Investigating the decline of the Martial Eagle (*Polemaetus bellicosus*) in South Africa

Daniël Cloete

Supervisor: Dr. Arjun Amar

Percy FitzPatrick Institute of African Ornithology
DST/NRF Centre of Excellence
University of Cape Town
Rondebosch
7701
South Africa
cloda07@gmail.com
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Abstract

The Martial Eagle (*Polemaetus bellicosus)* is an African endemic that occurs over a wide range of habitats but at naturally low densities. There is concern throughout its range that it is declining and it now appears to be strongly reliant on protected areas. It is classified globally as *Near Threatened* by the IUCN with a current consultation underway to up-list it to *Vulnerable* or *Endangered*. In this project I describe and explore the decline of the species across South Africa, using data from two repeated national bird surveys - South African Bird Atlas Projects (SABAP 1: 1987-1993; SABAP 2: 2007-2012). These analyses suggest a relatively uniform decline across South Africa in Martial Eagle reporting rates of 59% over the last 20 years. Alarmingly, these declines also occurred in protected areas, including the traditional strongholds of the Kruger National Park (54% decline) and the Kalahari National Park (44% decline). Independent survey data, undertaken in the Kalahari National Park, confirmed these declines and reinforced the validity of using the two SABAP surveys to examine population change. Within protected areas, the species is still encountered five times more frequently and is six times as abundant as compared to outside protected areas. Between the biomes, the species is encountered the least and has the lowest abundance in the Grassland biome where tree density is low, and has the highest abundance in the Savanna biome where tree density is higher. Examining environmental correlates of these population changes provided some support for two hypotheses on the causes of these declines, with climate change (increases in temperature) and power line densities negatively associated with changes in reporting rates. Although this analysis also suggested support for declines being associated with avian prey declines, this was unlikely to be a major driver nationally, because I found that overall avian prey species actually increased over this time period. Within Kruger National Park changes in reporting rates were negatively associated with Elephant densities, which may be related to a reduction in nesting opportunities (large trees) for the species. These results are an initial attempt to assess the broad drivers of decline and should help focus and prioritize further detailed research to elucidate the mechanisms behind this species decline.

Chapter 1: Understanding the Martial Eagle and its potential causes of decline

1.1. Species Declines

Over the last century many species have gone extinct or have undergone dramatic declines throughout the world (Brook et al. 2008; Sodhi et al. 2009). Most of these extinctions or declines can be ascribed to the multitude of anthropogenic pressures being exerted, ranging from over-exploitation through to habitat destruction, alien species invasions and climate change (Lande 1998; Goudie 2009; Sodhi et al. 2009). In some cases where the causes for the declines were identified early enough, attempts to stabilise and recover the population could be planned and implemented. Raptors in particular offer many good examples of such cases. For example, after undergoing major declines during the mid-20th century as a result of restricted reproduction caused by the effect of DDT on their eggs and embryos, Peregrine falcons (*Falco peregrinus*) have now recovered globally as a result of reintroduction efforts and regulatory restrictions on the use of DDT and similar pesticides (Millsap et al. 1998; Elliot et al. 2005). Another example, concerns the *Gyps* vultures in the Indian subcontinent; after contamination of their carrion food supply with Diclofenac was identified as the cause of their rapid declines in the mid-90s, a ban on this drug has now resulted in a slowing of the decline (Cuthbert et al. 2011; Prakash et al. 2012). However, it is also acknowledged that further efforts to better enforce the ban is required if there is any hope to realistically reverse the declines (Cuthbert et al. 2011; Prakash et al. 2012). On the other hand, numerous species have unfortunately been driven to extinction or have undergone declines without an understanding as to the underlying factors (Didham et al. 2007). For example, attempts to understand the changes in environmental conditions (between 1980s to early 2000s) which led to the major declines of amphibians in eastern Puerto Rico, failed to identify a single factor as the primary driver (Stallard 2001). Further research is still required into the combined effect of the multitude of factors identified as potential causes to better understand the declines of these amphibians (Stallard 2001).

To manage species that have undergone declines and to provide measures to halt or reverse these declines, there is a need to identify and understand the multitude of factors that may be responsible for the decline of a species (Caughley 1994; Peery et al. 2004). In his declining population paradigm, Caughley (1994) suggests that based on a species' ecology and life history, it is plausible to generate a number of hypotheses as to what factors are likely responsible for the decline of a species. This approach was successfully employed by Amar et al. (2011) to determine if the declines in abundance of upland waders in Britain were determined by a variety of competing hypotheses. This study compared data from repeat upland surveys and tested whether population changes were associated with habitat cover, forest edge exposure, grouse moor management and crow abundance (Amar et al. 2011). This approach was used in this study to investigate the suspected decline of the Martial Eagle (*Polemaetus bellicosus*) (ME) in South Africa. As a precursor to this, the natural history of the ME will be reviewed to identify potential agents of decline from which a set of sensible hypotheses will be proposed. These competing hypotheses will then be tested using change in reporting rates of MEs between the Southern African Bird Atlas Projects (SABAP) 1 and 2 to see if they correlate with covariates associated with each hypothesis. These correlative analyses should not be seen as a definitive test of the factors driving the declines of MEs in South Africa, but rather as an initial attempt to assess the broad drivers and mechanisms of decline and to prioritize and guide further focused and detailed research (Amar et al. 2011).

1.2. Martial Eagle

1.2.1. Natural History

Range and habitat

The Martial Eagle (ME) is endemic to Africa and occurs widely south of the Sahel in a wide range of habitats (Steyn 1982; Boshoff 1997; Simmons 2005; BirdLife International 2012). Even though they are widespread, they are relatively uncommon except in some extensive protected areas e.g. Kruger National Park (Brown 1970; Boshoff 1997; Barnes 2000; Ferguson-lees and Christie 2001). MEs are mostly found in open woodland and savanna plains, the drainage line wooded areas of open scrubland and forest edges (Brown 1970; Steyn 1982; Simmons 2005). They occur rarely or are totally absent in mountainous, densely wooded and forested areas (West and Central Africa) and true desert areas as well as extensive grasslands and highly transformed landscapes (Steyn 1982; Boshoff 1997; Ferguson-lees and Christie 2001; Thiollay 2006a). They mainly occur below 1500m above sea-level but can be found up to 3000m above sea-level (Ferguson-lees and Christie 2001). In South Africa they are also found in more open country and even in sub-desert with scattered trees, wooded hillocks or artificial pylons providing nesting sites (Ferguson-lees and Christie 2001). They do not occur in Lesotho probably due to a lack of breeding sites and are only occasionally recorded in the south-west Cape (Steyn 1982).

Territories and movements

Depending on the habitat, breeding MEs maintain territories ranging from 125 km^2 to 300km^2, but up to 990 km^2 has been recorded in Hwange National Park, Zimbabwe (Brown 1970; Boshoff and Vernon 1980; Steyn 1982; Boshoff 1993; Ferguson-lees and Christie 2001). MEs maintain large territories relative to their body size but may even maintain abnormally large territories if the population density is for example lowered through persecution (Van Zyl 1992). Juveniles and immature adults will move widely outside of the areas where they were born and it is believed that they stay nomadic until establishing a territory (Brown 1970; Ferguson-lees and Christie 2001). Inter nesting distance ranges from 11 km to 35 km depending on the habitat (Steyn 1982; Tarboton and Allan 1984; Hustler and Howells 1987; Boshoff 1993; Ferguson-lees and Christie 2001). Machange et al. (2005) found that MEs in the Nama Karoo preferred not to utilize cultivated land for their territories. They further found that the density of ME territories is significantly higher in game and mixed farming areas than in areas with only small-stock (Machange et al. 2005). They also suggest that the availability of prey between different areas due to e.g. vegetation cover differences, will affect the distribution of territories e.g. less active nests in areas with higher vegetation cover (Machange et al. 2005).

Nesting and breeding

ME nests are made of sturdy sticks and can be 2 m across and 2 m deep as they get older with a 40 - 50 cm wide central depression lined with leaves (Steyn 1982; Ferguson-lees and Christie 2001). They usually build in big trees but also on transmission line pylons (Steyn 1982; Tarboton and Allan 1984; Boshoff 1993; Anderson 2000; Ferguson-lees and Christie 2001; Machange et al. 2005). Breeding on pylons and in exotic trees has allowed the ME to increase its abundance in areas where they would not have naturally occurred due to a lack of breeding trees e.g. the Karoo (Boshoff 1993; Anderson 2000; Machange et al. 2005). Nests in trees are often located in the neck of a valley in the fork of often the largest tree and are found at a height of 5-19 m, and in an extreme case 70 m above the ground (Steyn 1982; Tarboton and Allan 1984; Ferguson-lees and Christie 2001). A breeding pair often has more than one nest and they can alternate breeding attempts between the nests (Steyn 1982; Tarboton and Allan 1984; Boshoff 1993; Machange et al. 2005). Breeding tends to be annual or biennial but is known to be erratic and is influenced by factors such as prey availability and the length of the post-nest dependence period (Steyn 1982; Hustler and Howells 1987; Boshoff 1993; Ferguson-lees and Christie 2001).

A single egg (and rarely a second) is laid and has an incubation period of 47 to 53 days (Steyn 1982; Tarboton and Allan 1984; Hustler and Howells 1987; Boshoff 1993; Ferguson-lees and Christie 2001). In southern Africa clutches are usually laid between March - August with the highest proportion being laid in May and June (Steyn 1982; Tarboton and Allan 1984; Hustler and Howells 1987; Boshoff 1993; Ferguson-lees and Christie 2001). In other regions of Africa the breeding period differs markedly e.g. November - April in Senegal and January - June in Sudan depending on the local conditions (Ferguson-lees and Christie 2001). Nestling period is estimated to be between 90 to 109 days but the chick may still be fed at the nest up to 3 months after leaving and can stay with the adults for 8 - 12 months (Steyn 1982; Ferguson-lees and Christie 2001). Young reared per year range from 0.52 to 0.58 per breeding attempt depending on the country and habitat (Steyn 1982; Hustler and Howells 1987; Boshoff 1993). The transition of juvenile plumage to adult plumage is a gradual process that can take up to 6 years (Steyn 1982).

Feeding behavior and diet

MEs have exceptional eyesight and can spot prey from a distance of 6 km (Steyn 1982; Ferguson-lees and Christie 2001). They can take prey in flight and have been known to pirate prey from other raptors (Steyn 1982; Ferguson-lees 2001). Most of their prey is probably less than 5 kg but they have been known to take animals up to the size of a Common Duiker (*Sylvicapra grimmia*)(~15 – 20 kg)(Ferguson-lees and Christie 2001). They will take their prey to a safe perching spot or nest to eat unless it is too large to lift where they will eat it on the ground (Ferguson-lees and Christie 2001). They prey on a large range of smaller mammals (e.g. rabbits, hares, hyrax, mongoose, ground squirrel), birds and reptiles (mostly monitor lizards, snakes, terrapins and tortoises) (Brown 1970; Steyn 1982; Ferguson-lees and Christie 2001; Simmons 2005). Preferred bird species in southern Africa include Helmeted Guineafowl (*Numida meleagris*), Coqui (*Peliperdix coqui*) and Crested Francolins (*Dendroperdix* *sephaena*), Swainson's Spurfowl (*Pternistis swainsonii*), Ibises (family Threskiornithidae), Bustards (family Otididae) and Korhaans (family Otididae) (Steyn 1982; Tarboton and Allan 1984; Allan and Osborne 2005; Simmons 2005; Ferguson-lees and Christie 2001; Ratcliffe 2005; Thiollay 2006a). Prey species taken will differ between location and habitat depending on natural occurrence, availability and abundance of different species (Steyn 1982; Boshoff et al.1990; Boshoff 1993). Boshoff et al. (1990) found in the Western Cape that there was a general change in species composition and a noticeable shift in the avian and reptile proportion to the mammalian proportion of prey taken through a range of habitats ranging from drier scrubland in the West to wetter grassland sites of the study area in the East. MEs, especially non-territorial adults and sub-adults, are known to take domestic livestock (young sheep and young goats) but in many cases these are only taken as carrion (Brown 1970; Boshoff and Palmer 1980; Boshoff et al. 1990; Ferguson-lees and Christie 2001; Simmons 2005). They have also been known to take young farmed Ostriches (*Struthio camelus*) and chickens (Tarboton and Allan 1984; Ferguson-lees and Christie 2001).

1.2.2. Conservation Status

The IUCN Red List category status for MEs is *Near Threatened* (BirdLife International 2012). The reason for its current status is that it is estimated that its numbers have declined rapidly throughout its range over the species' last three generations (56 years) even though it still has a broad distribution (Simmons 2005; BirdLife International 2012). Since 1988 the status has been upgraded from *Lower Risk/Least Concern* to *Least Concern* in 2004 and to its current *Near Threatened* status in 2009. There is currently a consultation by Birdlife International to upgrade the species status to *Vulnerable* or *Endangered* (Taylor, M. pers. com.). In southern Africa the ME will be classified as Endangered in the current 2013 Red List update (Taylor, M. pers. com.) In southern Africa the ME is probably the most threatened of all eagle species, with the possible exception of the Bateleur Eagle (*Terathopius ecaudatus*) (Steyn 1982).

Even with marked declines in their numbers being recorded in some parts during the 1980s, the old Transvaal (now Limpopo, Mpumalanga and parts of the North West and KwaZulu-Natal) alone was at that time estimated to hold 550 breeding pairs (Steyn 1982; Tarboton and Allan 1984). The declines in the 1980s were most noticeable when comparing the abundance of MEs inside and outside of protected areas indicating that human induced pressures were at play in their decline (Kemp and Kemp 1975; Boshoff and Vernon 1980; Tarboton and Allan 1984). The current population estimate for South Africa of around 600 pairs indicate that declines in their numbers have continued since the 1980s (Barnes 2000). It is estimated that Kruger National Park and the Kalahari National Park maintain a substantial proportion of South Africa's current ME pairs (Barnes 2000). Kruger National Park alone may support 100 - 120 breeding pairs and appears to be the only park of sufficient size to provide habitat without human settlement for MEs (Steyn 1982; Tarboton and Allan 1984; Ferguson-lees and Christie 2001). In all other reserves pairs need to range onto neighboring land where humans have settled (Tarboton and Allan 1984).

By considering the ME's natural history there are certain traits that make them particularly vulnerable to human pressures (Owens and Bennett 2000; Purvis 2000). MEs are apex predators and thus more vulnerable to the cumulative effects of disturbance to species lower down the food chain (Crooks and Soulé 1999; Purvis 2000). MEs have a slow reproductive rate (maximum of 1 offspring every one to two years) so the population is not able to recover quickly if numbers have been reduced or if adult survival declines. Breeding pairs maintain large territories, keeping the population naturally at low densities and they roam over large tracts of land where they are exposed to various land uses and users (Boshoff 1993). Sub-adults and non-territorial adults roam over even larger distances and inexperienced animals are more likely to prey on poultry and livestock (Ferguson-lees and Christie 2001). In Kenya it is estimated that 55% of immature MEs don't survive to adulthood and the percentage in South Africa is estimated to probably be even higher (Ferguson-lees and Christie 2001). In areas without suitable nesting sites, a factor that is recognised in playing a role where they occur, their territories will be even further apart even if the habitat is otherwise suitable (Brandl et al. 1985; Boshoff 1997). Taking all these factors into consideration, the species may not be able to breed fast enough to counter the losses and in this case the replacement rate may become too low to maintain the population exposing the species as a whole to the threat of extinction (Ferguson-lees and Christie 2001).

1.3. Potential drivers of change for the Martial Eagle

An inevitability associated with the global growth in the human population is that it has and will lead to the transformation and degradation of natural habitat and a reduction of wildlife through a range of activities (Goudie 2009). Habitat destruction and transformation is brought about through human activities such as urbanization, agricultural crop conversion, deforestation, overgrazing and subsequent bush encroachment and/or soil erosion and alien plant invasion (Meadows and Hoffman 2002; Goudie 2009; Gillson et al. 2012). In addition to wildlife reduction through indirect pressures, direct reduction also occurs as a result of human activities such as hunting and poisoning (Goudie 2009).

The decline in Martial Eagles (ME) in South Africa and the rest of its range are believed to be associated with the growing human population and they are thought to be linked to direct or indirect human impacts such as a loss of and degradation of habitat and subsequently less natural prey, persecution by small-stock farmers, drowning in farm reservoirs, incidental poisoning and electrocution and collisions on electricity pylons (Boshoff and Vernon 1980; Boshoff 1997; Anderson 2000; BirdLife International 2012). Next, I consider and review key aspects of ME's natural history and relate them to various potential drivers of change.

1.3.1. Could changes in climate be responsible for the decline?

The warming of the earth's mean surface temperature by 0.8 °C since the early 20th century and the projected continuation of this increase in temperature is recognised to be caused by human activities, especially the burning of fossil fuels (Pachauri and Reisinger 2007). These changes in temperature and also the associated changes in precipitation patterns are predicted to lead to an increased risk of extinction for many species of birds, animals and plants (Greenwood 2007; Jetz et al. 2007; Tadross et al. 2011; Warren et al. 2013). Extinctions in the quaternary period ascribed to climate change indicate that if species are not able to respond to climate change by shifting their location fast enough or adapting to changing conditions in their current location they will be at risk of extinction (Simmons et al. 2004; Huntley et al. 2006). Shifts in ranges by species in response to climate change have already been observed e.g. more pole wards or towards higher altitudes (Huntley et al. 2006; Tadross et al. 2011). In southern Africa, range restriction and a reduction of abundance are predicted for most bird species as a result of climate change (Huntley et al. 2012).

Bird species with specialized traits in body size, nomadism and residency, migration and diet are likely to be more vulnerable to climate change (Bond 1994; Lemoine and Böhning-Gaese 2003; Simmons et al. 2004; Both et al. 2006; Huntley et al. 2006). Large bodied species may better survive periods of lower food availability but smaller, shorter generation time species, will likely over the long term adapt better (Simmons et al. 2004). Species that are dependent on protected area networks may be at risk if the extent of the network is not sufficient to compensate for future range shifts of the species (Mazaris et al. 2013).

Both et al. (2009) in a Netherlands study on the changes in phenology across four trophic levels showed that the higher trophic levels (e.g. avian predators) are less flexible to respond to climate change and are thus more vulnerable than the lower trophic levels (e.g. caterpillars and passerines) as they are slower to shift their highest food demand periods (e.g. hatching dates for raptors) to the peaks in food availability. It is suggested that this may be a result of a longer time lag for the higher trophic levels between the decision making environment (e.g. hatching dates) and the selection environment (e.g. peak food availability).

In one of the few investigations of the potential impact of climate change on a raptor species in Africa, Wichmann et al. (2003; 2004) studied the Tawny Eagle (*Aquila rapax*) in the arid savanna of the southern Kalahari; they suggested that even a slight increase in year to year precipitation variation, even with an unchanged average annual precipitation, could lead to the species' extinction.

Although it is uncertain how climate change will affect MEs in South Africa, it is important to determine if a significant impact has already occurred so that the mechanisms involved can be investigated further.

1.3.2. Could declines in prey abundance be responsible for the decline?

The growing human population and its impact on the natural environment has led to the reduction in many species either directly (e.g. hunting) or indirectly (e.g. land management) (Goudie 2009). South Africa, with a higher population density and its associated land use changes, was found to generally have a lower species diversity and density of raptors than the adjoining Botswana and Zimbabwe (Brandl et al. 1985). The same trend was also seen when comparing raptor abundance in and out of protected areas in South Africa and Botswana, with less of a contrast found in Botswana (Herremans and Herremans-tonnoeyr 2000). The constant hunting pressure for food by the human population in central west Africa led to a severe decline in large game birds species outside compared to inside of protected areas between two roadside count surveys 30 years apart (Thiollay 2006b). Different land use and associated management practices (from overgrazed farms to a protected area) have been shown to result in marked structural changes in vegetation in historically the same natural vegetation type in the Kalahari thornveld (Hudson and Bouwman 2007). It was found that a decrease in structural variation with more grazing intensity and management for grazing capacity corresponded with a decrease in bird species diversity (Hudson and Bouwman 2007). Structural changes to lower vegetation levels and bush encroachment as a result of overgrazing was also found in studies in Botswana and the southern Kalahari, South Africa, to lead to a reduction in bird species and other small animals (Herremans 1998; Herremans-tonnoeyr 2000; Seymour and Dean 2010). Drastic land transformation such as when natural areas are transformed to intensive agriculture or commercial forestry or when farming methods intensify, can be too much for species to adapt to and can lead to large declines and a threat to the species' persistence if there are not large enough areas of their natural habitat still remaining (Allan et al. 1997; Donald et al. 2001; Moreira 2004; Newton 2004; Jetz et al. 2007; Amano and Yamaura 2007; Amar et al. 2011; Hofmeyer 2012).

