
THE REINTRODUCTION OF BEARDED VULTURES IN SOUTH AFRICA: A FEASIBILITY ANALYSIS

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ABSTRACT

The southern African population of bearded vultures, *Gypaetus barbatus*, is declining rapidly and plans for windfarm developments within the core of this species' range threaten to accelerate the population's passage to extinction. As an insurance against such a situation a reintroduction has been proposed to establish a second bearded vulture population within their historic South African range. Before such a scheme could occur suitable areas, if present, will first need to be identified and the requirements and best implementation strategy will need to be determined. Therefore, the aims of this study were (1) to identify the most suitable site for such a reintroduction and (2) to provide some insight into the potential outcomes of different release strategies. Habitat modelling and GIS techniques were used to identify potential reintroduction sites, most notably based on the presence of cliffs. Potential reintroduction sites were then compared based on a range of habitat attributes, of which the amount of human settlement and power line density was considered most important. Five potential reintroduction sites were identified with the two highest ranking sites situated mostly within the Eastern Cape Province. Various release strategies ranging from captive breeding prioritization to release prioritization were modelled using stochastic modelling software. Results indicated that straight releases, without any captive breeding support, had a high probability of failure (defined <34 individuals) ranging between 78.3 and 95.7% across different mortality scenarios over a 30 year period. Supplementation from captive breeding reduced this to between 25.5 and 49.8%. Although it is important for mortality rates to be lower at the reintroduced site this study shows that a reintroduction initiative can be valuable even if this is not the case, as a reintroduction initiative can reduce the probability of extinction (one sex remains) of the species in southern Africa after 50 years by approximately 30%. This study concludes that a captive breeding programme is imperative for the success of the reintroduction and is a prudent measure considering the continuing decline of the species. However, a complementary study examining release sites on the ground as well as stakeholder attitudes and the socio-economic impacts of bearded vultures will be required before the reintroduction can be implemented.

1. INTRODUCTION

1.1 *Global species loss*

Species are going extinct at a rate that is 100 to 1000 times higher than pre-human extinction rates (Pimm *et al.* 1995). This situation has resulted in the proclamation of the Earth's sixth mass extinction event (Barnosky *et al.* 2011). Human enterprise is at the root of this extinction event and has brought about environmental issues such as habitat loss, pollution, global warming, and over exploitation to name only a few. As a result of this, one-fifth of the world's vertebrate species are classified as threatened and on average 52 vertebrate species move one category closer to extinction each year (Hoffmann *et al.* 2010). Without conservation efforts this rate of decline would have been one fifth higher (Hoffmann *et al.* 2010). Nevertheless biodiversity loss does not appear to be reducing (Butchart *et al.* 2010). This is despite the 2002 commitment by world leaders, through the Convention on Biological Diversity (CBD), to have achieved a significant reduction in the rate of biodiversity loss by 2010 (Butchart *et al.* 2010). Consequently there is a reason to increase conservation efforts globally, both for reasons of moral obligation and self-preservation.

1.2 *In situ and ex situ conservation*

In situ and ex situ conservation are viewed as two separate conservation strategies aiming to prevent the loss of biodiversity. These two strategies are often seen to compete with each other for limited conservation funds and as a result the utility of ex situ conservation has often been questioned (Balmford *et al.* 1995). In situ conservation refers to the protection of species in their natural habitat with the view of conserving ecosystem integrity (CBD 1993). Climate change projections indicate that many species will experience range shifts and those species that find themselves unable to track these shifts will be vulnerable (Thomas *et al.* 2007; Thuiller *et al.* 2006; Pritchard *et al.* 2011). Some ranges are consequently already receding (Thuiller 2007). Thus, in future protected areas may no longer be suitable for the species they were created for. This, and the fact that species respond differently to climate change, challenges the underlying principle of in situ conservation, that component species can be protected through site specific ecosystem conservation (Pritchard *et al.* 2011).

