari Reserve between June 2017 and May 2018.

	df	Sum Sq	Mean Sq	F	p
Season	2	380520.0	190260.0	0.2	0.794
Residuals	3	2288956.0	762985.0		

On a species level, I found no significant seasonal variation in mesopredator detections from cameras ($p > 0.05$) (Table 8). My comparison of large, meso, and small predator detections (Figure 17) showed that small predator detections were highest in the hot-wet season (n = 77, $\bar{X}$ = 12.8, SD = 28.1), followed by the cold-dry season (n = 50, $\bar{X}$ = 8.3, SD = 14.6), with the lowest detections in the hot-dry season (n = 6, $\bar{X}$ = 1.0, SD = 1.5). I found mesopredator detections were highest in the hot-wet season (n = 176, $\bar{X}$ = 14.7, SD = 31.9), followed by the cold-dry season (n = 154, $\bar{X}$ = 12.8, SD = 25.6), with the lowest detections in the hot-dry season (n = 88, $\bar{X}$ = 7.3, SD = 11.0). My large predator detections were highest in the cold-dry season (n = 90, $\bar{X}$ = 9.0, SD = 19.5), followed by the hot-wet season (n = 23, $\bar{X}$ = 2.3, SD = 3.5), with the lowest detections in the hot-dry season (n = 12, $\bar{X}$ = 1.2, SD = 1.8). The black-backed jackal was the species that I detected most frequently (both year round and on a seasonal basis). The seasonal variation in mesopredator detections, as a collective, from the camera trap data was highest in the hot-wet season (Appendix I - Table A14).

Table 8: Seasonal comparison of mesopredator detections (per camera trap) using an ANOVA on Tswalu Kalahari Reserve between June 2017 and May 2018.

Predator Species	df	Sum Sq.	Mean Sq.	F	p
Large Predator					
Acinonyx jubatus	2	0.5	0.2	0.5	0.607
Hyaena brunnea	2	18111.0	9056.0	1.6	0.206
Lycaon pictus	2	40.0	20.0	1.0	0.370
Panthera leo	2	1586.0	792.8	1.2	0.303
Panthera pardus	2	6333.0	3167.0	0.9	0.418
Mesopredator					
Lupulella mesomelas	2	4774.0	2387.0	0.4	0.677
Caracal caracal	2	111.0	55.6	0.5	0.607
Felis silvestris lybica	2	81.0	40.6	0.5	0.584
Genetta genetta	2	2.0	1.0	2.0	0.134
Otocyon megalotis	2	66.0	32.9	0.5	0.609
Vulpes chama	2	0.5	0.2	0.0	0.973
Small Predator					
Cynictis penicillata	2	1904.0	952.1	1.0	0.372
Ictonyx striatus	2	1.5	0.7	1.0	0.366
Suricata suricatta	2	641.0	320.7	0.9	0.421

Table 9: Comparison of the relative abundance (per 100 trap nights) of predator detections (per camera trap) between seasons, using ANOVA, on Tswalu Kalahari Reserve between June 2017 and May 2018.

Predator Species	df	Sum Sq	Mean Sq	F	p
Lupulella mesomelas	2	48.0	23.9	0.4	0.677
Caracal caracal	2	0.0	0.0	0.5	0.607
Cynictis pencillata	2	43.0	21.4	1.0	0.372
Felis silvestris lybica	2	0.8	0.4	0.5	0.584
Genetta genetta	2	0.0	0.0	2.0	0.134
Hyaena brunea	2	1.8	0.9	1.6	0.206
Otocyon melagotis	2	1.5	0.7	0.5	0.609
Panthera pardus	2	0.6	0.3	0.9	0.418
Vulpes chama	2	0.0	0.0	0.3	0.938

I found that the seasonal index of predator abundance (detections per 100 trap nights) did not differ within seasons (cold-dry season: χ^2 = 140.0, df = 130, p = 0.259, hot-dry season: χ^2 = 112.0, df = 104, p = 0.279, and hot-wet season: χ^2 = 140.0, df = 130, p = 0.259). Seasonally, I found no significant difference between my indices of predator abundance (cold-dry season: χ^2 = 195.0, df = 182.0, p = 0.242, hot-dry: χ^2 = 150.0, df = 140, p = 0.267, and hot-wet season: χ^2 = 150.0, df = 140, p = 0.267) (Table 9).

I compared seasonal detections per section and found that predator detections in the cold-dry season were not significantly different between Korannaberg and Lekgaba. The seasonal predator detections were similar on Korannaberg and Lekgaba in the cold-dry season (t = -1.8, df = 15, p = 0.088) as well as for the hot-dry season (t = -0.5, df = 19, P = 0.638), and the in the hot-wet season, (t = 1.8, df = 13, p = 0.097).

For seasonal predator diversity (Appendix I – Table A15), I found that the predator richness was highest in the cold-dry season (S = 11.0), followed by the hot-wet and hot-dry seasons (S = 10.0). Yet, species diversity was consistent across all three seasons (H' = 0.1) (Figure 19). I compared seasonal detections of mesopredator diversity (Figure 20) and found that species richness was the same across all three seasons (S = 5.0), although species diversity did differ slightly, the highest diversity was in the hot-dry season (H' = 0.18), followed by the hot-wet season (H' = 0.15), and the lowest in the cold-dry season (H' = 0.14)

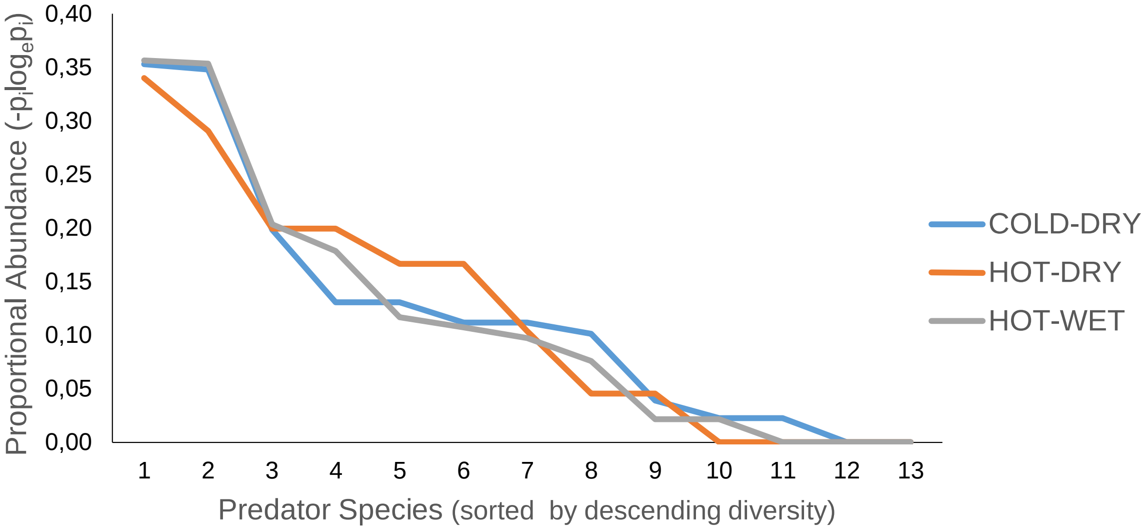

Figure 19: Seasonal relative diversity of predators detected on Tswalu Kalahari Reserve between June 2017 and May 2018.

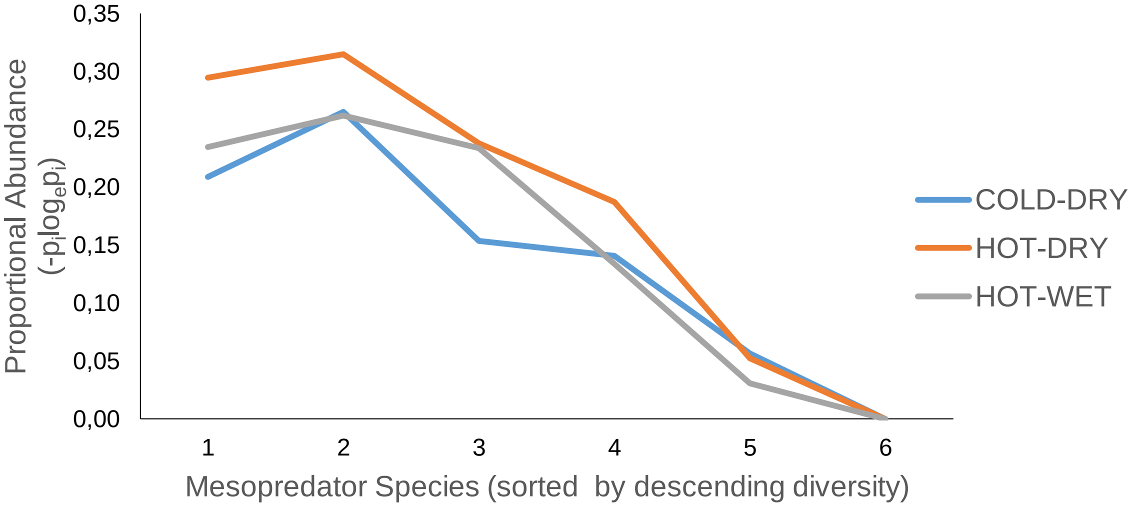

Figure 20: Seasonal relative diversity of mesopredator species detected on Tswalu Kalahari Reserve between June 2017 and May 2018.

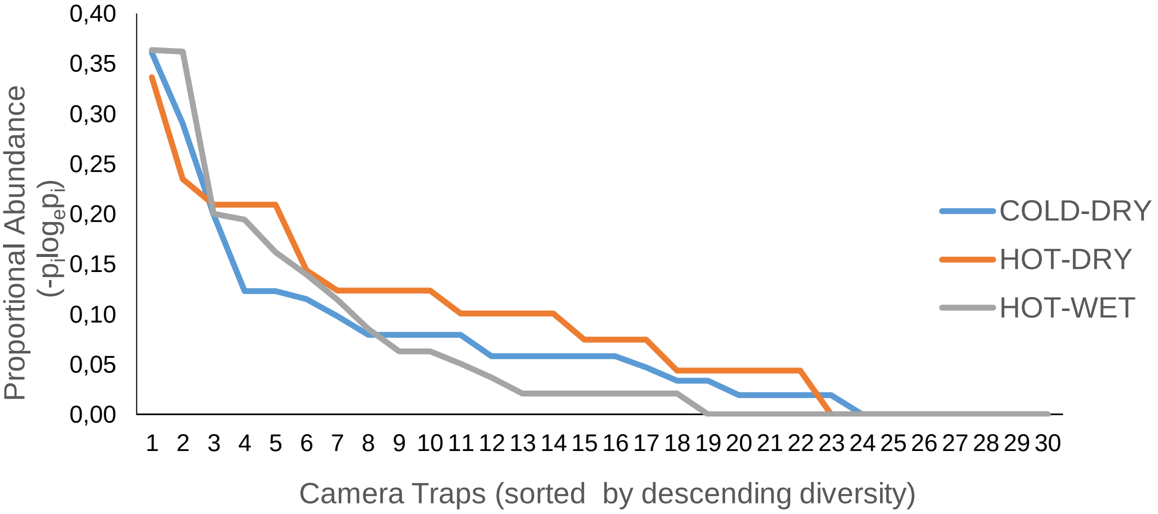

Figure 21: Seasonal relative diversity of predator species detected per camera trap on Tswalu Kalahari Reserve between June 2017 and May 2018.

I found that seasonal predator diversity per camera (Figure 21) indicated that richness was highest in the cold-dry season (S = 24.0), followed by the hot-dry season (S = 23.0), with the lowest richness in the hot-wet season (S = 19.0). Species diversity per camera trap was highest in the hot-dry season (H' = 0.09), with the diversity in the cold-dry and hot-wet seasons being equal (H' = 0.07).

I compared seasonal mesopredator species richness per camera (Figure 22) and found that species richness was highest in the hot-dry season (S = 23.0), followed by the cold-dry season (S = 22.0), with the lowest richness in the hot-wet season (S = 18.0). Species diversity was highest in the hot-wet season (H' = 0.09), followed by the cold-dry season (H' = 0.08), with the lowest diversity in the hot-dry season (H' = 0.06).

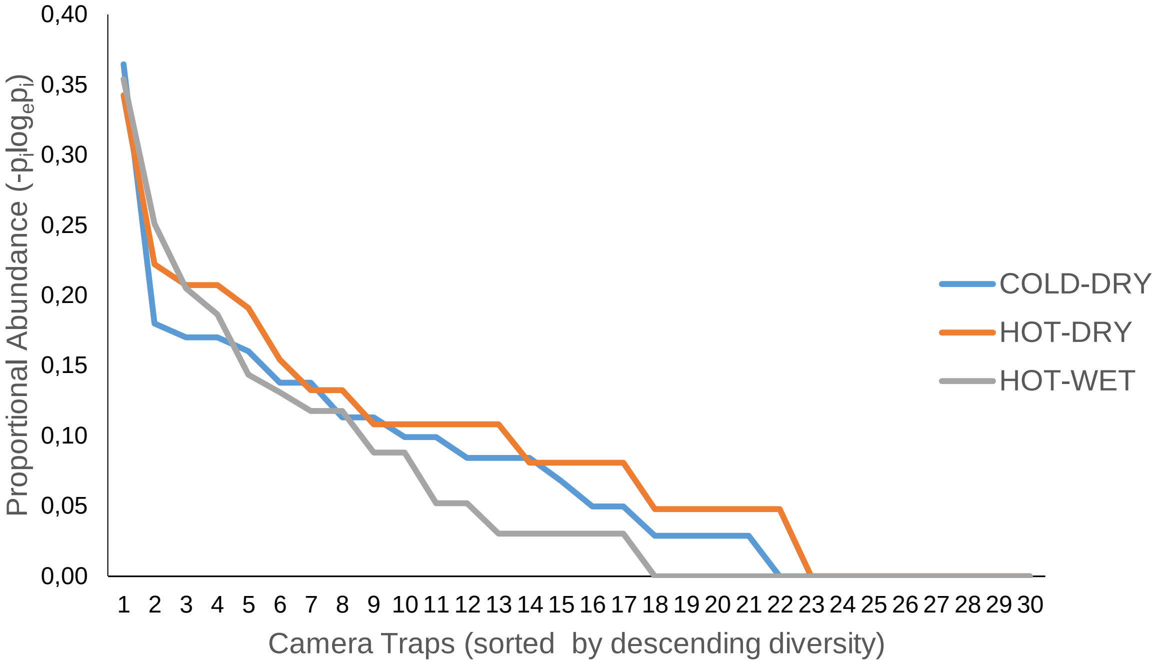

Figure 22: Seasonal relative diversity of mesopredators detected per camera trap on Tswalu Kalahari Reserve between June 2017 and May 2018.

I compared seasonal predator diversity and richness between Korannaberg and Lekgaba. In the cold-dry (Figure 23) season both Korannaberg and Lekgaba had equal richness scores (S = 11.0), but Lekgaba had a slightly higher diversity (H' = 0.12) than Korannaberg (H' = 0.11). In the hot-dry season (Figure 24) I calculated higher species richness values for Lekgaba (S = 9.0), than Korannaberg (S = 8.0). My species diversity values were also higher on Lekgaba (H' = 0.12), than Korannaberg (H' = 0.11). In the hot-wet season (Figure 25) I found species richness values were higher on Korannaberg (S = 10.0) than Lekgaba (S = 9.0). My estimates for species diversity values were higher on Lekgaba (H’ = 0.12) than Korannaberg (H’ = 0.11).

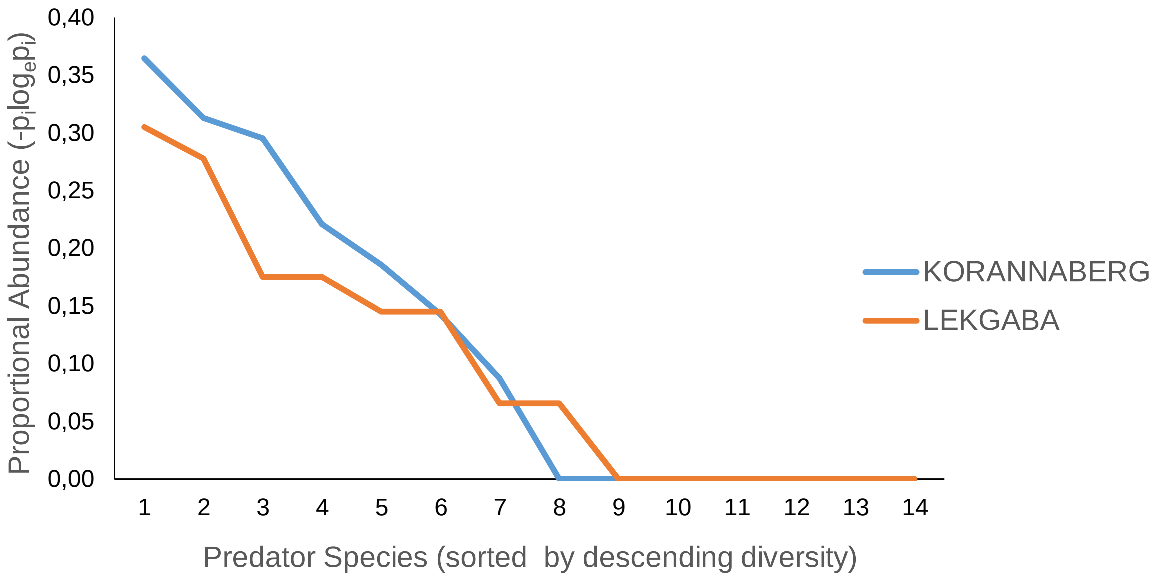

Figure 23: Predator relative diversity in the cold-dry season between Korannaberg and Lekgaba, recorded on Tswalu Kalahari Reserve from June 2017 to May 2018.

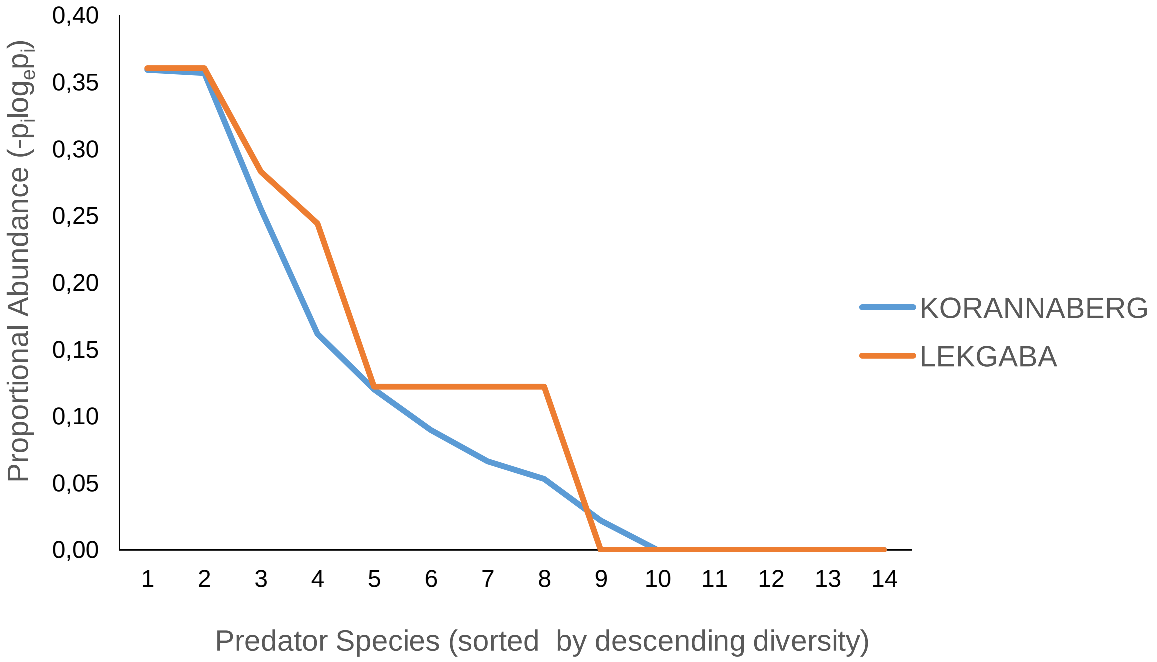

Figure 24: Predator relative diversity in the hot-dry season between Korannaberg and Lekgaba, recorded on Tswalu Kalahari Reserve from June 2017 to May 2018.

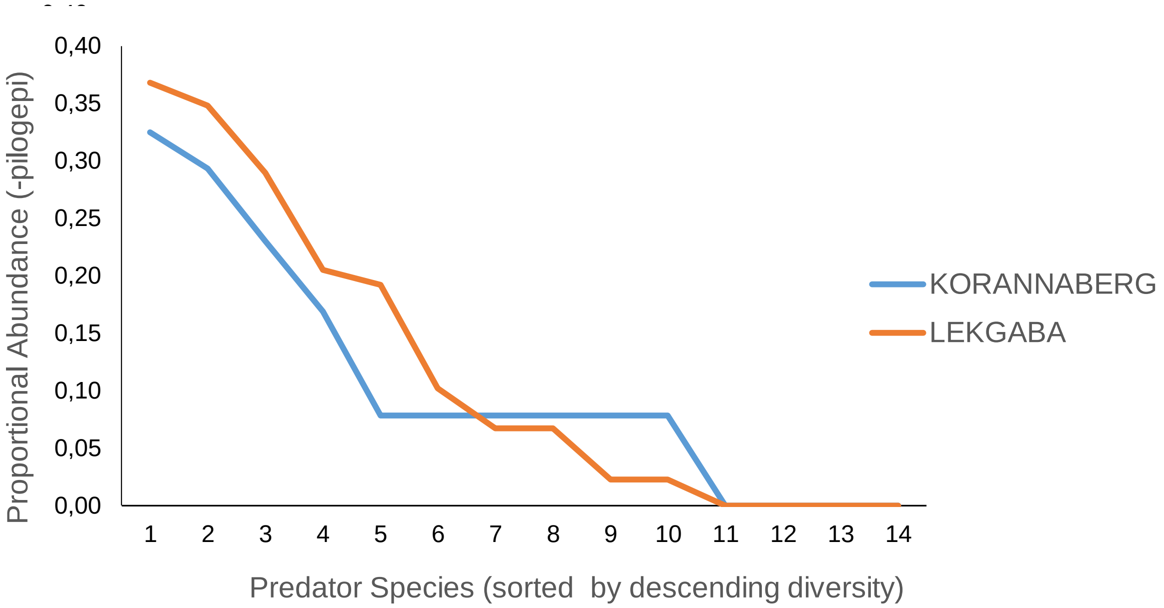

Figure 25: Predator relative diversity in the hot-wet season between Korannaberg and Lekgaba, recorded on Tswalu Kalahari Reserve from June 2017 to May 2018.

4.1.2 Ungulates

4.1.2.1 Korannaberg vs. Lekgaba comparison

I recorded a total of 26 281 detections of ungulates from 8441 camera trap clips, of these detections 78.2% fell in the large ungulate category (n = 20 559, $\overline{X}$ = 1142.2, SD = 1603.3) and 21.8% were classified as small ungulates (n = 5 722, $\overline{X}$= 508.6, SD = 753.8) (Appendix I - Table A16). I compared the ungulate detections on Korannaberg and Lekgaba and recorded 17.5% more ungulate detections on Lekgaba (n = 15 438, $\overline{X}$ = 13.2, SD = 89.5) than on Korannaberg (n = 10 843, $\overline{X}$ = 9.3, SD = 48.0) (Figure 26). I recorded more total ungulate detections (χ^2 = 64 071.0, df = 12, p < 0.001) and mean ungulate detections (χ^2 = 771.9, df = 12, p < 0.001) on Lekgaba than on Korannaberg. However, there was no significant difference between the ungulate detections (t = -0.7, df = 20, p = 0.498) between Korannaberg and Lekgaba.

When comparing large to small ungulates detections between sections, I found a higher detection of large to small ungulates on Lekgaba (71.2%) than Korannaberg (35.5%) (Figure 26). I recorded higher numbers of both large ungulate detections on Korannaberg (n = 7 345, $\overline{X}$ = 10.2, SD = 52.8) and Lekgaba (n = 13 214, $\overline{X}$ = 18.4, SD = 134.4), than small ungulate detections on Korannaberg (n = 3 498, $\overline{X}$ = 7.8, SD = 40.3) and Lekgaba (n = 2 224, $\overline{X}$ = 4.9, SD = 17.6).

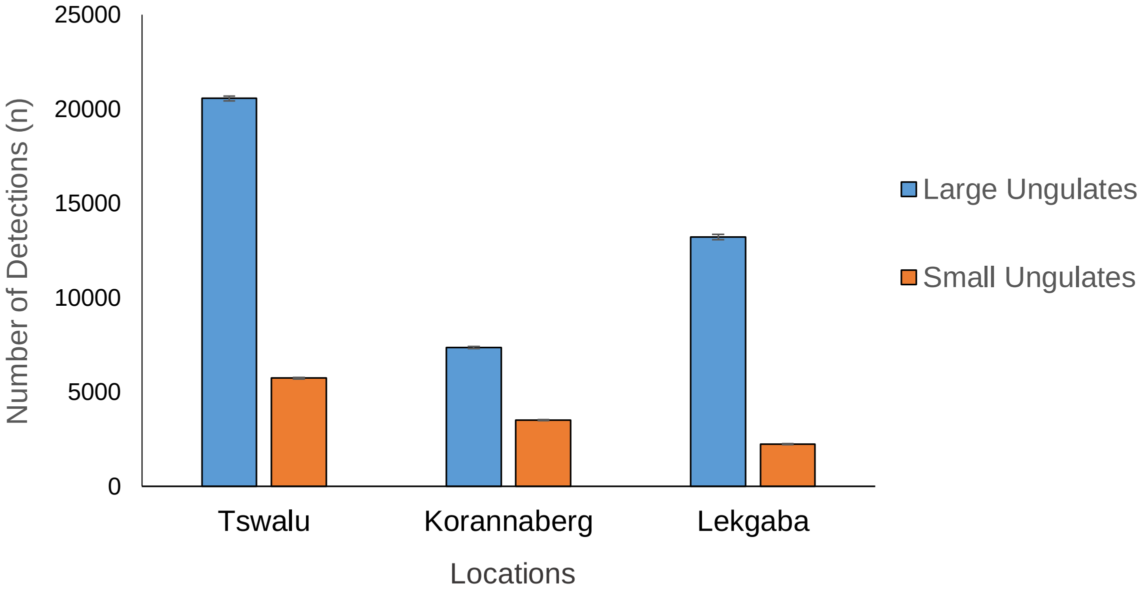

Figure 26: Large and small ungulate detections between sections recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

In my comparison of ungulate relative abundance (detections per 100 camera nights) (Appendix I - Table A18), I found that total (χ^2 = 156.0, df = 144, p = 0.234), and mean ungulate detections per 100 camera nights (χ^2 = 0.3, df = 1, p = 0.582) were not significantly different between Korannaberg and Lekgaba. I compared relative abundance on a species level (Table 10 & 12) and found that ungulate detections were not significantly different among camera locations (p >0.05). I found a significant difference in total ungulate detections between Korannaberg and Lekgaba (t = 2.9, df = 12.0, p = 0.011), however, the mean number of ungulate detections did not differ significantly (t = -0.7, df = 20, p = 0.498) between Korannaberg and Lekgaba.

Table 10: Results from Chi-squared test comparing relative abundance (detections per 100 trap nights) of ungulates between Korannaberg and Lekgaba recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

Ungulate Species	χ^2	df	p
Large Ungulates			
Alcelaphus caama	510.0	493	0.289
Connochaetes taurinus	690.0	667	0.261
Equus quagga	480.0	464	0.294
Equus zebra hartmannae	270.0	261	0.338
Giraffa camelopardalis	540.0	522	0.284
Oryx gazella	780.0	754	0.249
Taurotragus oryx	570.0	551	0.279
Tragelaphus strepsiceros	570.0	551	0.279
Small Ungulates			
Aepyceros melampus	390.0	377	0.311
Antidorcas marsupialis	600.0	580	0.274
Phacochoerus africanus	450.0	435	0.300
Raphicerus campestris	330.0	319	0.324
Sylvicapra grimmia	450.0	435	0.300

Ungulate detections were normally distributed for Korannaberg and Lekgaba (p >0.05) (Appendix I –Table A17). I included a comparison on ungulate detections per camera for Korannaberg (Appendix I – Tables A19) and Lekgaba (Appendix I – Table A20) and trap rates of ungulates per camera for Korannaberg (Appendix I – Table A21) and Lekgaba (Appendix I – Table A22). These showed no significant difference between total detections or detections per trap rate of ungulates recorded on Korannaberg and Lekgaba (t = -0.7, df = 20, p = 0.498) (Table 11 & 12).

Table 11: Outcome from t-test comparing the number of ungulate detections (per camera trap) between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

Ungulate Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Large Ungulate					
Alcelaphus caama	4.4	14.9	98	-1.2	0.226
Connochaetes taurinus	10.1	38.4	90	-0.8	0.431
Equus quagga	1.0	59.5	89	-1.8	0.242
Equus zebra hartmannae	1.0	6.7	91	-1.0	0.332
Giraffa camelopardalis	8.4	1.1	90	1.2	0.247
Oryx gazella	40.1	15.2	95	1.0	0.326
Taurotragus oryx	12.1	8.7	171	0.3	0.727
Tragelaphus strepsiceros	4.5	2.3	176	1.5	0.141
Small Ungulate					
Aepyceros melampus	3.4	4.8	173	-0.6	0.577
Antidorcas marsupialis	17.8	15.7	147	0.2	0.844
Phacochoerus africanus	16.1	2.0	91	1.4	0.158
Raphicerus campestris	0.4	0.7	153	-1.2	0.197
Sylvicapra grimmia	1.2	1.5	162	-0.4	0.699

[1] Lions absent; [2] Lions present

Table 12: Outcome from t-test comparing the relative abundance (per 100 trap nights) of ungulate detections (per camera trap) between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

Ungulate Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Large Ungulates					
Alcelaphus caama	29.0	99.0	98	-1.2	0.226
Connochaetes taurinus	67.1	256.0	90	-0.8	0.431
Equus quagga	6.5	396.4	89	-1.2	0.242
Equus zebra hartmannae	6.8	44.7	91	-1.0	0.332
Giraffa camelopardalis	56.0	7.6	90	1.2	0.247
Oryx gazella	267.6	101.6	95	1.0	0.326
Taurotragus oryx	80.9	57.9	171	0.3	0.727
Tragelaphus strepsiceros	30.1	15.6	176	1.5	0.141
Small Ungulates					
Aepyceros melampus	22.8	32.1	174	-0.6	0.577
Antidorcus marsupialis	118.4	104.5	147	0.2	0.844
Phacochoerus africanus	108.0	13.6	91	1.4	0.158
Raphicerus campestris	2.8	4.8	154	-1.3	0.197
Sylvicapra grimmia	7.7	9.7	163	-0.4	0.699

[1] Lions absent; [2] Lions present

I compared the diversity of large (Figure 27) and small (Figure 28) ungulates detected using cameras and found that species richness was equal for large ungulates on Korannaberg and Lekgaba (S = 8.0). Species diversity for large ungulates (H' = 0.2) was also equal for both sides of the reserve. Small ungulates had equal species richness (S = 5.0) and species diversity (H' = 0.2) for both sides of the reserve.

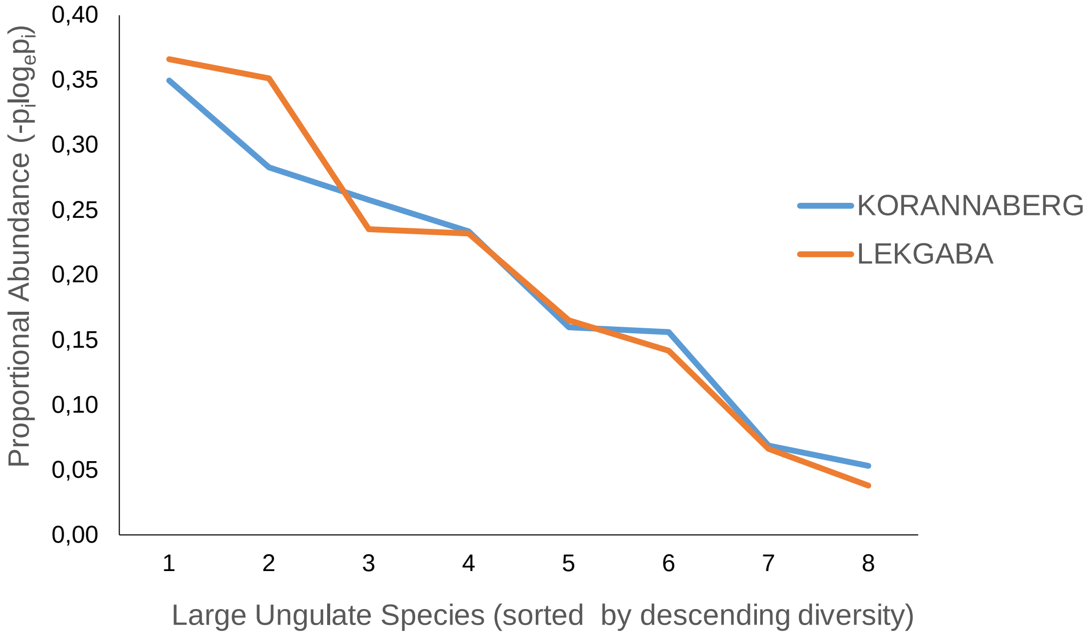

Figure 27: Relative diversity of large ungulates on detected Tswalu Kalahari Reserve between June 2017 and May 2018.

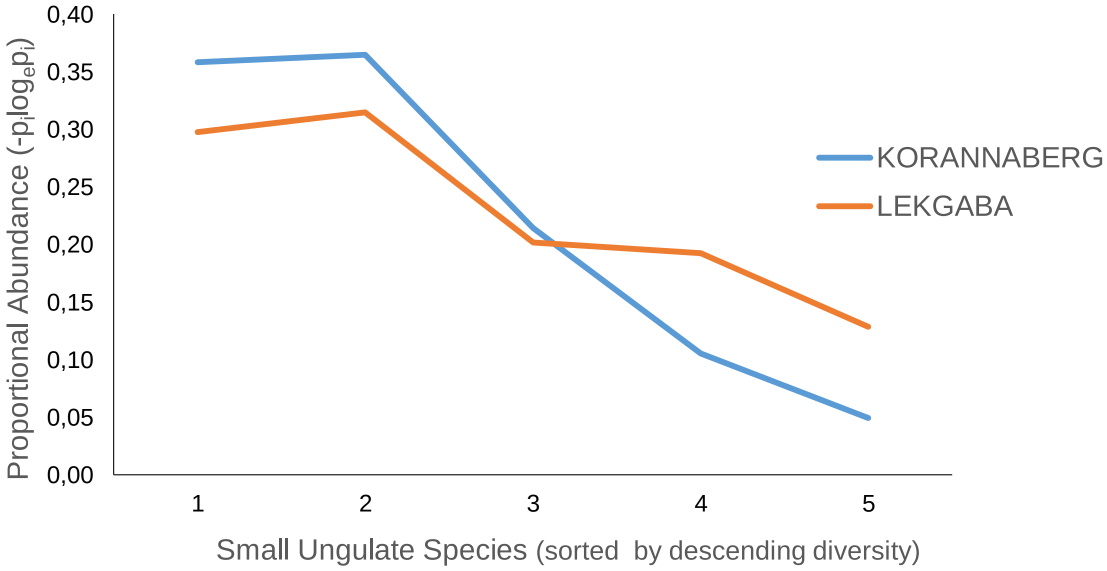

Figure 28: Relative diversity of small ungulates detected on Tswalu Kalahari Reserve between June 2017 and May 2018.

I compared ungulate I diversity per camera trap (Appendix I – Table A23) and found that Korannaberg (S = 29.0) had a slightly higher species richness, than Lekgaba (S = 28.0). Ungulate diversity per camera trap was lower on Lekgaba (H' = 0.05) than Korannaberg (H' = 0.08) (Figure 29).

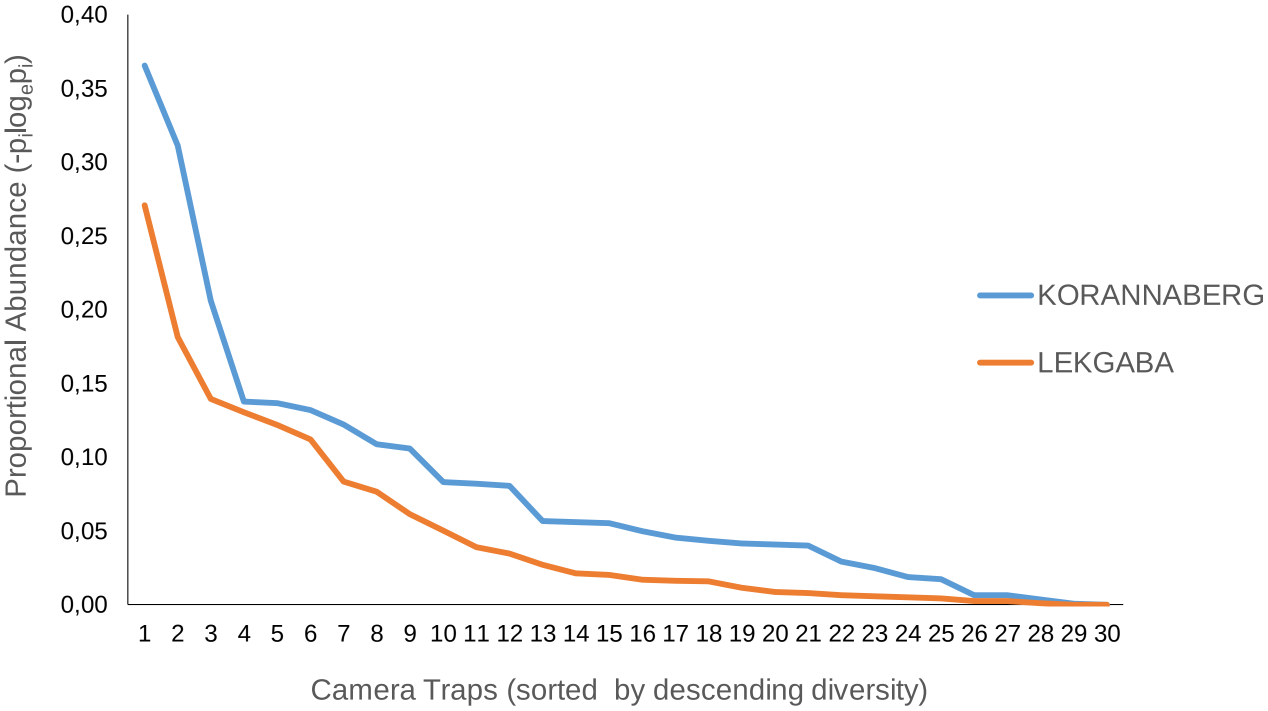

Figure 29: Relative diversity of detections of all ungulate species detected per camera trap on Tswalu Kalahari Reserve between June 2017 and May 2018.

4.1.2.2 Seasonal comparison

I found that ungulate detections from the cameras on TKR were highest in the cold-dry season (n = 14 663, $\overline{X}$ = 18.8, SD = 114.3), followed by the hot-wet season (n = 7 429, $\overline{X}$ = 9.5, SD = 51.7) with the lowest number of ungulate detections recorded in the hot-dry season (n = 4 189, $\overline{X}$ = 5.4, SD = 18.7) (Figure 30). My total number of seasonal ungulate detections (χ^2 = 10 085.0, df = 24, $p <$ 0.001) and mean number of detections per season (χ^2 = 168.5, df = 24, $p < 0.001$) were significantly different among the three seasons. I found that seasonal detections of ungulates differed significantly (t = 4.3, df = 38, $p < 0.001$).

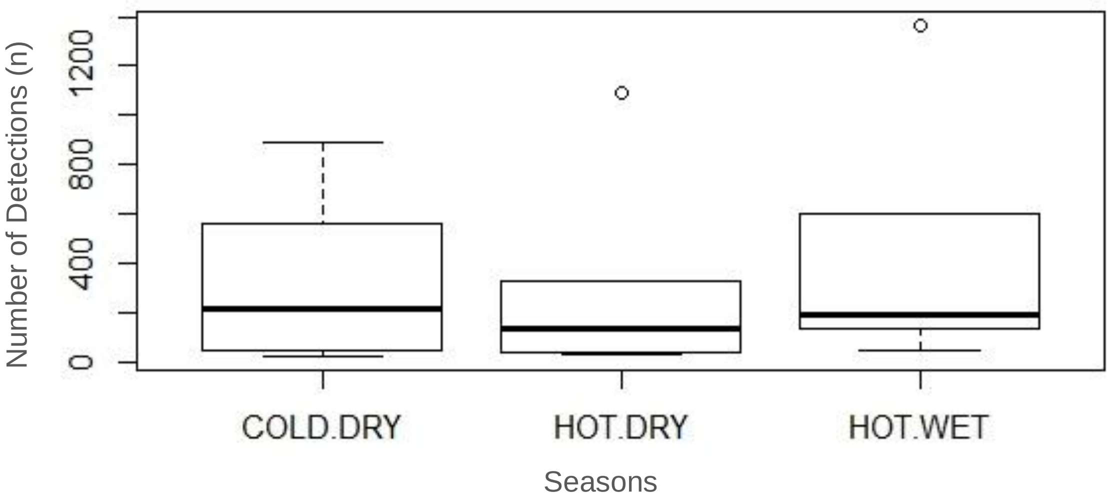

Figure 30: Seasonal variation in ungulate detections recorded on Tswalu Kalahari Reserve from June 2017 to May 2018.

I compared seasonal detections of large and small ungulates (Figure 31). I recorded the highest number of large ungulate detections in the cold-dry season (n = 12 914, $\overline{X}$ = 26.9, SD = 167.0), followed by the hot-wet season (n = 5 077, $\overline{X}$ = 10.6, SD = 59.9), with the lowest in the hot-dry season (n = 2 568, $\overline{X}$ = 5.4, SD = 19.1). I found small ungulate detections were highest in the hot-wet season (n = 2 352, $\overline{X}$ = 7.8, SD = 38.6), followed by the cold-dry season (n = 1 749, $\overline{X}$ = 5.8, SD = 167.0). Similar to large ungulates, I recorded the lowest number of small ungulate detections in the hot-dry season (n = 1 621, $\overline{X}$ = 5.4, SD = 18.0) (Appendix I -Table A25).