Large predators such as the ME, are apex predators at the top of the food chain (Crooks and Soulé 1999; Purvis 2000; Dobson et al. 2006). In a disturbed ecosystem where the richness and abundance of species are reduced, these apex predators will be directly affected as their prey base is diminished (Thiollay 2006a; Dobson et al. 2006). With a reduction in a reliable food source the predators' breeding success and chance of survival will become reduced and they will be required to move to a more suitable location to persist (Dobson et al. 2006).

1.3.3. Could power lines be responsible for the decline?

Globally, power lines (and other manmade structures) are a significant cause of death for bird species through collisions and electrocution (Bevanger 1994; 1998; Manville and Albert 2005). Not all species are equally exposed and equally susceptible to collisions: for example large terrestrial, wetland and fast-flying bird species are likely to be impacted the most (Bevanger 1998; Jenkins et al. 2010). Flocking species that fly at low altitudes and on set flight paths have further been identified as high risk for collisions and large, heavy and relatively small-winged birds are deemed to be more susceptible to collisions (Bevanger 1994, 1998; Manville and

Albert 2005; Jenkins et al. 2010). Additional factors that have been ascribed to further contribute to a species' collision risk are the maturity of the bird, dusk or dawn flying, adverse environmental conditions (e.g. misty or windy conditions), speed chasing of prey or elaborate aerial displays and the visibility of power lines (Bevanger 1994; Jenkins et al. 2010). Although there is no single approach or device effective for all species, a range of preventative (e.g. reviewing the placement of new lines) and mitigation options (e.g. fitting of markers on lines) have been found to be effective in reducing bird collision frequency (Bevanger 1994; Jenkins et al. 2010).

Raptors may benefit by making use of power line structures for nesting, roosting, hunting and feeding perches but they are also at risk from electrocution and collisions (Ledger and Hobbs 1999; Lehman 2001; Manville and Albert 2005). Although incidents are reported, and raptors can be susceptible to collisions (Martin and Shaw 2010), they are deemed not to be as much at risk from collisions (Ledger and Hobbs 1999; Anderson 2000). However, electrocution (rather than collision) is recognized as a significant cause of death for many species of raptors (Bevanger 1998; Ledger and Hobbs 1999; Lehman 2001; Manville and Albert 2005). For Bald Eagles (*Haliaeetus leucocephalus*) in the USA, electrocution has been identified as one of the leading causes of death along with road kills and persecution (Lehman 2001). Electrocution has been found to be one of the main causes of death (along with persecution) for (especially sub-adult) Bonelli's Eagle (*Aquila fasciata*) in Spain, one of Europe's rarest raptors (Real et al. 2001).

The risk of electrocution for raptors depends on a number of factors such as the species involved (e.g. size and wingspan), maturity of the bird, structure design (e.g. pole-top configuration and clearance among electrical components) and location of each structure (e.g. lack of alternative perches) (Bevanger 1994; Anderson and Kruger 2004; Lehman et al. 2007). In South Africa up to 95% of electrocutions occur only on 4 types of power pole structures;

22kV wooden T-structures, 88kV steel kite transmission towers, terminal H-frame wood structures and 88kV or 132kV Delta suspension structures (Krüger 1999 cited in Lehman et al. 2007). T-structures and terminal H-frame structures are responsible for killing a broad array of raptor species while the kite and Delta suspension mostly kill large species with wide-wingspans such as Cape Griffons (*Gyps coprotheres*), African White-backed vultures (*Gyps africanus*) and MEs (Krüger 1999 cited in Lehman et al. 2007; van Rooyen and Ledger 1999). Although a range of effective preventative and mitigation measures (e.g. staggered insulators) have been designed, cost and time constraints have prevented the widespread retrospective implementation of these measures (Van Rooyen and Ledger 1999; Anderson 2000). Lehman et al. (2007) in a systematic review of global raptor electrocution literature, concluded that over 30 years of sustained effort that in only a few studies have a reduction in raptor electrocution incidents been shown.

In South Africa, apart from a few selective species such as the Cape Griffons, power line mortality is generally not seen to be high enough to affect the long term population viability of raptors (Lehman et al. 2007). Deaths from both collisions and electrocutions have been reported for MEs but it is uncertain to what extent it is impacting the population (Ledger and Hobbs 1999; van Rooyen and Ledger 1999). In Kenya, the ME has been identified as one of the species that face a high risk of direct interaction with electrical infrastructure (Smallie and Virani 2010), and it seems likely that this conclusion may be just as valid in South Africa. During the 1996 - 1999 period, 10 MEs killed by electrocutions and two by collisions with power lines were reported to the Eskom-Endangered Wildlife Trust Strategic Partnership in South Africa (van Rooyen and Ledger 1999) whilst during the last 5 years an additional 22 power line deaths were reported (Kruger, R. pers. com), and because this depends on dead birds being found this will almost certainly be a huge underestimation of the real numbers.

1.3.4. Could changes in persecution by stock farmers be responsible for the decline?

Human activities have destroyed or transformed large areas of the world's natural land with a resulting decline in the number and richness of species that rely on these associated habitats (Lande 1998; Laurance and Cochrane 2001; Goudie 2009). Currently in South Africa, 80% of land can be considered as being in a natural or semi-natural state but even by the late 1980s it was estimated that only 7% of the land was undisturbed with over 83% being farmed (condition ranging from natural to transformed) and 10% being permanently transformed (Macdonald 1989; Gillson et al. 2012). With a reduction in available habitat, predators are increasingly under pressure directly (e.g. conflict with farmers) and indirectly (e.g. reduction in prey densities) to meet their survival needs (Laurance and Cochrane 2001; Dobson et al. 2006).

Farmers perceive predators as a threat to their livestock, game and poultry and as a result resort to eliminate this threat (Anderson 2000; Thirgood et al. 2005; Berger 2006; Moleón et al. 2011). Worldwide, sheep farmers experience a loss of around 5% from predation; this is considerably less than from other causes of loss, which account for a further 14% (Davies 1999). Considering various global studies on eagle diet and their impact on sheep production, it was estimated that out of a 20 - 30% yearly lamb mortality only 0 - 3% could be ascribed to eagle losses (Thirgood et al. 2005). White-tailed Eagles (*Haliaeetus albicilla*) on the island of Mull, Scotland were found to predate on sheep but these losses were seen not to be damaging to sheep farming on a broad scale (Marquiss et al. 2003). In a study on Black Eagles (*Aquila verreauxii*) in the Karoo it was demonstrated that even though lamb losses occur, the farmer was in fact more likely to benefit from a resident pair of eagles as they controlled the number of Rock Hyraxes (*Procavia capensis*) that could negatively impact on grazing availability (Davies 1999).

Territorial pairs of large eagles that are conserved are also likely to keep out roving young birds that are more likely to take domestic livestock (Anderson and Kruger 2004). Even though the true risk of predation is relatively low, it is the perceived risk to an individual farmer that is being reacted to, as the consequences of an attack can be severe and financially damaging (Joffe 2003).

The efficiency of reducing livestock loss has not been proven but farmers in general are persisting in persecuting predators in an attempt to reduce their losses (Berger 2006; Moleón et al. 2011). In South Africa it is hard to assess the impact of persecution on raptors as they have full legal protection and farmers will not readily admit to killing raptors (Anderson 2000). Raptors are not only directly persecuted by farmers by shooting and poisoning, but are also inadvertently killed when non-selective poisons and baited gin traps are used indiscriminately to control mammalian predators (Anderson 2000; Anderson and Kruger 2004). In East, West and southern Africa the widespread use of poisons have been implicated in the decline of several avian raptors (Brown 1991; Anderson 2000; Thiollay 2007 a,b; Virani et al. 2011).

1.3.5. Could increases in Elephants numbers within protected areas be responsible for the decline (in protected areas)?

Due to the Ivory trade African Elephant (*Loxodonta Africana*) numbers have been reduced significantly even to the point where they were close to being extirpated south of the Zambezi River (Cumming et al. 1997). In southern Africa the populations have recovered and are, in some cases, overpopulating areas (Cumming et al. 1997; Trollope et al. 1998). Contributing factors to areas being overpopulated are a moratorium on culling, fencing of protected areas preventing Elephants from completing seasonal movements, and artificial water points providing Elephants with year round access to all potential habitat (Cumming et al. 1997; Owen-Smith et al. 2006; Young et al. 2009). In Kruger National Park (KNP), the Elephant population has recovered from 20 animals in 1900 to more than 13,050 animals in 2007 (Trollope et al. 1998; Owen-Smith et al. 2006; Whyte 2007). Prior to the moratorium on Elephant culling in 1994 the Elephant population was maintained at the estimated carrying capacity of 7000 animals (Trollope et al. 1998; Owen-Smith et al. 2006), but has subsequently doubled (Whyte 2007).

African Elephants have the potential to significantly alter the vegetation communities and modify the structure of their habitat through their feeding behavior and movement (Cumming et al. 1997; Young et al. 2009; Haynes 2012). In KNP and surrounding areas there is a concern that the high density of Elephants are having a substantial negative impact on habitat structure and ecosystem functioning and that this may eventually lead to the loss of species (Trollope et al. 1998; Whyte 2001; Helm et al. 2009; Young et al. 2009). One major concern is that high Elephant density may be causing the elimination of large trees (Trollope et al. 1998; Whyte 2001; Helm et al. 2009; Young et al. 2009; Rode 2010). Shannon et al. (2008) found that 60% of large trees (>5 m in height) in the south of KNP showed elephant utilization and 4% died due to Elephant foraging behaviour. Trollope et al. (1998), in comparing aerial photos between 1960 and 1986/89, found a dramatic decline in the density of large trees in KNP, especially in the Arid vegetation types, and these changes have been ascribed mostly to the impact of Elephants.

For many animals including raptors, large trees are focal points for activity, providing them with shade, shelter, resting sites, nesting sites and observation sites for hunting (Dean et al. 1999). 40% of KNP bird species are estimated to depend on trees for some part of their life cycle (Owen-Smith et al. 2006). By felling large trees, Elephants are removing the existing or potential nesting sites of tree nesting birds (Rode 2010). On the basis of their findings that vultures completely avoided Elephant enclosures in two Swaziland reserves where all trees were debarked and killed, Monadjem and Garcelon (2005) suggested that high Elephant densities could be responsible for low vulture nest densities in KNP.

With the Elephant numbers in southern Africa continually rising, especially in fenced protected areas such as KNP, NGOs have expressed concern with regards to the potential impact on the nesting sites of vultures and raptors (Botha 2005). A range of vultures and raptors, including the African White-backed Vulture, Tawny Eagle, Martial Eagle and Bateleur Eagle, which are listed as *Vulnerable* or *Near Threatened* in southern Africa are potentially threatened by this impact (Botha 2005).

1.4. The value of protected areas for raptors

Human activities and their associated impacts have left protected areas as the only refuge for many species. A range of studies undertaken in West Africa and southern Africa showed that many raptor species as well as their game bird prey have higher abundance within protected areas than outside, or they are now only found inside and in buffer areas around protected areas (Brandl et al. 1985; Herremans and Herremans-tonnoeyr 2000; Hudson and Bouwman 2007; Thiollay 2007a,b). This can be explained by the fact that in areas outside of protected areas often only a fraction of the birds' original habitat remains, and in these areas they are frequently confronted with additional anthropogenic pressures, whereas within protected areas these pressures are in general absent (Thiollay 2007a,b).

There is a concern that protected areas are not sufficient to conserve certain species with large territory requirements, such as Martial Eagles (ME). This concern seems to be confirmed by the findings of Herremans and Herremans-tonnoeyr (2000) in Botswana that large eagles such as the ME, even while still being in the protected area, had a lower abundance (56%) on the edge than in the core of the protected area. This is explained by the ME territories spanning outside of the protected area where they are exposed to humans and their associated direct and indirect impacts (Herremans and Herremans-tonnoeyr 2000). This edge effect is given as a possible reasons for Kruger National Park (being only 60 km wide on average) having a relatively lower abundance of raptors compared to protected areas in Botswana (Herremans and Herremans-tonnoeyr 2000). Therefore, if these pressures surrounding protected areas have also increased, this may have had a knock on effect inside the park and may also explain declines within the protected areas.

1.5. Project Outline

1.5.1. Hypotheses Proposed

Based on the review above of Martial Eagle (ME) natural history as it relates to various potential drivers of change, I propose the following plausible hypotheses, which could be driving the decline (as recommended by Caughley (1994)), and associated predictions that I will subsequently test in this thesis.

Hypothesis 1 (H1): That the decline of MEs in South Africa is being caused through changes in temperature and/or precipitation.

Predictions:

1. That climate has changed in South Africa over the last 20 years, with changes in temperature and/or precipitation evident.
2. That population declines will correlate with changes in temperature and/or precipitation; in other words declines would be greatest where temperature or precipitation changes have been greatest.

The almost complete lack of knowledge on how the key demographic parameters of MEs are influence by weather or climate means that predictions on the direction of any such relationship are not possible.

Hypothesis 2 (H2): That the decline of MEs in South Africa is being caused through a decline in prey abundance as a result of changes in land use.

Predictions:

1. That a selection of important avian prey species has declined over the last 20 years.

2. That the prey declines would be greatest outside of protected areas. The assumption is that inside protected areas prey will be better protected from anthropogenic pressures.
3. That ME population declines will be negatively correlated with declines in a selection of important avian prey species.

With this hypothesis only a selection of avian prey is considered as these were the only data available at an appropriate scale to test this hypothesis and predictions. The assumption is made that if there is a relationship in ME declines with this selection of avian prey species then similar relationships with other prey species (e.g. medium sized mammals and reptiles) would be likely, since they would likely be influenced by the same land use changes and other pressures.

Hypothesis 3 (H3): That the decline of MEs in South Africa is being caused through high mortality caused by power line electrocution and collisions.

Prediction: That ME population declines will be negatively correlated with the density of power lines. This prediction was made with the assumptions that direct mortality due to power lines would reduce the chance that juvenile and non-territorial adult birds would settle or that a breeding pair would persist in an area.

A fourth plausible hypothesis and prediction is proposed here, but it could not be tested in this thesis as I was not able to acquire suitable data for the purpose.

(Hypothesis): That the decline of MEs in South Africa is being caused through high levels of persecution by stock farmers.

(Prediction): That ME population declines will be negatively correlated with the distribution of small stock grazing land in South Africa. The assumption is made that even though breeding pairs will also be persecuted, juveniles are more likely to be persecuted as they are roaming into new territories and are more prone as inexperienced hunters to take stock as prey.

In addition to the four hypotheses proposed as drivers for overall decline in MEs, I propose a hypothesis for the decline of MEs in protected areas with Elephants.

Hypothesis 4 (H4): That the decline of MEs in protected areas with Elephants in South Africa is due to an increase in Elephant densities.

Prediction: That population declines will be negatively correlated with Elephant density over the last 20 years in Kruger National Park. This assumes that in areas with highest densities of Elephants nesting or foraging opportunities will be reduced to the point where they are limiting and therefore the numbers of ME will be lowered.

1.5.2. Aims

In this thesis I use the change in reporting rates between the South African Bird Atlas Projects (SABAP 1 and SABAP 2) as they provide data on change in reporting rates between the late 1980s and late 2000s at a spatial resolution that allows population changes to be compared with a variety of environmental covariates. These changes in reporting rates will be used to quantify and spatially describe the overall decline for MEs in South Africa over the last 20 years as well as for the decline in different provinces and biomes. The change in reporting rates will also be used to determine whether the declines differ within and outside of protected areas. To investigate the potential drivers of change that are postulated as being responsible for the ME declines in South Africa, the above set of plausible hypothesis and predictions will be tested using the change in reporting rates to see if they correlate with covariates relating to each hypothesis.

An attempt will further be made to first determine if the change in reporting rates represent a real change rather than being an artefact of changing methods between SABAP projects, as some suspect. The protocols for data collection for the SABAP projects differ but the intention was always for the datasets to be compared (ADU 2013). The unit of spatial measurement for SABAP 1 was the quarter-degree grid cell (QDS) and the time period for collection was over one month while for SABAP 2 it is pentads (a ninth the size of a QDS) and over 5 days (Bonnevie 2011). These differences introduced many possible sources of bias including the comparison of reporting rates (e.g. lower apparent reporting rates in SABAP 2 than in 1) (Bonnevie 2011). This is one of the main concerns in using data with different protocols to examine change and as such I aim to establish additional independent evidence to confirm that the decline is real. I will aim to achieve this by comparing the change in reporting rates in selected QDSs with data collected from more intensive field research carried out in the same area, to see if these declines are also revealed on the ground.

Lastly, I will use the current SABAP 2 data (at the finer pentad scale) to describe occupancy and abundance differences between biomes, in relation to power line densities and between protected and non-protected areas.

Chapter 2: Quantifying and exploring the cause of the decline of Martial Eagles

2.1. Introduction

Understanding the cause of declines are vital if appropriate management practices aimed at halting or reversing declines are to be appropriately designed (Caughley 1994; Peery et al. 2004). Otherwise, resources can be misdirected to areas and issues which do not effectively address the real causes of declines (Caughley 1994).

Globally many species of raptor have undergone population declines as a result of direct and indirect anthropogenic pressures (Abbot 1933; Real et al. 2001; Cuthbert et al. 2011; Prakash et al. 2012). Studies in West, East and southern Africa have shown that many raptor species, particularly vultures and large eagles have undergone dramatic declines and are in some cases now only found inside protected areas and their surrounding buffer areas (Brandl et al. 1985; Herremans and Herremans-tonnoeyr 2000; Hudson and Bouwman 2007; Thiollay 2007a,b; Virani et al. 2011).

The Martial Eagle (ME) is endemic to Africa and occurs widely south of the Sahel in a wide range of habitats (Steyn 1982; Boshoff 1997; Simmons 2005; BirdLife International 2012). A number of natural history characteristics make MEs vulnerable to rapid population declines (Owens and Bennett 2000; Purvis 2000). MEs are apex predators that naturally live at low densities with pairs maintaining large territories for their size (125 km^2 to 300 km^2) (Brown 1970; Steyn 1982; van Zyl 1992; Ferguson-lees and Christie 2001), their preferred habitat of open woodland and savannas are also prime areas for large and small stock farming (Brown 1970; Steyn 1982; Simmons 2005), they have a low and erratic breeding rate with only 0.52 - 0.58 young reared annually (Steyn 1982; Hustler and Howells 1987; Boshoff 1993; Machange et al. 2005) and sub-adults are believed to roam widely outside of the presumably safer areas where they were raised and will stay nomadic until they establish a territory (Ferguson-lees and Christie 2001).

Throughout its range the species is known to be declining in abundance, if not in distribution, as recognised in its current *Near Threatened* IUCN Red List category status (Birdlife International 2012). BirdLife International is currently under consultation to upgrade the species status to *Vulnerable* or *Endangered* due to concerns that the species is declining further (Taylor, M. pers. com.). The continent wide decline of MEs, has also apparently occurred throughout South Africa over the last 20 years. This decline has been highlighted through a simple assessment of change in reporting rates from the two South African Bird Atlas Projects (SABAP 1 & 2) (Underhill 2012). Additionally, this assessment highlighted declines within the traditional strongholds for the species: large protected areas (Underhill 2012). In the South African Red List, the species will be upgraded from *Vulnerable* to *Endangered* in the 2013 revision (Taylor, M. pers. com.).

