A CONSERVATION ASSESSMENT OF THE SUNGAZER *(Smaug giganteus)*

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ABSTRACT

The Sungazer (*Smaug giganteus*) is an endemic lizard species that is threatened by habitat destruction and illegal harvesting, and as a result, is listed as 'Vulnerable' on the IUCN Red Data List. The species is restricted to the Highveld grasslands of South Africa, where over 40% of the area is used for crop monoculture, and much of the remainder has been transformed for human habitation and the construction of roads, dams, mines and power plants. This poses serious threats to the persistence of the species, as the Sungazer is a habitat specialist, and is strongly associated with pristine *Themeda* grassland. In addition, the species is illegally harvested from the wild for the traditional medicine, and pet trades. The rate at which these threats are removing habitat and affecting Sungazer populations is unknown, and the lack of such knowledge impedes effective conservation planning. This has prompted the call for research on the population ecology and life history of the species, so that the species can be managed.

Area of occupancy. A minimum convex hull was created around all QDGCs containing species occurrence records, and an Extent of Occurrence (EOO) of 5 833 800 ha was calculated. The distribution of the species (area of QDGCs and portions of QDGCs containing occurrence records that fall within Free State and Mpumalanga Provinces) was calculated as 3 819 600 ha. Of this area, 2 053 035 ha is currently natural. To assess the proportion of EOO and distribution actually occupied by Sungazers, I surveyed 120 random sites for Sungazer presence, and found 5 containing Sungazers (4.17%) within the EOO, and 4 (5.05%) within the distribution. This measure was used to calculate the Area of Occupancy (AOO), which was 103 678 ha.

Population size. I recorded a mean burrow density (MBD) of 6.14 ± 0.87 burrows/ha for 80 sites across the distribution of the species. To estimate the number of burrows within the distribution, I multiplied the MBD by the AOO. I calculated 636 325 ± 90 282 burrows. Burrow occupancy data reported in the literature indicates that only 85.7% of burrows are occupied at a given time, and there is an average occupancy of 1.83 lizards/burrow in these burrows. When applied to the number of burrows calculated, a total figure of 998 247 ± 141 632 lizards is estimated to occupy a total of 545 490 ± 77 395 burrows. Population demographics data reported in the literature indicates that 61.2% of a population is made up of mature (sexually reproductive) individuals, and when applied to the total population size, total mature individual count is 610 927 ± 86 679 Sungazers.

Population decline. I visited 39 sites where Sungazer populations were reported in 1978, and found a population decline of 20.51% at these sites (0.59% decline/year). I assessed the change in land cover between 2001 and 2009 using geographic information systems (GIS) techniques and found a 13.3% decline in natural habitat across the distribution of the species over this time (1.48% decline/year). The loss of natural habitat was due primarily to an increase in cultivated areas.

Priority conservation areas. Five priority zones, representing the top 20% of optimal Sungazer habitat were identified using an ecological niche model. These zones are spread across the distribution, with sites situated in the west (Welkom), north centre (Vrede, Edenville), south east (Harrismith) and north east (Volksrust). In total, the priority zones cover 1.7% of the AOO, but are estimated to contain 3-4.4% of the total population based on the habitat quality. The population size estimated contained within these zones is four to five times the mean minimum viable population (MVP) estimated for vertebrate species.

Conclusion. I used my demographic measures to assess the conservation status of *S. giganteus* using Version 3.1. of the IUCN Categories and Criteria for conservation assessments. This assessment improves the precision of the measure of population reduction and includes geographic range for the species. My conservation assessment confirms the current listing of *S. giganteus* as ‘Vulnerable’ under criteria A2bcd and B2ab. I highlight the need for developing a protocol for translocations, a phylogeographic study to assess the landscape genetics of the species, an investigation of dispersal patterns and colonisation strategies.

DECLARATION

I declare that this dissertation is my own, unaided work unless specifically acknowledged in the text. It has not been submitted previously for any degree or examination at any other university, nor has it been prepared under the aegis or with the assistance of any other body or organization or person outside of the University of the Witwatersrand, Johannesburg, South Africa.

Shivan Parusnath
23rd May 2014

DEDICATION

I dedicate this work to my parents, Vinesh and Sashi Parusnath, who nurtured my love for nature since the very beginning.

Thank you for everything.

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All protocols used during this research were passed by the Animal Ethics Screening Committee of the University of the Witwatersrand under permit 2011/47/2A.

TABLE OF CONTENTS

ABBREVIATIONS AND ACRONYMS

The following abbreviations and acronyms are used throughout the dissertation:

AOO	Area of Occupancy
AUC	Area Under Curve
BOI	Burrow Occupancy Index
CI	Confidence Interval
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
DEAT	Department of Environmental Affairs and Tourism
EMPSSR	Eskom Majuba Power Station Sungazer Reserve
ENM	Ecological Niche Model
EOO	Extent of Occurrence
EWT	Endangered Wildlife Trust
GIS	Geographic Information Systems
Ha	Hectare
IUCN	International Union for the Conservation of Nature
MBD	Mean Burrow Density
MVP	Minimum Viable Population
NLC	National Land Cover
PA	Priority Area
PEB	Percentage of Empty Burrows
PMI	Percentage of Mature Individuals
QDGC	Quarter Degree Grid Cell
RDB-AR	Red Data Book - Amphibians and Reptile
ROC	Receiver Operating Characteristic
SANBI	South African National Biodiversity Institute
SARCA	South African Reptile Conservation Atlas
SVL	Snout-to-vent Length

PROJECT OVERVIEW

The research in this dissertation was conducted in response to the perceived need for a re-evaluation of the conservation status of *S. giganteus*. Thus, the dissertation is structured around the investigation of aspects of demographics and distribution of the species. The data collected allow for the first estimates of area occupied, population size, rate of decline of habitat and populations, and provides sufficient data for a realistic conservation assessment.

Chapter 1: General introduction

In the introduction, I summarise the current global status of reptiles, and the processes that threaten them. I consider risks facing reptiles in South Africa and synthesise current knowledge on the distribution, conservation status, ecology of the study species. The historical and current levels of the threat processes being faced by the species are assessed in relation to reptiles in general. I also highlight the need for research.

Chapter 2: Sungazer population density

Sungazer population density has previously been assessed through time-consuming and labour-intensive capture techniques. These studies have focused on population densities in small areas and are prone to the effects of a complex interaction of density-dependant factors. They are also likely to focus only on prime habitat where the species is most abundant. I developed a methodology using burrow counts as a rapid alternative to capture techniques, and measured burrow density at 80 sites across the distribution. This provides the first distribution-wide measure of burrow density for the species and, at least partially, controls for environmental density-dependant variables on population estimates. A weighted metascore aggregating burrow occupancy measures from previous studies was used as in index to calculate population density across the distribution based on the burrow density measures from this Chapter.

Chapter 3: Quantifying EOO, distribution and AOO

The distribution of the *S. giganteus* was first mapped by De Waal (1978) and Jacobsen (1989), and various occurrence records have expanded our knowledge of the range of the species since. However, it has been noted that there are large areas across the distribution where the species does not occur, and the relationship between the Extent of Occurrence and Area of Occupancy therefore becomes important in understanding spatial use of the landscape by Sungazers. I define the EOO of the species and calculate the percentage of untransformed land area within the EOO using the most recent GIS national land cover map (NLC 2009). The AOO was quantified by assessing the likelihood of finding Sungazers at randomly selected sites in untransformed land area across the distribution of the species.

Chapter 4: Sungazer population size: Present, historical, and rates of decline

A lack of knowledge of burrow density and spatial use of the EOO by Sungazers has prevented accurate estimations of population size being made. Alongside an understanding of the area occupied by a species and how this landscape is used, knowing the number of animals left in the wild is crucial to the management of the species. I estimated the population size of the species by developing a formula that builds on the findings of previous Chapters. A case study assessed the efficacy of this model by estimating the population size of a small reserve, and compared the estimated number to the actual population size. Measures of Sungazer population decline have previously only been inferred from decline in habitat across the EOO, as opposed to direct observation of declines at a sub-population scale and the true rate of decline of the species was therefore not well understood. I investigated population decline by visiting localities that had previously been confirmed, and assessed them for the presence of Sungazers.

Chapter 5: Identifying priority areas for Sungazer conservation using ecological niche modelling

Despite the Sungazer's 'Vulnerable' conservation status and the numerous threats that it faces, the species does not occur within any formal conservation areas in population sizes that can be sustained in the long-term. I created an ecological niche model using 536 occurrence records, and 19 environmental variables to model the suitable habitat of the species across the distribution. I selected the top 20% of habitat identified as optimal by the model, and created polygons around these zones. I estimated the population size for each zone and total area of all zones, using the formula developed in Chapter 4, and compared this to the minimum viable population (MVP) necessary for the species to persist.

Chapter 6: Red List assessment, conservation recommendations, future work and conclusions

In this Chapter, three primary sections are covered: the current status of the species in the wild, recommendations for the *in situ* and *ex situ* conservation of the species, and recommendations for future studies. Firstly, I discuss the current status of the Sungazer in the wild based on the findings of this study. Declines in habitat and populations, as well as overall population size are considered, and these findings are assessed using the latest set of IUCN Red Data List Categories and Criteria (Version 3.1.), with the goal of assessing the species' conservation status on the IUCN Red Data List. Secondly, I discuss various approaches to the *in situ* and *ex situ* conservation of the species, and recommendations of the most promising methods are made in the context of the natural history and ecology of the species. The use of the Sungazer as a flagship species and an education tool are investigated and justified. Finally, recommendations for the direction of future research on the species are highlighted.

DETAILS OF PUBLICATIONS AND PRESENTATIONS

Published articles

Parusnath, S. 2012. Natural history note: *Smaug giganteus* A. Smith, 1844 Predation. African Herp News 58, 13-15.

Manuscripts in preparation

Parusnath, S., Cunningham, M.J., Jansen, R., Little, I.T, Alexander, G.J. A conservation assessment of the Sungazer (*Smaug giganteus*). (In preparation for African Journal of Herpetology)

Parusnath, S., Cunningham, M.J., Jansen, R., Little, I.T, Alexander, G.J. Spatial distribution and population size of *Smaug giganteus.* (In preparation for Journal of Herpetology)

Parusnath, S., Alexander, G. J. The desolation of Smaug? The effects of habitat loss, degradation and fragmentation on the Sungazer (*Smaug giganteus*). (In preparation for Journal of Herpetology)

Parusnath, S., Alexander, G.J. Using burrow counts to rapidly assess Sungazer (*Smaug giganteus*) population density. (In preparation for Copeia)

Parusnath, S. Using ecological niche models to identify priority conservation areas for the Sungazer (*Smaug giganteus*). (In preparation for Diversity and Distributions)

Conference presentations delivered

November 2012. Parusnath, S., Alexander, G.J., Kotze, A., Dalton, D., Jansen, R., Little, I. Genetics, distribution and population density of a threatened endemic lizard, the Sungazer (*Smaug giganteus*). Third Annual National Zoological Gardens Research Symposium. Pretoria, South Africa.

February 2013. Parusnath, S., Jansen, R., Cunningham, M.J., Little, I. T., Alexander, G. J. Population ecology and conservation of the Sungazer (*Smaug giganteus*). 11th Annual Herpetological Association of Africa Symposium. Pretoria, South Africa. (Best student award)

CHAPTER 1:
INTRODUCTION

1.1. Global and local decrease in reptile biodiversity

1.1.1. Conservation status of the world's reptiles

Of the 9909 reptile species currently described worldwide (Uetz and Hošek, 2014), only 42% have had their conservation status assessed for the IUCN Red Data List (IUCN, 2013). A fifth of the species assessed are classified as Threatened and face various levels of extinction risk in the wild; 18.7% 'Critically Endangered', 37.4% 'Endangered' and 43.9% 'Vulnerable' (IUCN, 2013). The paucity of data on how threat processes affect reptile species that have not yet been assessed results in these taxa being overlooked in conservation and management decisions (Böhm *et al.,* 2013). In the face of scant data on the conservation status of the remainder of the world's reptile species, Böhm *et al.,* (2013) reviewed the conservation status of 1500 randomly selected species representing all known reptile taxa to gauge the proportion of reptiles facing imminent extinction risk. Their results mirrored those of the IUCN very closely; 19% of species were classified as Threatened with 12% of those classified as 'Critically Endangered', 41% 'Endangered' and 47% 'Vulnerable' (Böhm *et al.,* 2013). Furthermore, 21% of species assessed were classified as 'Data Deficient', meaning that abundance and/or distribution data are lacking, such that the status of the species cannot be assessed (IUCN, 2012). The IUCN recommends that 'Data Deficient' species receive the same attention as Threatened species until data can be collected and that their status can be assessed (IUCN, 2012). Understanding the dire situation of these species, facilitates planning to ensure their persistence and the amelioration of the threat processes.

1.1.2. Threats to reptile biodiversity

Anthropogenically-driven loss, degradation and fragmentation of habitat are the primary contributors to biodiversity loss worldwide (Soule, 1991; Freemark, 1995; Duelli, 1997; White *et al*., 1997; Jeanneret *et al.,* 2003; Schumaker *et al*., 2004; DEAT, 2005; Driver *et al.,* 2005; Santellman *et al*., 2006; Lötter, 2010), particularly in reptile species (Branch, 1988; Shine, 1991; Gibbons *et al.,* 2000; Glor *et al*., 2001; Fabricius *et al*., 2003; Smart *et al*., 2005; Santellman *et al*., 2006; Masterson *et al.,* 2009; Böhm *et al.,* 2013). Over 80% of Threatened reptile species are affected by more than one threat process, with agricultural land cover change, biological resource use, urban development and invasive alien species being the primary threats to terrestrial reptiles, affecting 74%, 64%, 34% and 22% of species respectively (Branch, 1998; Vitt *et al.,* 1998; Glor *et al.,* 2001; Santellman *et al.,* 2006; Böhm *et al.,* 2013). Climate change has already led to reptile extirpations, however the worst effects are yet to come, with reptile species extinction rates expected to reach 20% by 2080 (Sinervo *et al.,* 2010). Rare and threatened reptile species frequently fall victim to the pet trade, with legal and illegal harvesting contributing to population declines and extirpations (Bartlett, 1997; Grismer *et al.,* 1999; Jenkins *et al.,* 1999; Webb *et al.,* 2002; Auliya, 2003; Reed and Gibbons, 2003; Sy, 2012). Species that occur in close vicinity of rural communities are often used in traditional medicine and as food (Klemens and Thorbjarnarson, 1995; Van Dijk *et al.,* 2000; Huang *et al.,* 2008; Simelane and Kerley, 2008). The impact of these various threats vary from country to country, and understanding the habitat use across the landscape of a country is integral to understanding how changes in landscape affect local reptile species diversity.

1.1.3. Biodiversity loss in South Africa

In South Africa, biodiversity is threatened by habitat loss processes, primarily urbanisation, agriculture and mining (Minter *et al.,* 2004; DEAT, 2005; Driver *et al.*, 2005). Burgeoning human populations require more land for living space, with a trend towards urbanisation. Approximately 60% percent of South Africa's population reside in urban areas, and this is expected to rise to 80% by 2020 (Driver *et al.,* 2005). Urbanisation is correlated with irreversible transformation of the landscape for houses, roads, recreational and industrial areas. Agricultural transformation has had the biggest impact on natural habitat in South Africa, through the clearing of natural vegetation for crop cultivation (Driver *et al.,* 2005). Agricultural area occupies over 80% of South Africa's land area, providing employment to 13% of the country (DEAT, 2005). Crop monoculture dominates 13% of this area, while virtually all untransformed grassland is used as rangeland (DEAT, 2005; Lötter, 2010). South Africa is rich in mineral sources such as gold, platinum, diamonds and coal. As a result, mining is a large industry in South Africa, generating ~41% of foreign exchange, and contributing between 6-9% to national GDP (DEAT, 2005). Often, areas of high agricultural potential and mineral deposits overlap with important biodiversity areas (Driver *et al.*, 2005). This leads to a conflict in land use allocation.

South Africa's rich biodiversity also fuels strong traditional medicine and pet trades. The traditional medicine trade in South Africa has over 28 million consumers (DEAT, 2005; Mander *et al.,* 2007), 71% of which use animal products (Simelane and Kerley, 1997). The diversity of reptile species reported in the traditional medicine trade varies across provinces, with the Faraday Market in Gauteng offering at least 33 species for sale (Whiting *et al.*, 2011), various herbalist shops in the Eastern Cape providing a total of 31 species (Simelane and Kerley, 1998), and 21 species across KwaZulu-Natal (Ngwenya, 2001). Quantifying the effects of harvesting for the traditional medicine trade is difficult because of the unwillingness of traders to reveal the sources of their stock (Whiting *et al.,* 2011). Approximately 20% of CITES listed reptiles exported out of South Africa for the pet trade over the last decade were wild caught (CITES, 2014). However, this may be higher, as it is believed that some reptile species exported as captive may have actually been wild caught (David Newton, pers. comms.).

South Africa is home to 480 reptile species, 36% of which are endemic to the country, and 24% classified as Threatened (Bates *et al.*, 2014). A third of South Africa's terrestrial ecosystems are also Threatened, yet only 5.4% of the country's land area is formally conserved (Driver *et al.*, 2005). These protected area networks are skewed towards biomes such as savanna, and many of the other nine recognised biomes in South Africa are under-conserved. Grassland is one such biome, covering 29.5% of the country, and hosting the second highest diversity of indigenous and endemic species after the Cape Floristic Region (DEAT, 2005). Over 60% of grassland ecosystems are Threatened, yet less than 3% of the biome is formally conserved (Bredenkamp, 2002; DEAT, 2005; Driver *et al.*, 2005).This situation poses grave conservation risks to both grassland fauna and flora (Lötter, 2010).

1.2. The Sungazer and habitat destruction

The Sungazer (*Smaug giganteus* formerly *Cordylus giganteus*), also known as the Giant Dragon Lizard (formerly Giant Girdled Lizard), is a Threatened Dragon Lizard species endemic to the Highveld grasslands of the north-eastern Free State and southern Mpumalanga Provinces of South Africa (De Waal, 1978; Jacobsen, 1989). *Smaug giganteus* is currently listed as 'Vulnerable' (Mouton, 2014). However this classification is based substantially on distribution data from 1978 (De Waal, 1978) and it is possible that the species is at an even greater risk of extinction than is suggested by this IUCN status. Since De Waal's (1978) study there has been a proliferation of trade in the species and a continued trend of transformation across its distribution. The need for a re-assessment of the conservation status based on current population size and distribution has long been recognised (Van Wyk, 1988; Van Wyk, 1992; McIntyre, 2006; IUCN, 2013).

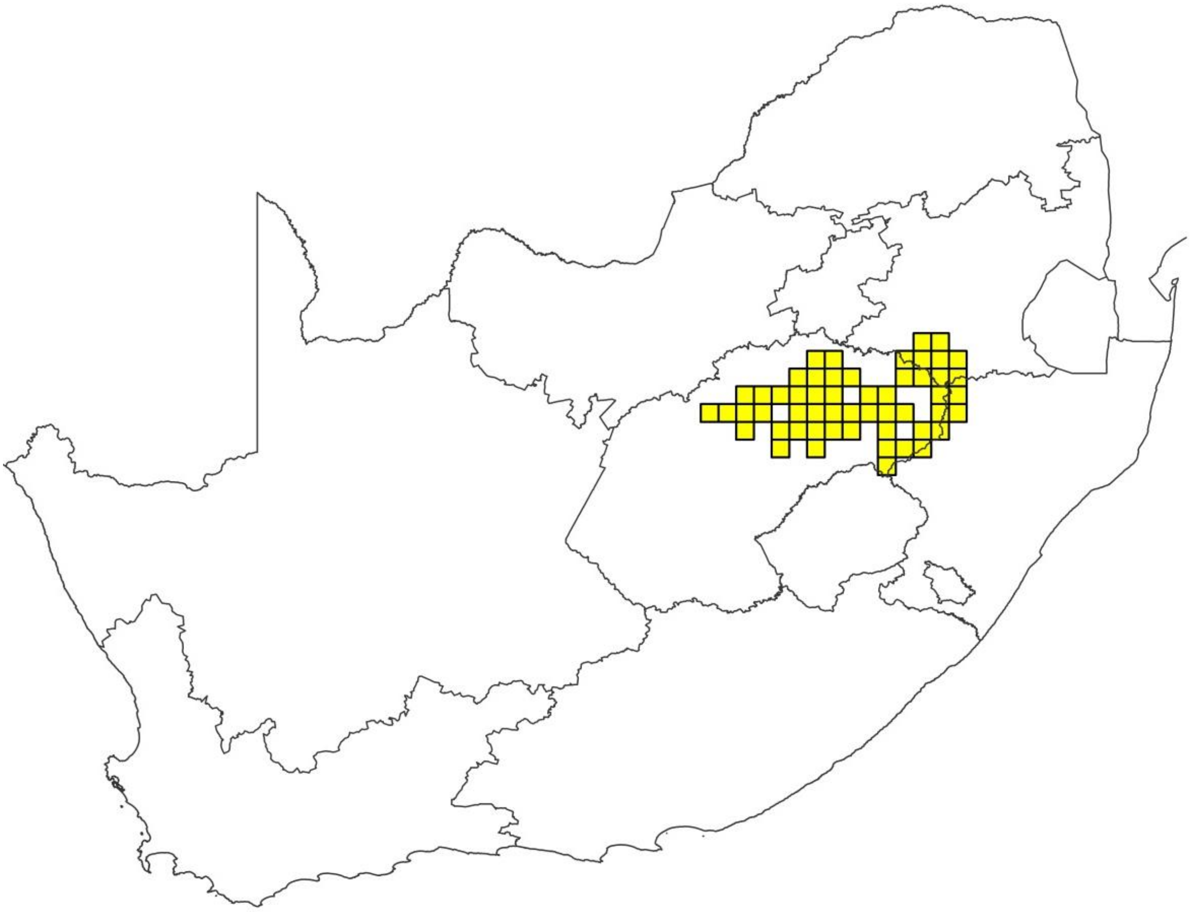


Figure 1.1. Map of South Africa showing Quarter Degree Grid Cells where *Smaug giganteus* occurrence has been recorded (Source: Mouton, 2014).

The distribution of *S. giganteus* falls within the Highveld Agricultural Region, 40% of which has been irreversibly transformed for the monoculture of maize, wheat, sugar, sorghum, sunflowers and potatoes (Van Wyk, 1992; DEAT, 2005). The Sungazer is a unique Dragon Lizard species as it is not rupicolous, but rather shelters in self-excavated burrows in grassland that are prone to destruction during crop cultivation (Van Wyk, 1992; DEAT, 2005). Of the natural Highveld, *Themeda* grassland comprises the optimal habitat, 81.5% of which is underlain by arable soil, 40% of which is deeper than 400 mm and therefore ideal for both crop production and Sungazer burrows (Branch and Patterson, 1975; De Waal, 1978; Van Wyk, 1988; Jacobsen *et al.*, 1990; Van Wyk, 1992). The non-arable soil in the grasslands forms only marginal and minimally inhabited habitat for Sungazers, as burrowing is restricted by the presence of rocks and bedrock (Van Wyk, 1992). Agricultural practises are therefore a major and direct threat to the species, destroying large tracts of habitat and fragmenting populations (Marais, 1984; Van Wyk, 1988; Jacobsen *et al.*, 1989).

Reptiles respond differently to land-use scenarios in comparison to other vertebrate taxa (Santelmann *et al.*, 2006), and should be considered separately from other fauna when developing land use plans (Masterson *et al.*, 2009). The life history requirements of individual species need to be considered when investigating the risks that landscape changes pose to diversity. Species that are purely terrestrial, such as *S. giganteus*, are the most vulnerable to human mediated changes in land cover (Van Wyk, 1992). Grasslands existing on previously cultivated land support fewer reptile species than primary grassland (Masterson *et al.*, 2009). Crop monoculture significantly alters soil properties, and changes to soil and vegetation affect diversity and abundance of invertebrates, and consequently, the vertebrate taxa that feed on them (Driver *et al.*, 2005). Correspondingly, Sungazers have not been recorded recolonizing fallow lands (Jacobsen, 1989; Newbery and Jacobsen, 1994), despite earlier suggestions by Marais (1984).

Apart from agriculturally-driven land cover change, *Smaug giganteus* is also threatened by mining developments, and the construction of dams, roads, power stations and other developments (Van Wyk, 1992). The Highveld grasslands lie above uranium and gold deposits, and coal fields, and all three of these resources are mined within the distribution of *S. giganteus* (Van Wyk, 1992; McIntyre, 2006). Mining sites in South Africa are typically not rehabilitated to original condition after use, and Sungazers have never been recorded in any of these 'rehabilitated' sites. Mining waste products have also been found to accumulate in the tissue of Sungazers that occupy areas around mining sites, resulting in poor physiological condition of the animals (McIntyre, 2006). The construction of Eskom's Majuba Power Station in Mpumalanga destroyed 1.8% of the land area within *S. giganteus* distribution (Jacobsen *et al*., 1990), and ten more power stations are planned for construction within the EOO. These developments will transform a significant portion of the Sungazer habitat (Petersen *et al.,* 1985).

1.3. Illegal Sungazer harvesting

1.3.1. Pet trade

Sungazers are highly sought-after in the international pet trade (Auliya, 2003), and are one of the top five reptile species exported out of South Africa due to this trade (CITES, 2014). Auliya (2003) reports that rare and endemic reptiles from South Africa are in high demand in the European trade, with the Sungazer being in the top eight reptiles in the greatest demand across Europe. Japan, the United States and Germany are the primary import countries of Sungazers, collectively making up 74.8% of imports over the past three decades (Fig. 1.2.). Species such as *S. giganteus* that have a resticted distribution, low reproductive rate and high level of protection, are rarities in the trade and command premium prices due to their high demand (Reed and Gibbons, 2003; Auliya, 2003). Sungazers are known to have been offered for sale for ₩6 328 000 (~R64 000) in the South Korean pet trade (Brian Lee; pers. comms.), ¥680 000 (~R70 000) in Japan, €2 000 (~R28 000) in Europe

(Fraser Gilchrist, pers. comm.) and $2 000 (~R21 000) in the USA (Auliya, 2003). The high prices demanded for Sungazers serve as an incentive for their illegal capture and exportation (Haacke, pers. comms. in McLachlan, 1978; Van Wyk, 1988; Auliya, 2003), with cases of poaching for the European and American trades being reported since the 1970s (Van Wyk, 1988).

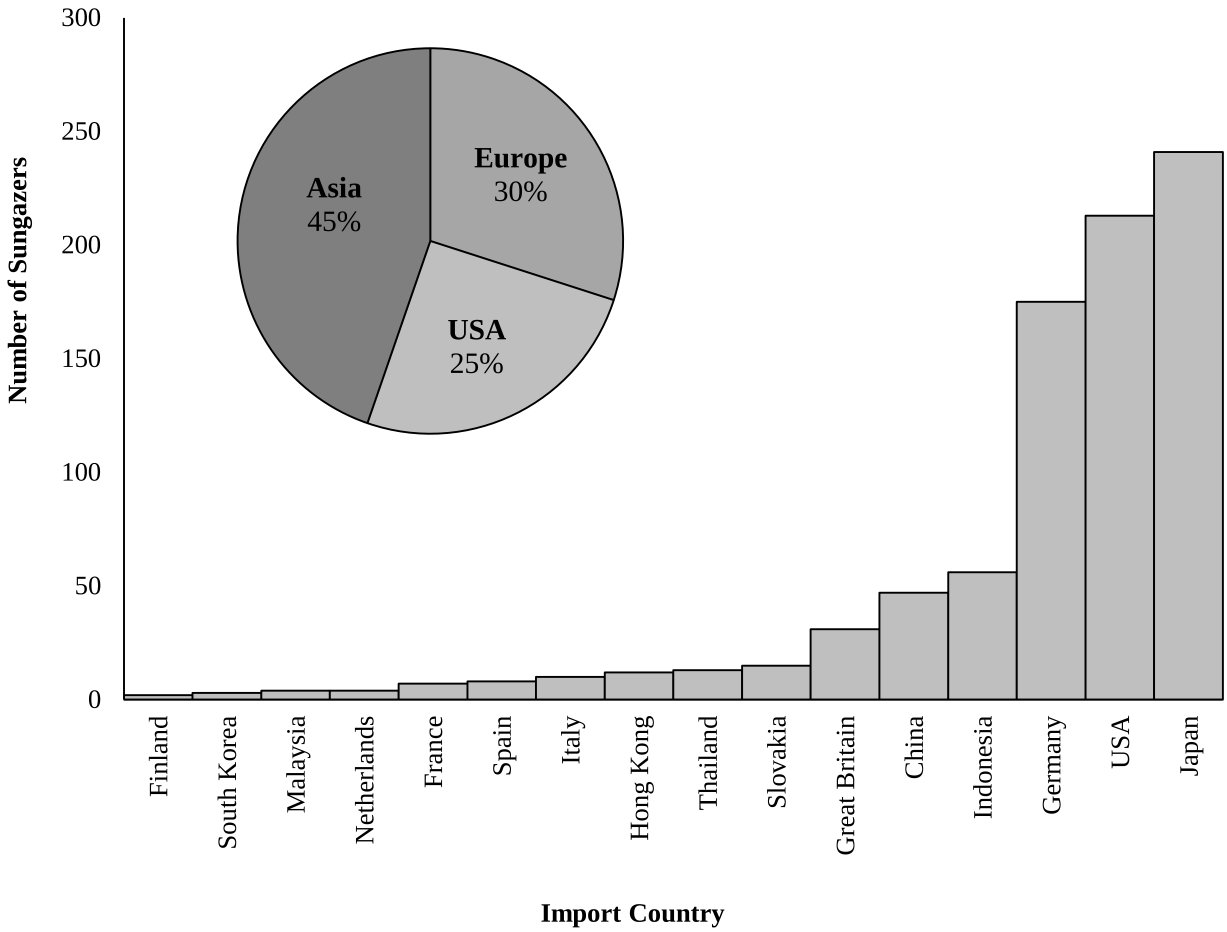


Figure 1.2. Number of Sungazers exported from South Africa to various destinations between 1983 and 2012. The pie diagram insert indicates the percentage contribution of three major continents to the total number of Sungazers imported over this period (Source: CITES, 2014).

The popularity of Sungazers as pets, along with the loss of habitat due to agricultural development, led to a request in 1980 from the scientific counsellor at London's South African embassy for the British government to ban the import of the species (TRAFFIC, 1980). This was in accordance with the claim that "as from 1st June, South Africa will impose a total ban on exports of *Cordylus giganteus*" (TRAFFIC, 1980). Despite these communications, a total ban of the export from either side was not realised. Instead, the species has been listed on CITES Appendix II since 1981 (UNEP-WCMC, 2014), and permits for the export of Sungazers are strictly regulated and supposedly issued only under exceptional circumstances (Branch, 1990). However, Sungazers continue to be exported in large numbers (Fig. 1.3) (CITES, 2014). Between 1983 and 2012, 761 Sungazers were exported from

South Africa, with a mean of 25 Sungazers exported per year. The trade has increased over the past decade, with 2003-2012 representing 47% of total export numbers on record. A multitude of articles on the captive husbandry of the Sungazer and new listings on the European Studbook over the past decade are testament to the thriving captive trade of the species (Donovan, 1997; Fogel, 2000; Langwerf, 2001; McKeown, 2001; Zwartepoorte, 2003; Zwartepoorte, 2006; Schwier, 2007; Gilchrist, 2009; Gilchrist 2010a; Gilchrist 2010b; Zwartepoorte, 2010; Gilchrist, 2013).