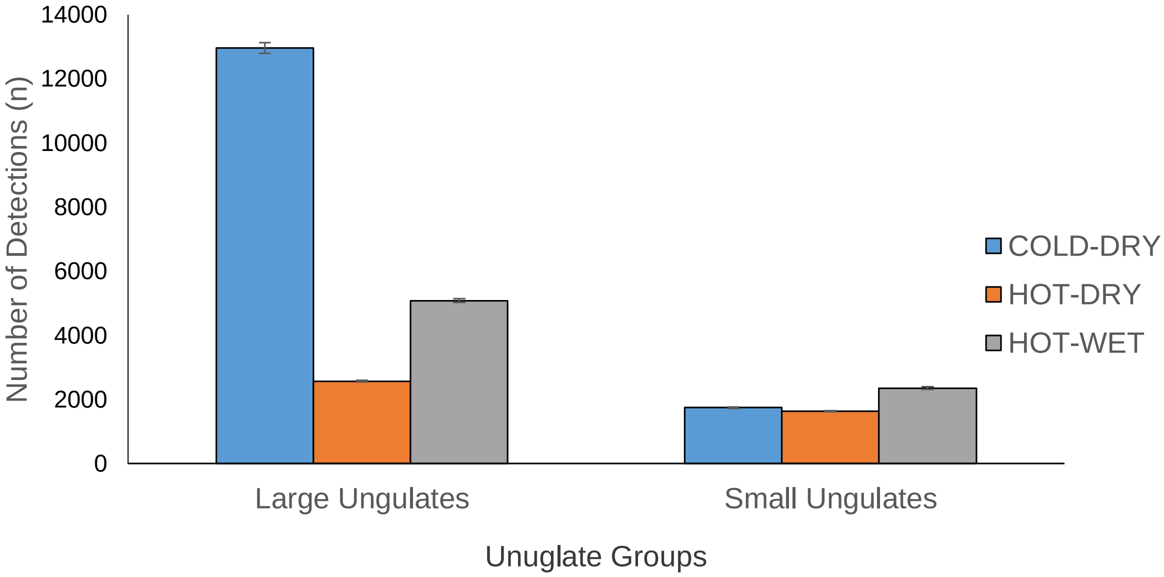

Figure 31: Seasonal comparison of large and small ungulate detections recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

*Note: Estimates are mean ± 1 standard deviation.

Table 13: Outcome from Kruskal-Wallis test comparing the ungulate detections (per camera trap) between seasons on Tswalu Kalahari Reserve between June 2017 and May 2018.

Ungulate Species	df	χ^2	p
Large Ungulates			
Alcelaphus caama	2	0.41	0.814
Connochaetes taurinus	2	1.31	0.521
Equus quagga	2	1.52	0.468
Equus zebra hartmannae	2	0.55	0.758
Giraffa camelopardalis	2	0.02	0.989
Oryx gazella	2	3.33	0.189
Taurotragus oryx	2	0.45	0.799
Tragelaphus strepsiceros	2	0.35	0.839
Small Ungulates			
Aepyceros melampus	2	0.77	0.678
Antidorcas marsupialis	2	1.48	0.477
Phacochoerus africanus	2	0.61	0.739
Raphicerus campestris	2	0.35	0.838
Sylvicapra grimmia	2	3.16	0.206

My seasonal data for ungulates were non-normal ($p < 0.05$). I found no significant difference in ungulate detections (Table 13) or relative abundance (detections per 100 trap nights) (Table 14) between seasons ($p > 0.05$).

I found that Korannaberg and Lekgaba had similar numbers of ungulate detections in all seasons: cold-dry season ($\chi^2 = 24.0$, df = 23, p = 0.404), hot-dry season ($\chi^2 = 26.0$, df = 24, p = 0.353), hot-wet season ($\chi^2 = 24.0$, df = 23, p = 0.404). There was no significant seasonal difference in the number of ungulate detections between Korannaberg and Lekgaba: (cold-dry season (t = -1.8, df = 13, p = 0.093), hot-dry season (t = 0.8, df = 13, p = 0.463), hot-wet season (t = 2.1, df = 12, p = 0.053)).

Table 14: Outcome of Kruskal-Wallis test comparing seasonal relative abundance (per 100 trap nights) of ungulates (per camera trap) on Tswalu Kalahari Reserve between June 2017 and May 2018.

Ungulate Species	df	χ^2	p
Large Ungulates			
Alcelaphus caama	2	3.65	0.162
Connochaetes taurinus	2	1.90	0.386
Equus quagga	2	1.52	0.468
Equus zebra hartmannae	2	0.63	0.729
Giraffa camelopardalis	2	1.09	0.580
Oryx gazella	2	3.33	0.189
Taurotragus oryx	2	0.45	0.799
Tragelaphus strepsiceros	2	0.35	0.839
Small Ungulates			
Aepyceros melampus	2	3.81	0.149
Antidorcus marsupialis	2	3.01	0.222
Phacochoerus africanus	2	0.61	0.739
Raphicerus campestris	2	0.35	0.838
Sylvicapra grimmia	2	3.16	0.206

My comparison of ungulate diversity showed that richness was equal in all three seasons (S = 13.0). Species diversity was also equal in all three seasons (H' = 0.2) (Figure 32). I compared seasonal ungulate detections per camera (Figure 33) and found I recorded the highest species richness (S = 29.0) in the cold-dry season, followed by the hot-dry season (S = 26.0) and the hot-wet season had the lowest richness (S = 25.0). I recorded the highest seasonal ungulate diversity per camera in the hot-dry season (H' = 0.09), followed by the hot-wet season (H' = 0.07), with the lowest richness in the cold-dry season (H' = 0.04).

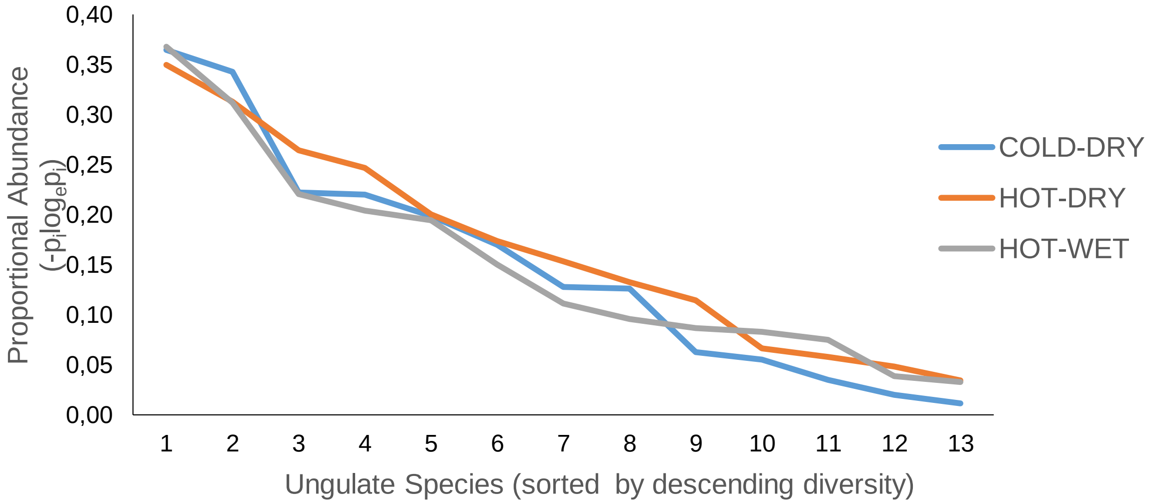

Figure 32: Seasonal relative diversity of ungulates detected on Tswalu Kalahari Reserve between June 2017 and May 2018.

My analyses of seasonal ungulate detections included a comparison between sections; during the cold-dry season (Figure 34) species richness was marginally higher on Korannaberg (S = 25.0) than Lekgaba (S = 24.0). Additionally, I recorded a higher species diversity on Korannaberg (H' = 0.08) than Lekgaba (H' = 0.05). In the hot-dry season (Figure 35). I found species richness was the same (S = 13.0) on Korannaberg and Lekgaba, whereas species diversity was higher on Korannaberg (H' = 0.15), than on Lekgaba (H' = 0.12). In the hot-wet season (Figure 36) I recorded a higher species richness on Lekgaba (S = 13.0), than Korannaberg (S = 12.0), as well as a higher species diversity on Lekgaba (H' = 0.13) than Korannaberg (H' = 0.10).

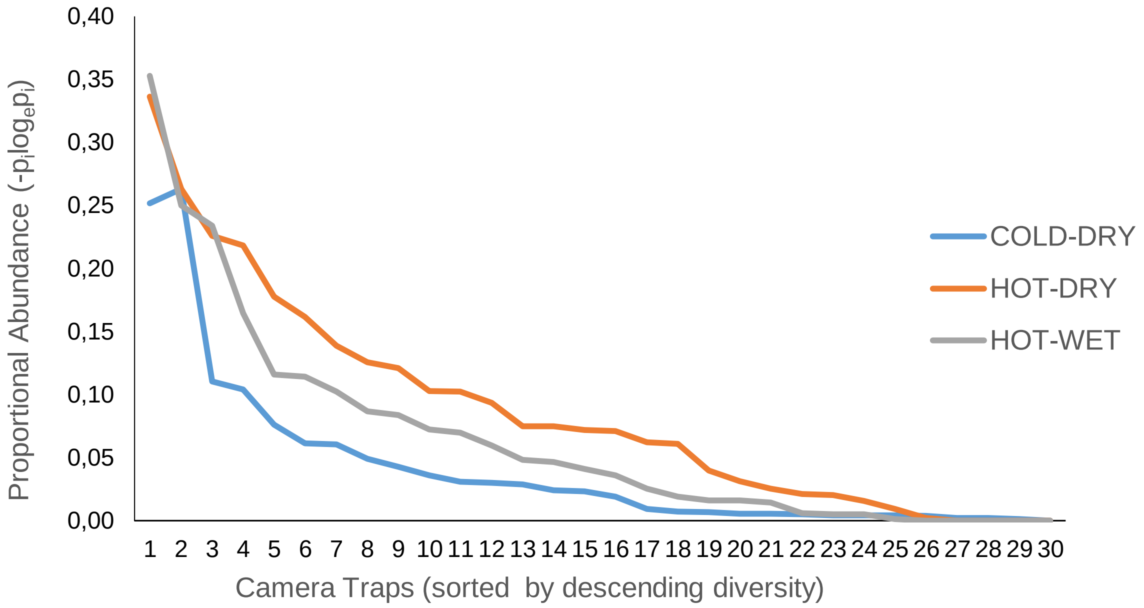

Figure 33: Seasonal relative diversity of ungulates detected per camera trap, on Tswalu Kalahari Reserve from June 2017 to May 2018.

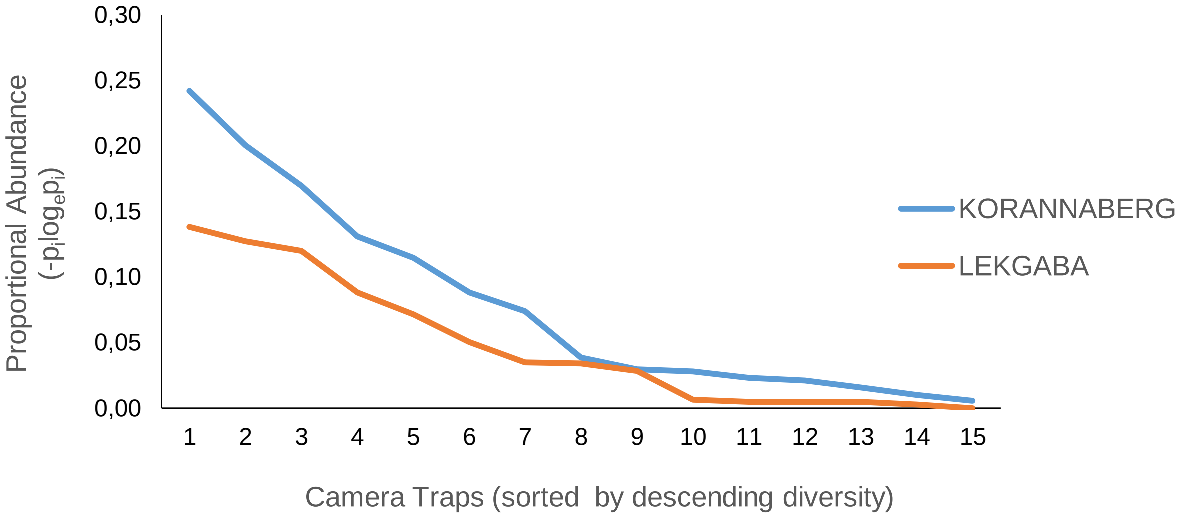

Figure 34: Relative diversity of ungulate detections in the cold-dry season between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

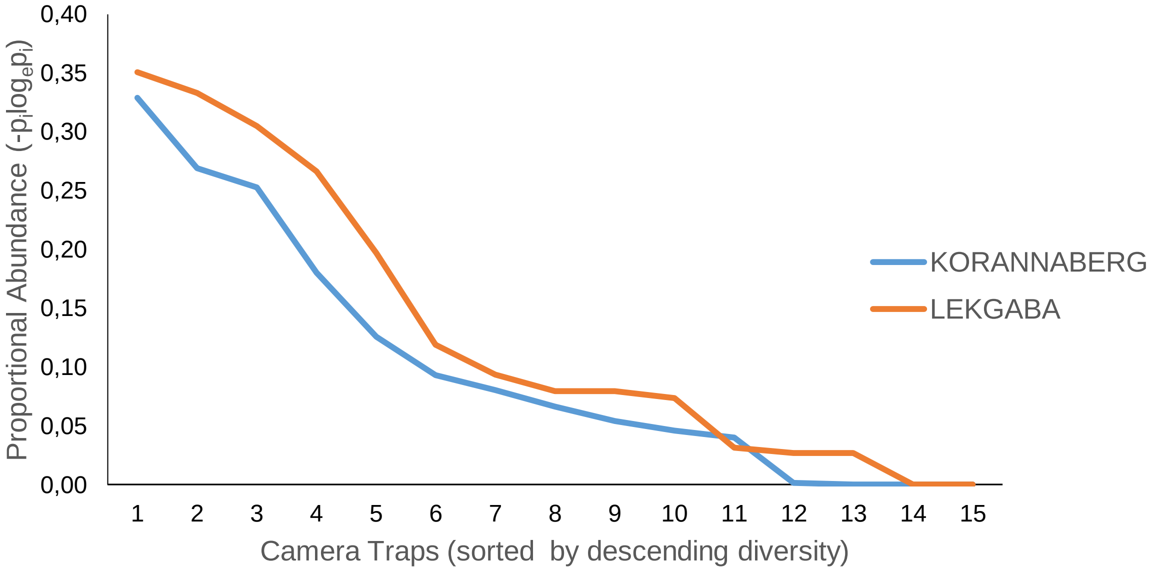

Figure 35: Relative diversity of ungulate detections in the hot-dry season between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

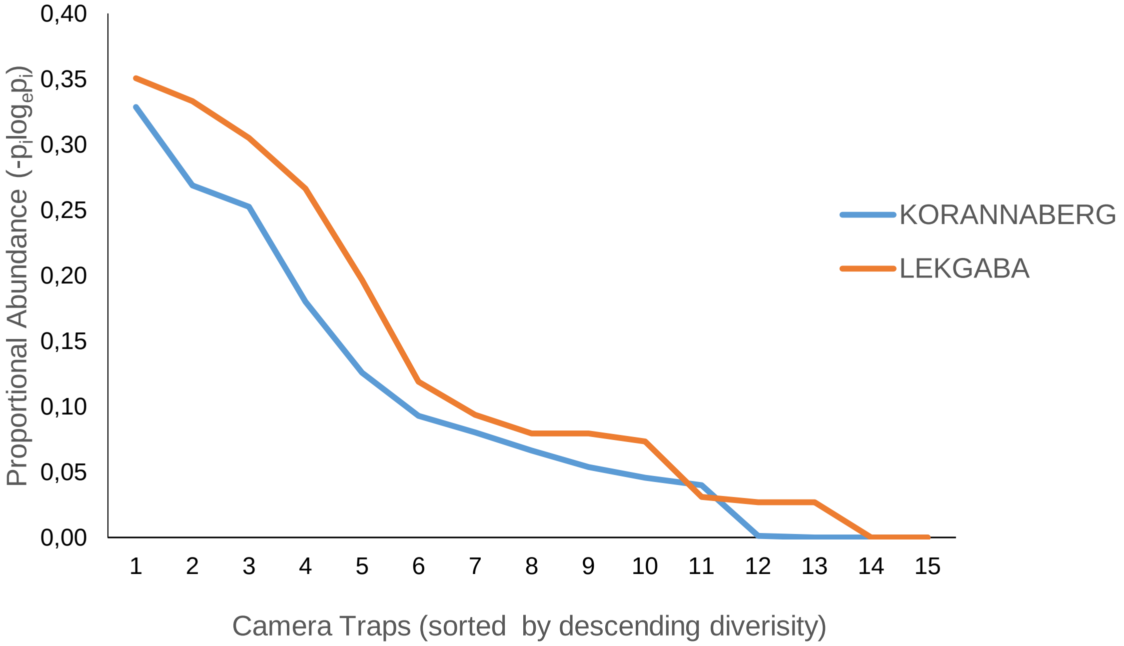

Figure 36: Relative diversity of ungulate detections in the hot-wet season between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

4.2 Small Mammals

4.2.1 Korannaberg vs. Lekgaba comparison

I sampled a total of 72 small mammal transects, of 30 traps each, (Figure 37) during my study (a total of 21600 trap nights). During this period, I made 1672 captures ($\bar{X}$ = 69.7, SD = 28.9), comprising 1319 individual small mammals from 15 species, with a mean capture rate of 3.7 captures per transect for the duration of the study. On Korannaberg, I made 795 captures comprising 613 individual small mammals from 10 species ($\bar{X}$ = 3.4, SD = 8.6), while on Lekgaba I made 877 captures comprising 706 individual small mammals from 14 species ($\bar{X}$ = 3.9, SD = 8.6) (Appendix I – Table A27).

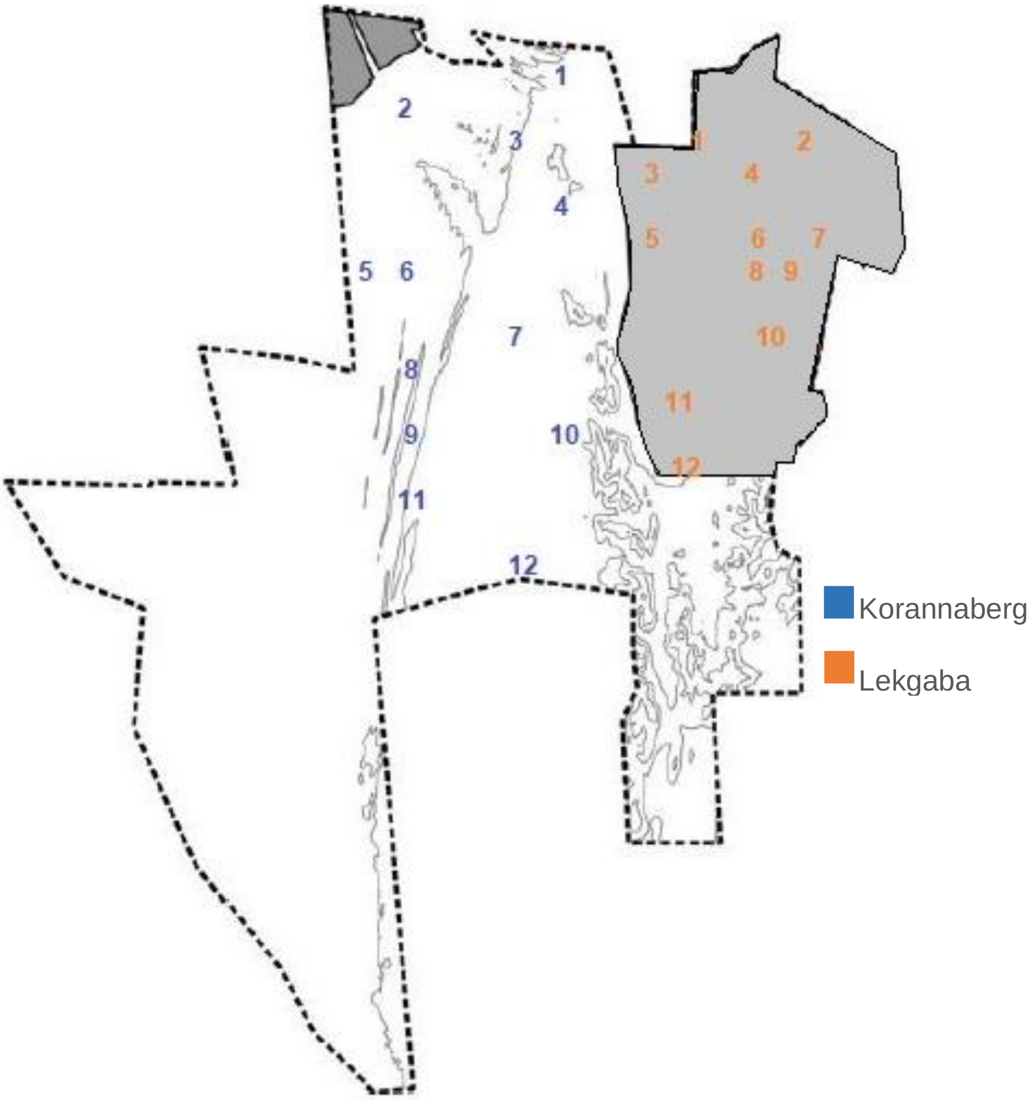

Figure 37: Locations of small mammal transects on Tswalu Kalahari Reserve between June 2017 and May 2018 (GPS co-ordinates in Appendix I – Table A26).

I found that even though my total number of small mammal captures was higher on Lekgaba (n = 877) than on Korannaberg (n = 795), my number of recaptures was marginally higher on Korannaberg (n = 181, $\overline{X}$ = 15.1, SD = 10.1) than Lekgaba (n = 171, $\overline{X}$ = 14.3, SD = 6.6) (Figure 38).

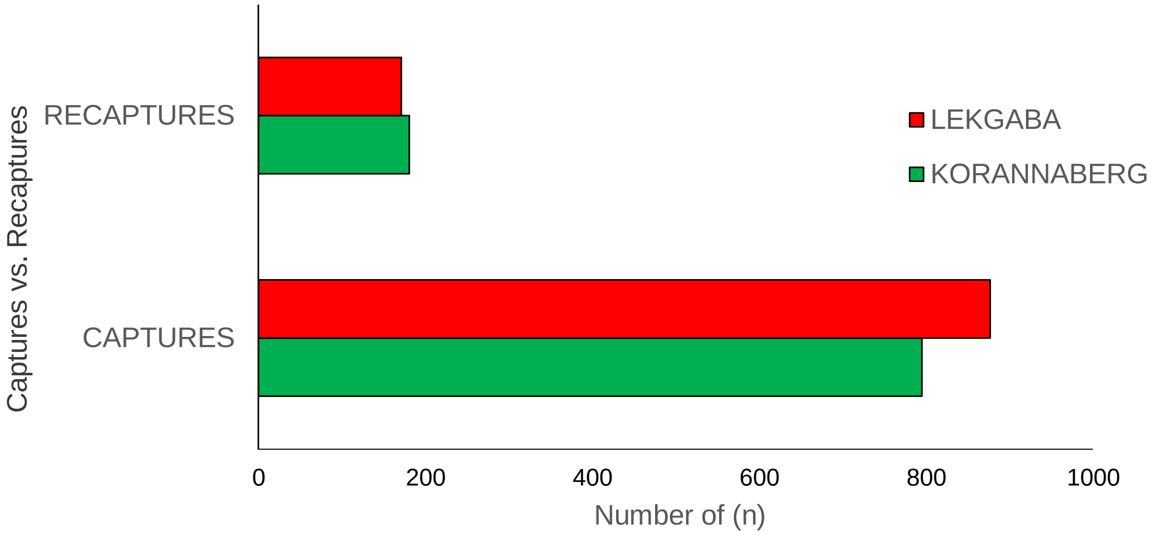

Figure 38: Comparison between the numbers of small mammals captured and those recaptured on Tswalu Kalahari Reserve from June 2017 to May 2018.

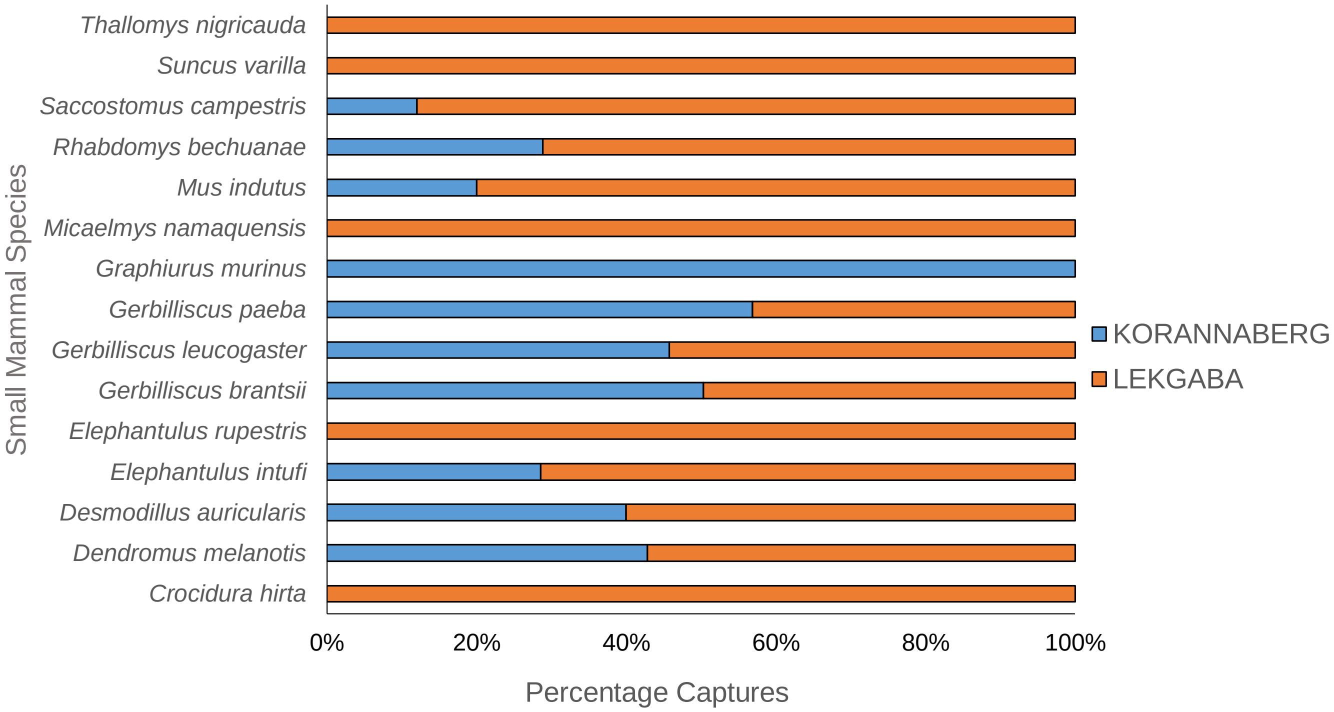

Figure 39: Percentage distribution of small mammal captures between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

I found the proportion of total small mammal captures ($\chi^2 = 210.0$, df = 196, p = 0.234) and mean number of small mammal captures (t = -0.2, df = 27, p = 0.839) did not differ significantly between Korannaberg and Lekgaba. The total number of captures I made per transect, for the duration of the study, ranged from 9 to 106 (Appendix I - Table A28), while mean captures per transect ranged from 0.6 to 7.1 individuals for the entire study. I found the mean number of captures per transect, for the duration of the study, was similar for both Korannaberg ($\bar{X} = 3.4$, SD = 8.6) and Lekgaba ($\bar{X} = 3.9$, SD = 8.6). I recorded the highest number of individual captures on transect 11 on Korannaberg (86 captures from five species) and on transect 17 on Lekgaba (106 captures from six species) (Appendix I - Table A28). The transect on which I captured the most species (n = 10) was also on Lekgaba. Yet, I found no significant difference in the captures per transect (t = -0.6, df = 22, p = 0.583) between Korannaberg and Lekgaba. The species for which I recorded the highest number of captures on TKR were the pygmy hairy-footed gerbil (*Gerbilliurus paeba* A. Smith, 1836) (n = 522), the bushveld gerbil (*Gerbilliscus leucogaster* Peters, 1852) (n = 411), and the highveld gerbil (*Gerbilliscus brantsii* A.Smith, 1836) (n = 169). The non-gerbilline species for which I recorded the highest number of captures was the pouched mouse (*Saccostomus campestris* Peters, 1846) (n = 58). I found that the species-specific capture rates were normally distributed ($p > 0.05$). I recorded a significant difference in total captures of only two small mammal species, the desert pygmy mouse (*Mus indutus* Thomas, 1910) and the pouched mouse ($p < 0.05$) (Table 15), between Korannaberg and Lekgaba.

My MARK population estimates (ESS) per transect (Appendix I – Table A27) were non-normally distributed (W = 0.9, p = 0.019). I found that the species-specific population estimates were significantly different for the TKR ($\chi^2 = 3551.2$, df = 14, $p < 0.001$), Korannaberg ($\chi^2 = 2314.2$, df = 14, $p < 0.001$), and Lekgaba ($\chi^2 = 1365.9$, df = 14, $p < 0.001$). Although I found no significant difference for the ESS between Korannaberg and Lekgaba (W = 85.0, p = 0.260), my comparison of species showed that the ESS for three of the species differed significantly between Korannaberg and Lekgaba, namely Desert pygmy mouse, west-central South Afriacn four-striped grass rat (*Rhabdomys bechuanae* Thomas Morong, N Britton, 1839) and pouched mouse (Table 16).

Table 15: Comparison of small mammal total captures between Korannaberg and Lekgaba, using a t-test, for the duration of the study on Tswalu Kalahari Reserve between June 2017 and May 2018.

Small Mammal Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Crocidura hirta	0.0	0.0	35	-1.0	0.324
Dendromus melanotis	0.1	0.1	68	-0.3	0.733
Desmodillus auricularis	0.1	0.2	49	-0.4	0.666
Elephantulus intufi	0.2	0.6	41	-0.9	0.384
Elephantulus rupestris	0.0	0.1	35	-1.7	0.103
Gerbilliscus brantsii	2.4	2.3	54	0.0	0.977
Gerbilliscus leucogaster	5.2	6.2	69	-0.6	0.527
Gerbillurus paeba	8.3	6.3	68	1.3	0.191
Graphiurus murinus	0.0	0.0	35	1.0	0.324
Micaelmys namaquensis	0.0	0.8	35	-1.7	0.095
Mus indutus	0.1	0.6	57	-2.4	0.022*
Rhabdomys bechuanae	0.4	1.1	45	-1.7	0.095
Saccostomus campestris	0.2	1.4	37	-2.6	0.014*
Suncus varilla	0.0	0.0	35	-1.0	0.324
Thallomys nigricauda	0.0	0.0	35	-1.0	0.324

[1]Lions absent; [2]Lions present

*Statistically significant results evaluated at α = 0.05

Table 16: Comparison using Mann-Whitney U-test on non-normally distributed ESS of small mammal captures between Korannaberg and Lekgaba from Tswalu Kalahari Reserve between June 2017 and May 2018.

Small Mammal Species	W	p
Crocidura hirta	0.0	1.000
Dendromus melanotis	14.0	0.533
Desmodillus auricularis	27.5	0.726
Elephantulus intufi	55.0	0.721
Elephantulus rupestris	0.0	0.059
Gerbilliscus brantsii	527.0	0.146
Gerbilliscus leucogaster	1539.0	0.245
Gerbillurus paeba	2213.5	0.400
Graphiurus murinus	1.0	1.000
Micaelmys namaquensis	0.0	0.064
Mus indutus	52.0	0.002*
Rhabdomys bechuanae	203.0	0.021*
Saccostomus campestris	110.5	0.003*
Suncus varilla	0.0	1.000
Thallomys nigricauda	0.0	1.000

*Statistically significant results evaluated at α = 0.05

Species diversity is made up of two indices: species diversity and richness. Small mammal species richness differed substantially between sites (Figure 40), whereby Korannaberg had a much lower richness (S = 10.0) compared with Lekgaba (S = 14.0). Additionally, Lekgaba (H' = 1.2) also had a higher species diversity than Korannaberg (H' = 0.09). Small mammal richness and diversity between transects (Figure 41) was similar for both Korannaberg and Lekgaba (S = 12.0; H' = 0.2).

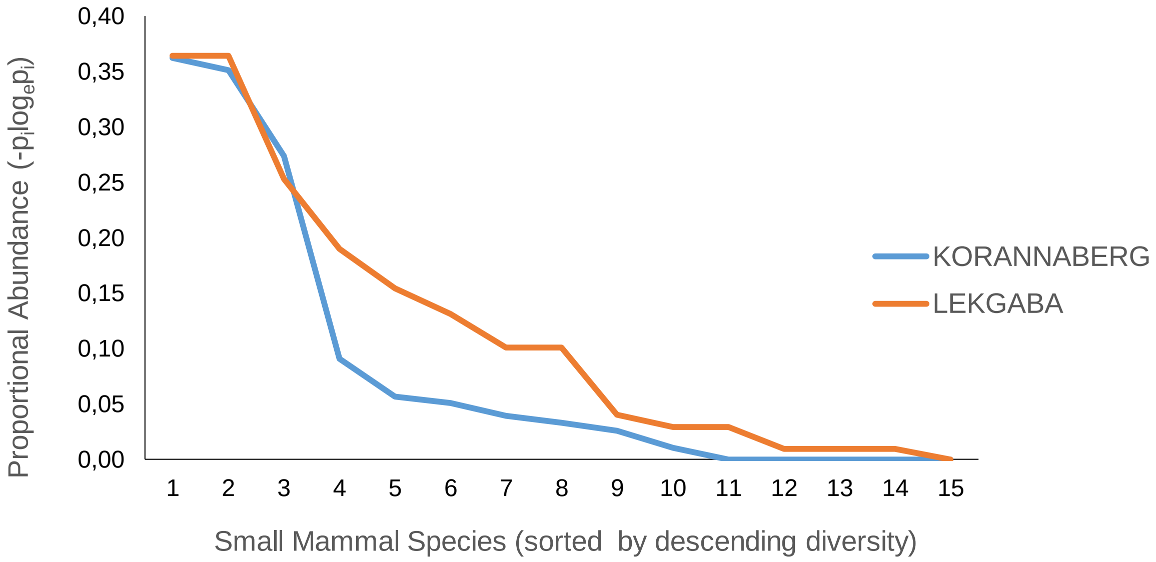

Figure 40: Relative diversity of small mammal species captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

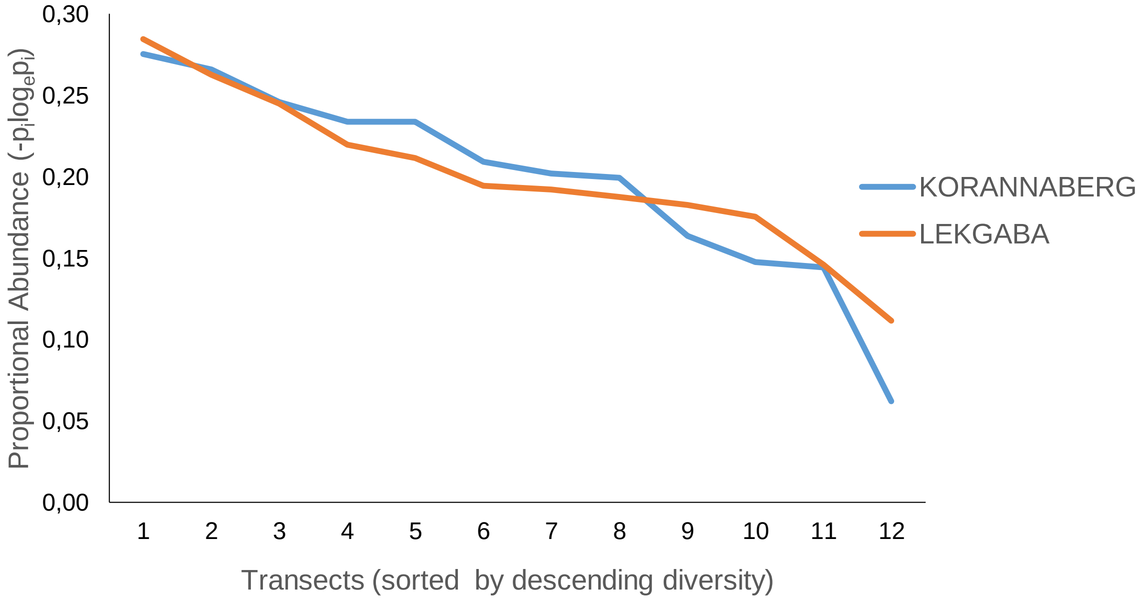

Figure 41: Relative diversity of small mammal species captured per transect on Tswalu Kalahari Reserve between June 2017 and May 2018.

4.2.2 Seasonal comparison (captures and population estimates)

I found that seasonal small mammal captures on TKR (Appendix I - Table A30) were highest in the cold-dry season (n = 714, $\overline{X}$ = 2.0, SD = 1.9) and the other two seasons returned similar numbers of captures (hot-dry; n = 318, $\overline{X}$ = 0.9, SD = 1.4 and hot-wet; n = 290, $\overline{X}$ = 0.9, SD = 1.3) (Figure 42 & 43). During the cold-dry season I captured more small mammals per transect on Korannaberg (n = 465, $\overline{X}$ = 38.8, SD = 16.8) than Lekgaba (n = 443, $\overline{X}$ = 36.9, SD = 13.1). However, during the warmer seasons I found the opposite. I made a higher number of small mammal captures on Lekgaba in the hot-dry (n = 212, $\overline{X}$ = 17.7, SD = 12.0) and hot-wet seasons (n = 222, $\overline{X}$ = 18.5, SD = 14.3) than Korannaberg in the hot-dry (n = 174, $\overline{X}$ = 14.5, SD = 10.5) and hot-wet seasons (n = 156, $\overline{X}$ = 13.0, SD = 9.1). I found that the total number of captures per transect differed significantly among seasons (χ^2 = 136.8, df = 28, $p < 0.05$), although the seasonal mean number of captures per transect did not differ (χ^2 = 5.7, df = 28; p = 1.000).

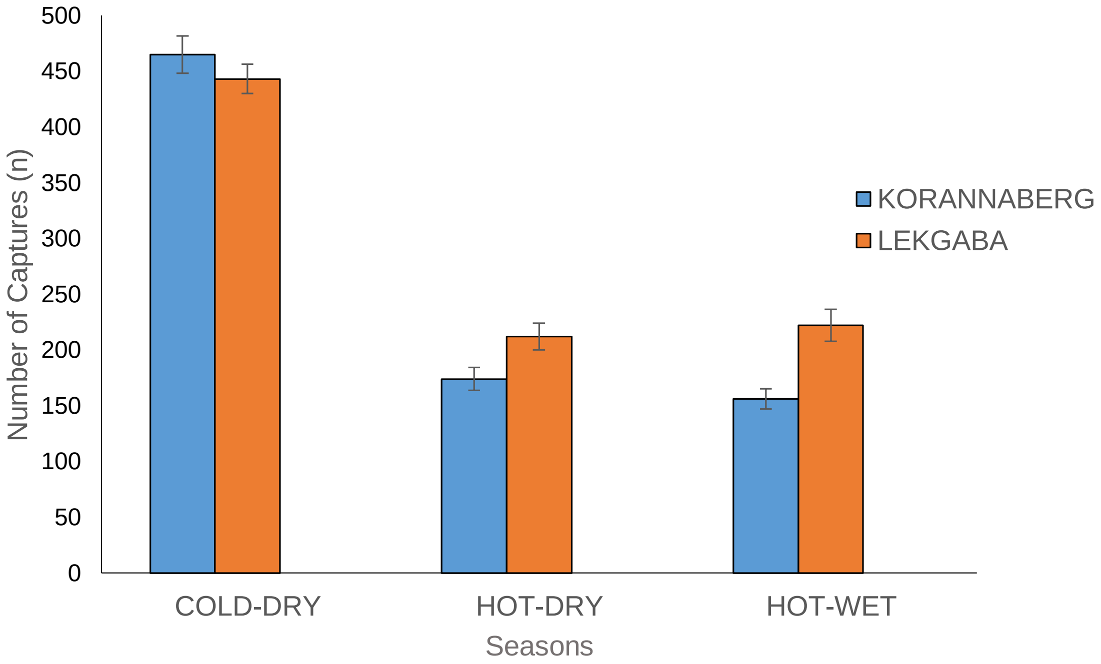

Figure 42: Seasonal variation in small mammals captures of on Tswalu Kalahari Reserve from June 2017 to May 2018.

*Note: Estimates are mean ± 1 standard deviation.

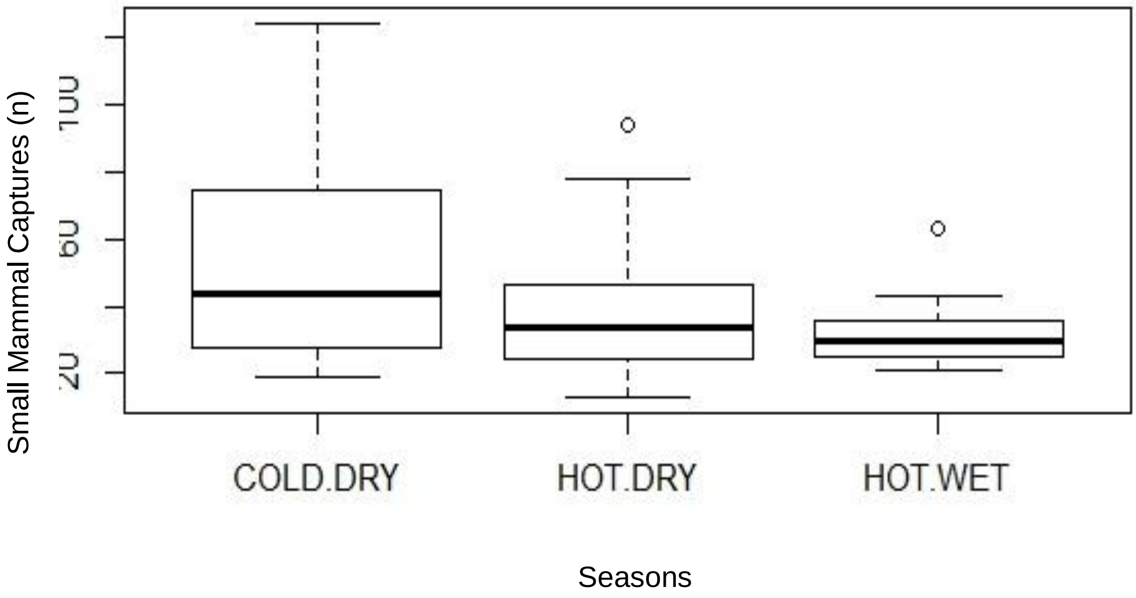

Figure 43: Seasonal distribution of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

The seasonal data that I collected for small mammals on TKR over the period of a year were normally distribution (W = 0.9; p = 0.992).I recorded a significant difference between seasonal captures of small mammals for TKR ($p < 0.001$) and captured significantly more small mammals in the cold-dry season than in either the hot-dry or the hot-wet seasons (Table 17).