There is an urgent need to identify and understand the factors that are responsible for the decline of this species so that research and management can be focused to arrest these declines (Peery et al. 2004). In this chapter, I aim to examine the change in reporting rates between SABAP 1 (1987 – 1993) and SABAP 2 (2007 – 2012) to explore and quantify the apparent decline of MEs in South Africa. I then aim to test the hypotheses and predictions discussed next for potential drivers of ME decline based on the hypothetico-deductive path proposed by Caughley (1994). This is possible as the SABAP data has been collected spatially, so that changes in specific areas can be compared to environmental variables in these same areas, and thereby facilitating the tests of these predictions.

Climate change is known to affect many bird species by making existing areas less suitable (in terms of habitat or prey), affecting the range of their prey available and their breeding phenology (Bond 1994; Simmons et al. 2004; Both et al. 2006). Climate is known to have changed in South Africa in recent decades in that the western half, northeast and east have shown relative increases in warm extremes and decreases in cold extremes (Kruger and Sekele 2012). I therefore hypothesize (**H1**) that climate change may be causing the decline in MEs, and if so I would predict that declines would be greatest in areas where climate has changed the most. Although, due to the lack of knowledge on the interactions between MEs and climate/weather, I do not predict the direction of any such response.

In the absence of persecution, prey abundance is considered to be the main limiting factor for raptors (Newton 1979). Several studies have shown declines in prey to be the cause of population declines in raptors, for example Hen Harriers (*Circus cyaneus*) in Orkney, UK (Amar and Redpath 2002) and Lesser Kestrels (*Falco naumanni*) in European pseudo-steppes (Ursua et al. 2005). Indeed, Thiollay (2006b) suggested that the decline in large eagle species in West Africa could be directly attributed to the simultaneous decline of large bird prey. MEs prey on a wide variety of prey, including medium and large sized bird species e.g. Bustards, Guineafowl and Spurfowl. Many such species have apparently declined at least locally in South Africa in recent decades e.g. Ludwig's Bustard (*Neotis ludwigii*) (Jenkins et al. 2011) and Swainson's Spurfowl (*Pternistis swainsonii*) (van Niekerk 2011). Therefore another plausible hypothesis (**H2**) for the decline is that it has been caused by a decline in prey abundance. If this is the case I would predict that the species has declined most in areas where prey has declined the most. To test this prediction I used changes in the reporting rates of large bird prey species (that feature in the ME's diet) within each QDS and examined whether declines in these prey correlate with ME declines.

Birds of prey are known to be vulnerable to collisions and particularly electrocutions with power lines (Bevanger 1998; Ledger and Hobbs 1999; Lehman 2001; Manville and Albert 2005). Electrocutions with power lines are known to be among the greatest causes of mortality for some eagle species e.g. Bonelli's Eagle in Spain (Real et al. 2001). Within South Africa a number of species are threaten by high collision and electrocution rates e.g. Cape Griffon (Lehman et al. 2007), and MEs are known to suffer mortality from this threat (Ledger and Hobbs 1999; van Rooyen and Ledger 1999). Therefore another plausible hypothesis (**H3**) is proposed: that declines in MEs have been caused through high mortality through power line electrocution and collisions. If so I predict that declines will be greatest in areas with higher density of power lines.

Given that declines have also occurred within protected areas, which should be buffered against many of the factors linked to the above hypotheses, I suggest another additional hypothesis to explain the declines within these areas. In South Africa Elephant numbers have recovered to the extent that they overpopulate some areas (Cumming et al. 1997; Trollope et al. 1998). High Elephant densities may lead to habitat structure and ecosystem function changes with a subsequent loss of species (Trollope et al. 1998; Dean et al. 1999; Whyte 2001; Young et al. 2009). Tree nesting vultures and eagles including MEs can also be negatively affected by the high Elephant densities if their nesting trees are removed by Elephants (Monadjem and Garcelon (2005)). I thus propose another plausible hypothesis (**H4**), that the decline of MEs in protected areas in South Africa is due to an increase in Elephants. If so I predict that population declines of MEs in Kruger National Park will be correlated with the density of Elephants.

Further in this chapter, I will also attempt to verify whether the changes detected in the comparison of the two SABAP projects are real by comparing the change in reporting rates from the SABAP data with data collected from more intensive field research carried out in the same area. This will be done because the protocol of the two projects differed both spatially and temporally, and this may have introduced possible sources of bias when comparing the data (See Bonnevie 2011 for further details).

Finally, SABAP 2 reporting rates (which are carried out at a finer scale) will be used to describe and quantify the sighting probability and relative abundance of MEs in relation to a range of environmental covariates.

2.2. Methods

2.2.1. Survey data used to examine change and current abundance measures

Data from the Southern African Bird Atlas Projects (SABAP 1 and SABAP 2) was obtained from the Avian Demographic Unit (ADU), University of Cape Town, who coordinate these national citizen science surveys. SABAP 1 (running from 1987 to 1993) was conducted at a Quarter-degree grid cell (QDS) resolution (approx. 26 km x 27 km) and data were collected over a 30 day period, while SABAP 2 (2007 and ongoing) is conducted at a pentad resolution (one ninth of a QDS - approximately 9 km x 9 km) with data being collected over a 5 day period (Harrison and Underhill 1997; ADU 2013). All vetted records (check lists of the birds seen in an area) submitted until 31 August 2012 for SABAP 2 were included in the project (checklists were submitted for 63% of all the pentads in South Africa). To ensure spatial comparability between the two datasets, the SABAP 2 pentad data was combined at the QDS resolution. For each QDS in South Africa the data for the total number of cards submitted as well as the number of cards with Martial Eagles (ME) recorded on was provided for both SABAP projects. The reporting rate was the main metric of importance and is used as an index of ME abundance. This rate was determined by dividing the number of cards with MEs by the total number of cards for each QDS.

A minimum card threshold to be included for subsequent analysis was explored to try and increase the robustness of any change recorded. A five card minimum was chosen for both SABAP projects, which represented the best compromise to ensure a reasonably robust change in reporting rate within a QDS analysis while minimising the extent of information lost. This threshold removed 720 (36.6% of all QDSs which had cards for both periods) QDSs from further analysis. No minimum card threshold was implemented for the analysis describing occupancy and relative abundance at the pentad level using only SABAP 2 data.

2.2.2. Validation of the comparability of SABAP 1 and 2 to detect change

Because of the changes in the spatial and temporal resolution of surveys between the two SABAP projects, it was important to establish the validity of any changes detected by the SABAP comparison using independent field survey data, which could then be compared with reporting rate changes from SABAP 1 and 2 within the same area.

Herholdt and Kemp (1997) monitored ME nesting territories in trees in the Auob and Nossob riverbeds in the Kalahari National Park from 1988 to 1993 (i.e. covering a similar time period as SABAP 1). In 2011 and 2012, ME nesting territory data in this region was again collected by the Endangered Wildlife Trust through repeat road surveys, using similar techniques and some of the same surveyors as were used in the Herholdt and Kemp (1997) field survey (Whittington, M. pers. com.). Each nesting territory from the two surveys were allocated to a QDS in ArcGIS 9.3 (ESRI 2011). I then compared the change in reporting rates (from the SABAP projects) with the changes detected from these field surveys.

2.2.3. Environmental Covariates

To test many of the proposed hypotheses, and to explore correlates of contemporary patterns of occupancy and abundance, I extracted environmental data which was applicable to each of the Quarter-degree grid cells (QDS) and pentad cells. To achieve this, relevant data sources were collated and extracted using ArcGIS 9.3 (ESRI 2011). To ensure accurate calculation of areas for each grid cell, all data layers were re-projected to Albers Equal Area (central meridian =24.00°E, standard parallels =18.00°S and 32.00°S). Data were extracted only for grid cells clipped to the South African geo-political boundary and, as such, cells that border the coast or neighboring countries would only be partial cells. Refer to Appendix A for maps of the environmental covariates used.

Geopolitical boundaries and biomes

National and Provincial data were obtained from the Municipal Demarcation Board of South Africa (MDB 2013). The proportion of each of the 9 provinces in each QDS was extracted and a single category assigned for each QDS based on the province covering the highest proportion. These steps used to extract and assign a single value for each QDS were also used in the assignment of a biomes type to each QDS and pentad.

Data on the 9 South African biomes (Albany Thicket, Desert, Forest, Fynbos, Grassland, Indian Ocean Coastal Belt, Nama Karoo, Savanna and Succulent Karoo) were sourced from Mucina and Rutherford (2006). For ease of analysis any biome type that was dominant in less than 10 cells were reclassified to the second most dominant biome in a cell or to the dominant biome in the surrounding cells if there was no other biome in a cell. For both the QDS and the pentad analysis this approach reclassified all Desert and Forest biome QDSs and pentads.

Protected areas

Data on formal terrestrial protected areas (statutory designated sites) in South Africa were obtained from Biodiversity GIS (BGIS 2007a) which was part of the 2011 National Biodiversity Assessment (Driver et al. 2012). To ensure comprehensive coverage, data on informal protected areas (unproclaimed private nature reserves, game reserves and game farms) were also included, these data came from both the Biodiversity GIS (BGIS 2007b) as part of the 2008 National Protected Area Expansion Strategy (GSA 2010) and The World Database of Protected Areas (Protected Planet 2012). For analysis, formal and informal protected areas were combined, and a cell was classified as protected if $\geq$50% of the cell area was protected (QDS and pentad).

Power line data

Transmission and distribution line data for South Africa were obtained from Eskom (2011). From these data I included the following categories of lines: existing, commissioned and decommissioned, and excluded those that were designed, dismantled, invalid, planned or surveyed. Decommisioned lines were included as although they are not electrified they still pose a collision hazard. The three categories of power lines were combined for each cell (QDS and pentad) to calculate a single value, and were also analysed separately as distribution ($\leq$33kV and 44-110kV) and transmission ($\geq$132 kV) lines (QDS). The calculated values were the length of power line (meters) per cell area (km^2). 79% of the power lines were $\leq$33kV, 14% were 44-110 kV and only 7% were the larger $\geq$132 kV transmission lines.

Climate change data

Precipitation (P) and temperature (T) data for years 1987 - 2010 were obtained from the University of Delaware's Centre for Climate Research (Matsuura and Willmott 2012a,b). These data are derived through interpolation onto a half-degree grid cell resolution (equal to four QDSs) of irregularly spaced global weather station data (1,600 - 12,300 global station records for monthly air T and 4,100 - 22,000 global station records for monthly total rain gage-measured P for the period 1900 - 2010) (Matsuura and Willmott 2012a,b). A mean P and T for each year for each QDS was derived from averaging these monthly data. For each QDS, I calculated a single change measure for T and P, as the slope of a linear regression fitted to the 1987 - 2010 mean annual time series.

Changes in prey abundance

Data on the reporting rates of bird species that were identified as important ME prey species were obtained for both SABAP 1 and SABAP 2 from the ADU. Thirteen medium and large sized South African bird species that are known prey species of MEs (Steyn 1982; Tarboton and Allan 1984; Ferguson-lees and Christie 2001; Ratcliffe 2005; Simmons 2005; Thiollay 2006a) were used. These species are African Sacred Ibis (*Threskiornis aethiopicus*), Coqui Francolin (*Peliperdix coqui*), Crested Francolin (*Dendroperdix sephaena*), Denham's Bustard (*Neotis denhami*), Glossy Ibis (*Plegadis falcinellus*), Hadeda Ibis (*Bostrychia hagedash*), Helmeted Guineafowl (*Numida meleagris*), Karoo Korhaan (*Eupodotis vigorsii*), Kori Bustard (*Ardeotis kori*), Ludwig's Bustard (*Neotis ludwigii*), Natal Francolin (*Francolinus natalensis*), Northern Black Korhaan (*Afrotis afraoides*) and Swainson's Spurfowl (*Francolinus swainsonii*). For each QDS a single measure representing the change in combined reporting rates (rr) for all species between the SABAP projects was calculated:

$$\text{Combined Species Change Measure} = \frac{\text{Sum of SABAP 2 rr}}{(\text{Sum of SABAP 2 rr} + \text{Sum of SABAP 1 rr})}$$

African Sacred Ibis, Hadeda Ibis and Helmeted Guineafowl were recorded in 80% or more of QDSs in one or both SABAP projects while Glossy Ibis, Swainson's Spurfowl and Natal Francolin were recorded between 45 and 50% and the remaining species were below 31%. Testing only the three most widely distributed species in the change measure showed a similar change in prey trend as using the full 13 species change measure but the full change measure was deemed to be a more robust and relevant measure for illustrating subtler changes. Whilst it is recognized that the proportion of avian prey in the diet of MEs differs between regions (Boshoff et al. 1990), with higher proportions of mammals and reptiles in some areas, I was limited to data for avian prey, as birds were the only major prey type for which data existed to allow change in prey abundance to be examined across the entire South African range and at an appropriate scale for this analysis.

Elephant density data from Kruger National Park

Elephant aerial census data for Kruger National Park (KNP) for the years 1987 - 2008 were obtained from the South African National Park Data Repository (SDR 2012). Surveys were conducted annually using a helicopter flying a set pattern along the river systems in KNP as per the standardized technique adopted since 1967 (Whyte 2007). For each QDS the average number of Elephants per year was extracted. A single measure providing the density of Elephants per QDS was calculated by averaging the number of Elephants for the 1987 - 2008 time series and dividing it by the total QDS area.

2.2.4. Statistical Analysis

Generalised linear models were used to quantify and explore variables associated with the changes in reporting rates between the two SABAP projects and also to examine variables associated with the current abundance of MEs. All analysis were carried out using the statistical package R 2.15.2 (R Core Development Team 2013).

My analysis of the change in reporting rates between the SABAP projects at the QDS scale used two modeling approaches. For these analyses, QDSs where MEs were never reported were excluded from all subsequent analysis.

The first analysis aimed to quantify changes (model 1) in reporting rates at various levels: 1) nationally, 2) within each province, 3) between biomes and 4) within or outside of protected areas. This model used a binomial structure with a logit link function, with the number of positive cards for a QDS (for each project) as the numerator and the number of negative cards for that QDS (for each project) as the denominator. "Period" (SABAP 1 or SABAP 2) was fitted as a categorical fixed effect in the model to examine the change nationally and as an interaction to examine the change in reporting rate for each of the other variables fitted as a main effect (e.g. province, biome, protected area status). This interaction therefore tested whether the reporting rate change differed between the two projects in different provinces, biomes or depending on whether QDSs were protected. The nature of this analysis accounted directly for the differing sample size of the number of cards submitted for each QDS. This same approach was also used to quantify the overall changes in avian prey abundance (combined and for each species individually) and whether it changed within and outside of protected areas (combined). For the combined avian prey abundance the average number of positive cards for a QDS (for each project) for the 13 species was the numerator and the average number of negative cards for that QDS (for each project) was the denominator.

I further examined whether change in reporting rates within QDSs was correlated with a number of potential factors associated with my key hypotheses. For these analysis a change metric was my response variable, this again had a binomial structure with a logit link function with the reporting rate from SABAP 2 specified as the numerator and the sum of the reporting rates for SABAP 1 and SABAP 2 as the denominator (model 2). A similar analytical approach was used by Amar et al. (2010; 2011) to examine changes in British upland waders in relation to a number of covariates. This approach yield an output range from 0 - 1 with 0=extinction, 0.5=stability, 0.66=100% increase and 1=colonisation from SABAP 1 to SABAP 2. As recommended by Amar et al. (2011) all QDSs with zero counts during SABAP 1 were excluded from analysis, as these "colonisation events" would have a disproportionately higher value in the response variable than QDSs showing large increases of pre-existing populations; this removed 78 (10%) colonised QDSs from the analysis i.e. those outputs with a value of 1. To account for sample size variation between QDSs, the model was weighted using the minimum numbers of cards submitted for either of the two projects. With any data of this nature, spatial autocorrelation could influence the relationships with covariates; therefore to account for spatial auto-correlation which might be present, latitude (X) and longitude (Y) and their interaction were fitted as fixed effects in the model. To allow for ease of interpretation, I present the back-transformed output from model 2, the change metric, as a percentage change in reporting rate between the SABAP projects. The association between change in ME records and the four covariates (change in temperature, change in rainfall, power line density and prey changes) were examined within a full model including all terms, and I report their type III (partial) significance. Prior to building the full model I found that none of the covariates were correlated with any of the other covariates (refer to Appendix B - Table 6). Lastly, under this set of analytical procedures, I analyzed a subset of these data, which included only QDSs falling within Kruger National Park (n=40), using the same structure as model 2, but without the spatial terms, to explore for any associations between Elephant densities and ME declines.

The other analytical component of this chapter concerned the analysis of the current occupancy and relative abundance based on the SABAP 2 pentad scale data and used two modeling approaches. Model 3 examined current occupancy and used a binary measure of presence/absence as the response variable and was fitted with a binomial error structure and a logit link function. This model was used to describe and quantify where MEs are currently found in relation to the explanatory variables (biome category, power line density and protected area status). Thus for this analysis all pentads that had been surveyed were used, including those in which no MEs were seen in any survey. Model 4 examined relative abundance and used the number of positive cards for MEs as the response variable and the total number of cards in each pentad was specified as an offset to account for variation in sample size. Model 4 was fitted with a Poisson error structure with a log link function. This model examined the relative abundance of MEs in relation to the explanatory variables. A variation of model 4, which included only pentads where MEs were seen, was used to examine how abundant the species was in areas it occupied in relation to the explanatory variables.

To correct for overdispersion identified for both change in abundance and current abundance models, a quasibinomial or quasipoisson distribution was used. A restriction of using these types of distribution is that the fitting of different models could not be compared using an Information Theoretic approach (e.g. AIC).

The lsmeans (least-squares means) R package (Lenth 2013) was used for categorical variables analyzed with the models to derive the differences in output within each term. The lsmeans outputs were back-transformed (to account for the link function) to provide the differences in terms of the models response variable input measure and confidence limits for each level of factors. Significance of differences between each level were evaluated through pairwise comparison derived from the post-hoc lsmeans test.

2.3. Results

2.3.1. Martial Eagle change in abundance

From the 1245 Quarter-degree grid cells (QDS) considered for this analysis, 475 QDSs (38.2%) contained no records of Martial Eagles (ME) during either South African Bird Atlas Project (SABAP) and so were excluded from any further analyses. This left 770 (61.9%) QDSs that were used in the analysis. When considering the change categories for the 770 QDSs, MEs were completely lost from 381 QDSs (49.5%), and declined in 198 QDSs (25.7%) (Figure 1, Figure 2). MEs increased in 113 QDSs (14.7%) and colonised 78 QDSs (10.1%) (Figure 1, Figure 2).