In situations where in situ conservation has failed, or is likely to fail, at stopping population declines, ex situ strategies such as captive breeding, translocation and reintroduction could be used to save species from extinction, or at least provide an insurance against it (Burney and Burney 2007). Successful ground breaking reintroductions such as the reintroduction of the Arabian oryx

(*Oryx leucoryx*) to Oman (Stanley Price 1989), golden lion tamarins (*Leontopithecus rosalia*) in Brazil (Kleiman and Mallinson 1998), and peregrine falcons (*Falco peregrines*) in North America (Cade and Burnham 2003) inspired a large increase in the number of reintroductions attempted (Seddon *et al.* 2007). Many of these reintroductions were overenthusiastic and ill-conceived. For example, a review of reintroductions showed that only 5% of the 74 projects reviewed were considered to be successful in 1987 but that these had declining populations by 1993 (Wolf *et al.* 1996). This low success rate has often been used as an argument against reintroduction and captive breeding initiatives. Many of these unsuccessful reintroductions can, however, be ascribed to a lack of adequate planning and monitoring (Seddon *et al.* 2007). For these reasons the World Conservation Union (IUCN) has produced a set of guidelines to prevent the implementation of inappropriate reintroductions (IUCN 2013). These guidelines stipulate the need for a feasibility analysis before reintroductions are attempted. The feasibility analysis should include:

- An assessment of the availability and suitability of the habitat at proposed reintroduction sites;
- Affirmation that the threats that caused the previous extinction have been correctly identified and have been removed;
- A risk assessment to assess the potential social, economic and ecological effects of the project. These risks should be balanced against the potential benefit of the project and if a high degree of uncertainty remains alternative solutions should be sought; and
- Some type of modelling to predict the outcome of the project under various scenarios to provide valuable insight for selecting the optimal strategy.

The past few decades have shown many improvements in the field of reintroduction biology in terms of technical knowledge and practices that overcome limitations faced in the past (Pritchard *et al.* 2011). Consequently the number of successful reintroductions seem to be increasing (Wanless *et al.* 2002; King *et al.* 2012; Reynold *et al.* 2012) and raptors especially seem to be well suited to this management strategy (Evans *et al.* 1999; Jones *et al.* 2008; Schaub *et al.* 2009; BirdLife 2013; Monti *et al.* 2014). These successes illustrate that reintroductions and captive breeding can prove to be vital conservation tools.

1.3 Vulture population declines

Species loss compromises ecosystem integrity and influences the delivery of ecosystem services on which human well-being depends (Ehrlich and Mooney 1983). The scavenger guild for example plays a vital role in waste removal, disease control and nutrient cycling (Prakash *et al.*

2003). Vultures are the only known vertebrates that are obligate scavengers (Ruxton and Houston 2004) and are notoriously good at providing this free sanitation service. Ogada *et al.* (2012b) showed that the mean carcass decomposition time nearly tripled in the absence of vultures. Furthermore, the mean number of mammals at carcasses as well as the mean time spent at carcasses increased three-fold. This increased amount of contact between mammalian scavengers leads to the potential for increased transmission rates of infectious diseases such as rabies and canine distemper. By disposing of infected carcasses vultures most likely contribute to the control of livestock diseases such as brucellosis, tuberculosis and anthrax as well, by removing potential sources of infection (Swan *et al.* 2006). Quicker decomposition of carcasses facilitated by vultures can thus reduce infection risk in humans, wildlife and livestock. The value of vultures to human well-being is clearly illustrated by the increase of \$34 billion in healthcare costs in India between 1993 and 2006, principally due to increases in human rabies infections, which is associated with severe reductions in vulture numbers (Markandya *et al.* 2008).

The loss of vultures can also negatively affect economic activities, most notably through the loss of a cost effective disposal method of organic waste. In Socotra off the Horn of Africa, Egyptian vultures consume up to 22% of the annual waste produced by towns (Gangoso *et al.* 2013). In India vultures pick carcasses clean, enabling tanners to avoid the more costly processes of burial or incineration (Markandya *et al.* 2008). If these processes are used the bones of the skinned animals are no longer available to local bone collectors, who supply these to the fertiliser industry (Markandya *et al.* 2008). Vultures can also be an important ecotourism attraction with the potential to raise extra income for local impoverished communities (Svoronou and Holden 2005; Markandya *et al.* 2008). All levels of society thus stand to lose if vultures disappear, from taxpayers who have to cover the increased healthcare burden to poverty stricken bone collectors.

It is thus disturbing that vultures are one of the fastest declining avian groups globally (Xirouchakis *et al.* 2001; Gilbert *et al.* 2002; Virani *et al.* 2011; Ogada *et al.* 2012a; Krüger *et al.* 2015; Ogada *et al.* 2015), with 61% of vulture species worldwide being threatened with extinction (Ogada *et al.* 2012a). Vulture declines have been reported in Asia, the Middle East, Central America, South America, North America, and Europe (Ogada *et al.* 2012a). Vulture declines in Europe and North America started as early as the mid-19th century and various populations of bearded vultures and Californian condors were on the brink of extinction within a hundred years (Snyder 1983; Mingozi and Estève 1997). In Greece, for example, the bearded vulture experienced a 75% decrease in breeding distribution and an 84% decrease in population size over two decades (Xirouchakis *et al.* 2001). In Asia the use of the nonsteroidal anti-inflammatory veterinary drug, diclofenac-sodium, on livestock led to catastrophic declines in vultures (Prakash

et al. 2003; Oaks *et al.* 2004; Gilbert *et al.* 2006; Shultz *et al.* 2015), including a 95% decline of three vulture species in India within 10 years (Gilbert *et al.* 2002).