Table 17: Tukey HSD comparison between seasons for small mammal captures on Tswalu Kalahari Reserve from June 2017 to May 2018.

Seasonal Comparison	Difference	Lower	Upper	p
Cold-dry vs Hot-dry	-21.8	-30.6	-12.9	< 0.001*
Cold-dry vs Hot-wet	-22.1	-30.9	-13.3	< 0.001*
Hot-dry vs Hot-wet	-0.3	-9.2	8.5	0.995

*Statistically significant results evaluated at α = 0.05

I found significant seasonal species-wise differences in the populations of gray climbing mouse (*Dendromus melanotis* A. Smith, 1834), Cape short-eared gerbil (*Desmodillus auricularis* A. Smith, 1834), bushveld gerbil, and pygmy hairy-footed gerbil (Table 18). I found that the seasonal ESS data conformed to a normal distribution for all three seasons: cold-dry (W = 1.0, p = 0.607), hot-dry (W = 1.0, p = 0.810), hot-wet (W = 1.0, p = 0.858). However, I found no significant difference in the total MARK ESS between the three seasons ($\chi^2 = 107.8$, df = 88, p = 0.075).

In spite of this, I found significant differences in seasonal ESS estimates for three species (bushveld gerbil, pygmy hairy-footed gebil, pouched mouse) (Table 19).

Table 18: ANOVA comparing seasonal captures of small mammal (per transect) on Tswalu Kalahari Reserve between June 2017 and May 2018.

Small Mammal Species	df	Sum Sq	Mean Sq	F	p
Crocidura hirta	2	0.0	0.0	1.0	0.373
Dendromus melanotis	2	0.9	0.4	4.0	0.023*
Desmodillus auricularis	2	2.0	1.0	3.8	0.028*
Elephantulus intufi	2	3.7	1.8	0.7	0.495
Elephantulus rupestris	2	0.0	0.0	0.2	0.847
Gerbiliscus brantsii	2	83.5	41.8	2.7	0.072
Gerbiliscus leucogaster	2	1008.0	504.1	17.8	<0.001*
Gerbillurus paeba	2	767.5	383.8	11.2	<0.001*
Graphiurus murinus	2	0.0	0.0	1.0	0.373
Micaelmys namaquensis	2	0.9	0.4	0.1	0.902
Mus indutus	2	3.0	1.5	2.7	0.072
Rhabdomys bechuanae	2	5.8	2.9	1.1	0.330
Saccostamus campestris	2	4.2	2.1	0.5	0.626
Suncus varilla	2	0.0	0.0	1.0	0.373
Thallomys nigricauda	2	0.0	0.0	1.0	0.373

*Statistically significant results evaluated at α = 0.05

Table 19: ANOVA comparing seasonal captures of the calculated ESS of small mammals (per transect), using the Cormack Jolly-Seber model, on Tswalu Kalahari Reserve between June 2017 and May 2018.

Small Mammal Species	df	Sum Sq	Mean Sq	F	p
Crocidura hirta	2	0.7	0.3	-	-
Dendromus melanotis	2	1.2	0.6	3.0	0.160
Desmodillus auricularis	2	1.5	0.8	2.9	0.146
Elephantulus intufi	2	10.6	5.3	0.9	0.433
Elephantulus rupestris	2	0.7	0.3	-	-
Gerbilliscus brantsii	2	16.2	8.1	0.6	0.540
Gerbilliscus leucogaster	2	537.8	268.9	12.8	< 0.001*
Gerbillurus paeba	2	266.6	133.3	5.5	0.006*
Graphiurus murinus	2	0.7	0.3	-	-
Micaelmys namaquensis	2	18.7	9.3	-	-
Mus indutus	2	0.8	0.4	0.8	0.455
Rhabdomys bechuanae	2	3.6	1.8	0.7	0.500
Saccostomus campestris	2	73.2	36.6	11.1	0.001*
Suncus varilla	2	0.7	0.3	-	-
Thallomys nigricauda	2	0.7	0.3	-	-

*Statistically significant results evaluated at α = 0.05

I recorded significantly more captures of gray climbing mouse in the cold-dry than the hot-wet season (Table 20). I found that the captures of Cape short-eared gerbil were significantly higher in the cold-dry than the hot-wet season (Table 20). I found that the captures of bushveld gerbil and pygmy hairy-footed gerbil were significantly higher in the cold-dry season than both the hot-dry and hot-wet seasons (Table 20). For the seasonal differences in the ESS of the small mammals captured, I found that the bushveld gerbil population estimates were significantly higher in the cold-dry season than both the hot-dry and hot-wet seasons (Table 21). I found that the pygmy hairy-footed gerbil populations had significantly higher population estimates in the cold-dry than the hot-dry seasons, and significantly higher estimates in the hot-wet than the hot-dry seasons (Table 21). My calculations of the pouched mouse population estimates showed that estimates were significantly higher in the hot-dry than the cold-dry season and significantly higher in the hot-wet season than the hot-dry season (Table 21).

Table 20: Tukey HSD comparison for small mammal species with significant differences between seasonal captures on Tswalu Kalahari Reserve from June 2017 to May 2018.

Small Mammal Species	Seasons	Difference	Lower	Upper	p
	Cold-dry vs Hot-dry	-0.21	-0.44	0.02	0.079
Dendromus melanotis	Cold-dry vs Hot-wet	-0.25	-0.48	-0.02	0.028*
	Hot-dry vs Hot-wet	-0.04	-0.27	0.19	0.899
	Cold-dry vs Hot-dry	-0.33	-0.69	0.03	0.074
Desmodillus auricularis	Cold-dry vs Hot-wet	-0.38	-0.73	0.32	0.039*
	Hot-dry vs Hot-wet	-0.04	-0.40	0.32	0.958
	Cold-dry vs Hot-dry	-8.25	-11.93	-4.57	<0.001*
Gerbiliscus leucogaster	Cold-dry vs Hot-wet	-7.58	-11.27	-3.90	<0.001*
	Hot-dry vs Hot-wet	0.67	-3.02	4.35	0.902
	Cold-dry vs Hot-dry	-7.13	-11.17	-3.08	<0.001*
Gebillurus paeba	Cold-dry vs Hot-wet	-6.71	-10.75	-2.67	<0.001*
	Hot-dry vs Hot-wet	0.42	-3.63	4.46	0.967

*Statistically significant results evaluated at α = 0.05

Table 21: Tukey HSD comparison for the ESS of small mammal species with significant differences between seasonal population estimates, using the Cormack Jolly-Seber model, on Tswalu Kalahari Reserve from June 2017 to May 2018.

Small Mammal Species	Seasons	Difference	Lower	Upper	p
Gerbiliscus leucogaster	Cold-dry vs Hot-dry	-5.82	-9.20	-2.45	< 0.001*
	Cold-dry vs Hot-wet	-6.58	-10.18	-2.99	< 0.001*
	Hot-dry vs Hot-wet	-0.76	-4.46	2.94	0.874
Gerbillurus paeba	Cold-dry vs Hot-dry	-4.04	-7.73	-0.36	0.028*
	Cold-dry vs Hot-wet	-4.34	-7.83	-0.86	0.011*
	Hot-dry vs Hot-wet	-0.30	-4.06	3.45	0.980
Saccostomus campestris	Cold-dry vs Hot-dry	6.71	3.00	10.43	0.001*
	Cold-dry vs Hot-wet	0.71	-1.49	2.92	0.692
	Hot-dry vs Hot-wet	-6.00	-9.54	-2.46	0.001*

*Statistically significant results evaluated at $\alpha = 0.05$

I compared seasonal small mammal captures by sections and found no significant difference in captures between Korannaberg and Lekgaba (t = 0.3, df = 21, p = 0.769) in the cold-dry season. In the hot-dry season I found no significant difference in captures between Korannaberg and Lekgaba (t = -0.7, df = 21, p = 0.499). Additionally, I found no significant differences in captures between Korannaberg and Lekgaba (t = -1.1, df = 19, p = 0.275) in the hot-wet season.

For small mammal seasonal diversity (Figure 44), I recorded the highest species richness in the hot-dry season (S = 13.0), followed by the cold-dry season (S = 12.0) with the lowest richness in the hot-wet season (S = 10.0). However, species diversity was equal across all three season (H' = 0.1).

Small mammal species richness (S = 24.0) and diversity (H' = 0.1) was equal across transects for all three seasons (Figure 45).

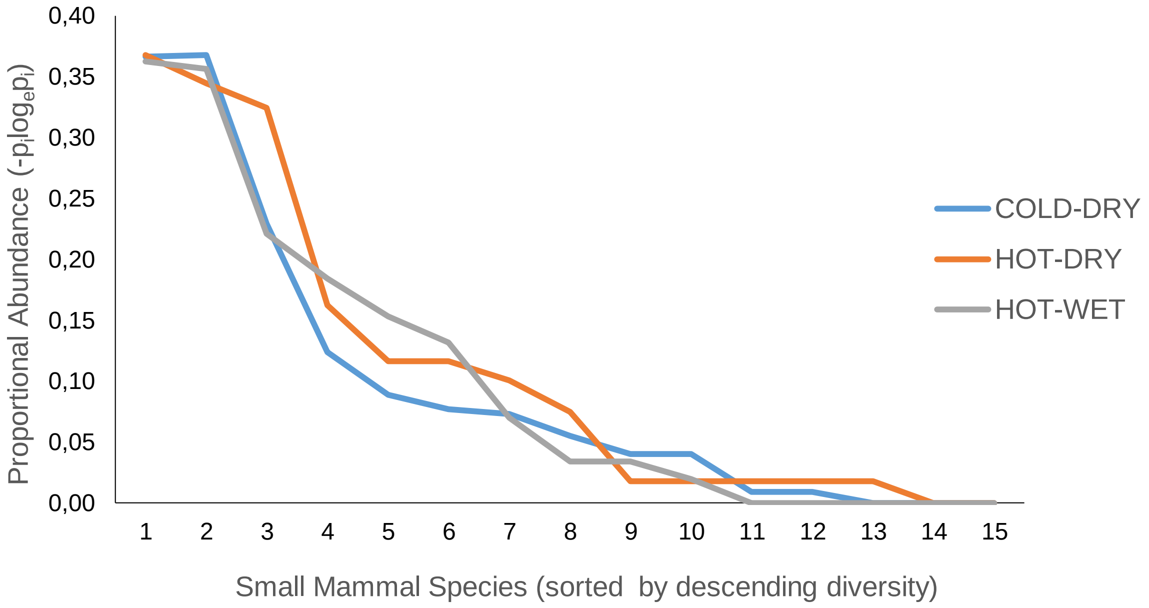

Figure 44: Seasonal relative diversity of the small mammal species captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

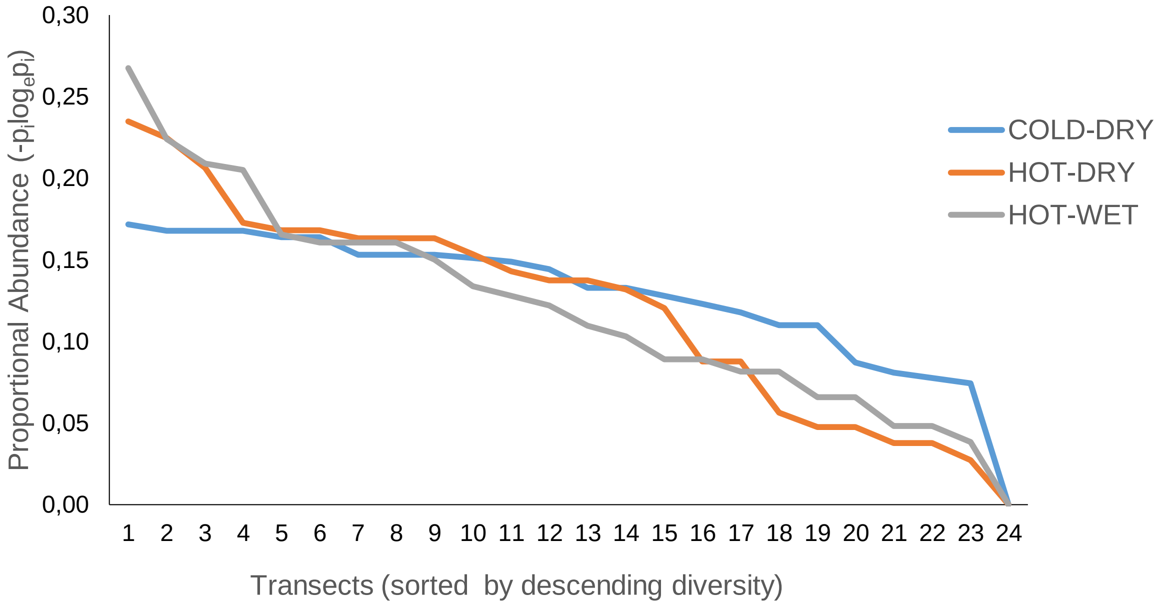

Figure 45: Seasonal relative diversity of the small mammals captured, per transect on Tswalu Kalahari Reserve between June 2017 and May 2018.

4.2.3 Dietary functional groups

I categorised the small mammals based on their functional groups by comparing their main food source that they consume – majority of species are omnivorous eating mostly seeds, but also eat other parts of plants and several species of insects (Table 22). I compared the three functional groups by site (Figure 46) and season (Figure 47). The capture data for the three functional groups conformed to a normal distribution (W = 1.0, p= 0.933). I found no significant difference in captures for any of the functional groups between Korannaberg and Lekgaba (herbivores: t = -1.1, df = 1, p = 0.478, insectivores: t = -0.9, df = 4, p = 0.416, omnivores: t = -0.1, df = 15, p = 0.918). Additionally, I found no significant differences between seasonal captures of the herbivores, insectivores or omnivores (p > 0.05 – Table 23).

Table 22: Feeding preferences of small mammals (Skinner and Chimimba 2005; Stuart and Stuart 2015) captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

Small Mammal Species	Commmon Name	General Dietry Group
Desmodillus auricularis	Cape short-eared gerbil	Omnivorous
Gerbilliscus brantsii	Highveld gerbil	Omnivorous
Gerbilliscus leucogaster	Bushveld gerbil	Omnivorous
Gerbilliurus paeba	Pygmy hairy-footed gerbil	Omnivorous
Graphiurus murinus	Woodland dormouse	Omnivorous
Mus indutus	Desert pygmy mouse	Omnivorous
Rhabdomys bechuanae	West-central South African four-striped grass rat	Omnivorous
Saccostomus campestris	Pouched mouse	Omnivorous
Dendromus melanotis	Gray climbing mouse	Omnivorous
Micaelamys namaquensis	Namaqua rock rat	Herbivorous
Thallomys nigricauda	Black-tailed tree rat	Herbivorous
Crocidura hirta	Lesser red musk shrew	Insectivorous
Elephantulus intufi	Bushveld elephant shrew	Insectivorous
Elephantulus rupestris	Western rock elephant shrew	Insectivorous
Suncus varilla	Lesser dwarf shrew	Insectivorous

Table 23: Outcome of ANOVA comparing seasonal captures of the functional groups of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

Small Mammal Dietry Groups	df	Sum Sq.	Mean Sq.	F	p
Herbivores	2.0	13.0	6.5	0.1	0.874
Insectivores	2.0	22.2	11.1	0.4	0.671
Omnivores	2.0	12167.0	6083.0	1.1	0.353

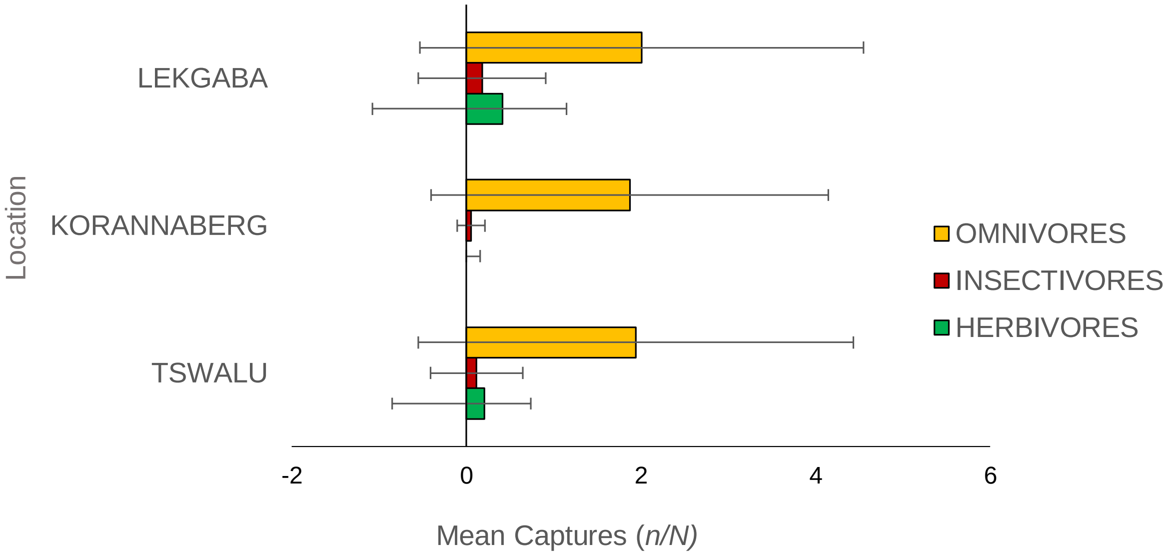

Figure 46: Comparison of the mean captures of small mammals, by functional group, between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Estimates are mean ± 1 standard deviation.

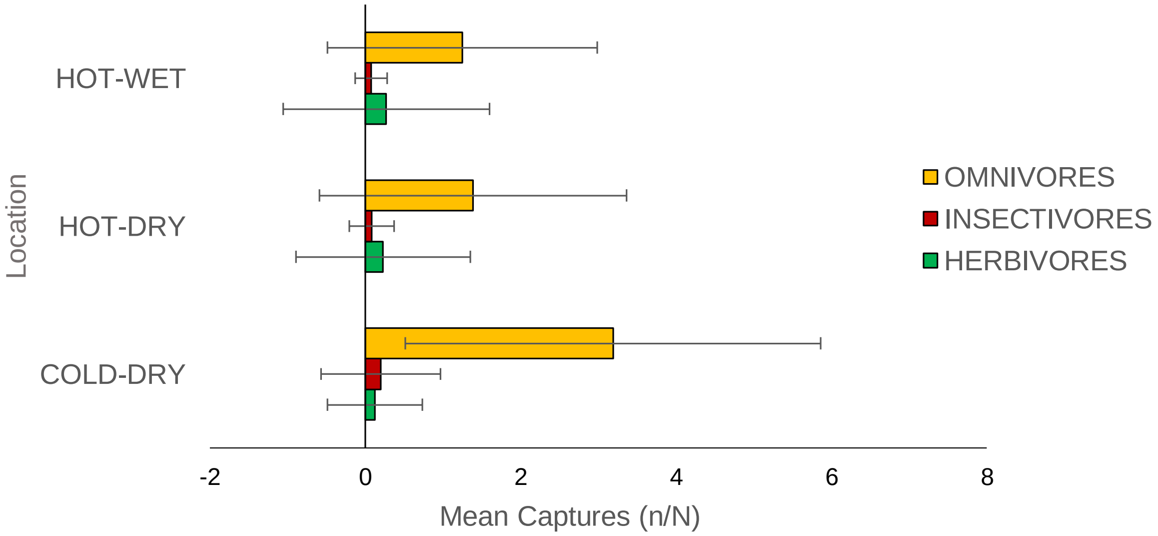

Figure 47: Seasonal captures of small mammals, by functional group, on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Estimates are mean ± 1 standard deviation.

When I compared variation in functional groups per transect (Appendix I – Table A31). I found that the number of captures per transect differed significantly between sections for all three functional groups (herbivores: $\chi^2 = 623.2$, df = 23, p < 0.001, insectivores: $\chi^2 = 194.7$, df = 23, p < 0.001, omnivores: $\chi^2 = 236.3$, df = 23, p < 0.001). My captures of herbivores (t = 1.1, df = 23, p = 0.296) and insectivores (t = 2.0, df = 23, p = 0.057 - marginally non-significant) did not differ significantly between transects, but my captures of omnivores did differ significantly (t = 11.1, df = 23, p < 0.001) between transects.

I recorded richness in herbivorous small mammals on Korannaberg as zero (S = 0.0), unlike Lekgaba (S = 2.0). I found the same pattern for herbivore species diversity, (Korannaberg: (H' = 0.0), Lekgaba: (H' = 0.07)). My comparison of seasonal herbivore captures showed that species richness was highest in the hot-wet season (S = 2.0). Whereas species richness was the same (S = 1.0) in both dry seasons (only one species was caught per season). I found the same pattern for species diversity, which was highest in the hot-wet season (S = 1.0) and zero in the two dry seasons.

My comparison of the insectivores between sections (Figure 48) showed that species richness (S = 1.0 vs. S = 4.0) and diversity (H' = 0.0 vs. H' = 0.19) was lower on Korannaberg than Lekgaba. I found that seasonal insectivore richness (Figure 49) was highest in the cold-dry and hot-dry seasons (S = 3.0), and lowest in the hot-wet season (S = 2.0). While I recorded the highest species diversity in the hot-dry season (H' = 0.18), followed by the hot-wet season (H' = 0.15), and the lowest in the cold-dry season (H' = 0.10).

I found that the omnivorous (Figure 50) species richness was higher on Korannaberg (S = 9.0) than Lekgaba (S = 8.0). However, I recorded higher species diversity on Lekgaba (H' = 0.17) than Korannaberg (H' = 0.14). Seasonally (Figure 51) I recorded the highest species richness in the hot-dry season (S = 9.0), followed by the cold-dry season (S = 8.0), with the lowest richness in the hot-wet season (S = 6.0). I recorded the highest species diversity in the hot-dry season (H' = 0.17), with the cold-dry and hot-wet season having equal species diversity values (H' = 0.15).

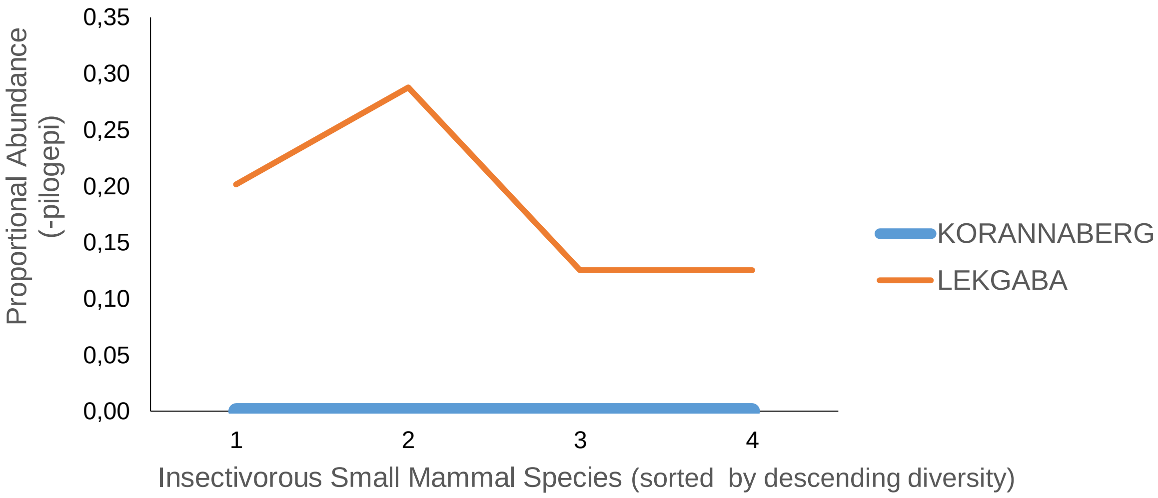

Figure 48: Relative diversity of insectivorous small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

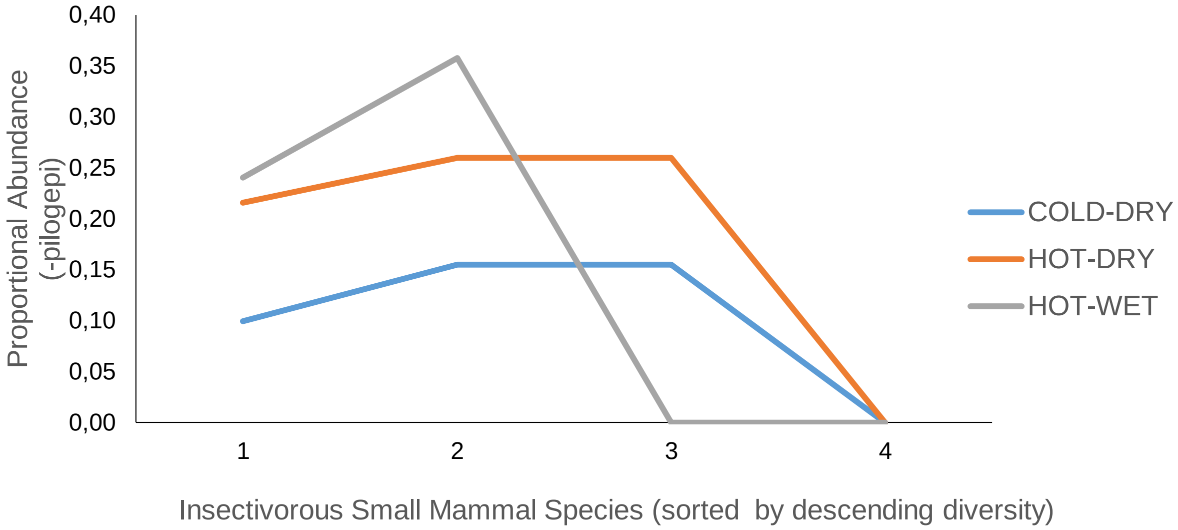

Figure 49: Seasonal relative diversity of insectivorous small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

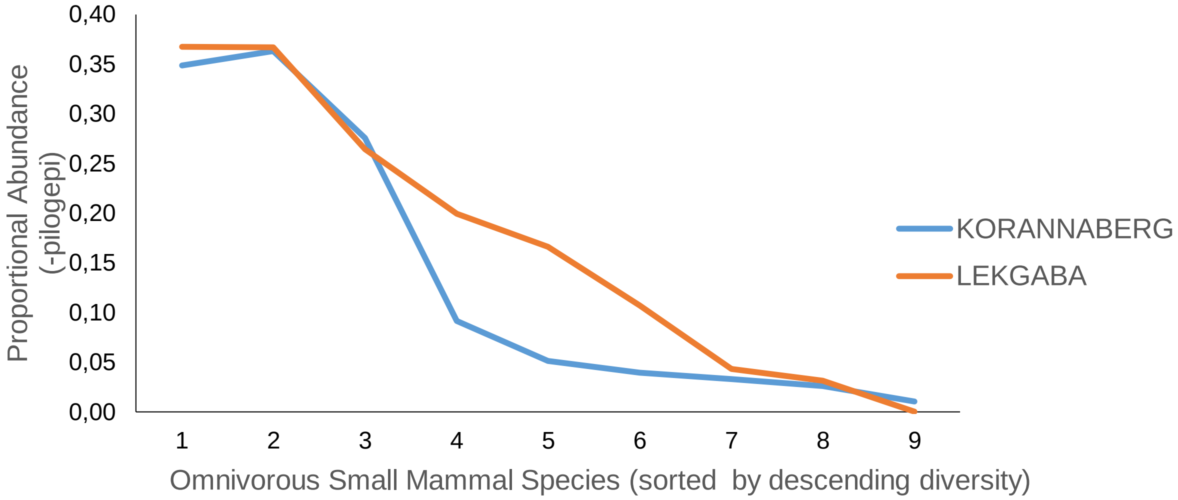

Figure 50: Relative diversity of omnivorous small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

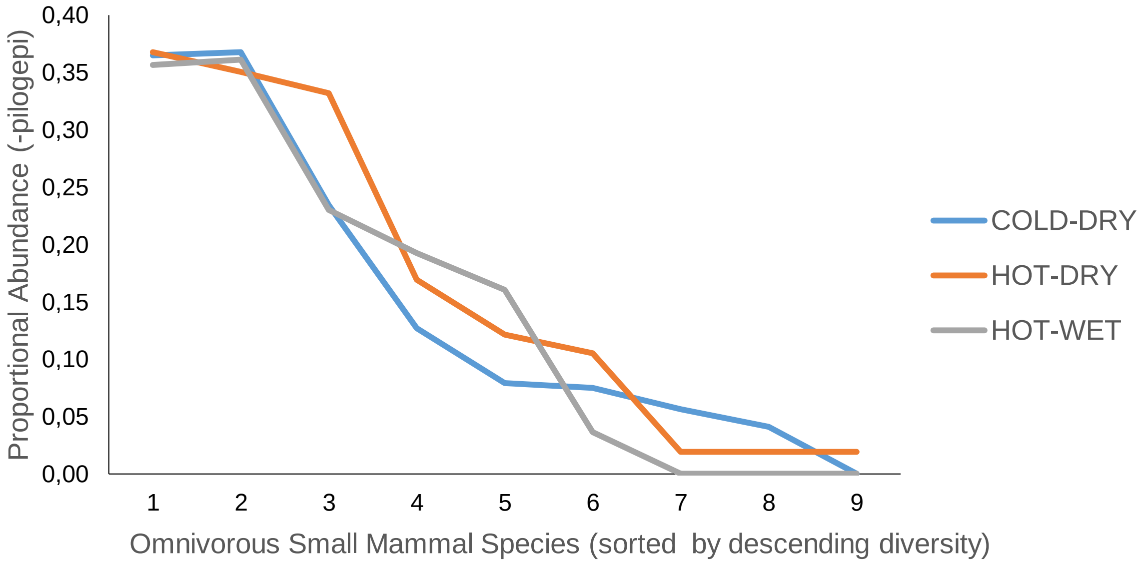

Figure 51: Seasonal relative diversity of omnivorous small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

My diversity comparison by transect (Figure 52) showed low species richness for both the herbivorous (S = 2.0) and the insectivorous (S = 8.0) small mammal groups, but a high species richness for the omnivorous species group (S = 24.0). I found a similar pattern for species diversity, with the lowest diversity for herbivorous species (H' = 0.01), followed by the insectivorous species (H' = 0.07) with the highest diversity for the omnivorous group (H' = 0.13)

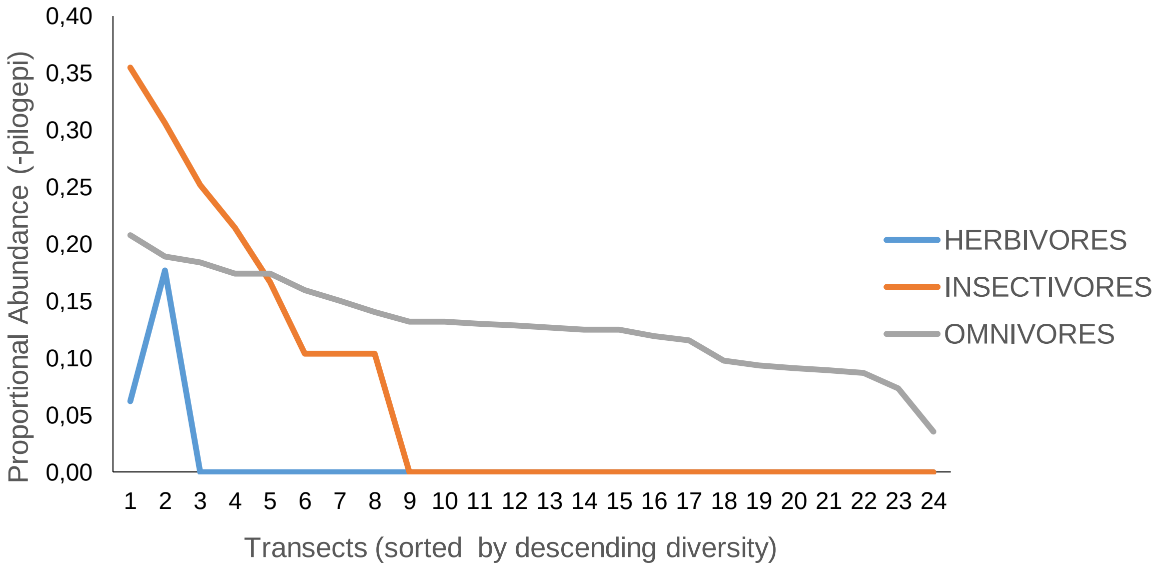

Figure 52: Relative diversity of different small mammal functional groups, per transect, captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

4.2.4 Body morphometric comparison

I compared morphometries (head length, body length, and total length) for the small mammals captured on TKR. The morphometric data all conformed to a normal distribution (p > 0.05) (Appendix I -Table A33). My comparison of head lengths (Appendix I – Table A32) showed that all except one (highveld gerbil) of the seven species had longer head lengths on Lekgaba than Korannaberg (Figure 53). My comparison of body lengths (Figure 54) showed that three (highveld gerbil, pygmy hairy-footed gerbil, pouched mouse) of the seven species had a larger body size on Korannaberg than Lekgaba (Appendix I – Table A32). I found that on Korannaberg four species (highveld gerbil, pygmy hairy-footed gerbil, west-central South African four-striped grass rat, and pouched mouse) had larger total lengths than those on Lekgaba (Figure 55).

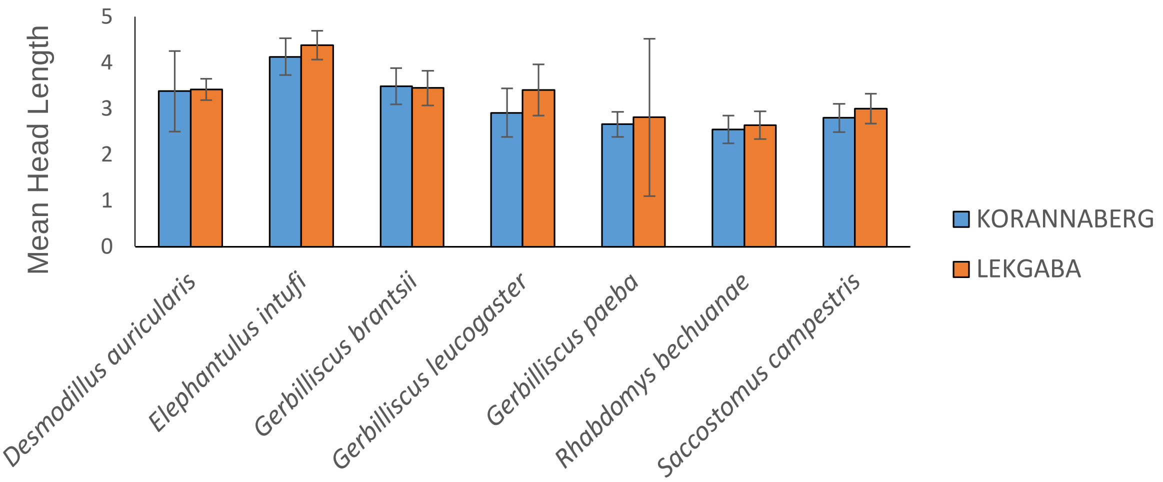

Figure 53: Mean head lengths of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Estimates are mean ± 1 standard deviation.

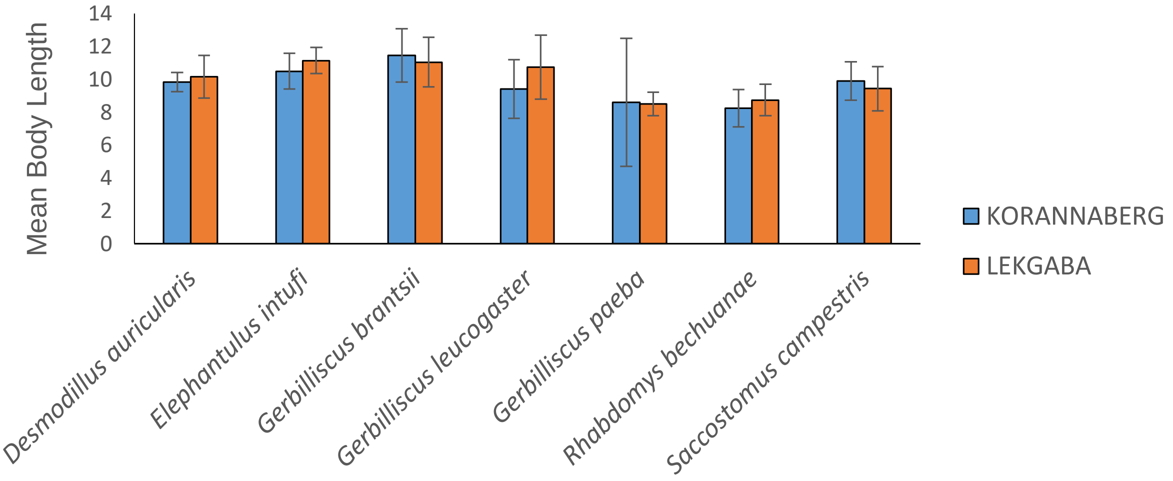

Figure 54: Mean body lengths of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Estimates are mean ± 1 standard deviation.

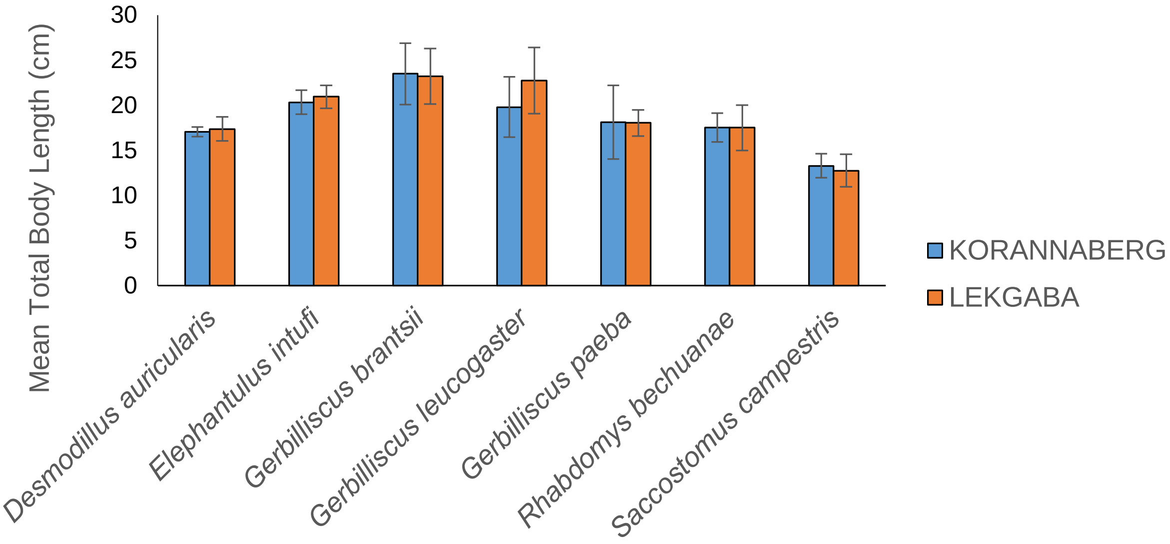

Figure 55: Mean total lengths of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

Note: Estimates are mean ± 1 standard deviation.

I compared differences in head (Table 24), body (Table 25) and total lengths (Table 26) by sites and found only a single species (bushveld gerbil) returned significantly larger measurements on Lekgaba than Korannaberg.

Table 24: Comparison of small mammal head lengths between Korannaberg and Lekgaba, using a t-test, on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Head Lengths (mm)				
Small Mammal Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Desmodillus auricularis	33.8	34.2	3	-0.1	0.931
Elephantulus intufi	41.3	43.8	11	-1.6	0.139
Gerbilliscus brantsii	34.9	34.5	166	0.8	0.453
Gerbilliscus leucogaster	29.1	34.1	397	-9.2	< 0.001*
Gerbillurus paeba	26.6	2.81	466	-1.6	0.117
Rhabdomys bechuanae	25.5	26.4	21	-0.8	0.425
Saccostomus campestris	28.0	30.0	8	-1.6	0.146

[1]Lions absent; [2]Lions present

*Statistically significant results evaluated at α = 0.05

Table 25: Comparison of small mammal body lengths between Korannaberg and Lekgaba, using a t-test, on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Body Lengths (mm)				
Small Mammal Species	Korannaberg[1] $\overline{X}$	Lekgaba[2] $\overline{X}$	df	t	p
Desmodillus auricularis	98.3	101.5	7	-0.5	0.610
Elephantulus intufi	104.9	111.4	10	-1.6	0.151
Gerbilliscus brantsii	114.5	110.4	165	1.6	0.117
Gerbilliscus leucogaster	94.0	107.3	400	-7.2	< 0.001*
Gerbillurus paeba	85.9	85.0	497	-2.0	0.050
Rhabdomys bechuanae	82.3	87.4	19	-1.3	0.195
Saccostomus campestris	99.0	94.3	8	1.0	0.354

[1]Lions absent; [2]Lions present

*Statistically significant results evaluated at α = 0.05

Table 26: Comparison of small mammal total lengths between Korannaberg and Lekgaba, using a t-test, on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Total Lengths (mm)				
Small Mammal Species	Korannaberg[1] $\overline{X}$	Lekgaba[2] $\overline{X}$	df	t	p
Desmodillus auricularis	170.8	173.8	7	-0.5	0.626
Elephantulus intufi	203.4	209.5	12	-1.1	0.290
Gerbilliscus brantsii	235.0	232.2	165	0.6	0.570
Gerbilliscus leucogaster	198.1	227.4	400	-8.5	< 0.001*
Gerbillurus paeba	181.2	180.6	384	0.5	0.598
Rhabdomys bechuanae	175.4	175.3	32	0.0	0.984
Saccostomus campestris	133.0	127.5	9	1.0	0.353

[1]Lions absent; [2]Lions present

*Statistically significant results evaluated at α = 0.05

My seasonal measurements of small mammal head, body and total lengths (Appendix I – Table A34) conformed to a normal distribution (p > 0.05) (Appendix I – Table A33). I captured all species (apart from the Cape short-eared gerbil) in all three seasons. I did not capture the Cape short-eared gerbil in the hot-wet season. I found significant seasonal differences in head lengths for bushveld elephant shrew, bushveld gerbil and pouched mouse (Table 27). I found significant seasonal differences in body lengths for Cape short-eared gerbil, and bushveld gerbil (Table 28). I found significant seasonal differences in total body lengths for Cape short-eared gerbil, bushveld gerbil and pygmt hairy-footed gerbil (Table 29).