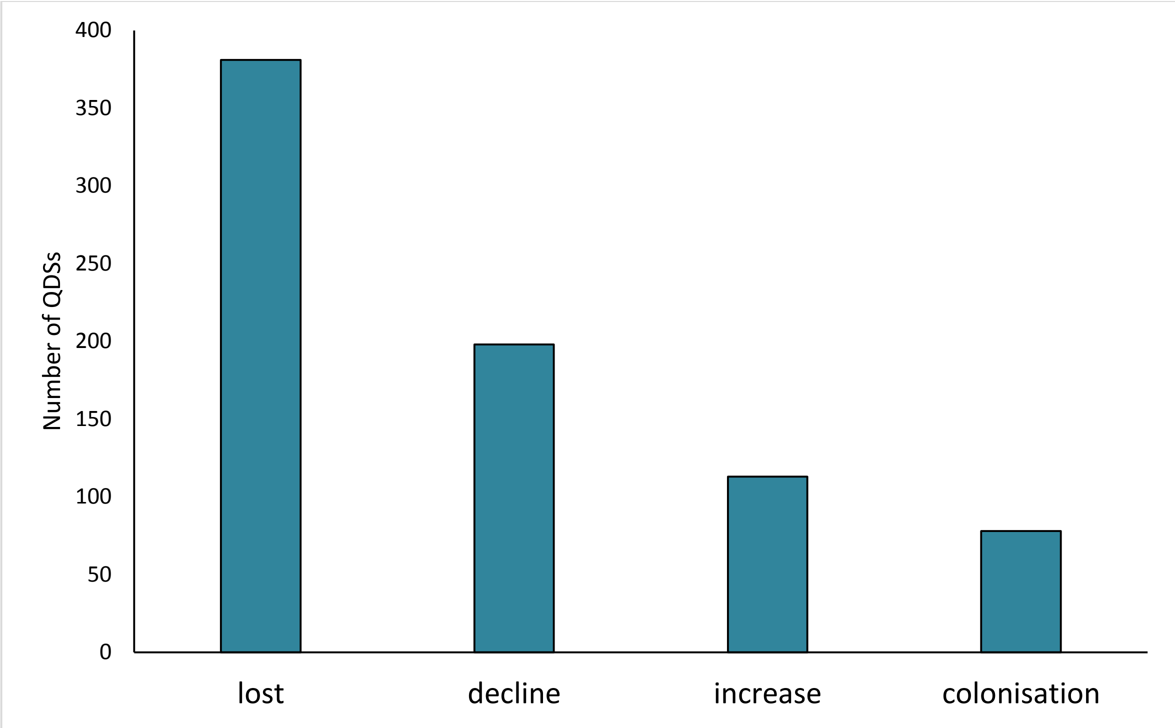
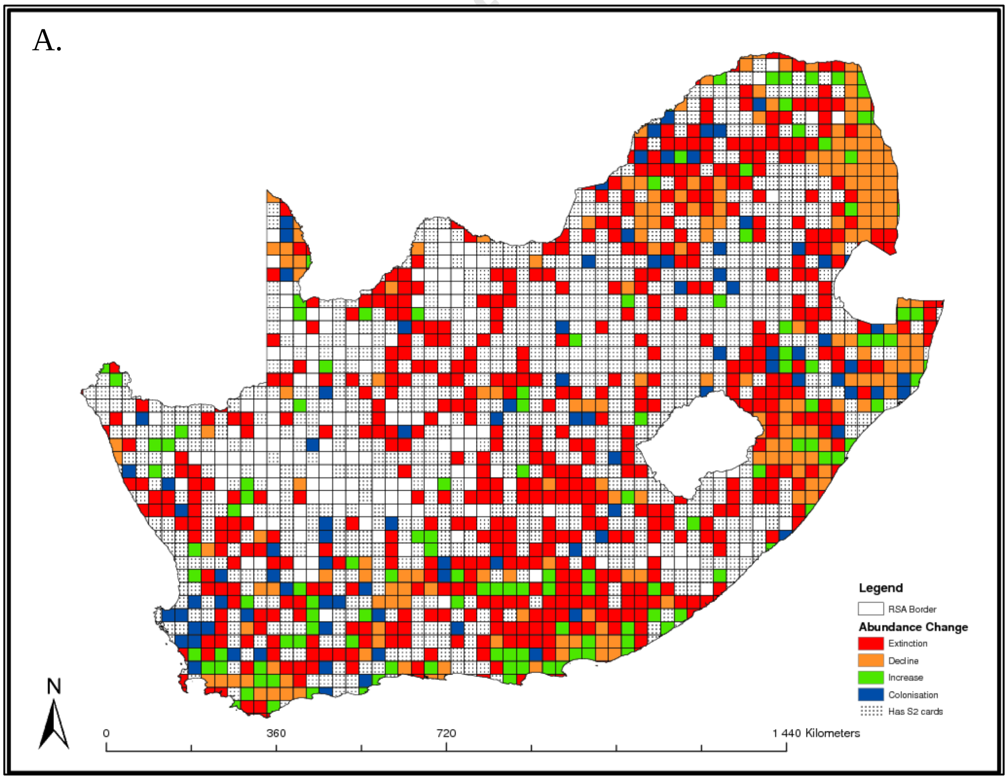


Figure 1: Graph summarising into categories the distribution of ME changes in reporting rates among QDSs across South Africa included in the analysis. In the majority (75%) of QDSs, MEs were either completely lost or declined in reporting rates.



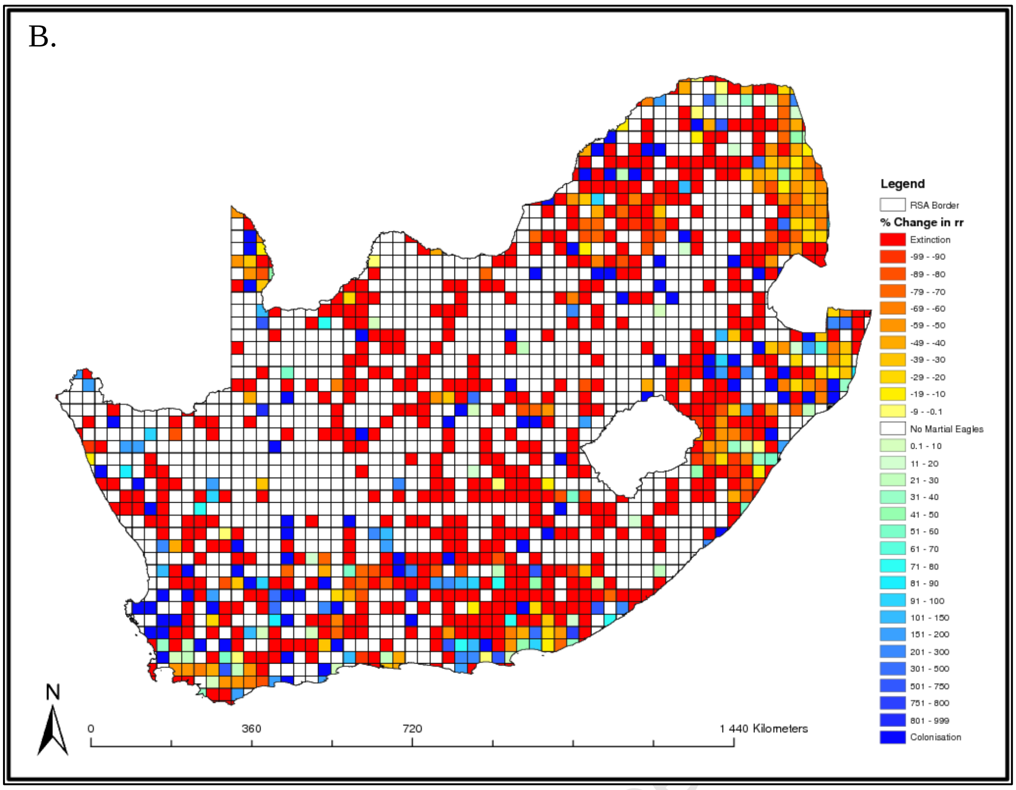


Figure 2: Map showing the ME reporting rate changes in South Africa. Map [A] shows the changes using the same categories used in other SABAP reporting. Map [B] shows the percentage change for each QDC ranging from decreases being shown through the hotter colours and increases being shown by the cooler colours. Blank QDSs are QDSs that had 5 or less cards submitted in either period or where there is no ME abundance change information.

Analysing change (model 1), explicitly incorporating sample size of cards in each project, a significant decrease in ME reporting rates per QDS between the two SABAP projects was found (Figure 3, Table 1). There has been a 59% decline in reporting rates per QDS, with a drop from a 7.3% (6.6 - 8%) during SABAP 1 to only 3% (2.5 - 3.6%) during SABAP 2 (Figure 3).

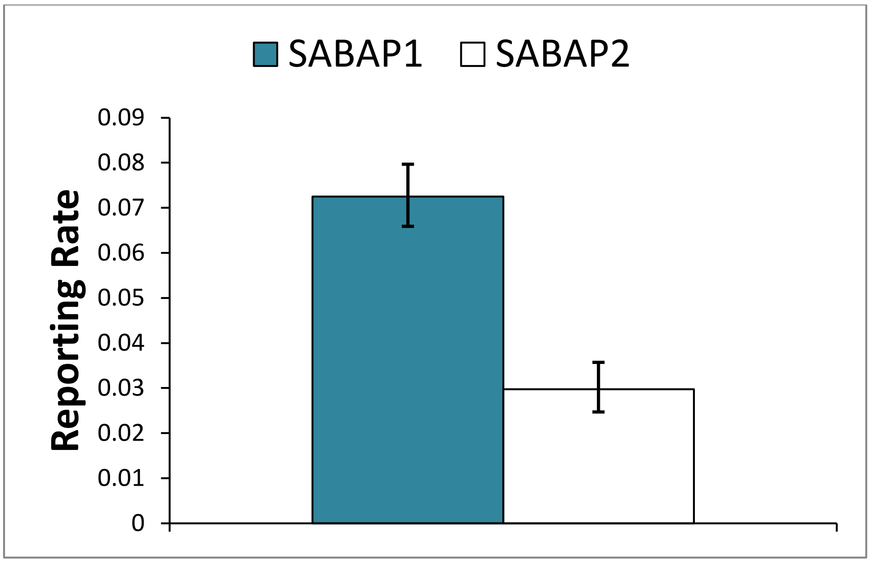


Figure 3: The overall difference in South Africa in the probability of reporting MEs between the SABAP projects (mean ± 95% confidence limit).

There was no significant interaction between province and period indicating that the changes in reporting rates for MEs between the two SABAP projects were similar between provinces (Figure 4, Table 1).

Changes in reporting rates of MEs differed between biomes, with a significant interaction between biomes and period (Figure 5, Table 1). Pairwise comparisons revealed significant decreases in the Grassland (-77.9%, n=184, t=5.08, $p<0.001$), Indian Ocean Coastal Belt (-81.5%, n=27, t=3.87, $p<0.001$), Nama Karoo (-76%, n=83, t=4.36, $p<0.01$) and Savanna (-60.9%, n=289, t=9.22, $p<0.001$) biomes (Figure 5). Whereas no significant changes were found in Albany Thicket, Succulent Karoo, and non-significant increases were detected in the Fynbos biome (Figure 5).

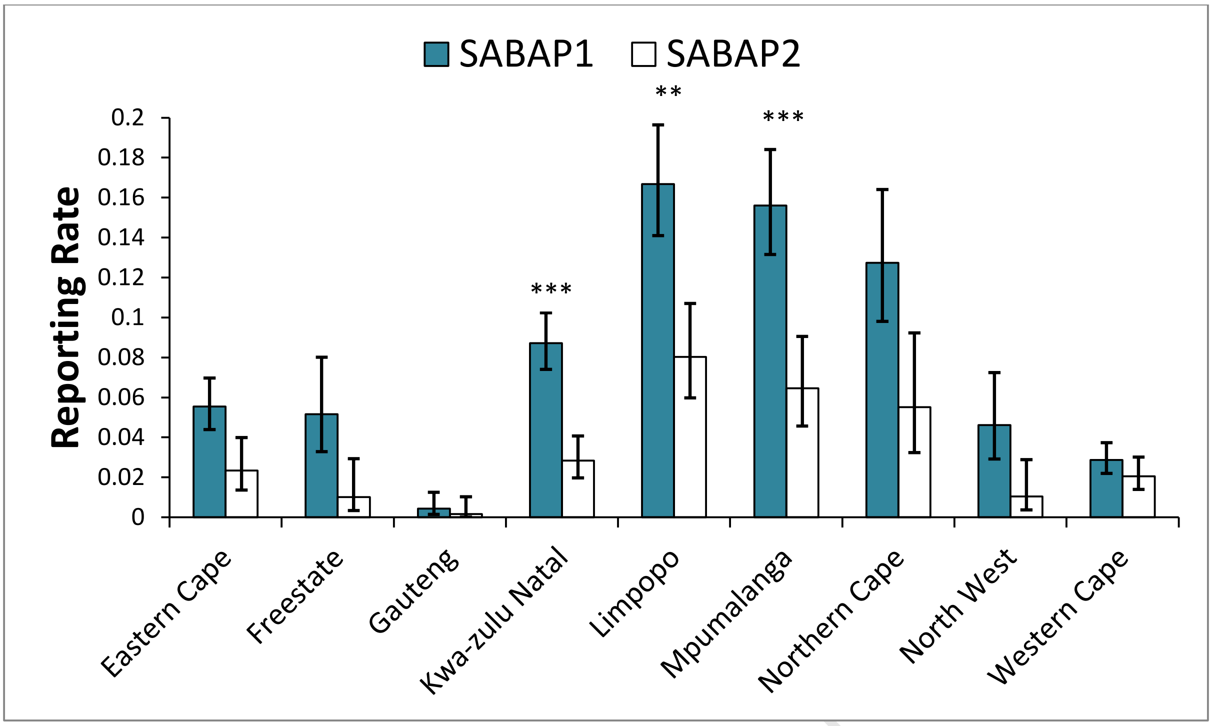


Figure 4: Reporting rates of MEs across the nine South African provinces between the SABAP projects (mean ± 95% confidence limit). Significant changes are marked as follows: *p<0.05, **p<0.01, ***p<0.001.

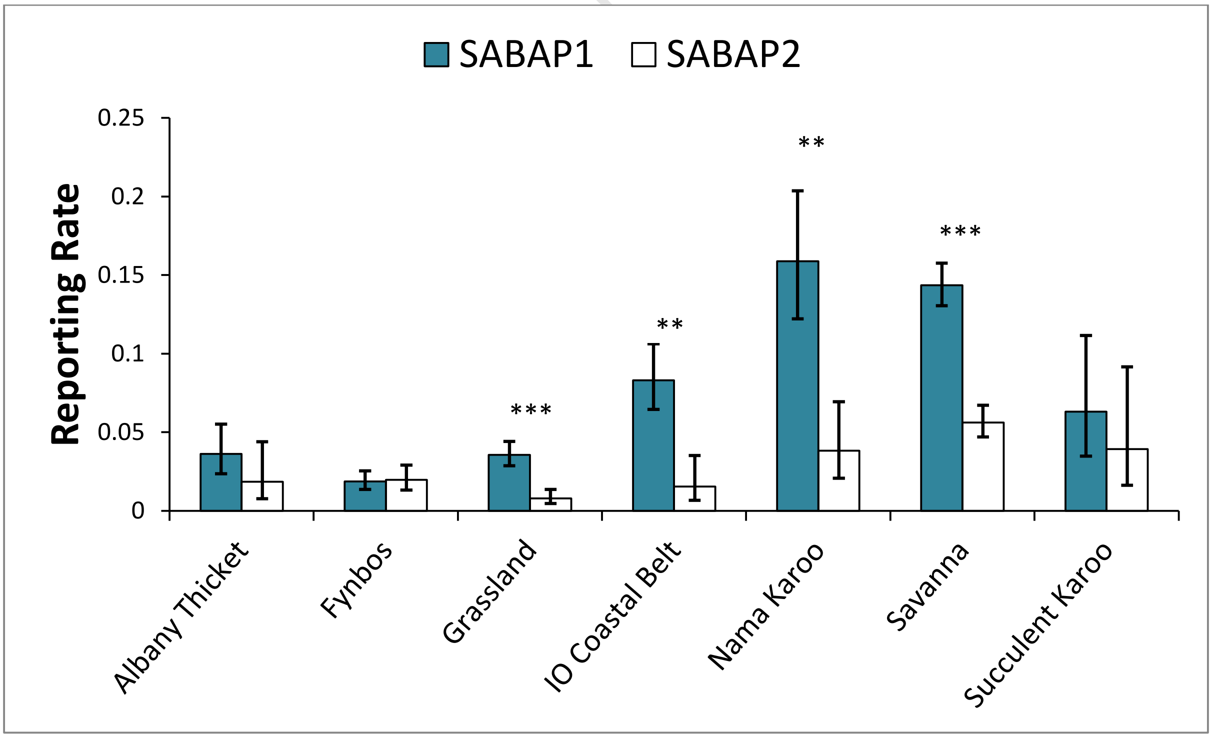


Figure 5: Reporting rates of MEs across the dominant biomes in South African provinces between the SABAP projects (mean ± 95% confidence limit). Significant changes are marked as follows: *p<0.05, **p<0.01, ***p<0.001. IO=Indian Ocean.

I found significant differences between the levels of decline inside and outside of protected areas, as evidenced by the significant interaction between protected area status of a QDS and period (Figure 6, Table 1). Pairwise comparison revealed significant decreases in both areas, but with a larger decline of 64% outside of protected areas (n=691, t=8.39, p<0.001) as compared with a 42% decline inside of protected areas (n=79, t=3.80, p<0.001) (Figure 6).

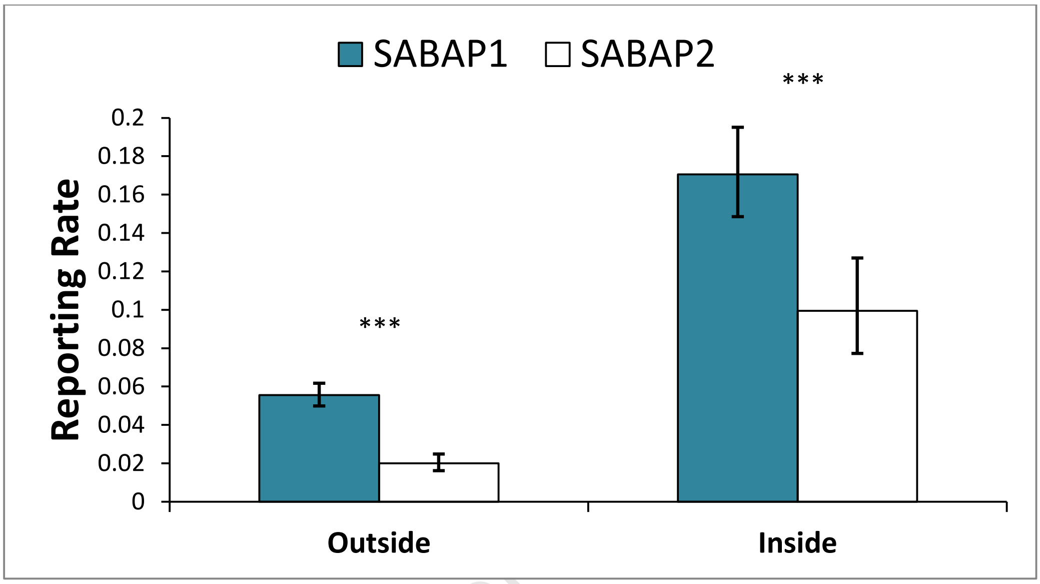


Figure 6: Reporting rates of MEs outside and inside of protected areas in South Africa between the SABAP projects (mean ± 95% confidence limit). Significant changes are marked as follows: *p<0.05, **p<0.01, ***p<0.001.

To examine this pattern further, I looked at changes in reporting rates inside three of the bigger protected areas in South Africa. Declines were apparent in all three areas, with significant decreases for Kruger and Kalahari National Parks and a near significant (p<0.1) decrease for Hluhluwe-iMfolozi (HI) Park (Figure 7, Table 1). In Kruger National Park there has been a 54.1% decline in reporting rates with a drop from on average 32.2% (28.2 - 36.5%) reporting rate per QDS during SABAP 1 to only 14.8% (11.5 - 18.9%) during SABAP 2 (Figure 7A). In the Kalahari National Park the reporting rates declined by 44% with a drop from 32.1% (25.7 - 39.2%) to 17.9% (11.6 - 26.8%) (Figure 7B), while for HI there has been a 54.4% decline with a drop from 17.8% (10.4 - 28.6%) to 8.1% (3.5 - 17.4%) between the two SABAP projects (Figure 7C).

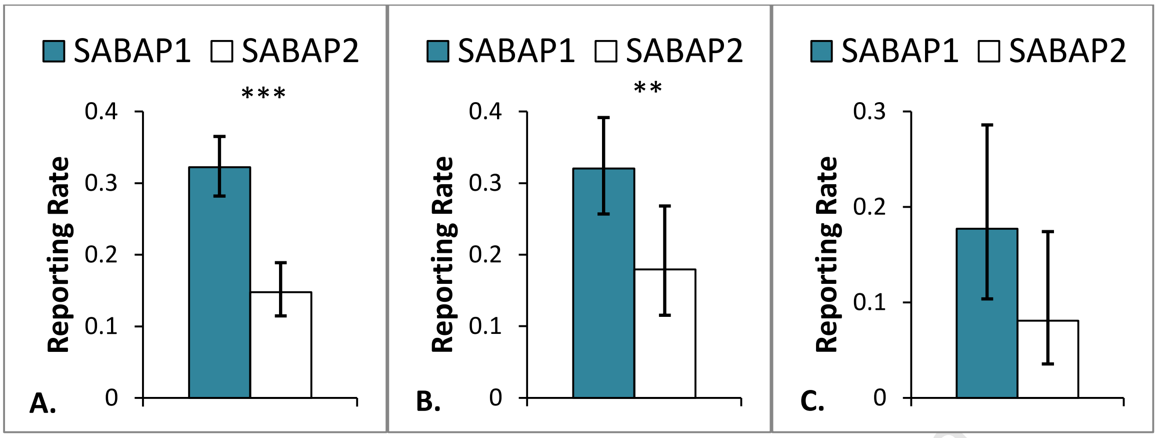


Figure 7: Difference in Kruger National Park [A], Kalahari National Park [B] and Hl Park [C] in the probability of reporting MEs between the SABAP projects (mean ± 95% confidence limit). Significant changes are marked as follows: *p<0.05, **p<0.01, ***p<0.001.