In Africa the situation looks no better. In West Africa most vulture species have declined by an average of 95% outside of protected areas between 1973 and 2004 (Thiollay 2007). In Kenya vulture declines of 70% have been recorded in a period as short as 3 years (Ogada and Keesing 2010). Vultures are even declining within protected areas, at an average of 42% inside the Sudanese zone (Thiollay 2007).

1.4 Causes of vulture declines

Vulture declines are most notably driven by poisoning and human persecution, which are present in nearly every declining population (Ogada *et al.* 2012a and 2015; Krüger *et al.* 2015; Brown 1991). In Europe, 69% of bearded vulture deaths were due to shooting and poisoning (Margalida *et al.* 2008). These declines can partly be attributed to the life histories and foraging behaviour of vultures that have low productivity, delayed maturity and high adult survival rates (Ogada *et al.* 2012a). Their longevity make them particularly vulnerable to toxic substances as these can easily accumulate in their bodies over a long period (Ogada *et al.* 2012a). Vulture poisoning is commonly unintentional and most often takes place as a result of baited carcasses put out by farmers to control predator species that are seen as a threat to livestock (Ogada *et al.* 2012a).

Another form of unintentional poisoning, other than the already mentioned effect of diclofenac-sodium on Asian vultures, occurs when vultures ingest lead fragments from spent ammunition when feeding on shot animals. Lead poisoning is a growing concern in many vulture species. Of 20 bearded vultures involved in a satellite tracking study in South Africa, 53% died due to poisoning and 80% of those that died exhibited high levels of lead (Krüger 2014). Lead poisoning is the leading cause of death among reintroduced free ranging California condors in Arizona (Parish *et al.* 2009). Chronic exposure can also have sub-lethal effects on reproductive success (Bueger *et al.* 1996; Scheuhammer 1987), behaviour (Scheuhammer 1987), immune response (Snoeijs *et al.* 2004) and physiology (Fair and Ricklefs 2002). The Convention on Migratory Species is therefore attempting to phase out the use of lead ammunition in favour of non-lead alternatives (Convention on Migratory Species 2014). In 2008 the use of lead ammunition within the Californian condor's range was banned, resulting in reduced lead associated mortalities in the condors (Parish *et al.* 2009), as well as reduced lead exposure in other scavenging birds such as golden eagles and turkey vultures (Kelly *et al.* 2011).

Some poisonings are however intentional. Direct poisoning and persecution also occurs and is motivated by a perceived threat to livestock (Brown 1991), poachers who wish to conceal the locations of their activities (Roxburg and McDougall 2012; Ogada 2014), food demands (Thiollay 2006), traditional medicine (Thiollay 2007; Mander *et al.* 2007; McKean *et al.* 2013) and because of superstitious beliefs (McKean *et al.* 2013).

The high mortality rates caused by poisoning is likely to be additive to the effects of other anthropogenic factors. Negative effects of infrastructure, such as power lines and windfarms, on vultures and other large raptor populations through collisions and electrocutions have been widely documented (Lehman *et al.* 2007; Markandya *et al.* 2008; Smallie and Virani 2010; Boshoff, *et al.* 2011; Ogada *et al.* 2012a). These events account for 22% of bearded vulture deaths in Europe (Margalida *et al.* 2008). This danger is especially present in Southern and North Africa where there is increasing electrical infrastructure development (Jenkins *et al.* 2010; Rushworth and Krüger 2014)

1.5 Southern Africa's bearded vultures

One of southern Africa's most threatened birds is the bearded vulture, *Gypaetus barbatus*. This large raptor is a non-colonial, monogamous cliff nester and pairs generally occupy territories above 1800 m (Brown 1988). This species is unique in that its diet consists almost exclusively of bone (Brown 1990). Although it occurs both in Africa and Eurasia, numbers are dwindling and only isolated pockets remain in Africa, with Ethiopia and southern Africa being strongholds for the species in sub-Saharan Africa. In southern Africa the species has recently regionally been up-listed to critically endangered (Krüger 2015), due to a decline in breeding pairs of between 32 and 51% over the last five decades, leaving only around 100 breeding pairs (Krüger *et al.* 2014a). The breeding range has also declined during the same period by 27% and the species is currently restricted to the Maloti-Drakensberg mountains of Lesotho and South Africa (Krüger *et al.* 2014a). Stochastic models predict an 89% probability of extinction at current demographic rates over the next 50 years (Krüger 2014). The reasons for this decline is the same as for other vultures, with 53% of mortalities being attributed to poisoning and 21% to power line collisions (Krüger 2014).