Table 27: Seasonal comparison, using an ANOVA, of head lengths of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Head Lengths (mm)				
Small Mammal Species	df	Sum Sq	Mean Sq	F	p
Desmodillus auricularis	1	0.0	0.0	0.1	0.718
Elephantulus intufi	2	1.2	0.6	6.7	0.005*
Gerbilliscus brantsii	2	0.5	0.2	1.5	0.216
Gerbilliscus leucogaster	2	34.2	17.1	62.6	< 0.001*
Gerbillurus paeba	2	1.7	0.9	0.6	0.524
Rhabdomys bechuanae	2	0.5	0.2	2.8	0.073
Saccostomus campestris	2	0.9	0.4	4.7	0.013*

*Statistically significant results evaluated at α = 0.05

Table 28: Seasonal comparison, using an ANOVA, of body lengths of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Body Lengths (mm)				
Small Mammal Species	df	Sum Sq	Mean Sq	F	p
Desmodillus auricularis	1	6.3	6.3	14.1	0.006*
Elephantulus intufi	2	4.1	2.1	2.8	0.079
Gerbilliscus brantsii	2	7.0	3.5	1.4	0.245
Gerbilliscus leucogaster	2	74.4	37.2	9.9	< 0.001*
Gerbillurus paeba	2	15.0	7.7	0.9	0.420
Rhabdomys bechuanae	2	1.0	0.5	0.4	0.643
Saccostomus campestris	2	0.9	0.4	0.2	0.785

*Statistically significant results evaluated at α = 0.05

Table 29: Seasonal comparison, using an ANOVA, of total lengths of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 20018.

	Total Lengths (mm)				
Small Mammal Species	df	Sum Sq	Mean Sq	F	p
Desmodillus auricularis	1	8.3	8.3	37.9	< 0.001*
Elephantulus intufi	2	1.2	0.6	0.3	0.709
Gerbilliscus brantsii	2	30.3	15.2	1.5	0.237
Gerbilliscus leucogaster	2	680.0	339.9	26.7	< 0.001*
Gerbillurus paeba	2	80.0	39.8	3.9	0.022*
Rhabdomys bechuanae	2	8.3	4.2	0.8	0.448
Saccostomus campestris	2	0.9	0.4	0.1	0.876

*Statistically significant results evaluated at α = 0.05

Of the species for which I recorded significant seasonal morphometric variation, I compared between seasons. I found that head lengths for bushveld elephant shrew were significantly longer in the cold-dry than the hot-dry season, and longer in the hot-wet than the hot-dry season (Table 30). I found that head lengths for bushveld gerbil were significantly different between all three seasons and were longest in the hot-wet season, followed by the hot-dry season, with the shortest head lengths in the cold-dry season (Table 30). I found that head lengths for pouched mouse were significantly longer in the hot-wet than the hot-dry and the cold-dry season (Table 30).

Of the species with significant seasonal differences in body lengths, I found that the Cape short-eared gerbil body lengths were significantly longer in the hot-dry season than the cold-dry season (Table 31). For bushveld gerbil lengths, I found significantly longer body lengths in the hot-wet season than the cold-dry, as well as longer lengths in the hot-dry season compared with the cold-dry season (Table 31).

I found that the Cape short-eared gerbil and the pygmy hairy-footed gerbil had significantly longer total body lengths in the hot-dry season than the cold-dry season (Table 32). For bushveld gerbil I found that total body lengths were significantly longer in the hot-dry season than the cold-dry season, and significantly longer total body lengths in the hot-wet season than the cold-dry season (Table 32).

Table 30: Outcome of Tukey HSD comparison for seasonal differences in small mammal head lengths measured on Tswalu Kalahari Reserve from June 2017 to May 2018.

	Head Lengths				
Small Mammal Species	Seasons	Difference	Lower	Upper	p
Elephantulus intufi	Cold-dry vs Hot-dry	-0.36	-0.69	-0.03	0.032*
	Cold-dry vs Hot-wet	0.22	-0.13	0.57	0.283
	Hot-dry vs Hot-wet	0.58	0.18	0.99	0.004*
Gerbiliscus leucogaster	Cold-dry vs Hot-dry	0.35	0.18	0.52	< 0.001*
	Cold-dry vs Hot-wet	0.74	0.58	0.89	< 0.001*
	Hot-dry vs Hot-wet	0.39	0.18	0.60	< 0.001*
Saccostomus campestris	Cold-dry vs Hot-dry	0.01	-0.26	0.29	0.995
	Cold-dry vs Hot-wet	0.26	0.00	0.51	0.049*
	Hot-dry vs Hot-wet	0.25	0.02	0.47	0.029*

*Statistically significant results evaluated at α = 0.05

Table 31: Outcome of Tukey HSD comparison for seasonal differences in small mammal body lengths measured on Tswalu Kalahari Reserve from June 2017 to May 2018.

	Body Lengths				
Small Mammal Species	Seasons	Difference	Lower	Upper	p
Desmodillus auricularis	Cold-dry vs Hot-dry	2.64	1.02	4.27	0.006*
Gerbiliscus leucogaster	Cold-dry vs Hot-dry	0.70	0.07	1.33	0.025*
	Cold-dry vs Hot-wet	1.02	0.43	1.61	< 0.001*
	Hot-dry vs Hot-wet	0.32	-0.45	1.08	0.594

*Statistically significant results evaluated at α = 0.05

Table 32: Outcome of Tukey HSD comparison for seasonal differences in small mammal total body lengths measured on Tswalu Kalahari Reserve from June 2017 to May 2018.

		Total Body Length			
Small Mammal Species	Seasons	Difference	Lower	Upper	p
Desmodillus auricularis	Cold-dry vs Hot-dry	3.04	1.90	4.19	< 0.001*
Gerbiliscus leucogaster	Cold-dry vs Hot-dry	2.18	1.01	3.34	< 0.001*
	Cold-dry vs Hot-wet	3.06	1.97	4.14	< 0.001*
	Hot-dry vs Hot-wet	0.89	-0.53	2.29	0.308
Gerbillurus paeba	Cold-dry vs Hot-dry	0.93	0.10	1.77	0.024*
	Cold-dry vs Hot-wet	0.58	-0.24	1.39	0.221
	Hot-dry vs Hot-wet	-0.36	-1.33	0.62	0.667

*Statistically significant results evaluated at α = 0.05

4.3 Avian populations

4.3.1 Korannaberg vs. Lekgaba comparison

During this study I recorded 2010 separate records of avian species. I recorded fewer detections on Korannaberg (n = 939, $\bar{X}$ = 13.0, SD = 10.2) than Lekgaba (n = 1046, $\bar{X}$ = 14.5, SD = 10.8) (Appendix I -Table A36). These sightings comprised 64 species (Korannaberg = 55; Lekgaba = 52). The three species that I recorded most frequently were, sociable weaver (*Philetairus socius* Latham, 1790), (n = 417, $\bar{X}$ = 26.1, SD = 24.8), scaly-feathered weaver (*Sporopipes squamifrons* A. Smith, 1836), (n = 304, $\bar{X}$ = 19.0, SD = 17.4) and Namaqua sandgrouse (*Pterocles namaqua* Gmelin, 1789), (n = 131, $\bar{X}$ = 8.2; SD = 15.6). Data for observations of the five most common species (sociable weaver, black-chested prinia (*Prinia flavicans* Vieillot, 1821), Namaqua sandgrouse, scaly-feathered weaver and ring-necked dove (*Streptopelia capicola* Sundevall, 1857) were normally distributed (p > 0.05) (Appendix I – Table A37).

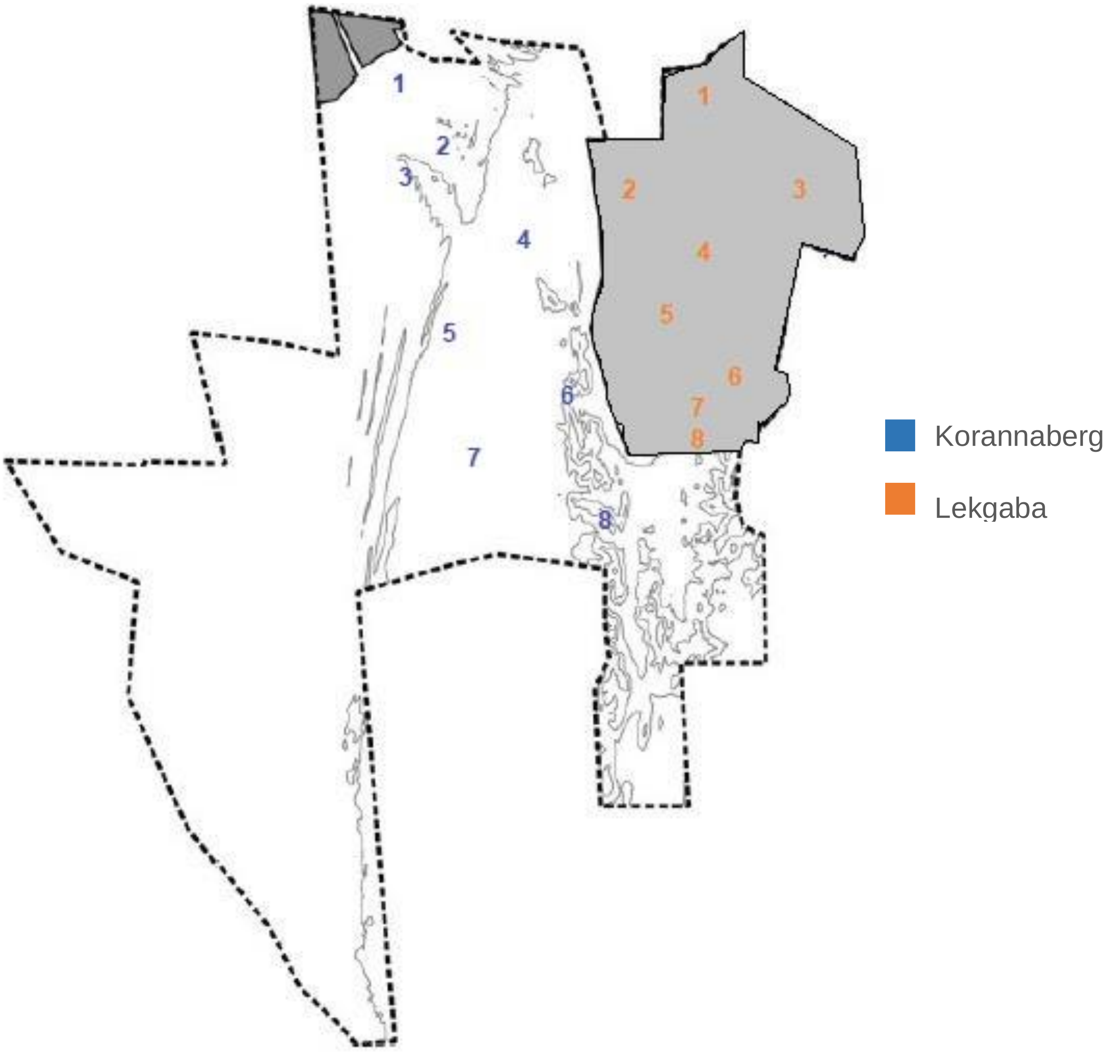

Figure 56: Locations of avian transects sampled on Tswalu Kalahari Reserve between June 2017 and May 2018 (GPS co-ordinates in Appendix I – Table A35).

I found no significant differences in the number of detections of the five most frequently observed species between Korannaberg and Lekgaba (Table 33). Of the 64 species recorded, 78% were classified as songbirds, 14% as ground birds, 6% as raptors and only one species of waterfowl (Egyptian goose, *Alopochen aegyptiaca* Linnaeus, 1766) was recorded on Lekgaba, which I included in the ground bird group (Table 34) (Figure 57). Data for each of these groups were normally distributed (p > 0.5) (Appendix I – Table A38).

Table 33: Comparison of detections of five most observed avian species between sections on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	T	p
Philetairus socius	27.1	25.0	14	0.1	0.867
Prinia flavicans	6.4	6.4	10	0.0	1.000
Pterocles namaqua	9.5	6.9	13	0.3	0.744
Sporopipes squamifrons	12.9	25.1	11	-1.4	0.201
Streptopelia capicola	6.4	6.4	10	0.0	1.000

[1]Lions Absent; [2]Lions Present

Table 34: Avian groups observed between the different sections on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Ground Birds			Raptors			Songbirds		
Location	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Tswalu	212	4.4	9.3	9	0.2	0.5	1764	36.8	22.3
Korannaberg	131	5.5	11.5	7	0.3	0.6	801	33.4	18.3
Lekgaba	81	3.4	6.5	2	0.1	0.3	963	40.1	25.7

[1]Lions Absent; [2]Lions Present

Detections for each bird group (songbirds, ground birds, and raptors) were also normally distributed (Appendix I -Table A38). However, there was no significant difference between detections of birds from the three groups between Korannaberg and Lekgaba (p > 0.05) (Table 35).

Avian sightings, per transect, across Tswalu had a normal distribution (W = 1.0; p = 0.5).I recorded a similar number of sightings, per transect, on both Korannaberg and Lekgaba (Appendix I - Table A39). The transects associated with the highest number of sightings on Korannaberg and Lekgaba have similar numbers of sightings (Korannaberg = 181; Lekgaba = 192). I found the total number of avian sightings per transect were significantly different from each other ($\chi^2 = 260.5$; df = 63; $p < 0.001$).

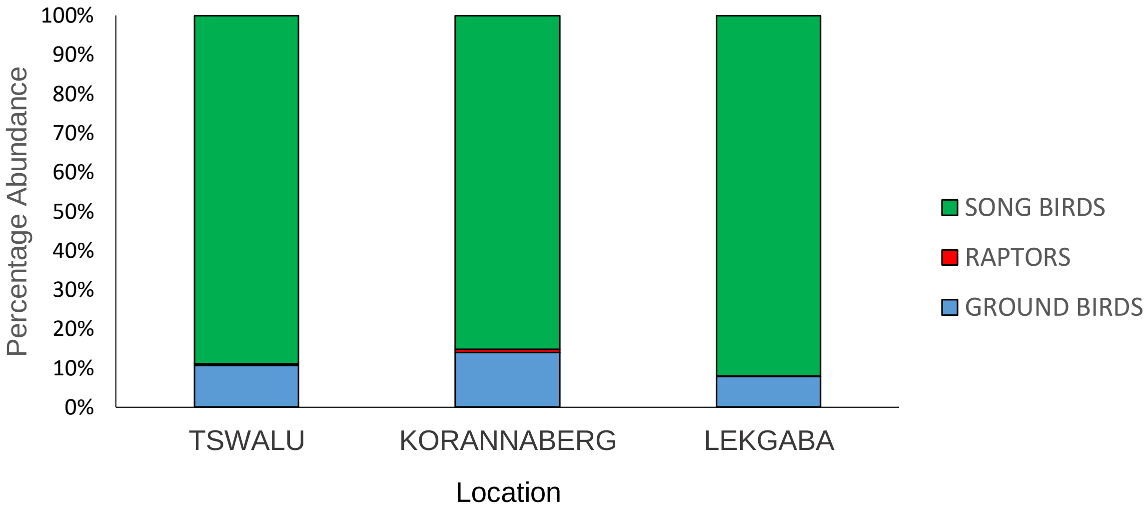

Figure 57: Different avian groups and the frequency of their detections at different sections on Tswalu Kalahari Reserve between June 2017 and May 2018

I found no significant difference between the overall number of bird detections per kilometre between Korannaberg and Lekgaba (t = -0.5, df = 43, p = 0.638). My group wise comparison (Table 36) showed that there was no significant difference for detections per kilometre between sections ($p > 0.05$).

Table 35: Outcome of a t-test comparison of three different avian groups (detection per transect), between Korannaberg and Lekgaba on Tswalu Kalahari Reserve from June 2017 to May 2018.

Avian Groups	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Ground birds	13.6	8.4	36	0.8	0.445
Raptors	0.7	0.2	32	1.5	0.146
Songbirds	83.4	100.3	41	-1.0	0.300

[1]Lions Absent; [2]Lions Present

Table 36: Outcome of a t-test comparison of index (detections per km) detections of avian groups between Korannaberg and Lekgaba on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Groups	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Ground birds	5.5	3.3	36	0.8	0.419
Raptors	0.3	0.1	32	1.5	0.146
Songbirds	33.4	40.1	41	-1.0	0.300

[1]Lions Absent; [2]Lions Present

Furthermore, my species-wise index comparison of detections per km (Table 37) showed no significant difference ($p > 0.05$) in the relative abundance of any of the five most frequently observed species recorded on TKR, between Korannaberg and Lekgaba.

I compared index of relative abundance per transect and found a significant difference in detections among transects ($p < 0.001$).

Table 37: Outcome of a t-test comparison of index (detections per km) of relative abundance of avian species between sections on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species	Korannaberg[1] $\bar{X}$	Lekgaba[2] $\bar{X}$	df	t	p
Philetairus socius	22.6	20.8	45	0.1	0.882
Prinia flavicans	4.7	4.5	46	0.2	0.845
Pterocles namaqua	7.9	5.7	35	0.4	0.726
Sporopipes squamifrons	9.7	20.9	33	-1.8	0.077
Streptopelia capicola	5.3	5.3	40	1.0	1.000

[1] Lions Absent; [2] Lions Present

My diversity comparison of the bird species showed that Korannaberg (S = 55.0) had a slightly higher species richness than Lekgaba (S = 52.0), but the species diversity was equal for both sites (H’ = 3.0) (Figure 58). I compared diversity by groups; for ground birds (Figure 59) I found that species richness was slightly higher for Korannaberg (S = 8.0) than Lekgaba (S = 7.0). Yet, I found species diversity to be equal for both sides (H’ = 0.12).

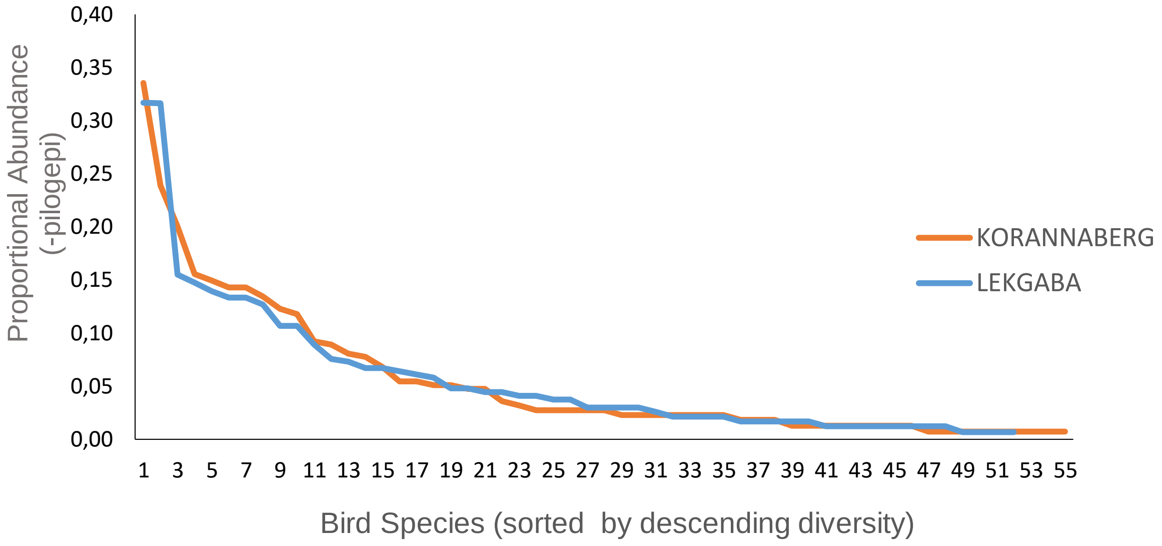

Figure 58: Relative diversity of all avian species observations on Tswalu Kalahari Reserve between June 2017 and May 2018.

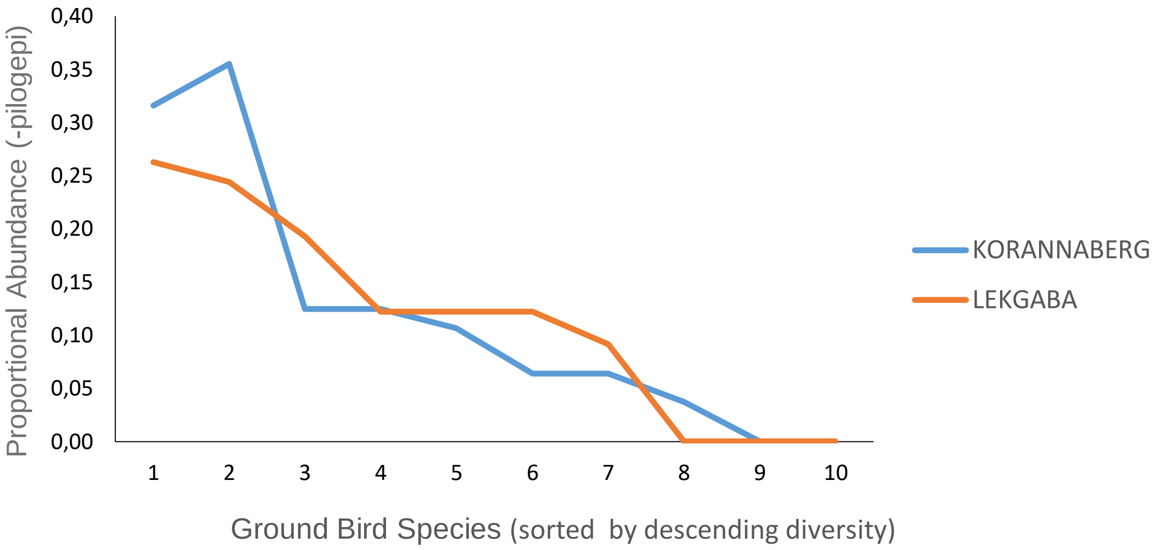

Figure 59: Relative diversity of ground bird observations from Tswalu Kalahari Reserve between June 2017 and May 2018.

Due to only having made four raptor sightings during my study, one on Lekgaba and three on Korannaberg, my comparison of raptors on TKR is skewed. The diversity on Lekgaba was one (H’ = 1.0), compared with Korannaberg (H’ = 3.0) and species richness was zero on Lekgaba (S = 0.0) whereas Korannaberg had a small measurable richness (S = 0.3) (Figure 60).

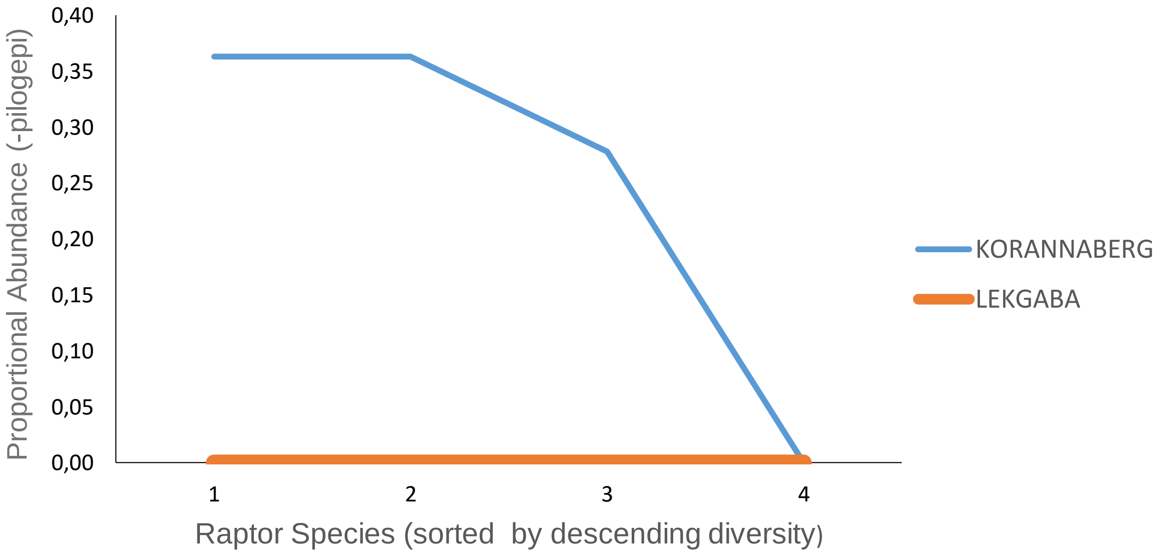

Figure 60: Relative diversity of raptors observed on Tswalu Kalahari Reserve between June 2017 and May 2018.

I found songbirds (Figure 61) species richness was similar on Korannaberg (S =45.0) and Lekgaba (S = 44.0) and that species diversity was equal between sites (H' = 0.06).

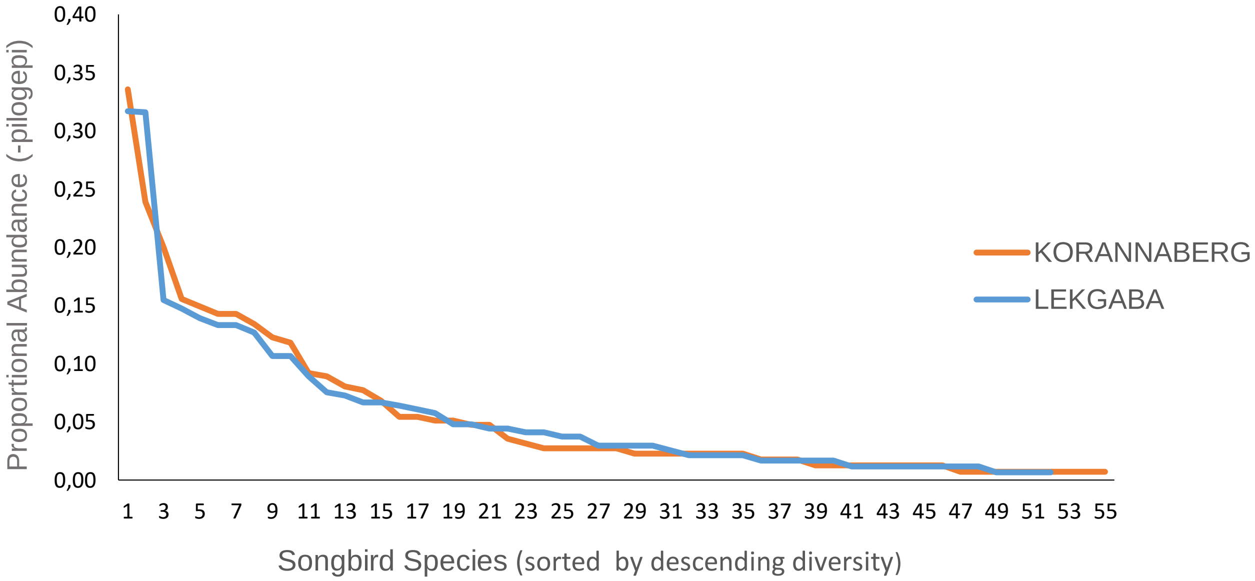

Figure 61: Relative diversity of songbirds observed on Tswalu Kalahari Reserve between June 2017 and May 2018.

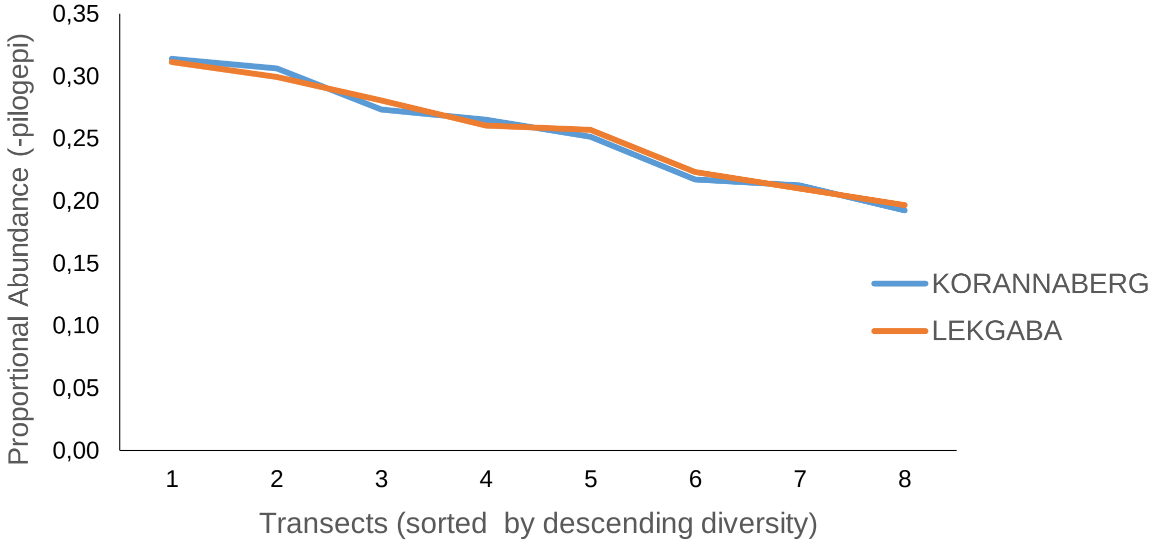

Figure 62: Relative diversity of all bird species observed per transect on Tswalu Kalahari Reserve between June 2017 and May 2018.

My comparison of avian species diversity, per transect (Figure 62), showed that richness (S = 8.0) and diversity (H' = 0.3) was equal on both Korannaberg and Lekgaba.

4.3.2 Seasonal comparison

I recorded the highest number of avian sightings in the cold-dry season (n = 858, $\overline{X}$ = 53.6, SD = 31.9) followed by the hot-dry season (n = 639, $\overline{X}$ = 39.9, SD = 23.3) and the lowest number of detections in the hot-wet season (n = 513, $\overline{X}$ = 32.1, SD = 10.5) (Figure 63) (Table 38). My avian detections differed significantly among seasons (χ^2 = 322.4, df = 30, p < 0.001). I recorded significantly more detections in the cold-dry than the hot-wet and hot-dry seasons (p = 0.039) (Table 39).

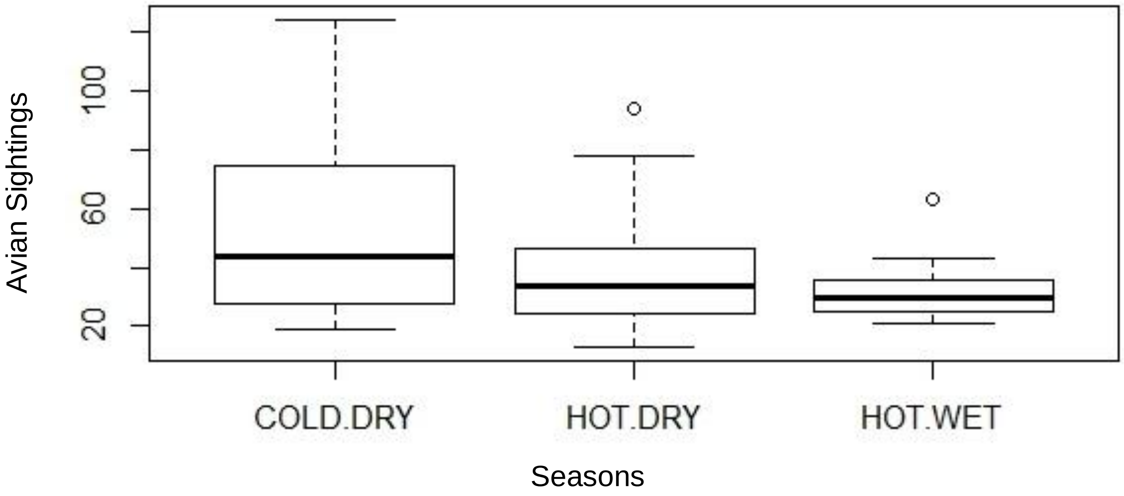

Figure 63: Seasonal variation in avian sightings on Tswalu Kalahari Reserve from June 2017 to May 2018.

Table 38: Seasonal variation in avian detections on Tswalu Kalahari Reserve between June 2017 and May 2018.

Seasons	n	$\overline{X}$	SD
Cold-dry	858	53.6	31.9
Hot-dry	639	39.9	23.3
Hot-wet	513	32.1	10.5

Table 39: Tukey HSD difference comparison between seasonal bird observations on Tswalu Kalahari Reserve between June 2017 and May 2018.

Seasonal Comparison	Difference	Lower	Upper	p
Cold-Dry vs Hot-Dry	-13.7	-33.9	6.5	0.239
Cold-Dry vs Hot-Wet	-21.6	-41.8	-1.4	0.034*
Hot-Dry vs Hot-Wet	-7.9	-28.1	12.3	0.615

*Statistically significant results evaluated at α = 0.05

On my comparison of seasonal observations per transect I recorded more observations on Lekgaba (n = 524, $\bar{X}$= 65.5, SD = 36.5) than Korannaberg (n = 334, $\bar{X}$ = 41.75, SD = 22.8) in the cold-dry season (Figure 64). In the hot-dry season I recorded similar numbers of detections per transect on Korannaberg (n = 334.0, $\bar{X}$= 41.75, SD = 29.1) and Lekgaba (n = 305, $\bar{X}$= 38.1, SD = 17.7) (Figure 65). In the hot-wet season I recorded a higher number of bird observations on Korannaberg (n= 296, $\bar{X}$= 37.0, SD = 12.8) than on Lekgaba (n = 217, $\bar{X}$= 27.1, SD = 3.9) (Figure 66).

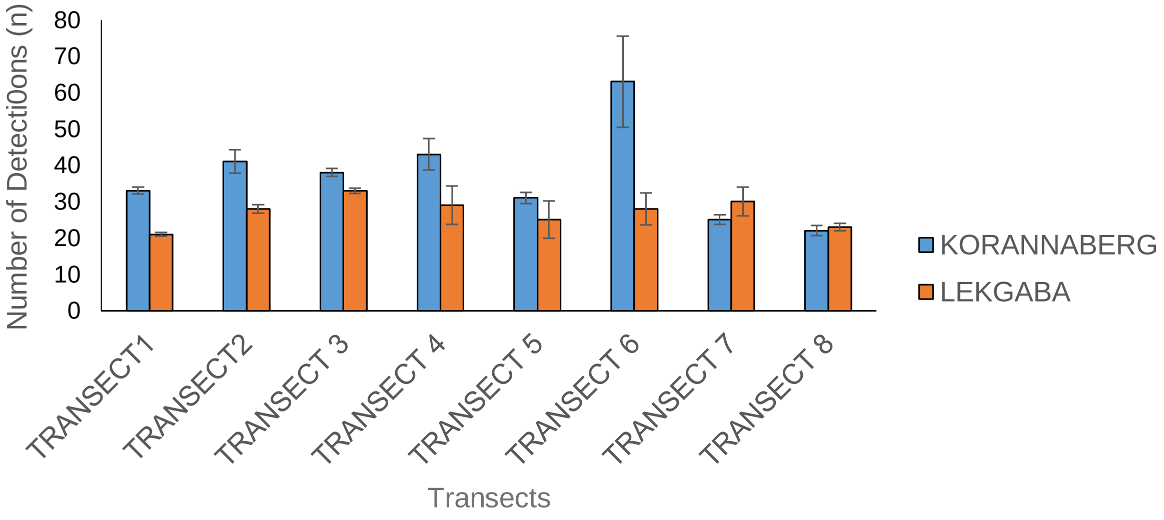

Figure 64: Seasonal avian observations, per transect, in the cold-dry season on Tswalu Kalahari Reserve from June 2017 to May 2018.

Note: Estimates are mean ± 1 standard deviation.

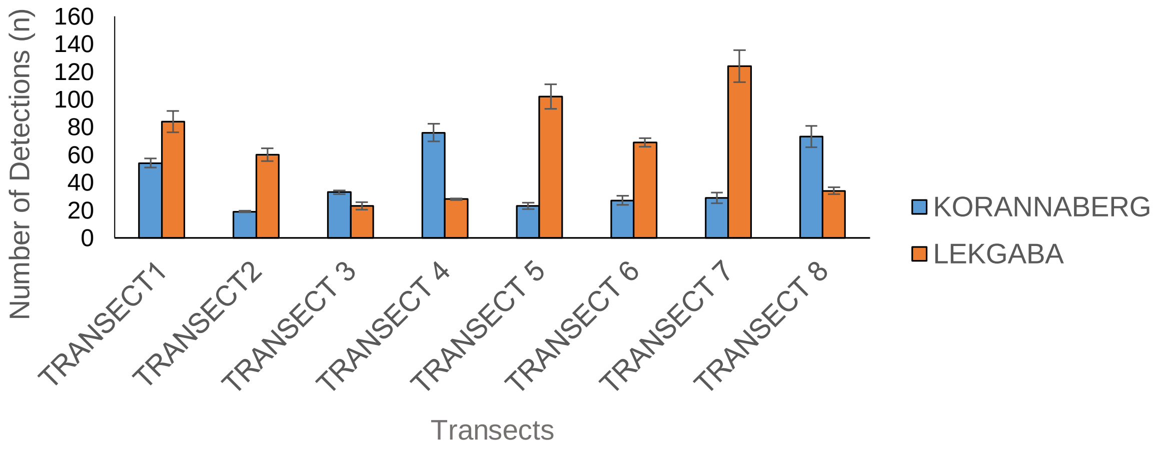

Figure 65: Seasonal avian observations, per transect, in the hot-dry season on Tswalu Kalahari Reserve from June 2017 to May 2018.

Note: Estimates are mean ± 1 standard deviation.

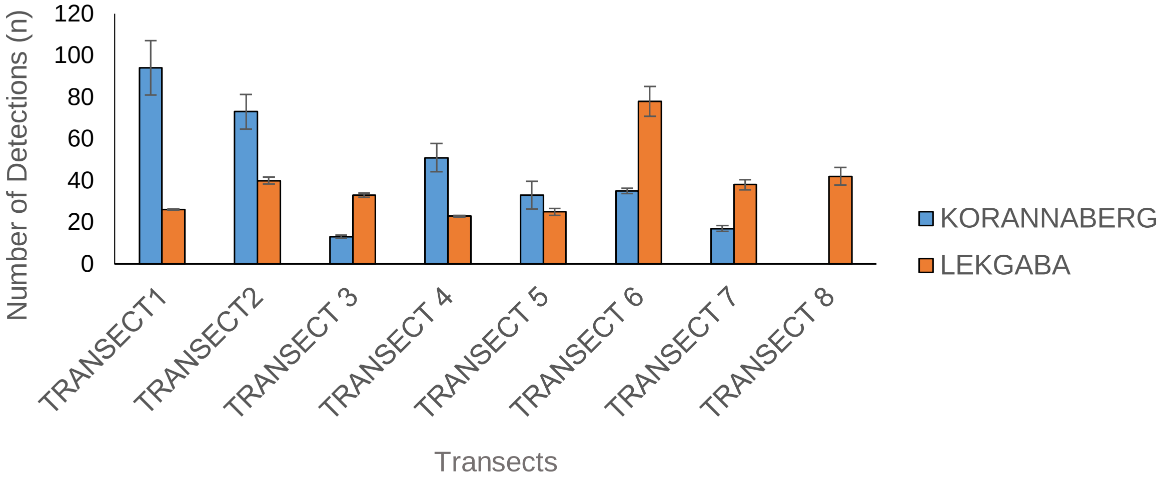

Figure 66: Seasonal avian observations per transect in the hot-wet season on Tswalu Kalahari Reserve from June 2017 to May 2018.

Note: Estimates are mean ± 1 standard deviation

My comparison of seasonal differences between bird detections for TKR had a normal distribution (W = 1.0, p = 0.208). I found that seasonal comparisons for bird groups (ground birds. raptors and songbirds) detections also all conformed to a normal distribution (Table 40)

Table 40: Shapiro-Wilk normality test for avian groups sampled on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Groups	W	p*
Ground birds	1.0	0.419
Raptors	0.9	0.435
Songbirds	1.0	0.789

*All values above 0.05 are normally distributed.

I compared seasonal differences in sightings within groups (ground birds, raptors, and songbirds) and found no significant difference (Table 41). For the five most frequently observed species (Appendix I – Table A40) I recorded the highest number of sightings in the cold-dry season, followed by the hot-dry season while the hot-wet season returned the lowest observations. Of my top five most observed bird species, I only found one species was associated wirth a significant difference in seasonal detections, black-chested prinia,(p = 0.036) (Table 42).

Table 41: Seasonal comparison, using an ANOVA, of detections of avian groups from Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Groups	df	Sum Sq	Mean Sq	F	p
Ground birds	2	204.0	101.9	1.2	0.314
Raptors	2	0.4	0.2	0.8	0.466
Song birds	2	2150.0	1074.8	2.3	0.114

Table 42: Seasonal comparison, using an ANOVA, of detections of the five most observed bird species from Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species	df	Sum Sq	Mean Sq	F	p
Philetairus socius	2	660.0	329.8	1.3	0.290
Prinia flavicans	2	13.5	6.8	3.6	0.036*
Pterocles namaqua	2	355.0	177.5	2.6	0.084
Sporopipes squamifrons	2	254.0	126.8	1.7	0.194
Streptopelia capicola	2	45.1	22.6	1.4	0.260

*Statistically significant results evaluated at α = 0.05

I found that my detections of black-chested prinia differed significantly between the cold-dry and the hot-wet seasons (Table 43).

Table 43: Outcome of Tukey HSD test comparing seasonal detections of *Prinia flavicans* recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species	Seasons	Difference	Lower	Upper	p
Prinia flavicans	Cold-dry vs Hot-dry	-0.94	-2.12	0.24	0.143
	Cold-dry vs Hot-wet	-1.25	-2.43	-0.07	0.035*
	Hot-dry vs Hot-wet	-0.31	-1.49	0.87	0.797

*Statistically significant results evaluated at α = 0.05

My comparison between seasonal indices of detections per kilometre (Table 44) showed that I recorded significantly more avian sightings in the cold-dry than the hot-wet seasons (p = 0.034).

Table 44: Outcome of Tukey HSD difference comparison between seasonal bird index detections on Tswalu Kalahari Reserve between June 2017 and May 2018.

Seasonal Comparison	Difference	Lower	Upper	p
Cold-Dry vs Hot-Dry	-34.2	-84.7	16.3	0.239
Cold-Dry vs Hot-Wet	-53.9	-104.4	-3.4	0.034*
Hot-Dry vs Hot-Wet	-19.7	-70.2	30.8	0.615

*Statistically significant results evaluated at α = 0.05

I compared the index of detections by groups (Table 45) and there were no significant differences between Korannaberg and Lekgaba for any of the groups ($p > 0.05$).