Table 1: Outputs from the generalised linear model 1 exploring changes in reporting rates. The Period χ^2 values for National and Kruger National Park, Kalahari National Park and Hl Park relate only to whether there was a significant change in reporting rates between the SABAP projects. For provinces, biomes and within and outside of protected areas the Main term χ^2 values (whether there is a significant change in ME reporting rates between categories) and the Interaction χ^2 values (whether there is a significant change in ME reporting rates between the SABAP projects for the Main term categories) are further reported. Significant changes are marked as follows: .p<0.1, *p<0.05, **p<0.01, ***p<0.001. NP = National Park. Df = Degrees of freedom.

Main Term	Significance			χ^2						Intercept
	Period	Main Term	Interaction	Period	Df	Main term	Df	Interaction	Df	
National	***	-	-	83.7	1	-	-	-	-	-2.6
Provincial	**	***	NS	10.1	1	317.6	8	10.1	8	-2.8
Biomes	NS	***	***	2.1	1	377.1	6	25.2	6	-3.3
Protected Areas	***	***	*	83.6	1	134.1	1	4.3	1	-2.8
Kruger NP	***	-	-	36	1	-	-	-	-	-0.7
Kalahari NP	**	-	-	7.6	1	-	-	-	-	-0.8
Hl Park	.	-	-	3.7	1	-	-	-	-	-1.5

For one of these protected areas, the Kalahari National Park, I was able to examine the trends detected from the SABAP analysis in more detail by using intensive monitoring done by Herholdt and Kemp (1997) that was independent of the SABAP projects. Comparison of the number of nesting territories between 1993 and 2011/2012 showed a decline of 43.8% (16 in 1993; 9 in 2011/2012) (Figure 8, Table 2) which is in close agreement with the 44% decline found from examining the SABAP data. For comparisons on an individual QDS level, the change in nesting territories with the SABAP reporting rate change category showed that 67% (six of the nine QDSs with nesting territories in either monitoring survey) agreed on the change direction (Figure 8, Table 2). This agreement included both QDSs with declines and increases. With the three non-agreement QDSs all showing a SABAP decline, even though the nests stayed the same, it can be assumed that roaming non-territorial birds were further lost from the park as the number of territories declined (Table 2).

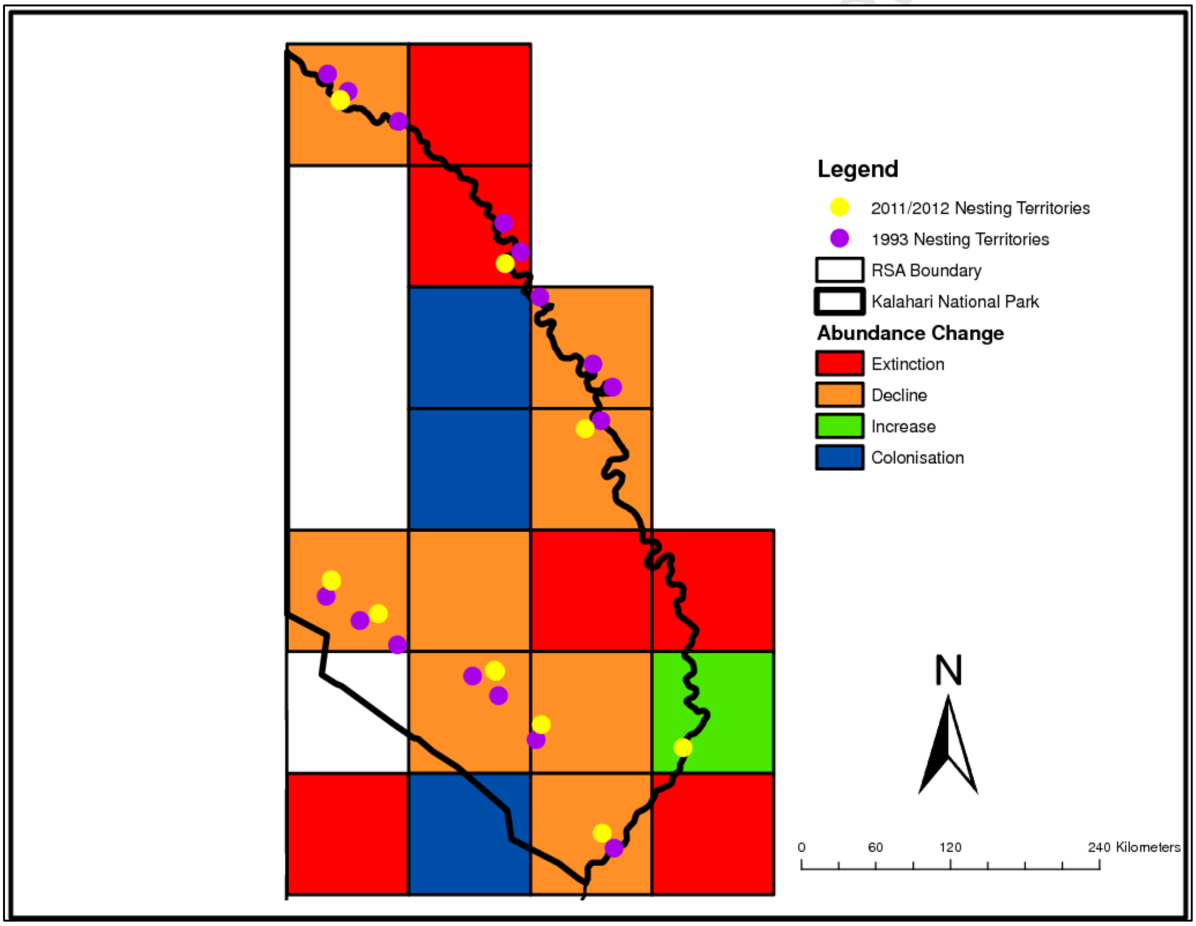


Figure 8: Map showing the approximate location of ME nesting territories for both the 1993 and 2011/2012 periods along the Auob and Nossob riverbeds in the Kalahari National Park. The nesting territories are overlaid on the ME reporting rate change categories so that a comparison can be made between the changes in nesting territories and the SABAP reporting rate categories. Nesting territories are found in 9 of the 18 QDSs in one or both of the monitoring surveys.

Table 2: Comparison on an individual QDS level of the change in nesting territories with the SABAP change in reporting rate categories in the Kalahari National park.

QDS	Nesting Territories		Difference	Pattern	SABAP	Comparison
	1993	2011/12			Change	Agreement
2420CC	3	1	-2	decline	decline	Yes
2520BC	3	0	-3	decline	decline	Yes
2520CC	3	2	-1	decline	decline	Yes
2620AB	2	1	-1	decline	decline	Yes
2520AB	2	1	-1	decline	"extinction"	Yes
2520DA	1	1	0	same	decline	No
2620BC	1	1	0	same	decline	No
2620BA	1	1	0	same	decline	No
2620BB	0	1	1	increase	increase	Yes

2.3.2. Avian Prey change in abundance

Analyzing change (model 1) in combined avian prey reporting rate, a significant increase in reporting rates per QDS between the two SABAP projects was found (Figure 9; Table 3). There has been an 8% increase in reporting rates per QDS, with an increase from 16.1% (15.7 – 16.5%) during SABAP 1 to 17.5% (16.9% - 18%) during SABAP 2 (Figure 9). In analysing individual species change in reporting rates, significant increases were found for four species, significant decreases for three species, non-significant increases for five species and non-significant decreases for one species (Figure 10; Table 3).

No significant difference was detected for the level of decline of combined avian prey inside and outside of protected areas, as evidenced by the non-significant interaction between protected area status of a QDS and period (Table 3).

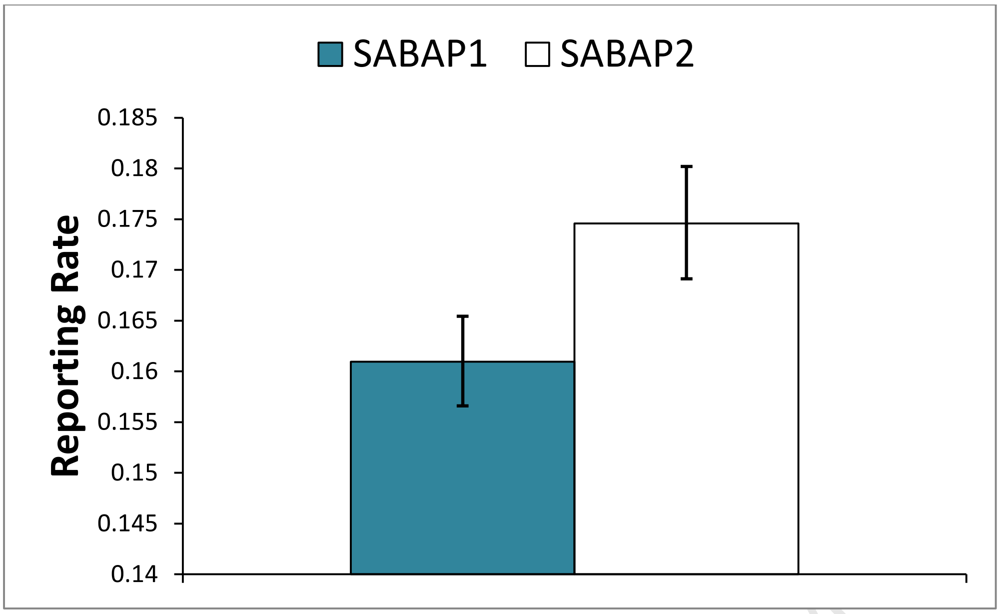


Figure 9: The overall difference in South Africa in the probability of reporting the combined avian prey species between the SABAP projects (mean ± 95% confidence limit).

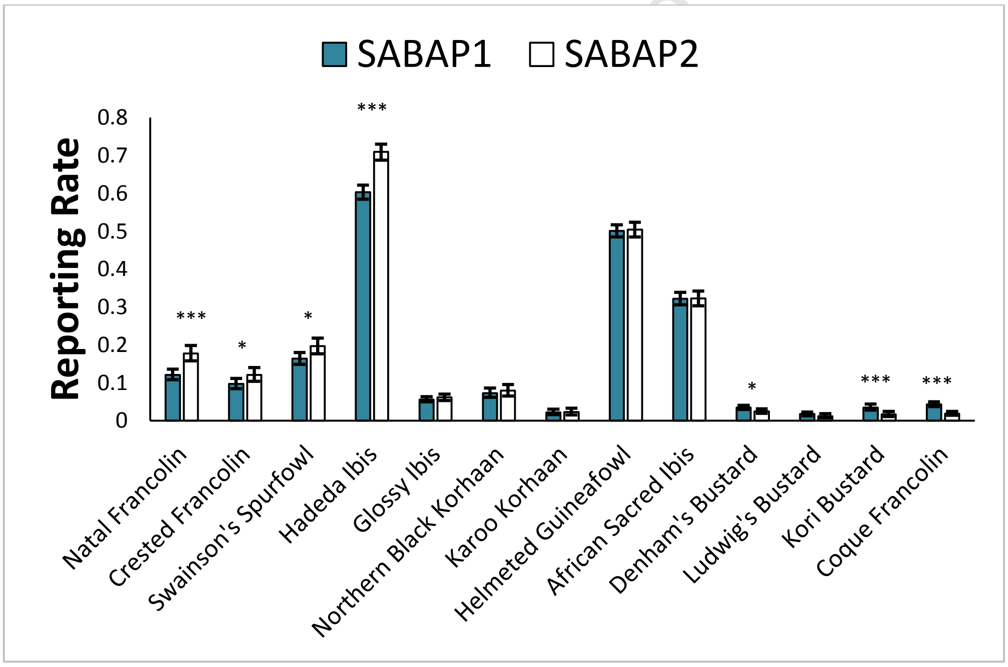


Figure 10: The overall difference in South Africa in the probability of reporting each of the thirteen ME avian prey species used in the study between the SABAP projects (mean ±95% confidence limit). Significant changes are marked as follows: *p<0.05, **p<0.01, ***p<0.001.

Table 3: Outputs from the generalised linear model 1 exploring changes in reporting rates. The Period χ^2 values for avian prey overall (combined species and individual species) relate only to whether there was a significant change in reporting rates between the SABAP projects. For within and outside of protected areas (combined species) the Main term χ^2 values (whether there is a significant change in ME reporting rates between within and outside protected areas) and the Interaction χ^2 values (whether there is a significant change in ME reporting rates between the SABAP projects for the Main term categories) are further reported. Significant changes are marked as follows: .p<0.1, *p<0.05, **p<0.01, ***p<0.001.

	Main Term	Significance			χ^2						Intercept
		Period	Main Term	Interaction	Period	Df	Main term	Df	Interaction	Df	
Combined	National	***	-	-	14.4	1	-	-	-	-	-1.7
	Protected Areas	**	.	NS	9.7	1	3	1	1.1	1	-1.6
Individual	African Sacred Ibis	NS	-	-	0.001	1	-	-	-	-	-0.7
	Coqui Francolin	***	-	-	30.6	1	-	-	-	-	-3.1
	Crested Francolin	*	-	-	4.3	1	-	-	-	-	-2.2
	Denham's Bustard	*	-	-	5.6	1	-	-	-	-	-3.3
	Glossy Ibis	NS	-	-	1.1	1	-	-	-	-	-2.8
	Helmeted Guineafowl	NS	-	-	0.6	1	-	-	-	-	0.01
	Hadeda Ibis	***	-	-	51.6	1	-	-	-	-	0.4
	Kori Bustard	***	-	-	11.6	1	-	-	-	-	-3.3
	Karoo Korhaan	NS	-	-	0.01	1	-	-	-	-	-3.8
	Ludwig's Bustard	NS	-	-	2	1	-	-	-	-	-4
	Northern Black Korhaan	NS	-	-	0.4	1	-	-	-	-	-2.5
	Natal Francolin	***	-	-	21.1	1	-	-	-	-	-2
	Swainson's Spurfowl	*	-	-	6.2	1	-	-	-	-	-1.6

2.3.3 Changes in reporting rates in relation to environmental covariates

Next I explored how ME reporting rates changed in relation to a range of covariates and controlling for any potential spatial autocorrelation in the analysis. The full model indicated significant negative associations between power line density (Figure 11), increases in temperature (Figure 12) and decreases in avian prey abundance (Figure 13); no significant association was found with changes in rainfall (Table 4).

The negative relationship with power line densities indicated greater declines in QDSs with higher power line densities than in QDSs with lower densities (Figure 11; Table 4). Including the distribution and transmission power line categories separately in the full model showed that the relationship was best explained by the smaller distribution lines (Intercept=-

14.3, Parameter Estimate=-0.001, χ^2=38.1, DF=1, p<0.001) rather than the transmission lines. A positive relationship was found with the transmission lines, however, showing that there was less of a decline in QDSs with a higher density of transmission lines (Intercept=-14.3, Parameter Estimate=0.002, χ^2=14.3, DF=1, p<0.001).

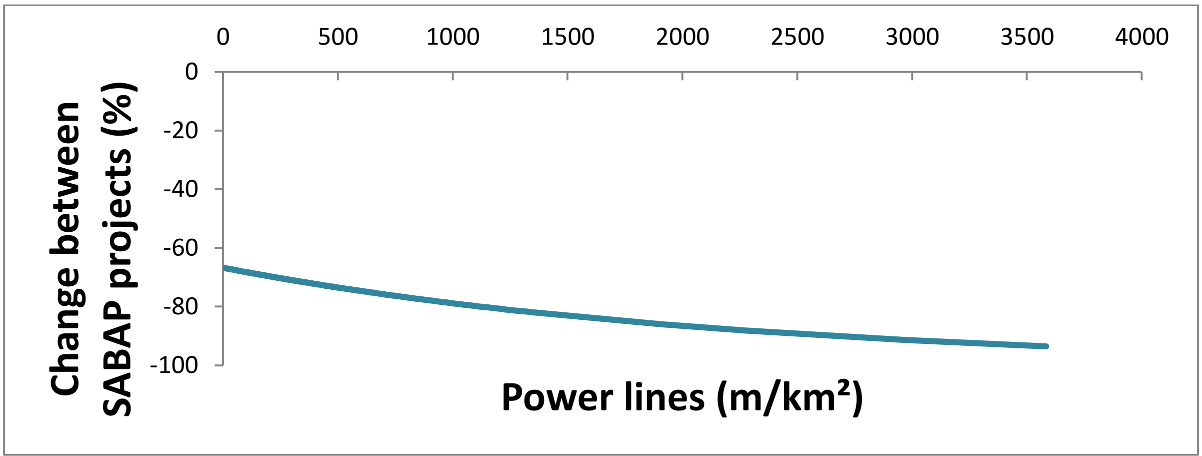


Figure 11: Plotted line of relationship between power lines and ME changes in reporting rates between the SABAP projects. A significant negative relationship is found, with MEs declining with higher power line densities.

The negative association with temperature changes indicated that declines were greater in QDSs which became relatively warmer between 1987 and 2010, than in QDSs that became relatively cooler (Figure 12, Table 4).

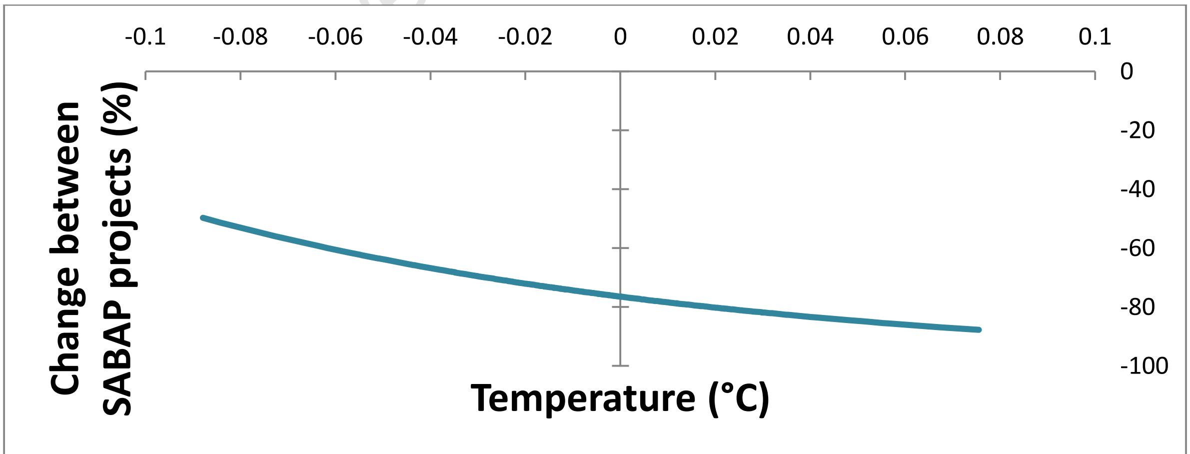


Figure 12: Plotted line of relationship between changes in temperature between 1987 and 2010 and ME changes in reporting rates between the SABAP projects. A significant relationship is found, with decreases in MEs found in areas that had become relatively hotter.

The relationship with avian prey change indicate that declines were greater in QDSs where avian prey was decreasing, than in QDSs where avian prey was increasing (Figure 13, Table 4). The full model therefore suggests that power lines, temperature and avian prey are correlated with changes in ME reporting rates (Table 4). Although significant results were found, this full model only explains about 11% of the deviance in the changes in ME reporting rates.