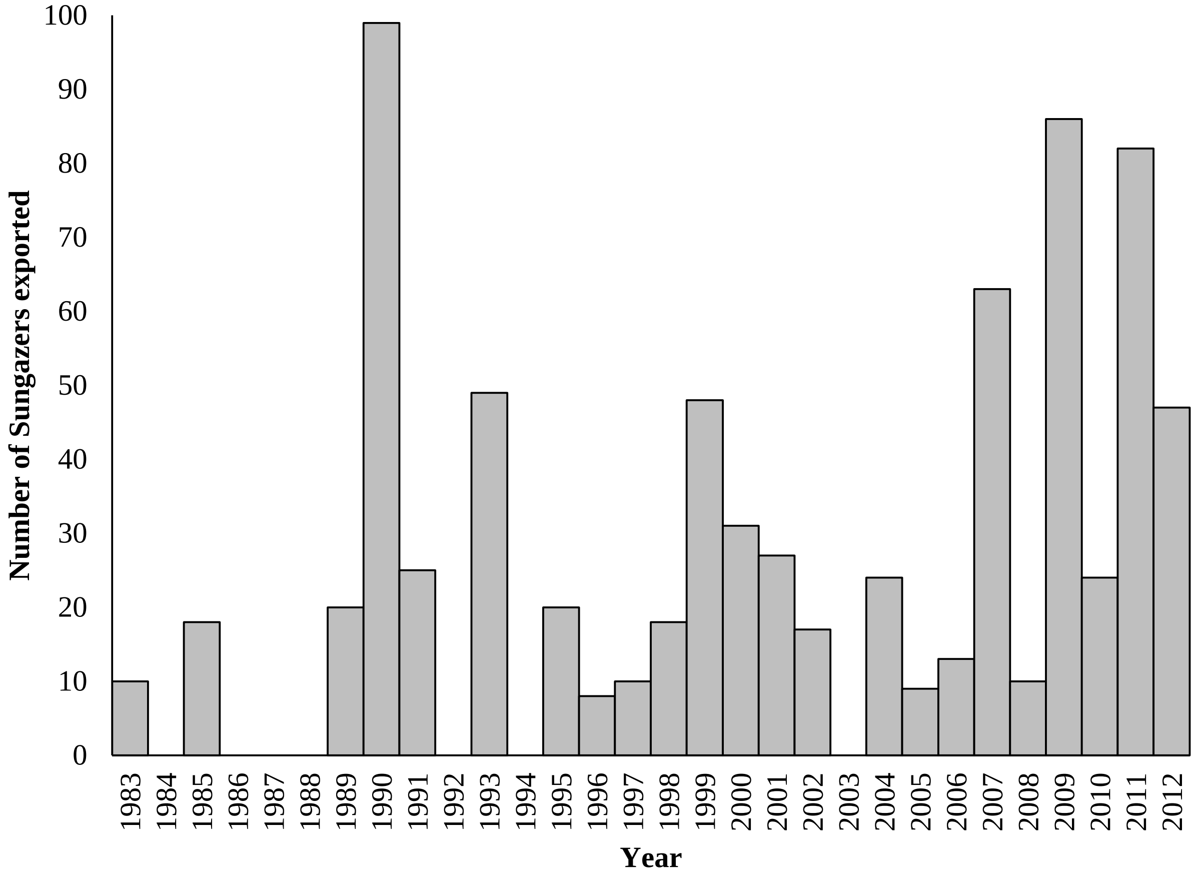


Figure 1.3. Number of Sungazers exported out of South Africa per year between 1983 and 2011 (Source: CITES, 2014).

All CITES listed records of Sungazers being traded during the period 1983-2012 have been reported as captive bred, however no breeding programmes are known of within the country that can supply a trade with such a large quantity of lizards. Furthermore, only one case of successful captive breeding of Sungazers has ever been officially reported (Langwerf, 2001). The trade in captive-bred animals is less strictly regulated than trade in wild-caught animals, and there are indications that wild-caught animals are laundered and imported as captive bred (Auliya, 2003). It has been suggested that many Sungazers being exported from South Africa are done so under a false declaration of the source of the animals when applying for permits (David Newton; pers. comm.; Fraser Gilchrist; pers. comm.).

1.3.2. Traditional medicine (Muti) trade

Sungazers have historically been harvested for use in traditional Sotho medicine across the distribution of the species. In recent times however, Sungazers have been found in muti markets outside of the distribution of the species, in the neighbouring provinces of KwaZulu-Natal and Gauteng (Whiting *et al.*, 2011; pers. obs.). Sungazers are purchased by Sangomas (witch-doctors) who use powdered body parts to make potions that purportedly allow a man to achieve harmonious consent from his wife or girlfriend to have multiple partners (Peterson *et al.*, 1985; pers. obs.). These potions are sold in small quantities (50 g) and use a small portion of Sungazer skin, but sell for R200 per ‘treatment’. Whiting *et al.*, (2011) found that 21.9% of traders at the Faraday market in Johannesburg, (South Africa’s second largest traditional medicine market) had cordylid species for sale. Five traders were recorded as selling Sungazer body parts and whole Sungazers, and recently killed Sungazers were observed on sale. Quantifying the large-scale effects of harvesting of Sungazers for the traditional medicine trade is difficult because of the unwillingness of traders to reveal the sources of their stock, and the turnover rate of lizards (Whiting *et al.*, 2011).

Previously, the illegal reptile trade has been largely ignored in conservation evaluations of this species, but it has the potential to significantly impact wild reptile populations (Simelane and Kerley, 1998; Auliya, 2003; Whiting *et al.*, 2011). Sungazers occur in discrete colonies across the grassland matrix and even at low population densities it is generally possible to excavate and remove a large number of animals at a time (De Waal, 1978). McLachlan (1978) reports that three collectors have been known to collect 200 Sungazers in a day. The plight of the Sungazer is exacerbated by the fact that females breed only biennially or triennially, depending on resource availability (Van Wyk, 1992; Van Wyk, 1994). McKinney (1997) suggests that species with these life-history characteristics are the most at risk from unsustainable harvesting.

The combination of increasing irreversible habitat loss and fragmentation through anthropogenic land cover change, and the loss of populations from harvesting to fuel illegal pet and muti trades leaves the Sungazer in a potentially dire situation. Conserving a species can only be effective when the effects of current and future threats to its longevity are recognised and quantified. The Sungazer, despite its iconic status, has been overlooked in this regard. The effects of these threats to its persistence need to be urgently assessed in order to understand how the population size has changed over time, and to estimate its risk of extinction.

1.4. Conservation history of the Sungazer

Smaug giganteus was classified as ‘Vulnerable’ in the first South African Red Data Book - Amphibians and Reptile (RDB-AR) (McLachlan, 1978), due to the amount of habitat destruction across the distribution of the species, and collection of animals for the pet trade and for use in laboratory dissections. The status was retained in the subsequent RDB-AR (Van Wyk, 1988), chiefly because the reassessment was based on the original data with few additions (Van Wyk, 1992). The species has since been listed as 'Vulnerable' in the first assessment for the IUCN Red Data List (Groombridge, 1994), and a subsequent update (WCMC, 1996), based on criteria A2cd (suspected > 30% population reduction in the past three generations based on habitat quality and actual or potential levels of exploitation; IUCN 2001 Red Data List Category and Criteria Version 2.3). The most recent assessment of the species' conservation status in the Atlas and Red Data List of the Reptiles of South Africa, Lesotho and Swaziland (Mouton, 2014), retains the 'Vulnerable' status for the species, based on a population reduction of 30% over the last 27 years, inferred from habitat destruction across the Grassland Biome. This assessment represents the first in several decades to take recent changes in the habitat across the distribution of the species into account, and offers a more realistic picture of the current situation. Despite this update, there remain gaps in the knowledge of the life history and ecology of the species that impede an accurate assessment of the conservation status of the species. With knowledge of how the land within the distribution is used by the species, population density and abundance across the distribution and observed changes in known populations, a thorough investigation can be conducted for the species.

1.5. Study animal: Sungazer (*Smaug giganteus*)

1.5.1. Introduction and classification of species

Smaug giganteus has the largest body size of any species in the Cordylidae, a family of lizards that is endemic to sub-Saharan Africa (Branch, 1998). A recent molecular study assigned the species to the new genus *Smaug* along with seven other species previously belonging to the genus *Cordylus* (Stanley *et al.*, 2011). *Smaug giganteus* is a heavily armoured species, with a mean SVL of 183 mm (McIntyre, 2006), and is easily distinguishable from other cordylids by the elongated pair of occipital spines and the enlarged keeled caudal spines (Van Wyk, 1988). The species is known as the Sungazer because of its distinctive thermoregulatory behaviour of elevating the anterior parts of the body by extending its forearms, usually near the entrance of its burrow as if looking at the sun (Branch, 1998). The species is well known throughout its distribution, and goes by several different common names, in different languages. The most common name is 'Ouvolk', given by Afrikaans landowners who liken the thermoregulatory basking position of the species to retired farmworkers, who spend much of their days sitting in the sunlight. The Sungazer is also known ubiquitously as 'Pathakalle' by Sotho speaking people and 'Mbedla' by Zulu speaking people.

1.5.2. Distribution

Smaug giganteus is endemic to the northern Free State and the adjacent southern parts of Mpumalanga. The species has previously been listed as occurring in KwaZulu-Natal (Bourquin, 1993, Bourquin, 2004; Lambiris and Bourquin, 1993). However Armstrong (2011) suggested that *S.*

giganteus be removed from the list of reptiles occurring in KwaZulu-Natal based on the results of a survey conducted across sample sites selected using an ecological niche model. Furthermore, Armstrong (2011) reported that most records of *S. giganteus* in KwaZulu-Natal are of animals that were released into farmland properties from other provinces, and not based on indigenous populations. Records from western Lesotho (Ambrose, 2006) have also been considered to be doubtful (Mouton, 2014).

1.5.3. Environmental niche

Climatic variables

Smaug giganteus occurs between the altitudes of 1400 m and 1800 m above sea level, with the highest population densities between 1500 m and 1670 m (Jacobsen, 1989; Van Wyk, 1992). The Grassland Biome in which it occurs is characterised by summer rainfall and winter frost (DEAT, 2005). Mean annual precipitation across the distribution ranges from 500 mm in the west to 800 mm in the east, and 600-700 mm is typical for the central area (Van Wyk, 1992). Maximum rainfall (70%) occurs between November and March (Van Wyk, 1992). Mean temperature ranges from 18 °C in the east to 24 °C in the west during summer (January), from 7 °C in the east to 9 °C in the west during winter (July) (Van Wyk, 1992).

Vegetation type

The EOO of the of *S. giganteus* falls across 31 vegetation groups described by Mucina *et al.*,(2006), yet the Sungazer occupies only 10 of the vegetation types within the EOO (Table 1.1.). 53.92% of these vegetation types are Endangered, and 6.13% are Vulnerable, yet only 0.74% of the total area of these vegetation types is formally protected.

Landform, geology and soil types

Sungazers are typically found on flat or gently sloping land (De Waal, 1978; Van Wyk, 1988; Van Wyk, 1992; Newbery and Jacobsen, 1994), and 85.42% of the EOO is comprised of level land with a gradient of less than 30% (Table 1.2.). The range of the species is underlain by sandstones, mudstones and shales of the Beaufort series (Van Wyk, 1992). Dolerite intrusions form ridges and escarpments across most of the area (Scheepers, 1975; Newbery and Jacobsen, 1994). The soil is generally brown and loamy, dominated by Avelon, Escort, Kroonstad and Longlands soil groups (Branch, 1988; MacVicar, 1991). These soils are highly arable, and may be deeper than 400 mm, making ideal conditions for monoculture of commercial crops.

Table 1.1. Percentage contribution, conservation status, percentage protected and percentage remaining of vegetation types found in the EOO (minimum convex hull) of *S. giganteus*. (Source: Mucina *et al.*, 2006).

Legend	**Conservation status**	**% contribution to Sungazer EOO**	**% protected**
Eastern Free State Sandy Grassland	Endangered	18.37	1.8
Frankfort Highveld Grassland	Vulnerable	18.27	0
Central Free State Grassland	Vulnerable	16.52	0.8
Eastern Free State Clay Grassland	Endangered	15.89	0.1
Vaal-Vet Sandy Grassland	Endangered	11.64	0.3
Soweto Highveld Grassland	Endangered	8.019	0.2
Amersfoort Highveld Clay Grassland	Vulnerable	5.167	0
Low Escarpment Moist Grassland	Least threatened	2.882	2
Highveld Alluvial Vegetation	Least threatened	1.7	9.2
Western Free State Clay Grassland	Least threatened	1.543	0

Table 1.2. Percentage contribution, gradient and relief intensity of landform across the EOO of *S. giganteus* (Dijkshoorn *et al.*, 2008).

Landform	**% contribution to Sungazer EOO**	**Gradient (%)**	**Relief Intensity (m/km^2)**
Level Land - Plain	52.09	<10	<50
Sloping Land – Dissected Plain	33.33	10 - 30	50 - 100
Sloping Land – Medium-gradient Mountain	2.46	15 - 30	150 - 300
Water	0.49	/	/
Level Land – Valley Floor	1.26	<10	<50
Steep Land – High-gradient Valley	2.54	>30	>150
Sloping Land – Medium-gradient Hill	7.29	10 - 30	100 - 250
Steep Land – High-gradient mountain	0.55	>30	>300

1.5.4. Burrow structure, density and occupancy

Sungazers live in self-excavated burrows (Branch, 1988), and are not known to adopt burrow systems from other species. Burrows typically range from 1.5 m to 2.5 m in length, and slope to an average depth of 0.5 m (Jacobsen *et al*., 1990; Van Wyk, 1992). They are usually dug into the slope of the topography, with entrances facing the aspect of the slope (Van Wyk, 1992). Mean burrow densities range from 4-6.8 burrows/ha, (Stolz and Blom, 1981; Jacobsen, 1989; Jacobsen *et al.*, 1990; Van Wyk, 1992), however densities as high as 19 burrows/ha have been recorded (Van Wyk, 1992). Burrows are typically occupied by single adults, or an adult with juveniles (Jacobsen *et al.*, 1990). Occasionally, burrows occupied by up to six and seven Sungazers have been found (Branch and Patterson; 1975; Jacobsen *et al*., 1990; Van Wyk, 1992). The relatedness of animals sharing burrows has not been investigated, and the social structure of the species is therefore not well understood. The frog species *Cacosternum boettgeri*, *Kassina senegalensis*, and *Semnodactylus wealii* are commonly found sharing Sungazer burrows (Branch and Patterson, 1975; De Waal, 1978; Van Wyk, 1992). The burrows represent a distinct microclimate, with deep burrow temperatures remaining relatively constant despite variation in surface temperature (Van Wyk, 1992). Sungazers are generally slow and cannot easily flee from predators. They typically remain in close proximity to their burrows and the burrows are used as retreats from predation (Ruddock, 2000; Losos *et al.*, 2002).

1.5.5. Breeding and lifespan

Sungazers are active from spring to autumn, but remain in their burrows during winter and early spring (De Waal, 1978; Van Wyk, 1988). It is unknown whether Sungazers brumate during this period (Van Wyk, 1988). They feed during eight months of the year, with Coleoptera, Diplopoda, Hemiptera, Hymenoptera, Orthoptera, and Lepidoptera making up the six major taxa present in their diet (Van Wyk, 2000). Sungazers are visually-orientated, extreme ambush (sit-and-wait) foragers (Jacobsen, 1989; Van Wyk, 2000). Breeding is seasonal, although females only reproduce every two or three years, depending on resource availability (Van Wyk, 1992). Reproducing females give birth to two or three live young in autumn (Van Wyk, 1992). Males and females reach sexual maturity at four to five years of age, at an average SVL of 165 mm (Van Wyk, 1992). Growth is relatively slow and lizards may only reach maximum length in their eleventh year (Van Wyk, 1992). Longevity records for Sungazers in captivity show that the species can live for up to 25 years (HAGR, 2014). Anecdotal observations from various landowners, who have kept Sungazers in enclosures in the natural environment within the distribution of the species, have claimed maximum longevity of 35 years. The slow reproductive rate, age to maturity and lifespan are typical of a K-selection life history strategy (Van Wyk, 1992).

1.5.6. Failure of past translocation efforts

Translocations of Sungazers from areas marked for development have not been successful even in the short term. Groenewald (1992) reported that burrow occupancy rates as low as 10% were reported 49 days after a population of Sungazers were translocated to the Golden Gate Highlands National Park. It was argued that this was due to predation by Yellow Mongoose (*Cynictis penicillata*), Suricate (*Suricata suricatta*) and Secretary Bird (*Sagittarius serpentarius*), but is also likely to be a result of the lizards not having suitable burrows in which to live. This latter contention is supported by the fact that lizards generally did not make use of the artificial burrows created for them with soil augers and travelled as far as 1000 m from the initial point of translocation (Groenewald, 1992), exposing themselves to predation. Adult females, which are known to share burrows with neonates, (Branch and Patterson, 1975; De Waal, 1978; Van Wyk, 1992) were found to abandon them upon translocation (Groenewald, 1992). Although formal scientific studies have not been conducted on the Golden Gate Highlands National Park since 1992, it is estimated that less than 2% of Sungazers survived the translocation over the next two years (Groenewald, pers. comm.).

1.6. Need for research

To protect a threatened species, relevant knowledge pertaining to its ecology, life history and its response to potential threats should be elucidated. Without adequate information, rational decisions cannot be made to conserve the species, and mitigate the threat processes that they face. The Sungazer is one of South Africa's most iconic reptiles, yet a poor understanding of current distribution-wide population densities of Sungazers and the effect of agriculturally-driven land cover change on populations hinders the formulation and implementation of effective conservation strategies. In the spate of continuing and irreversible trend of habitat degradation, a re-assessment of the conservation status based on current population density and distribution is much-needed (Van Wyk, 1988; McIntyre, 2006; IUCN, 2013). My study aims to investigate aspects of the ecology and life history of the Sungazer that are directly pertinent to the conservation of the species. Specifically, I aim to:

1) Quantify mean burrow density across the distribution
2) Quantify the area occupied by the species within the distribution (AOO)
3) Estimate the population size (mature individuals) of Sungazers
4) Quantify declines in Sungazer habitat
5) Quantify declines in Sungazer populations
6) Identify priority zones where conservation efforts should be focused
7) Re-assess the species' status on the IUCN Red Data List using the primary findings of this study

CHAPTER 2:
SUNGAZER POPULATION DENSITY

2.1. Introduction

Direct observation methods for estimating population density, such as transect censusing and trapping, can be time consuming, laborious and often underestimate actual population densities (Dasmann and Mossman, 1962, Emlen, 1971; Beck-King *et al.*, 1990). In addition, some species are not active throughout the year, and the times that they can be observed is limited. In burrowing species, quantifying the number of conspicuous burrows made by resident animals is a simpler method to estimate population density (Conroy, 1996). Furthermore, burrow counts may prove to be more accurate than methods based on direct observation in species in which burrow occupancy (ratio of animals:burrows) can be measured, such as in Wood Rats (*Neotoma* spp.; Cameron and Rainey 1972, Cranford 1977) , Pacas (*Agouti paca*; Beck-King *et al.*, 1990), Gopher Tortoises (*Gopherus polyphemus*; McCoy and Mushinsky 1992) and Land Crabs (*Cardisoma guanhumi*; Govender and Rodríguez-Fourquet, 2004). Accurate measures of burrow use can be ensured by identifying active burrows through signs of recent activity such as claw and/or tail marks (Auffenberg and Franz, 1982), footprints and shed tissue (Beck-King *et al.*, 1990) or faecal matter (Govender and Rodriguez-Fourquet, 2004). Studies using rapid assessment techniques that count burrows rather than animals are therefore increasingly employed as simpler and more accurate alternatives to quantify population densities, particularly with endangered or threatened species (Beck-King *et al.*, 1990; Govender and Rodriguez-Fourquet, 2004).

The life history and ecology of the Sungazer is centred around the burrows in which the lizards live, brumate and retreat for protection (Van Wyk, 1992; Ruddock, 2000; Losos *et al.*, 2002). Burrows are typically occupied by single adults (often accompanied by a juvenile) over long time periods (Jacobsen *et al.*,1990; Van Wyk, 1992; Ruddock, 2000) and their construction represents a significant investment of time and energy. They are therefore defended aggressively against conspecifics, with chemical signals used to mark burrows by both sexes (Ruddock, 2000). Sungazers spend a large portion of the day in their burrows, as shuttling in and out of the burrow is the primary means of actively regulating body temperature (Van Wyk, 1992). In addition, Sungazers are difficult to approach on foot when they are basking or foraging, as they generally stay in close proximity to their burrows and retreat to the burrow before they can be observed. It is therefore difficult to quantify Sungazer population density by direct observation, without resorting to trapping or excavation, and the method of using burrow counts to estimate population density presents a viable, simple alternative.

Jacobsen *et al.*, (1990) and Van Wyk (1992) recorded burrow occupancy from a sample of 1139 excavated Sungazer burrows, and these two studies provide data with which Sungazer population density can be estimated from burrow counts using a correction factor. In the well-studied system of Gopher Tortoise burrows it has been found that not all burrows are occupied all the time (Auffenberg and Franz, 1982; McCoy and Mushinsky 1992), and a correction factor is employed to relate the number of burrows to the number of tortoises. Similarly, in studies of Sungazers, Jacobsen *et al.*, (1990) and Van Wyk (1992) found that between 7.0 and 18.5% of excavated burrows were unoccupied. Subsequently however, it has been found that 68% of Sungazers leave their burrows to visit their nearest-neighbour of opposite sex during mating seasons for several days at a time (Ruddock, 2000). Empty burrows during the mating season are therefore likely to not represent true absences, but burrows where adults temporarily leave their burrows to seek mates. Furthermore, occupied burrows typically display conspicuous claw and tail marks in the exposed soil (Van Wyk and Swart, 2002) (Fig. 2.1), as well as recently shed scales from the large spines on the body and tail. These scales are very light and prone to being blown away or swept into the burrow by wind soon after shedding, and exposed scales thus indicate recent lizard activity. Therefore, I have assumed that a Sungazer burrow that has signs of recent activity is in use, and a burrow count within a defined area should reflect the population density when multiplied by the burrow occupancy rates.

Previous studies have quantified Sungazer burrow density extensively at particular sites (Jacobsen *et al.*, 1990; n = 580), or a pair of sites (Van Wyk, 1992; n = 273), but no studies have surveyed Sungazer burrow density using a standardised sampling procedure across the distribution of the species. Population density is a function of environmental complexity and energy resources (Pianka, 1973; Turner, 1977; Beck-King *et al.*, 1990), and measures of population density at a single site may be representative of the relative state of these factors at that site. In order to measure population density independently of factors that vary between sites, population densities should be assessed as widely across the distribution as possible. The primary aim of this Chapter was to measure Sungazer burrow density at representative sites across the distribution of the species such that the range of burrow densities across the distribution can be understood.

Aims

1) Measure Sungazer burrow density across the distribution of the species.
2) Assess the accuracy of burrow count surveys.
3) Calculate an index of Sungazer burrow occupancy from previous studies and estimate Sungazer population density.

2.2. Methods

2.2.1. Identifying Sungazer burrows

Sungazer burrows were identified by A) their distinctive ovular shape, B) the ~3 cm wide ridge running along the centre of the burrow floor and C) the bare patch of the earth where Sungazers forage and bask (Fig. 2.1). These features allow Sungazer burrows to be distinguished from those of South African Ground Squirrels (*Xerus inauris*), Springhare (*Pedetes capensis*), Slender Mongoose (*Galerella sanguinea*), Yellow Mongoose (*Cynictis penicillata*) and Suricate (*Suricata suricatta*), which all occur commonly across the range of the Sungazer (Van Wyk, 1992; Ruddock, 2000; Stuart and Stuart, 2007).

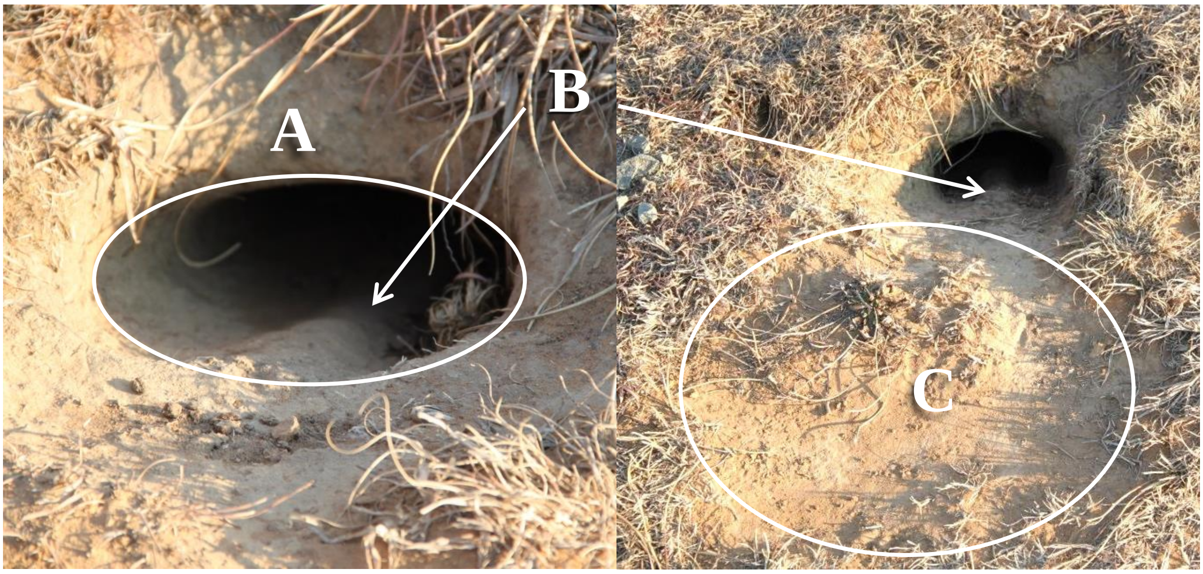


Figure 2.1. Photographs of Sungazer burrows showing a) the distinct ovular shape b) central ridge and c) bare patch preceding the burrow entrance.

Burrows were classified as active if a) there were claw and/or tail marks in the bare earth in front of the burrow entrance (Fig. 2.1C) (Auffenberg and Franz, 1982), b) Sungazer tissue was found at the mouth of the burrow and c) the burrow entrance was unobstructed by debris. If the state of occupancy of a burrow was questionable, a camera was inserted into the burrow and the footage was used to assess the presence of Sungazers (*sensu* Breininger *et al.*, 1991).

2.2.2. 1 ha Quadrat methodology

A standard quadrat for measuring burrow density was 1 ha. This was walked for 1 hour at a standard pace in parallel line transects at 5 m intervals (Fig. 2.2). I used a Garmin GPSmap 78s (datum WGS1984) with real-time path tracking to guide the length and spacing of transects. I recorded GPS co-ordinates and elevation at every burrow found. The location of each burrow was viewable on the GPS, allowing for the exclusion of burrows being counted more than once.

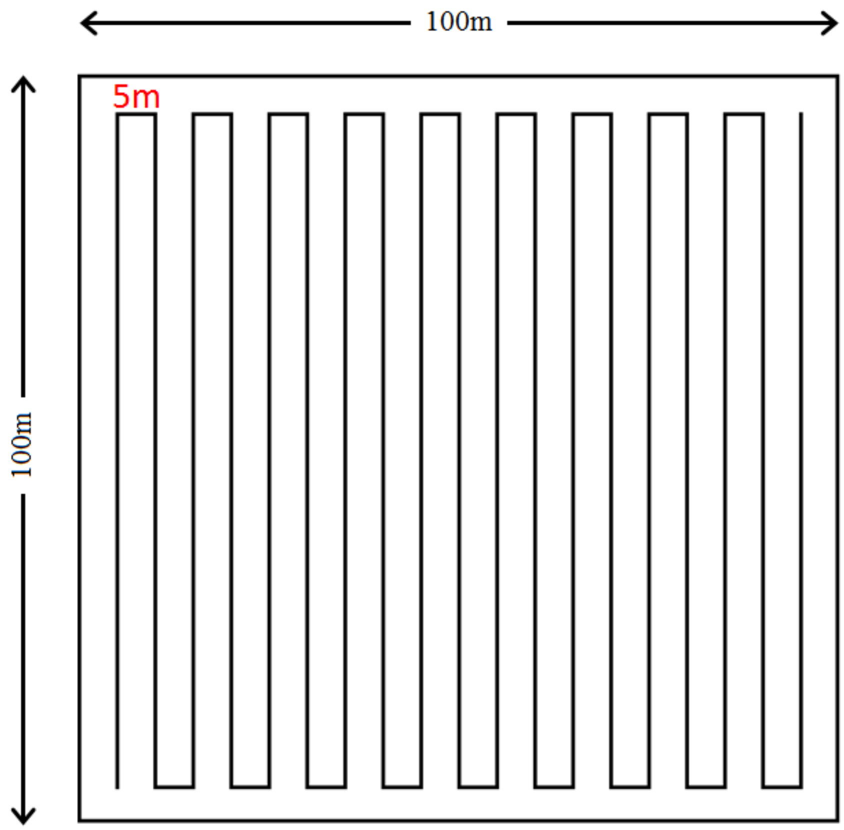


Figure 2.2. One hectare sampling plot showing 5 m line transects.

2.2.3. Sample sites

I quantified Sungazer burrow density at 80 sample sites across the distribution of the species (Fig. 2.3). Sample quadrats surveyed were a combination of targeted sites and random sites. Targeted sites were areas on farms where Sungazers were known to occur at the time of the survey, and the location of Sungazer colonies were indicated by the landowner. Random sites were areas on a farm within the distribution that were randomly selected on a map prior to being surveyed. These sites were then navigated to using a GPS, after obtaining the landowner's permission. The site categories are explained in Table 2.1., and explanations for sampling site selections are found in the relevant Chapters.