There is an emerging threat that has the potential to accelerate the decline of the Maloti-Drakensberg population. There are plans for extensive windfarm developments on the Lesotho highlands. Two proposals, one for 42 turbines and another for 100 turbines, have already been submitted and there is a long term goal of having up to 4000 turbines in this area (Jenkins and

Allan 2013). Bearded vultures are considered vulnerable to collisions with turbine blades and habitat modelling has shown that the location of one of the proposed windfarms is in one of the areas that is most heavily used by bearded vultures (Reid *et al.* 2015). Wind farm collisions have led to numerous deaths of especially vultures and large raptors and in extreme cases can push populations towards extinction (Carrete *et al.* 2009; Dahl *et al.* 2012; Bellebaum *et al.* 2013). There is therefore mounting concern about the future of the southern African population of bearded vultures. Ex situ conservation efforts that have been successful elsewhere are thus being considered.

1.6 Bearded vulture reintroductions

The Alpine population of bearded vultures went extinct in the early 1900s due to poisoning and shooting (Mingozzi and Estève 1997). A subsequent reintroduction project started releasing captive bred birds back into the Alps in 1986 (Frey 1992). This project has been deemed successful and it is estimated that no further releases are required (Schaub *et al.* 2009). Similar projects have been initiated in Andulasia (Spain), and Grands Causses (France), with the larger aim of restoring the species across its former European range (Frey and Llopis 2014). Since the start of this initiative a total of 435 bearded vultures have been bred in captivity and 235 of those have been released into the wild across all three projects (Frey and Llopis 2014).

In light of this success, the declining population trend of the southern African population and the looming threat of windfarm developments, a reintroduction of bearded vultures into part of their historic range in South Africa is being considered by the Bearded Vulture Task Force. Bearded vultures used to occur across mountainous areas between Lesotho and Cape Town (Brooke 1984, Figure 1). The aim is that the reintroduced population will function as an insurance population against the regional extinction of the species. Local attitudes, agricultural practices and habitat in this area may have changed drastically since extirpation and suitable habitat and conditions for bearded vultures could again be present. Especially when considering that extirpation in some of these areas began in the 1700s and that many of the misconceptions that led to bearded vulture persecution, are less widespread today (Brown 1991).

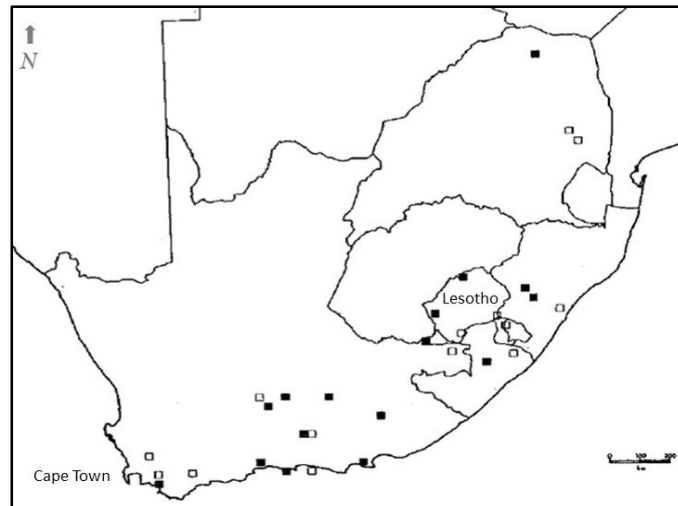


Figure 1: Historic distribution of bearded vultures in Southern Africa. Closed squares indicate published and unpublished material records and open squares indicate published and unpublished sight records (adapted from Boshoff *et al.* 1978).

1.7 Identifying a reintroduction site and acquiring birds for release

Geographical Information Systems (GIS) are a useful tool for identifying suitable habitat for species (McShea *et al.* 2005; Schadt *et al.* 2002) and have been applied in various reintroduction feasibility analyses to identify suitable reintroduction sites (Leaper *et al.* 1999). I used a similar approach in this study to determine which site would be most suitable for a bearded vulture reintroduction based on landscape features, habitat requirements and the most common causes of mortality.