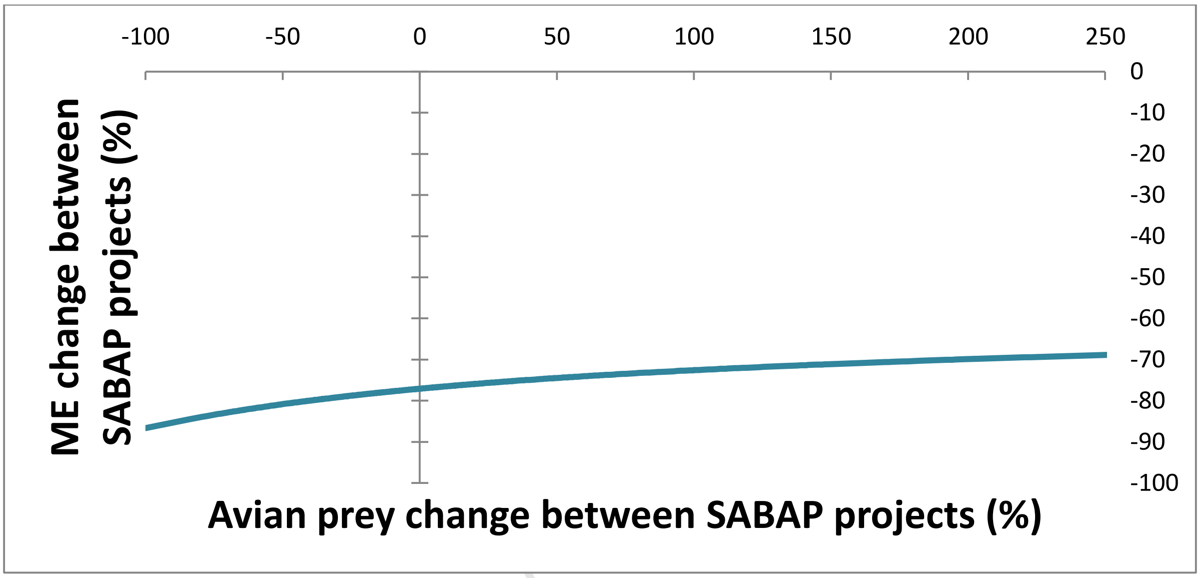


Figure 13: Plotted line of relationship between avian prey changes in reporting rates between the SABAP projects and ME changes in reporting rates between the SABAP projects. A significant relationship is found with less of a decrease in MEs as avian prey increases.

Table 4: Output from the generalised linear model 2 with the full model fitted. The intercept for the full model is indicated in the Power line row and is relevant for all the variables. The χ^2 values relate to whether there was a significant relationship between each variable and the change in ME reporting rates between the SABAP projects. To account for spatial auto-correlation, which might be present, latitude (X) and longitude (Y) and their interaction were fitted as fixed effects in the model. Significant changes are marked as follows: .p<0.1 *p<0.05, **p<0.01, ***p<0.001.

Main Term	Main term significance	χ^2	Df	Intercept	Parameter estimate
Power line	***	18	1	13.9	-0.0004
Precipitation	NS	0.1	1	-	-0.04
Temperature	*	5.2	1	-	-6.4
Avian Prey	**	7.6	1	-	1.6
X	.	3.2	1	-	0.3
Y	*	4.1	1	-	-0.3
X:Y	NS	1.4	1	-	0.01

For a subset of the data including only QDSs from Kruger National Park (KNP), I examined the changes in reporting rates in relation to Elephant densities (with no spatial terms) (Figure 14). I found a significant association between decreases in the reporting rates of MEs in KNP and increases in the average density of Elephants per QDS between 1987 and 2010 (Intercept=-0.1, Parameter Estimate=-1.2, χ^2=7.7, DF=1, p<0.01, Figure 15).

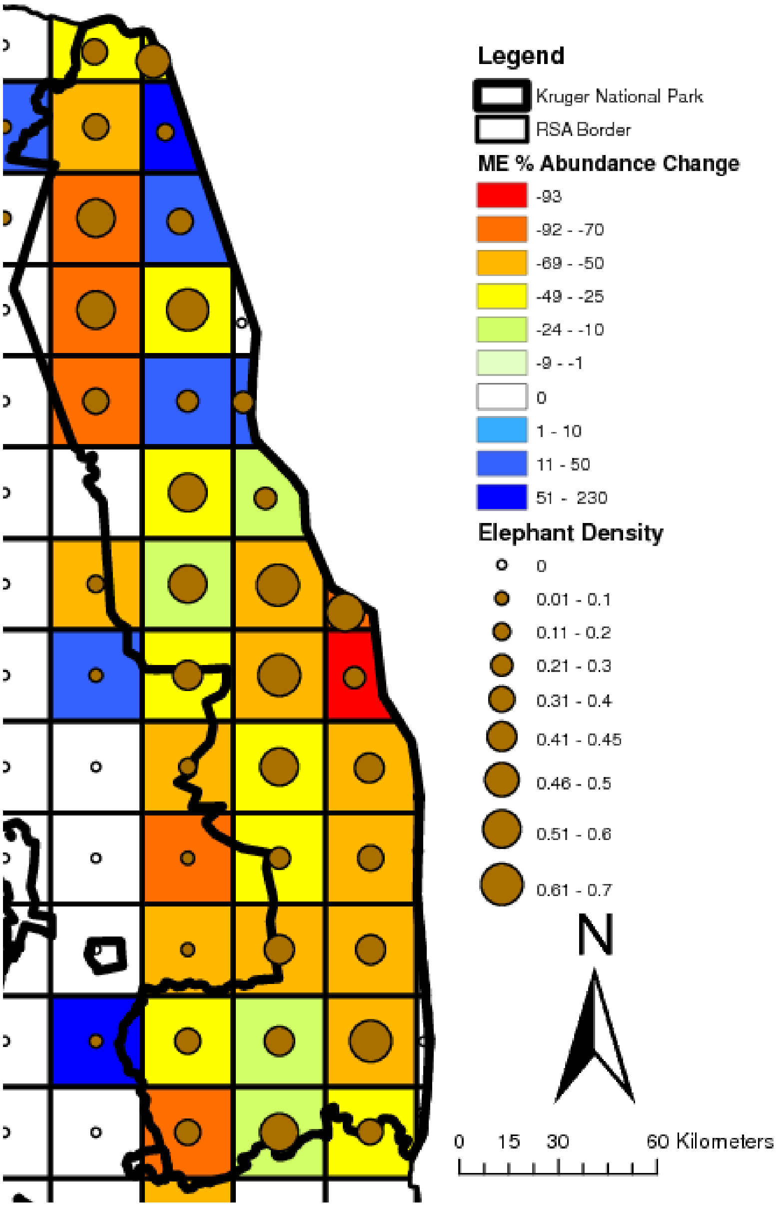


Figure 14: Map showing the average density of Elephants per QDS over the 1987 - 2010 period in Kruger National Park. Small circles indicate low average densities ranging through to large circles that indicate high average densities. The average density symbols are overlaid on the ME changes in reporting rate indicated as percentage change for each QDS. Negative changes ranges from green through to red while positive changes range through shades of blue.

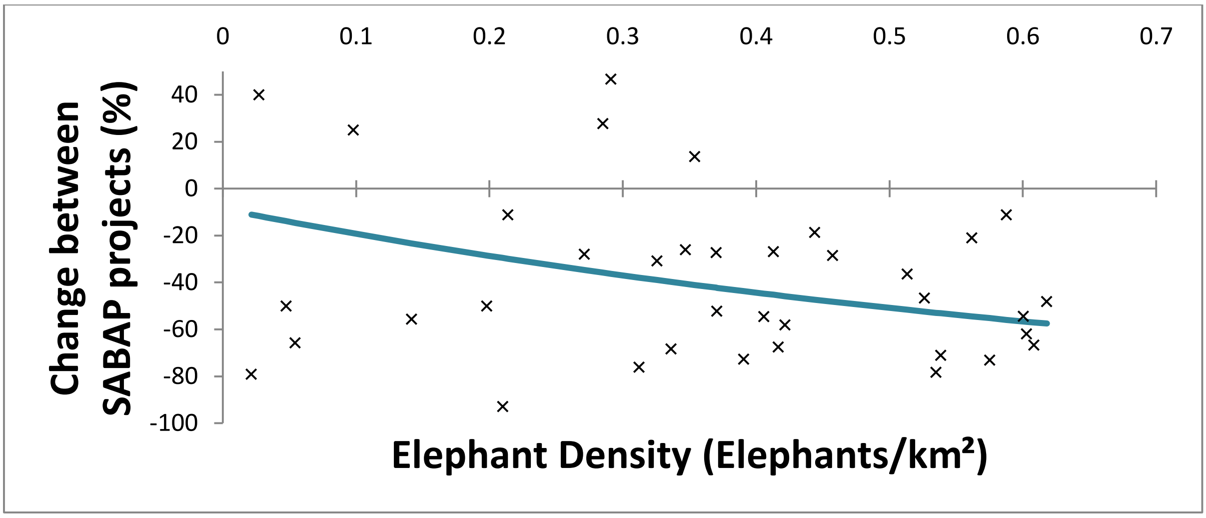


Figure 15: Plotted line of the relationship between the average density of Elephants in Kruger National Park and ME changes in reporting rates between the SABAP projects. A significant negative relationship is found with MEs decreasing in areas where the average density of Elephants increased.

2.3.4. Martial Eagle current abundance

South Africa comprises of 16,829 pentads of which 10,717 (63%) had been surveyed at least once as part of SABAP 2 by August 2012. From these 10,717 pentads, 841 (7.9%) had MEs recorded at least once. Of the pentads with MEs, 33.7% (n=283) showed a reporting rate of between 0.1-10% , 24.9% (n=209) between 10.1-20%, 11.9% (n=100) between 20.1-30%, 8.2% (n=69) between 30.1-40%, 9.3% (n=78) between 40.1-50%, 1% (n=8) between 50.1 and 99.9% and 11.2% (n=94) had a 100% reporting rate (Figure 16, Figure 17)

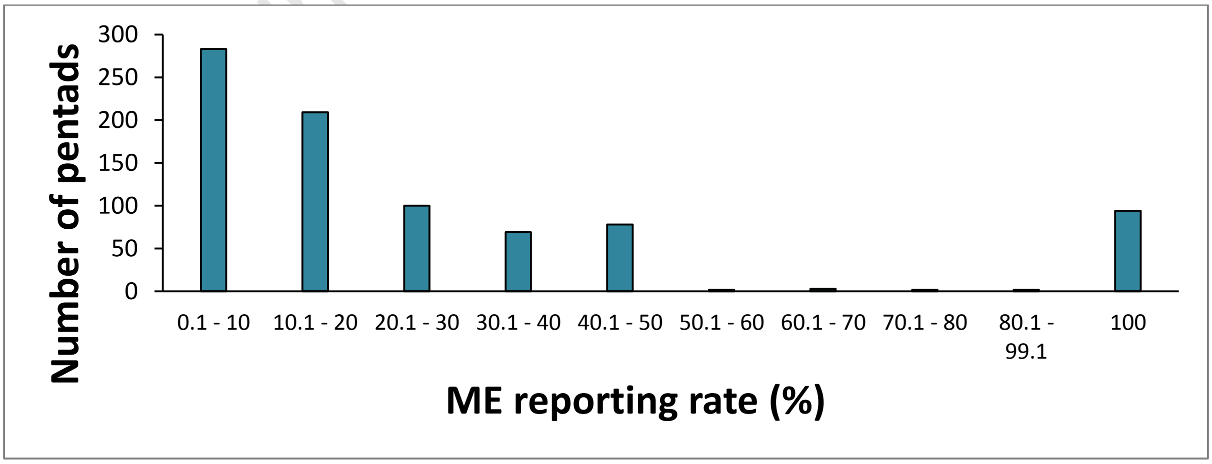


Figure 16: Graph illustrating the distribution of ME reporting rates among pentads with SABAP 2 cards submitted. The high number of pentads with a 100% reporting rate is a result of a high proportion of pentads (with MEs recorded) with only a low number of cards submitted.

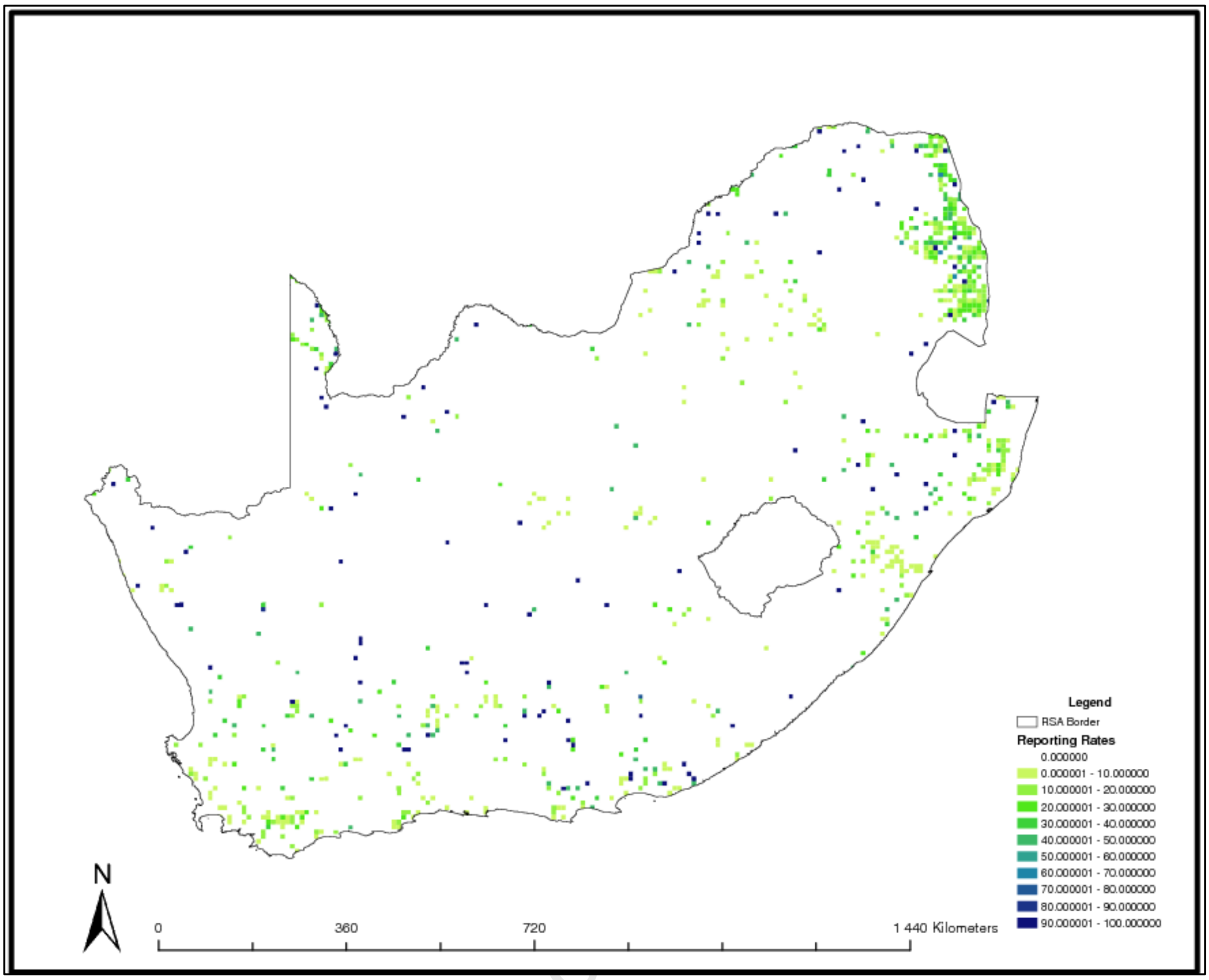


Figure 17: Map showing the SABAP 2 reporting rates of MEs in South Africa ranging from lime for the lower rates, through green to blue for the higher rates. Only pentads where MEs have been reported are indicated on the map.

Comparing pentads with and without ME sightings (model 3), it was found that there were significant differences in the probability of sighting a ME at least once between biome types (Figure 18A, Table 5). The Grassland biome had the lowest probability (mean=0.03, ±95% confidence limits = 0.01 - 0.06) and differed significantly from all other biomes (Figure 18A).

Comparing the relative abundance (model 4) between biomes, also revealed significant differences (Figure 18B, Table 5). MEs again had the lowest relative abundance in the Grassland biome (0.3, 0.2 - 0.3) and differed significantly from all other biomes except for the Succulent Karoo whilst MEs has the highest relative abundance in the Savanna biome (2, 1.9 - 2.2) (Figure 18B). Examining the relative abundance, using only pentads with MEs present, I also found significant differences (Figure 18C, Table 5). The Savanna biome (5.6, 5 - 6.3) again had the highest relative abundance but only differed significantly from the Albany Thicket and Grassland biomes (Figure 18C).

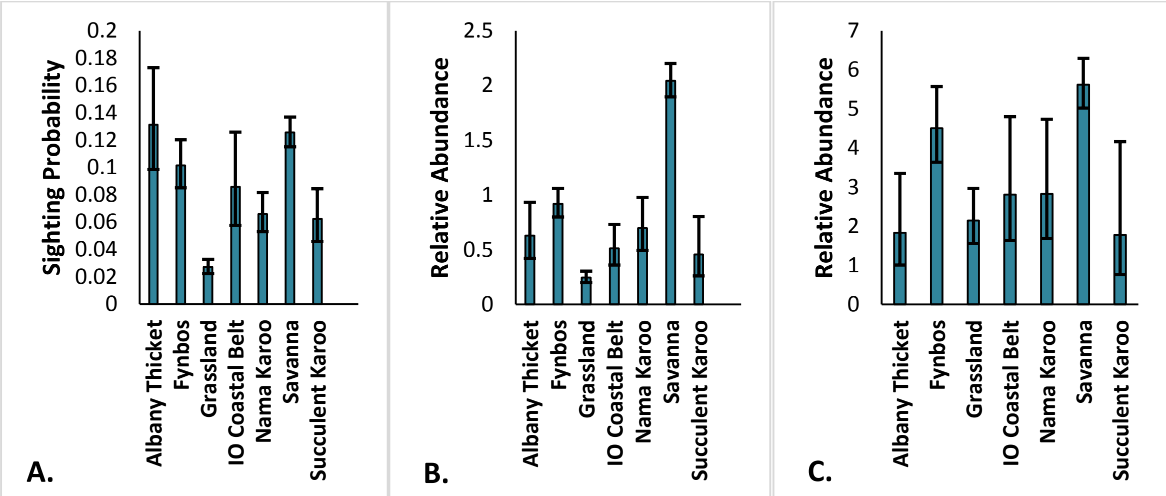


Figure 18: Sighting probability [A.], relative abundance [B.] and relative abundance (model 4 variation) [C.] of MEs across the dominant biomes in South Africa for SABAP 2. Mean values are indicated with ± 95% confidence limits. IO = Indian Ocean

In exploring whether sighting probability or abundance was influenced by the density of power lines it was found in all analyses that there were negative associations (Figure 19, Table 5).

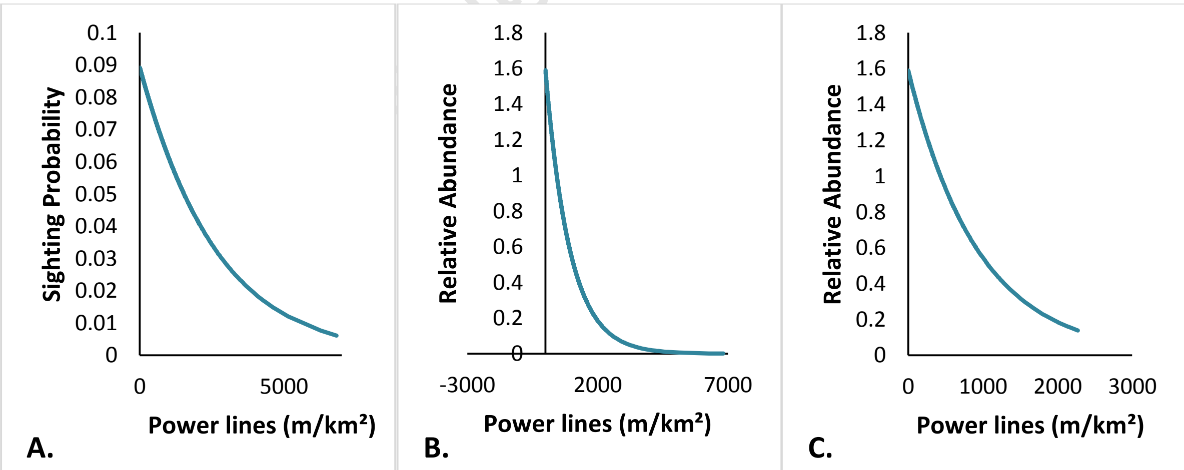


Figure 19: Plotted lines of the relationships between power lines and sighting probability [A.], relative abundance [B.] and relative abundance (model 4 variation) [C.] of MEs for SABAP 2. Power lines showed a significant negative relationship with all measures used.