Table 45: Seasonal comparison, using an ANOVA test, of avian detections per km between groups detected on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Groups	Sum Sq.	Mean Sq.	df	F	p
Ground birds	1274.0	637.1	2	1.2	0.314
Raptors	2.4	1.2	2	0.8	0.466
Songbirds	13435.0	6718.0	2	2.3	0.114

I compared the seasonal index of detections for the five most observed species and found that only one species (black-chested prinia) showed significant seasonal variation in index values (Table 46).

Table 46: Seasonal comparison, using an ANOVA, of index (detections per km) of five most common bird species detected on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species	Sum Sq.	Mean Sq.	df	F	p
Philetairus socius	4123.0	2061.0	2	1.3	0.294
Prinia flavicans	84.6	42.3	2	3.6	0.036*
Pterocles namaqua	2219.0	1109.5	2	2.6	0.084
Sporopipes squamifrons	1584.0	792.2	2	1.7	0.194
Streptopelia capicola	282.0	141.0	2	1.4	0.260

*Statistically significant results evaluated at α = 0.05

My seasonal comparison of the index of detections showed that black-chested prinia was associated with significantly more observations in the cold-dry than the hot-wet season (Table 47).

Table 47: Outcome of Tukey HSD test comparing the seasonal differences in the index (detections per km) of *Prinia falvicans* observed on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species	Seasons	Difference	Lower	Upper	p
Prinia flavicans	Cold-dry vs Hot-dry	-2.34	-5.29	0.60	0.143
	Cold-dry vs Hot-wet	-3.13	-6.07	-0.18	0.035*
	Hot-dry vs Hot-wet	-0.78	-3.73	2.17	0.797

*Statistically significant results evaluated at α = 0.05

I compared seasonal avian sightings between sections and recorded no significant difference between Korannaberg and Lekgaba in the cold-dry season (t = -1.6, df = 12, p = 0.145), or in the hot-dry season (t = 0.3, df = 11, p = 0.769), or in the hot-wet season (t = 2.1, df = 8, p = 0.069).

Seasonally I found species richness was highest in the hot-wet season (S = 54.0), followed by the hot-dry season (S = 48.0), with lowest richness in the cold-dry season (S = 38.0). However, I found species diversity was equal across all three seasons (S = 0.1) (Figure 67).

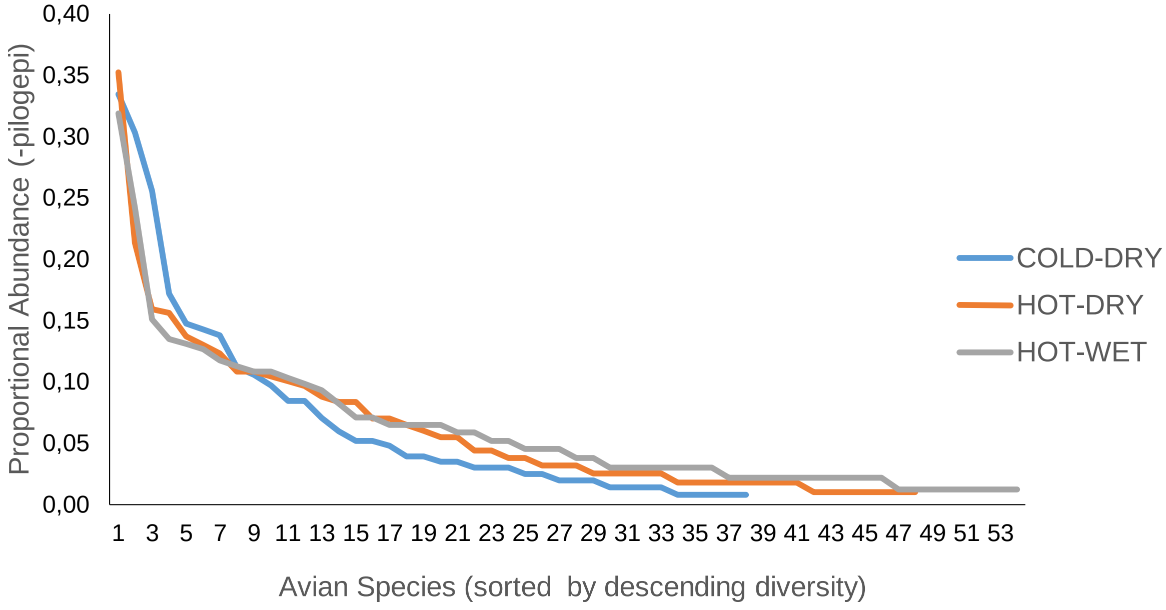

Figure 67: Seasonal relative diversity of bird species recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

I found that species richness for ground birds (Figure 68) was highest in the hot-wet season (S = 8.0), followed by the cold-dry and hot-dry season (S = 6.0). I found species diversity was highest in the hot-wet season (H’ = 0.17), followed by the hot-dry season (H’ = 0.14), with the lowest ground bird diversity in the cold-dry season (H’ = 0.06). With the raptor group I found that species richness and diversity of raptors in the cold-dry season was zero (Figure 69). Yet, species richness was highest in the hot-wet season (S = 4.0), followed by the hot-dry season (S = 2.0) with diversity also highest in the hot-wet season (H’ = 0.33), followed by the hot-dry season (H’ = 0.17).

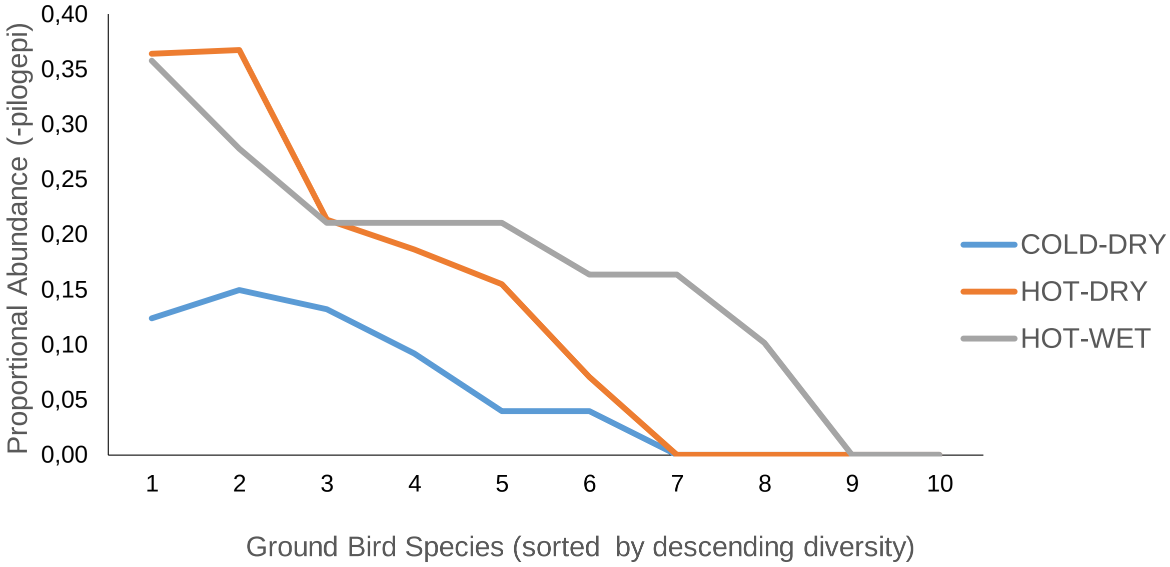

Figure 68: Seasonal relative diversity of ground bird detections from Tswalu Kalahari Reserve from June 2017 to May 2018.

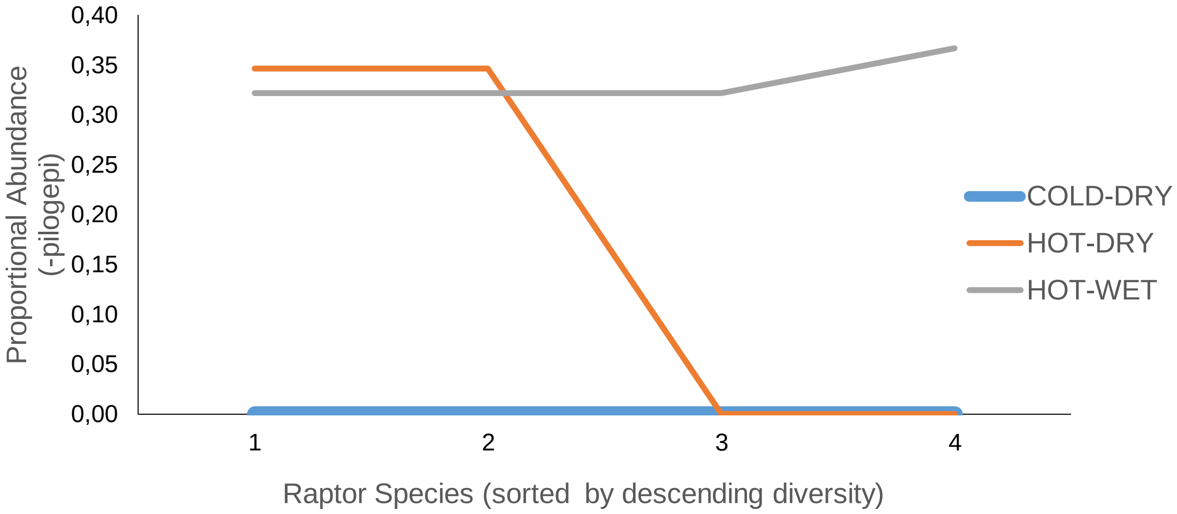

Figure 69: Seasonal relative diversity of raptor detections from Tswalu Kalahari Reserve from June 2017 to May 2018.

I found that songbird species richness was highest in the hot-wet season (S = 43.0), followed by the hot-dry season (S = 40.0) and lowest in the cold-dry season (S = 32.0). However, I found songbird diversity was equal in the hot seasons (H’ = 0.06) with a lower diversity in the cold-dry season (H’ = 0.05) (Figure 70). I compared seasonal observations of birds by transect (Figure 71) and found that species richness was similar (S = 16.0) across all three seasons, while species diversity was highest in the hot-wet season (H’ = 0.17), with the cold-dry and the hot-dry seasons reflecting the same species richness (H’ = 0.16).

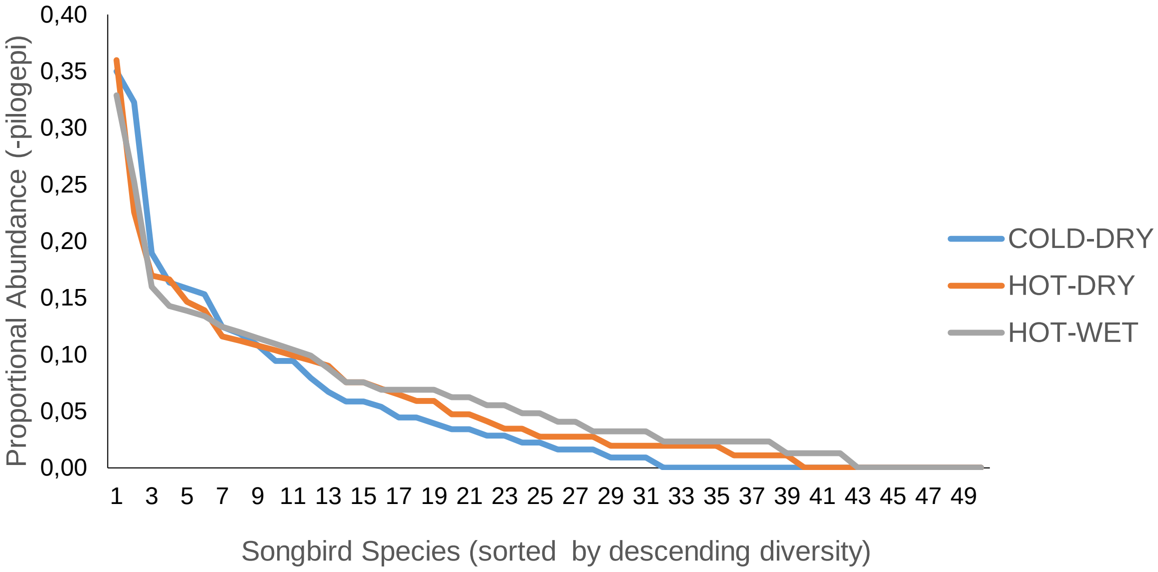

Figure 70: Seasonal relative diversity of songbird detections from Tswalu Kalahari Reserve from June 2017 to May 2018.

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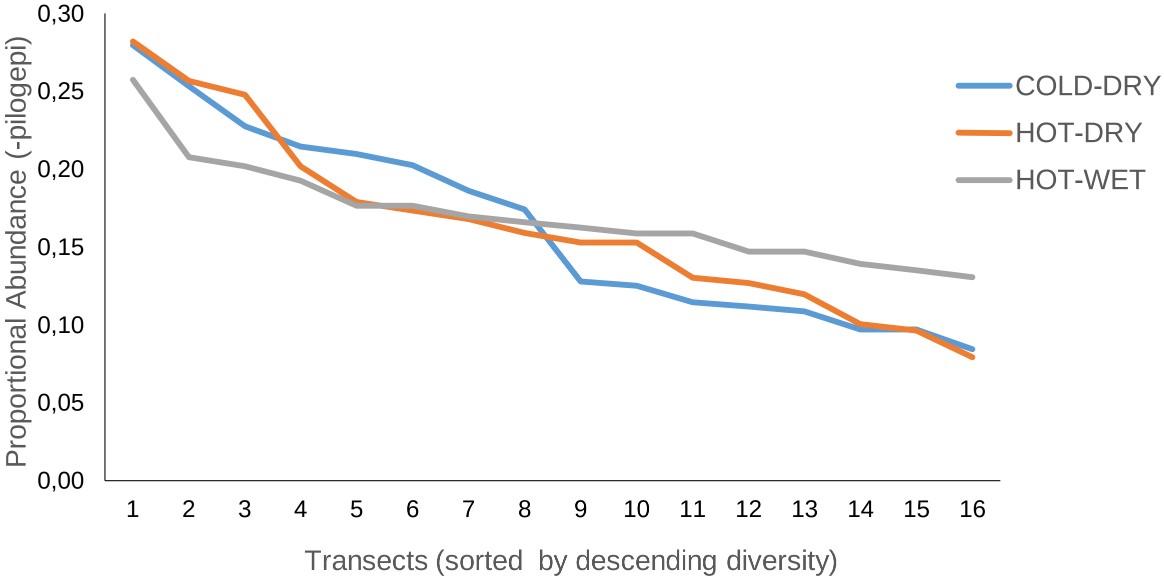

Figure 71: Seasonal relative diversity of avian species, per transect, observed on Tswalu Kalahari Reserve from June 2017 to May 2018.

4.3.3 Morning and evening comparison

I compared my species detections relative to the time of day (morning or evening). I recorded a total of 949 sightings across 60 species during my morning observations, and 1061 sightings across 50 species during my evening observations. Both my morning and evening observation data conformed to a normal distribution ($p > 0.05$). Although I recorded a higher number of observations in the evening than the morning, I found no significant difference between the total detections recorded in the morning vs. the evening ($t = -0.3$, $df = 112$, $p = 0.775$). On a group wise basis, I found that I recorded a higher diversity of species in the mornings while I recorded a higher abundance of individuals in the evenings (Table 48). I found observations of ground birds were more prevalent in the evening ($n = 116$, $\bar{X} = 11.6$, $SD = 28.5$), than the morning ($n = 96$, $\bar{X} = 9.6$, $SD = 14.8$). Yet, I found no significant difference ($t = -0.2$, $df = 13$, $p = 0.847$) between morning and evening detections for ground birds. I found the raptors were also more prevalent during my evening observations ($n = 6$, $\bar{X} = 1.5$, $SD = 0.6$), than my morning ones ($n = 3$, $\bar{X} = 0.8$, $SD = 1.0$). Yet, I found no significant difference ($t = -1.3$, $df = 5$, $p = 0.238$) in the raptor detections between morning and evening. Unlike the other two groups, I found that morning and evening detections of songbirds differed significantly ($\chi^2 = 719.5$, $df = 576$, $p < 0.001$), and I detected significantly more songbirds in the evening ($n = 939$, $\bar{X} = 18.4$, $SD = 43.5$) than the morning ($n = 850$, $\bar{X} = 16.7$, $SD = 29.7$).

Table 48: Number of detections and species richnes of avian groups detected in the morning vs. the evening on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Morning		Evening	
Avian Groups	Detections	Richness	Detections	Richness
Ground birds	96	9	116	6
Raptors	3	2	6	4
Songbirds	850	49	939	40

I compared bird detections by transect in the morning and the evening (Table 49) and found that the transects in the evenings were associated with a higher number of detections and diversity (detections; $n = 1108$, $\bar{X} = 69.3$, diversity; $\bar{X} = 15.4$), than the morning (detections; n = 902, $\bar{X}$ = 56.4, diversity; $\bar{X}$ = 14.3). I found that the number of detections per transect (t = -0.7, df = 28, p = 0.479) and the diversity of birds detected per transect (t = -0.4, df = 29, p = 0.719) did not differ significantly between morning and evening.

Table 49: Comparison of detections and species richness of bird s observed per transect in the morning and evening from Tswalu Kalahari Reserve from June 2017 to May 2018.

	Morning		Evening	
Transect no.	Detections	Richness	Detections	Richness
Transect 1	94	12	87	23
Transect 2	133	31	0	0
Transect 3	38	18	46	28
Transect 4	0	0	170	22
Transect 5	54	17	33	8
Transect 6	35	12	90	17
Transect 7	29	10	42	14
Transect 8	113	24	47	22
Transect 9	0	0	84	13
Transect 10	128	28	0	0
Transect 11	61	21	28	11
Transect 12	51	16	29	9
Transect 13	102	13	50	17
Transect 14	0	0	175	25
Transect 15	30	11	162	19
Transect 16	34	16	65	19

I compared morning and evening bird diversity on a species level (Figure 72). I found that species richness was higher in the morning (S = 61.0), than the evening (S = 51.0). Additionally, species diversity was marginally higher in the morning (H' = 0.05) than the evening (H' = 0.04).

I compared group wise diversity between morning and evening detections; I found ground birds (Figure 73) had a higher species richness in the morning (S = 10.0) than evening (S = 7.0). Additionally, ground bird species diversity was higher in the morning (H' = 0.15) than evening (E= 0.08). For raptors I found species richness was higher in the morning (S = 4.0) than the evening (S = 3.0) (Figure 74). However, raptor species diversity was higher in the evening (H' = 0.33) than the morning (H' = 0.16). For the songbirds (Figure 75). I found species richness higher in the morning (S = 50.0) than in the evening (S = 40.0), while species diversity I found to be higher in the morning (H' = 0.06) than in the evening (H' = 0.05).

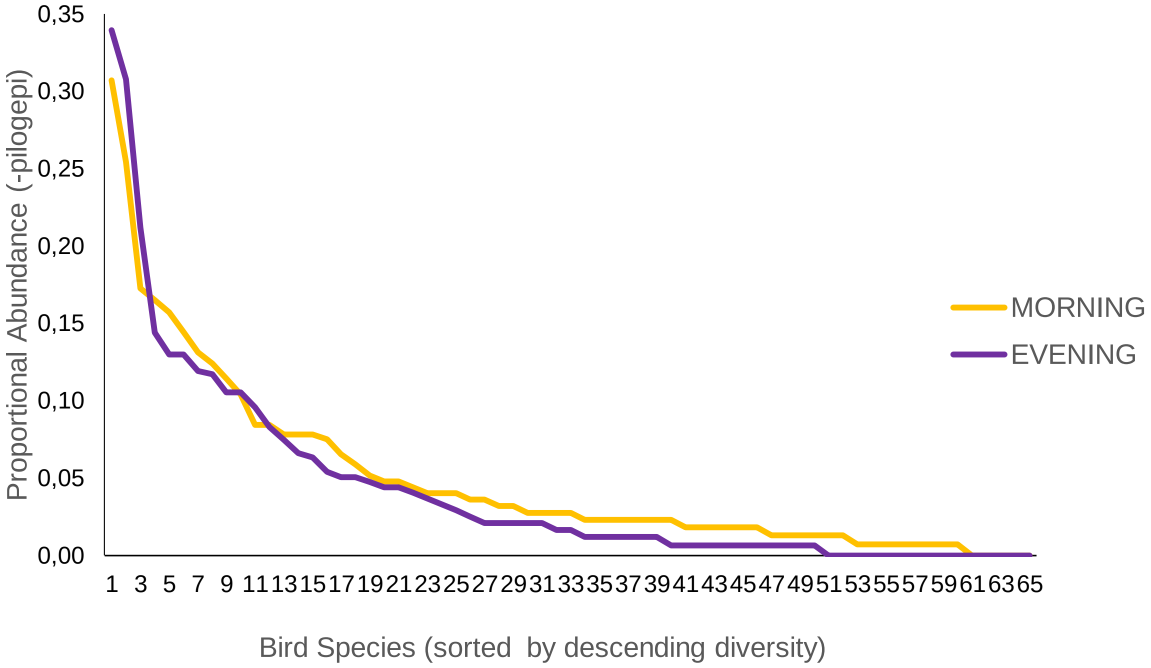

Figure 72: Relative diversity of all the species of birds detected in the morning vs. the evening on Tswalu Kalahari Reserve between June 2017 and May 2018.

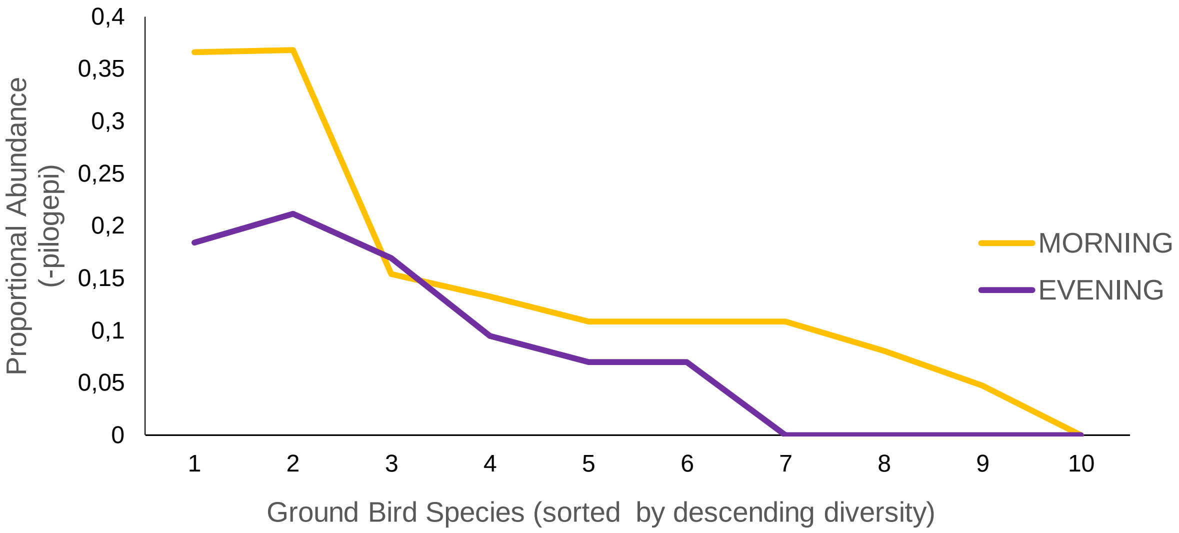

Figure 73: Relative diversity of ground birds detected in the morning vs. the evening on Tswalu Kalahari Reserve between June 2017 and May 2018.

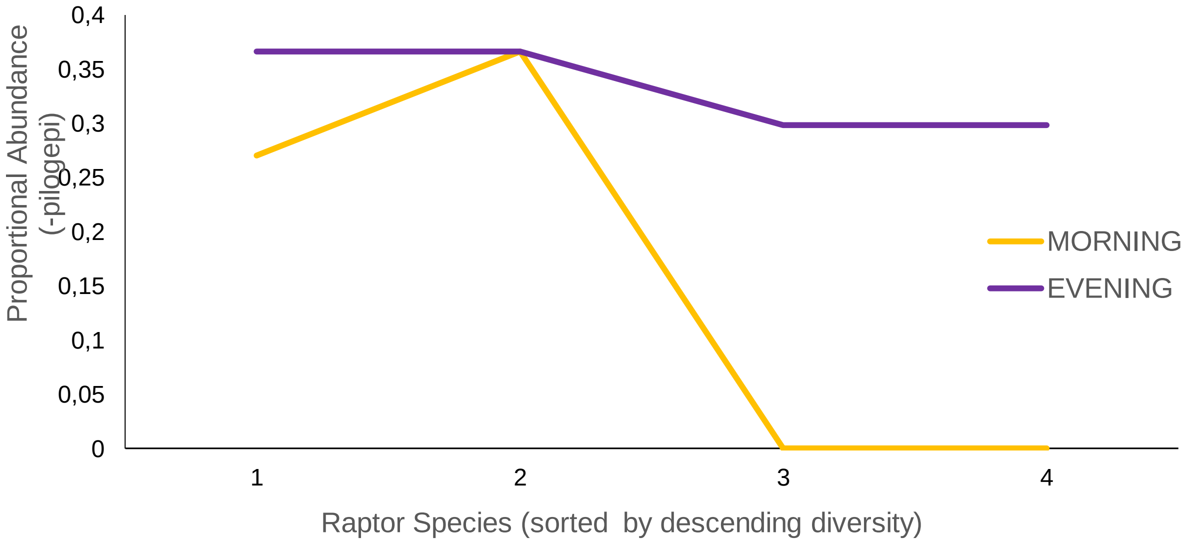

Figure 74: Relative diversity of raptors detected in the morning vs. the evening on Tswalu Kalahari Reserve between June 2017 and May 2018.

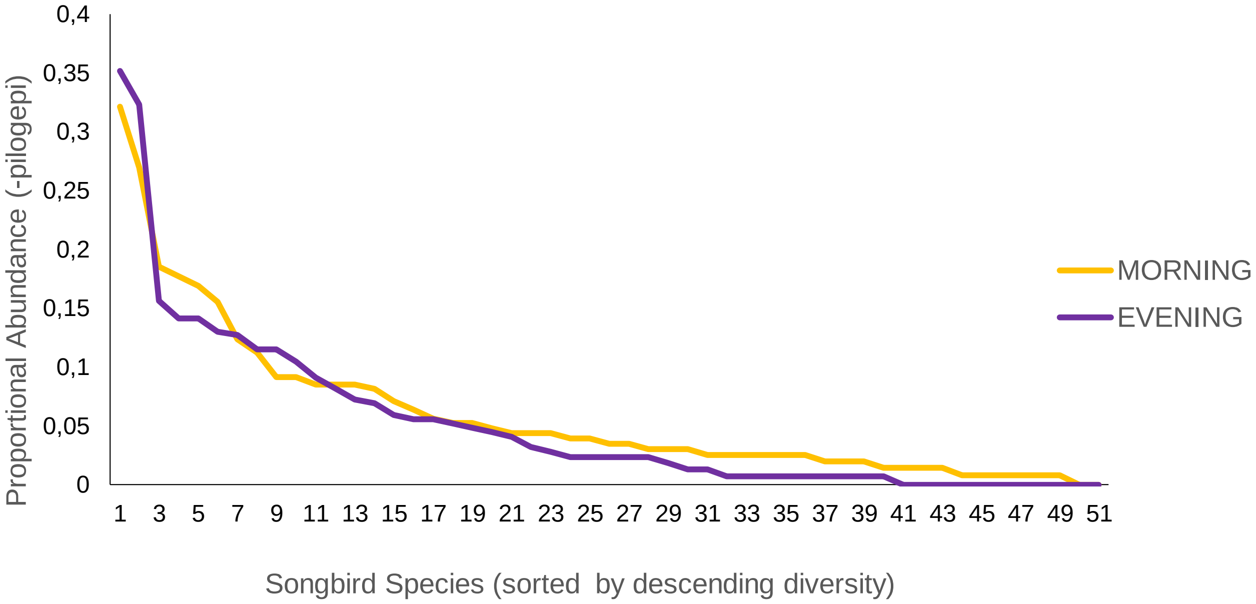

Figure 75: Relative diversity of songbirds detected in the morning vs. the evening on Tswalu Kalahari Reserve between June 2017 and May 2018.

When I compared diversity per transect between the morning and the evening (Appendix I - Table A39), I found that species richness was higher in the morning (S = 15.0) than the evening (S = 14.0), yet species diversity was equal for the morning and evening (H’ = 0.15) (Figure 76).

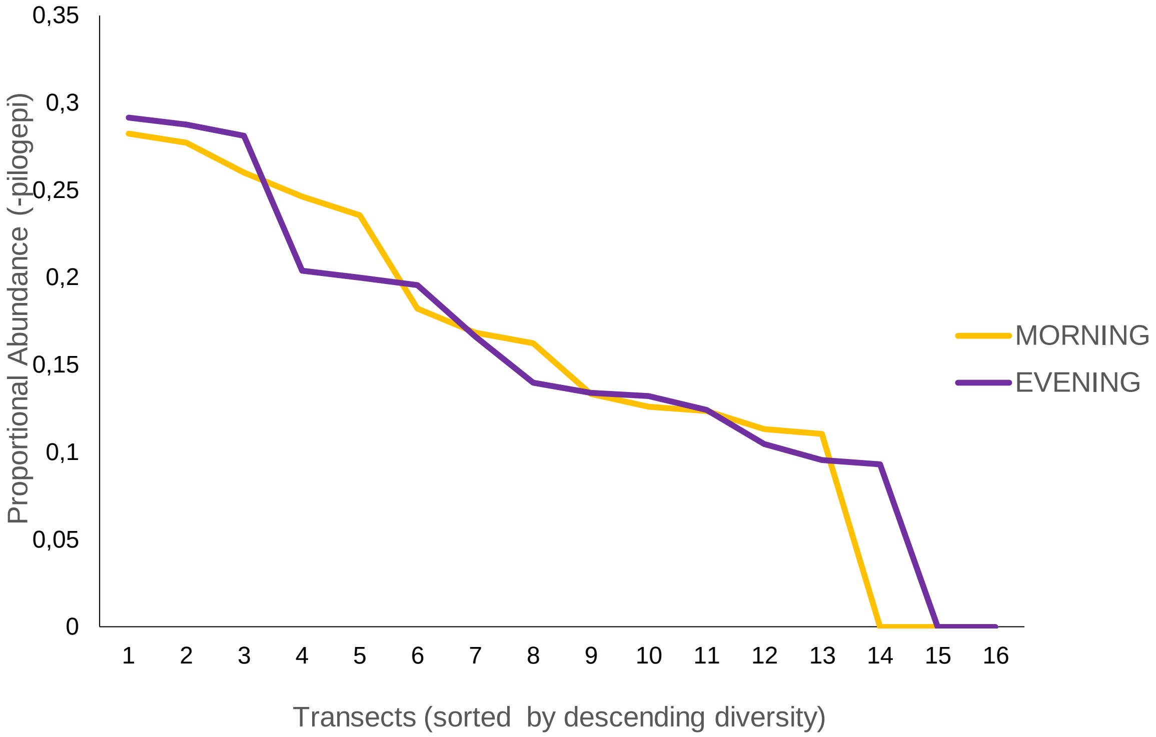

Figure 76: Relative diversity of avian detections, per transect, in the morning vs. the evening on Tswalu Kalahari Reserve between June 2017 and May 2018.

CHAPTER 5 – DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

“It may be better to be a live jackal than a dead lion, but it is better still to be a live lion. And usually easier.”
-Robert A. Heinlein

My study followed a similar design to that of Crooks and Soulé (1999), who investigated the mesopredator release hypothesis and how the decline of coyotes (*Canis latrans* Say, 1823), in conjunction with habitat fragmentation, affected the abundance and distribution of smaller carnivores and their avian prey species. They concluded that trophic interactions, combined with habitat fragmentation, structures ecological communities. I conducted this study to determine whether I could detect evidence of mesopredator release in a South African context, specifically in the Kalahari. Tswalu Kalahari Reserve (TKR) comprises two separate components with different top predator management regimes. The Korannaberg section has cheetah (*Acinonyx jubatus* Schreber, 1775), and wild dog (*Lycaon pictus* Temminck 1820) as top predators and lions (*Panthera leo* Linnaeus, 1758) and spotted hyaenas (*Crocuta crocuta* Erxleben, 1777) have been excluded. On the Lekgaba section a resident lion population is actively managed and species such as cheetah and wild dog have been excluded. This, therefore seemed an ideal location to test whether the predictions of the mesopredator release hypothesis held in the Kalahari context. I observed that on TKR there is a four-tiered predator hierarchy; lions as apex predators, wild dog, cheetah, leopard (*Panthera pardus* Linnaeus, 1758), and brown hyaena (*Hyaena* brunnea Thunberg, 1820) as second-tier, top predators, black-backed jackals (*Lupulella mesomelas* Schreber, 1775), caracals (*Caracal caracal* Schreber, 1776), African wild cat (*Felis silvestris* Schreber, 1777), bat-eared fox (*Otocyon megalotis* Desmarest, 1822), and Cape fox (*Vulpes chama* A. Smith, 1833), as third-tier mesopredators, and with the fourth and lowest tier comprising small predators like mongooses (*Herpestes* Illiger, 1811),
and genets (*Genetta* G.[Baron] Cuvier, 1816). The mesopredator release hypotheis postulates that the distribution, abundance and behaviour of the middle-ranking predators will alter in response to a decrease in density or distribution of

apex/ top predators (Crooks and Soulé 1999; Prugh *et al*. 2009). In terms of TKR I expected lions to exert top-down control on all other predators. Yet, in the absence of lions I expected the second-tier predators to affect the mesopredators relative abundance and diversity. If this top-down control was evident on the TKR I would have expected a higher relative abundance of mesopredators on Korannaberg (lions absent) than on Lekgaba (lions present), accompanied by a lower abundance of their main prey species likely small ungulates, small mammals, and birds (Crooks and Soulé 1999).

5.1 Camera trap survey

My camera survey to investigate the relative abundance of mesopredators and their prey suggested that there was no difference, in the relative abundance or diversity, of these assemblages between Korannaberg and Lekgaba. Had there been evidence of mesopredator release then I would have expected a significantly higher relative abundance of mesopredators on Korannaberg than Lekgaba. Several studies, local and international, have shown that mesopredator release is not ubiquitous and does not necessarily succeed changes in top predator identity (Johnson *et al.* 2007; Woodgate 2014; Jachowski *et al*. 2020). In their review of 38 mesopredator release studies, Jachowski *et al.* (2020) found that only 11 comprised a tiered predator hierarchy that had more than two levels, similar to that on the TKR, where the structure of the predator assemblage comprised more levels than simply the apex and mesopredators. Of these 11 studies approximately 46% showed strong support for mesopredator release, with another 46% showing no support for mesopredator release and only one study showing, what they term “mixed support” (where multiple metrics for a common variable differed in supporting/ not supporting mesopredator release) (Jachowski *et al.* 2020). In Australia, the elimination of dingoes (*Canis lupus dingo* Meyer, 1793) from large parts of the continent has allowed smaller introduced predators to devastate small marsupial populations and, in some cases, resulting in their extinction (Johnson *et al.* 2007). Many Australian mammals benefit from the management of dingoes and top predators are crucial in maintaining prey biodiversity in large terrestrial ecosystems (Hayward and Marlow 2014; Johnson *et al.* 2007). Yet, a comparative study in Kwazulu-Natal, within the savanna and Indian Ocean coastal belt biomes, showed that the relative abundance of mesopredators (caracal, black-backed jackal, side-striped jackal (*Lupulella adustus* Sundevall, 1847), honey badger (*Mellivora capensis* Schreber, 1776) and serval (*Leptailurus serval* Schreber 1776)) were not suppressed by apex predators (leopard, spotted and brown hyaena) (Woodgate 2014). This suggests that in KwaZulu-Natal bottom-up drivers had a greater influence on the relative abundance of predators than top-down forces (Woodgate 2014). Direct comparisons of species assemblages between different locations, in the past, have been known to be complicated, as approaches vary in the probability of detecting various species and consequently vary in detecting changes in dynamics associated with mesopredator release hypothesis (Harmsen *et al*. 2010). This may be attributable to several reasons including, but not limited to, the home ranges of species being limited by the lower limit of their nutritional requirements in relation to food production in its environment and an upper limit of the animal's ability to range beyond this area to find high quality food patches (Mott 2010). Additionally, the dispersal potential and home-ranges of arid-adapted mammals are much larger than those of similar species in wetter habitats (du Toit 1990; Fennessy 2009).

My camera trap study did, however, show that there was a significant seasonal difference in relative abundance of mammals on the TKR and that considerably more detections were recorded in the cold season than in either of the two warmer seasons. Seasonal movement patterns of terrestrial animals vary among assemblages but most have been well documented (McNaughton 1990; Nathan *et al.* 2008). Common causes of animal movement are linked to environmental change and include anthropogenic habitat fragmentation, changes in land-use patterns, climate, and exotic species introduction (Nathan *et al.* 2008). These movements can also range from foraging to migration, depending on the distance and purpose of the movement (Nathan *et al.* 2008). African ungulates have been shown to occupy different geographical regions centred on seasonal constraints i.e. some species have been known to concentrate around permanent water sources in the drier seasons and move to the drier areas in the wet season (McNaughton 1990). Grazers like zebras (*Equus* Gray, 1821) and wildebeest (*Connochaetes taurinus* Burchell 1823) feed on short-grasses of high nutritional-value during the wet season and taller-grasses of lower nutritional-value, during the dry season (Tyrrell *et al.* 2017). Browsers like greater kudu (*Tragelphus strepsiceros* Pallas, 1766) and Grant's gazelle (*Nanger granti* Brooke 1872) move from the open grassland and bushveld areas in the wet season to the denser, thicker vegetation in the drier season (Tyrrell *et al.* 2017). The relatively high number of animal detections that I recorded in the cold-dry season when compared with either of the hot seasons might be as a result of constrained resources leading animals to move more in search of food, water and shelter (Durant *et al*. 1988; Johnson *et al*. 1992; Fennessy 2009; Avgar *et al.* 2013). As this study took place in a closed system, there was unlikely any migrations into or out of the reserve.

5.1.1 Predators

The mesopredator release hypothesis suggests that apex predators control mesopredators through top-down forces and when the apex predators are removed from a system the mesopredators will increase in diversity and abundance (Crooks and Soulé 1999; Prugh *et al.* 2009; Brook *et al.* 2012; Beasley *et al.* 2013; Ripple *et al.* 2013; Newsome *et al.* 2017; Jachowski *et al.* 2020). This was found to be the case in southern California where the decline of coyotes was associated with an increase in local populations of mesopredators (striped skunk (*Mephitis mephitis* Schreber 1776), raccoon (*Procyon lotor* Linnaeus 1758), gray fox (*Urocyon cinereoargenteus* Schreber 1775)) and exotic carnivores (domestic cat (*Felis catus* Linnaeus 1758) and opossum (*Didelphidae* Gray 1821)) (Crooks and Soulé 1999). An increase in mesopredators resulted in increased predation on local scrub-breeding birds (Crooks and Soulé 1999).

Tswalu Kalahari Reserve has been characterised as being a collection of a diverse array of habitats, each with its own characteristic vegetation which plays host to a diverse gathering of faunal assemblages (van Rooyen and van Rooyen 2017). This unique terrain spans Lekgaba and Korannaberg, the proximity of one to another made it viable for me to conduct this comparison between the two sections. The predator management strategy on the TKR presented a unique opportunity for me to compare predator relative abundance and diversity between the two sections, which have similar climatic conditions and vegetation, but differed in terms of top predators. I hypothesized that because of the presence of

Lion on Lekgaba there would be a lower relative abundance of mesopredators, like black-backed jackal, caracal, and Cape fox, while Korannaberg would support a higher diversity and abundance of mesopredators. I found that top predators varied between sites because of management, but there was no significant difference in the relative abundance or diversity of mesopredators between Korannaberg and Lekgaba. It seems that the relative abundance of species that contribute to this guild were similar, regardless of Korannaberg being approximately four times larger than Lekgaba. The key difference between Korannaberg and Lekgaba is that Lekgaba was stocked with two prides of lions (which numbered approximately 20 individuals) whereas lions are excluded from Korannaberg and its top predator tier comprised cheetah (which numbered approximately 4) and wild dogs (approximately 20). Despite the differences in top predators, the mesopredator and small predator assemblages were similar on Korannaberg and Lekgaba. There is a possible argument to be made that the top predators in the TKR system, regardless of section, have a similar influence on the mesopredators and small predators. This contention is supported by various studies on different apex and mesopredator species and the results showing changes in behaviour of the mesopredators in the presence of the larger predators (Prugh *et al.* 2009; Brook *et al*. 2012; Ramesh *et al.* 2017; Haswell *et al.* 2018; O'Regan 2019; Curveira-Santos *et al.* 2021). Another reason for the similarities between Korannaberg and Lekgaba, in terms of mesopredators, is that there is evidence that predator antagonisms may be dampened in lower productivity areas (like the Kalahari) (Hunter *et al.* 2007; Ritchie and Johnson 2009). An example of this was when the wild dog population declined to extinction in the Serengeti National Park, even though there was a high abundance of prey available, because the presence of the many top predators made hunting dangerous and energetically costly (Creel 2001). The wild dogs persist, however, in low numbers in areas associated with lions and hyaenas but in areas of lower prey abundance there was a lower top predator abundance which was associated with lower levels of interference and exploitation competition with the wild dogs and therefore increased numbers of them (Creel 2001; Ritchie and Johnson 2009). Recent research from Australia has contradicted the long held contention that dingoes restrict feral cat populations, and the removal of dingoes from pastoral rangelands in Western Australia has caused mesopredator release (Brook *et al.* 2012).

Kreplins *et al*. (2021) found that the restriction of dingo control will not benefit biodiversity, as was previously thought to be the case.