The Analytical Hierarchy Process (AHP) is often used during such analyses (Thatcher *et al.* 2006; LaRue and Nielsen 2011). The AHP, developed by Saaty (1980), is a multi-criteria analysis method that assists in decision making. This method objectively assigns priority scales to the various elements relevant to a specific problem. This is done through experts assigning a scale of absolute judgements during pairwise comparisons on how important one element is in comparison to another. These comparisons provide weightings for each element of the given problem. Measuring and improving the consistency of these judgements is also a concern of the AHP.

This method has been used successfully in solving a range of environmental and ecological problems (Clevenger *et al.* 2002; Kovacs *et al.* 2004), and has also been used to determine the relative importance of each habitat parameters assessed with regards to site selection for reintroduction projects (Thatcher *et al.* 2006; Cruz *et al.* 2014). The technique was therefore considered useful for this study.

After a reintroduction site has been determined, finding individuals for release is the next challenge. Harvesting wild birds could accelerate the decline of the wild population (Margalida *et al.* 2015) and reduce the time available to address the causes of the declines before extinction occurs. Eggs may however be harvested from the wild population in a way that is not detrimental. Females of this species usually lay two eggs but as is the case with many large raptor species siblicide ensures that only one chick is raised (Margalida *et al.* 2004). Due to this situation these second eggs, that are obligate mortalities, can be harvested to contribute to a reintroduction or captive breeding programme without any detrimental impacts on the donor population, which is a requirement of the IUCN guidelines for reintroductions. As a pilot trial, two eggs have already been successfully harvested and raised in this way from the Maloti-Drakensberg population during 2015 (Krüger, S. pers. comm.). It is thus an opportune time to explore various sites for future releases as well as the best method by which to conduct such releases. For example, the establishment of captive populations may be valuable insurance against the extinction of a species as this population may be used for subsequent reintroductions after the impacts in their native habitat have been addressed (Hoffmann *et al.* 2010).

1.8 Population modelling

The alpine reintroduction has shown that re-establishing a bearded vulture population is a long-term and costly venture and can potentially detract funds from other conservation efforts (Bustamante 1998). The accumulated cost was estimated at €70 000 per released individual (Frey 1998). It is thus important to determine how much investment is needed to ensure the success of such a project before it commences so that financial resources can be optimally allocated (Schaub *et al.* 2009). Population modelling is often used to model population trends and gain insight into population dynamics, the probable outcomes of management strategies (Haines *et al.* 2005; Anderson *et al.* 2015) and the outcomes and feasibility of reintroductions (Beissinger and Westphal 1998; South *et al.* 2000; Bach *et al.* 2010). Models can give us a good indication of what is needed with regard to reintroductions as they; i) encourage the formation of quantitative goals, ii) estimate the probability of reaching those goals in a specified amount of time, iii) allow the comparison of alternative management actions based on their effectiveness and cost, and iv) allow the evaluation of available information (Bustamante 1998). It is for such reasons that the IUCN guidelines stipulate that population modelling should be used to assess the feasibility of reintroduction projects (IUCN 2013).

1.9 Aims and objectives

In this study I aim to assess the feasibility of a potential South African bearded vulture reintroduction by firstly identifying the most suitable site for a reintroduction, based on landscape features, habitat requirements and the features associated with the most common causes of mortality and territorial abandonment. Secondly, I aim to provide some insight into which release strategy would be most efficient in establishing an insurance population and decreasing the decline of the species as a whole.

2. METHODS

2.1 Reintroduction site identification

Site identification was done using ArcMap 10.2 (ESRI 2013) GIS software. Bearded vultures are cliff nesting raptors and consequently the first step was to identify suitable cliffs within their historic range (Figure 1). Only areas within their historic range were considered because such areas are most likely to have suitable habitat and present less of a risk as global evidence shows that translocations of species outside their indigenous range can frequently cause extremely negative ecological, economic and social impacts (IUCN 2013).

Cliffs were identified using a 30 m resolution digital elevation model (DEM), the STRM 1-Arc Second Global DEM (available at <http://earthexplorer.usgs.gov/>). Data gaps in this DEM were filled with data from the 30 m resolution ASTER Global DEM, version 2 (available at <http://earthexplorer.usgs.gov/>), using the query:

$$\text{Con}(\text{IsNull}(\text{STRM30}), (\text{ASTER GDEM} + (\text{ASTER GDEM} + 7.68)), \text{STRM})$$

This algorithm fills each data gap in the STRM DEM with data from the ASTER GDEM while adding the average difference between the two DEMs (7.68 m), at the site where data was missing, to compensate for constant shift between the two data sets.