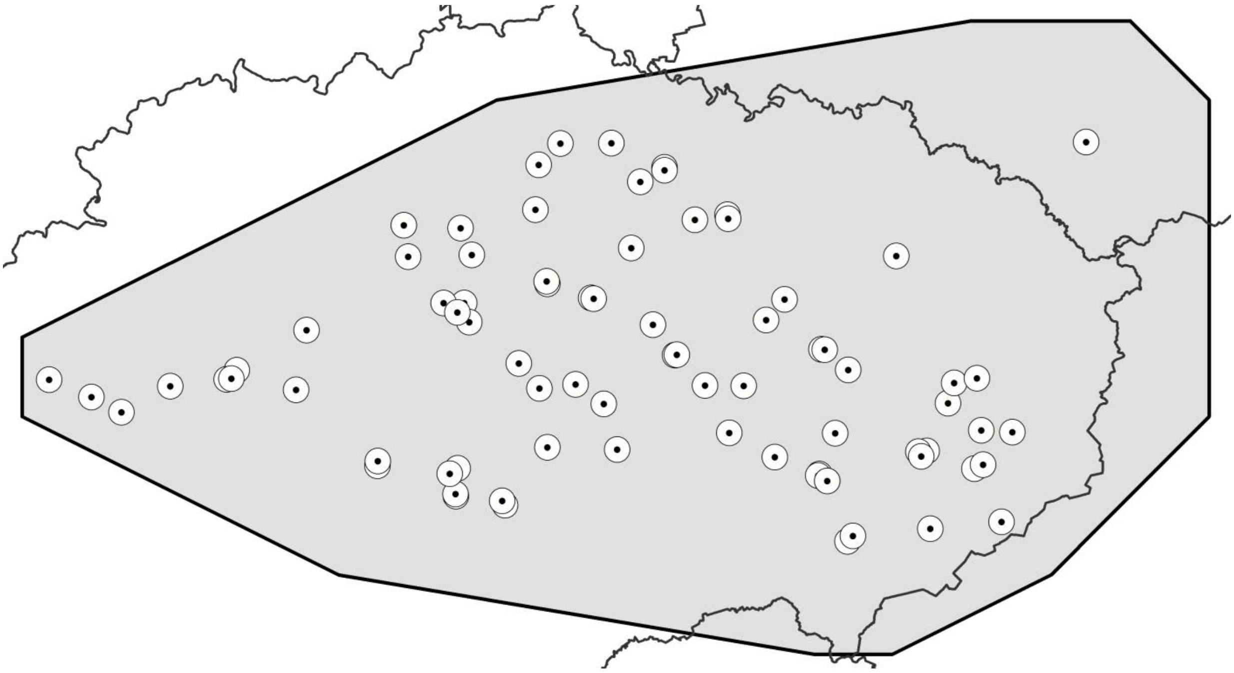


Figure 2.3. Location of 80 occurrence sites across the distribution of the Sungazer where burrow density was recorded in 1 ha plots (EOO polygon as defined in Chapter 3).

Table 2.1. Description of site categories and number of sites per category where burrow density was quantified.

Site description	**Code**	**Number of sites**	**Detailed in**
Targeted sites on farms evenly spread across the distribution	SP	18	Chapter 3
Targeted sites on farms where the species was previously recorded	DWP	30	Chapter 4
Targeted sites on farms where the species was previously unrecorded	P/E/DWN	17	Chapter 4
Targeted sites on randomly selected farms across the distribution	RA	10	Chapter 3
Randomly selected sites on targeted farms across the distribution	RB	5	Chapter 3

2.2.4. Burrow count accuracy

The distance at which burrows were observed from the line transect was measured for 116 burrows. This was done to quantify the average distance from which Sungazer burrows were observed, and the rate of detection decay. Sampling accuracy could therefore be estimated by quantifying the mean distance at which burrows were observed. The data were normally distributed (SW-W = 0.9003, $p < 0.001$).

2.2.5. Burrow photographs

Geo-tagged photographs of all recorded burrows, and four photographs of the surrounding landscape (orientated north, west, south and east) were taken to collect a visual archival record of burrow entrance structure, and suitable Sungazer habitat.

2.2.6. Burrow occupancy index (BOI)

In order to relate Sungazer burrow densities recorded in this study to Sungazer population density, the relationship between number of Sungazers and number of burrows was considered. Since I did not quantify Sungazer burrow occupancy in this project, I used measures from published studies on burrow occupancy measured at three sites across the distribution (Jacobsen *et al.*, 1990; Van Wyk, 1992). The number of observations made at these sites varied and I thus weighted scores according to the number of observations made. An overall weighted mean was then derived using the formula below.

$$\text{Weighted metascore for mean } \textit{S. giganteus} \text{ burrow occupancy} = \frac{A_n . A_{MBO}}{(A_n + B_n + C_n)} + \frac{B_n . B_{MBO}}{(A_n + B_n + C_n)} + \frac{C_n . C_{MBO}}{(A_n + B_n + C_n)}$$

A_n, B_n, C_n = number of observations per site

A_{MBO}, B_{MBO}, C_{MBO} = mean burrow occupancy at each site

2.2.7. Sungazer population density

Previous studies recorded burrow density at various sites, and population densities can be estimated from these figures using the burrow occupancy index calculated above. The number of observations varied between sites, and the mean population density of each study was weighted according to the number of observations made. The mean population density aggregate from previous studies provides a comparison for the mean population density measured in this study.

$$\text{Weighted metascore for mean } \textit{S. giganteus} \text{ population density} = \frac{A_n . A_{BD}}{(A_n + B_n + C_n)} + \frac{B_n . B_{BD}}{(A_n + B_n + C_n)} + \frac{C_n . C_{BD}}{(A_n + B_n + C_n)}$$

A_n, B_n, C_n = number of observations per site

A_{BD}, B_{BD}, C_{BD} = mean population density at each site

2.3. Results

2.3.1. Burrow count accuracy

Sungazer burrows were observed from a mean distance of 3.6 m ± 0.2/2.15 m (SE/SD) from the burrow entrance (Fig. 2.4). The median distance at which burrows were observed was 3 m, and 82.76% of burrows were observed within 5 metres of the line transect. The apparent reduced observability at less than a metre is a result of most burrows being observed before the observer can get within a metre of the burrow. The furthest distance at which a burrow was observed was 10.5 m. At distances further than 10.5 m, burrow entrances are obscured by grass and are not visible on foot. Occasionally, Sungazers were observed basking at the burrow entrance and the burrows were then approached and recorded. These records were not taken into account as they did not provide a true measure for assessing the distance at which the burrow was observable.

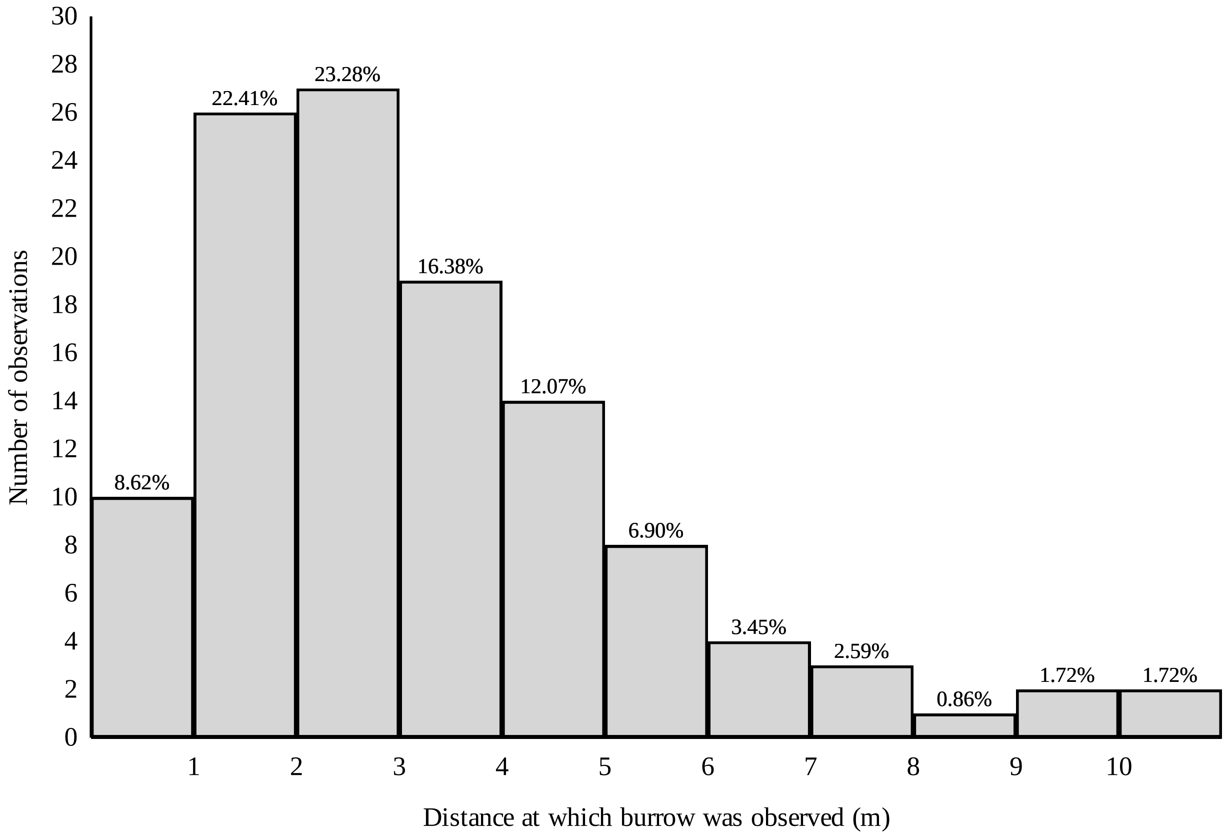


Figure 2.4. Frequency of distances at which Sungazer burrows were observed.

2.3.2. Elevation

Burrows were recorded between altitudes of 1260-1840 m asl (Fig. 2.5). The lowest elevation records for the species were on the western edge of the distribution in the districts of Welkom, Odendaalsrus and Wesselsbron, while the highest elevations were in Bethlehem, Reitz and Harrismith in the east. The elevations at which burrows were found were normally distributed (Shapiro-Wilk = 0.96, $p < 0.001$).

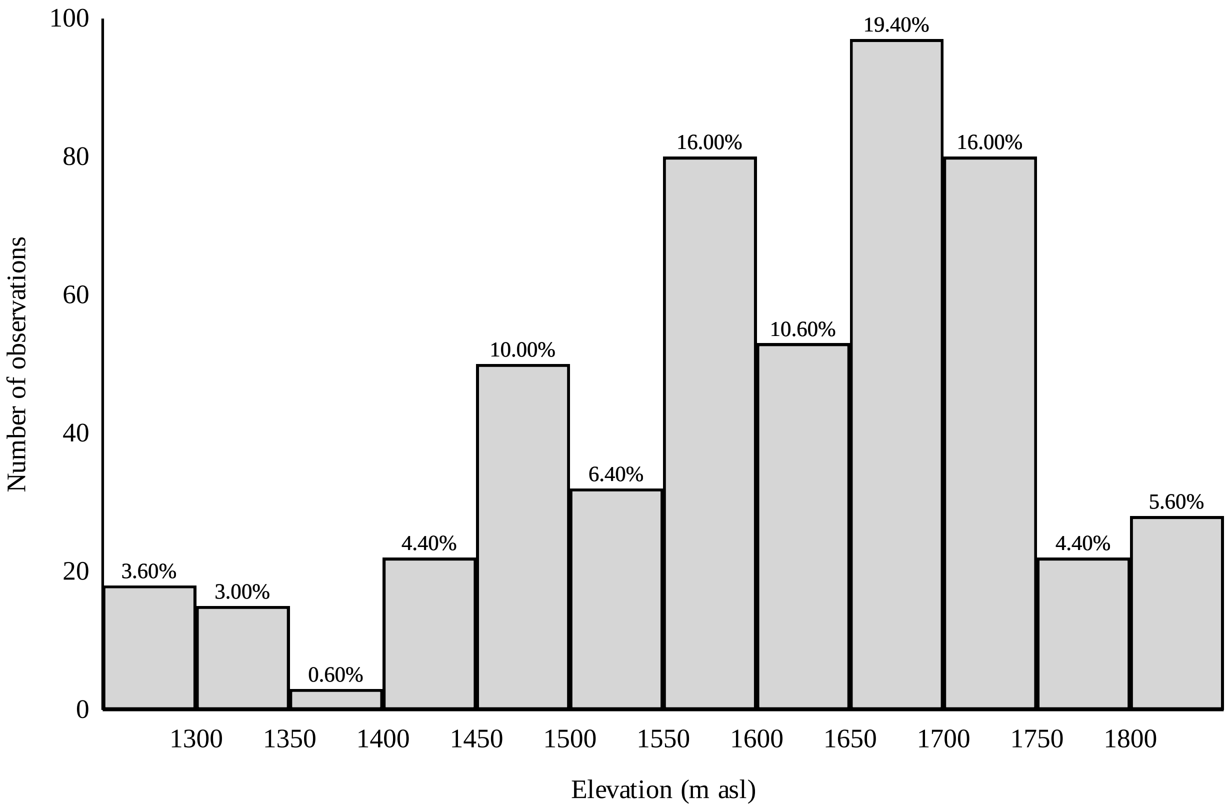


Figure 2.5. Distribution of Sungazer burrows as a function of elevation.

2.3.3. Burrow density

A total of 491 burrows were recorded at 80 occupied sites, yielding the mean density of 6.14 ± 0.87 burrows/ha. The number of burrows per hectare ranged from 1 to 16. Data for burrow density were normally distributed (SW-W = 0.9, $p < 0.001$). Lower population densities (< 6 burrows/ha) are more common than high density populations (> 6 burrows/ha), with a trend showing a decrease in sites recorded for higher density populations (Fig. 2.6). The most frequently encountered burrow densities were 1 and 5 burrows/ha.

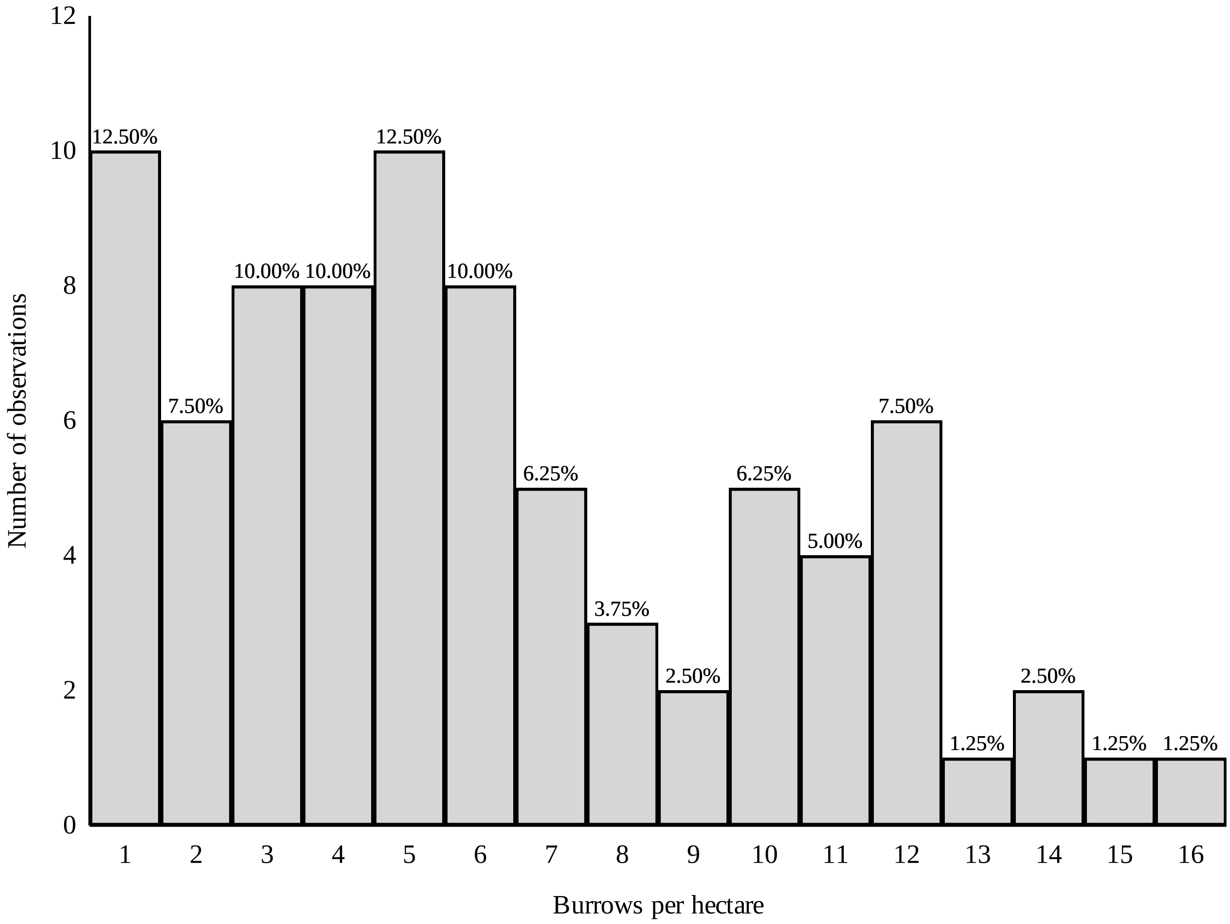


Figure 2.6. Frequency of Sungazer burrow densities per hectare.

2.3.4. Burrow occupancy index (BOI)

Using the weighted metascore formula for burrow density, the mean number of Sungazers per burrow was calculated as 1.83 Sungazers/burrow (Table 2.1). Jacobsen *et al.*, (1990) recorded burrow occupancy for 841 burrows, contributing 73.8% of data to the metascore calculation. Burrow occupancy records from Van Wyk (1992) contribute the remaining 16.2% to the calculation at 15.8% (Greenlands) and 10.4% (Middelpunt) each. Mean burrow occupancy ranged from 1.42-2.11 Sungazers/burrow. Jacobsen *et al.*, (1990) and Van Wyk (1992) recorded the frequency of burrow occupancy of all burrows excavated and the results are presented in Figure 2.7. Approximately half of all burrows were occupied by a single adult (51-53%), a quarter (22-25%) contained two individuals, 14-16% contained three individuals, 5-7% contained four individuals, 2% contained five individuals and burrows containing six, seven and eight individuals each made up less than 1% of the total burrows excavated.

Table 2.1. Weighted metascore of mean Sungazers per burrow derived from the literature.

Study	**Mean burrow occupancy**	**No. burrows recorded**	**% of total burrow count**	**Weighted contribution**
Jacobsen *et al.*, (1990)	1.83	841	73.8	1.35
Van Wyk (1992) - Greenlands	2.11	180	15.8	0.33
Van Wyk (1992) – Middelpunt	1.42	118	10.4	0.15
Total		**1139**	**100**	**1.83**

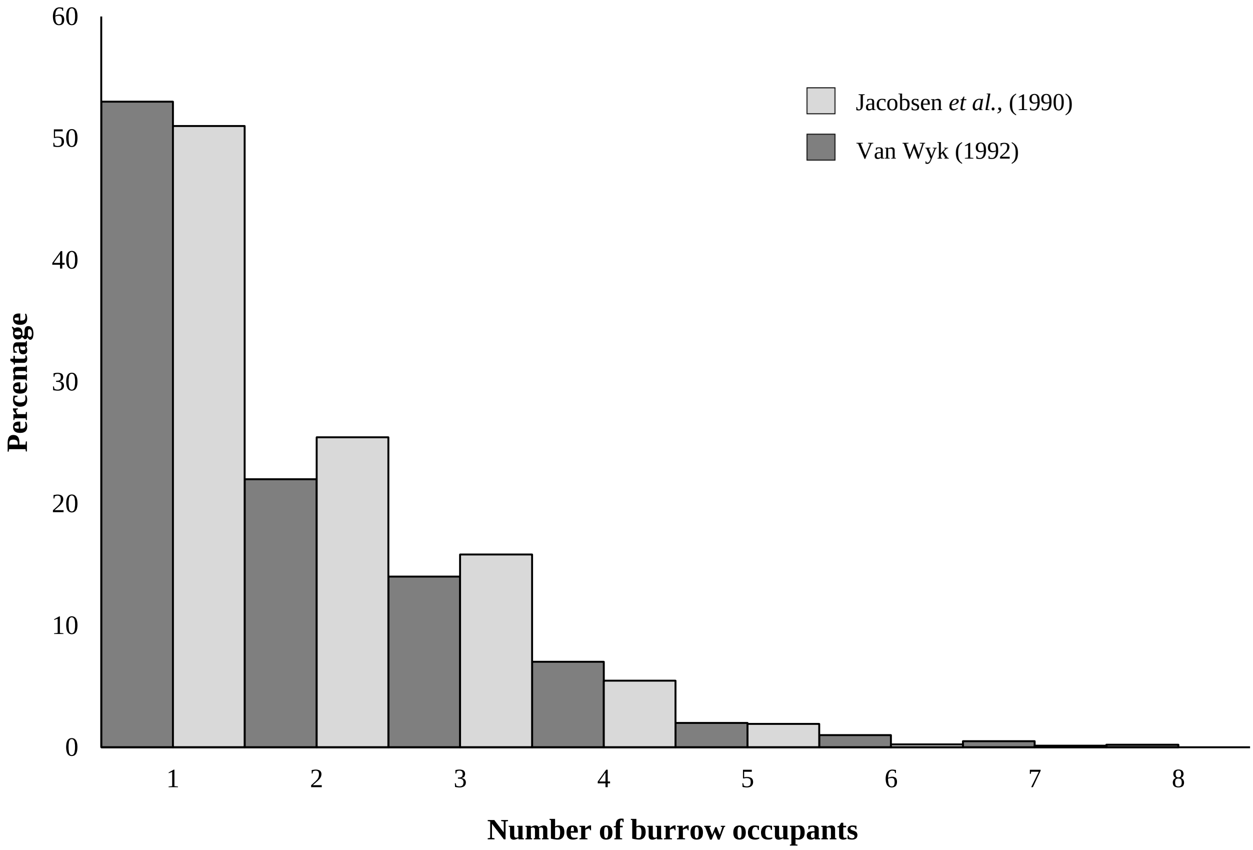


Figure 2.7. Frequency of number of burrow occupants from previous studies (n=841 burrows, Jacobsen *et al.*, 1990; n=273 burrows, Van Wyk, 1992).

2.3.5. Sungazer population density

Mean Sungazer population density calculated in my study was 11.24 ± 7.27 individuals/ha. The lowest Sungazer population density measured was 1.83 lizards/ha, while maximum population density recorded in this study is 29.28 lizards/ha. The range calculated using standard deviation for burrow density is 4.0-18.51 individuals/ha.

Table 2.2. Weighted metascore of mean Sungazer burrow density derived from the literature.

Study	Mean burrow density	No. burrows recorded	% of total burrow count	Weighted contribution
Jacobsen *et al.*, (1990)	4.04	580	66.06	2.67
Van Wyk (1992) - Greenlands	5.1	180	20.50	1.05
Van Wyk (1992) - Middelpunt	6.8	118	13.44	0.92
Total		**878**	**100**	**4.63**

2.4. Discussion

Density of Sungazer populations across the distribution was generally low (mean = 11.24 ± 7.27 lizards/ha), but varied widely across the 80 sites surveyed (range = 1.83-29.28 lizards/ha). Burrows were easily identifiable and most (82.76%) were detected within the 5 m line transects of the 1 ha sampling plot. Given the mean distance at which burrows were identified (3.55 m), I am confident that all burrows within the sampling plot were recorded. Colonies appear to be restricted between altitudes of 1260-1840 m asl, and my measures extend the known elevation range by 12.8%. The majority (60%) of burrow were recorded between 1550-1750 m asl. The lowest altitudes where Sungazers occurred were on the western edge of the distribution in the districts of Welkom, Odendaalsrus and Wesselsbron, while the highest altitudes were in Bethlehem, Reitz and Harrismith in the east.

The findings of this study represent one of the most robust measures of population density of any South African lizard species. Population density at a site is a product of a complex interaction of environmental variables, and measures from one area are therefore not necessarily representative of the typical density of the species. My measures of burrow density spanned the distribution of the species, and the mean density calculated, controlled for environmental variability between sites. One potential weakness in the calculation is that the BOI was based on data from only three sites that were likely selected as study sites due to the abundance of lizards. It is therefore possible that burrow occupancy in these areas represents the upper limit of the range, potentially skewing the BOI towards higher occupancies, and overestimating population density. Although the mean occupancy rates are consistent between studies, with a clear pattern emerging (Fig. 2.7), the reliability of this measure in calculating population densities can only be tested when more comprehensive studies on distribution wide burrow occupancy are conducted. Such a study should quantify burrow occupancy at representative sites across the distribution, similarly to how burrow density was quantified in this study. The relationship between number of burrows and number of lizards within an area is a temporally variable measure that is influenced by a combination of density-dependant factors at a point in time. For instance, variables that allow for the proliferation of prey items might result in higher reproductive rate over a period, and therefore an increased lizard:burrow ratio for that period. It is therefore also important for a study to assess burrow occupancy throughout the year, so that these variations can be taken into account.

The mean burrow density calculated in this study (6.14 burrows/ha) falls within the upper end of the range of mean burrow densities recorded for the species in previous studies (4-6.8 burrows/ha, Stolz and Blom, 1981; Jacobsen *et al.*, 1990; Van Wyk, 1992), and is higher than the mean burrow density calculated from these studies. The range of burrow densities recorded (1-16 burrows/ha) was similar to the range reported by Van Wyk (1992) (1-19 burrows/ha). Similarly to this study, Van Wyk (1992) noted that despite high densities in some areas, large tracts of adjacent land were found with no burrows, and concluded that the distribution of burrow density appears to be close to random. Other cordylid species have been found to occur at a wide range of population densities (*Cordylus cordylus* = 4.2-288 lizards/ha; Burrage, 1974, *Cordylus macropholis* = 38.1-77.1 lizards/ha; Nieuwoudt *et al.*, 2003), with higher mean densities (*Cordylus cordylus* = 146 lizards/ha; Burrage, 1974, *Cordylus macropholis* = 58 lizards/ha; Nieuwoudt *et al.*, 2003), than *S. giganteus*. *Smaug giganteus* is one of few burrowing species of cordylid; most other members of the family are rupicolous, and significantly smaller in size. Direct comparisons to population densities of other cordylid species may therefore not be ecologically relevant in contextualising Sungazer population density.

Other lizard species of a comparable body size to *S. giganteus* have been recorded as occurring at similarly low population densities, such as *Sauromalus obesus* (SVL = 170 mm, mean PD = 19 lizards/ha; Kwiatkowski and Sullivan, 2002), *Dipsosaurus dorsalis* (SVL = 130 mm, PD = 22.4 lizards/ha; Alberts, 1993), and *Crotaphytus collaris* (SVL = 100 mm; PD = 17.2 lizards/ha; Abts, 1987). The larger *Uromastyx aegyptius* (SVL = 350 mm) occurs at the population density of 4.4 – 6.3 lizards/ha in the arid Kuwait desert (Robinson, 1995). However, many lizard species of similar body size to the Sungazer occur at significantly higher population densities, such as *Egernia cunninghami* (SVL = 180 mm) which occurs at local population densities of 85 lizards/ha on mainland Australia (Barwick, 1965), but up to 368 lizards/ha on West Island, South Australia (Van Weenen, 1995). *Anolis bimaculatus*, which reaches a maximum SVL of 114 mm has been recorded as occurring at population densities of 1042-1667 lizards/ha (Diaz *et al.*, 2005). While body size might function as an upper threshold to population density, a complex interaction of a multitude of density-dependant factors such as competition and food availability are likely to play a more functional role in moderating population density (Pianka, 1973; Turner, 1977; Hengeveld, 1990). Shuttleworth (2006) found that population density of *Oroborous cataphractus* (SVL = 85 mm), a moderate-sized rupicolous cordylid, is strongly correlated with the population density of their primary prey, the termite species *Microhodotermes viator*. Similarly, Dunham (1982) recorded *Urosaurus ornatus* occurring at population densities of 39-60 lizards/ha in some areas, and 286-721 lizards/ha in others, and found that population density correlated strongly with precipitation levels, which are linked to availability of their food resources (Dunham, 1981). The sit-and-wait foraging strategy of the Sungazer requires high prey densities, high prey mobility and low energy demand by the lizard (Schoener, 1969; Schoener, 1971). Given the foraging strategy of the Sungazer and the wide variation in population densities across the distribution, it is possible that population densities are limited by prey density.

In conclusion, my estimate of burrow density for Sungazers across the distribution is similar to the previous published measures (Jacobsen *et al.*, 1990; Van Wyk, 1992). Estimates of population density for the species were derived from burrow occupancy data at three different sites, representing a total of 1 139 burrows. These data present a clear picture of the trends of burrow occupancy, but assessing burrow occupancy across the distribution of the species might reveal a trend free from density-dependant factors that influence population density at a site. I found that counting burrows is a rapid and simple alternative to capture techniques when quantifying Sungazer population density. The technique is recommended as a standardised sampling tool for assessing Sungazer burrow density in future studies, and further contributes to the growing number of studies that recommend this technique as an alternative to other more time consuming, laborious and inaccurate measures of population density (Beck-King *et al.*, 1990; Govender and Rodriguez-Fourquet, 2004).

CHAPTER 3:
QUANTIFYING EOO, DISTRIBUTION AND AOO

3.1. Introduction

The geographic range of a species is a fundamental aspect of its evolutionary history and ecology, and along with an understanding of natural and human-induced changes over time, can reflect the extinction risk it faces (Gaston, 2003; Gaston, 2009; Gaston and Fuller, 2009; Tella *et al.*, 2013). As such, approximately 50% of conservation assessments of mammal, amphibian, bird and gymnosperm species have been based solely on geographic range (IUCN, 2007). Geographic range however, can be interpreted in more than one way (Gaston, 1991). Definitions have been as coarse as a list of countries in which the species is found (Corbet and Hill, 1990) or the number of degrees of longitude and latitude where the species has been recorded (Reaka, 1980). Finer scale attempts have represented geographic range as the number of quadrats (of varying sizes) occupied by the species (Schoener, 1987; Ford, 1990), or the total number of localities at which the species has been recorded as occurring (Gaston and Lawton 1988). Arriving at standard approaches to quantify geographic range is important for understanding how the geographic range of a species reflects its relative vulnerability to extinction. Gaston (1991) distinguished two distinct measures to meet this need for standardisation: extent of occurrence (EOO) and area of occupancy (AOO) (Fig. 3.1). Both of these measures have since been widely employed in pure and applied contexts (Gaston and Fuller, 2009).

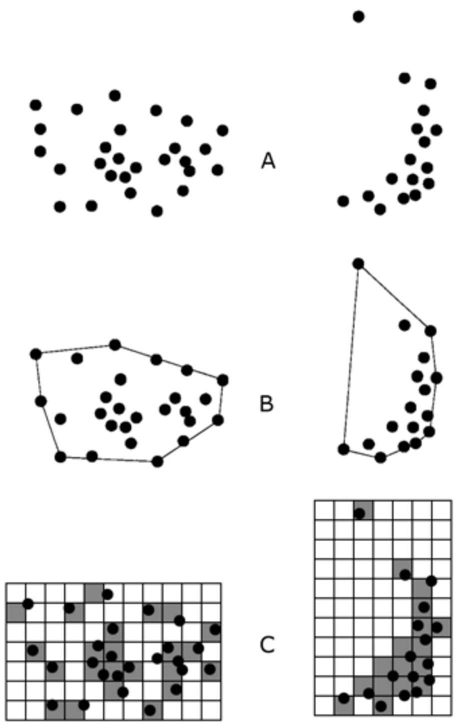


Figure 3.1. Two examples illustrating EOO and AOO. (A) distribution of records of occurrence (B) minimum convex hulls around records of occurrence (C) a measure of area of occupancy achieved by summing the area of the occupied grid squares (adapted from IUCN, 2012).