Ritchie and Johnson (2009) reviewed 61 studies on interactions between apex and mesopredators and found that the only instances where mesopredators were not suppressed by apex predators was when the mesopredators had specialized defences (like spines in porcupines (*Hystrix africaeaustralis* Peters, 1852) or chemical defences (like skunks (*Mephitis mephitis* Schreber, 1776) which are capable of shooting a noxious oily spray from their scent glands when threatened) (Prange and Gehrt, 2007). An alternative reason might have been because resource availability was so high that competitive interactions were significantly reduced (Gehrt and Prange, 2007; Prange and Gehrt, 2007).Coyotes and wolves (*Canis lupus* Linnaeus, 1758) are known to have a wide overlap in niches, spatially and temporally, when there is an abundance of food resources (Paquet 1992; Newsome and Ripple 2015). Another instance where mesopredators were not suppressed by top predators is where the mesopredators employed behavioural avoidance to elude top predators (Gehrt and Prange 2007; Haswell *et al.* 2018). An example of this is where cheetah seek out “refuge” areas which contain low densities of lion and hyaenas to persist in heterogeneous environments (Durant 1998). In their analysis, Ritchie and Johnson (2009) found that 95% of the studies showed evidence consistent with the predictions of the mesopredator release hypothesis, while the remaining 5% found no evidence of mesopredator release. Another review of mesopredator release studies conducted in North America (Jachowski *et al.* 2020) found that of the 38 relevant studies that they reviewed, almost half (46%) found no or limited support for mesopredator release in multi-predator communities that had lost an apex predator. They suggested that the widely accepted assumption that mesopredator release is universal in multi-predator communities, which have lost an apex predator, should be discarded as mesopredator release is context dependant and based on the spatial (i.e. home range overlapping), temporal (i.e. seasonal resource abundance availability) and ecological (i.e. intraguild competition) scales of the predators included (Prange and Gehrt 2007; Jachowski *et al.* 2020).

Past research (Brashares *et al.* 2010; Woodgate 2014; Panebianco *et al.* 2022) has shown that systems with animal assemblages comprising a variety of predators, mesopredators and prey species tend to be subject to weak top-down forces, which means that these systems are likely to display lower levels of mesopredator release if top predators are removed. Past research on mesopredator dynamics in the Kalahari found that black-backed jackals constrain subordinate mesopredators' (like Cape fox and bat-eared fox) populations through lethal and non-lethal mechanisms (Mills and Bester 2005; Blaum *et al.* 2009; Bagniewska and Kalmer 2013). Even though I detected black-backed jackal frequently on Korannaberg and Lekgaba, my detections of small and mesopredators were similar in both sections. I found that predator diversity and richness was similar on Korannaberg and Lekgaba, implying an equal diversity of predators across TKR. The only difference in predator assemblages between Korannaberg and Lekgaba were a consequence of top predator management strategy. I detected no differences in predator diversity beside the active exclusion of several top predators on by the TKR management strategy.

The only difference between Korannaberg and Lekgaba in terms of predator assemblage composition, that was not a function of active management, was two detections of caracal on Lekgaba, and none on Korannaberg. I suspect this is because caracals seem to favour elevated rocky habitats (Avenant *et al.* 2016) that are more prevalent on Lekgaba than on Korannaberg (van Rooyen and van Rooyen 2017). Jackals tend to prefer open plains, such as those that are typical of Korannaberg (Bothma 2012; Avenant *et al.* 2016). A further variable that might have influenced the lack of detections of caracal on Korannaberg might be associated with the randomisation process for camera deployment. Had I stratified the habitats and then randomised within habitat types, I might have detected caracal on Korannaberg. Despite the lack of camera trap detections, the presence of caracals on Korannaberg was verified by observations by field rangers and other researchers on site*. Another possible reason I did not detect caracal on Korannaberg could be because caracal densities and home range size are highly variable and are based on a variety of environmental variables like, habitat type, vegetation density, prey composition and human disturbance (Bothma 2012; Avenant *et al.* 2016). Furthermore, where black-backed jackals and caracals are sympatric, jackals are known to be competitively dominant over caracals and prey on caracal kittens (Ferreira 1988; Bothma 2012; Du Plessis *et al.* 2015) and I recorded more jackal detections on Korannaberg than Lekgaba.

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* Prince Ngomane, Sustainability Officer, Tswalu Kalahari Reserve, Personal Comms.June 2017 to May 2018

I found no seasonal differences in the relative abundance or diversity of mesopredators on TKR. Black-backed jackal was the mesopredator that I detected most frequently (both year round and on a seasonal basis). This might be attributable to jackals being both a food and habitat generalist. However, it also helps that a consistent prey species is available as a result of resource availability being relatively consistent throughout the year on TKR because of the provision salt licks and supplemental feeding (van Rooyen and van Royen 2017). Black-backed jackals are omnivores with wide variation in their diets, which include hunted and scavenged meat and a variety of plant materials including fruits and seeds (Grafton 1965; Mills and Bester 2005; Klare *et al.* 2010; Bothma 2012). The composition of their diet is likely related to resource availability (Tambling *et al.* 2018). Black-backed jackal diets in the eastern Karoo were primarily comprised of small ungulates (Van de Ven *et al.* 2013). In this region, jackals maintained a consistent biomass intake year-round, however, the composition of their diet varied seasonally in line with resource availability (Van de Ven *et al.* 2013). In their study, Van de Ven *et al.* (2013) hypothesized that the stability of the ungulate resource for jackals was facilitated by larger predators, specifically cheetahs, which provided scavenging opportunities for jackals. Other carnivores like lions and leopards are also instrumental in providing carrion sources for jackals (Minnie *et al.* 2016)*,* therefore there is likely to be facilitation for the jackals both on Korannaberg and Lekgaba by top predators, but confirmation of this requires further investigation. Facilitation might also explain the seasonal diversity of the diversity and abundance of mesopredators in the TKR.

5.1.2 Ungulates

The mesopredator release hypothesis predicts that I should have expected to record a higher diversity and abundance of large ungulates on Korannaberg, where lions are absent, than on Lekgaba (Carbone *et al.* 1999; Owen-Smith and Mills 2008; Ritchie and Johnson 2009; Brashares *et al.* 2010; Tambling *et al.* 2018). Furthermore, I would have expected a lower diversity and abundance of small ungulates on Korannaberg than Lekgaba as a consequence of an anticipated higher number of mesopredators (Owen-Smith and Mills 2008; Ritchie and Johnson 2009; Brashares *et al.* 2010; Tambling *et al.* 2018). I recorded more ungulate detections on Lekgaba than on Korannaberg, yet there was no significant difference in detections between sites on a species level, by index or between camera traps. Environments typified by low productivity, like arid and semi-arid habitats, have their structure influenced more by abiotic factors, like climate variation, than biotic interactions (Roemer *et al.* 2009). If not controlled by top-down forces, ungulate diversity and distribution can be attributed to a variety of factors including resource distribution and availability, habitat, and ungulate body size (Jarman 1974; Gordon 2003; Szemán *et al.* 2021; Forbes and Kerley 2022). Smaller-bodied herbivores are typically concentrate selectors (Hanley, 1982; Muller *et al.* 2013) and have more specialized dietary requirements than larger herbivores (Hanley 1982; Muller *et al.* 2013). Small herbivores forage on the leaves and fruits of nutritious, low-density shrub and tree species, and tend to monopolize such resources within their territories (Jarman 1974; McNaughton and Georgiadis 1986; Muller *et al.* 2013). Larger-bodied herbivores are typically less selective in their feeding habits and forage on lower quality, abundant vegetation, like grass (Jarman 1974; McNaughton and Georgiadis 1986; Muller *et al.* 2013), in the Kalahari.

Over and above bottom-up forces, herbivore abundance is also highly susceptible to top-down mechanisms such as predation (Hairston *et. al*. 1960; Frank 2008). In Yellowstone National Park the reintroduction of wolves has been shown to affect not only their ungulate prey, but also plants, and the microbial activity that influences the grasslands energy and nutrient dynamics (Frank 2008). A study conducted on wild guanacos (*Lama glama* Linnaeus 1758) in Patagonia, Argentina, showed males of this species were more abundant in areas of increased primary productivity which also contained a higher number of female groups (Panebianco *et al*. 2022). This study concluded that bottom-up processes and social traits are stronger regulators of large herbivore spatial distribution than top-down processes like predation risk (Panebianco *et al*. 2022). In Africa, several studies have shown that bovid grazers such as buffalo (*Syncerus caffer* Sparrman, 1779) and wildebeest (*Connochaetes taurinus* Burchell, 1823) are more affected by bottom-up processes, like grass cover, than by top-down processes like predation (Schein 1987; Grange and Duncan 2006). Understanding the feeding ecology of African herbivores (top-down vs. bottom-up controlled) is fundamental in understanding patterns of herbivore species diversity, while herbivore body size and home range is an important determinant of resource partitioning, and community properties (McNaughton and Georgiadis 1986).

In addition to the natural vegetation, herbivores are provisioned with salt licks on the TKR (Abraham *et al.* 2021). Artificial salt licks are commonly used on agricultural and wildlife establishments to encourage the health, growth and development of domesticated and wild animals (Milner *et al.* 2014; Murray *et al.* 2016; Simpson *et al.* 2020). Their use of salt licks is a good illustration of how wild animals seek out nutrition which is essential to their survival (Lameed and Adetola 2012; Murray *et al.* 2016). Salt licks also concentrate animal activity in specific locations and provide opportunities for a variety of anthropogenic pursuits including; photography, tourism, hunting and wildlife tracking and research (Lameed and Adetola 2012). Salt licks play a pivotal role in the distribution of animals. This may have influenced detection rates on some of my cameras as some cameras were deployed near watering points where salt licks were also available.

When assessed together, I found a significant seasonal difference in the detections of ungulates (large and small combined). However, there was no seasonal significant difference in detections for small ungulates on either Korannaberg or Lekgaba. The ungulates on TKR are aided in terms of bottom-up resources, whether natural or anthropogenic, either in the form of natural foliage, supplemental feeding in the dry season, artificial salt-licks or pumped water provided at watering holes year round. The question remains whether they are subject to top-down pressures from top and mesopredators.

In the Kruger National Park, lion predation has a strong influence on ungulate populations, however, there are several factors that provide “relief” to ungulates from high predation rates (Owen-Smith 2008). One of these factors is the distribution of surface water provisioned at managed water holes which has been shown to expand the distribution of several ungulate species and ultimately reduces predation pressure (Owen-Smith 2008; Van Rooyen and Van Rooyen 2017). There are presently 25 permanent watering points on the TKR. Theoretically, the reserve only requires 23 such points, based on a 4 km radius around each watering point, this is established on the average range distance of water-dependant ungulates (i.e. wildebeest and impala (*Aepyceros melampus* Lichtenstein, 1812) (Van Rooyen and Van Rooyen 2017). Therefore, there is likely a surfeit of water for ungulates on TKR, which might dilute the local threat of predation. Another factor that provides relief is that some predators, lions in particular, have seasonal shifts in prey preferences specifically in species and sex (Owen-Smith 2008). Of the five small ungulate species that I detected on the cameras, only two species are water independent (springbok (*Antidorcus marsupialis* Zimmermann, 1780) and steenbok (*Raphicerus campestris* Thunberg, 1811)). In the past, the lions on Lekgaba have shown a prey-preference for smaller ungulates like warthog (*Phacochoerus aethiopicus* Pallas 1766) and steenbok (Roxburgh 2008). Their interest in smaller prey is said to be characteristic of Kalahari lions as they tend to hunt alone or in smaller groups as predation on larger species requires larger prides (Eloff 2002; Roxburgh 2008). Studies on the seasonal prey preferences of lion on TKR, however, still need to be conducted.

Predators have been found to synchronize their birthing seasons so that they align with periods of high resource availability (Mills and Bester 2005). At uKhahlamba-Drakensberg Park ungulate carcasses are most abundant at the end of the dry winter season, and black-backed jackal births were found to be timed to coincide with this abundant food source (Rowe-Rowe 1978; Mills and Bester 2005). Jackal reproduction in the Northern Cape is seasonal and the birthing season is from July to October (Mills and Bester 2005), which coincides with the cold-dry and hot-dry seasons. Caracal births have been recorded year-round, however the peak in births occur from October to February (Mills and Bester 2005) which coincides with the two hot seasons on TKR. Cheetahs and Leopards are not subject to a breeding season and cubs can be born at any time in the year, whereas lions have been found to synchronize their birthing season to their prey i.e. when ungulates have their young so do lions (Mills and Bester 2005). Another factor that provides relief to predation for ungulates is low rainfall patterns (Owen-Smith 2008). High rainfall provides for an increase in vegetation cover and grass height which provides better concealment for stalking and hunting predators (Owen-Smith 2008). The lions on Lekgaba were found to have a distinct selection for certain habitat types on TKR (Roxburgh 2008). They favoured the open and bushy plains, and avoided the mountainous and hilly areas, and the dune crests (Roxburgh 2008). All of which plays an influential role in the hunting potential of the lions (Grant *et al*. 2005).

African savanna ungulate behaviour and reproduction patterns are strongly correlated to seasonal changes in environmental conditions (Ogutu *et al.* 2008; Van Beest *et al.* 2013). Ungulate diversity and abundance are associated with fluctuations in rainfall, which drives ungulate movements, reproduction, and survival (Ogutu *et al.* 2008). Reproductive activities of ungulates, changing vegetation and resource availability correspond to seasonal changes, specifically to rainfall variation (Ogutu *et al.* 2008; Owen-Smith 2008). Consequently, rainfall influences ungulates' vulnerability to predation. In the context of TKR, ungulate births are generally at their highest in the rainy season (hot-wet season in the TKR) (Sinclair *et al.* 2000), therefore the elevated large ungulate detections in the cold-dry season are possibly because of the ungulate births from the previous season (hot-wet) and the small ungulates' elevated detection in the hot-wet season coincide with their birthing season.

My data also showed that species diversity and richness was equal for both sections of the reserve, Korannaberg and Lekgaba, for large and small ungulates across all three seasons. Therefore, even though the two sides support different assemblages of ungulates there was no seasonal difference in terms of diversity of ungulate species. Ungulate diversity is a product of spatial, temporal, and taxonomic diversity of the vegetation structure (Du Toit and Cumming 1999). The consistency of the ungulate diversity is likely due to the border fence that prohibits the movement of ungulates and therefore their diversity and richness remains constant (Hayward and Kerley 2009).

5.2 Small mammals

In line with the predictions of the mesopredator release hypothesis, I anticipated higher small mammal abundance and diversity on Lekgaba (in the presence of lions) than Korannaberg (Ritchie and Johnson 2009; Tambling *et al.* 2018). Yet, I found no significant difference in the number of captures of small mammals or in my indices of relative abundance of small mammals between Korannaberg and Lekgaba. Equally, my small mammal population estimates did not differ significantly between Korannaberg and Lekgaba for majority of species with the exception of the desert pygmy mouse (*Mus indutus* Thomas, 1910), West-central

South African four-striped grass rat (*Rhabdomys bechaunae* Thomas Morong, N. Britton, 1893) and pouched mouse (*Saccostomus campestris* Peters, 1846).

The desert pygmy mouse has been found to have a far-reaching habitat tolerance, ranging from arid scrub of the savanna to water-watered vegetation along the banks of the Okavango Delta (Skinner and Chimimba 2005). The four-striped grass mouse occurs in a wide range of habitats so long as there is good grass cover (Skinner and Chimimba 2005). The pouched mouse is categorised as a generalist murid that has been captured in all provinces in South Africa, yet has been found to be partial to specific habitat and vegetation types (SANBI 2016). They occur in a diverse collection of habitats ranging from the fringes of lowland forests to savanna open *Vachellia* spp. bushveld but in the Kalahari they have be found to prefer short grass habitats surrounding dry pans (Skinner and Chimimba 2005). Even though all three small mammal species have a wide tolerance for different habitat types, the higher number of desert pygmy mouse, four-striped grass mouse and pouched mouse captures on Lekgaba than Korannaberg is likely indicative of habitat preference or a more suitable habitat type. In general, the vegetation on Lekgaba is more bushveld like than that of Korannaberg and is more of an open bushveld with a medium-tall tree layer (Van Rooyen and Van Rooyen 2017).

My results showed that small mammal diversity differed significantly between Lekgaba and Korannaberg. Species richness and diversity were substantially higher on Lekgaba than Korannaberg. Small mammal diversity is directly associated with abiotic factors and vegetation variables features including, grass height, tree density and ground cover (Delcros *et al.* 2015). On a finer scale, the spatial distribution of desert rodents is influenced by microhabitat characteristics (Brown 1973; Price 1978; Harris 1986; Delcros *et al*. 2015). The high diversity of seed-eating rodents in arid environments is linked to high number of desert plants that produce seeds (Brown 1973; Abramsky and Rosenzweig 1984; Yip and Dickman 2023).

The results from my seasonal small mammal capture experiments were similar to those from the Eastern Cape where small mammal captures peaked in winter (Do Linh San *et al.* 2011). Furthermore, small mammal species richness and diversity in the Eastern Cape were highest in spring (Do Linh San *et al.* 2011). I recorded my highest number of captures in the cold-dry (winter) season on TKR. In the savanna biome in Kwazulu-Natal the highest rodent diversity was associated with winter (Delcros *et al.* 2015). This study also showed that rodent and shrew diversity in the savanna biome has substantial seasonal variation associated with microhabitat features (Delcros *et al.* 2015). Arid and semi-arid regions that are associated with relatively low primary productivity, like the TKR, have small mammal populations that are more strongly influenced by rainfall than predation (Jaksic *et al*. 1997; Shenbrot 2014). My population estimates showed no significant seasonal variation for most of the species, except for bushveld gerbil (*Gerbiliscus leucogaster* Peters, 1852), pygmy hairy-footed gerbil (*Gerbillurus paeba* A. Smith, 1836), and pouched mouse. These species were more abundant in the cold-dry season than in either of the other seasons. Rainfall has been shown to play a pivotal role in the abundance patterns of gerbils and mice across southern Africa (Skinner and Chimimba 2005). My results for small mammal diversity showed a difference in species richness among seasons, however species diversity was similar across all three seasons implying consistent diversity for small mammals year-round. This may be a consequence of the diversity being influenced by changes in habitat structure and not season (Do Linh San *et al.* 2011). Past research has shown significant seasonal variation in pouched mouse abundance (Keesing 1998) with a peak during the warm, wet summer months when their young are born (Skinner and Chimimba 2005). Changes in availability of foods for rodents affect their foraging behaviour, diet quality, maternal nutrition and juvenile recruitment, and population dynamics (Curtis and Perrin 1979; Skinner and Chimimba 2005). Individual rodent populations within multispecies assemblages fluctuate in a non-synchronized manner because of species-specific responses to environmental variability (Shenbrot 2014). This seems to be the case in TKR where certain species have displayed season-specific variation in both abundance and relative abundance while others do not.

Seasonal shifts in small mammal diets are common and various species may use leaves or seeds during cooler seasons and then use insects in the summer months (Nel 1978; Nel *et al.* 1984; Skinner and Chimimba 2005). The small mammals that I captured were primarily granivorous (60%). This is consistent with work conducted in Israel (Brown 1973) and North-West America (Abramsky and Rosenzweig 1984) where arid zones have high levels of seed availability. My results contrasted with those of similar semi-arid locations in South Africa (the

Kalahari (Kerley *et al.* 1990; Kuechly *et al.* 2011) and the Karoo (Kerley 1992), South America (Mares *et al.*1977), and Australia (Morton 1979; Yip and Dickman 2023). Several of the species that I captured tend to show seasonal variation in their feeding habits which is thought to be because of the unpredictability of resource availability in arid zones (Skinner and Chimimba 2005). The Kerley *et al.* (1990) study, conducted in the Kalahari, resulted in the capture of seven small mammal species, all of which were represented in my data set. This is important for my study because granivory is thought to be the most logical diet for small mammals in semi-arid environments, like TKR, as the availability of foliage and insects tend to be seasonally variable in these areas (Nel *et al.* 1984; Kerley *et al.* 1990). Despite this, the pouched mouse is the only species that I caught that has a specific adaptation to granivory (cheek pouches) (Kerley *et al.* 1990). Pouched mice have strongly seasonal diets which are comprised primarily of seeds and arthropods in the dry seasons and then they seem to ‘switch’ to herbivory on forbs during the wet season (Keesing 1998; Skinner and Chimimba 2005). In addition to their diets, rodent reproductive cycles are linked to seasonal food availability (Curtis and Perrin 1979; Skinner and Chimimba 2005). This is also likely to have influenced the seasonal variation in the small mammal assemblage on the TKR.

I found that small mammal morphometries varied seasonally for several species. Typically, I found larger individuals were associated with the warmer seasons. Bushveld gerbils have an annual cycle of reduced size in winter months when food is limited (Korn 1989). Pouched mice have body sizes strongly correlated with seasonal rainfall patterns, with smaller body sizes being recorded in drier seasons than in wet seasons (Keesing 1998; Skinner and Chimimba 2005). This aligns with my findings as the individuals that I captured in the hot-wet season were larger than those I captured in either of the other seasons. Pouched mice reproduce in the hot-wet season, and this contributes to the increase in size and weight due to enlarged scrotal testes in males and females gravid with young (Skinner and Chimimba 2005). Four-striped grass mice and bushveld elephant shrews (*Elephantulus intufi* A. Amith, 1836) are seasonal breeders and tend to breed in warmer and wetter seasons (Skinner and Chimimba 2005) which, along with a greater availability of resources, may account for the increase in size of these small mammals in the warmer months. However, I did not find seasonal differences in the morphometries of four-striped grass mice on TKR. Bushveld elephant shrews differed significantly in head lengths among seasons, with the smaller head lengths being associated with the hot-wet season when compared with the other two seasons. A possible reason for this could be that the smaller head lengths associated with the hot-wet season could be because of a higher number of juveniles as their young are born in the warmer, wet summer months (Skinner and Chimimba 2005).

5.3 Avian populations

Birds, particularly ground-nesting birds, form an important component of predator diets (Ritchie and Johnson 2009; Berger *et al.* 2010; Ritchie *et al.* 2012). I assumed that this was the case in the TKR as ground-nesting birds make up a substantial portion of black-backed jackal diets (Grafton 1965; Mills and Bester 2005). Cape foxes actively hunt ground birds in open arid habitats (Mills and Bester 2005). African wild cats' diets include a diversity of birds such as doves, korhaans, quails, and weavers (Mills and Bester 2005). I expected the avian diversity and abundance of the TKR to be higher on Lekgaba than on Korannaberg in line with the predictions of the mesopredator release hypothesis. The withdrawal of apex predators from natural systems has been shown to result in increased numbers and activity of mesopredators and a resultant increase in predation pressure on native avian species (Crooks and Soulé 1999). Coyote removal caused an increase in mesopredator activity, which in turn caused an increased local extinction of scrub-breeding birds (Crook and Soulé, 1999). Gordon *et al.* (2017) found that the presence of an apex predator, the dingo, was associated with increases in little button-quail (*Turnix velox* Gould 1841) populations. The abundance of little button-quail was highest in areas where dingoes were present and seemingly limited mesopredators like red fox (*Vulpes vulpes* Linnaeus, 1758) and feral cats (*Felis catus* Linnaeus, 1758) (Gordon *et al.* 2017). In the UK, several avian species (pigeons, raptors, owls, woodpeckers and passerines), mostly tree-dwelling birds, are not limited by predation by foxes, however, several ground-nesting birds (seabirds, gamebirds and waders) are limited by predation (Roos *et al.* 2018). In the arctic tundra, the expansion of the red fox population is playing a pivotal role in the decline of bird populations (Henden *et al.* 2021). Long-term management practices to control the red fox populations seem to limit the decline of willow ptarmigan, *Lagopus lagopus* (Linnaeus, 1758), a socio-economically important arctic-native grouse species (Henden *et al.* 2021). Conversely, Lloyd (2007) showed that bird nest predation rates by mesopredators did not differ between the rangelands, where predator control was practiced, and protected areas, where there was no predator control. This was the case on TKR, as I found no differences in avian populations between the two sections.

My results showed no significant differences in detections of specific avian species or bird functional groups (ground birds, raptors, or songbirds) between Korannaberg and Lekgaba. Avian species richness was higher on Korannaberg than Lekgaba, but species diversity was similar on both sections. Bird diversity in the Kalahari is directly linked to vegetation structure (Hudson and Bouwman 2007; Seymour and Dean 210) and areas that have vegetation that is more structurally diverse have higher levels of bird diversity (Hudson and Bouwman 2007). Predation on avian species in the Kalahari is not a strong driver of diversity. It is more likely that vegetation cover is more important for avian diversity in this landscape (MacArthur and MacArthur 1961; Fleishman *et al.* 2003; Pascoe *et al.* 2021). Predation on birds by black-backed jackals has been found to be predominately opportunistic (Tambling *et al.* 2018) and it is unlikely that predation by mesopredators substantially influences bird populations on the TKR. Caracals too, prey on a variety of birds (Bothma 2012; Melville *et al.* 2004). Ground-nesting birds comprise a small percentage of Cape fox diets in semi-arid environments (Mills and Bester 2005). Mongooses are thought to be primary predators of bird nests in arid and semi-arid environments (Lloyd 2007). As is the case elsewhere, it is apparent that mesopredators prefer ground-nesting birds and their eggs; this is likely the case on TKR as well.

Although I recorded no difference in the bird assemblages between sections, I did note a seasonal difference in the relative abundance of total avian populations on TKR with significantly more detections, of all species, in the cold-dry season than during the hot seasons. Of the 64 avian species that I encountered on TKR, five are migratory species (two bee-eater species, two shrike species and one cuckoo) which visit the Kalahari in the summer months and two are partially migratory (one flycatcher and one wheatear) which travel east in the winter (Roberts VII Multimedia, 2012). Therefore, I anticipated that there should have been a decline in diversity and relative abundance of birds on TKR during the cold-dry season, yet I recorded more detections during this time. Bonter *et al.* (2013) found that during colder weather birds' foraging behaviour alters and is biased towards avoidance of starvation rather than avoidance of predation-risk. Therefore, birds forage more frequently during the colder months and will feed until they reach satiation to accumulate more fat (Bonter *et al.* 2013). This might explain why my results report more avian detections in the cold-dry season than in warmer seasons.

Avian foraging periods are typically bimodal with an early morning peak in foraging activity which declines to a period of relative inactivity, during the middle of the day, followed by a second peak in foraging activity just before sunset (Bednekoff and Houston 1994; McNamara *et al.* 1994; Bonter *et al.* 2013). However, I found no significant difference between the total avian detections in the morning or the evening on TKR.

My results for avian diversity showed species richness and diversity was higher in the morning than in the evening. A study conducted relatively near to TKR at Kuruman River Reserve, showed that the daily increase in temperature had no effect on the foraging effort (amount of time spent actively foraging) on the population of southern pied bablers (*Turboides bicolor* Jardine, 1831*)* native there (Du Plessis *et al.* 2012), which is consistent with my findings on TKR. In south-eastern Alaska birds' peak detectability was in the morning (Kessler and Milne 1982) and the authors advised that sampling periods should conform to the activity patterns of individual species and if only an abundance index is required, counts should take place at midday. However, in Punjab, Pakistan, an urban environment surrounded by cultivated land, showed no difference in bird detections between morning and evening (Yousaf *et al.* 2020). Foraging behaviour of birds is not constant and is dependent on several abiotic and biotic factors including, microhabitat, predation risk, and vegetation structure and abundance (Bonter *et al.* 2013; Yousaf *et al.* 2020). The primary factor influencing avian diversity in arid regions is rainfall because rainfall controls vegetation structure which serves as sustenance, a dwelling place, and a place of refuge for birds (Fleishman *et al.* 2003; Tischler *et al.* 2013). It seems that similarity in my morning and evening detections may be because of a combination of abiotic factors and ecological interactions.

5.4 Conclusion

The motivation for my study can be thoroughly summed up in the proceeding statement; Mesopredators play a vital role in predator–prey interactions and ecosystem functioning. This role may be even more pronounced in areas where large carnivores are absent (e.g. farmlands, nature reserves with a comprehensive predator management programme), leaving mesopredators, like black-backed jackals and caracals, elevated to the role of top predators (Minnie *et al.* 2016). Few studies have evaluated evidence (or lack thereof) for the mesopredator release hypothesis in the African (Creel and Creel 1996; Durant 1998, 2000; Creel 2001; Schuette *et al.* 2013; Taylor *et al.* 2016) and more specifically South African context (Lloyd 2007; Taylor 2015). To my knowledge, my study is the first of this nature in the Kalahari. I wanted to establish whether I could detect signals possibly associated with mesopredator release within the Kalahari system. The primary objective of my study was to determine whether the presence of lions might have a cascading impact of alternating trophic levels on the TKR. My strategy was to assess the relative abundance of mesopredators and their prey assemblages. I found no evidence of differences in relative abundance or diversity of assemblages of mesopredators, small ungulates, small mammals, or birds under either of the top-predator management regimes practiced on TKR.

There are several possible reasons that might explain why I did not find evidence of the mesopredator release hypothesis on TKR. Mesopredator populations have been shown to have a relatively fast recovery rate when population numbers are reduced either through predation or hunting and can increase quickly over a short span of time (Conner and Morris 2015). Mesopredators may be regulated more by bottom-up (prey abundance) than top-down forces (Brashares *et al.* 2010). Since the vegetation types are similar on both Korannaberg and Lekgaba bottom-up control may be similar in both areas. Another possible reason that might explain the lack of evidence for mesopredator release could be experimental error. During my assessments of relative abundance of specific groups I could rather have been seeing a reflection of the species' range use instead or absolute abundance counts (Crooks and Soulé 1999). Relative abundance measures the number of animals and their activity at one camera compared with another (Gilbert *et al.* 2021) whereas absolute abundance is considered less biased as such estimates predict the number of individuals present in an area (Hopkins and Kennedy 2004). My sample time scale could have been too short (Brashares *et al.* 2010). Another factor that may have affected my data collection was the unusually high amount of rain that fell in the hot-wet season of 2017 while my sampling took place (SAWS 2020). My study aimed to investigate multiple species interactions on several trophic levels. The complex food-web dynamics of the TKR may not be reduced to a single casual mechanism (Brashares *et al.* 2010). Past research suggests that the mesopredator release will only occur if certain conditions are met, firstly, top-down forces on mesopredators, by top predators, needs to exceed that on their shared prey species and secondly, the carrying capacity of mesopredators needs to exceed a certain threshold (Takimoto and Nishijima 2022). I am of the view that these conditions were not met in the case of TKR because it seems that the pressure on prey species by both meso- and top predators was much higher than that imposed on the mesopredators by top predators. I determined this from my results of the high relative abundance of mesopredators recorded on both Korannaberg and Lekgaba and the lack of difference in relative abundance of mesopredators between the two different predator regimes.

Due to active management on TKR there will always be food resources available year-round; the provisioning of salt licks, supplemental feeding (Lucerne, antelope pellets or nutritional supplements) and artificial watering points ensure that the ungulates are well nourished (van Rooyen and van Rooyen 2017) which provides an abundant food source for the predators which results in little competition and weak top-down forces within the predator assemblage (Letnic and Dickman 2010). The lack of evidence for mesopredator release on the TKR suggests that there is little concern for trophic cascading, from a management standpoint (Ripple *et al.* 2016). Even though the apex predators do not seem to have a substantial effect on the mesopredators, the presence of black-backed jackal on TKR may influence sympatric mesopredators like caracal, bat-eared foxes and Cape foxes (Tambling *et al*. 2018), but that will require further investigation. Elmhagen and Rushton (2007) demonstrated that mesopredator release is weaker in arid environments where trophic regulation is more bottom-up controlled, than top-down forced. Another pattern that emerged from my study was the seasonal differences in relative abundances of ungulates, small mammals, and birds, associated with relatively high diversity levels across all groups being evident throughout the year.

This is important as it suggests that species diversity and abundance in arid areas relies more on bottom-up forces and changes in abiotic factors, like rainfall, than top-down forces like predation (Ogutu *et al.* 2008; Tischler *et al.* 2013; Pascoe *et al.* 2021).

My study was based on the assumption that both Korannaberg and Lekgaba are situated within essentially the same vegetation type. I therefore did not take variation in vegetation into account in my study design. This might have masked some of the finer scale dynamics at play in my study system. However, I anticipated that the randomisation process that I employed, associated with the deployment of my sampling effort, should have partially mitigated vegetation variability. I am aware that the two sections of the TKR are slightly different in terms of vegetation structure, however I am not aware of any other sites in the region that have strict predator management regimes, in relatively similar vegetation types, that might be better suited to an assessment of the possible manifestation of mesopredator release. Korannaberg is larger than Lekgaba and during my study I applied an equal sampling effort on both sides, however, because of the size difference, I ultimately sampled five times more intensively on Lekgaba than on Korannaberg. Therefore, I may have under-sampled the Korannaberg section.

My recommendations for future mesopredator studies on TKR are that a similar study be conducted, over a prolonged period of time, to dispel any question that the year in which I conducted my study was an anomaly. I would suggest that camera trapping be conducted over an extended period with more cameras to include more comprehensive sampling of Korannaberg which included a stratified approach that accounted for variation in vegetation. My results highlighted the high relative abundance of jackals within the TKR system. Therefore, it might be wise to initiate an investigation into the functional role of this mesopredator in this system.

CHAPTER 6 – REFERENCES

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APPENDICES

Appendix I

Appendix I - Table A1: GPS co-ordinates of camera trap locations set up for this study on Tswalu Kalahari Reserve between June 2017 and May 2018.

Korannaberg[1]		Lekgaba[2]	
Location	Co-ordinates	Location	Co-ordinates
1	-27.209078, 22.351832	1	-27.193429, 22.410175
2	-27.208009, 22.376036	2	-27.203200, 22.385799
3	-27.215795, 22.347883	3	-27.201979, 22.393352
4	-27.214574, 22.385821	4	-27.201063, 22.400219
5	-27.221748, 22.339472	5	-27.200299, 22.406055
6	-27.221449, 22.355952	6	-27.200299, 22.421848
7	-27.221596, 22.382731	7	-27.207170, 22.392837
8	-27.241515, 22.349235	8	-27.206559, 22.407943
9	-27.241362, 22.380306	9	-27.205948, 22.414638
10	-27.250519, 22.348549	10	-27.205643, 22.424595
11	-27.249603, 22.379791	11	-27.216177, 22.383739
12	-27.261354, 22.345459	12	-27.215719, 22.390434
13	-27.260743, 22.358333	13	-27.215108, 22.399017
14	-27.259980, 22.368633	14	-27.214039, 22.408458
15	-27.264253, 22.370521	15	-27.213276, 22.416698
16	-27.275239, 22.370865	16	-27.213276, 22.428028
17	-27.273255, 22.390091	17	-27.222130, 22.396442
18	-27.281189, 22.367431	18	-27.221062, 22.406913
19	-27.280426, 22.376873	19	-27.220298, 22.427513
20	-27.279511, 22.387687	20	-27.228694, 22.400219
21	-27.305292, 22.343742	21	-27.227625, 22.409317
22	-27.304835, 22.352325	22	-27.227167, 22.428371
23	-27.303004, 22.371036	23	-27.239683, 22.399360
24	-27.302089, 22.380306	24	-27.239073, 22.418758
25	-27.301631, 22.390777	25	-27.23892, 22.429401
26	-27.318409, 22.346145	26	-27.244262, 22.395241
27	-27.317494, 22.356102	27	-27.243346, 22.414638
28	-27.316579, 22.368461	28	-27.242888, 22.426826
29	-27.325425, 22.375671	29	-27.248688, 22.414982
30	-27,324662, 22.391807	30	-27,248230, 22.430088

[1]Lions Absent; [2]Lions Present

Appendix I – Table A2: Total camera trap triggers captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

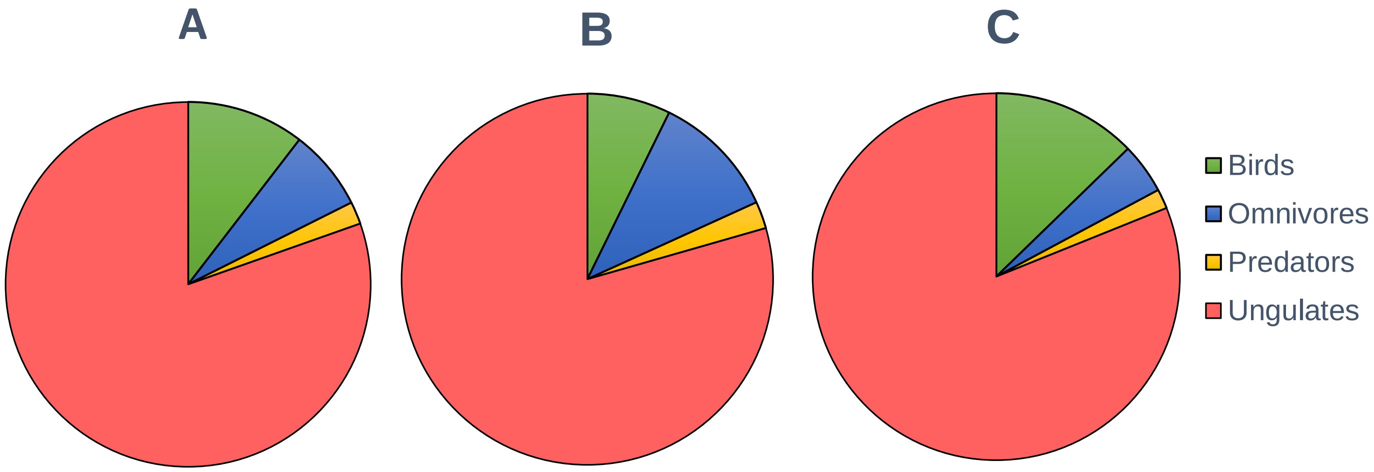
	Korannaberg[1]			Lekgaba[2]		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Camera 1	713	237.7	290.9	322	107.3	117.6
Camera 2	323	107.7	158.6	476	158.7	88.0
Camera 3	1231	410.3	607.1	454	151.3	216.2
Camera 4	987	329.0	312.6	614	204.7	145.8
Camera 5	977	325.7	446.7	729	243.0	211.5
Camera 6	515	171.7	99.1	2397	799.0	825.1
Camera 7	981	327.0	310.0	104	34.7	50.0
Camera 8	1248	416.0	289.4	261	87.0	87.5
Camera 9	614	204.7	345.0	167	55.7	44.9
Camera 10	54	18.0	16.1	280	93.3	66.0
Camera 11	368	122.7	63.9	490	163.3	68.6
Camera 12	1057	352.3	439.2	2927	975.7	975.6
Camera 13	548	182.7	192.7	691	230.3	116.7
Camera 14	218	72.7	26.7	388	129.3	107.0
Camera 15	728	242.7	223.1	1126	375.3	626.7
Camera 16	1268	422.7	450.6	1649	549.7	829.2
Camera 17	1363	454.3	157.2	1248	416.0	219.1
Camera 18	742	247.3	365.2	624	208.0	278.2
Camera 19	1295	431.7	370.7	1219	406.3	339.8
Camera 20	1297	432.3	523.3	479	159.7	136.4
Camera 21	1464	488.0	423.7	284	94.7	83.1
Camera 22	1105	368.3	488.2	781	260.3	241.1
Camera 23	1402	467.3	487.9	973	324.3	545.3
Camera 24	1156	385.3	514.2	394	131.3	202.9
Camera 25	1684	561.3	495.8	1311	437.0	425.6
Camera 26	434	144.7	194.1	1265	421.7	289.4
Camera 27	32	10.7	18.5	121	40.3	69.9
Camera 28	192	64.0	110.9	0	0.0	0.0
Camera 29	1084	361.3	298.2	1707	569.0	457.6
Camera 30	814	271.3	89.6	522	174.0	98.9

[1]Lions Absent; [2]Lions Present

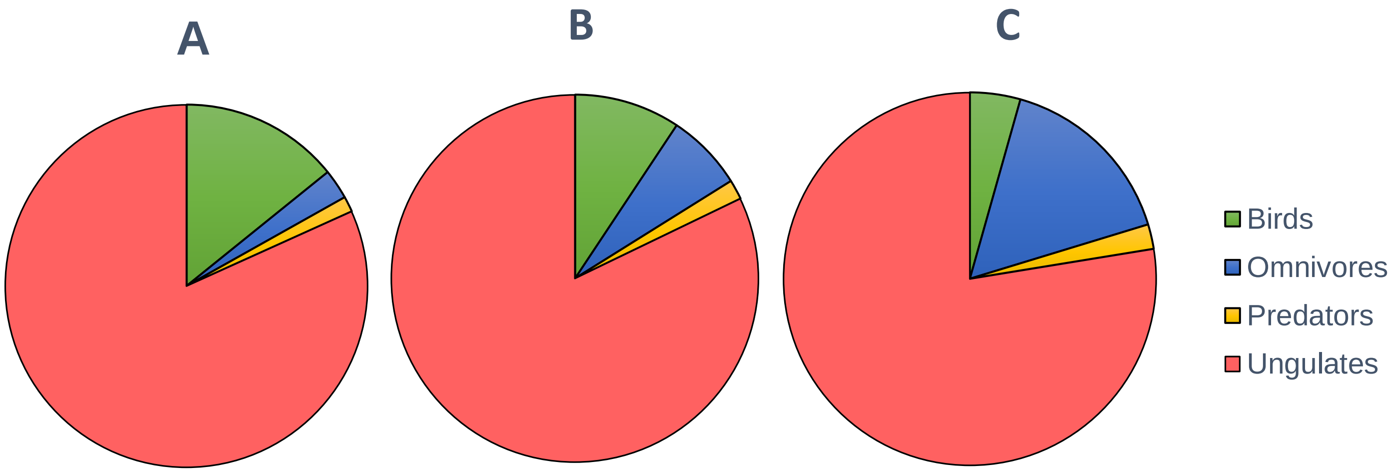
Appendix I – Table A3: Camera trap detections of animals on Tswalu Kalahari Reserve between June 2017 and May 2018

	Korannaberg[1]			Lekgaba[2]		
Camera Trap no.	n	$\bar{X}$	SD	N	$\bar{X}$	SD
Camera 1	232	77.3	63.1	139	46.3	66.9
Camera 2	120	40.0	53.4	232	77.3	48.0
Camera 3	106	35.3	20.6	18	6.0	6.6
Camera 4	247	82.3	102.8	307	102.3	123.6
Camera 5	342	114.0	184.5	70	23.3	15.9
Camera 6	144	48.0	33.7	50	16.7	23.8
Camera 7	205	68.3	74.2	31	10.3	16.2
Camera 8	10	3.3	2.1	14	4.7	8.1
Camera 9	77	25.7	40.2	62	20.7	34.1
Camera 10	41	13.7	11.8	190	63.3	46.5
Camera 11	74	24.7	18.8	133	44.3	38.7
Camera 12	775	258.3	416.4	272	90.7	70.3
Camera 13	46	15.3	17.2	218	72.7	34.9
Camera 14	70	23.3	33.5	73	24.3	14.0
Camera 15	51	17.0	14.0	32	10.7	10.5
Camera 16	35	11.7	12.5	1418	472.7	771.4
Camera 17	112	37.3	57.7	53	17.7	16.5
Camera 18	54	18.0	14.9	92	30.7	33.9
Camera 19	35	11.7	6.7	14	4.7	3.1
Camera 20	134	44.7	50.1	9	3.0	1.7
Camera 21	6	2.0	3.5	5	1.7	2.9
Camera 22	16	5.3	9.2	58	19.3	25.3
Camera 23	454	151.3	197.0	18	6.0	3.0
Camera 24	101	33.7	27.2	16	5.3	9.2
Camera 25	90	30.0	20.7	10	3.3	2.3
Camera 26	260	86.7	139.8	116	38.7	56.7
Camera 27	20	6.7	11.5	1	0.3	0.6
Camera 28	0	0.0	0.0	0	0.0	0.0
Camera 29	365	121.7	86.0	14	4.7	6.4
Camera 30	214	71.3	35.0	340	113.3	121.3

[1] Lions absent; [2] Lions present



Appendix I – Figure A1: Proportional contribution of various vertebrate groups to the camera trap detections of animals on (A) Tswalu Kalahari Reserve (B) Korannaberg and (C) Lekgaba from June 2017 to May 2018.