The sighting probability and relative abundance was significantly higher inside than outside of protected areas (Figure 20, Table 5). There was 5.1 times less change of sighting MEs outside of protected areas compared to inside with a drop from 0.29 (0.26 - 0.32) sighting probability to 0.058 (0.054 - 0.063) (Figure 20A). The relative abundance is also 6 times lower with a drop from 3.49 (3.1 - 3.7) in relative abundance to 0.56 (0.51 - 0.62) (Figure 20B). Even within pentads where MEs were recorded, relative abundance was 2.1 times lower outside of protected areas compared to inside with a drop from 6.5 (5.69 - 7.42) in relative abundance to 3.15 (2.74 - 3.62) (Figure 20C).

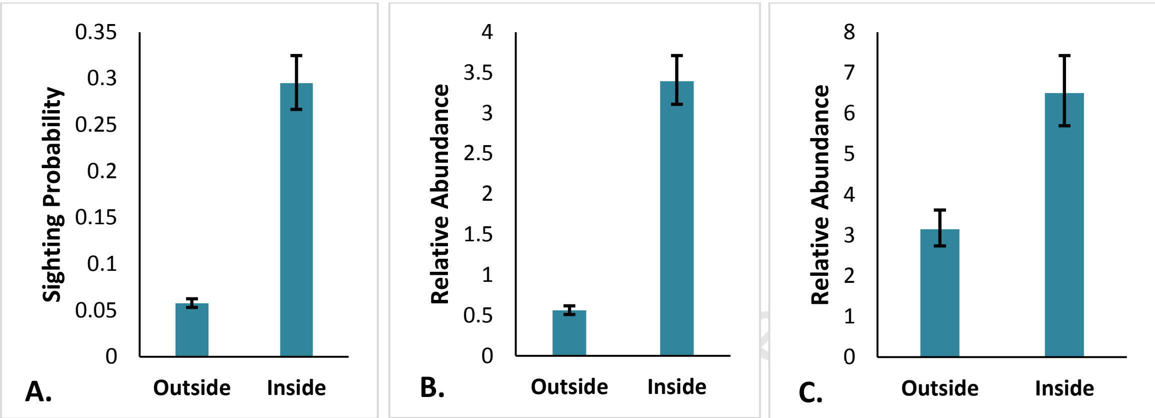


Figure 20: Sighting probability [A.], relative abundance [B.] and relative abundance (model 4 variation) [C.] of MEs outside and inside of protected areas in South Africa for SABAP 2. Mean values are indicated with ± 95% confidence limits.

Table 5: Outputs from the generalised linear models 3 and 4 (including the model 4 variation) exploring the sighting probability and relative abundance of MEs between biomes, in relation to power line densities and inside and outside of protected areas. Significant changes are marked as follows: *p<0.05, **p<0.01, ***p<0.001.

	Main Term	Main term significance	χ^2	df	Intercept	Parameter estimate
Model 3	Biomes	***	290.3	7	-1.9	-
	Power lines	***	15.9	1	-2.3	-0.0004
	Protected areas	***	446.6	1	-2.8	-
Model 4	Biomes	***	615.1	7	-0.5	-
	Power lines	***	228.7	1	0.5	-0.001
	Protected areas	***	683.3	1	-0.6	-
Model 4 Variation	Biomes	***	64.9	6	0.6	-
	Power lines	***	42.1	1	1.8	-0.0008
	Protected areas	***	54	1	1.2	-

2.4. Discussion

My analysis revealed that the Martial Eagle (ME) has dramatically declined in South Africa over the last 20 years, with field data from one area further confirming the scale of these declines. Greatest correlational support was found for the drivers of declines being linked to increasing temperatures and power line densities.

2.4.1. Population changes

The large declines of around 60% detected for MEs over the last 20 years in South Africa are worrying for a species with the life history traits displayed by MEs. These broad scale declines appeared to occur at a similar magnitude throughout the provinces, but differed between biomes, with similar declines found in most biomes, with the exception of Albany Thicket, Succulent Karoo and Fynbos biomes where declines were less severe. The declines detected in this study are similar to those declines found elsewhere in Africa for MEs and other large raptors over recent decades, where direct and indirect impacts from the growing human populations were seen as being the drivers of decline (Brandl et al. 1985; Herremans and Herremans-tonnoeyr 2000; Hudson and Bouwman 2007; Thiollay 2007a,b; Virani et al. 2011).

My analysis examining the variables influencing the species' contemporary abundance suggested that the lowest probability of sighting MEs, and where they also have the lowest relative abundance, is in the Grassland biome, whilst the relative abundance for MEs is highest in the Savanna biome. This reflects the known habitat preferences for MEs in that even though they occur widely, they prefer the Savanna biome as habitat and are found less in the Grassland biome due to a lack of nesting opportunities (Brown 1970; Steyn 1982; Boshoff 1997; Ferguson-lees and Christie 2001; Simmons 2005).

Although my study found that declines inside protected areas were less severe than outside of protected areas, it is still of great concern that MEs also underwent large declines, of over 40%, inside protected areas over the last 20 years. Thus, it does not appear that protected areas are able to fully safeguard this species against the pressures being exerted from outside the protected areas, and the effect of pressures inside protected areas e.g. Elephant and other species' overbrowsing and overgrazing (Trollope et al. 1998, this study). It does suggest, however, that protected areas do offer a buffer from the main pressures driving the declines. These findings relate to the findings from Herremans and Herremans-tonnoeyr (2000) in Botswana where large raptors, such as MEs, with large territories (van Zyl 1992) that extend outside of protected areas, may still be exposed to human related threats. Although protected areas have been seen as strongholds for MEs (and other large raptors) in Africa (Herremans and Herremans-tonnoeyr 2000; Thiollay 2006, a,b; 2007a,b) it appears that protection may not be adequate in South Africa (and probably throughout its range) and that urgent effort should be made to better understand the reasons behind the declines both outside and inside of protected areas. Despite these declines, the species is still encountered around five times more frequently within protected areas, and in areas where they do occur they are also twice as abundant within protected areas compared to outside of these areas, suggesting that protected areas have provided some degree of safety from the declines and that habitats are still preferable within these areas than in areas outside protected areas.

The strong agreement in the trend of decline found for the Kalahari National Park from the South African Bird Atlas Project (SABAP) analysis and the intensive field monitoring substantiates the SABAP analysis changes found. Although it does not quell all the concerns expressed with regards to comparing the SABAP data due to the change in protocol (Bonnevie 2011), it gives some increased confidence in the findings from this project.

MEs are currently listed as globally *Near Threatened* (BirdLife International 2012) and regional as *Vulnerable* (Barnes 2000). However, these statuses are currently being reassessed both globally and regionally. Regionally, the species will be up listed to *Endangered* (M. Taylor pers. com.), and the declines reported in this study strongly support this regional up listing. My results also provide some support for the global up listing of this species and the information will form part of the decision process for this consultation process.

2.4.2. Correlates of change

Some support was found for two of the hypotheses on the causes of these ME declines from my analysis on the correlates of change with climate change (**H1**), specifically change in temperature, and power line densities (**H3**) being negatively associated with the declines. Although in my analysis a relationship with avian prey abundance change was found (**H2**), with declines being greatest in areas with largest avian prey declines, it was unlikely that prey declines have been important in the widespread national declines of the species because overall across the country avian prey abundance generally increased rather than declined in contradiction of one of **H2**'s predictions. This relationship may though be important on a local scale.

As predicted (**H1**) declines were greater in areas where climate has changed the most (specifically where the temperature has increased the most), suggesting that increases in temperature may be negatively affecting some aspect of ME fecundity and/or survival. Overall, as predicted, within South Africa climate has changed over the last 20 years and visually areas of increases and decreases in temperatures are about even (Refer to Appendix A – Figure 25). The mechanisms involved in this relationship are, however, less clear. For some species e.g. the southern Pied Babbler (*Turdoides bicolor*) higher temperatures can result in less efficient feeding since more time is devoted to regulating their temperature and as a result they can lose body condition which can result in fitness costs; for example lower productivity (du Plessis et al. 2012). The northern Goshawk (*Accipiter gentilis*) in Sweden was found to have advanced hatching times with warmer earlier spring temperatures and had larger clutches with higher temperature and lower precipitation during laying and brooding (Lehikoinen et al. 2013).

Breeding success of this Goshawk population is expected to increase with predicted climate change increases in winter and spring temperatures (Lehikoinen et al. 2013). The Black Kite (*Milvus migrans*) in Italy was found to increase its frequency of hunting and had a higher success rate with increases in temperature and decreases in rainfall (Sergio 2003). Higher temperatures in the days prior to laying resulted in higher breeding success as males were more successful in hunting and females were in better condition to lay eggs (Sergio 2003). These different studies on the effect of temperature demonstrate that although weather and changes in climate can impact raptors and other bird species either positively or negatively, the exact mechanism is important to understand in order to manage the species effectively. This kind of understanding is currently completely lacking for MEs.

As predicted, MEs were found to decline most in areas with the highest power line densities (**H3**). This finding indicates that the known, but admittedly under reported collisions and electrocutions of MEs (Kruger, R. pers. com) are potentially a driver for the decline of the South African ME population. This finding is further strengthened by the finding that the current sighting probability and relative abundance of MEs is lower in areas with higher power line density. Dispersing immature or sub-adult eagles may be at most risk from power lines, as was found to be the case for the Golden Eagle (*Aquila chrysaetos*) in North America (Lehman et al. 2007) and also to a degree for Bonelli's Eagle (*Aquila fasciata*) in Spain and has been ascribed to the fact that inexperienced birds are less adept at landings and take-offs (Real et al. 2001). However, it must be recognized that these relationships are correlational and therefore are only suggestive of a potential causal mechanism involved in the decline. One concern with this finding is that any such relationship might actually be attributable to other variables which are associated with power line networks. For example, one suggestion was that power lines are less abundant in the former apartheid homelands where persecution might actually be lower. However, I found no evidence of this, for there was no obvious differences between power line densities inside or outside of these areas.

A further analysis of the different sizes of power lines, showed that this negative relationship was driven by the smaller power lines (i.e. the distribution lines <33kV) which form the majority of the power lines (c 80%) and are also the power line type for which most deaths are recorded (Kruger, R. pers. com.). Indeed analysis using only the larger transmission lines showed a positive relationship, which might be expected given their importance as nesting structures (Machange et al. 2005). Overall, therefore, there appears good evidence to suggest that power lines are having a negative impact on MEs, and as such it should be examined further and all efforts should be made to ensure that the risk from power lines for MEs are reduced.

Unlike my predictions (**H2**), avian prey abundances were found to have increased nationally and the level of these increases were found to be similar inside and outside of protected areas. Therefore, although ME declines were found to be the greatest in areas with largest avian prey declines, it is highly unlikely that declines in avian prey abundance are key driver of ME declines nationally as **H2** predicted. On a local scale, prey abundance can though be seen as being a driver of decline as the relationship shows that where avian prey species have decreased most, MEs have also decreased more. Visually inspecting the change in prey abundance map (Appendix A – Figure 26), it appears that substantial declines have taken place over large parts of KwaZulu-Natal as well as part of the southern/south-western Cape, Mpumalange and Limpopo. In the KwaZulu-Natal midlands the loss of a habitat mosaic with associated edge habitat to more intensive agricultural land-use patterns in combination with increased pesticide use contributed to the decline of overall birds species diversity and specifically the Helmeted Guineafowl (*Numida meleagris*) (Malan and Benn 1999; Ratcliffe and Crowe 2001), one of the preferred prey species of MEs (Simmons 2005). The decline of large eagles in West Africa was, in part, attributed to the decline in large bird prey (Thiollay 2006b) and indicates that a loss of prey should not be completely discounted as a possible driver for ME declines, especially as only avian prey was investigated in this study and not also their mammalian and reptilian prey.

Declines were found to have occurred inside all of the protected areas in South Africa, including Elephant free reserves such as the Kalahari National Park. It was, however, considered a possibility that increases in Elephants may have been a driver for declines in some protected areas, particularly in Kruger National Park, which is the stronghold for the species nationally, and probably supports at least 15% of the national population (Steyn 1982; Tarboton and Allan 1984). In support of this hypothesis (**H4**), declines were found to be associated with higher densities of Elephants in Kruger National Park as was predicted. The mechanism for this relationship is thought to be linked to Elephants reducing nesting opportunities for MEs and other raptors, a concern raised by Monadjem and Garcelon (2005) or by changing habitat structure more generally (Trollope et al. 1998; Dean et al. 1999; Whyte 2001; Young et al. 2009). Nest site limitation of raptors is known from a number of studies (Lõhmus 2003; Dixon and Batbayar 2010; Ibarra et al. 2012) and habitat changes are also known to influence prey abundance and breeding performance (Whitfield et al. 2007; Thorup et al. 2010; Amar et al. 2011; Ibarra et al. 2012). Elephant numbers in Kruger National Park have nearly doubled over the last 20 years (Whyte 2007). With the moratorium on Elephant hunting still in place (Owen-Smith et al. 2006), the Elephant density in Kruger National Park is expected to continue to increase. Unless the moratorium is lifted, it can be expected that the negative effect of Elephants on nesting trees and/or habitat and thus on MEs and other raptors in Kruger National Park will continue and probably increase.

Even though there is some support for two of the hypotheses proposed for the national decline of MEs, temperature change (**H1**) and power lines (**H3**), and on a smaller spatial scale for avian prey abundance change (**H2**), it is important to recognise that an additional plausible hypothesis, persecution by farmers, could not be tested in this study due to a lack of adequate data. Direct (e.g. shooting) and indirect (e.g. poisoning of mammalian predators) persecution has been shown in various raptor studies worldwide to have been a substantial cause of death (Lehman 2001; Real et al. 2001; Whitfield et al. 2004a,b; Margalida et al. 2008) and a potential driver of decline and as such it is strongly suggested that it may also be an additional driver (and perhaps the main driver) of decline for MEs in South Africa. Indeed given that my full model explained a relatively small proportion of the variance (i.e. around 11%), it is highly likely that other unexplored variables are also contributing to the declines of this species. Current research focused on the Kruger National Park will explore survival rates of dispersing juvenile and sub-adult birds (van Eeden, R. pers. com.) to specifically explore whether high levels of persecution are resulting in overall low levels of recruitment and, ultimately, declines in the populations (Penteriani et al. 2005)

CHAPTER 3: Study Review and Synthesis

3.1. Conclusions

The population size of the Martial Eagle (ME) is always going to be difficult to assess, as the species has a wide range, maintains large territories and occurs at low natural abundance (Barnes 2000; Ferguson-lees and Christie 2001). Determining population changes will also therefore present difficulties, even when real concerns are expressed that this has occurred (Barnes 2000; Underhill 2012). The aim of this study was to determine the extent of ME declines in South Africa and to see if any broad causes for these declines could be identified.

The most important finding from this study was that MEs have declined dramatically and in a relatively uniform manner throughout South Africa. A great concern is that these declines have also occurred (albeit to a lesser extent) within protected areas including the Kruger National Park and the Kalahari National Park, which are both seen as strongholds for MEs. Strong independent evidence was found in the Kalahari National Park that the declines found through the South African Bird Atlas Project (SABAP) analysis were real. This finding is important and suggests that declines detected in the comparison of the two SABAP surveys are unlikely to simply be the result of changes in protocols. Furthermore, my finding that in almost 50% of quarter-degree grid cells, the species had completely disappeared further reinforces this conclusion. Taken together, these two findings lend confidence to the results from this study.

From the hypotheses proposed as potentially driving the decline, some evidence was found to support two potential overall drivers of decline. 1) In areas where the temperature has shown the greatest relative increase over the last 20 years MEs are declining more, although the mechanism for this decline is not yet clear. 2) In areas with higher densities of power lines MEs are declining more, probably due to collisions and electrocution. Avian prey abundance changes over the last 20 years do not appear to be an overall driver for ME declines, because even though a relationship was found, the overall abundance of these avian prey species has actually increased. However, at a local scale, avian prey may well have had an influence on the declines (e.g. Kwa-Zulu-Natal where they seem to have declined) given that MEs were found to have decline more in the areas where prey has also declined. In Kruger National Park, where MEs should be protected from many threats faced by individuals outside of protected areas (e.g. power lines, persecution etc.), higher densities of Elephants were related to larger declines in MEs, probably as a result of a reduction in nesting sites or changes in habitat quality.

It is important to note that the aim of this study was not to provide a definitive test of the factors driving the declines of MEs but rather provide an initial attempt to assess the broad drivers and mechanism of decline, and to prioritize and guide further focused and detailed research (Amar et al. 2011). As such the findings of this study confirm the concern with regards to the decline of MEs in South Africa. The reasons proposed and supported for the decline of MEs in South Africa are either directly or indirectly linked to the growing human population and the associated pressures (Barnes 2000; Simmons 2005; Owen-Smith et al. 2006; Goudie 2009). To ensure the continued survival of MEs in South Africa focused field research is urgently required to better understand the causes of decline and the mechanism through which MEs are being impacted so that pragmatic conservation actions for the species can be implemented.

This study represents one of the first attempts to assess nationally, using empirical data, the likely drivers of decline for a bird species in South Africa. Additionally, it is one of the first studies to explore whether climate change may be influencing range contractions at a very broad scale in an African context. The findings suggest that such an approach as this may be useful in elucidating the likely environmental drivers behind such declines and could be used on other similarly declining species in South Africa, and hopefully in the future in Southern Africa as the SABAP repeat surveys extend to some of these areas (ADU 2013).

3.2. Complications with the study

Some uncertainty has been expressed with regards to the comparison of South African Bird Atlas Project (SABAP) 1 and SABAP 2 due to the changes in protocol (Bonnevie 2011). In this thesis I recognized this as a potential issue and so addressed this concern by comparing the change according to the SABAP projects with changes from independently collected field data for the same area in the Kalahari National Park. Changes in Martial Eagle (ME) numbers and reporting rates were found to be in strong agreement. Although I also searched for other similar independent data sets from other areas, unfortunately none of these three other sources (Hluhluwe-iMfolozi, Central Karoo power lines and Birds in Reserve Project) was suitable to be used. These data sets took considerable time and effort to acquire.

SABAP 2 still has low overall coverage compared to SABAP 1 even after 5+ years. The data used for this study, up to the end of August 2012, only had records submitted for 66% of all pentads in South Africa whilst each Quarter-degree grid cell in RSA, except one, was surveyed during SABAP 1. It is therefore hoped that an improved analysis may be possible in a couple of years' time when better coverage is achieved for SABAB 2 especially as the ME is a species that naturally occurs at low abundance (Ferguson-lees and Christie 2001).

One of the plausible hypothesis, that the decline of MEs in South Africa is due to persecution, could not be tested as I was unable to acquire a suitable data layer (e.g. stock farming data) in the time available to be used for this project. An alternative layer was considered, the National Land Cover Map 2000, but it was found to be inadequate as its categories was too coarse e.g. to show the stock farming areas that I was interested in with this layer strongly mirroring the spatial configuration of the biomes layer. What would have been of more use is specific land use information which is only available for certain selected areas of the country e.g. Western Cape (http://www.elsenburg.com/gis/apps/cfm).