EOO is defined as the area within the outermost limits of occurrence (Gaston, 1991; IUCN, 2012), and can be delineated by a minimum convex hull that contains all occurrence points (Gaston, 1991; Hartley and Kunin, 2003) (Fig. 3.1). The EOO of a species is not a measure of its distribution however, as it typically contains tracts of unsuitable and unoccupied habitat (Gaston and Fuller, 2009), and is greatly affected by outliers of populations of individuals (Gaston 1994a; IUCN, 2012). Rather, EOO is a measure of the spatial spread of occurrence records of a species, and is used to assess the likelihood of simultaneous extirpation of all populations as a result of stochastic or directional threat processes (SPWG, 2006; Gaston and Fuller, 2009). Some studies have attempted to remove discontinuities in occurrence and unsuitable habitat from their calculations of EOO (Lewison and Oliver, 2008; Marino and Sillero-Zubiri, 2011), but this misrepresentation of EOO biases measures towards a measure of AOO. It is recommended that discontinuities be included in EOO (Gaston and Fuller, 2009), particularly in the assessment of threatened species, where the degree of spatial risk spreading can be interpreted (Gaston and Fuller, 2009).

The AOO provides a measure of occupancy of a species within the EOO, and can be calculated as the total area of grid cells that contain occurrence records, the proportion of the EOO in which the species occurs, the total area used by populations or the amount of suitable habitat (Gaston and Fuller, 2009 and references therein). Measures of AOO are highly dependent on the resolution of distribution data and the method of measurement (Gaston 1991; Keith *et al.*, 2000). The use of large grid cells may result in an overestimation the AOO (Keith *et al*., 2000; WCU, 2001), while small grid cells tend to produce values that correlate with population counts (He and Gaston, 2000). The scale and intensity at which AOO is measured is influential in whether IUCN Red List thresholds are met (Keith *et al.*, 2000). AOO generally correlates with population size, and provides a useful estimate of extinction probability based on changes in within-range habitat extent, fragmentation or suitability, and demographic processes (Joseph and Possingham, 2008).

The first attempts at mapping the geographic range of *Smaug giganteus* were undertaken in 1978 in the Free State Province (De Waal, 1978), and in 1989 in Mpumalanga (Jacobsen, 1989). These surveys were conducted at different resolutions, with the Mpumalanga survey covering several farms in every QDGC in the province (Jacobsen, 1989). The Free State survey sampled farms in every alternate Eighth Degree Cell in a checkerboard pattern, and as a result presented the Free State distribution of the species as a scattered set of localities (Fig. 3.2), with blank areas not necessarily representing absences. Since the initial mapping, a combination of records from EIAs, scientific studies, museum specimens and the SARCA project have added distribution data to the database. However, there are still conspicuous absences in-between occurrence records. At this point, it is not known whether these gaps represent real disjunctions in the distribution of the species, or a result of the survey strategy of De Waal (1978), as there have been no systematic surveys of the species since.

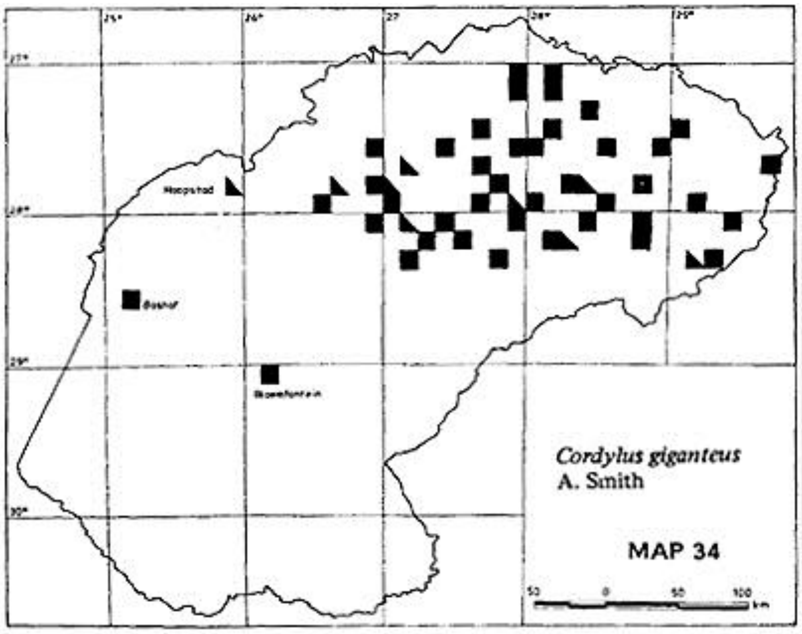


Figure 3.2. Map of the Free State Province showing the initial mapping of *Smaug giganteus* occurrence by De Waal (1978) during an extensive herpetological survey (Source: De Waal, 1978).

The EOO polygon that portrays the limits of distribution has not been calculated for *Smaug giganteus* as per the IUCN definition, excluding the species from being assessed based on the geographic criteria. The same applies to the AOO, with the most recent estimations based on a qualitative assessment of satellite imagery as opposed to the direct evaluation of occupied habitat. An empirical quantification of these aspects of the geographic range of the species is necessary to assess the species in context of the risk it faces from spatial perspective. The proportion of natural land cover remaining across the distribution of the species is not known, and this hinders a realistic assessment of the extinction threat to the species. Furthermore, even in areas where the species is recorded as occurring, the lizards do not occur ubiquitously and uniformly across the landscape (Branch, 1998), and areas of high population density are often surrounded by large areas without Sungazers (Van Wyk, 1992). Without a realistic estimate of the area occupied by the species, it is impossible to estimate the number of Sungazers remaining in the wild.

In this Chapter, I aim to quantify the EOO, further an understanding of the spatial distribution of the species, quantify the proportion of natural land cover across the distribution of the species, and finally, to quantify the AOO.

Aims

1) Define and calculate the EOO of *S. giganteus*
2) Quantify the spatial distribution of the species within the EOO
3) Calculate the area of natural land cover within the EOO and specifically the distribution
4) Calculate the AOO of the Sungazer

3.2. Methods

3.2.1. Extent of Occurrence (EOO)

The EOO typically includes all known point localities recorded for a species within the shortest imaginary boundary (IUCN, 2012). However most records for *Smaug giganteus* that lie on the outer edge of the distribution are at QDGC resolution (SARCA REF?), with no GPS co-ordinates provided. In lieu of a finer resolution of locality records for the species across the distribution, I used the QDGCs containing occurrence records (Mouton, 2014) as the smallest unit of occurrence. A minimal convex hull (IUCN, 2013) was created in ArcMap 10.0. around the outermost QDGCs in which *S. giganteus* has been recorded (Mouton, 2014). The dataset included 229 records from 54 QDGCs. Records from the KwaZulu-Natal province and Lesotho were excluded on the basis of being questionable (Mouton, 2014) or confirmed as false (Armstrong, 2011). I used the Calculate Areas tool in ArcMap 10.0. to measure the area of the EOO polygon.

3.2.2. Spatial distribution

To augment the current mapping of *Smaug giganteus*, I visited 42 QDGCs in a checkerboard pattern across the EOO of the species (Fig. 3.3). I designed the sampling regime such that my sites covered as many QDGCs within the EOO in which the species has not previously been recorded. I targeted farms near the centre of QDGCs (at the intersect between two perpendicular lines drawn from corner to corner of a QDGC), and if these farms were not accessible, I visited the closest adjacent farms. At each site, I asked landowners about the presence of Sungazers on their property. Sungazers and their burrows are visually conspicuous in the grassland, and their presence or absence is known by landowners. If the species was confirmed as occurring on the property, a 1 ha plot was surveyed as described in Chapter 2 to record burrow density.

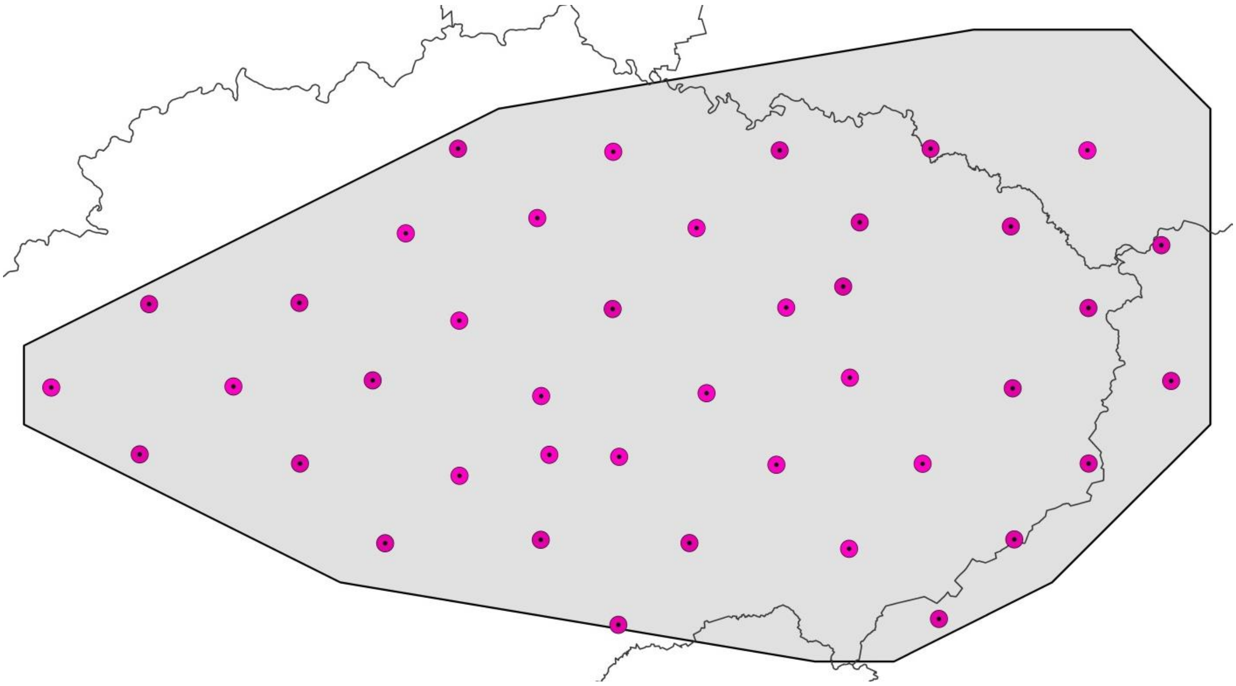


Figure 3.3. Sample sites in alternate QDGCs in a checkerboard pattern across the EOO (minimum convex hull) of *Smaug giganteus*.

Ad hoc sites

During the course of fieldwork, 42 additional '*ad hoc* sites' were surveyed to further augment the mapping of *S. giganteus* across the EOO. These sites were a combination of sites aimed specifically at QDGCs where Sungazers had not been recorded before, and sites where I thought it likely that Sungazers were present based on habitat characteristics (flat/gentle slope, primarily *Themeda* cover, termitaria present indicating that the land had not been previously ploughed).

Total area of distribution

The QDGCs where Sungazers were found in this study were combined with QDGCs where the species has been found previously into a polygon shapefile in ARCMap 10.0. As the species does not occur in Lesotho or the KwaZulu-Natal province (Armstrong, 2011; Mouton, 2014), and there are substantial physical barriers (Drakensberg mountain range) that lie along the political boundaries, the portions of the QDGCs that fall within these areas can be removed from the distribution. I clipped the area of the 60 QDGCs against the Free State and Mpumalanga provinces, and used to the Calculate Areas tool in ArcMap 10.0. to calculate the area of the distribution.

3.2.1. Land cover

To measure the proportion of natural land cover within the EOO and distribution of the species, I clipped the most recent South African land cover GIS map (NLC 2009) with the EOO polygon, and distribution polygon (including new records from this study) in ArcMap 10.0. The 2009 land cover

GIS layer did not include land cover for Lesotho, a portion of which falls within the EOO and distribution. I used the previous iteration of the national land cover map (NLC2000) which included land cover information for Lesotho, and clipped the relevant portion of Lesotho against the EOO and distribution. I used the attributes table to quantify the contribution of each land cover type to the land cover of the EOO and distribution.

3.2.2. Area of Occupancy (AOO)

The area of occupancy represents the actual area of land within the EOO that is occupied by a species. In order to assess this, I calculated the proportion of natural area being occupied by Sungazers within the known distribution. I visited 120 sites within the EOO, 98 of which fell within the distribution of the species (Fig. 3.4). Farms 10 km north, north-west, north-east, west, south-west, south, south-east and east of the towns of Harrismith, Kestell, Bethlehem, Verkykerskop, Warden, Vrede, Reitz, Lindley, Edenville, Kroonstad, Koppies, Heilbron, Frankfort, Petrus Steyn, Memel were sampled. Although this sampling regime is not random in the placement of sites, it is random in the sense that the land cover type at a particular site on a farm, suitability of the area for Sungazers and presence of the lizards themselves were not known prior to the visit. A 1 ha plot was randomly selected within each farm on a map. This site was then surveyed for the presence of Sungazer burrows as described in Chapter 2. If the area was natural, but unambiguously unsuitable for the species i.e. extremely rocky, montane, unsuitable aspect or vegetation type, the 1 ha plot was not surveyed as described above, but marked as a negative site. If the area was transformed (i.e. cultivated, disturbed, developed), another random site within the farm was selected, since the aim of the exercise was to assess the proportion of natural landscape occupied by Sungazers.

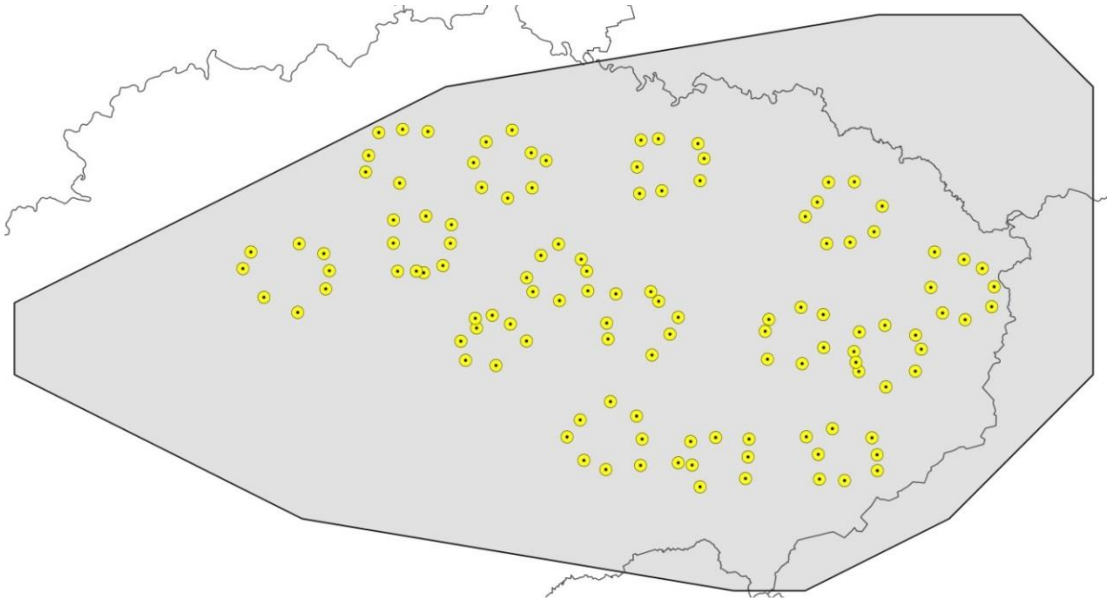


Figure 3.4. Location of 120 random sites surveyed for *Smaug giganteus* presence for AOO calculation, within EOO (minimum convex hull).

3.3. Results

3.3.1. Extent of Occurrence

The area of the minimum convex hull representing the EOO of *Smaug giganteus* was measured as 58 338 km^2 (Fig. 3.5). This comprises 72 complete QDGCs and parts of 27 QDGCs.

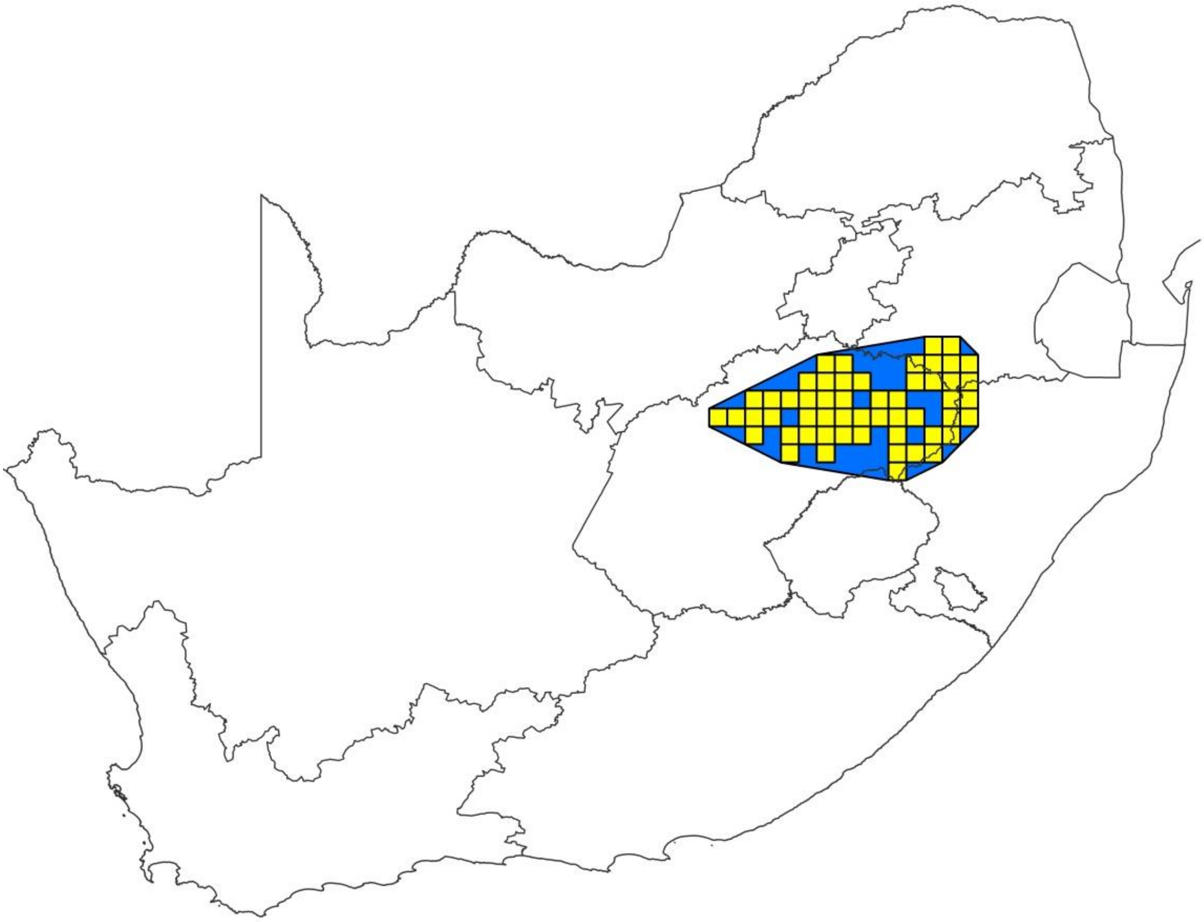


Figure 3.5. Map of South Africa showing the minimum convex hull of *Smaug giganteus* EOO (minimum convex hull) around the QDGC distribution records.

3.3.2. Spatial distribution

I found Sungazers at 18 out of 42 sites surveyed (Fig. 3.6). Three of these sites were in QDGCs where the species has not previously been recorded (2727AD, 2828BA, 2829AA).

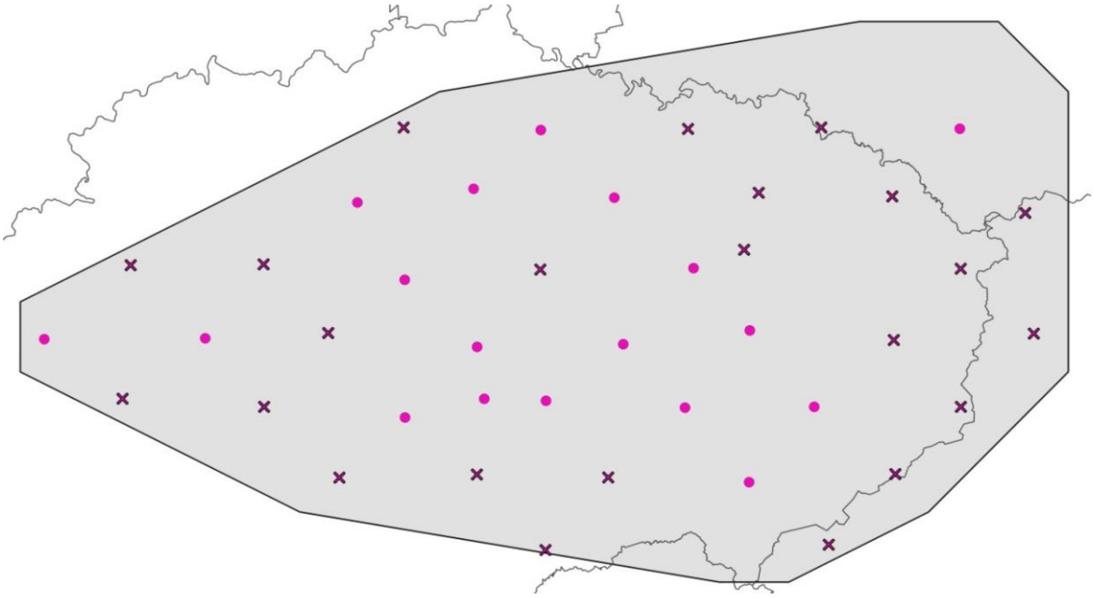


Figure 3.6. *Smaug giganteus* EOO (minimum convex hull) showing presence (O) and absence (X) of Sungazer populations at sites near the centre of every alternate QDGC.

Ad hoc sites

I found Sungazers at 11 out of 42 *'ad hoc sites'* surveyed (Fig. 3.7). Three of these sites were in QDGCs where the species has not been recorded before (2827BC, 2728AB, 2729CD).

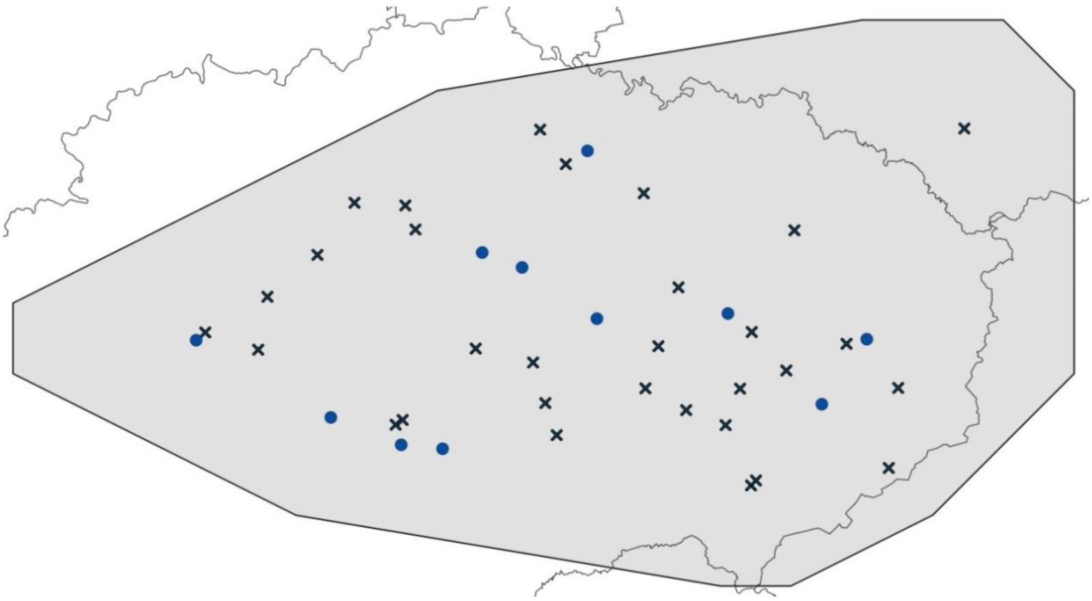


Figure 3.7. *Smaug giganteus* EOO (minimum convex hull) showing presence (O) and absence (X) of Sungazer populations at *'ad hoc'* sites.

Total area of distribution

The total area of all 60 QGDCs is 40 969 km^2 (Fig. 3.8). The 6 new QDGC records do not affect the area of the EOO polygon calculated above. Within South Africa, 33 491 km^2 of the EOO is in the Free State Province, and 4 705 km^2 is in Mpumalanga Province. The area of the distribution of the species is taken as the sum of the areas of QDGCs and portions of QDGCs that fall within Free State and Mpumalanga provinces, which is 38 196 km^2.

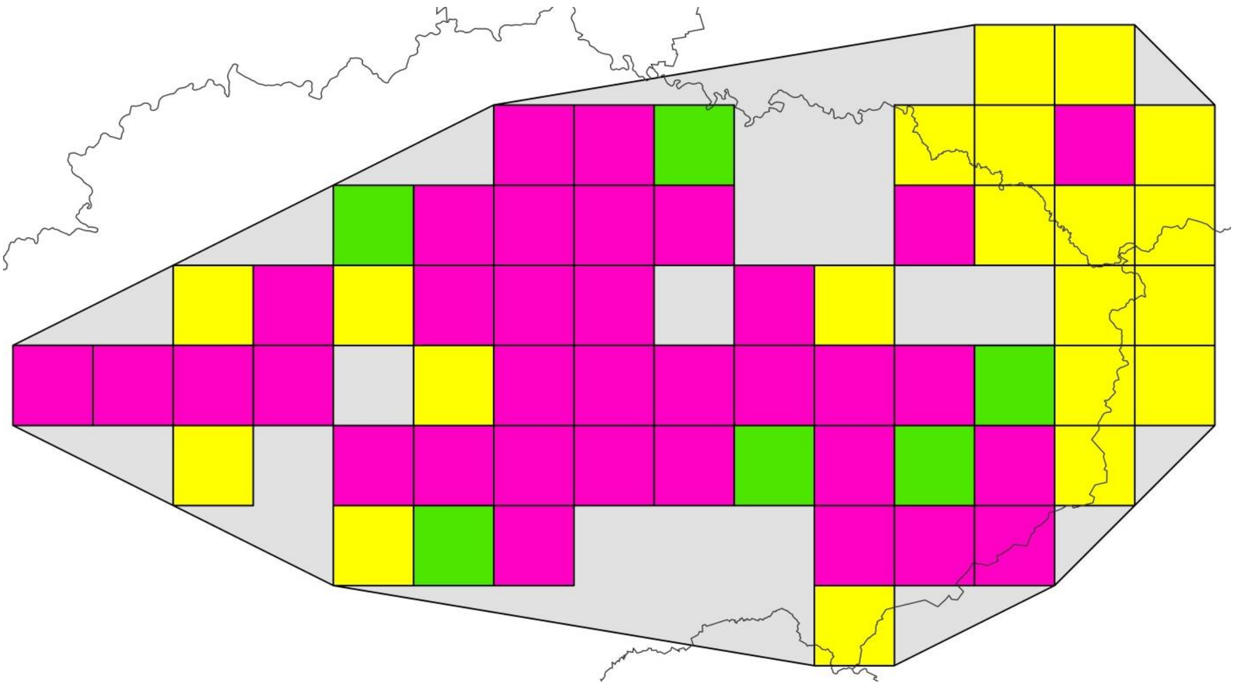


Figure 3.8. *Smaug giganteus* EOO (minimum convex hull) showing 1) QDGCs where the species was recorded in this study (Pink), 2) QDGCs where the species was not recorded in this study but has been recorded previously (Yellow) and 3) new QDGC records for the species found during this study (Green).

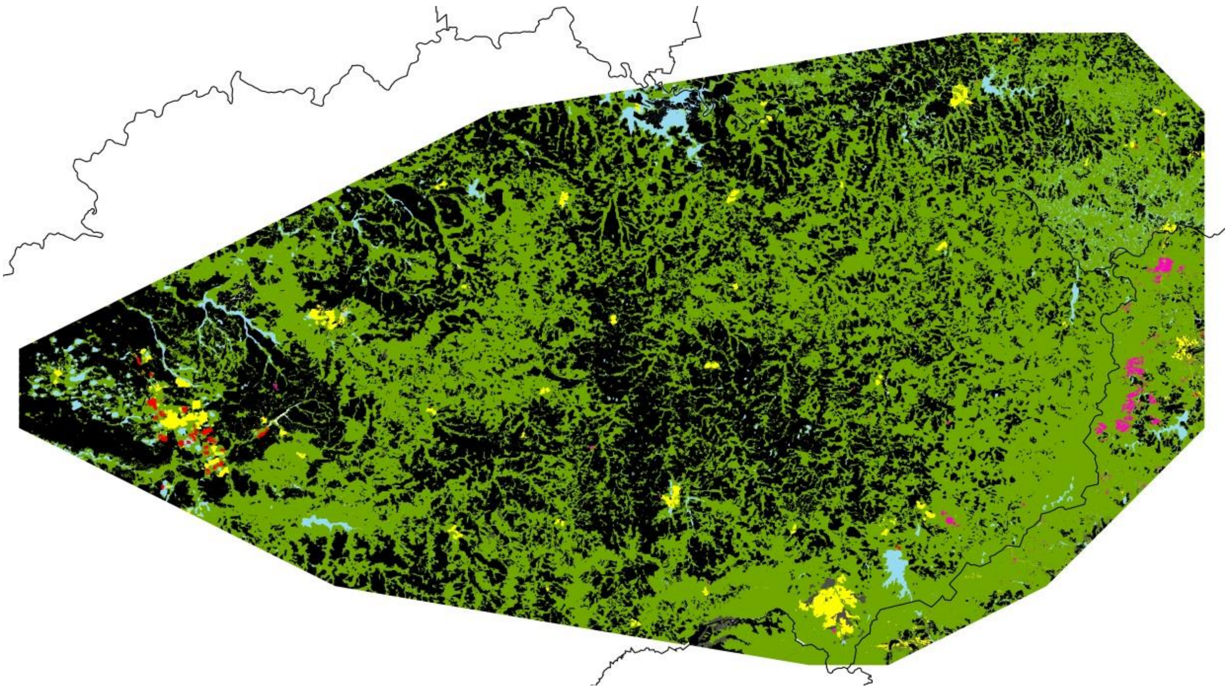
Absences

Sungazers were recorded as absent at all of 45 sites within the 19 QDGCs enclosed by the minimal convex hull where the species has not been previously recorded.