Slope was extracted from the resulting DEM and slopes above a threshold of 45° were considered indicative of cliffs (Loye *et al.* 2009). This simple morphometric approach of using a slope threshold is often used to identify cliff sites that can be source areas of potential rock falls (Guzetti *et al.* 2003; Jaboyedoff and Labiouse 2003; Frattini *et al.* 2008) and has also been applied to identify cliffs and potential nesting sites for a proposed peregrine falcon reintroduction project (Wakamiya 2008).

The ability of the 45° threshold to accurately identify cliffs was tested in two ways. Firstly, visual assessment was done in randomly identified cliff areas using the free map based software Google Earth®. The areas viewed indicated that cliff sites were accurately identified. Secondly, data on the locations of known nest sites were obtained from Ezemvelo KwaZulu-Natal Wildlife and used to assess whether cliffs, that all nests are located on, were identified by the cliff model. This proved to be true in almost all cases, with only a few nest sites not corresponding to any identified cliffs. This is likely due to these nest sites being located on isolated small cliff faces that cannot be detected at the 30m resolution of the DEM used in this study. However, such isolated sites with single breeding territories would not be considered as suitable for reintroduction in any case. Based on available literature (Loye *et al.* 2009) and from this overall assessment, I thus concluded that my cliff model did indeed accurately identify cliffs.

To isolate potentially suitable reintroduction sites I used the species distribution modelling software MaxEnt (Phillips *et al.* 2006) to identify areas with potentially suitable environmental conditions for bearded vultures. MaxEnt models the likelihood of species occurrence based on known locations of occurrence and measurements of a set of variables at these locations. I used MaxEnt to model the potential availability of nesting habitat. Known nest sites in the current range were used as sampling data. Environmental variables included in the model were the presence or absence of cliffs, elevation obtained from my DEM, climate data (BioClim, Hijmans *et al.* 2005) and terrain ruggedness (standard deviation of slope). Elevation and terrain ruggedness had previously been shown to be important in predicting habitat use by this species (Reid *et al.* 2015). These environmental variables were sampled at the resolution of the coarsest layer, which had a cell size of 898 x 898 m.

Preparing the cliff layer for inclusion in the model involved producing a binary layer for cliff presence (slopes above 45°) in each 898 x 898 m cell. During resampling to 898 x 898 m cells, some discrepancies were encountered in that the resampled larger cells, with cliffs present, in some cases excluded nest sites that were clearly associated with them. This arose due to the fact that the nest sites are mostly inaccessible and when recorded in the field with a handheld GPS device, were often recorded a distance away from their actual location. These discrepancies would bias the model to incorrectly associate occurrence data with a “cliffs absent” cell. To rectify this I produced an Euclidean distance map around the identified cliffs and ran a query that specified that if a nest site was within 1 km from a cliff area, then the cell in which the nest site is should be considered to contain a cliff.

I used the standard deviation of slope as an estimate of terrain ruggedness. This was identified as the most efficient method for estimating ruggedness by Grohmann *et al.* (2011). Terrain ruggedness has been found to be an important variable with regards to bearded vulture breeding density (Donzaar *et al.* 1993) and habitat use (Reid *et al.* 2015). It also correlates with greater availability of cliffs for nesting, rocky outcrops that can be used as ossuaries and air current formation suited to bearded vulture flying behaviour (Donzaar *et al.* 1993).

For the MaxEnt models I started with cliff presence, variables were then subsequently added and the changes in predicted areas of occurrence was observed. Using the model output for all variables (Figure 3), areas with more than a 1% probability of occurrence were used as a mask to extract the identified cliffs with which this layer overlapped. This low criterion of 1% was used as MaxEnt was not used in the classical sense of predicting species occurrence but rather as an objective guide to highlight areas where reintroductions might be possible. The overlapping cliffs represented the most suitable nesting sites for bearded vultures with regards to elevation, climate and terrain ruggedness. All extracted cliffs that were within 40 km of each other were then aggregated to form discrete reintroduction sites. Only sites that were at least 100 km away from the Maloti-Drakensberg population's current range were considered as the aim of this project was to establish a separate population. Site 1 and 2 were manually split into two separate sites, because of the large size and longitudinal shape of the area (Figure 4, 5 and 6). The aggregated cliffs were then buffered at a radius of 10 km, about the size of an adult home range (Krüger *et al.* 2014b). Although non-adult birds range more widely than adults, a lack of spatial segregation between age classes has been shown in Europe (Margalida *et al.* 2008). This suggests that non-adults stay within the total area used by all adults. This has been hypothesized to be due to food availability and the presence of feeding stations (Margalida *et al.* 2008). I therefore assumed that the adult home range size buffer encapsulated the entire area that will be used by bearded vultures of all age classes, at each site. Each one of the buffered sites was considered a potential reintroduction site.