Appendix I – Figure A2: Proportional contribution of various vertebrate groups to the camera trap detections of animals recorded in the (A) Cold-Dry (B) Hot-Dry and (C) Hot-Wet season captured on Tswalu Kalahari Reserve from June 2017 to May 2018.

Appendix I – Table A4: Full list of all the species captured by camera traps, on Tswalu Kalahari Reserve between June2017 and May 2018.

Species		Number of Camera Captures per Location			Number of Camera Captures per Season		
Latin Name	Common Name	Tswalu	Korannaberg	Lekgaba	Cold-dry	Hot-dry	Hot-wet
Large Predator							
Acinonyx jubatus	Cheetah	2	2	0	1	1	0
Hyaena brunnea	Brown Hyaena	32	10	22	19	8	5
Lycaon pictus	Wild Dog	9	9	0	0	0	9
Panthera leo	Lion	73	0	73	62	3	8
Panthera pardus	Leopard	9	8	1	8	0	1
Mesopredator							
Lupulella mesomelas	Black-backed Jackal	290	155	135	117	53	120
Caracal caracal	Caracal	2	0	2	0	1	1
Felis silvestris cafra	African Wild Cat	41	22	19	8	16	17
Genetta genetta	Small-spotted Genet	2	1	1	2	0	0
Otocyon megalotis	Bat-eared Fox	50	28	22	20	9	21
Vulpes chama	Cape Fox	20	8	12	7	6	7
Small Predator							
Cynictis penicillata	Yellow Mongoose	86	78	8	10	6	70
Ictonyx striatus	Striped Polecat	3	1	2	2	0	1
Suricata suricatta	Suricate	51	9	42	41	4	6

Appendix I – Table A4: Full list of all the species captured by camera traps, on Tswalu Kalahari Reserve between June2017 and May 2018.

		Number of Camera Captures per Location			Number of Camera Captures per Season		
Latin Name	Common Name	Tswalu	Korannaberg	Lekgaba	Cold-dry	Hot-dry	Hot-wet
Large Ungulate							
Connochaetes taurinus	Blue Wildebeest	4362	906	3456	3516	474	372
Damaliscus lunatus	Tsessebe	57	57	0	30	4	23
Equus quagga	Plains Zebra	5440	88	5352	4660	540	240
Equus zebra hartmannae	Hartmanns Mountain Zebra	695	92	603	580	56	59
Giraffa camelopardalis	Giraffe	859	756	103	103	67	689
Hippotragus equinus	Roan Antelope	52	52	0	14	6	33
Hippotragus niger	Sable Antlope	104	104	0	13	6	85
Kobus ellipsiprymnus	Waterbuck	0	1	0	0	0	1
Oryx gazella	Gemsbok	4984	3613	1371	1376	779	2829
Taurotragus oryx	Common Eland	1873	1092	781	1141	174	558
Tragelaphus strepsiceros	Greater Kudu	617	406	211	185	262	170
Redunca fulvorufula	Mountain Reedbuck	14	14	0	0	14	0
Syncerus caffer	African Savanna Buffalo	78	78	0	77	0	1
Small Ungulate							
Aepyceros melampus	Impala	742	308	434	217	330	195
Antidorcus marsupialis	Springbok	3010	1599	1411	568	1077	1365
Phacochoerus africanus	Common Warthog	1632	1449	183	887	141	604
Raphicerus campestris	Steenbok	103	38	65	26	29	48
Sylvicapra grimmia	Common Duiker	235	104	131	51	44	140

Appendix I – Table A4: Full list of all the species captured by camera traps, on Tswalu Kalahari Reserve between June2017 and May 2018.

Species		Number of Camera Captures per Location			Number of Camera Captures per Season		
Latin Name	Common Name	Tswalu	Korannaberg	Lekgaba	Cold-dry	Hot-dry	Hot-wet
Large Ungulates							
Diceros bicornis	Hook-lipped (Black) Rhinoceros	135	0	135	105	7	23
Hystrix africaeaustralis	Cape Porcupine	64	19	45	37	13	14
Lepus saxatilis	Scrub Hare	41	13	28	21	10	10
Orycteropus afer	Aardvark	7	3	4	3	1	2
Papio ursinus	Chacma Baboon	2209	1470	739	427	315	1467
Pedetes capensis	Southern African Springhare	27	14	13	9	5	13
Proteles cristatus	Aardwolf	13	2	11	6		2
Xerus inauris	Southern African Ground Squirrel	44	18	26	1	2	41
Reptile							
Stigmochelys pardalis	Leopard Tortoise	8	5	3	0	3	5
Avian							
Afrotis afraoides	Nothern Black Korhaan	4	3	1	0		
Alopochen aeyptiaca	Egyptian Goose	86	0	86	0	84	2
Aquila rapax	Tawny Eagle	3	0	3	0	3	0
Ardeotis kori	Kori Bustard	84	21	63	67	10	7
Bubo africanus	Spotted Eagle Owl	1	0	1	1	0	0
Burhinus capensis	Spotted Thick-knee	4	2	2	1	0	3
Calendulauda africanoides	Fawn-coloured Lark	2	0	2	2	0	0
Erythropygia paena	Kalahari Scrub Robin	2	1	1	1	1	0
Gyps africanus	White-backked Vulture	1	1	0	0	0	1
Lamprotornis nitens	Cape Glossy Starling	3	0	3	3	0	0
Lanius colllaris	Common Fiscal	1	1	0	1	0	0
Lophotis ruficrista	Red-crested Korhaan	4	0	4	2	2	0
Melierax canorus	Pale Chanting Goshawk	9	0	9	9	0	0
Merops hirundineus	Swallow-tailed Bee-eater	1	1	0	0	1	0

Appendix I – Table A4: Full list of all the species captured by camera traps, on Tswalu Kalahari Reserve between June2017 and May 2018.

Species Latin Name	Common Name	Tswalu	Korannaberg	Lekgaba	Cold-dry	Hot-dry	Hot-wet
Avian							
Numida meleagris	Helmeted Guineafowl	2897	741	2156	2266	325	306
Oena capensis	Namaqua Dove	4	0	5	4	1	0
Philetairus socius	Sociable Weaver	50	50	0	0	50	0
Plocepasser mahali	White-browed Sparrow Weaver	25	14	11	5	7	13
Pterocles bicinctus	Double-banded Sandgrouse	75	0	75	75	0	0
Sagittarius serpentarius	Secretarybird	34	34	0	1	0	33
Streptopelia capicola	Cape-turtle Dove	4	0	4	4	0	0
Struthio camelus	Common Ostrich	166	127	39	93	25	48
Tadorna cana	South African Shelduck	2	0	2	2	0	0
Tockus leucomelas	Southern Yellow-billed Hornbill	27	0	27	24	3	0
Vanellus coronatus	Crowned Lapwing	41	28	13	25	2	14

Appendix I - Table A5: Seasonal camera trap detections of all species detected on Tswalu Kalahari Reserve from June 2017 to May 2018.

	n			$\bar{X}$			SD		
	Cold-Dry	Hot-Dry	Hot-Wet	Cold-Dry	Hot-Dry	Hot-Wet	Cold-Dry	Hot-Dry	Hot-Wet
Camera 1	128	137	106	64.0	68.5	53.0	83.4	74.2	75.0
Camera 2	209	39	104	104.5	19.5	52.0	4.9	3.5	70.7
Camera 3	50	60	14	25.0	30.0	7.0	17.0	35.4	9.9
Camera 4	76	250	228	38.0	125.0	114.0	7.1	169.7	120.2
Camera 5	13	38	361	6.5	19.0	180.5	2.1	17.0	207.2
Camera 6	90	63	41	45.0	31.5	20.5	56.6	17.7	27.6
Camera 7	34	52	150	17.0	26.0	75.0	17.0	33.9	106.1
Camera 8	1	5	18	0.5	2.5	9.0	0.7	3.5	7.1
Camera 9	6	73	60	3.0	36.5	30.0	2.8	50.2	42.4
Camera 10	132	81	18	66.0	40.5	9.0	63.6	29.0	12.7
Camera 11	74	86	47	37.0	43.0	23.5	22.6	49.5	30.4
Camera 12	39	152	856	19.5	76.0	428.0	12.0	96.2	439.8
Camera 13	104	69	91	52.0	34.5	45.5	73.5	0.7	47.4
Camera 14	17	24	102	8.5	12.0	51.0	6.4	11.3	15.6
Camera 15	18	11	54	9.0	5.5	27.0	2.8	7.8	8.5
Camera 16	1369	77	7	684.5	38.5	3.5	959.5	17.7	0.7
Camera 17	23	37	105	11.5	18.5	52.5	9.2	21.9	72.8
Camera 18	91	54	1	45.5	27.0	0.5	30.4	2.8	0.7
Camera 19	23	20	6	11.5	10.0	3.0	4.9	8.5	1.4
Camera 20	30	7	106	15.0	3.5	53.0	18.4	2.1	67.9
Camera 21	6	0	5	3.0	0.0	2.5	4.2	0.0	3.5
Camera 22	64	10	0	32.0	5.0	0.0	22.6	7.1	0.0
Camera 23	380	20	72	190.0	10.0	36.0	264.5	5.7	38.2
Camera 24	32	20	65	16.0	10.0	32.5	0.0	14.1	46.0
Camera 25	9	38	53	4.5	19.0	26.5	3.5	24.0	29.0
Camera 26	105	258	13	52.5	129.0	6.5	72.8	168.3	6.4
Camera 27	21	0	0	10.5	0.0	0.0	13.4	0.0	0.0
Camera 28	0	0	0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 29	72	84	223	36.0	42.0	111.5	50.9	42.4	154.9
Camera 30	78	353	123	39.0	176.5	61.5	43.8	98.3	34.6

Appendix I - Table A6: Seasonal camera trap detections per vertebrate group captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Tswalu			Korannaberg[1]			Lekgaba[2]		
	Cold-Dry	Hot-Dry	Hot-Wet	Cold-Dry	Hot-Dry	Hot-Wet	Cold-Dry	Hot-Dry	Hot-Wet
Birds	5169	930	848	1014	378	638	4155	552	210
Omnivores	971	677	3084	57	119	2883	914	558	201
Predators	504	173	414	105	77	366	399	96	48
Reptiles	0	3	5	0	0	5	0	3	0
Ungulates	29760	8192	15031	5584	3432	13020	24176	4760	2011

[1] Lions Absent; [2] Lions Present

Appendix I - Table A7: Index of predator detections (per 100 trap nights) on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Tswalu			Korannaberg[1]			Lekgaba[2]		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Large Predator									
Acinonyx jubatus	13.3	0.1	0.7	13.3	0.1	1.0	0.0	0.0	0.0
Hyaena brunnea	213.4	1.2	5.0	66.7	0.7	2.7	146.7	1.6	6.6
Lycaon pictus	60.0	0.4	0.1	60.0	0.7	6.3	0.0	0.0	0.0
Panthera leo	486.7	2.7	25.6	0.0	0.0	0.0	486.7	5.4	36.3
Panthera pardus	60.0	0.4	4.0	53.3	0.6	5.6	6.7	0.1	0.7
Mesopredator									
Lupulella mesomelas	1933.3	10.8	51.7	1033.3	11.5	56.8	900.0	10.0	46.8
Caracal caracal	13.3	0.1	0.7	0.0	0.0	0.0	13.3	0.1	1.0
Felis silvestris lybica	273.4	1.5	5.7	146.7	1.6	6.6	126.7	1.4	4.8
Genetta genetta	13.4	0.1	0.7	6.7	0.1	0.7	6.7	0.1	0.7
Otocyon megalotis	333.4	1.9	8.1	186.7	2.1	8.5	146.7	1.6	7.8
Vulpes chama	133.3	0.8	0.1	53.3	0.6	2.2	80.0	0.9	3.6
Small Predator									
Cynictis penicillata	573.3	3.2	0.6	520.0	5.8	43.6	53.3	0.6	3.3
Ictonyx striatus	20.0	0.1	0.9	6.7	0.1	0.7	13.3	0.1	1.0
Suricata suricatta	340.0	1.9	19.1	60.0	0.7	4.7	280.0	3.1	26.7

[1] Lions Absent; [2] Lions Present

Appendix I - Table A8: Shapiro-Wilk normality test results for mesopredator detections on Tswalu Kalahari Reserve from June 2017 to May 2018.

Species	W	p*
Lupulella mesomelas	1.0	0.8
Caracal caracal	1.0	0.7
Cynictis pencillata	1.0	0.6
Felis silvestris lybica	1.0	0.7
Genetta genetta	1.0	0.5
Hyaena brunnea	1.0	0.4
Otocyon melagotis	1.0	0.8
Panthera pardus	1.0	0.2
Vulpes chama	1.0	0.9

*(All values above 0.05 are normally distributed)

Appendix I - Table A9: Predator detections per camera trap on the Korannaberg (lions absent) section of Tswalu Kalahari Reserve between June 2017 and May 2018.

	Acinonyx jubatus	*Lupulella mesomelas*	*Caracal caracal*	*Cynictis penicillata*	*Felis silvestris lybica*	*Genetta genetta*	*Hyaena brunnea*	*Ictonyx striatus*	*Lycaon pictus*	*Otocyon megalotis*	*Panthera leo*	*Panthera pardus*	*Suricata suricatta*	*Vulpes chama*	n	X̄	SD
Camera 1	0	0	*0*	1	1	0	2	0	0	0	0	0	0	0	4	0.3	0.6
Camera 2	0	3	*0*	0	0	0	1	0	0	0	0	0	0	0	4	0.3	0.8
Camera 3	0	2	*0*	0	0	0	0	0	0	0	0	0	3	0	5	0.4	0.9
Camera 4	0	9	*0*	0	0	1	0	0	0	0	0	0	0	1	11	0.8	2.4
Camera 5	1	79	0	0	1	0	1	0	0	8	0	0	0	0	90	6.4	21.0
Camera 6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 7	0	1	0	3	0	0	0	0	0	0	0	0	0	0	4	0.3	0.8
Camera 8	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0.1	0.3
Camera 9	0	0	0	0	0	0	2	0	0	0	0	0	0	0	2	0.1	0.5
Camera 10	0	1	0	0	4	0	0	1	0	5	0	0	0	1	12	0.9	1.6
Camera 11	1	2	0	0	0	0	0	0	0	0	0	0	0	0	3	0.2	0.6
Camera 12	0	20	0	1	2	0	1	0	0	0	0	0	0	0	24	1.7	5.3
Camera 13	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0.1	0.3
Camera 14	0	1	0	0	0	0	1	0	0	0	0	0	0	0	2	0.1	0.4
Camera 15	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0.1	0.3
Camera 16	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	0.1	0.5
Camera 17	0	0	0	1	2	0	0	0	0	0	0	0	0	0	3	0.2	0.6
Camera 18	0	3	0	0	0	0	0	0	0	0	0	0	0	0	3	0.2	0.8
Camera 19	0	5	0	1	0	0	0	0	0	0	0	0	0	1	7	0.5	1.3
Camera 20	0	5	0	63	8	0	0	0	0	0	0	0	6	0	82	5.9	16.7
Camera 21	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0.1	0.3
Camera 22	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 23	0	7	0	0	0	0	0	0	0	0	0	0	0	0	7	0.5	1.9
Camera 24	0	4	0	0	0	0	0	0	0	6	0	0	0	0	10	0.7	1.9
Camera 25	0	3	0	4	0	0	0	0	0	4	0	0	0	0	11	0.8	1.6
Camera 26	0	7	0	0	0	0	0	0	0	0	0	0	0	1	8	0.6	1.9
Camera 27	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 28	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 29	0	2	0	2	2	0	2	0	9	0	0	0	0	0	17	1.2	2.4
Camera 30	0	0	0	3	2	0	0	0	0	3	0	8	0	0	16	1.1	2.3

Appendix I - Table A10: Predator detections per camera trap on the Lekgaba (lions present) section of Tswalu Kalahari reserve between June 2017 and May 2018.

	Acinonyx jubatus	*Lupulella mesomelas*	*Caracal caracal*	*Cynictis penicillata*	*Felis silvestris lybica*	*Genetta genetta*	*Hyaena brunnea*	*Ictonyx striatus*	*Lycaon pictus*	*Otocyon megalotis*	*Panthera leo*	*Panthera pardus*	*Suricata suricatta*	*Vulpes chama*	n	X̄	SD
Camera 1	0	1	0	0	1	0	1	0	0	10	0	0	0	0	13	0.9	2.6
Camera 2	0	2	0	0	0	0	0	0	0	0	11	0	0	0	13	0.9	2.9
Camera 3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 4	0	23	0	0	0	0	4	0	0	0	2	0	0	1	30	2.1	6.1
Camera 5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 6	0	4	0	0	2	0	1	0	0	0	2	0	0	0	9	0.6	1.2
Camera 7	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0.1	0.3
Camera 8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 9	0	0	0	0	3	0	0	0	0	0	8	0	0	0	11	0.8	2.2
Camera 10	0	0	0	0	0	0	1	0	0	5	0	0	2	6	14	1.0	2.0
Camera 11	0	2	0	0	0	0	0	0	0	0	0	0	2	1	5	0.4	0.7
Camera 12	0	2	0	2	4	0	0	1	0	0	0	0	0	0	9	0.6	1.2
Camera 13	0	11	0	0	0	0	0	0	0	1	0	1	0	0	13	0.9	2.9
Camera 14	0	0	0	0	0	0	0	0	0	1	0	0	0	1	2	0.1	0.4
Camera 15	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0.1	0.3
Camera 16	0	65	0	4	3	1	8	0	0	1	50	0	0	2	134	9.6	20.6
Camera 17	0	1	1	0	0	0	0	0	0	0	0	0	0	0	2	0.1	0.4
Camera 18	0	4	0	0	2	0	1	0	0	0	0	0	0	0	7	0.5	1.2
Camera 19	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 20	0	2	0	0	0	0	1	1	0	0	0	0	0	0	4	0.3	0.6
Camera 21	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 22	0	4	0	0	2	0	2	0	0	0	0	0	38	1	47	3.4	10.0
Camera 23	0	4	0	0	0	0	0	0	0	0	0	0	0	0	4	0.3	1.1
Camera 24	0	3	0	0	0	0	3	0	0	0	0	0	0	0	6	0.4	1.1
Camera 25	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 26	0	0	0	0	2	0	0	0	0	4	0	0	0	0	6	0.4	1.2
Camera 27	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 28	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 29	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.0	0.0
Camera 30	0	3	1	1	0	0	0	0	0	0	0	0	0	0	5	0.4	0.8

Appendix I - Table A11: Predator detections (per 100 tap nights) captured per camera on the Korannaberg (lions absent) section of Tswalu Kalahari Reserve from June 2017 to May 2018.

	Acinonyx jubatus	*Lupulella mesomelas*	*Caracal caracal*	*Cynictis penicillata*	*Felis silvestris lybica*	*Genetta genetta*	*Hyaena brunnea*	*Ictonyx striatus*	*Lycaon pictus*	*Otocyon megalotis*	*Panthera leo*	*Panthera pardus*	*Suricata suricatta*	*Vulpes chama*	n	$\bar{X}$	SD
Camera 1	0.0	0.0	0.0	6.7	6.7	0.0	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	26.7	1.9	4.1
Camera 2	0.0	20.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	26.7	1.9	5.5
Camera 3	0.0	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	0.0	33.3	2.4	6.2
Camera 4	0.0	60.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	73.3	5.2	15.9
Camera 5	6.7	526.7	0.0	0.0	6.7	0.0	6.7	0.0	0.0	53.3	0.0	0.0	0.0	0.0	600.0	42.9	140.0
Camera 6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 7	0.0	6.7	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	26.7	1.9	5.5
Camera 8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	6.7	0.5	1.8
Camera 9	0.0	0.0	0.0	0.0	0.0	0.0	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	1.0	3.6
Camera 10	0.0	6.7	0.0	0.0	26.7	0.0	0.0	6.7	0.0	33.3	0.0	0.0	0.0	6.7	80.0	5.7	10.7
Camera 11	6.7	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	1.4	3.9
Camera 12	0.0	133.3	0.0	6.7	13.3	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	160.0	11.4	35.3
Camera 13	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	6.7	0.5	1.8
Camera 14	0.0	6.7	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	1.0	2.4
Camera 15	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	6.7	0.5	1.8
Camera 16	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	13.3	1.0	3.6
Camera 17	0.0	0.0	0.0	6.7	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	1.4	3.9
Camera 18	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	20.0	1.4	5.3
Camera 19	0.0	33.3	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	46.7	3.3	9.0
Camera 20	0.0	33.3	0.0	420.0	53.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	40.0	0.0	546.7	39.0	111.2
Camera 21	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.5	1.8
Camera 22	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 23	0.0	46.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	46.7	3.3	12.5
Camera 24	0.0	26.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	40.0	0.0	0.0	0.0	0.0	66.7	4.8	12.4
Camera 25	0.0	20.0	0.0	26.7	0.0	0.0	0.0	0.0	0.0	26.7	0.0	0.0	0.0	0.0	73.3	5.2	10.5
Camera 26	0.0	46.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	53.3	3.8	12.5
Camera 27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 29	0.0	13.3	0.0	13.3	13.3	0.0	13.3	0.0	60.0	0.0	0.0	0.0	0.0	0.0	113.3	8.1	16.2
Camera 30	0.0	0.0	0.0	20.0	13.3	0.0	0.0	0.0	0.0	20.0	0.0	53.3	0.0	0.0	106.7	7.6	15.2

Appendix I - Table A12: Predator detections (per 100 tap nights) captured per camera on the Lekgaba (lions present) section of Tswalu Kalahari Reserve from June 2017 to May 2018.

	Acinonyx jubatus	*Lupulella mesomelas*	*Caracal caracal*	*Cynictis penicillata*	*Felis silvestris lybica*	*Genetta genetta*	*Hyaena brunnea*	*Ictonyx striatus*	*Lycaon pictus*	*Otocyon megalotis*	*Panthera leo*	*Panthera pardus*	*Suricata suricatta*	*Vulpes chama*	n	X̄	SD
Camera 1	0.0	6.7	0.0	0.0	6.7	0.0	6.7	0.0	0.0	66.7	0.0	0.0	0.0	0.0	86.7	6.2	17.6
Camera 2	0.0	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	73.3	0.0	0.0	0.0	86.7	6.2	19.6
Camera 3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 4	0.0	153.3	0.0	0.0	0.0	0.0	26.7	0.0	0.0	0.0	13.3	0.0	0.0	6.7	200.0	14.3	40.7
Camera 5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 6	0.0	26.7	0.0	0.0	13.3	0.0	6.7	0.0	0.0	0.0	13.3	0.0	0.0	0.0	60.0	4.3	8.1
Camera 7	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.5	1.8
Camera 8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 9	0.0	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	53.3	0.0	0.0	0.0	73.3	5.2	14.8
Camera 10	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	33.3	0.0	0.0	13.3	40.0	93.3	6.7	13.3
Camera 11	0.0	13.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	6.7	33.3	2.4	5.0
Camera 12	0.0	13.3	0.0	13.3	26.7	0.0	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	60.0	4.3	8.1
Camera 13	0.0	73.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	6.7	0.0	0.0	86.7	6.2	19.5
Camera 14	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	6.7	13.3	1.0	2.4
Camera 15	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.5	1.8
Camera 16	0.0	433.3	0.0	26.7	20.0	6.7	53.3	0.0	0.0	6.7	333.3	0.0	0.0	13.3	893.3	63.8	137.6
Camera 17	0.0	6.7	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	1.0	2.4
Camera 18	0.0	26.7	0.0	0.0	13.3	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	46.7	3.3	7.7
Camera 19	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 20	0.0	13.3	0.0	0.0	0.0	0.0	6.7	6.7	0.0	0.0	0.0	0.0	0.0	0.0	26.7	1.9	4.1
Camera 21	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 22	0.0	26.7	0.0	0.0	13.3	0.0	13.3	0.0	0.0	0.0	0.0	0.0	253.3	6.7	313.3	22.4	67.0
Camera 23	0.0	26.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	26.7	1.9	7.1
Camera 24	0.0	20.0	0.0	0.0	0.0	0.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	40.0	2.9	7.3
Camera 25	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 26	0.0	0.0	0.0	0.0	13.3	0.0	0.0	0.0	0.0	26.7	0.0	0.0	0.0	0.0	40.0	2.9	7.7
Camera 27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 29	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 30	0.0	20.0	6.7	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	33.3	2.4	5.6

Appendix I - Table A13: Predator species diversity index per camera captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Korannaberg[1]		Lekgaba[2]	
	Detections	Proportional Abundance	Detections	Proportional Abundance
Camera 1	4	0.05	13	0.13
Camera 2	4	0.05	13	0.13
Camera 3	5	0.06	0	0.00
Camera 4	11	0.11	30	0.22
Camera 5	90	0.35	0	0.00
Camera 6	0	0.00	9	0.10
Camera 7	4	0.05	1	0.02
Camera 8	1	0.02	0	0.00
Camera 9	2	0.03	11	0.11
Camera 10	12	0.12	14	0.13
Camera 11	3	0.04	5	0.06
Camera 12	24	0.19	9	0.10
Camera 13	1	0.02	13	0.13
Camera 14	2	0.03	2	0.03
Camera 15	1	0.02	1	0.02
Camera 16	2	0.03	134	0.37
Camera 17	3	0.04	2	0.03
Camera 18	3	0.04	7	0.08
Camera 19	7	0.08	0	0.00
Camera 20	82	0.35	4	0.05
Camera 21	1	0.02	0	0.00
Camera 22	0	0.00	47	0.28
Camera 23	7	0.08	4	0.05
Camera 24	10	0.11	6	0.07
Camera 25	11	0.11	0	0.00
Camera 26	8	0.09	6	0.07
Camera 27	0	0.00	0	0.00
Camera 28	0	0.00	0	0.00
Camera 29	17	0.15	0	0.00
Camera 30	16	0.15	5	0.06

[1] Lions Absent; [2] Lions Present

Appendix I - Table A14: Seasonal detections of predators captured on Tswalu Kalahari Reserve from June 2017 to May 2018.

	Cold-Dry			Hot-Dry			Hot-Wet		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Large Predator									
Acinonyx jubatus	1	0.0	0.1	1	0.0	0.1	0	0.0	0.0
Hyaena brunnea	19	0.3	1.2	8	0.1	0.4	5	0.1	0.3
Lycaon pictus	0	0.0	0.0	0	0.0	0.0	9	0.2	1.2
Panthera leo	62	1.0	6.6	3	0.1	-0.3	8	0.1	1.0
Panthera pardus	8	0.1	1.0	0	0.0	0.0	1	0.0	0.1
Mesopredator									
Lupulella mesomelas	117	2.0	8.2	53	0.9	2.9	120	2.0	10.4
Caracal caracal	0	0.0	0.0	1	0.0	0.1	1	0.0	0.1
Felis silvestris lybica	8	0.1	0.5	16	0.3	0.8	17	0.3	1.2
Genetta genetta	2	0.0	0.2	0	0.0	0.0	0	0.0	0.0
Otocyon megalotis	20	0.3	1.4	9	0.2	1.0	21	0.4	1.2
Vulpes chama	7	0.1	0.4	6	0.1	0.4	7	0.1	0.6
Small Predator									
Cynictis penicillate	10	0.2	0.7	6	0.1	0.4	70	1.2	8.0
Ictonyx striatus	2	0.0	0.2	0	0.0	0.0	1	0.0	0.1
Suricata suricatta	41	0.7	4.9	4	0.1	0.4	6	0.1	0.8

Appendix I - Table A15: Seasonal predator species diversity index captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Cold-Dry		Hot-Dry		Hot-Wet	
	Detections	Prop. Abundance	Detections	Prop. Abundance	Detections	Prop. Abundance
Camera 1	128	0.13	137	0.18	106	0.12
Camera 2	209	0.17	39	0.07	104	0.12
Camera 3	50	0.06	60	0.10	14	0.02
Camera 4	76	0.09	250	0.25	228	0.19
Camera 5	13	0.02	38	0.07	361	0.25
Camera 6	90	0.10	63	0.10	41	0.06
Camera 7	34	0.05	52	0.09	150	0.15
Camera 8	1	0.00	5	0.01	18	0.03
Camera 9	6	0.01	73	0.12	60	0.08
Camera 10	132	0.13	81	0.12	18	0.03
Camera 11	74	0.09	86	0.13	47	0.06
Camera 12	39	0.05	152	0.19	856	0.36
Camera 13	104	0.11	69	0.11	91	0.11
Camera 14	17	0.03	24	0.05	102	0.11
Camera 15	18	0.03	11	0.03	54	0.07
Camera 16	1369	0.36	77	0.12	7	0.01
Camera 17	23	0.03	37	0.07	105	0.12
Camera 18	91	0.10	54	0.09	1	0.00
Camera 19	23	0.03	20	0.04	6	0.01
Camera 20	30	0.04	7	0.02	106	0.12
Camera 21	6	0.01	0	0.00	5	0.01
Camera 22	64	0.08	10	0.03	0	0.00
Camera 23	380	0.25	20	0.04	72	0.09
Camera 24	32	0.05	20	0.04	65	0.08
Camera 25	9	0.02	38	0.07	53	0.07
Camera 26	105	0.11	258	0.26	13	0.02
Camera 27	21	0.03	0	0.00	0	0.00
Camera 28	0	0.00	0	0.00	0	0.00
Camera 29	72	0.08	84	0.13	223	0.19
Camera 30	78	0.09	353	0.30	123	0.13

Appendix I - Table A16: Detections of ungulates captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Tswalu			Korannaberg[1]			Lekgaba[2]		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Large Ungulate									
Alcelaphus caama	1729	9.6	57.7	392	4.4	18.2	1337	14.9	79.7
Connochaetes taurinus	4362	24.2	239.2	906	10.1	28.4	3456	38.4	338.5
Equus quagga	5440	30.2	332.7	88	1.0	6.6	5352	59.5	471.3
Equus zebra hartmannae	695	3.9	38.9	92	1.0	5.4	603	6.7	55.0
Giraffa camelopardalis	859	4.8	41.7	756	8.4	58.9	103	1.1	4.6
Oryx gazella	4984	27.7	168.6	3613	40.1	234.9	1371	15.2	44.9
Taurotragus oryx	1873	10.4	66.0	1092	12.1	59.6	781	8.7	72.5
Tragelaphus strepsiceros	617	3.4	9.8	406	4.5	10.3	211	2.3	9.3
Small Ungulate									
Aepyceros melampus	742	4.1	16.7	308	3.4	15.4	434	4.8	18.1
Antidorcas marsupialis	3010	16.7	70.9	1599	17.8	85.9	1411	15.7	52.7
Phacochoerus africanus	1632	9.1	66.2	1449	16.1	93.0	183	2.0	10.9
Raphicerus campestris	103	0.6	1.6	38	0.4	1.2	65	0.7	1.8
Sylvicapra grimmia	235	1.3	5.2	104	1.2	5.9	131	1.5	4.3

[1] Lions Absent; [2] Lions Present

Appendix I - Table A17: Shapiro-Wilk statistical results for ungulates captured on camera traps on Tswalu Kalahari Reserve between June 2017 and May 2018.

	W	p*
Large Ungulates		
Alcelaphus caama	1.0	0.7
Connochaetes taurinus	1.0	0.9
Equus quagga	1.0	0.2
Equus zebra hartmannae	1.0	0.6
Giraffa camelopardalis	1.0	0.7
Oryx gazella	1.0	0.7
Taurotragus oryx	1.0	0.9
Tragelaphus strepsiceros	1.0	0.7
Small Ungulates		
Aepyceros melampus	1.0	0.9
Antidorcas marsupialis	1.0	0.7
Phacochoerus africanus	1.0	0.1
Raphicerus campestris	1.0	0.3
Sylvicapra grimmia	1.0	1.0

*(All values above 0.05 are normally distributed)

Appendix I - Table A18: Ungulate detections (per 100-nights) index captured on Tswalu Kalahari Reserve from June 2017 to May 2018.

	Tswalu			Korannaberg[1]			Lekgaba[2]		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Large Ungulate									
Alcelaphus caama	11526.7	64.0	384.9	2613.3	29.0	121.3	8913.3	99.0	531.4
Connochaetes taurinus	29080.0	161.6	1595.0	6040.0	67.1	189.2	23040.0	256.0	2256.4
Equus quagga	36266.7	201.5	2217.9	586.7	6.5	43.8	35680.0	396.4	3141.7
Equus zebra hartmannae	4633.3	25.7	259.6	613.3	6.8	35.9	4020.0	44.7	366.4
Giraffa camelopardalis	5726.7	31.8	278.1	5040.0	56.0	392.9	686.7	7.6	30.4
Oryx gazella	33226.7	184.6	1123.9	24086.7	267.6	1565.7	9140.0	101.6	299.1
Taurotragus oryx	12486.7	69.4	440.1	7280.0	80.9	397.2	5206.7	57.9	483.4
Tragelaphus strepsiceros	4113.3	22.9	65.5	2706.7	30.1	68.8	1406.7	15.6	61.9
Small Ungulate									
Aepyceros melampus	4946.7	27.5	111.5	2053.3	22.8	102.8	2893.3	32.1	120.6
Antidorcas marsupialis	20066.7	111.5	472.4	10660.0	118.4	572.7	9406.7	104.5	351.0
Phacochoerus africanus	10880.0	60.4	441.6	9660.0	107.3	620.2	1220.0	13.6	72.4
Raphicerus campestris	686.7	3.8	10.4	253.3	2.8	8.1	433.3	4.8	12.2
Sylvicapra grimmia	1566.7	8.7	34.5	693.3	7.7	39.6	873.3	9.7	28.9

[1] Lions Absent; [2] Lions Present

Appendix I - Table A19: Detections of ungulates per camera trap on the Korannaberg (lions absent) section of Tswalu Kalahari reserve between June 2017 and May 2018.

	Aepyceros melampus	*Alcelaphus caama*	*Antidorcas marsupialis*	*Connochaetes taurinus*	*Equus quagga*	*Equus zebra hartmannae*	*Giraffa camelopardalis*	*Oryx gazella*	*Phacochoerus africanus*	*Raphicerus campestris*	*Sylviacapra grimmia*	*Taurotragus oryx*	*Tragelaphus strepsiceros*
Camera 1	80	0	27	35	0	0	0	109	67	0	0	0	23
Camera 2	0	17	7	131	2	0	0	35	8	2	1	21	12
Camera 3	0	14	45	92	0	0	2	46	0	1	1	1	24
Camera 4	0	5	763	14	0	0	8	85	0	5	3	7	7
Camera 5	0	18	12	150	1	44	66	47	2	1	0	33	28
Camera 6	4	9	14	11	0	0	18	12	0	0	0	0	27
Camera 7	0	98	324	13	0	0	1	26	0	0	0	4	8
Camera 8	0	0	0	0	0	0	9	5	1	0	1	463	0
Camera 9	58	0	0	3	0	1	0	3	2	1	1	0	71
Camera 10	0	0	7	1	0	0	5	14	3	0	1	0	5
Camera 11	0	5	38	20	0	0	0	5	3	0	2	8	20
Camera 12	0	74	8	137	55	00	557	2205	502	2	2	1	9
Camera 13	0	0	0	10	0	0	0	22	1	4	1	1	11
Camera 14	0	0	0	6	0	0	7	40	8	1	0	31	0
Camera 15	0	0	0	3	0	0	6	74	1	4	0	3	0
Camera 16	0	1	0	47	0	0	0	3	1	3	0	6	0
Camera 17	0	0	0	0	0	0	0	0	1	0	0	0	0
Camera 18	0	6	1	30	0	0	1	32	12	1	0	59	0
Camera 19	0	5	8	1	0	0	0	9	0	1	1	5	2
Camera 20	0	1	7	2	0	0	0	23	2	1	37	45	2
Camera 21	0	0	0	0	0	0	0	2	0	1	0	2	0
Camera 22	0	0	0	2	0	0	0	5	0	0	0	0	3
Camera 23	90	132	81	100	30	11	27	401	737	0	0	326	60
Camera 24	0	0	3	22	0	0	6	35	0	0	0	39	2
Camera 25	11	0	16	62	0	13	3	17	1	0	0	0	14
Camera 26	0	0	146	4	0	2	0	119	0	0	0	30	28
Camera 27	0	4	3	0	0	0	0	1	0	0	0	2	0
Camera 28	0	0	0	0	0	0	0	0	0	0	0	0	0
Camera 29	65	0	88	10	0	0	39	78	96	1	43	2	27
Camera 30	0	3	1	0	0	21	1	160	1	9	10	3	23

Appendix I - Table A20: Detections of ungulates per camera trap on the Lekgaba (lions present) section of Tswalu Kalahari reserve between June 2017 and May 2018.

	Aepyceros melampus	*Alcelaphus caama*	*Antidorcas marsupialis*	*Connochaetes taurinus*	*Equus quagga*	*Equus zebra hartmannae*	*Giraffa camelopardalis*	*Oryx gazella*	*Phacochoerus africanus*	*Raphicerus campestris*	*Sylviacapra grimmia*	*Taurotragus oryx*	*Tragelaphus strepsiceros*
Camera 1	110	0	0	0	0	12	0	4	0	6	21	0	19
Camera 2	0	60	0	10	134	12	0	386	0	4	2	4	18
Camera 3	3	0	0	0	0	15	0	3	0	1	0	0	6
Camera 4	162	43	56	100	445	26	28	11	41	0	1	16	3
Camera 5	17	0	0	0	0	0	0	3	0	11	6	0	4
Camera 6	2	1	26	13	2	0	0	15	0	0	0	0	0
Camera 7	3	0	0	0	0	0	0	1	0	1	0	0	0
Camera 8	0	0	0	0	0	0	0	0	19	0	1	0	0
Camera 9	0	10	0	0	0	0	0	4	0	0	21	0	10
Camera 10	0	395	58	24	28	0	13	14	0	0	21	0	17
Camera 11	2	7	14	1	75	0	1	33	0	7	7	0	78
Camera 12	0	0	309	10	28	0	4	152	0	0	0	3	0
Camera 13	59	127	27	12	51	17	0	19	5	11	6	4	0
Camera 14	0	2	19	0	46	0	14	42	0	0	0	2	0
Camera 15	0	1	6	0	1	0	1	22	0	1	1	1	9
Camera 16	68	648	238	3213	4456	521	0	294	92	7	23	687	34
Camera 17	1	18	0	10	6	0	0	18	0	3	0	0	0
Camera 18	2	1	38	26	5	0	11	23	0	1	1	0	0
Camera 19	0	0	0	0	4	0	0	4	0	1	2	0	0
Camera 20	0	0	0	0	0.0	0	0	0	0	2	0	0	0
Camera 21	0	0	0	0	0	0	0	0	0	1	4	0	0
Camera 22	2	0	1	14	43	0	0	14	2	0	1	0	3
Camera 23	0	0	0	0	0	0	0	1	0	0	8	0	0
Camera 24	0	0	0	0	3	0	0	1	0	6	0	0	4
Camera 25	0	3	0	0	0	0.0	0	2	8	0	0	0	0
Camera 26	3	0	128	1	13	0	27	80	0	0	3	45	0
Camera 27	0	0	0	0	0	0	0	0	0	0	0	0	0
Camera 28	0	0	0	0	0	0	0	0	0	0	0	0	00
Camera 29	0	0	0	0	0	0	0	1	16	0	1	0	0
Camera 30	0	21	491	22	12	0	4	119	0	1	1	19	6

Appendix I - Table A21: Trap rates of ungulate detections (per 100 tap nights) captured per camera on Korannaberg (lions absent) of Tswalu Kalahari Reserve from June 2017 to May 2018.

	Aepyceros melampus	*Alcelaphus caama*	*Antidorcas marsupialis*	*Connochaetes taurinus*	*Equus quagga*	*Equus zebra hartmannae*	*Giraffa camelopardalis*	*Oryx gazella*	*Phacochoerus africanus*	*Raphicerus campestris*	*Sylviacapra grimmia*	*Taurotragus oryx*	*Tragelaphus strepsiceros*
Camera 1	533.3	0.0	180.0	233.3	0.0	0.0	0.0	726.7	446.7	0.0	0.0	0.0	153.3
Camera 2	0.0	113.3	46.7	873.3	13.3	0.0	0.0	233.3	53.3	13.3	6.7	140.0	80.0
Camera 3	0.0	93.3	300.0	613.3	0.0	0.0	13.3	306.7	0.0	6.7	6.7	6.7	160.0
Camera 4	0.0	33.3	5086.7	93.3	0.0	0.0	53.3	566.7	0.0	33.3	20.0	46.7	46.7
Camera 5	0.0	120.0	80.0	1000.0	6.7	293.3	440.0	313.3	13.3	6.7	0.0	220.0	186.7
Camera 6	26.7	60.0	93.3	73.3	0.0	0.0	120.0	80.0	0.0	0.0	0.0	0.0	180.0
Camera 7	0.0	653.3	2160.0	86.7	0.0	0.0	6.7	173.3	0.0	0.0	0.0	26.7	53.3
Camera 8	0.0	0.0	0.0	0.0	0.0	0.0	60.0	33.3	6.7	0.0	6.7	3086.7	0.0
Camera 9	386.7	0.0	0.0	20.0	0.0	6.7	0.0	20.0	13.3	6.7	6.7	0.0	473.3
Camera 10	0.0	0.0	46.7	6.7	0.0	0.0	33.3	93.3	20.0	0.0	6.7	0.0	33.3
Camera 11	0.0	33.3	253.3	133.3	0.0	0.0	0.0	33.3	20.0	0.0	13.3	53.3	133.3
Camera 12	0.0	493.3	53.3	913.3	366.7	0.0	3713.3	14700.0	3346.7	13.3	13.3	6.7	60.0
Camera 13	0.0	0.0	0.0	66.7	0.0	0.0	0.0	146.7	6.7	26.7	6.7	6.7	73.3
Camera 14	0.0	0.0	0.0	40.0	0.0	0.0	46.7	266.7	53.3	6.7	0.0	206.7	0.0
Camera 15	0.0	0.0	0.0	20.0	0.0	0.0	40.0	493.3	6.7	26.7	0.0	20.0	0.0
Camera 16	0.0	6.7	0.0	313.3	0.0	0.0	0.0	20.0	6.7	20.0	0.0	40.0	0.0
Camera 17	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	0.0
Camera 18	0.0	40.0	6.7	200.0	0.0	0.0	6.7	213.3	80.0	6.7	0.0	393.3	0.0
Camera 19	0.0	33.3	53.3	6.7	0.0	0.0	0.0	60.0	0.0	6.7	6.7	33.3	13.3
Camera 20	0.0	6.7	46.7	13.3	0.0	0.0	0.0	153.3	13.3	6.7	246.7	300.0	13.3
Camera 21	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	0.0	6.7	0.0	13.3	0.0
Camera 22	0.0	0.0	0.0	13.3	0.0	0.0	0.0	33.3	0.0	0.0	0.0	0.0	20.0
Camera 23	600.0	880.0	540.0	666.7	200.0	73.3	180.0	2673.3	4913.3	0.0	0.0	2173.3	400.0
Camera 24	0.0	0.0	20.0	146.7	0.0	0.0	40.0	233.3	0.0	0.0	0.0	260.0	13.3
Camera 25	73.3	0.0	106.7	413.3	0.0	86.7	20.0	113.3	6.7	0.0	0.0	0.0	93.3
Camera 26	0.0	0.0	973.3	26.7	0.0	13.3	0.0	793.3	0.0	0.0	0.0	200.0	186.7
Camera 27	0.0	26.7	20.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	0.0	13.3	0.0
Camera 28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 29	433.3	0.0	586.7	66.7	0.0	0.0	260.0	520.0	640.0	6.7	286.7	13.3	180.0
Camera 30	0.0	20.0	6.7	0.0	0.0	140.0	6.7	1066.7	6.7	60.0	66.7	20.0	153.3

Appendix I - Table A22: Trap rates of ungulate detections (per 100 tap nights) captured per camera on the Lekgaba (lions present) section of Tswalu Kalahari Reserve from June 2017 to May 2018.