3.3.3. Land cover

Over half (56.27%) of the land cover in the EOO of *Smaug giganteus* is natural, while the remainder has been irreversibly transformed as a result of crop cultivation, urbanisation, plantations and mines (Fig. 9a, 10). This pattern remains the same for the distribution of the species within Free State and Mpumalanga provinces, with 53.75% natural, and the remainder irreversibly transformed (Fig. 3.9b, 3.10). These percentages translate to 32 827 km^2 natural land cover within the EOO and 20 531 km^2 within the distribution.

a)



b)

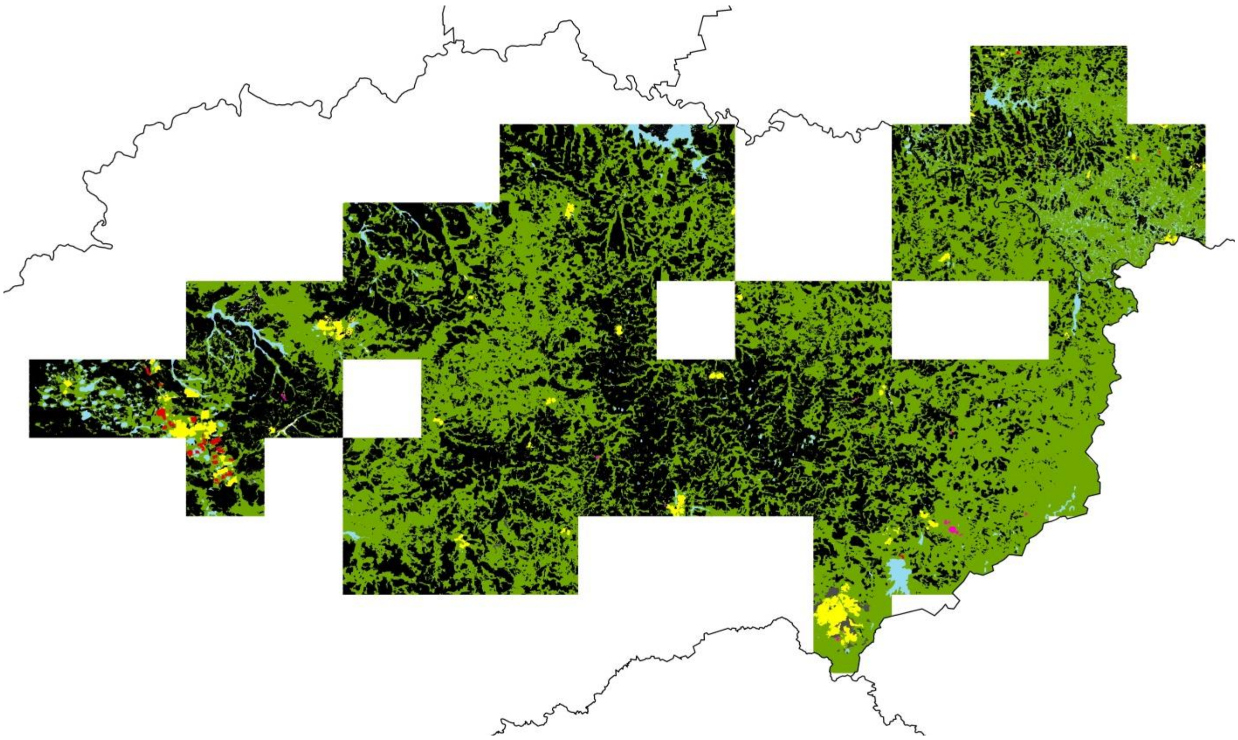
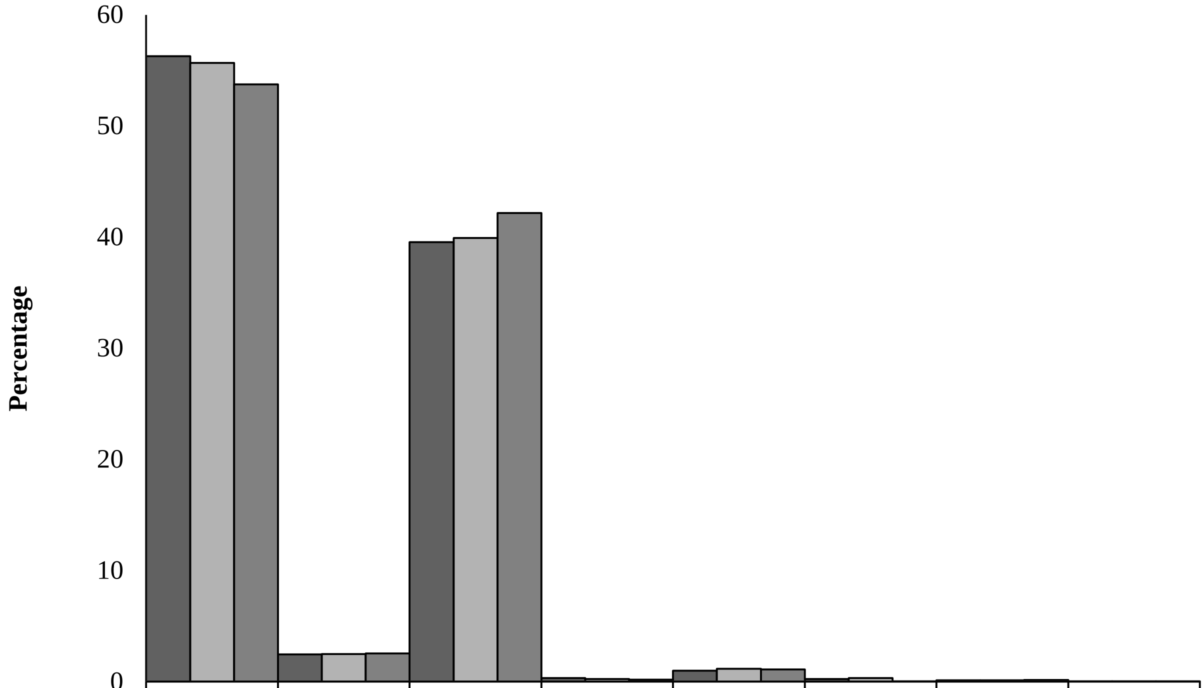


Figure 3.9. Land cover types within *Smaug giganteus* a) EOO (minimum convex hull) and b) distribution (Green = Natural untransformed area, Aqua = Water bodies, Black = Cultivated land, Grey = Degraded land, Yellow = Urban Built-up land, Pink = Plantations, Red = Mines).



	Natural	Waterbodies	Cultivation	Degraded	Urban Built-up	Plantations	Mines	Undefined
EOO	56.27	2.46	39.54	0.34	1.00	0.25	0.12	0.03
QDGCs	55.68	2.48	39.91	0.24	1.18	0.34	0.14	0.04
Distribution	53.75	2.55	42.17	0.18	1.11	0.06	0.15	0.04

Land cover type

Figure 3.10. Percentage contribution of land cover types to area of *Smaug giganteus* EOO, QDGCs in which the species has been recorded, and distribution as defined in this study.

3.3.4. Area of Occupancy

I found Sungazers at 5 out of 120 random sites across the EOO, 99 of which fell within the distribution of the species (including the 6 new records found in this study) (Fig. 3.11). This represents a 4.17% chance of finding occupied habitat at a randomly chosen spot within the EOO of the species, and 5.05% chance of finding occupied habitat at a randomly chosen spot within the distribution of the species.

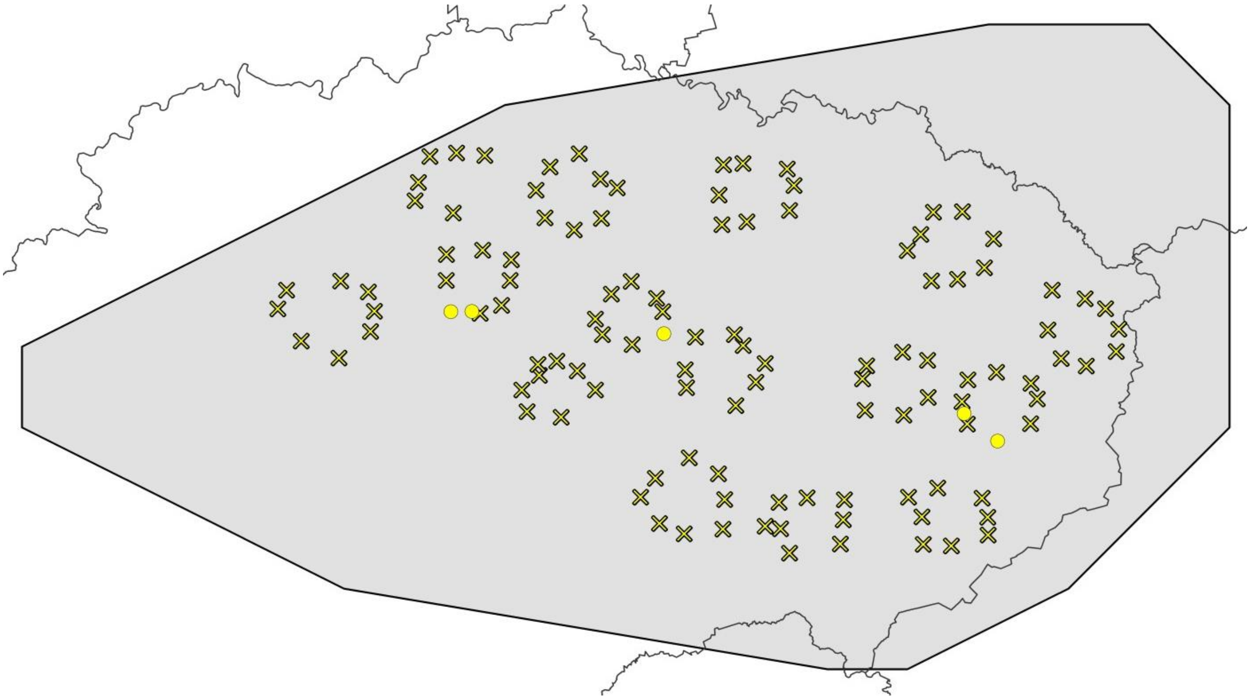


Figure 3.11. *Smaug giganteus* EOO (minimum convex hull) showing presence (O) and absence (X) at random sites visited for AOO estimation.

The definition of AOO recognises that the EOO comprises area that is unsuitable and therefore uninhabitable by the species, and the AOO estimation must therefore be applied to the natural portion of the distribution in order to accurately understand the area occupied by the species. I calculated AOO as 5.05% of the natural land area within the distribution where the species has been recorded as occurring. The total area of natural land within the distribution as calculated above is 20 531 km^2. AOO therefore equals 1 037 km^2.

3.4. Discussion

In this Chapter, I defined and calculated the extent of three important ecological areas for *Smaug giganteus*. First, the EOO was defined by the creation of a maximal convex hull that extends along the outer edge of all QDGCs that contain Sungazer occurrence records. The EOO was measured as 58 338 km^2, 57 877 km^2 of which falls within South Africa, and the remaining 461 km^2 in Lesotho. Second, the spatial distribution of the species was defined. The EOO is a spatial representation of the geographic spread of a species, but does not indicate how the species is distributed across this area. The distribution of the species was defined as the sum of the areas of the QDGCs and the portions of the QDGCs in which Sungazers have been reported occurring, excluding the portions that fall within Lesotho and the KwaZulu-Natal Province of South Africa. The total area of the 60 QDGCs in which the species is found is 40 969 km^2, and 38 196 km^2 of this is considered to be the distribution of the species. While both the EOO and distribution of the species could be refined using point locality data for the species that represent that the outer limits of occurrence, the majority of historical records for the species were recorded in the QDGC in which the record was made, and the exact locality within the ~650 km^2 unit is not defined. While this study did investigate aspects of the species' ecology across the distribution, it was not an aim to define the outer limits of Sungazer occurrence. There is therefore a potential for the actual EOO and distribution to be ~10% smaller than calculated here. Although some records are available as point localities, or a finer resolution (16 degree cells – De Waal, 1978), the majority of records have been record at the QDGC scale. This did not allow EOO to be calculated at a finer scale.

Just over half (53.75%) of the distribution as defined in the Chapter remains in a natural state, while the balance is irreversibly transformed for crop cultivation and plantations, and the construction of urban areas and mines. Sungazers occupy only 5.05% of natural area, which translates to an AOO of 1 037 km^2. The EOO viewed on its own is significantly/3x larger than the maximum area necessary to classify the species as Threatened using IUCN criteria. The AOO however, is small enough to classify the species as 'Vulnerable' using the same criteria. The AOO of the Sungazer is just over twice the area necessary to classify the species as 'Endangered', and a further 50% reduction in natural habitat will lead to the species being placed into this category.

Further to the absolute value of the EOO and AOO revealing the extinction risk of the species, evaluating these measures in context of each other is key in assessing how at risk the species is from stochastic processes (Gaston and Fuller, 2009). The AOO is only 1.78% of the EOO of the species. This means that for every unit of area that is occupied by the species, 56 are unoccupied. The species is therefore relatively safe from natural threat processes such as fire or disease, as the chance that a single threat process causes extirpations at all localities is highly unlikely. AOO is typically a much smaller area than the EOO of a species, often being just a small fraction (Hurlbert and Jetz, 2007; Boitani *et al.,* 2008). Species that have restricted distributions and are habitat specialists often have their ranges overestimated, leading to a false optimism as to the distribution of the species (Jetz *et al.,* 2008).

I found Sungazers in 6 QDGCs where the species has not previously been recorded, representing an 11% increase in the area on the QDGC resolution in which the species has been recorded. However, these new records are unlikely to be due to a recent range expansion, but rather due to better sampling. A total of 45 sites across 19 QDGCs where *Smaug giganteus* has not been previously recorded were surveyed in an effort to improve coverage of the species across the EOO. The fact that no Sungazers were found in these sites is not necessarily indicative that the species does not occur within that QDGC, as a QDGC is approximately 650 – 680 km^2 and a complete survey of a QDGC is impractical. Mapping the distribution of the species at a finer resolution than QDGC will alter the measure of total area of the distribution of the species, as some areas within QDGCs that record the species as present will be removed, and some areas are likely to be found containing Sungazers in QDGCs where they currently marked as absent (Gaston, 1991; Gaston, 1994; Cowley *et al.,* 1999; Goehring *et al.,* 2007).

Van Wyk (1992) estimated the EOO as 45 623 km^2 for *S. giganteus* (Van Wyk, 1992) based on the number of magisterial districts in which the Sungazer had been recorded. More recently, Mouton (2014) calculated the EOO as 47 450 km^2, based on the area of QDGCs contained within a polygon drawn around distribution records. This method only considered the complete QDGCs contained within this polygon, and did not fulfil the requirements of the IUCN definition of EOO. I calculated an area of 58 338 km^2. This calculation is approximately 23% larger in area than previous estimations, but follows the official IUCN definition and is made with a high level of confidence, over the medium confidence estimation made previously. Similarly, an AOO of 3510 km^2 was calculated based on the number of QDGCs where the species has been recorded (50), and the percentage of this area (10%) estimated to be suitable based on an examination of Google Earth imagery (Mike Bates, pers. comms. in Mouton (2014)). Branch (1988) suggested that 50% of the arable grassland within the Sungazers range is already irreversibly transformed for crop monoculture. Although this estimation was made two decades ago, the current percentage of transformed land across the distribution (46.25%) is similar. The knowledge that only 53.75% of the distribution of the Sungazer remains natural, creates an awareness of the dire situation that the species and its habitat face.

My study was the first to quantify an EOO for *S. giganteus* strictly following IUCN protocols, as well as measure of spatial distribution based on natural habitat within the QDGCs where the species has been recorded, and finally a measure of how much area is occupied by the species. The empirical quantification of the AOO of *S. giganteus* allows for the species to be assessed using the IUCN's criteria on geographic range for the first time. A full assessment of the conservation status of *S. giganteus* is presented in Chapter 6, taking these new measures of geographic range into account.

CHAPTER 4:

SUNGAZER POPULATION SIZE: PRESENT, HISTORICAL, AND RATES OF DECLINE

4.1. Introduction

Anthropogenically-mediated land cover change is the most significant threat to global biodiversity (Wilson, 1988; Lawton and May, 1995; Sala *et al.*, 2010), with just over half of the Earth's land area directly modified through human action (UNEP, 2002; Clay, 2004; Hooke, 2012). Agriculture is the primary effector of land cover change (Ramankutty and Foley, 1999; Asner *et al.*, 2004), contributing to approximately 72% of global land transformation (Hooke, 2012). Crop land area has increased by 70% over the past 40 years (Rosegrant *et al.*, 2002; Gleick, 2003), and is set to increase further with burgeoning demands from a growing human population (Sala *et al.*, 2010). Land cover change can affect populations directly through habitat loss i.e. reducing the area that a species can occupy, or indirectly, by fragmenting the landscape into patches separated spatially by unfamiliar and/or hostile environments (Weins, 1989; Gilpin and Hanski, 1991). Together, these processes have led to population declines and extinctions of a wide array of species globally (Burbridge and McKenzie, 1989; Groombridge, 1992; Burkey, 1995; Ehrlich, 1995; Fahrig, 1997; Alford *et al.*, 2001; Driscoll, 2004; Winne et L., 2007; Collen *et al.*, 2009). In the light of these extensive declines, quantifying population sizes and rates of decline is critical in identifying species that need urgent conservation attention.

Total population size and fluctuations are important determinants of the survival of a species (Reed *et al.*, 2003 and references therein), and as such, are commonly used to assess the conservation status of species (e.g., Rabinowitz *et al.*, 1986; Martins *et al.*, 2008). Population declines can be detected directly in species where populations have been monitored over long-term periods (Ceballos and Ehrlich, 2002). However, many species lack such historical monitoring programmes, with the need for conservation only becoming apparent when much of the decline has already occurred (Balmford *et al.*, 2003). Population declines are intrinsically linked to habitat loss and fragmentation, and in cases where the relationship between area and population density are known, population size and declines can be inferred from habitat loss (Ceballos and Ehrlich, 2002; Balmford *et al.*, 2003). This indirect method of estimating declines is more commonly used, with modern geographical information system (GIS) techniques allowing for accurate measures to be made of land cover change over pre-defined spatial and temporal scales (Driver *et al.*, 2011).

The distribution of *S. giganteus* falls across the Highveld Agricultural Region, 81.5% of which is underlain by highly arable soil, and as a result, just under half of the area (46.25%; Chapter 3) has been irreversibly transformed for crop monoculture. Agricultural practises are therefore a major and direct threat to the species, destroying large tracts of optimal habitat and creating a network of isolated patches across the distribution (Marais, 1984; Van Wyk, 1988; Jacobsen *et al.*, 1989). Despite the Sungazer being one of South Africa's most iconic reptile species, the current population size and decline have not been empirically quantified. The most recent assessment of the species' conservation status is based on perceived population reduction based on trends in habitat loss and fragmentation within the grassland biome (Mouton, 2014). This is problematic, as without a meaningful measure of the current population size contextualised by changes over time, conservation decisions made for the species may be misguided.

Prior to this study, an accurate quantification the population size of *Smaug giganteus* was not possible due to a lack of data on population densities and the relationship between natural land area and Sungazer presence. In Chapter 3, the AOO of the species was calculated, and used in conjunction with the measures of burrow density and occupancy quantified in Chapter 2, it was possible to develop a model to calculate the population size of Sungazers likely to be found within a given area of natural land. The historical population size of the species prior to anthropogenic change can also be calculated retrospectively, using the distribution as defined in Chapter 3, adjusting the natural area to the scale of the entire distribution. Changes in the landscape over time and the associated declines can be monitored using recent national land cover maps (Driver *et al.*, 2011). The most recent land cover map for South Africa was published in 2009 (NLC 2009), to replace the outdated 2000 land cover map (NLC 2000) (SANBI, 2009). The 2009 map is estimated to be 90% accurate, and is regarded as a good indicator of change in land use over the previous 10 years, providing up-to-date digitization of cultivated areas (ARC) and buildings (ESKOM) (SANBI, 2009).

In this Chapter, I use a series of population models and geographic information systems (GIS) techniques to calculate the current and historical population size of *Smaug giganteus*, as well as how populations have changed over the past decade, by analysing trends in land cover change.

Aims

1. Calculate current population size
2. Assess accuracy of population size calculation
3. Calculate historical population size
4. Assess change in known populations over time
5. Calculate habitat loss and associated population decline
6. Assess accuracy of previous survey

4.2. Methods

4.2.1. Current population size

Total burrow count

I calculated the total number of *Smaug giganteus* burrows across its distribution by multiplying the mean burrow density (6.14 ± 0.87 (95% CI) burrows/ha; derived in Chapter 2) by the area occupied by the species (AOO) (103 682 ha; derived in Chapter 3):

Total number of burrows = MBD ± 95% CI* x AOO

*The 95% confidence interval around the mean burrow density is presented throughout the calculations as a representation of the variance around the means.

Unoccupied burrows

Previous studies that investigated population demography and burrow inhabitancy of *S. giganteus* found that 11.49-18.5% of burrows within a colony are unoccupied (Stolz and Blom, 1981; Jacobsen, 1989; Van Wyk, 1992). Sungazers move between burrows for mating purposes (Van Wyk, 1992; Ruddock, 2000), and an unknown percentage leave the colony to disperse each season. Along with mortality, this leaves a proportion of burrows empty at any time point. Understanding the typical number of burrows empty within a colony allows for a meaningful relationship to be gleaned between the total number of burrows and the number of Sungazers within a colony. The number of occupied burrows can be calculated by subtracting the mean percentage of empty burrows (14.28%) (PEB) quantified from previous studies, from the total number of burrows calculated above:

Total number of occupied burrows = (MBD ± 95% CI x AOO) – PEB

Total population size

Similarly, burrow occupancy has been quantified in previous studies (Jacobsen *et al.*, 1990; Van Wyk, 1992) (Fig. 2.7.), and the weighted burrow occupancy index (BOI) (calculated in Chapter 2; 1.83 Sungazers/burrow) can be applied to the total number of occupied burrows to calculate the total population size of Sungazers that occupy burrows within an area:

Total population size = Occupied burrows x BOI

Mature Individuals

Population size is interpreted in the IUCN Categories and Criteria Version 3.1. as the total number of members of a population capable of reproduction (IUCN, 2012). This standardised measure of population size across life forms allows for categorisations to be made based on the reproductive capacity of the population (IUCN, 2012). The number of mature individuals in the total population of Sungazers must therefore be quantified in order to contextualise the population size in terms of the IUCN Categories and Criteria Version 3.1.

Van Wyk (1992) found that the minimum SVL at which males and females are reproductively active is 165 mm and 170 mm respectively. In a study of population demographics over six seasons, Van Wyk (1992) recorded the percentage contributions of Sungazers considered to be reproductively mature (above SVL of 165 mm), which ranged from 53.6-74.2% over the study period, with an average of 62.4% (Table 4.1).

Table 4.1. Percentage contribution of mature adults to Sungazer population over six seasons (data derived from Van Wyk, 1992).

Season	**% contribution of mature adults to population**
October 1985	68.3
March 1986	61.3
October 1986	60.9
March 1987	53.6
October 1987	56.1
March 1988	74.2
Average	**62.4**

Jacobsen *et al.,* (1990) recorded the population demographics of 1539 lizards at a single time point, and presented categories of lizards classed according their developmental stages (Table 4.2.) Adult Sungazers made up a total of 54% of the population.

Table 4.2. Population demographics of a Sungazer population (data derived from Jacobsen *et al.*, 1990).

Demographic	**% contribution**
Adult Female	32.6
Adult Male	21.4
Sub-adult Female	6.2
Sub-adult Male	4.2
Juvenile	35.6

If the measure derived from Jacobsen *et al.,* (1990) is considered as a single measure of contribution of mature individuals to a population, and is pooled with the seasonal data from Van Wyk (1992), an average percentage of mature individuals (PMI) is 61.2%. The number of mature individuals in the total population can be calculated by applying the appropriate percentage to the total population of Sungazers:

Total mature individuals = (Occupied burrows x BOI) x PMI

The complete formula to calculate the number of mature individuals in a population is therefore:

Total mature individuals = ((MBD ± 95% CI x AOO) – PEB) x BOI) x PMI

4.2.2. Accuracy of population model

The model for calculating total burrows within an area forms the basis for estimating total population size, and the accuracy of this measure is critical in gaining a realistic idea of the total population size of the species. The mean burrow density (MBD) and proportion of area occupied within a site (AOO) were created from extensive sampling across the distribution and form a robust basis of which populations can be calculated. I only used measures collected during my study, and therefore only total number of burrows as calculated and ground-truthed in this study were assessed in this Chapter.

The Eskom Majuba Power Station Sungazer Reserve (EMPSSR) near Amersfoort in Mpumalanga (Fig.4.1), is a protected area created specifically for the conservation of Sungazers as an offset to the construction of a power station on the property. The reserve did not contribute to the calculation of burrow density and AOO, and serves here as an independent sample.

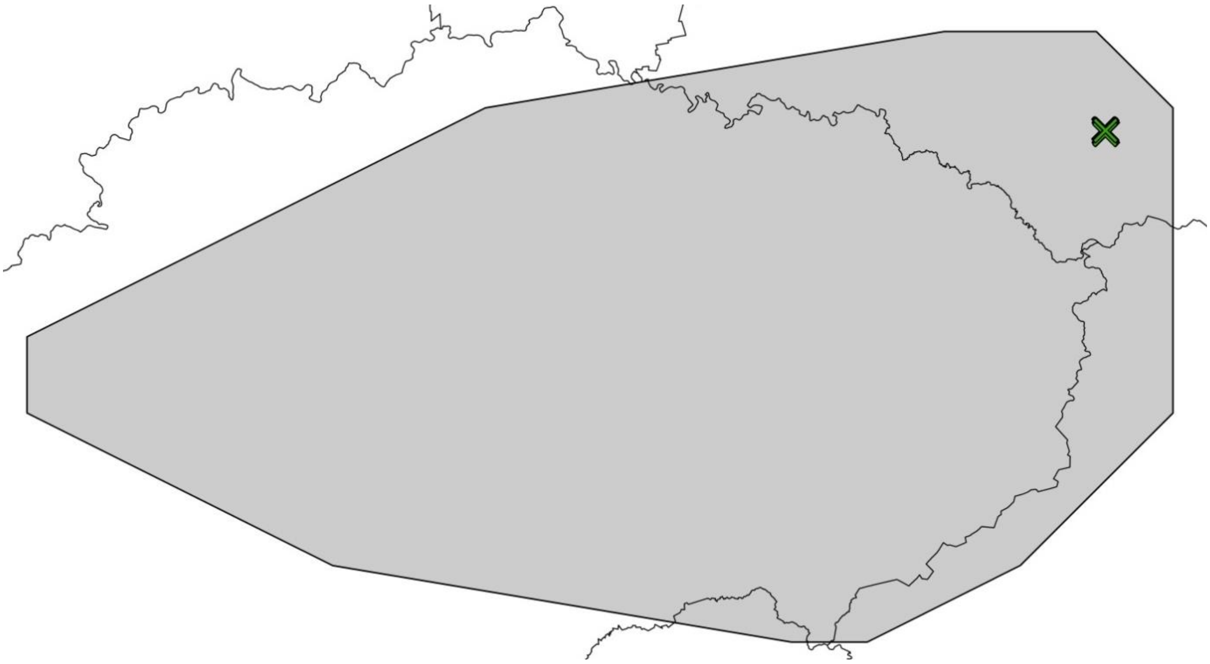


Figure 4.1. *Smaug giganteus* EOO (minimum convex hull) showing the location of the EMPSSR site (marked with an X) near Amersfoort, Mpumalanga Province.

Population modelling

I created a polygon around the borders of the EMPSSR in ArcMap and used the 'Calculate Area' function to calculate the area of the reserve. I clipped the 2009 land cover GIS layer (NLC 2009) to the EMPSSR polygon and assessed the percentage of natural land area within the reserve. Using the occupancy percentage of 5.05% calculated in Chapter 3, I calculated the area within the natural area of the reserve likely to be occupied by Sungazers. I then used the formula created above to calculate the population size based on the total natural area within the reserve. I used the formula: **Total number of burrows = MBD ± 95% CI x AOO**, to calculate the total number of burrows in the reserve.

Population survey

The Sungazer population at the EMPSSR was informally surveyed circa 2000, and steel poles were erected at the entrance of all Sungazer burrows. I assessed the status of the burrow at the site of each pole, and if active, recorded the GPS co-ordinates of the burrow. In addition, I extensively surveyed all areas within the reserve that represented suitable habitat (gentle slope/flat, untransformed *Themeda* dominated grassland). I was therefore confident that my survey was both comprehensive and accurate.

4.2.3. Historical population size

To contextualise the current population size of Sungazers, an estimate of the historical population size of the species is critical. However, base measures for original population size do not exist, nor are there any land cover maps that represent the unaltered landscape. The historical population size can therefore only be modelled based on the data used to calculate current population size. In order to estimate this population size, the following assumptions were made:

1) The extent of the natural distribution of the species has not changed considerably since the beginning of agricultural change in the region (following Ceballos and Ehrlich, 2002). The distribution to be used in the calculation is therefore the same as the current distribution. The natural area used in the calculation will be the entire extent of the distribution.
2) The proportion of suitable habitat occupied by the species within the distribution has not changed, and the percentage of 5.05% used to calculate AOO is applicable.

Accepting these assumptions, the historical population size of the species across the distribution can be calculated using the formulas developed above.

4.2.4. Population declines

I used two methods to assess declines in population size over time. The first method was to directly measure declines in known population. The second method inferred population changes from changes in the amount of natural area in the distribution, based on knowledge of population densities and occupancy and area. These methods are detailed below.

Method 1: Direct measures of populations

De Waal (1978) visited 76 sites within the EOO of *S. giganteus*, and recorded Sungazer populations at 39 sites (Fig. 4.2). I visited all the farms on which De Waal found Sungazers, to assess the current status of these populations. I adopted the same approach as De Waal (1978) during my survey: I interviewed the land owner and enquired about the presence of the species on the property. If the population was still present, I was directed to the population by the landowner and a 1 ha quadrat was surveyed as described in Chapter 2.

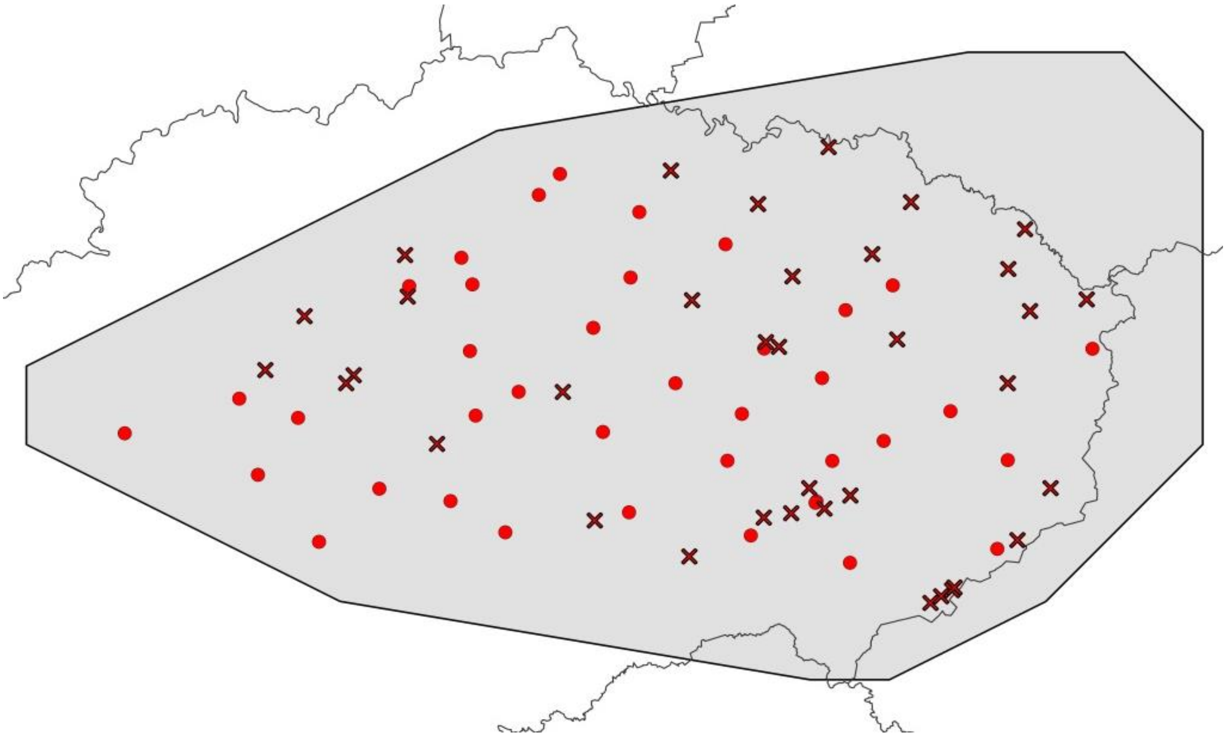


Figure 4.2. *Smaug giganteus* EOO (minimum convex hull) showing presence (O) and absence (X) of Sungazer populations at De Waal (1978) survey sites.