2.2 Reintroduction site ranking

To identify the most suitable potential reintroduction site, a range of parameters corresponding to bearded vulture habitat requirements were compared across sites. These variables were power lines, windfarms, human settlements and unsuitable habitat (negative variables) and livestock and protected areas (positive variables), a full description of each can be found in Table 1. ArcMap was used to determine the densities or proportions of these parameters within each proposed

Table 1: Parameters assessed to determine site suitability for a bearded vulture reintroduction.

Variable	Rationale	Measure	Data Source and/or layer name
Livestock density	This was used as a proxy for food availability.	Number of ungulate livestock (pigs, sheep, cattle and goats) per km ² .	2013 National Livestock Statistics ¹
Power line density	Cause of mortality and territory abandonment (Krüger <i>et al.</i> 2015).	Total distance (km) of 11 kV, 22 kV and 1322 kV power lines per km ² .	Eskom 2015.
Proportion Windfarms	Wind farms can have severe consequences for raptors who are killed when colliding with turning blades (Reid <i>et al.</i> 2015).	Proportion of reintroduction site that has windfarm developments, either existing, in process or proposed.	“REEA_OR_2015_Q3” Department of Environmental Affairs 2015, Republic of South Africa.
Proportion Human settlement	Human settlement is assumed to be related to increased instances of poisoning and persecution which are the main cause of mortality and territory abandonment (Krüger <i>et al.</i> 2015).	Proportion of reintroduction site that is classified as human settlement.	“Dea_cardno_2014_sa_lcov” Department of Environmental Affairs 2015, Republic of South Africa.
Proportion Protected	Protected areas represent a safe haven from all threats that is ensured for the foreseeable future.	Total area (km ²) under formal protection.	“SAPAD_OR_2014_Q2” Department of Environmental Affairs 2015, Republic of South Africa.
Proportion Unsuitable habitat	Unsuitable habitat decreases the amount of habitat available to bearded vultures at a given site.	Proportion of reintroduction site considered as habitat that is unusable by bearded vultures (thickly vegetated areas, wetlands, waterbodies, urban areas, mines, commercially cultivated fields).	“Dea_cardno_2014_sa_lcov” Department of Environmental Affairs 2015, Republic of South Africa.
Number of potential nests	Provides an estimate of the potential number of breeding pairs and therefore whether a viable population can be supported at the site.	Calculated number of nest sites based on MaxEnt model outputs and 9km inter-nest distances.	Calculated from MaxEnt extracted cliff sites (see above for explanation).

¹2013 National Livestock Statistics, Directorate: Statistics and Economic Analysis of the Department of Agriculture, Forestry and Fisheries.

Available at: (<https://www.environment.gov.za/mapsgraphics#protectedareas>)

reintroduction site. These measurements were then normalized by dividing each measurement, for a given parameter, by the total of that parameter across all sites. This provided a score for each site in relation to each parameter.

Livestock density, which I used as a proxy for food availability, was only available at the provincial level. I calculated livestock densities for each potential reintroduction site based on the assumption that livestock are evenly distributed across suitable land cover. All open vegetated areas outside of protected areas were considered suitable for livestock. The proportion of such areas per province that fell within my sites was calculated. The total number of livestock per province was then multiplied by the proportion of suitable livestock area that fell within the reintroduction site to provide an estimate of the total number of livestock in each site. From this, livestock density was calculated for the entire potential reintroduction site as the number of ungulate livestock per square kilometre.

The number of potential breeding territories at each reintroduction site was also calculated, but not included in the calculation of site scores. This parameter was only used to assess whether sites were of sufficient size to support the nesting requirements of a viable population. The reintroduced Alpine population was considered to be viable once around 10 breeding pairs had established (Schaub *et al.* 2009), I consequently assumed that 10 breeding territories would be adequate to sustain a viable population. Breeding territories were allocated manually within ArcMap by using a circular guiding shape with a radius of 9 km, which is the mean inter-nest distance for the southern African bearded vulture population (Krüger 2014). Starting with an arbitrarily assigned nest site, nest sites were assigned to cliffs while ensuring that all assigned nest sites were more than 9km away from each other. Only the cliff sites extracted from the MaxEnt model output were considered. Each nest site corresponded to a single breeding territory. In this manner the total number of breeding territories in each potential reintroduction site was calculated.