3.3. Future research

Due to the nature and some uncertainty with the data used for this study, the findings should not be seen as a definitive test of the factors driving the declines but rather as an attempt to identify and highlight factors that are of concern with regards to the decline of Martial Eagles (ME) in South Africa. As such, the serious concerns expressed in this study with the decline of MEs, warrant further focused and detailed research. Areas identified through the study and suggested for further detailed field research include; the impact and the mechanism of influence on MEs of power lines and persecution; the availability and stability of suitable prey; the identification of factors within e.g. Kruger National Park that are impacting on MEs; the influence of increasing temperature on ME hunting and breeding success. Most of these areas of research suggested are being addressed by Rowen van Eeden, a PhD candidate at the Percy FitzPatrick Institute at UCT in a current nest monitoring and sub-adult GPS study in Kruger National Park.

A shortcoming of using the South African Bird Atlas Project (SABAP) 2 data is that there has been relatively low participation in some areas, especially in the former homelands and over the large commercial farm areas that are inaccessible to the general public (ADU 2013). Even though this is an area that is actively being addressed by the Animal Demography Unit, it should be prioritized so that the confidence in the use of SABAP 2 data can be strengthened for future projects and so that a true national picture can emerge for South Africa's bird species.

3.4. Management recommendations

A point that was made through many of the sources consulted was that there is not enough protected areas to adequately conserve Martial Eagles (ME) and many other species (Anderson and Kruger 2004). Apart from a few large conservation areas there is not enough conserved habitat for some raptors with large territories (Anderson and Kruger 2004). As such, proactive policies and awareness and education programmes addressing farmers and other stakeholders are strongly recommended as the future of raptor conservation is almost entirely dependent on private landowners (Boshoff and Vernon 1980; Steyn 1982; Tarboton and Allan 1984; Barnes 2000; Anderson and Kruger 2004). Tarboton and Allan (1984) suggest that a priority for conservation action should focus attention on the threats to the species and the ecological role of the species; for example MEs, as apex predators, are sought after by ecotourists and as such they have high ecotourism value (Anderson and Kruger 2004). Sergio et al. (2008) cautioned though against the idea of indiscriminately using an apex predator such as the ME as an umbrella species for biodiversity conservation, especially as it has the potential to generate local conflict e.g. with stock farmers, but if they are used more cautiously they can be used effectively as part of a wider, context-dependent mixed conservation strategy.

With persecution being a concern, Hodkinson et al. (2007) suggested that when considering management of predators, farmers undertaking proactive management, prevention measures and damage controls should aim these at specific problem individuals and should aim to be as selective as possible rather than making use of indiscriminate techniques, such as poison baited carcasses. They further explain that it has been shown that raptors that are known to capture livestock can possibly be rehabilitated by e.g. using a bal-chatri - a wire cage containing live bait (Hodkinson et al. 2007). By baiting the cage using the species that is being predated upon, the caught raptor once released in the same area will associate this species with the unpleasant experience of being caught and will in future refrain from livestock predation (Hodkinson et al. 2007).

Although Eskom, especially in partnership with the Endangered Wildlife Trust, is actively involved in the prevention and mitigation of collision and electrocution of raptors and a range of other species in South Africa, it appears that the impact is still too high and unsustainable for some hard hit species (Shaw et al. 2010; Jenkins et al. 2011). In addition to actively implementing effective mitigation devices and building and replacing existing lines with bird friendly structures, Eskom should further improve its collection records of incidents so that a more accurate understanding can be gained with regards to the impact of power lines, although the planned use of satellite tags on MEs should also aid with this (van Eeden, R. pers. com.).

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5. Appendices

Appendix A: Maps of Environmental Covariates

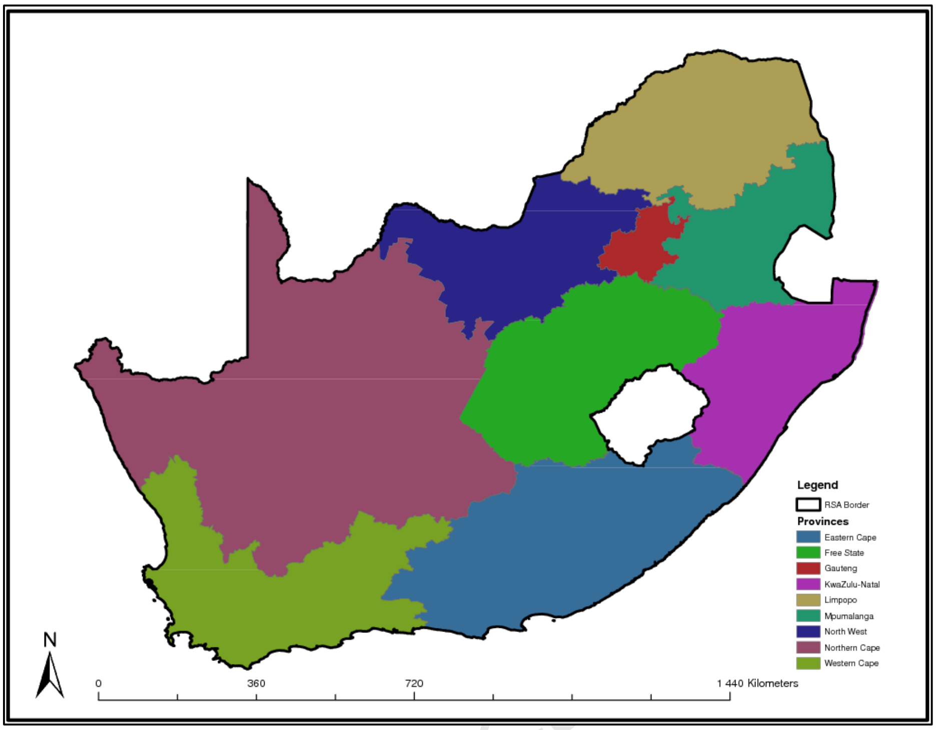

Figure 21: Map of the nine provinces in South Africa. Data were derived from the Municipal Demarcation Board of South Africa (MDB 2013).

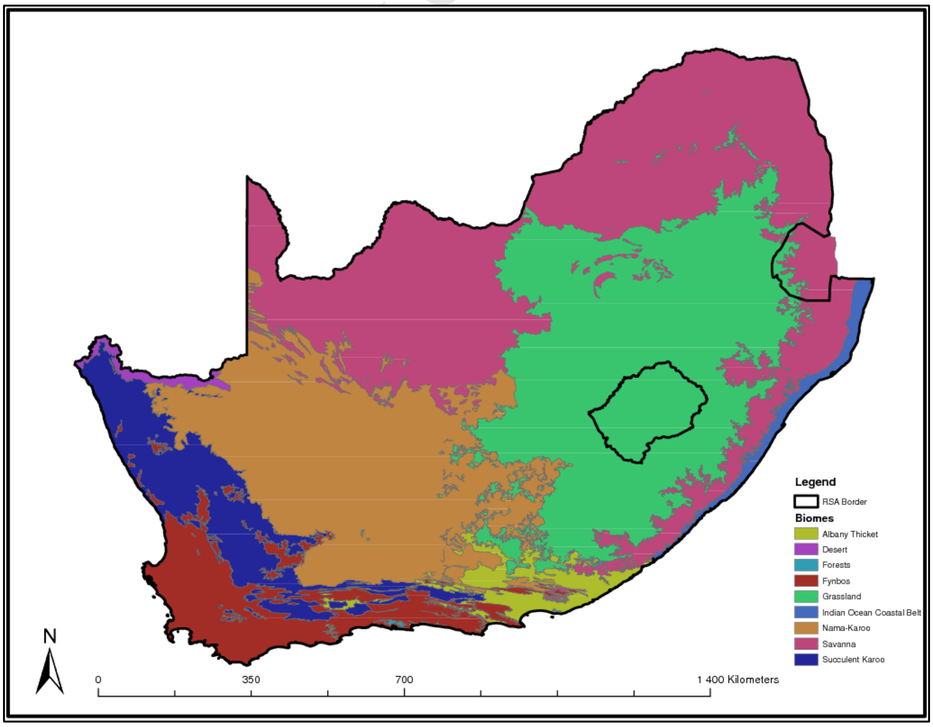

Figure 22: Map showing the distribution of the main biomes in South Africa. Data were derived from Mucina and Rutherford (2006).

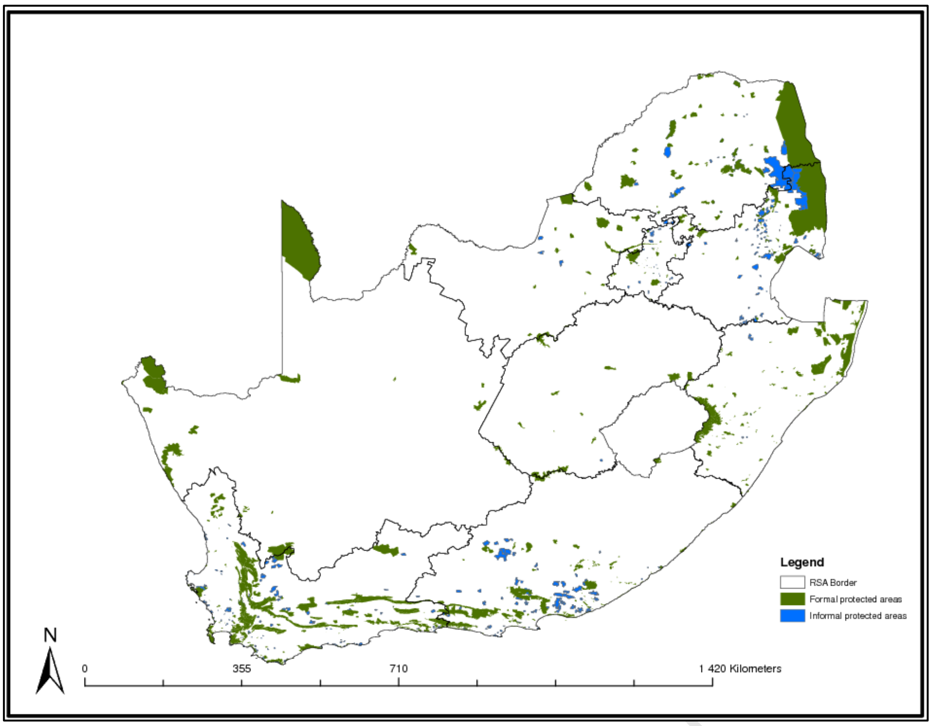

Figure 23: Map showing the protected areas in South Africa. Data were derived from National Protected Area Expansion Strategy 2008 (BGIS 2007b), World Database of Protected Areas (Protected planet 2012) and the 2011 National Biodiversity Assessment (BGIS 2007a).

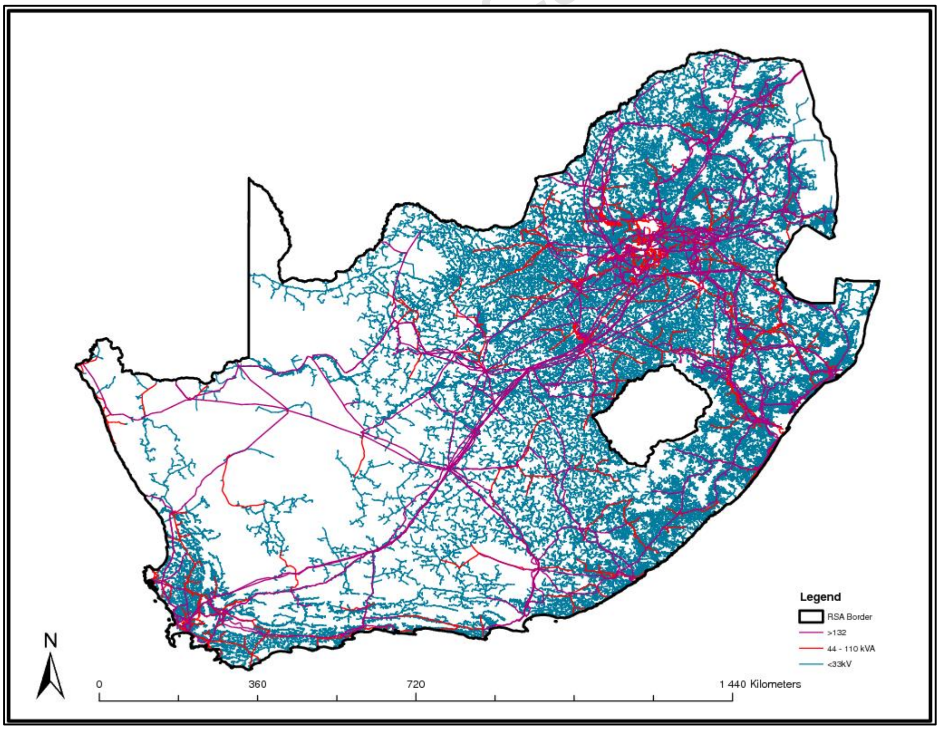

Figure 24: Map of power lines across South Africa. Data were derived from Eskom (2011).

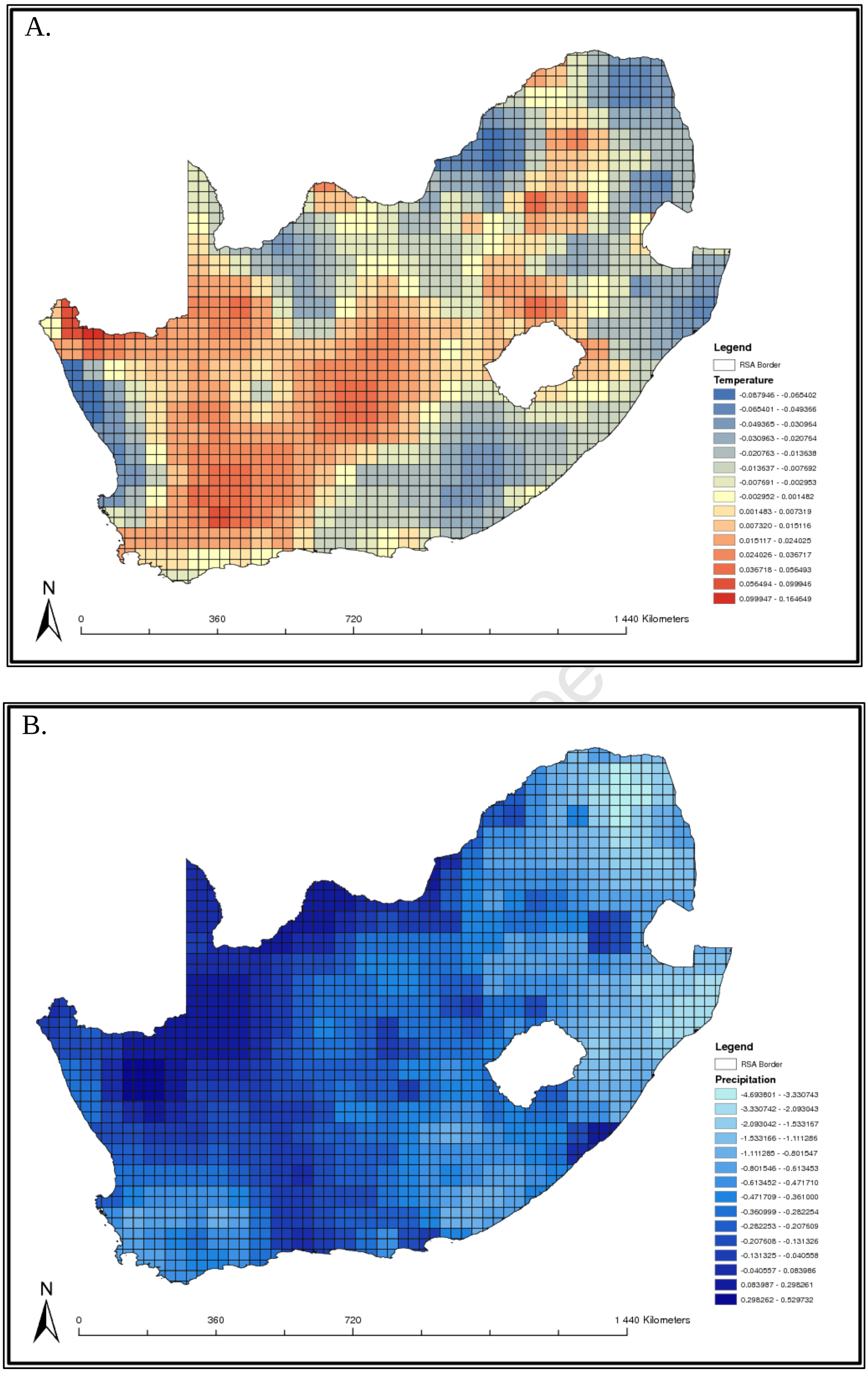

Figure 25: Maps of the trends in Temperature (°C) [A.] and rainfall (mm) [B.] for South Africa from 1987 - 2010. The values for temperature and rainfall for each QDS is derived from the slope over the 23 year period which coincides with the period between the South African Bird Atlas Projects (SABAP). Data were derived from the University of Delaware's Centre for Climate Research (Matsuura and Willmott 2012 a,b).

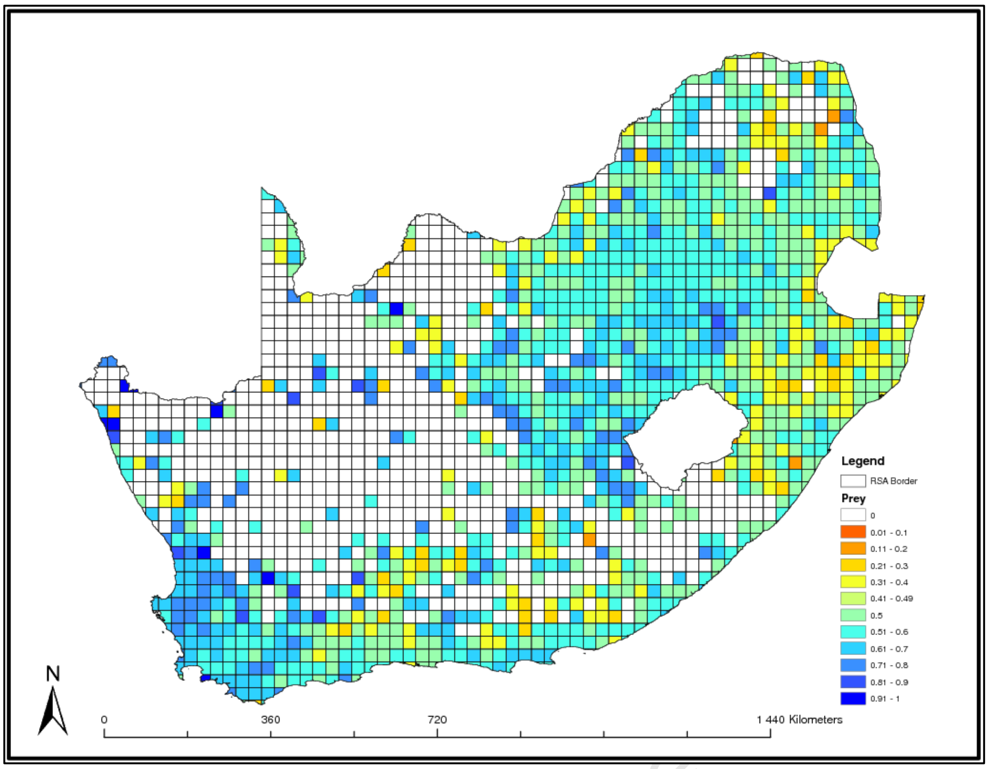

Figure 26: Map of the single measure calculated to representing the change in combined reporting rates for the thirteen medium and large sized South African bird species that are known prey species of Martial Eagles. Data are derived from the SABAP projects data (ADU 2013).

Appendix B: Correlation matrix for full model covariates

Table 6: Correlation matrices for the covariates used in the full model that was used to test the hypotheses proposed. [A] represent the correlation coefficient values and [B] represent the coefficient of determination values. None of the covariates are correlated with any of the other covariates.

[A]	Temperature	Precipitation	Prey
Power lines	-0.0423453	-0.1650742	0.0072013
Temperature	x	0.3567037	0.1475203
Precipitation	x	x	0.2512172
Prey	x	x	X

[B]	Temperature	Precipitation	Prey
Power lines	0.0017931	0.0272495	0.0000519
Temperature	x	0.1272376	0.0217622
Precipitation	x	x	0.0631101
Prey	x	x	x