Habitat

The current population size of the species was calculated based on the natural land area in the most recent land cover map for the country (NLC 2009). Similarly, the population size of the species nearly a decade ago can be inferred from the previous land cover map (NLC 2000), which was based on data collected up until 2000. I inferred declines in populations using changes in natural land area within the distribution of the species.

I clipped the 2001 land cover map (NLC 2000) to the distribution of the species in ArcMap, and quantified the percentage contribution of each land cover class to the overall area. The 2000 land cover map has 49 land cover classes, compared to the 9 classes presented in the 2009 map. The areas of each of the 49 classes within the distribution was grouped into the 9 classes as described in SANBI (2009) in order to standardise the interpretation of the land cover maps.

The total population size existing within the natural area in the 2001 land cover map is calculated using the formula:

Total mature individuals = ((MBD ± 95% CI x AOO) – 14.28%) x BOI) x 0.612

The change in population size as related to land cover can then be interpreted by comparing population sizes calculated from the 2001 and 2009 land cover maps.

4.2.5. Assess accuracy of previous survey

Sungazers occupy only a small proportion of the suitable land within any given area, and it is possible to overlook the presence of the species on a farm if not guided by the experience of the land owner. This may lead to underestimations of occurrences during surveys. I revisited 37 sites surveyed by De Waal (1978) within the EOO where he did not record the species. This was done in order to assess the error rate (omissions) of De Waal's survey. If Sungazers were recorded as occurring, a 1 ha quadrat was surveyed as described in Chapter 2, and land owners were questioned as to whether the populations had established since the 1978 survey, or if they have been present since 1978. This was to ascertain whether these populations represented new populations that have arisen since De Waal's (1978) survey, or were missed by De Waal during his survey.

4.3. Results

4.3.1. Population size

A total of 636 325 ± 90 282 burrows are calculated as occurring across the distribution of *S. giganteus*. With the estimated number of unoccupied burrows removed from this number, the total number of burrows is expected to be 545 490 ± 77 395. These burrows are estimated as being occupied by 998 247 ± 141 632 Sungazers, with 610 927 ± 86 679 of these being mature individuals. This is the population size that is referred to hereafter.

4.3.2. Accuracy of calculation

Model

The total area of the EMPSSR is 373 ha. The reserve's land cover is primarily natural (92.11%), with 312.57 representing natural land area (Fig. 4.3). The remaining area consists of 40 ha water-bodies, 24.3 ha cultivated land and 5.1 ha mining area. The area likely to be occupied by Sungazers within the reserve is thus calculated to be 15.78 ha. This equates to total of 97 ± 14 burrows, with 83 ± 12 burrows occupied.

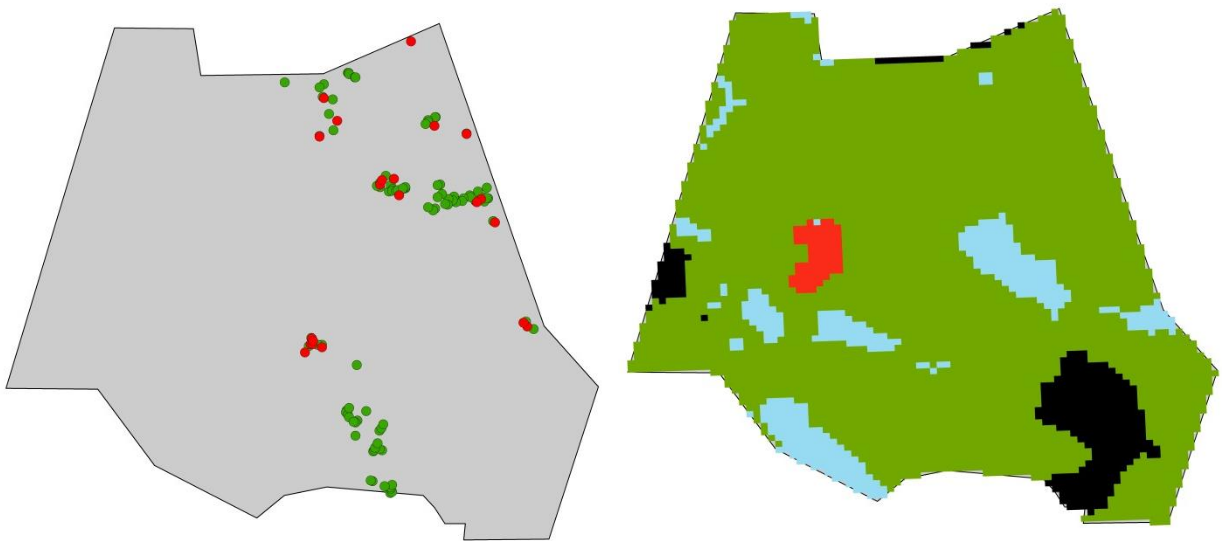


Figure 4.3. a) EMPSSR boundaries showing location of 98 burrows that were confirmed as active (green) and 21 burrows that were neglected/empty (red) in this study b) Land cover within EMPSSR (Green = Natural untransformed area, Aqua = Water bodies, Black = Cultivated land, Red = Mines).

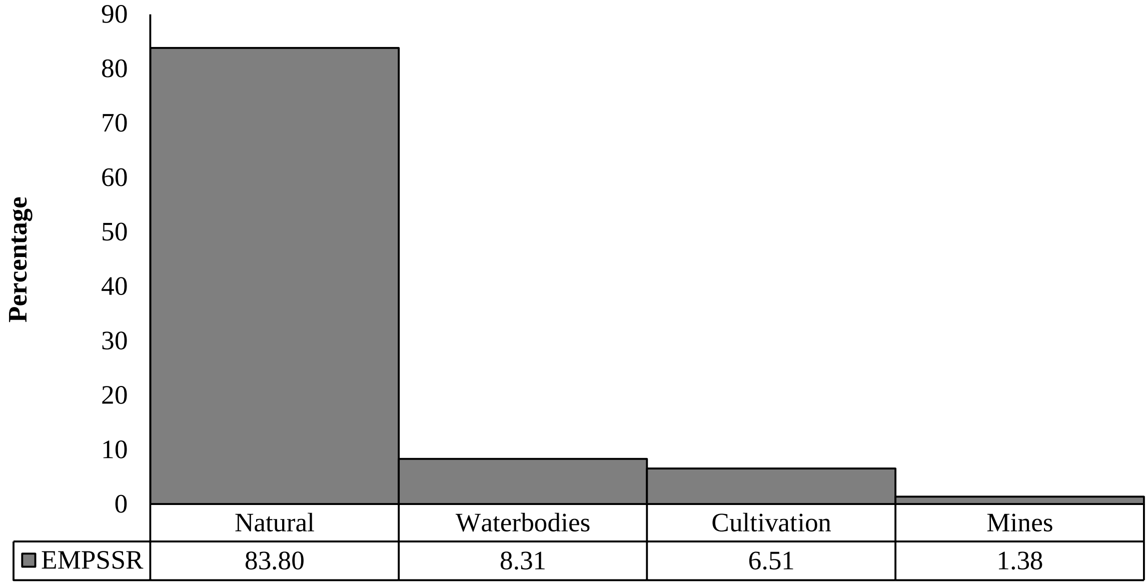


Figure 4.4. Percentage contribution of land cover types within EMPSSR.

Survey

The survey recorded a total of 119 Sungazer burrows. Burrow occupancy of 98 burrows was confirmed through sightings of Sungazers at the burrow entrance, fresh tail/claw marks or scat, or with the use of the intra-burrow camera. The status of the remaining 21 burrows could not be confirmed through any of these methods, and in most cases, debris blocked the burrow entrance. These burrows were assumed to be empty.

4.3.3. Historical population size

The total area of the distribution (occupied QDGCs within Free State and Mpumalanga provinces) is 38 196 km^2, and this entire area is assumed to be natural in the historical context. 1016 km^2 is made up of natural water bodies, leaving 39 953 km^2 as natural land area. This yields an AOO of 2018 km^2. The number of occupied burrows that can be expected to occur across this area is 1 183 861 ± 167 967, containing a total of 1 857 203 ± 263 502 Sungazers. This would yield a total of 1 002 889 ± 142 291 mature Sungazers as the historical population.

4.3.4. Population declines

Monitoring populations

De Waal (1978) found Sungazers at 39 out of 74 sites that he visited in his survey. I visited all 39 of the sites where he recorded the species (positive De Waal sites) and found that Sungazers were still present at 79.49% of these (Fig. 4.5). This represents a 20.51% decline in Sungazers at recorded sites over a 35 year period.

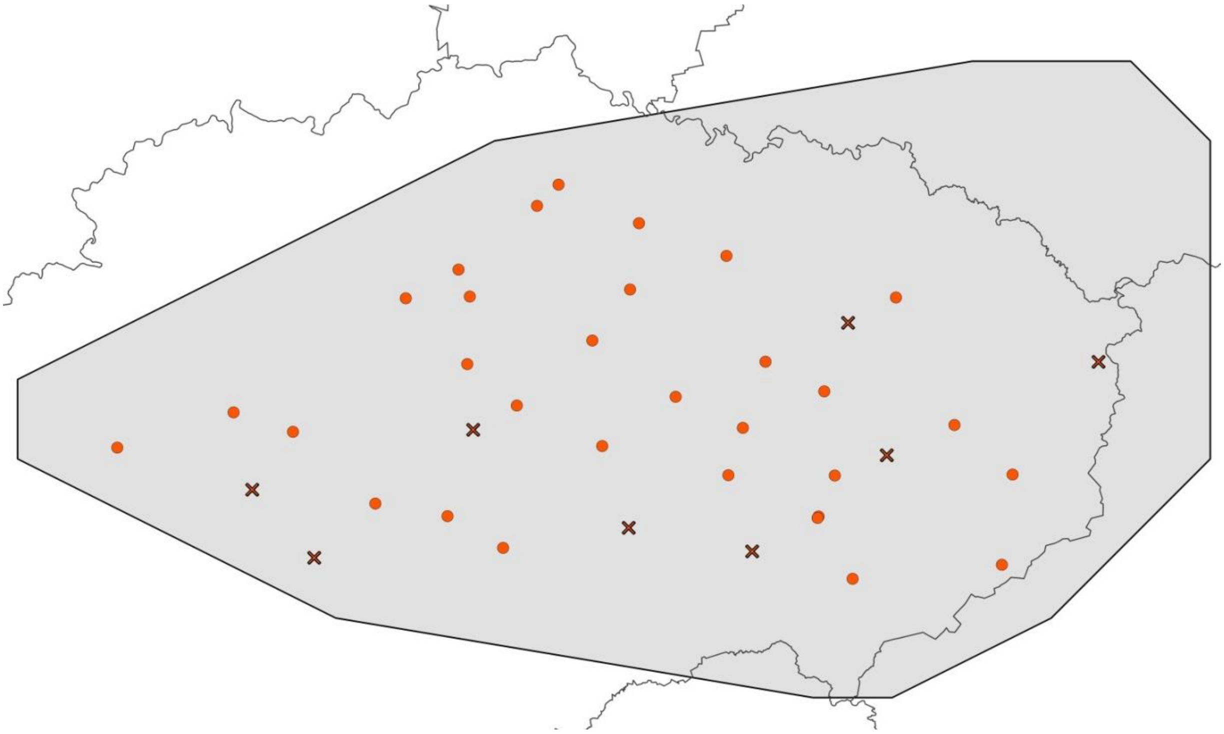
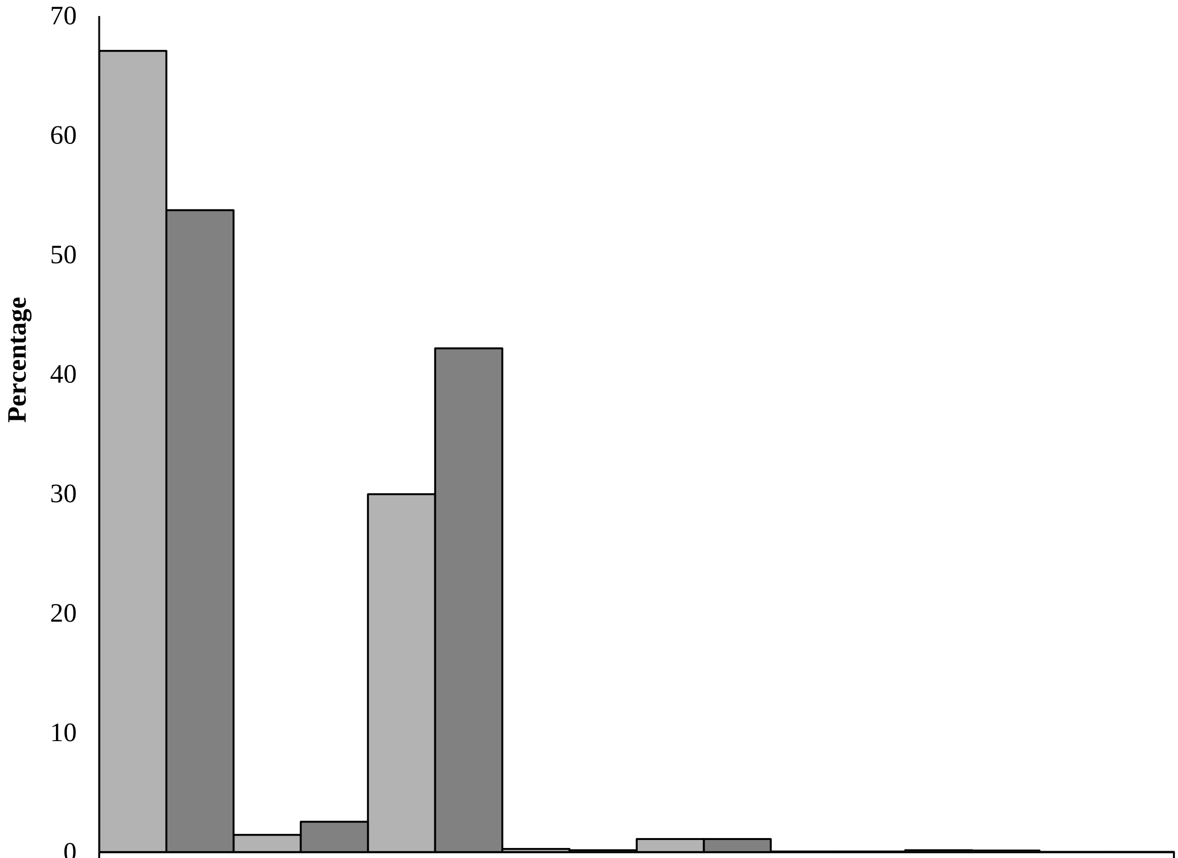


Figure 4.5. *Smaug giganteus* EOO (minimum convex hull) showing presence (O) and absence (X) of Sungazer populations at De Waal Positive sites.

Habitat

Natural land cover across the distribution of the species has declined by 13.33% over the 8 year time period between updated land cover maps (Fig. 4.6). This is congruent with a 12.21% increase in cultivated land, and a 1.1% increase in water-bodies.



	Natural	Waterbodies	Cultivation	Degraded	Urban Built-up	Plantations	Mines	Undefined
2001	67.08	1.45	29.96	0.28	1.12	0.06	0.18	0.00
2009	53.75	2.55	42.17	0.18	1.11	0.06	0.15	0.04

Land cover type

Figure. 4.6. Change in land cover in the distribution of *Smaug giganteus* between 2001 and 2009.

4.3.5. Accuracy of previous survey

I found Sungazers present at 8.11% (3/37) of sites where De Waal (1978) did not record any Sungazer populations (Fig. 7.). None of these sites were regarded as representing new populations, with land owners confirming the presence of the populations on the property for decades.

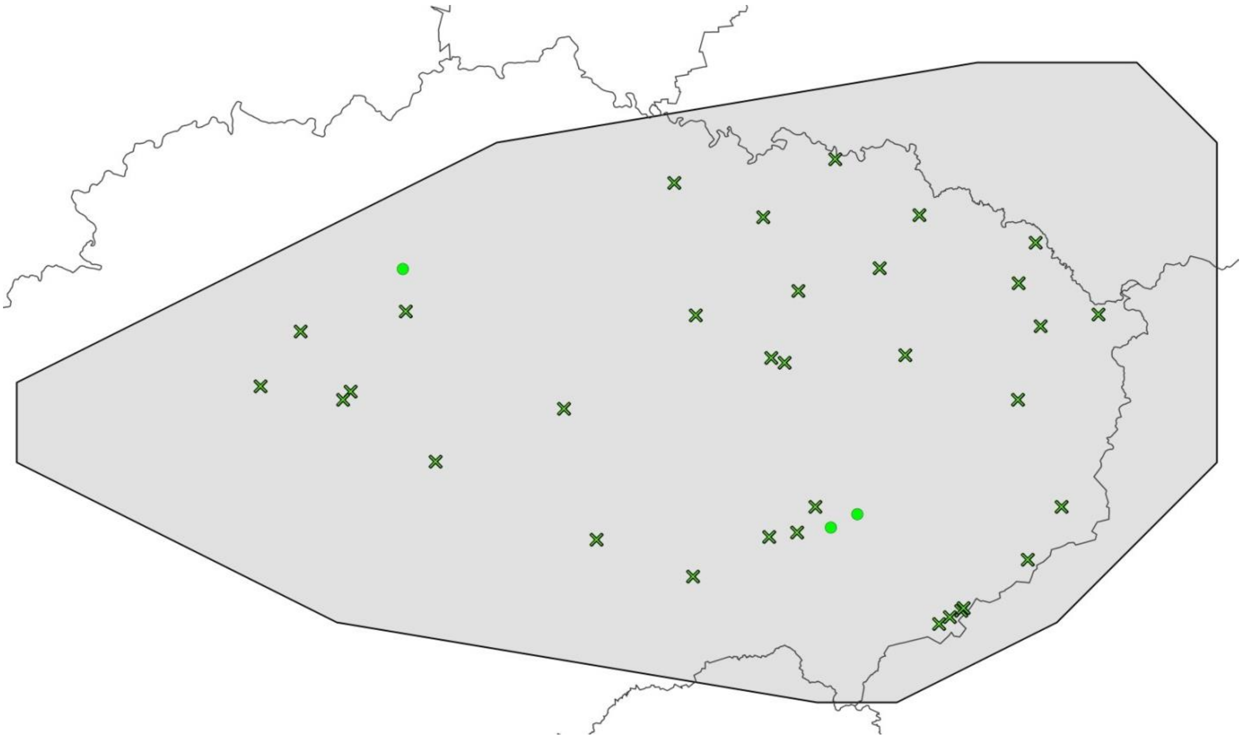


Figure 4.7. *Smaug giganteus* EOO (minimum convex hull) showing presence (O) and absence (X) of Sungazer populations at De Waal Negative Sites.

4.4. Discussion

Population size

The current population size of mature *S. giganteus* is estimated at 539 070 ± 76 484. This is a large population from a conservation viewpoint, being approximately 50 times larger than needed to be classified as Threatened by IUCN standards (IUCN, 2012). Just under half of the natural land area across the distribution of *S. giganteus* has been irreversibly transformed, with agricultural land cover contributing 91% towards this change (Fig 4.6.). This is similar to global trends of agriculture being the primary driver behind land cover change (Hooke, 2012). However, the distribution of the species falls within a region heavily utilised for crop production, and so the contribution of agriculture to overall land cover change is 19% higher than the global average. Based on the models formulated in this study, the historical population size prior to land cover change is estimated as 1 857 203 ± 263 502, with mature individuals making up 1 002 889 ± 142 291. This computation is based directly on the current relationship between amount of natural land and Sungazer occupancy. This does not take into account the potential confounding effects that fragmentation has had on the species. Landscape structure is typically hypothesised to play an important role in the regulation of populations (MacArthur and Wilson, 1967; Gilpin and Hanski, 1991; Saunders *et al.*, 1991), however model simulations (Kareiva and Wennergren, 1995; Fahrig, 1997) and empirical field studies (McGarigal and McComb; 1995) have found that habitat loss had a much larger effect on population extinction than habitat fragmentation. Until such time that the effects of fragmentation on Sungazer population processes are further understood, the assumptions made in this study represent the most realistic estimate of historical population size.

The 8.11% difference in population presence recorded between De Waal's (1978) survey and my study cannot be explained as an indication of the establishment of new populations, as all populations in my study were confirmed to have been in existence for at least 35 years by landowners. These sites are instead viewed as populations that were overlooked during De Waals survey. The survey that was conducted by De Waal aimed to assess the general herpetofaunal biodiversity, and even though he mentions asking landowners specifically about the presence of Sungazers, it is likely that in some cases he was not informed by the land owner (For example, the landowner may not have been present when the site visit was conducted). Owing to the large size of each farm and the discrete and patchy nature of Sungazer colony distribution, populations could be easily have been missed if the landowners were not asked about their presence. This error rate should be taken into account during future survey, with the possibility of being used as a correction factor.

The model for total burrow count predicted 97 ± 14 burrows being present in the EMPSSR, while 119 were counted in the survey. In this particular scenario, the upper limit of 111 of the modelled distribution slightly underestimated the actual burrow count (by 8 burrows). This is likely due to the 95% CI of 8.7% not accurately portraying the wide variation in burrow density across sites. The standard deviation of 3.97 burrows/ha around the mean yields a range of 34-160 burrows around the mean, and this use of this range might be more appropriate in future models, given the wide variation in burrow densities recorded. An ecological niche model is produced in Chapter 6, and predicts population densities across the distribution of the species. The model can potentially serve as a tool with which the 95% CI can be adjusted to best predict the density to be expected at a specific area.

My study provides the most robust and most realistic measure of current population size and population declines for *Smaug giganteus* to date. The inputs of the population model that were quantified during my study (burrow density, AOO) estimated the total burrow count of the EMPSSR relatively well. However surveys at further test sites should be conducted to further test the accuracy of the model. The input measures that were derived from metadata from other studies (PEB, BOI, PMI) were not verified during my study due to the time-consuming nature of capturing Sungazers. I recommend that further work be done to assess the accuracy of the model in its entirety. If certain inputs are found to be inaccurate in field studies, the variation in those indices can be quantified by sampling at sites across the distribution. This would potentially increase the value of the 95% CI around the mean population that is produced.

The calculation of historical population size relied on several assumptions. The assumption that the distribution of the species has not changed possibly underestimates historical population size if any populations that lay outside of the current EOO were extirpated through anthropogenic land cover change. Since the distribution is limited primarily by the vegetation and physical barriers, it is fairly likely that the historical EOO has not changed significantly. With species that decline without distribution shrinking, it is density that decreases with land cover change. It is therefore possible that the proportion of area occupied by the species within the natural area of the EOO was higher. However, no other data exist that allow for a more accurate estimation of historical population size. It is best to view the calculation with caution.

Jacobsen *et al.,* (1990) estimated that the population of Sungazers existing within Mpumalanga to be approximately 50 000. The total natural land area of the distribution within Mpumalanga is 451 348 ha, with an AOO of 22 793 ha. The total population size that can be expected to occur within this area is 219 459 ± 31 137, with mature individuals making up 118 508 ± 16 814. Assuming that Jacobsen *et al.,* (1990) were referring to total population size and not mature individuals, the number of Sungazers in Mpumalanga is expected to be roughly four times what the authors predicted, based on the model developed in my study. Jacobsen *et al.,* (1990)'s estimate of population size is the only one that exists for this species, prior to my study.

Population declines

A 20.51% decline of populations at known occurrence sites was recorded since 1978 (0.59% decline/year). None of these populations were lost directly to habitat loss through land cover change, as no new fields were ploughed at these sites since the De Waal survey. Most land owners took me to the exact locality where the population once existed, and in some cases pointed out the remnants of burrows. De Waal (1978) similarly noted that at several sites, land owners confirmed that the species was once found on the property, but had become extirpated. In several cases the land owners suspected that the population was likely to have been collected by poachers, and in at least one case, poaching was identified as the reason for the population's disappearance. It is possible that these population declines represent natural reductions, however it is more likely that the decline recorded is a result of habitat fragmentation. Animals that inhabit fragmented landscapes exhibit inhibited dispersal as a result from unwillingness and inability to traverse between fragments (Saunders *et al.*, 1991; Debinski and Holt 2000; Couvet, 2002). At present, the dispersal and colonisation strategies of the species are not understood, and this prevents the decline to be contextualised. A study of recruitment rates/mortality/dispersal and colonisation is essential in trying to elucidate the answers to these questions.

A comparison of the land cover types between 2000 and 2009 reveals a 13.3% decline in natural habitat across the distribution of *S. giganteus* (1.48% decline/year). This is explained by a 12.21% increase in cultivated land, and a 1.1% increase in water bodies. In the absence of further data that suggests a different relationship, the simplest and most intuitive relationship is assumed here: changes in area and populations correspond in a one to one fashion in ecological time (Hughes *et al.,* 1997). In this case, the 13.3% decline in natural land area should correspond directly with a 13.3% decline in population size. Linear declines are often assumed when projecting population declines into the future (e.g. Araújo and Williams, 2000; IUCN, 2012). Measures of decline are often quantified with only two points in time, and a more complex understanding cannot be gained without more frequent data points. Assuming that the rate of decline of 1.48% decline/year is linear, the Sungazer would be expected to go extinct in 68 years i.e. in 2082.

The systems of dispersal and colonisation are not understood for *S. giganteus*. This hinders a better understanding of the driving force of the 21% decline of known populations, within the natural framework of population fluctuations and extirpations. If populations are considered as distinct Mendelian entities, then the genes of the population might persevere at other sites. Population declines are difficult to detect, and studies of long-term population dynamics are integral in differentiating natural population fluctuations from true declines (Tinkle, 1979). Many herpetofaunal species exhibit dramatic fluctuations and extinctions at sub-population scale that are linked to natural causes (Gibbons, 1990; Pechmann *et al.,* 1991; Shine, 1991), and it is of great importance to understand the difference between natural variations and anthropogenic causation (Gibbons *et al.,* 2000). This decline should then be interpreted with prudence until such time that these systems are better understood, such that the decline is not taken as absolute.

The scale at which declines are measured is a determining factor in the detection of extirpation. Quarter Degree Grid Cells have become a ubiquitous resolution at which species distributions are measured, especially in course resolution assessments that focus on functional groups (Bates *et al.,* 2014). However QDGCs are large in area (~650km^2) and extinctions at population level are not necessarily detected at this resolution. Ideally, a fine resolution monitoring system is required to keep track of the species on a sub-population level. The finest scale and most effective method of population monitoring would be a farm-scale system, where the presence of the species on each farm within the distribution of the species is assessed, and land owners are given the responsibility to report to a national manager for the species. This could potentially be achieved through a stewardship programme, which is recommended as a primary conservation measure in the concluding chapter of this thesis.

The Living Planet Index (LPI), which synthesises trends across 3 000 populations of over 1 100 species of vertebrate species found that on average, terrestrial vertebrate species declined by 25% between 1970 and 2000 (0.83% decline/year) (Loh *et al.,* 2005). In context, the rate of 0.59% decline/year measured at known Sungazer sites is slightly lower than the average rate of decline found for most vertebrate species. Reading *et al.,* (2010) found that 65% of monitored snake populations representing eight species across five countries have experienced sharp declines over 15 years. While Reading *et al.*, (2010) note that their study only covers a few species, the average decline of 4.3% of populations per year is a noteworthy trend that should alert conservationists to the increasing declines. Some reptile species have experienced population declines that are almost 13 times (*Japalura polygonata polygonata*; JAE, 2000) and 19 times (*Caretta caretta*; Sato *et al.*, 1997) greater than that experienced by *S. giganteus*.

The most recent IUCN Red List assessment for *S. giganteus* was the first attempt prior to my study to estimate population declines from habitat decline. Mouton (2014) estimated a population decline of 30% over the past 27 years (1.1% decline per year), inferred from the amount of habitat destruction across the Grassland Biome (Rouget *et al.,* 2006). The rate of population decline inferred from habitat decline quantified in this study (1.48% decline per year) equates to 39.9 % decline over the same time period. This is quite close to the rate of decline estimated by Mouton. Even though his estimations were based on trends of decline in natural land cover across the Grassland Biome as a whole, these trends are in play across the Sungazer distribution, albeit at a slightly higher rate (0.38% decline/year) than the biome. Assuming that the trend in habitat decline calculated in this study is linear, it can be expected that Sungazer habitat will be completely transformed by 2062. This will lead to the extinction of the species unless populations are conserved within protected areas in substantial numbers. This rate of decline is similar to the rates of decline measured for North West (NWDED, 2011), KwaZulu-Natal (Jewitt, 2011) and Gauteng Provinces (GDARD, 2011), all of which are also expected to experience an extinction of natural habitat around 2050 (Driver *et al.*, 2011).

The population models/formulae developed in this study are relatively accurate within the 95% CI of the mean that is predicted, and can be applied to reserves and other properties to calculate the potential population size of Sungazers on the property. The measured and estimated declines based on populations and habitat is the first quantified declines of the species, and directly portrays the plight of the species. Long-term monitoring of undisturbed populations in areas that do not suffer greatly from habitat fragmentation will likely reveal the nature of natural population fluctuations and declines. Further, an understanding of how often new colonies are formed will allow a more educated interpretation of the measured declines of known populations. Using genetic techniques, the age of a colony and the source of its founder genes could be elucidated. Finally, these measures can be used in IUCN conservation assessments for the species, and provide a baseline measure of which future population sizes can be compared. The repercussions of the measured habitat and population declines on the conservation status of the species are discussed in detail in Chapter 5. The prolonged monitoring of both populations and habitats at fine scales through remote sensing and on-the-ground is imperative in keeping track of this species through time.