	Aepyceros melampus	*Alcelaphus caama*	*Antidorcas marsupialis*	*Connochaetes taurinus*	*Equus quagga*	*Equus zebra hartmannae*	*Giraffa camelopardalis*	*Oryx gazella*	*Phacochoerus africanus*	*Raphicerus campestris*	*Sylviacapra grimmia*	*Taurotragus oryx*	*Tragelaphus strepsiceros*
Camera 1	733.3	0.0	0.0	0.0	0.0	80.0	0.0	26.7	0.0	40.0	140.0	0.0	126.7
Camera 2	0.0	400.0	0.0	66.7	893.3	80.0	0.0	2573.3	0.0	26.7	13.3	26.7	120.0
Camera 3	20.0	0.0	0.0	0.0	0.0	100.0	0.0	20.0	0.0	6.7	0.0	0.0	40.0
Camera 4	1080.0	286.7	373.3	666.7	2966.7	173.3	186.7	773.3	273.3	0.0	6.7	106.7	20.0
Camera 5	113.3	0.0	0.0	0.0	0.0	0.0	0.0	20.0	0.0	73.3	40.0	0.0	26.7
Camera 6	13.3	6.7	173.3	86.7	13.3	0.0	0.0	100.0	0.0	0.0	0.0	0.0	0.0
Camera 7	20.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	6.7	0.0	0.0	0.0
Camera 8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	126.7	0.0	6.7	0.0	0.0
Camera 9	0.0	66.7	0.0	0.0	0.0	0.0	0.0	26.7	0.0	0.0	140.0	0.0	66.7
Camera 10	0.0	2633.3	386.7	160.0	186.7	0.0	86.7	93.3	0.0	0.0	140.0	0.0	113.3
Camera 11	13.3	46.7	93.3	6.7	500.0	0.0	6.7	220.0	0.0	46.7	46.7	0.0	520.0
Camera 12	0.0	0.0	2060.0	66.7	186.7	0.0	26.7	1013.3	0.0	0.0	0.0	20.0	0.0
Camera 13	393.3	846.7	180.0	80.0	340.0	113.3	0.0	126.7	33.3	73.3	40.0	26.7	0.0
Camera 14	0.0	13.3	126.7	0.0	306.7	0.0	93.3	280.0	0.0	0.0	0.0	13.3	0.0
Camera 15	0.0	6.7	40.0	0.0	6.7	0.0	6.7	146.7	0.0	6.7	6.7	6.7	60.0
Camera 16	453.3	4320.0	1586.7	21420.0	29706.7	3473.3	0.0	1960.0	613.3	46.7	153.3	4580.0	226.7
Camera 17	6.7	120.0	0.0	66.7	40.0	0.0	0.0	120.0	0.0	20.0	0.0	0.0	0.0
Camera 18	13.3	6.7	253.3	173.3	33.3	0.0	73.3	153.3	0.0	6.7	6.7	0.0	0.0
Camera 19	0.0	0.0	0.0	0.0	26.7	0.0	0.0	26.7	0.0	6.7	13.3	0.0	0.0
Camera 20	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.3	0.0	0.0	0.0
Camera 21	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	26.7	0.0	0.0
Camera 22	13.3	0.0	6.7	93.3	286.7	0.0	0.0	93.3	13.3	0.0	6.7	0.0	20.0
Camera 23	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	0.0	0.0	53.3	0.0	0.0
Camera 24	0.0	0.0	0.0	0.0	20.0	0.0	0.0	6.7	0.0	40.0	0.0	0.0	26.7
Camera 25	0.0	20.0	0.0	0.0	0.0	0.0	0.0	13.3	53.3	0.0	0.0	0.0	0.0
Camera 26	20.0	0.0	853.3	6.7	86.7	0.0	180.0	533.3	0.0	0.0	20.0	300.0	0.0
Camera 27	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 28	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Camera 29	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.7	106.7	0.0	6.7	0.0	0.0
Camera 30	0.0	140.0	3273.3	146.7	80.0	0.0	26.7	793.3	0.0	6.7	6.7	126.7	40.0

Appendix I - Table A23: Descriptive statistics of ungulate detections per camera, for Korannaberg and Lekgaba, captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Korannaberg[1]			Lekgaba[2]		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Camera 1	341	26.2	36.9	172	14.3	31.1
Camera 2	236	18.2	35.5	630	52.5	112.0
Camera 3	226.0	17.4	28.1	28	2.3	4.4
Camera 4	897	69.0	209.7	981	81.8	125.3
Camera 5	402	30.9	41.8	41	3.4	5.5
Camera 6	95	7.3	8.7	33	2.8	5.3
Camera 7	474	36.5	90.5	5	0.4	0.9
Camera 8	479	36.8	128.1	20	1.7	5.5
Camera 9	140	10.8	24.0	45	3.8	6.7
Camera 10	36	2.8	4.1	512	42.7	111.5
Camera 11	101	7.8	11.5	211	17.6	29.0
Camera 12	3552	273.2	611.1	197	16.4	43.5
Camera 13	50	3.8	6.7	311	25.9	37.1
Camera 14	93	7.2	13.1	106	8.8	16.9
Camera 15	91	7.0	20.2	37	3.1	6.4
Camera 16	61	4.7	12.8	10043	836.9	1447.2
Camera 17	1	0.1	0.3	56	4.7	7.0
Camera 18	142	10.9	18.3	70	5.8	9.3
Camera 19	32	2.5	3.2	11	0.9	1.6
Camera 20	120	9.2	15.5	2	0.2	0.6
Camera 21	5	0.4	0.8	5	0.4	1.2
Camera 22	10	0.8	1.6	79	6.6	12.6
Camera 23	1995	153.5	214.4	9	0.8	2.3
Camera 24	107	8.2	14.1	14	1.2	2.0
Camera 25	137	10.5	16.9	13	1.1	2.4
Camera 26	329	25.3	49.0	172	14.3	25.0
Camera 27	10	0.8	1.4	0	0.0	0.0
Camera 28	0	0.0	0.0	0	0.0	0.0
Camera 29	449	34.5	36.5	18	1.5	4.6
Camera 30	232	17.8	43.4	205	17.1	33.2

[1] Lions Absent; [2] Lions Present

Appendix I - Table A24: Descriptive statistics of ungulate detection trap rates, for Korannaberg and Lekgaba, captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Korannaberg[1]			Lekgaba[2]		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Camera 1	2273.3	174.9	245.8	1146.7	88.2	200.2
Camera 2	1573.3	121.0	236.5	4200.0	323.1	721.7
Camera 3	1506.7	115.9	187.1	186.7	14.4	28.5
Camera 4	5980.0	460.0	1398.3	6913.3	531.8	801.4
Camera 5	2680.0	206.2	278.3	273.3	21.0	35.6
Camera 6	633.3	48.7	58.3	393.3	30.3	54.8
Camera 7	3160.0	243.1	603.1	33.3	2.6	5.8
Camera 8	3193.3	245.6	853.8	133.3	10.3	35.0
Camera 9	933.3	71.8	160.1	300.0	23.1	43.1
Camera 10	240.0	18.5	27.6	3800.0	292.3	711.9
Camera 11	673.3	51.8	76.6	1500.0	115.4	185.1
Camera 12	23680.0	1821.5	4073.9	3373.3	259.5	608.0
Camera 13	333.3	25.6	44.4	2253.3	173.3	236.8
Camera 14	620.0	47.7	87.0	833.3	64.1	109.6
Camera 15	606.7	46.7	134.9	286.7	22.1	41.5
Camera 16	406.7	31.3	85.6	68540.0	5272.3	9303.5
Camera 17	6.7	0.5	1.8	373.3	28.7	45.2
Camera 18	946.7	72.8	122.3	720.0	55.4	84.0
Camera 19	213.3	16.4	21.4	73.3	5.6	10.1
Camera 20	800.0	61.5	103.2	13.3	1.0	3.7
Camera 21	33.3	2.6	5.1	33.3	2.6	7.5
Camera 22	66.7	5.1	10.6	533.3	41.0	80.9
Camera 23	13300.0	1023.1	1429.2	60.0	4.6	14.8
Camera 24	713.3	54.9	94.2	93.3	7.2	13.2
Camera 25	913.3	70.3	112.9	86.7	6.7	15.4
Camera 26	2193.3	168.7	326.9	2000.0	153.8	263.9
Camera 27	66.7	5.1	9.1	0.0	0.0	0.0
Camera 28	0.0	0.0	0.0	0.0	0.0	0.0
Camera 29	2993.3	230.3	243.2	120.0	9.2	29.4
Camera 30	1546.7	119.0	289.5	4640.0	356.9	901.6

[1]Lions Absent; [2]Lions Present

Appendix I - Table A25: Seasonal camera trap detections of ungulates captured on Tswalu Kalahari Reserve from June 2017 to May 2018.

	Cold-Dry			Hot-Dry			Hot-Wet		
	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Large Ungulate									
Alcelaphus caama	1353	22.6	98.3	216	3.6	12.8	160	2.7	9.6
Connochaetes taurinus	3516	58.6	414.5	474	7.9	19.8	372	6.2	24.2
Equus quagga	4660	77.7	574.9	540	9.0	57.5	240	4.0	18.8
Equus zebra hartmannae	580	9.7	67.2	56	0.9	4.2	59	1.0	5.9
Giraffa camelopardalis	103	1.7	5.5	67	1.1	4.1	689	11.5	72.1
Oryx gazella	1376	22.9	70.9	779	13.0	30.6	2829	47.2	283.1
Taurotragus oryx	1141	19.0	97.4	174	2.9	9.3	558	9.3	59.8
Tragelaphus strepsiceros	185	3.1	7.6	262	4.4	14.1	170	2.8	6.1
Small Ungulate									
Aepyceros melampus	217	3.6	15.5	330	5.5	20.1	195	3.3	14.4
Antidorcas marsupialis	568	9.5	33.8	1077	18.0	55.4	1365	22.8	104.9
Phacochoerus africanus	887	14.8	95.6	141	2.4	10.3	604	10.1	63.7
Raphicerus campestris	26	0.4	1.0	29	0.5	1.3	48	0.8	2.1
Sylvicapra grimmia	51	0.9	3.6	44	0.7	3.0	140	2.3	7.6

Appendix I - Table A26: GPS co-ordinates of small mammal transect locations set up for this study on Tswalu Kalahari Reserve between June 2017 and May 2018.

Korannaberg[1]		Lekgaba[2]	
Location	Co-ordinates	Location	Co-ordinates
1	-27.202055, 22.362732	1	-27.203887, 22.394575
2	-27.212131, 22.347197	2	-27.202666, 22.417578
3	-27.222817, 22.361616	3	-27.213963, 22.383246
4	-27.232281, 22.373289	4	-27.212742, 22.407278
5	-27.248154, 22.338957	5	-27.223733, 22.383589
6	-27.248154, 22.351660	6	-27.221291, 22.410025
7	-27.257615, 22.371229	7	-27.219764, 22.424788
8	-27.265855, 22.350287	8	-27.235944, 22.416205
9	-27.279892, 22.351317	9	-27.235944, 22.428908
10	-27.278062, 22.389425	10	-27.253953, 22.429938
11	-27.296393, 22.354063	11	-27.269822, 22.405562
12	-27.313147, 22.370199	12	-27.282944, 22.405905

[1]Lions Absent; [2]Lions Present

Appendix I – Table A27: Captures values, minimum know alive (MKA) values, and estimated sample size (estimated using MARK version 9.0 – Black-bellied Whistling Ducks) values, per transect, of small mammals trapped on Tswalu Kalahari Reserve between June 2017 and May 2018.

		Tswalu				Korannaberg[1]				Lekgaba[2]			
Species	Common Name	MKA	$\bar{X}$	SD	ESS	MKA	$\bar{X}$	SD	ESS	MKA	$\bar{X}$	SD	ESS
Crocidura hirta	Lesser Red Musk Shrew	1	0.0	0.1	0.1	0	0.0	0.0	0.0	1	0.0	0.2	0.1
Dendromus melanotis	Gray Climbing Mouse	7	0.1	0.3	0.6	3	0.1	0.4	0.3	4	0.1	0.3	0.3
Desmodillus auricularis	Cape Short-eared Gerbil	10	0.1	0.5	0.7	4	0.1	0.3	0.4	6	0.2	0.7	0.3
Elephantulus intufi	Bushveld Elephant Shrew	28	0.4	1.6	1.9	8	0.2	0.6	0.6	20	0.6	2.2	1.3
Elephantulus rupestris	Western Rock Elephant Shrew	4	0.1	0.3	0.3	0	0.0	0.0	0.0	4	0.1	0.4	0.3
Gerbilliscus brantsii	Highveld Gerbil	169	2.3	4.0	11.3	85	2.4	5.0	5.0	84	2.3	2.7	6.3
Gerbilliscus leucogaster	Bushveld Gerbil	409	5.7	6.4	28.3	187	5.2	6.8	14.5	222	6.2	6.2	13.8
Gerbillurus paeba	Pygmy Hairy-foooted Gerbil	526	7.3	6.6	37.8	300	8.3	6.0	21.5	226	6.3	7.1	16.3
Graphiurus murinus	Woodland Dormouse	1	0.0	0.1	0.1	1	0.0	0.2	0.1	0	0.0	0.0	0.0
Micaelamys namaquensis	Namaqua Rock Rat	29	0.4	2.0	1.8	0	0.0	0.0	0.0	29	0.8	2.8	1.8
Mus indutus	Desert Pygmy Mouse	25	0.3	0.8	1.7	5	0.1	0.5	0.4	20	0.6	0.9	1.3
Rhabdomys bechuanae	West-central South-African Four-striped Grass Rat	53	0.7	1.6	3.7	15	0.4	0.8	0.9	38	1.1	2.1	2.8
Saccostomus campestris	Pouched Mouse	58	0.8	2.1	4.1	7	0.2	0.5	0.6	51	1.4	2.8	3.5
Suncus varilla	Lesser Dwarf Shrew	1	0.0	0.1	0.1	0	0.0	0.0	0.0	1	0.0	0.2	0.1
Thallomys nigricauda	Black-tailed Tree Rat	1	0.0	0.1	0.1	0	0.0	0.0	0.0	1	0.0	0.2	0.1

[1] Lions Absent; [2] Lions Present

Appendix I - Table A28: Total captures of small mammals per transect captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Transects																							
	Korannaberg[1]												Lekgaba[2]											
Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
C. hirta	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
D. melanotis	0	0	0	0	0	0	2	0	0	1	0	0	1	0	1	1	0	0	1	0	0	1	0	0
D. auricularis	0	0	0	1	0	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	1	0	4
E. intufi	0	0	0	0	0	0	0	1	0	1	0	6	0	3	0	0	0	0	0	0	0	1	16	0
E. rupestris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0
G. brantsii	0	0	6	3	2	1	4	41	1	0	20	6	7	4	3	8	12	22	4	0	10	3	1	10
G. leucogaster	0	22	17	23	10	24	18	25	4	10	29	9	12	13	13	27	21	41	17	4	27	9	11	27
G.paeba	8	22	6	21	38	35	35	11	24	21	34	40	12	9	13	6	68	11	19	10	11	21	26	19
G. murinus	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
M. namaquensis	0	0	0	0	0	0	0	0	0	0	0	0.0	1	0	29	0	0	0	0	0	0	0	0	0
M.indutus	0	0	0	0	0	0	4	0	1	0	0	0	0	1	1	0	3	1	0	6	1	3	1	2
R. bechuanae	0	4	3	0	2	1	3	1	0	0	1	0	19	4	0	0	0	1	7	1	0	3	2	1
S. campestris	1	1	0	0	0	1	0	0	0	1	2	1	0	0	1	10	1	3	0	2	1	2	4	27
S.varilla	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
T. nigricauda	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
n	9	49	29	48	52	63	69	80	30	35	86	63	53	34	65	52	106	79	48	23	50	45	61	90
$\bar{X}$	0.6	3.3	1.9	3.2	3.5	4.2	4.6	5.3	2.0	2.3	5.7	4.2	3.5	2.3	4.3	3.5	7.1	5.3	3.2	1.5	3.3	3.0	4.1	6.0
SD	2.1	7.7	4.0	7.7	9.9	10.5	10.3	12.0	6.2	5.8	11.7	10.3	6.1	3.9	8.1	7.3	17.9	11.6	6.3	2.9	7.5	5.5	7.7	10.0

[1] Lions Absent; [2] Lions Present

Appendix I - Table A29: Shapiro-Wilk statistical comparison between sites for small mammals captured on Tswalu Kalahari Reserve between May 2017 and June 2018.

Species	W	p*
Crocidura hirta	0.9	0.5
Dendromus melanotis	1.0	0.9
Desmodillus auricularis	1.0	0.9
Elephantulus intufi	0.9	0.5
Elephantulus rupestris	1.0	0.5
Gerbilliscus brantsii	1.0	0.4
Gerbilliscus leucogaster	1.0	0.7
Gerbillurus paeba	1.0	0.8
Graphiurus murinus	1.0	0.5
Micaelamys namaquensis	1.0	0.3
Mus indutus	1.0	0.8
Rhabdomys bechuanae	1.0	0.6
Saccostomus campestris	1.0	0.1
Suncus varilla	1.0	0.9
Thallomys nigricauda	1.0	0.4

*(All values above 0.05 are normally distributed)

Appendix I - Table A30: Seasonal captures of small mammals trapped on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Cold-Dry			Hot-Dry			Hot-Wet		
Species	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Crocidura hirta	1	0.0	0.2	0	0.0	0.0	0	0.0	0.0
Dendromus melanotis	6	0.3	0.5	1	0.0	0.2	0	0.0	0.0
Desmodillus auricularis	9	0.4	0.9	1	0.0	0.2	0	0.0	0.0
Elephantulus intufi	17	0.7	2.7	6	0.3	0.7	5	0.2	0.4
Elephantulus rupestris	1	0.0	0.2	1	0.0	0.2	2	0.1	0.4
Gerbilliscus brantsii	71	3.0	3.7	78	3.3	5.4	20	0.8	1.5
Gerbilliscus leucogaster	263	11.0	7.8	65	2.7	2.6	81	3.4	4.2
Gerbillurus paeba	286	11.9	6.4	115	4.8	5.1	125	5.2	5.9
Graphiurus murinus	0	0.0	0.0	1	0.0	0.2	0	0.0	0.0
Micaelamys namaquensis	6	0.3	1.2	11	0.5	2.2	12	0.5	2.4
Mus indutus	14	0.6	1.1	9	0.4	0.6	2	0.1	0.4
Rhabdomys bechuanae	27	1.1	2.3	11	0.5	0.8	15	0.6	1.3
Saccostomus campestris	13	0.5	1.3	18	0.8	2.6	27	1.1	2.3
Suncus varilla	0	0.0	0.0	1	0.0	0.2	0	0.0	0.0
Thallomys nigricauda	0	0.0	0.0	0	0.0	0.0	1	0.0	0.2

Appendix I - Table A31: Captures of different small mammal dietary functional groups, per transect, captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Herbivores	Insectivores	Omnivores	n	$\bar{X}$	SD
Transect 1	0	0	9	9	3.0	5.2
Transect 2	0	0	49	49	16.3	28.3
Transect 3	0	0	29	29	9.7	16.7
Transect 4	0	0	48	48	16.0	27.7
Transect 5	0	0	52	52	17.3	30.0
Transect 6	0	0	63	63	21.0	36.4
Transect 7	0	0	69	69	23.0	39.8
Transect 8	0	1	79	80	26.7	45.3
Transect 9	0	0	30	30	10.0	17.3
Transect 10	0	1	34	35	11.7	19.3
Transect 11	0	0	86	86	28.7	49.7
Transect 12	0	6	57	63	21.0	31.3
Transect 13	2	0	51	53	17.7	28.9
Transect 14	0	3	31	34	11.3	17.1
Transect 15	29	4	32	65	21.7	15.4
Transect 16	0	0	52	52	17.3	30.0
Transect 17	0	1	105	106	35.3	60.3
Transect 18	0	0	79	79	26.3	45.6
Transect 19	0	0	48	48	16.0	27.7
Transect 20	0	0	23	23	7.7	13.3
Transect 21	0	0	50	50	16.7	28.9
Transect 22	0	2	43	45	15.0	24.3
Transect 23	0	16	45	61	20.3	22.8
Transect 24	0	0	90	90	30.0	52.0

Appendix I - Table A32: Differences in morphometries of small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Korannaberg[1]				Lekgaba[2]			
	Head Length		Body Length		Head Length		Body Length	
Species	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
Desmodillus auricularis	3.4	0.8	9.8	0.5	3.4	0.2	10.2	1.2
Elephantulus intufi	4.1	0.4	10.5	1.0	4.4	0.3	11.1	0.8
Gerbilliscus brantsii	3.5	0.4	11.5	1.6	3.4	0.4	11.0	1.5
Gerbilliscus leucogaster	2.9	0.5	9.4	1.8	3.4	0.6	10.7	1.9
Gerbillurus paeba	2.7	0.3	8.6	3.9	2.8	1.7	8.5	0.7
Rhabdomys bechuanae	2.6	0.3	8.2	1.1	2.6	0.3	8.7	0.9
Saccostomus campestris	2.8	0.3	9.9	1.1	3.0	0.3	9.4	1.3

[1] Lions Absent; [2] Lions Present

Appendix I - Table A33: Shapiro-Wilk normality test on the head and body lengths of the small mammals captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Head Length		Body Length	
Species	W	p*	W	p*
Desmodillus auricularis	0.9	0.1	0.9	0.6
Elephantulus intufi	1.0	1.0	1.0	0.8
Gerbilliscus brantsii	1.0	0.5	1.0	0.2
Gerbilliscus leucogaster	1.0	0.3	1.0	0.1
Gerbillurus paeba	1.0	0.9	1.0	0.6
Rhabdomys bechuanae	1.0	0.7	1.0	0.8
Saccostomus campestris	1.0	0.5	1.0	0.3

*(All values above 0.05 are normally distributed)

Appendix I - Table A34: Seasonal differences in small mammal body morphometries captured on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Head Lengths						Body Lengths					
	Cold-Dry		Hot-dry		Hot-Wet		Cold-Dry		Hot-Dry		Hot-Wet	
Species	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD	$\bar{X}$	SD
D. auricularis	3.4	0.6	3.6	-	0.0	0.0	9.6	0.5	12.4	-	0.0	0.0
E. intufi	4.3	0.2	4.0	0.4	4.6	0.4	11.3	0.9	10.5	0.8	10.6	0.7
G. brantsii	3.5	0.4	3.5	0.4	3.6	0.4	11.5	1.4	11.0	1.8	11.2	1.0
G. leucogaster	3.0	0.5	3.3	0.5	3.7	0.4	9.8	2.0	10.5	2.1	10.8	1.5
G. paeba	2.7	1.5	2.6	0.3	2.8	0.2	8.5	0.7	8.8	6.1	8.3	0.8
R. bechuanae	2.5	0.2	2.5	0.3	2.7	0.3	8.3	0.7	8.7	1.1	8.7	1.2
S. campestris	2.9	0.2	2.9	0.4	3.0	0.3	10.0	1.0	9.6	1.7	9.4	1.2

Appendix I - Table A35: GPS co-ordinates of bird transect locations set up for this study on Tswalu Kalahari Reserve between June 2017 and May 2018.

Korannaberg[1]		Lekgaba[2]	
Location	Co-ordinates	Location	Co-ordinates
1	-27.202055, 22.342734	1	-27.192894, 22.402815
2	-27.213963, 22.351317	2	-27.209994, 22.388396
3	-27.222206, 22.345480	3	-27.213963, 22.427191
4	-27.235028, 22.371229	4	-27.231365, 22.407622
5	-27.246627, 22.364020	5	-27.253342, 22.403673
6	-27.257310, 22.389425	6	-27.271653, 22.424959
7	-27.278977, 22.368139	7	-27.281723, 22.416376
8	-27.297284, 22.387022	8	-27.289656, 22.416033

[1] Lions Absent; [2] Lions Present

Appendix I - Table A36: All avian population detections recorded on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Species		Tswalu			Korannaberg[1]			Lekgaba[2]		
Latin Name	Common Name	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Afrotis afraoides	Northern Black Korhaan	14	0.9	1.4	5	0.6	1.1	9	1.1	1.7
Alopochen aegyptiaca	Egyptian Goose	3	0.2	0.4	0	0.0	0.0	3	0.4	0.7
Ardeotis kori	Kori Bustard	2	0.1	0.2	2	0.3	0.5	0	0.0	0.0
Batis pririt	Pririt Batis	18	1.1	1.2	10	1.3	1.3	8	1.0	1.2
Bradornis infuscatus	Chat Flycatcher	8	0.5	1.1	4	0.5	1.1	4	0.5	1.1
Bradornis mariquensis	Marico Flycatcher	18	1.1	1.9	7	0.9	1.4	11	1.4	2.4
Bubo africanus	Spotted Eagle Owl	1	0.1	0.2	1	0.1	0.4	0	0.0	0.0
Calendulauda africanoides	Fawn-coloured Lark	35	2.2	2.3	16	2.0	2.2	19	2.4	2.4
Campethera abingoni	Golden-tailed Woodpecker	3	0.2	0.3	3	0.4	0.5	0	0.0	0.0
Cercomela tractrac	Tractrac Chat	1	0.1	0.2	1	0.1	0.4	0	0.0	0.0
Cinnyris fuscus	Dusky Sunbird	26	1.6	1.3	1	0.1	0.4	25	3.1	2.3
Cinnyris mariquensis	Marico Sunbird	1	0.1	0.2	1	0.1	0.4	0	0.0	0.0
Cisticola fulvicapilla	Neddicky	1	0.1	0.2	1	0.1	0.4	0	0.0	0.0
Clamator jacobinus	Jacobin Cuko	1	0.1	0.2	0	0.0	0.0	1	0.1	0.4
Colius colius	White-backed Mousebird	78	4.9	5.1	34	4.3	3.0	44	5.5	7.1
Crithagra atrogularis	Black-throated Canary	1	0.1	0.2	0	0.0	0.0	1	0.1	0.4
Crithagra flaviventris	Yellow Canary	28	1.8	2.0	11	1.4	2.8	17	2.1	1.2
Emberiza capensis	Cape Bunting	7	0.4	0.9	2	0.3	0.7	5	0.6	1.2
Emberiza impetuani	Lark-like Bunting	4	0.3	0.5	2	0.3	0.5	2	0.3	0.5
Eremomela icteropygialis	Yellow-bellied Eremomela	22	1.4	1.7	5	0.6	1.2	17	2.1	2.2
Erythropygia paena	Kalahari Scrub Robin	88	5.5	2.7	41	5.1	2.6	47	5.9	2.7
Lamprotornis nitens	Cape Starling	16	1.0	1.0	12	1.5	0.9	4	0.5	1.1
Laniarius atrococcineus	Crimson-breasted Shrike	4	0.3	0.3	4	0.5	0.5	0	0.0	0.0
Lanius collaris	Southern Fiscal	31	1.9	1.7	20	2.5	2.3	11	1.4	1.2
Lanius collurio	Red-backed Shrike	5	0.3	0.9	4	0.5	1.4	1	0.1	0.4
Lanius minor	Lesser Grey Shrike	16	1.0	0.8	10	1.3	0.7	6	0.8	0.9
Lophotis ruficrista	Red-crested Korhaan	10	0.6	1.2	4	0.5	1.1	6	0.8	1.4
Melierax canorus	Pale Chanting Goshawk	2	0.1	0.2	0	0.0	0.0	2	0.3	0.5
Merops apiaster	European Bee-eater	28	1.8	4.2	24	3.0	7.0	4	0.5	1.4
Merops hirundineus	Swallow-tailed Bee-eater	12	0.8	1.3	4	0.5	1.1	8	1.0	1.6
Micronisus gabar	Gabar Goshawk	3	0.2	0.5	3	0.4	1.1	0	0.0	0.0
Mirafra fasciolata	Eastern Clapper Lark	7	0.4	0.8	4	0.5	1.1	3	0.4	0.5
Monticola brevipes	Short-toed Rock Thrush	7	0.4	0.5	1	0.1	0.4	6	0.8	0.7
Myrmecocichla formicivora	Ant-eating Chat	51	3.2	2.7	19	2.4	2.2	32	4.0	3.3
Numida meleagris	Helmeted Guineafowl	36	2.3	4.2	36	4.5	8.4	0	0.0	0.0
Oena capensis	Namaqua Dove	27	1.7	2.1	11	1.4	2.0	16	2.0	2.2
Oenanthe monticola	Mountain Wheatear	14	0.9	1.6	0	0.0	0.0	14	1.8	3.2
Oenanthe pileata	Capped Wheatear	1	0.1	0.2	1	0.1	0.4	0	0.0	0.0
Parus cinerascens	Ashy Tit	4	0.3	0.5	2	0.3	0.5	2	0.3	0.5
Passer melanurus	Cape Sparrow	20	1.3	3.1	5	0.6	0.9	15	1.9	5.3
Philetairus socius	Sociable Weaver	417	26.1	24.8	217	27.1	26.5	200	25.0	23.1
Plocepasser mahali	White-browed Sparrow-weaver	58	3.6	2.7	48	6.0	4.0	10	1.3	1.4
Ploceus velatus	Southern-masked Weaver	9	0.6	0.6	0	0.0	0.0	9	1.1	1.1
Polihierax semitorquatus	Pygmy Falcon	3	0.2	0.4	3	0.4	0.7	0	0.0	0.0
Prinia flavicans	Black-chested Prinia	89	5.6	1.7	45	5.6	1.8	44	5.5	1.6
Pterocles bicintus	Double-banded Sangrouse	3	0.2	0.4	0	0.0	0.0	3	0.4	0.7
Pterocles burchelli	Burchell’s Sandgrouse	3	0.2	0.5	1	0.1	0.4	2	0.3	0.7

Appendix I - Table A36: All avian population detections recorded on Tswalu Kalahari Reserve between June 2017 and May 2018

Avian Species Latin Name	Common Name	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Pterocles namaqua	Namaqua Sandgrouse	131	8.2	15.6	76	9.5	17.6	55	6.9	13.6
Pycnonotus nigricans	African Red-eyed Bulbul	55	3.4	3.0	23	2.9	3.2	32	4.0	2.8
Pytilia melba	Green-winged Pytilia	3	0.2	0.5	2	0.3	0.7	1	0.1	0.4
Rhinopomastus cyanomelas	Scimitarbill	7	0.4	0.7	5	0.6	0.9	2	0.3	0.5
Sagittarius serpentarius	Secetarybird	2	0.1	0.4	2	0.3	0.7	0	0.0	0.0
Spilopelia senegalensis	Laughing Dove	24	1.5	1.9	4	0.5	1.1	20	2.5	2.8
Sporopipes squamifrons	Scaly-feathered Weaver	304	19.0	17.4	103	12.9	12.8	201	25.1	22.1
Streptopelia capicola	Ring-necked Dove	102	6.4	6.0	51	6.4	7.9	51	6.4	4.0
Sylvia subcaerulea	Chestnut-vented Warbler	86	5.4	3.9	45	5.6	3.7	41	5.1	4.1
Tchagra australis	Brown-crowned Tchagra	4	0.3	0.4	1	0.1	0.4	3	0.4	0.5
Telophorus zeylonus	Bokmakierie	12	0.8	1.3	2	0.3	0.5	10	1.3	2.1
Tockus leucomelas	Southern Yellow-billed Hornbill	10	0.6	1.1	6	0.8	1.2	4	0.5	1.1
Tockus nasutus	African Grey Hornbill	2	0.1	0.2	2	0.3	0.5	0	0.0	0.0
Tricholaema leucomelas	African Pied Barbet	14	0.9	1.1	12	1.5	1.7	2	0.3	0.5
Uraeginthus granatinus	Violet-eared Waxbill	6	0.4	0.7	0	0.0	0.0	6	0.8	1.4
Vanellus coronatus	Crowned Lapwing	8	0.5	0.8	5	0.6	1.1	3	0.4	0.5
Vidua regia	Shaft-tailed Whydah	2	0.1	0.4	0	0.0	0.0	2	0.3	0.7

[1] Lions Absent; [2] Lions Present

Appendix I - Table A37: Shapiro-Wilk normality test for the five most observed bird species on Tswalu Kalahari Reserve between June 2017 and May 2018.

Avian Groups	W	p*
Ground birds	1.0	0.4
Raptors	0.9	0.4
Song birds	1.0	0.8

*(All values above 0.05 are normally distributed)

Appendix I - Table A38: Shapiro-Wilk normality test for groups of birds sampled on Tswalu Kalahari Reserve from June 2017 to May 2018.

Avian Species	W	p*
Philetairus socius	1.0	0.9
Prinia flavicans	1.0	0.9
Pterocles namaqua	0.9	0.4
Sporopipes squamifrons	1.0	0.7
Streptopelia capicola	1.0	0.9

*(All values above 0.05 are normally distributed)

Appendix I - Table A39: All avian species observed per transect on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Transects															
	Korannaberg								Lekgaba							
Avian Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Afrotis afraoides	0	0	0	1	1	0	0	3	0	0	0	0	1	5	2	1
Alopochen aegyptiaca	0	0	0	0	0	0	0	0	2	0	0	1	0	0	0	0
Ardeotis kori	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0
Batis pririt	3	3	2	0	1	1	0	0	2	2	1	0	3	0	0	0
Bradornis infuscatus	0	0	0	1	0	0	0	3	1	0	0	0	0	0	0	3
Bradornis mariquensis	3	0	0	0	0	3	0	1	0	1	0	0	1	7	2	0
Bubo africanus	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Calendulauda africanoides	1	1	3	0	7	1	2	1	5	0	0	2	4	2	6	0
Campethera abingoni	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Cercomela tractrac	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Cinnyris fuscus	0	1	0	0	0	0	0	0	4	1	4	4	7	0	1	4
Cinnyris mariquensis	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Cisticola fulvicapilla	0	1	0	0	0	0	0	0	0	0	2	0	0	0	0	0
Clamator jacobinus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Colius colius	5	4	6	9	5	0	0	5	10	0	13	1	0	2	0	18
Crithagra atrogularis	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Crithagra flaviventris	0	0	0	8	0	1	0	2	2	2	3	0	4	1	3	2
Emberiza capensis	0	2	0	0	0	0	0	0	0	2	3	0	0	0	0	0
Emberiza impetuani	1	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0
Eremomela icteropygialis	3	2	0	0	0	0	0	0	2	5	0	6	0	1	1	2
Erythropygia paena	8	8	8	4	1	5	4	3	1	4	9	5	9	8	6	5
Lamprotornis nitens	1	2	1	3	1	0	2	2	0	3	0	0	0	0	0	1
Laniarius atrococcineus	1	1	1	0	0	0	0	1	0	0	0	0	0	0	0	0
Lanius collaris	0	0	4	4	6	1	1	4	3	2	3	1	0	1	0	1
Lanius collurio	0	4	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Lanius minor	1	1	0	2	1	2	2	1	0	1	0	0	0	1	2	2
Lophotis ruficrista	0	0	0	0	3	0	1	0	0	0	0	0	0	1	1	4
Melierax canorus	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0
Merops apiaster	0	0	0	0	20	4	0	0	0	0	0	0	4	0	0	0
Merops hirundineus	0	1	0	0	0	0	3	0	0	4	3	0	0	0	0	1
Micronisus gabar	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
Mirafra fasciolata	0	0	1	0	0	0	0	3	1	0	0	1	1	0	0	0
Monticola brevipes	0	1	0	0	0	0	0	0	2	1	1	0	1	0	0	1

Appendix I - Table A39: All avian species observed per transect on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Transects															
	Korannaberg								Lekgaba							
Avian Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Myrmecocichla formicivora	1	0	2	4	1	2	2	7	3	5	3	2	0	11	5	3
Numida meleagris	20	16	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oena capensis	4	1	5	0	1	0	0	0	2	3	0	0	0	6	4	1
Oenanthe monticola	0	0	0	0	0	0	0	0	1	9	4	0	0	0	0	0
Oenanthe pileata	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Parus cinerascens	1	0	1	0	0	0	0	0	0	0	1	0	0	0	1	0
Passer melanurus	2	0	2	0	0	1	0	0	0	0	0	0	0	15	0	0
Philetairus socius	59	40	0	29	0	66	0	23	37	1	0	1	50	33	58	20
Plocepasser mahali	1	1	13	9	5	5	6	8	1	4	0	0	0	2	2	1
Ploceus velatus	0	0	0	0	0	0	0	0	2	0	2	1	0	1	3	0
Polihierax semitorquatus	0	0	0	0	0	2	0	1	0	0	0	0	0	0	0	0
Prinia flavicans	7	3	9	4	6	5	6	5	5	5	4	7	6	8	6	3
Pterocles bicinctus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	1
Pterocles burchelli	0	0	1	0	0	0	0	0	0	0	0	0	0	0	2	0
Pterocles namaqua	0	0	0	44	1	0	0	31	0	0	0	4	6	40	5	0
Pycnonotus nigricans	3	10	1	2	0	4	3	0	9	4	6	6	1	1	2	3
Pytilia melba	0	2	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Rhinopomastus cyanomelas	1	0	0	0	0	0	2	2	0	0	1	0	0	0	1	0
Sagittarius serpentarius	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Spilopelia senegalensis	1	3	0	0	0	0	0	0	1	8	3	4	0	0	0	4
Sporopipes squamifrons	13	0	9	38	10	7	25	1	22	38	0	17	36	18	68	2
Streptopelia capicola	24	3	3	1	12	1	4	3	3	14	8	1	5	5	6	9
Sylvia subcaerulea	11	10	7	2	4	6	5	0	4	4	7	13	7	0	1	5
Tchagra australis	0	1	0	0	0	0	0	0	1	0	1	1	0	0	0	0
Telophorus zeylonus	0	1	0	0	1	0	0	0	0	1	6	1	0	0	2	0
Tockus leucomelas	2	3	0	1	0	0	0	0	0	1	0	0	0	3	0	0
Tockus nasutus	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tricholaema leucomelas	2	4	1	0	0	4	0	1	1	0	0	0	0	0	0	1
Uraeginthus granatinus	0	0	0	0	0	0	0	0	0	0	1	0	4	1	0	0
Vanellus coronatus	0	0	1	1	0	3	0	0	0	0	0	1	0	1	0	1
Vidua regia	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
n	181	133	84	170	87	125	71	113	131	128	89	80	152	175	192	99
X̄	2.8	2.1	1.3	2.7	1.4	2.0	1.1	1.8	2.0	2.0	1.4	1.3	2.4	2.7	3.0	1.5
SD	8.4	5.6	2.7	8.1	3.4	8.3	3.4	4.9	5.5	5.2	2.6	3.0	7.8	7.1	11.0	3.6

Appendix I - Table A40: Seasonal comparison of the five most observed avian species on Tswalu Kalahari Reserve between June 2017 and May 2018.

	Cold-Dry			Hot-Dry			Hot-Wet		
Avian Species	n	$\bar{X}$	SD	n	$\bar{X}$	SD	n	$\bar{X}$	SD
Philetairus socius	191	11.9	18.2	170	10.6	17.4	56	3.5	23.2
Prinia flavicans	42	2.6	0.7	26	1.6	0.5	21	1.3	0.6
Pterocles namaqua	104	6.5	11.0	24	1.5	12.1	3	0.2	0.7
Sporopipes squamifrons	148	9.3	8.2	56	3.5	5.2	100	6.3	5.6
Streptopelia capicola	53	3.3	3.1	34	2.1	2.1	15	0.9	0.5

Appendix I - Table A41: Descriptive statistics of avian population detections per transect done on Tswalu Kalahari Reserve between sunrise and sunset from June 2017 to May 2018.

	Sunrise		Sunset	
	Abundance	Diversity	Abundance	Diversity
Transect 1	94	12	87	23
Transect 2	133	31	0	0
Transect 3	38	18	46	28
Transect 4	0	0	170	22
Transect 5	54	17	33	8
Transect 6	35	12	90	17
Transect 7	29	10	42	14
Transect 8	113	24	47	22
Transect 9	0	0	84	13
Transect 10	128	28	0	0
Transect 11	61	21	28	11
Transect 12	51	16	29	9
Transect 13	102	13	50	17
Transect 14	0	0	175	25
Transect 15	30	11	162	19
Transect 16	34	16	65	19

Appendix II

Appendix II - Field sampling sheet used for small mammal trapping on Tswalu Kalahari Reserve from June 2017 to May 2018.

DATE	TIME	TRAP NUMBER	MARKING	MEASUREMENTS (CM)					IDENTIFYING FEATURES
				HEAD LENGTH	HEAD GIRTH	BODY LENGTH	TAIL LENGTH	FULL BODY LENGTH	

Appendix II - Field sampling sheet used for avian point-count trapping on Tswalu Kalahari Reserve from June 2017 to May 2018.

DATE	SR/SS	TRANSECT	SPECIES INDENTIFIED	# OF INDVLS	MALE/ FEMALE	DISTANCE FROM CENTRE	IDENTIFYING FEATURES