CHAPTER 5:
IDENTIFYING SUNGAZER PRIORITY CONSERVATION AREAS USING ECOLOGICAL NICHE MODELS

5.1. Introduction

Protected areas are becoming increasingly important in arresting biodiversity loss (Shaffer, 1981; Pimm and Lawton 1998; Margules and Pressey 2000; Embling *et al.,* 2010). However the current global system of protected areas designed to conserve biodiversity is incomplete and insufficient (Soule and Sanjayan 1998; Rodrigues *et al.,* 2004), particularly for the protection of reptile species (Pawar *et al.*, 2007; Corbalan *et al.*, 2011; Vasconcelos *et al.*, 2012; Siler *et al.*, 2014). The establishment of protected areas that optimally conserve biodiversity in a region is challenging due to financial constraints, conflict with human interest over the land, improper design regarding the biodiversity that they contain, and a lack of relevant information (Martínez-Harms and Gajardo, 2008; Pressey and Tully, 1994; Pressey *et al.,* 1993; Ferrier *et al.,* 2000; Myers *et al.,* 2000; Mittermeier *et al.,* 2004). As a result, protected areas are often created in areas that are convenient and accessible for recreational purposes, or unsuitable for other purposes, such as agriculture or urban development. Many protected areas therefore, are not created to meet specific conservation objectives (Zhang *et al.,* 2012), and this places many species that occur in productive landscapes at risk (Margules and Pressey, 2000). In South Africa, where financial costs and habitat conflict are a hindrance to the creation of appropriate protected areas, rationalizing financial cost and utilising minimal area to achieve conservation targets are of key importance. One method of achieving conservation goals whilst minimising land used is prioritising conservation in areas where the target species occur at high densities. Areas with high population densities also experience a lower probability of extinction over a short temporal scale (Araújo and Williams, 2000; Donal and Greenwood, 2001; Araújo *et al.*, 2004). Occurrence data, geographic information systems (GIS) and ecological niche models (ENMs) can be employed to identify priority areas (PAs) for species conservation, based measures of probability of species presence and/or density (Araújo and Williams, 2000; Donal and Greenwood, 2001; Araújo *et al.*, 2004).

ENMs examine the associations between environmental characteristics and the known occurrences of a species (Guisan and Zimmermann, 2000; Scott *et al.*, 2002; Guisan and Thuiller, 2005), and allow the projection of a species' ecological niche into geographical space. This has the potential to identify where a species might occur within unexplored areas, map the distribution of a rare or threatened species, and project the distribution of a species into future or past climactic conditions. In the field of conservation biology, species distribution modelling has been coupled with reserve-selection analyses to identify priority sites for the expansion of protected area networks for mammals (Gibson *et al.,* 2004; Rondinini *et al.*, 2005; Greaves *et al.,* 2006; Moilanen and Wintle, 2007), birds (Loyn *et al.,* 2001; Suárez-Seoane *et al.,* 2002; Grand *et al.,* 2004; Moilanen and Wintle, 2007; Jensen *et al.,* 2008), amphibians (Rondinini *et al.,* 2005; Dayton and Fitzgerald, 2006; Goldberg and Waits, 2009), invertebrates (Smith *et al.,* 1996; Cabeza *et al.,* 2004; Grand *et al.,* 2004; Matern *et al.,* 2007; Steck *et al.,* 2007) and reptiles (Pawar *et al.,* 2007; Vasconcelos *et al.,* 2012, Siler *et al.,* 2014). While most studies of this nature focus on identifying areas that contain the highest biodiversity, or the highest proportion of targeted species, ENMs can be used successfully also in the planning of species-specific reserves. In cases where a single species is the target of a conservation programme, it is important to assess the area of target habitat and the population size needed so that the species is adequately conserved in the region of interest (Pressey *et al.,* 1993; Margules *et al.,* 1994; Pressey *et al.,* 1996; Csuti *et al.,* 1997).

Over the past several decades, studies have explored the concepts of minimal viable populations (Soule, 1980; Shaffer, 1981) and minimum habitat area (Fahrig, 2001), with the goal of estimating a minimum number of individuals needed to conserve a species within a PA system. The minimum habitat area necessary to conserve a species is highly variable (Fahrig, 2001), and is dependent on the range, reproductive rate, dispersal ability, emigration rate and other aspects of a species' life history and ecology (With and King, 1999). There has therefore not been a standardised index of habitat area that could be applied to a species' distribution in order to conserve a sufficient quantity of habitat. The concept of minimal viable population (MVP) presents an alternative to achieving a standard for a minimal index of conservation. Franklin (1980) suggested that a minimum effective population size should be no smaller than 50 individuals for short term population survival, keeping inbreeding to 1% per generation. Considering the maintenance of adequate genetic diversity for long term population survival, an effective minimum population size of 500 individuals is suggested (Franklin, 1980; Lande and Barrowclough, 1987). This 50/500 rule dictating minimal populations for short and long term survival has been used as a management goal for conserving species (Foose *et al.,* 1995). Since then, MVPs have been calculated for a plethora of vertebrate species based on well-understood life history traits, and meta-analyses of these studies suggest a mean MVP of ~5000 individuals, with a median range of 3 577 – 5 129 (Traill *et al.,* 2007; Traill *et al.,* 2010). This range of MVPs can be extrapolated to species where all relative life history characteristics are not fully understood, such that a reasonable conservation target is set. Despite the concept of a generalised MVP meeting criticism for being an overly simplistic 'magic number' (Flather *et al.,* 2011), the advantage of having a target population size for species where the relevant life history characteristics are not fully understood, is clear (Brook *et al.,* 2006; Brook *et al.,* 2011).

Smaug giganteus is endemic to South Africa, where its AOO is only 103 678 ha (Chapter 3). The species has experienced significant population and habitat declines over the past several decades (Chapter 4), and because the distribution falls within a highly transformed agricultural area, the species faces a high extinction risk. Sungazers have not bred successfully in captivity, nor been successfully translocated. Thus the conservation plans for the species rest entirely on *in situ* conservation and are dependent on a network of protected areas to ensure population sustainability over the long period. There are three formal reserves within the distribution where Sungazers have been recorded as occurring, however the populations were either introduced (Golden Gate Highlands National Park; Groenewald, 1992), no longer exist (Willem Pretorius Nature Reserve; Chapter 4) or are insubstantial for long term conservation purposes (Eskom Majuba Power Station Sungazer Reserve (EMPSSR); Chapter 4). The need for Sungazers to be protected within reserves has been highlighted since 1978, in the first South African Red Data Book: Reptilia (McLachlan, 1978). Peterson *et al.,* (1985) noted that a Sungazer reserve was urgently needed and that the Department of Environmental Affairs and Tourism (DEAT) was investigating suitable sites for protection, however nothing notable has been done in this regard since that time. Jacobsen (1989) raised the issue of the Sungazer not occurring within any formal conservation reserves in the Mpumalanga Province and noted that the population conserved within the EMPSSR is not sufficient to sustain a viable population. To create a protected area network for the long term conservation of *S. giganteus* while minimising financial costs and area used, conservation areas should be prioritised based on 1) habitat suitability and 2) areas with high population densities.

In this Chapter, I create an ecological niche model to identify areas that contain suitable Sungazer habitat and a high probability of containing high density populations. A variety of modelling techniques are commonly used to produce ENMs (Graham and Hijmans, 2006), although presence-only modelling techniques are commonly favoured as they simply require a set of known occurrences together with predictor variables such as topography, climate, soil and vegetation (Phillips and Dudik, 2008) and can be run with a small number of occurrence records (Elith *et al.*, 2006). Furthermore, presence-only models allow for the creation of ENMs for models for species where true absences are difficult to verify (Gu and Swihart, 2004). This is especially true when a species is rare, or if it is unclear whether a species is absent from a given locality because the site is outside of its climate envelope, or because of other factors such as biotic interaction, disturbance or dispersal limits. This can lead to misinterpretations, i.e. if the climate at a locality that is treated as an absence record is within the target species' climate envelope, the model algorithm misinterprets the climate at this site as unsuitable. For these reasons, the presence-only modelling programme MaxEnt (Version 3.3.3k; Phillips *et al.*, 2006) was chosen for the creating of ENMs for *S. giganteus*.

Aims

1) Create an ecological niche model for *S. giganteus*
2) Identify priority areas for conservation based on habitat suitability and probability of high density Sungazer populations
3) Assess the habitat suitability and likelihood of Sungazer presence in outlier QDGC records

5.2. Methods

5.2.2. Ecological niche model

Two primary types of data are needed to produce an ecological niche model for *S. giganteus* using the MaxEnt modelling programme: 1) *S. giganteus* occurrence records 2) environmental variable GIS layers for the area being modelled (*S. giganteus* EOO).

Occurrence records

GPS co-ordinates recorded at 511 Sungazer burrows across 80 sites were used in the creation of the model. Few of these records were from Mpumalanga Province due to the low number of sites covered in the study. To avoid a bias due to spatial bias in distribution records, I supplemented the database with 25 randomly-selected presence records from the Mpumalanga Province, provided by the Mpumalanga Parks Board. This resulted in a total of 536 presence records for input into the model. All records from the study were selected for use in the model, instead of a representative record from each site, in order for high population densities to be adequately represented in the model.

Environmental Layers

Smaug giganteus is a habitat specialist, and the presence of the species is known to be closely associated with distinct vegetation types and soil profiles, as well as falling within a well-defined altitudinal and climactic envelope (Van Wyk, 1988; Jacobsen, 1989; Van Wyk, 1992), and sensitive to the presence of rocks in sites where they dig burrows. A total of 24 environmental GIS layers representing all known aspects of the species' niche requirements were selected to model the fundamental niche of the species in geographic space. This included19 bioclimactic variables, altitude, vegetation type, soil type, underlying geology and land cover (Table 5.1).

Table 5.1. Climactic and environmental variables used in MaxEnt model

Environmental Variable Layer	**Code**	**Source**
Annual Mean Temperature	1	
Mean Diurnal Range	2	
Isothermality	3	
Temperature Seasonality	4	
Max Temperature of Warmest Month	5	
Min Temperature of Coldest Month	6	
Temperature Annual Range	7	
Mean Temperature of Wettest Quarter	8	
Mean Temperature of Driest Quarter	9	
Mean Temperature of Warmest Quarter	10	http://www.worldclim.org; Hijmans *et al.,* 2005.
Mean Temperature of Coldest Quarter	11	
Annual Precipitation	12	
Precipitation of Wettest Month	13	
Precipitation of Driest Month	14	
Precipitation of Seasonality (CoVar)	15	
Precipitation of Wettest Quarter	16	
Precipitation of Driest Quarter	17	
Precipitation of Warmest Quarter	18	
Precipitation of Coldest Quarter	19	
Altitude	alt_200	
Vegetation Type	vegmap	Mucina *et al.,* 2006.
Soil Type	soil	Dijkshoorn *et al.,* 2008.
Land cover Type	land cover	NLC, 2009.
Geology	geology	AGIS, 2009.

Running the model

Environmental variable layers were converted from their native formats into ASCII format in ArcMap 10.0. for input into the MaxEnt programme. A cell size of 0.016 (1 arc minute) was used. In order to increase the precision of the model, the layers were cropped to a rectangle that included that EOO and a surrounding buffer zone (Fig.5.1.). *Smaug giganteus* has historically had several records that fall outside of the currently accepted distribution (Mouton, 2014). Many of these records occur in areas that are obviously unsuitable and have been discarded, however some records exist within the grassland matrix, and the possibility of the species occurring in this areas could be tested with the model. The buffer zone was therefore defined to encompass seven of these records.

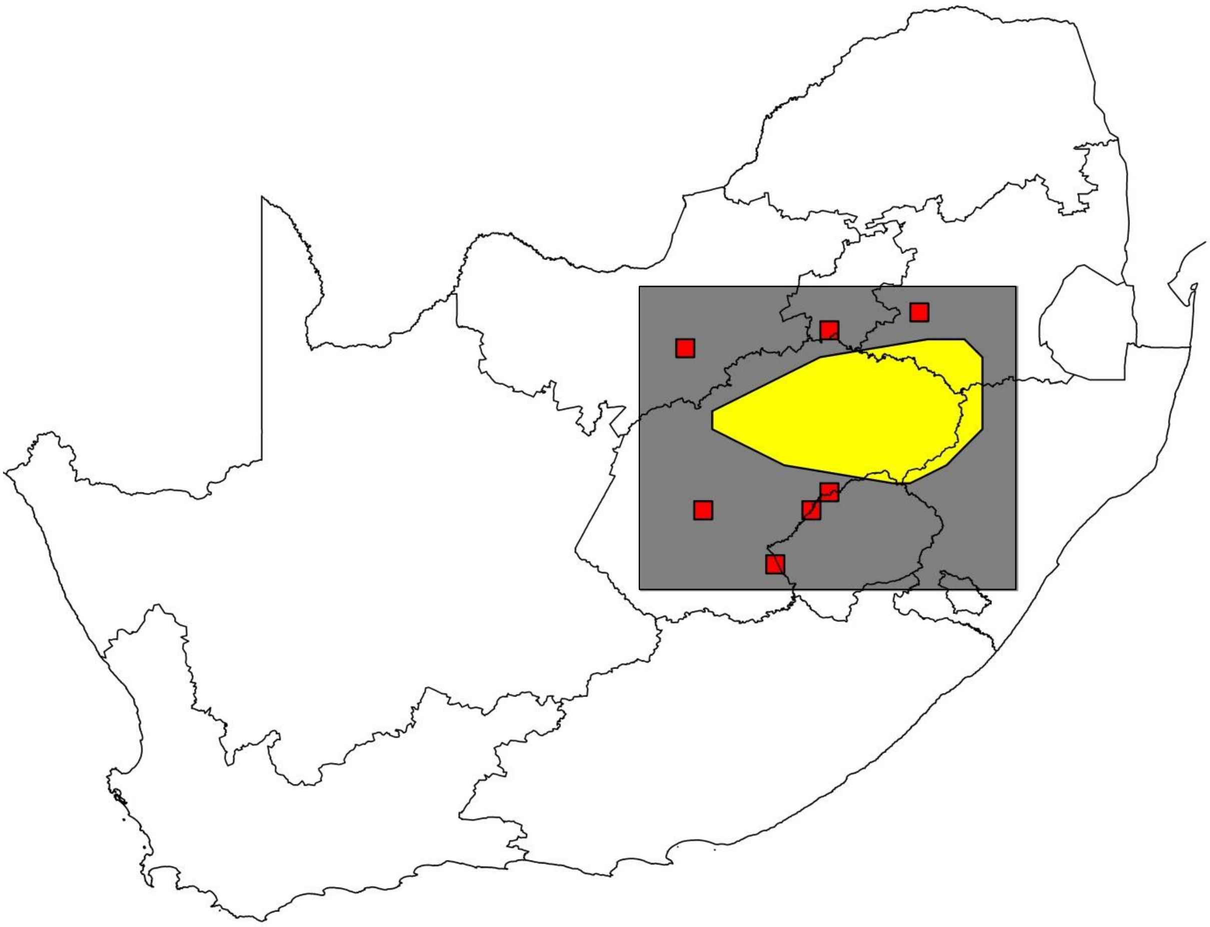


Figure 5.1. Map of South Africa showing *Smaug giganteus* EOO (minimum convex hull) (Yellow), QDGCs containing outlier records (Red) and area for which ENM was produced (Grey).

The model was run with 10 000 background points, 5 000 iterations and 15 replicates. The selected output grid format was 'logistic', in which pixel values ranged from 0 to 1. To assess model performance, I used Receiver Operating Characteristic (ROC) curves (Fielding and Bell, 1997; Philips *et al.,* 2004). The main advantage of ROC analyses is that the area under the ROC curve (AUC) provides a measure of model performance, independent of any choice of threshold (Phillips *et al.,* 2006). To estimate the importance of environmental variables in predicting Sungazer distribution, a jackknife analysis was carried out in MaxEnt. In this procedure, each variable is excluded in turn, and a model is created with the remaining variables, and with each variable in isolation (Phillips *et al.,* 2006).

5.2.3. Identifying priority areas

An ASCII file containing the spatial data for the ENM created in MaxEnt was converted into a raster file in ArcMap 10.0. I used Spatial Analyst in ArcMap to create binary raster files, using a series of thresholds to rank the sites according to habitat suitability. I started with a threshold of 0.99, and dropped by a factor of 0.01 until five distinct zones of optimal habitat were distinguished. Polygons were created around these zones, and the area, proportion of natural area, area expected to be occupied by Sungazers, estimated number of burrows and estimated population size of each zone was calculated. Considering that these nodes represent the optimal zones of habitat suitability for the species across the distribution, these areas should support the highest population densities (Pianka, 1973; Turner, 1977; Hengeveld, 1990). The total population sizes for these zones can therefore be calculated using the upper range of population densities as recorded in Chapter 2. If a node was delineated at a threshold of 0.81, representing 19% of the highest density populations as modelled, then the range of the top 19% of population densities (11 – 19 burrows/ha) was used to calculate the population in the area.

5.3. Results

5.3.1. Maximum entropy niche model

Model performance

The AUC of 0.915 achieved for the model (Fig. 5.2.) reflects a high predictive ability of species presence in geographic space. Values of AUC range from 0.5 (i.e. random) for models with no predictive ability to 1.0 for models that provide perfect predictions (Elith *et al.*, 2006). According to the classification of Swets (1988) AUC values > 0.9 describe ‘very good’, > 0.8 ‘good’ and > 0.7 ‘useful’ discrimination ability.

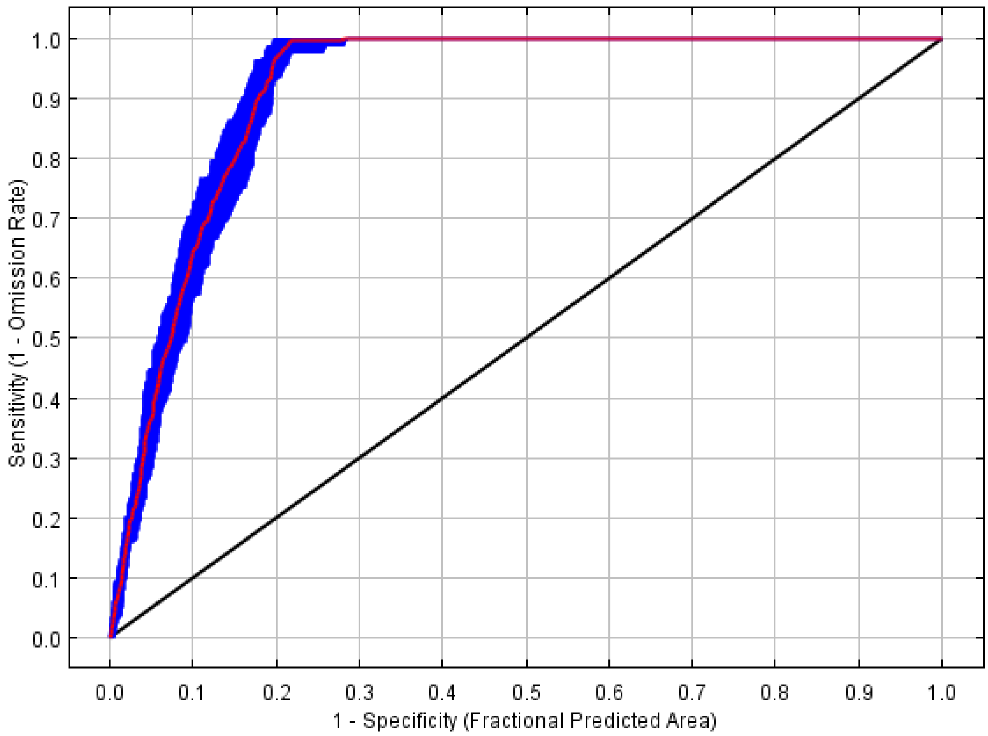


Figure 5.2. The average Receiver Operating Characteristic (ROC) for *Smaug giganteus* ENM created in MaxEnt. AUC mean = 0.915, Std Dev = 0.011.

Niche model

The total area of the distribution predicted by the ENM matches the known distribution of the species closely. The ENM shows several distinct areas that are predicted to represent optimal Sungazer habitat (Fig. 5.3), with extremely high probabilities of Sungazer presence (pixel value > 0.85). In between, and connecting these zones are large contiguous patches that are modelled as being highly suitable habitat, with a high probability (pixel value > 0.7) of Sungazer presence. None of the seven QDGCs that contain dubious presence records were predicted to contain suitable Sungazer habitat, with pixel values below 0.4, and in most cases, below 0.2. These values indicate that these areas are not more likely to contain Sungazers than any randomly selected spot across the modelled area.

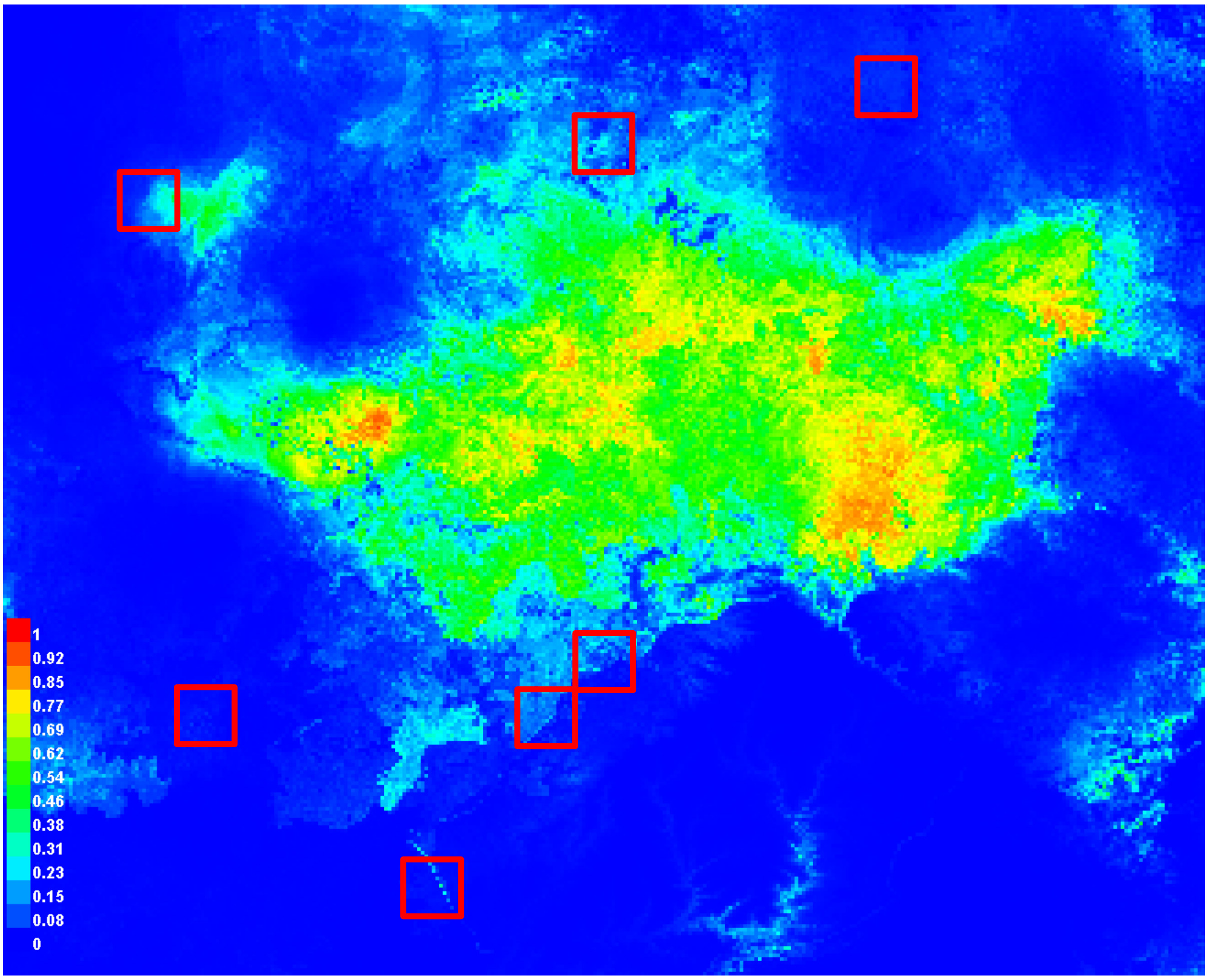


Figure 5.3. A graphical output of the MaxEnt model showing the suitability of Smaug giganteus habitat in 1min cells, where hot colours (pixel value > 0.9) indicates optimal habitat, warm colours (pixel value > 0.7) indicate highly suitable habitat, neutral colours (pixel value > 0.5) indicate habitat with low probably of Sungazer occurrence and cool colours (pixel values < 0.5) have no predictive quality.

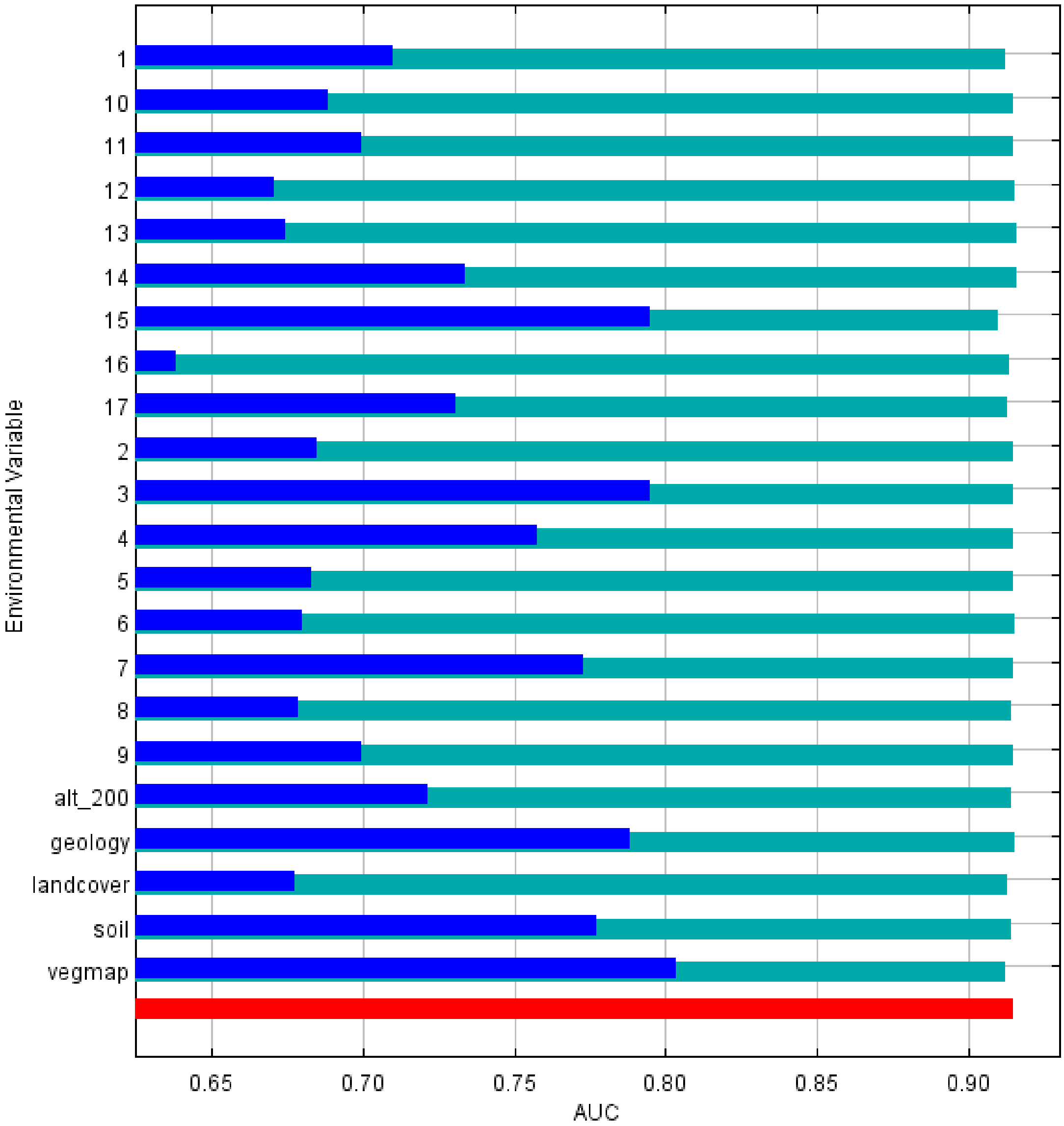


Figure 5.4. Results of jackknife evaluations of relative importance of predictor variables for *Smaug giganteus.* (Turquoise = model performance without variable, Blue = model performance using variable only, Red = model performance with all variables).

The model identified vegetation type, isothermality, precipitation seasonality, geology, soil, annual temperature range and temperature seasonality as the most important predictor variables for Sungazer presence. In addition, jackknife tests showed these variables to have the greatest test gain when used in isolation, with AUC values over 0.75, indicative of these variables being most informative when used in isolation.

5.3.2. Priority areas

Based on high suitability of environmental variables, five primary areas of high population density are evident across the EOO (Fig. 5.5a). When the top 5% and 10% of areas were isolated using Spatial Analyst in ArcMap, the pixels representing these optimal patches were incongruous and scattered, with no clear boundary containing them. Selecting the top ~20% of areas using a threshold value of ~0.8 in Spatial Analyst resulted in more contiguous patches. Thresholds ranging from 0.82 to 0.77 were used to delineate the five primary zones of optimal habitat suitability and the associated high density populations (Fig. 5.5b).

The PAs range in size from 4 477 to 50 641 ha, with a total area of 81 183 ha (Table 5.2.). The percentage of natural land area in each proposed PA ranges from 49-64% (Fig. 5.6), with a total of 52% of the total area remaining natural. Of this natural land area, 1 751 ha can be expected to be occupied by Sungazers, based on the formula created in Chapter 3. Assuming that Sungazers occur at highest population densities in the areas identified by the model as optimal habitat, the upper range of population densities (10-16 burrows/ha) can be used to calculate population sizes contained within these zones. A total of 18 494-26 898 Sungazers can be estimated to be contained within the total proposed PA area. This calculation assumes that at high burrow density, burrow occupancy is constant and independent of burrow density. Although Sungazer burrow occupancy has only been recorded quantitatively at two sites across the distribution (Jacobsen *et al.*, 1990; Van Wyk, 1992), patterns of occupancy appear to be consistent (Fig. 2.7. in Chapter 2). As a conservative measure, the population size for each PA and the total population size across all zones can be calculated using the mean burrow density calculated in Chapter 2. The population size calculated using the mean burrow diversity is 7 255-11 149.

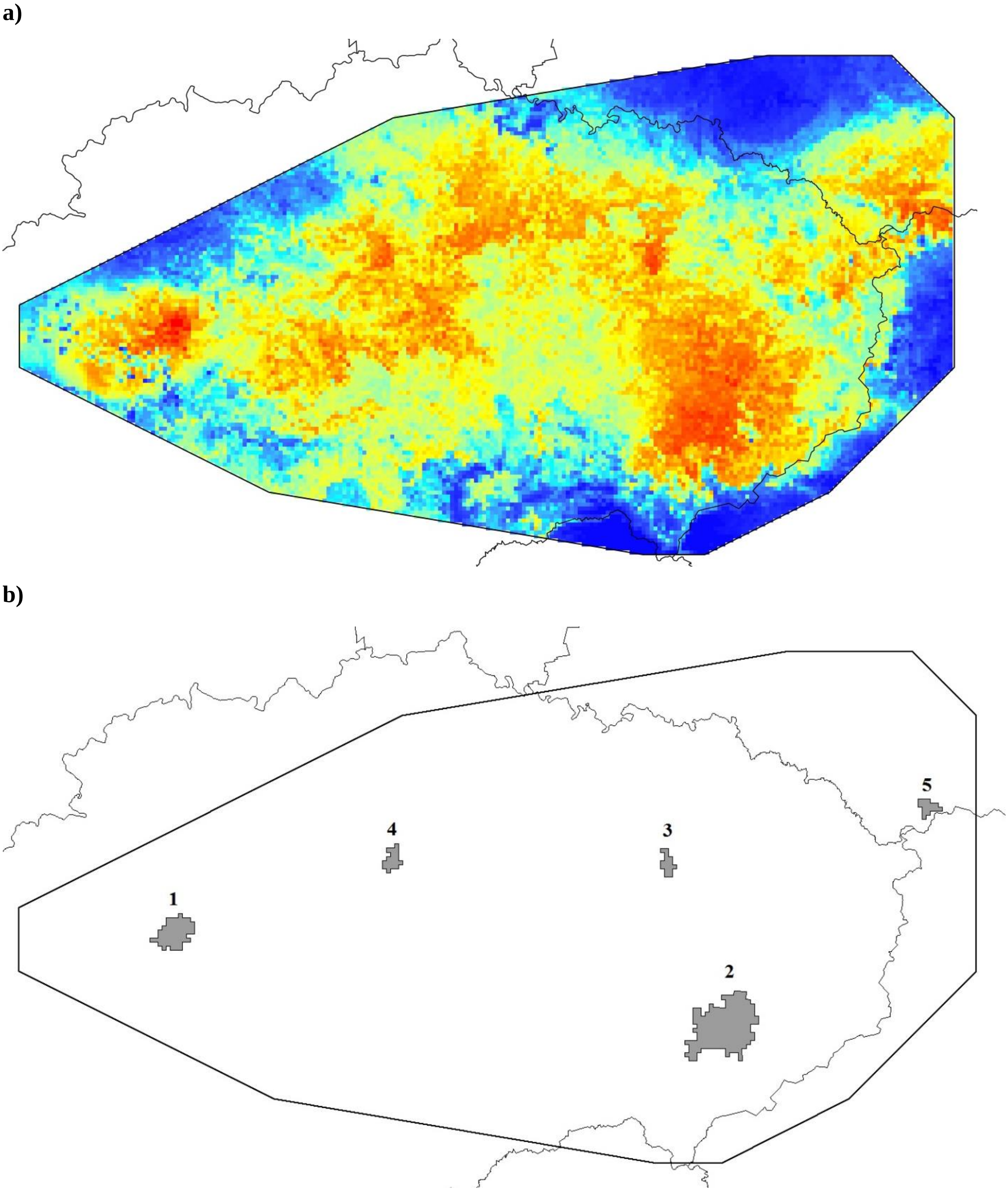


Figure 5.5. a) *Smaug giganteus* EOO (minimum convex hull) showing a gradient of most to least suitable Sungazer habitat along a red to blue gradient, b) polygons created around five primary areas of optimal habitat as predicted by the ecological niche model.

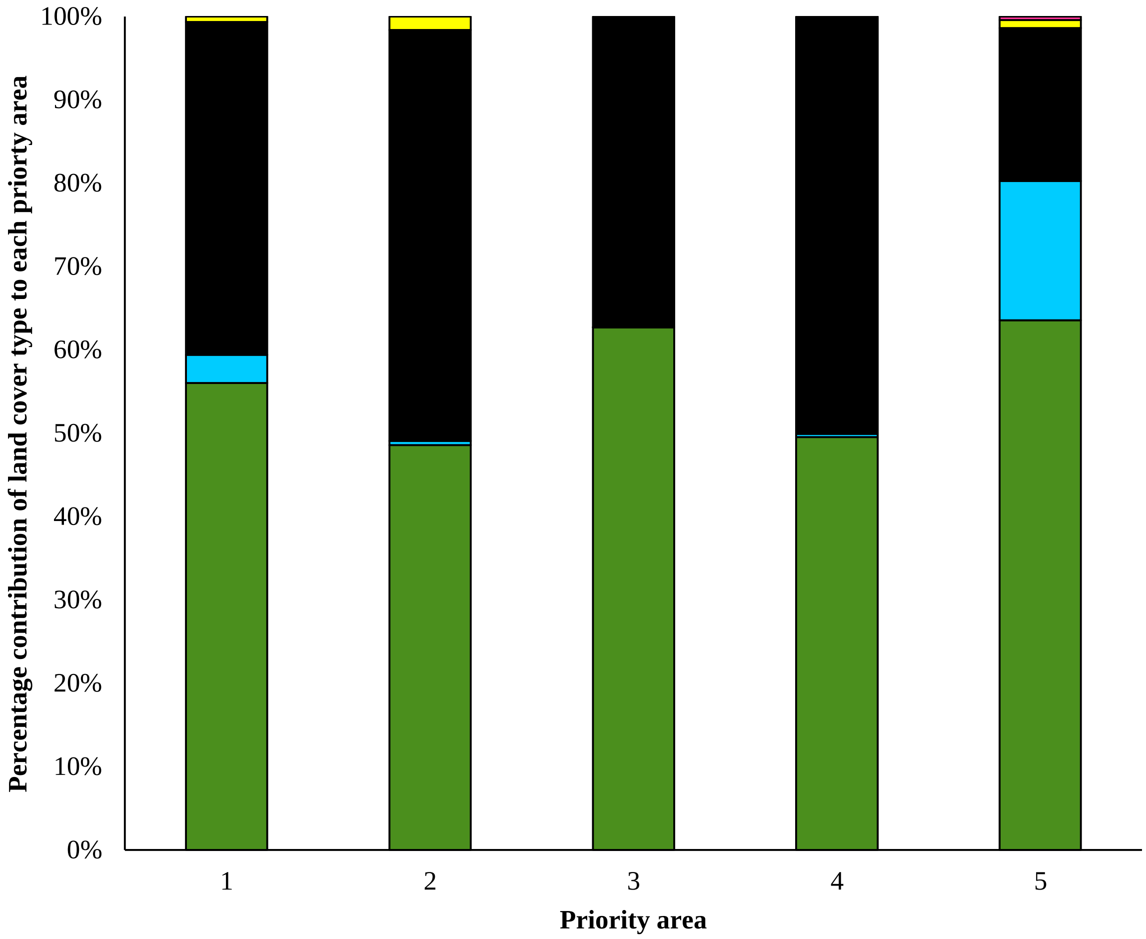


Figure 5.6. Land cover types within each PA as selected using the ecological niche model (Green = Natural untransformed area, Aqua = Water bodies, Black = Cultivated land, Grey = Degraded land, Yellow = Urban Built-up land, Pink = Plantations).

Table 5.2. Location, area and estimated Sungazer population size at five priority areas (PAs).

PA	Location	GPS coordinates of centre of area	Threshold used	Density range	Area (Ha)	Natural area (Ha)	AOO (Ha)	% contribution to AOO	Estimated population size (MBD)	Estimated population size (RBD)
1	Welkom	27°50'58.53"S 26°51'38.53"E	0.82	11-16	16 223	9 089	379	21.6	1 916-2 550	4 003-5 822
2	Harrismith	28°13'13.71"S 29° 0'32.46"E	0.81	11-16	50 641	24 595	1 026	58.6	5 188-6 904	10 836-15 761
3	Vrede	27°35'0.28"S 28°47'41.10"E	0.82	11-16	4 477	2 807	117	6.7	592-787	1 236-1 797
4	Edenville	27°34'26.53"S 27°42'50.58"E	0.78	10-16	5 338	2 643	110	6.3	556-740	1 162-1 690
5	Volksrust	27°21'40.77"S 29°48'36.82"E	0.77	10-16	4 504	2 861	119	6.8	602-801	1 257-1 828
Total					**81 183**	**41 996**	**1 751**	**1 751**	**7 255-11 149**	**18 494-26 898**

5.4. Discussion

Five zones, representing the top 20% of optimal Sungazer habitat across the distribution were delineated as priority areas for conservation focus. These PAs are spread out across the west (Welkom), north centre (Vrede, Edenville), south east (Harrismith) and north east (Volksrust) of the distribution. The total area covered by these zones is 81 183 ha, with 41 996 ha untransformed, and 1 751 ha expected to be occupied by Sungazers. This represents 2% of the EOO of the species, and 1.7% of the AOO, yet are estimated to contain 3-4.4% of the total population due to the higher densities expected in these areas. This assumption is based on the assertion that areas of optimal habitat yield higher population densities than less suitable habitat (Pianka, 1973; Turner, 1977; Hengeveld, 1990), and that the optimal 20% of habitat contains the highest 20% of density ranges recorded for Sungazer populations. The total population size for the proposed PA network based on this assumption is 18 494-26 898 mature individuals. This is compared to the more prudent estimate using the mean burrow density recorded for the species in this study, of 7 255-11 149 mature individuals. Whether these zones do actually support the high population densities as predicted should be validated through further field assessments (Elith *et al.,* 2006). Even at the lower range using the mean density, the total proposed PA network is estimated to contain a population size larger than the mean MVP calculated for vertebrate species by Traill *et al.,* (2006).The expected range however, contains four to five times the mean MVP.

The largest priority area (PA 2) occurs in the east of the distribution, in the Harrismith region. The area of this node is 50 641 ha and makes up 62% of the proposed PA system. This area alone is estimated to support 10 836-15 761 mature individuals, which is 1.8-2.6% of the total estimated population size of the species. This area would serve as the prime region, since it is estimated to contain the largest area that is predicted to contain Sungazers at high densities. The area also contains 2-3x the MVP necessary for the species to survive if the species was extirpated from the rest of its distribution. The second largest node (PA 1) is estimated to support the MVP of the species, and is estimated to contain 0.7-1% of the total population size. The three remaining PAs (3, 4 and 5) in Vrede, Edenville and Volksrust are the smallest nodes, and together contain 0.6-0.87% of the population. Although very little is known about the genetic variation of the haplotype across the distribution, its life history suggests there is potential for genetic isolation of populations. The geographic situation of each of the PAs allows the protection of whatever genetic structure may occur across the distribution. These areas are also well spaced across the EOO such that if stochastic processes such as fire or disease threaten on PA, the other remaining PAs will remain unaffected.

The ENM did not predict Sungazers occurring in any of the seven QDGCs containing dubious records. The QDGC in North West Province contained a patch of habitat ranked with medium-low probability of Sungazer presence (pixel value < 0.4). However, it can be interpreted from the low predictive ability of the pixels in this area that the species is not present within this QDGC. It is however worth investigating, as it is the only region outside of the currently accepted distribution that presents any possibility of an expansion of the EOO. The two records that border the north-western boundary of Lesotho have very low probability of Sungazer occurrence (pixel value < 0.2). The species has been reported as occurring in that country (Ambrose, 2006), but this record has been rejected (Mouton, 2014). The ENM does not predict any of the area within either of the QDGCs containing the records in question as being even weakly probable of containing Sungazers. It is safe to refute these records, along with the other five records of the species that were reported outside the currently known distribution.

MaxEnt can achieve high predictive accuracy with few (10-30) occurrence records (Pearson *et al.*, 2007; Costa *et al.*, 2010), however the predictive accuracy of a model increases with larger number of occurrence records (e.g. Kadmon *et al.*, 2003; Hernandez *et al.*, 2006; Wisz *et al.*, 2008). This study used a total of 536 occurrence records well spread across the known distribution. The species' ecological niche is therefore well represented by the occurrence records input into the MaxEnt programme. The outputs of ENMs represents a species' fundamental niche rather than realised niche (Guisan and Thuiller, 2005; Pearson, 2007; Kumar and Stohlgren, 2009). Areas that are modelled as ideal habitat for a species occurrence may not be occupied due to different causes, such as dispersal constraints (e.g. by presence of geographic barriers), competition, lack of prey, human activities, physiological constraints, stochastic events and historical factors, among others (Pulliam, 2000; Phillips *et al.*, 2006; Rodríguez *et al.*, 2007; Soberón, 2007; Jiménez-Valverde *et al.*, 2008; Kearney and Porter, 2009). This is true for instance in KwaZulu-Natal, where the ENM predicts a strip of the province as being prime Sungazer habitat. While the conditions appear to be suitable, the Drakensberg mountain range presents a physical barrier that limits the distribution of the species to this area. Diedericks and Daniels (2014) found that the MaxEnt predicted distribution for *Cordylus cordylus* matched the known distribution very closely, however presented a slightly wider range for the species, as would be expected without many of the physical barriers to distribution. However, despite methodological limitations, the use of predictive modelling of species distribution stands out as a useful tool for land-use planners seeking to make better decisions about biodiversity management and conservation (Rodríguez *et al.*, 2007).

Ecological niche models have become increasingly popular as conservation tools, and this study uses ecological niche models to guide the prioritisation of protected areas to aid the species' conservation.

The model used highlights areas that contain high density populations, and conservation of these areas allows for the maximal number of Sungazers to be protected, while keeping area to a minimum. Given the current state of the grasslands in South Africa, and the extremely limited area afforded formal protection, the implementation of protected areas across the distribution is vital not only to Sungazer conservation, but the conservation of the Grassland Biome as a whole. In this way, the Sungazer serves as a flagship species for grassland conservation, and adds to the growing system of reptiles being important flagship species for conservation. While the five priority areas as predicted by the model serve as guidance to where protected should be set up, it is also important to increase the heterogeneity of the gene pool that is conserved, by selecting ad-hoc protected areas between the priority areas. Ideally, a network of protected areas that cover the priority areas and link them through corridors would serve as the best approach to conserving species. The final chapter of this thesis deals with how this can be achieved.

CHAPTER 6:

RED LIST ASSESSMENT, CONSERVATION RECOMMENDATIONS, FUTURE WORK AND CONCLUSIONS

6.1. Red List assessment

I summarized the history of the conservation status of *S. giganteus* in Chapter 1, and raised concern that, since the first assessment, no empirical data had been collected to assess changes in populations and the habitat of the species. In my study, the EOO, AOO, population size, decline in habitat and populations were calculated for the species using empirical, quantitative data. The conservation status of the species can be assessed for the first time using declines of known populations, a loss of habitat across the distribution, and the geographic range. In this assessment, the most recent IUCN Red Data List Categories and Criteria (Version 3.1.) (IUCN, 2012) are used to assess the status of the species.

6.1.1. Generation length

In order to standardise rates of decline in the habitat and populations species for the IUCN Red Data List, declines are measured over the length of three generations (or 10 years, if three generations do not span more than 10 years) to take the life history of different life forms into account. A generation is defined by the IUCN as the average age of parents of the current newborn individuals in the population, and is greater than the age at first breeding, but less than the age of the oldest breeding individual (IUCN, 2012). Using information from the literature, the generation length of *S. giganteus* can be calculated.

Age of first reproduction

Sungazer reach sexual maturity when males and females reach the SVL of 165 and 170 mm respectively (Van Wyk, 1992). This is estimated to occur late during the fourth year, or early in the fifth year of a Sungazer's life (Van Wyk, 1992). Here I assume the age of first reproduction to be 5 years.

Longevity

There are no measures of longevity of Sungazers in the wild. Sungazers have relatively slow growth rates and may only reach maximum length in their eleventh year (Van Wyk, 1992). Longevity records for Sungazers in captivity show that the species can live for up to 25 years (HAGR, 2014). Landowners who have kept Sungazers in enclosures in the natural environment within the distribution of the species have claimed maximum longevity of 35 years. In absence of data confirming these claims, I have assumed the maximum longevity for the species as 25 years.

The formula below is used to calculate the generation length for the Sungazer, as defined by the IUCN (2012).

$$Generation\ Length = AFR + \frac{ML - AFR}{2}$$

$$AFR = \boldsymbol{age\ of\ first\ reproduction}$$

$$ML = \boldsymbol{maximal\ longevity}$$

Using these estimates, the generation length for *S. giganteus* is 15 years. The most recent assessment of the species assumed a generation length of 9 years (Mouton, 2014). The calculation in my study therefore extends the period over which declines in populations and habitat are measured from 27 to 45 years.

6.1.2. Criterion A: Population reduction

In Chapter 4 I calculated a population decline of 20.5% over 35 years, and habitat decline of 13.3% over 9 years. Assuming linear rates of decline, a population decline of 26.4%, and a habitat decline of 66.6% over three generations can be extrapolated. The habitat decline recorded was primarily due to transformation of natural land into cropland, and as mentioned previously, Sungazers and crop monoculture have the same habitat requirements within the Highveld Agricultural Region. Thus declines of natural area within the distribution are assumed to correlate directly with a reduction in Sungazer habitat, and therefore a reduction in populations. The population reduction can be classed as irreversible, as reduction is linked directly to destruction of habitat that Sungazers are likely to occupy. Based on anecdotal evidence from farmers across the distribution, *T. triandra* takes between 60-100 years to recolonize fallowed land, and the time taken for a reversion of soil quality necessary to sustain suitable prey items is unknown, but is suspected to be even longer. Thus far, there have been no reports of Sungazers recolonizing land previously used for crop plantations (Newbery and Jacobsen, 1994).

The rates of decline, and the irreversible nature of the transformation measured in this study places the Sungazer within the 'Vulnerable' category, on Criterion A2 ("population reduction observed, estimated, inferred, or suspected in the past where the causes of reduction may not have ceased OR may not be understood OR may not be reversible"), subcriterion b ("an index of abundance appropriate to the taxon"), c ("a decline in area of occupancy (AOO), extent of occurrence (EOO) and/or habitat quality") and d ("actual or potential levels of exploitation").

Decision using Criterion A: Vulnerable A2bcd

6.1.3. Criterion B: Geographic range

In Chapter 3 I calculated the EOO and AOO of the species as 58 338 km² and 1 037 km² respectively. The EOO is above the minimum area (< 20 000 km^2) to be classed as 'Vulnerable'. However, Sungazers were found to occupy only a small portion of the EOO, and the AOO of 1 037 km² is small enough (< 2 000 km^2) to classify the species as 'Vulnerable'. The criteria to be used is therefore B2, along with the sub-criteria of a ("severely fragmented"), b ("continuing decline in any of: (i) extent of occurrence; (ii) area of occupancy; (iii) area, extent and/or quality of habitat; (iv) number of locations or subpopulations; (v) number of mature individuals").

Decision using Criterion B: Vulnerable B2ab

6.1.4. Criterion C: Small population size and decline

The total wild population of Sungazers in the wild has been estimated in this study as approximately 998 247 ± 141 632 individuals. Adults comprise 61.2% of Sungazer populations, and the number of mature individuals in the population is therefore approximately 610 927 ± 86 679. Based on this population size, the species does not meet the criteria to be classed as 'Vulnerable' (< 10 000).

Decision using Criterion C: Not threatened

6.1.5. Criterion D: Very small or restricted population

As above, the population size of the species exceeds the limit required here to fall into any threat categories.

Decision using Criterion D: Not threatened

6.1.6. Criterion E: Quantitative analysis

The data currently available on the life history of the Sungazer are currently not sufficient to conduct a quantitative analysis for the species.

6.1.7. Conclusion

Smaug giganteus fits into the 'Vulnerable' category on the IUCN Red Data List, based on the small AOO (< 2 000 km^2), population reduction of known populations (26.4%) and inferred from habitat loss (66.6%), and the severe fragmentation within the distribution. However, there is no indication that the trends of habitat loss will cease in the future, as agricultural needs tend to increase with growing human population. The habitat loss is irreversible, and at least up to this point, no fallowed lands have been observed as being recolonized by the species. This is likely due to the significant changes to the soil structure, and the lack of invertebrate prey species that are associated with the natural *Themeda* grassland. With linear rates of decline assumed for habitat reduction recorded, the Sungazer can be expected to reach 'Endangered' status in 35 years (i.e. 2049).

6.2. Recommendations for conservation

6.2.1. Protected areas

In Chapter 5 I highlighted the need for the Sungazer to be conserved within officially protected areas, and made recommendations as to where conservation efforts should be focused. The area of land that is recommended for protection is 81 183 ha. The ideal situation for the long-term protection of the Sungazer is the declaration of this land as nature reserves. Purchasing land for conservation purposes is costly, and in developing countries such as South Africa, funds are seldom available to make such purchases (Urbina-Cardona and Flores-Villela, 2010). An alternative to the purchasing of land by conservation authorities are stewardship agreements.

In South Africa, biodiversity stewardship programmes have proved to be very successful in protecting terrestrial ecosystems (Driver *et al.*, 2011). Stewardship programmes involve communities and private land owners agreeing to restrictions on use of the land, in return for formal protected area status, an exclusion from property rates, and possible income tax benefits (Worrell and Appelby, 2000; Driver *et al.*, 2011). The conservation authority provides technical advice and management assistance, but the primary responsibility for management remains with the landowner. The cost of stewardships is small in comparison to the cost of managing and acquiring land. Evidence to date suggests that biodiversity stewardship contracts are approximately ten times cheaper than the acquisition of land. This is partly because the state does not bear the upfront cost of acquiring the land, and also because the landowners themselves bear most of the ongoing management costs, thus mobilising private resources for public benefit. This cost-effectiveness makes stewardships an attractive approach to creating and expanding protected areas. Establishment and roll-out of biodiversity stewardship programmes in all provinces is an urgent priority for supporting cost effective expansion of the protected area network.

In Chapter 3 I detailed the restricted occurrence of the Sungazer across the EOO, and calculated that only 4.17% of the EOO is occupied by the species. The sparse occurrence of the species, in distinct sub-populations means that within a piece of privately owned land, if 10% of the area is designated as a protected area, that land can be focused specifically around areas where the species occurs, and tracts of suitable habitat in a buffer zone around populations. This focus ensures that conservation efforts are maximised towards the most important areas across the distribution. The Endangered Wildlife Trust (EWT) initiated a stewardship programme late 2013, focusing on the protection of areas occupied by Sungazers to meet these needs. The research provided in this study will serve as a guide to where stewardships are focused, with the aim of the total recommended area of 81 183 ha protected.

6.2.2. The Sungazer as a 'flagship species'

Charismatic, well-known and well-liked animals are often branded as 'flagship species' and are used to arouse public interest in the conservation of the species and its habitat, and to promote the broader aspects of conservation, including ecological awareness and the economic value of conservation (Dietz *et al*., 1994; Smith and Sutton, 2008). While flagship species have historically tended to be well-known mammals, the appeal of less traditional charismatic species should be not be overlooked, as they can draw attention to local conservation issues as adequately as do large mammals (Entwistle, 2000; Entwistle and Dunstone, 2000). Reptiles such as the Bermuda Skink *Eumeces longirostris* (BAMZ, 1997) and the Antiguan Racer *Alsophis antiguae* (Daltry *et al.,* 2001) have been utilised as important focal conservation points in their native countries. The recently discovered Northern Sierra Madre Forest Monitor (*Varanus bitatawa*) is being touted as a flagship species for conservation in the Phillipines (Welton *et al.,* 2010). These large monitors are conspicuous and well known by locals and are hoped to direct attention at the remaining forests in Luzon. In the Mexican district of La Comarca Lagunera, there is high endemism of lizards, and the use of several species as flagships to protect other regional flora and fauana has been recommended (Gadseden *et al.,* 2012). This increase in the usage of reptiles as flagship species suggests that smaller animals, given that they fulfil the criteria, show the potential to assist as valuable conservation tools.

Given the charismatic and iconic appeal of the Sungazer, I propose that it be championed as a flagship species for the Highveld Grasslands. Under of the umbrella of the high habitat demands of a potential flagship species such as the Sungazer, species that co-inhabit these regions of the grassland, and the habitat itself are also afforded protection (Lambeck 1997). Traditionally, the criteria for a flagship species have been: 1) endemism 2) economic value 3) declining populations 4) potential as an umbrella species (Simberloff, 1998; Caro and Doherty, 1999; Entwistle and Stephenson, 2000; Bowen-Jones and Entwistle, 2002). Bowen-Jones and Entwistle (2002) suggest ten criteria by which a flagship species should be defined. The Sungazer meets these ten criteria (Table 6.1), and I therefore suggest that the species is touted as a flagship species for conservation of the Grassland Biome. Conservation measures implemented across the distribution of the Sungazer will also benefit other endemic and threatened species that are at risk from grassland degradation within the same area, such as Botha's Lark (*Spizocorys fringillaris*), the Blue Crane (*Anthropoides paradisea*), the Yellow-breasted Pipit (*Anthus chloris*) and the Orange Mouse (*Mus orangiae*).

Table 6.1. Justifications for the use of the Sungazer as a flagship species (*in sensu* Bowen-Jones and Entwistle, 2002).

Criteria	**Justification**
1. Geographic distribution	Endemic to the Highveld grasslands of South Africa
2. Conservation status	Vulnerable due to habitat destruction, habitat fragmentation and exploitation for the pet and traditional medicine trades
3. Ecological role	Numerous burrow systems influence soil structure and composition
4. Recognition	Highly recognisable - known ubiquitously across the distribution and well known in the country
5. Existing usage	Used in the logo of the South African Reptile Conservation Atlas and the Sungazer Working Group, the interests of which are aligned with the conservation of the species
6. Charisma	Archaic, heavily spiked appearance and large body lend to the Sungazer's unique and charismatic appearance
7. Cultural significance	The Sungazer is used in traditional medicine by Sotho and Zulu communities
8. Positive associations	The Sungazer is associated with healthy grassland, as the species only lives on pristine grassland. Landowners take pride in in Sungazer presence
9. Local knowledge	Local Sotho, Zulu and Afrikaans communities are overtly aware of the conspicuous lizard and its distinctive burrow
10. Local names	*Ouvolk* (old people) is a common Afrikaans name, *Sonkyker* and *Skuurwejantjie* are less commonly used Afrikaans names. Commonly known as *Pathakalle* in Sotho and *Mbedla* in Zulu.

6.2.3. Fine-resolution monitoring programme

Sungazer occurrence, as with the occurrence of most South African reptiles (Bates *et al.,* 2014), has historically been recorded at QDGC resolution. A QDGC covers approximately 650 km^2, depending on latitude (~667 km^2 across the distribution of the Sungazer). It is difficult to monitor population declines at this resolution, as the species can still be recorded as present within a QDGC, even though the density within the QDGC may have declined drastically. The most sensible method of measuring decline in a species such as the Sungazer that lives in discrete colonies is to record occurrence at the colony resolution. Patterns of loss of populations at a small (potentially farm scale) would provide meaningful insight to the trajectory of the decline of the species over time. I recommend that in parallel with stewardship programmes that aim to create protected areas, a monitoring programme is initiated, with landowners reporting to a programme manager on a regular basis.

6.2.4. Education programme

The harvesting of animal and plant species for use in traditional medicine is tied to traditional beliefs that often extend back millennia (Adeola, 1992; Anageletti, 1992; Lev, 2003; Alves and Rosa, 2005; Alves *et al.,* 2007). In modern times, such harvesting is often no longer sustainable, as demands for products derived from animals and plants increase with growing populations, leading to commonly-used species being threatened (Lee *et al.,* 2008). A potential solution to alleviate the pressure on species threatened by this trade is an education programme that focuses on teaching youths in rural areas about the importance of reptiles, and particularly Sungazers in ecosystem function. An appreciation for the species and its habitat may engender the support of the conservation of the Sungazer and other commonly used species, by communities within the distribution.

6.3. Recommendations for future work

6.3.1. Development of translocation protocol

The Highveld grasslands are constantly undergoing transformation as a result of the construction of dams, agriculture, power stations, roads, mines and other developments. These developments are often planned to occur directly over existing Sungazer colonies, and efforts are made to translocate affected populations to nearby areas. Unfortunately, all previously-documented translocations have been met with very low success rates and essentially do not aid in protecting the translocated populations. Understanding the habitat choices made by a habitat specialist such as the Sungazer is integral in selecting suitable areas for translocation populations. The development of an effective translocation protocol has been recommended by Van Wyk (1988) and Mouton (2014). The poor success of previous translocations has been associated with the unsuitability of constructed burrows, disruption of family and social structure within a colony. As a result, Sungazers do not remain in the burrows constructed for them and are thus exposed to predation by Secretary Birds, Yellow Mongoose and other predators (Groenewald, 1992). It is therefore very important to take the following aspects into consideration in order for future translocations to be conducted successfully:

1) Within-burrow and between-burrow social structure.
2) Burrow structure and why Sungazers appear require self-dug burrows.
3) Environmental variables and climate envelope required by the Sungazer.

A study testing a variety of translocation techniques (e.g. hard-releases vs soft-releases) and methods of burrow construction is necessary to develop a translocation protocol for the species that has a high success rate.

6.3.2. Investigation of genetic structure across the distribution

Sungazers are long-lived and individuals seldom travel further than a metre from their burrows, except for mating purposes (Van Wyk, 1992; Ruddock, 2000). There is therefore a potential for populations at distant ends of the distribution to have experienced a degree of genetic isolation over time. Microsatellites should be developed for the Sungazer, so that the landscape genetics of the species can be studied. An understanding of the genetic structure of the species across the distribution will allow for educated decisions to be made regarding translocation localities that fall within the gene pool of the species. Depending on the genetic structure that is elucidated, it might be possible to assess the origin of animals in the pet and muthi trades, and identify poaching hotspots across the distribution.

6.3.3. Investigation of long-term population dynamics

In this study, I found that 20.51% of populations were extirpated over a 35 year period. In order to properly interpret this decline as a natural fluctuation in populations or a true decline, a study of population dynamics over a large temporal scale is necessary. McIntyre (2004) tagged 200 Sungazers with Passive Integrated Transponders (PIT-tags) and recorded the GPS locations of the burrows they were found in. This presents an opportunity for a necessary study to examine longevity, turnover rate and fluctuation in population size. By understanding how populations function over a long time period in an undisturbed habitat, the decline recorded in this study can be interpreted as either a natural fluctuation in population or an anthropogenically mediated decline through habitat disturbance, or illegal harvesting.

6.3.4. Investigate effects of climate change

Climate change has been shown to lead to declines in extirpations of reptile populations, and reptile species extinction rates based on climate change are expected to reach 20% by 2080 (Sinervo *et al.*, 2010). *Smaug giganteus* has a restricted distribution, and the species' presence at sites has been shown to be associated with vegetation type, precipitation and temperature (see Chapter 5). Ecological niche models that project the distribution of the species into possible future scenarios of climate change will indicate how the distribution of the species will change, with climate change. An understanding of potential shifts in distribution will augment current conservation plans, in terms of assessing the changes in population size, and where protected areas should be situated to best conserve the species in future conditions.

6.4. Conclusions

The IUCN conservation status of *S. giganteus* remains ‘Vulnerable’. However, I provide evidence in this study to suggest that the species might reach the ‘Endangered’ category by ~2050 if current trends in habitat loss continue. While the EOO is relatively large in size, the area actually occupied by the species (AOO) is a small fraction of this area. This means that the species is likely to be safe from stochastic threat processes, but the population size contained within the broad EOO minimum convex hull is smaller than might be indicated from the large size of the polygon. However, it is not small enough to, in itself, justify the species being placed in any of the threat categories on the IUCN Red List. The rate of habitat decline compounded with the degree of fragmentation already existing across the distribution, currently present the biggest threats the species.

Conserving *S. giganteus* within strategically-located protected areas identified in my study is likely to be the most effective conservation strategy. This will ensure that relatively large populations can persist within a network of optimal habitat. Populations monitored at the colony level across the distribution will allow detection of trends in natural and anthropogenically-mediated fluctuations. Education programmes aimed at school children in rural areas are also likely to be an effective support for the conservation of the species. Future research should focus on the genetics of the species so that the genetic structure across the distribution. Measures of between-population and within-population genetic structure will elucidate levels of dispersal, shed light on new colony formation, and foster an understanding of how further habitat fragmentation is likely to affect gene flow. This information could also be used to assess the suitability of translocation as a conservation tool.

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