Not all assessed parameters are equally important in determining the suitability of the sites for reintroduction, therefore the AHP was used to determine the relative importance of each parameter. The AHP makes use of pairwise comparisons to reduce the conceptual complexity of a problem (Beach *et al.* 2004). These pairwise comparisons were done on my parameters in collaboration with Sonja Krüger, an expert on the southern Africa bearded vulture population, who contributed to and evaluated the pairwise comparisons. A continuous scale with values between 1 and 9 was used within the comparison matrix to compare the relative importance of each parameter with all others, and the reciprocal of these values was used to indicate a scale to which one parameter might be less important than another (Clevenger *et al.* 2002). The resulting comparison matrix was normalized and the normalized eigenvector of this matrix was calculated

(Saaty 1980). This provided weightings for each parameter. The consistency of the judgments was tested by calculating the consistency ratio of the comparison matrix. If this ratio was above the advised 0.1 threshold, the judgments were considered inconsistent and comparisons were adjusted (Saaty 1980). For further explanation and worked examples please refer to Saaty (2008).

The calculated weightings were then applied to the score of each habitat parameter and a total score for each site was calculated by subtracting the scores of the parameters that have a negative impact (power lines, windfarms, human settlement and unsuitable habitat) from those that are positive (livestock and protected areas) for bearded vultures (Table 4). The reintroduction sites were then ranked based on these scores. A similar scoring and comparing method based on site attributes has been used in other reintroduction site identification studies (O'Toole *et al.* 2002).

2.3 Release site selection

Due to the large size of the reintroduction sites, I endeavoured to identify suitable locations for the physical release of fledglings within each reintroduction site, i.e. a release site. The location at which these birds are released could have important consequences as these birds are the most vulnerable within their first year, which is when they are released, and thus also experience the highest mortality rates at this time (Krüger 2014). In order to identify the most suitable areas for the physical release of fledglings within each potential reintroduction site I divided the sites into 10 x 10 km grid cells. I repeated the same process as described above, under reintroduction site ranking, for each 10 x 10 km grid cell and mapped the results. Food availability in the form of livestock density was, however, not considered as the livestock density data was judged to be too coarse to provide a useful contribution to this assessment. Because of the finer scale and different aim of this analysis, the importance of variables during the calculation of weightings were considered to be different than that of the reintroduction site analysis.

The management of the reintroduction, in terms of the physical release of birds and maintenance of a hacking site, would be much easier if the release site was located within a protected area. Furthermore fledglings would benefit from additional protection during the most vulnerable part of their life cycle, before they disperse from their natal area (Krüger 2014). Once birds disperse, however, they are unlikely to stay within the confines of the protected area and will thus come into contact with threats outside of the protected area and the release site. Therefore protected areas were considered more important on the release site scale than on the larger reintroduction site scale.

2.4 Population modelling

I used the stochastic population simulation program Vortex 10 (Lacy and Pollak 2014) to set up a meta-population model and model the outcomes of various management strategies for a potential reintroduced population of bearded vultures. Management strategies focussed on various ways to use fledglings produced from wild harvested eggs. The harvest of these eggs was assumed to have no effect on the current population as these would be second eggs from nests, which are obligate mortalities in the wild. It is estimated that only six nest sites would be available for harvest each year and logistically harvesting is only envisaged possible for a five year period (Krüger, S. pers. Comm.). This harvest rate was determined by considering the accessibility of nest sites, the manpower involved and the potential available funding. This would most likely result in only four fledglings available to the reintroduction project per year, due to stochastic factors such as infertile eggs or weather conditions barring harvesting (Krüger, S. pers. comm.). I used this conservative estimate of four fledglings available for release per year over a period of five years to model the effect of three different likely management strategies (Figure 2):

- Model 1 (M1): All four fledglings are released directly into a reintroduction site each year.
- Model 2 (M2): All four fledglings are retained to build up a captive population which starts releasing birds to the wild only after it reaches a threshold capacity of 23 individuals.
- Model 3 (M3): A blended model with simultaneous and equal supplementation of fledglings to both the captive population and the reintroduced population. With releases from the captive population only commencing after the captive population has built up to 23 individuals.

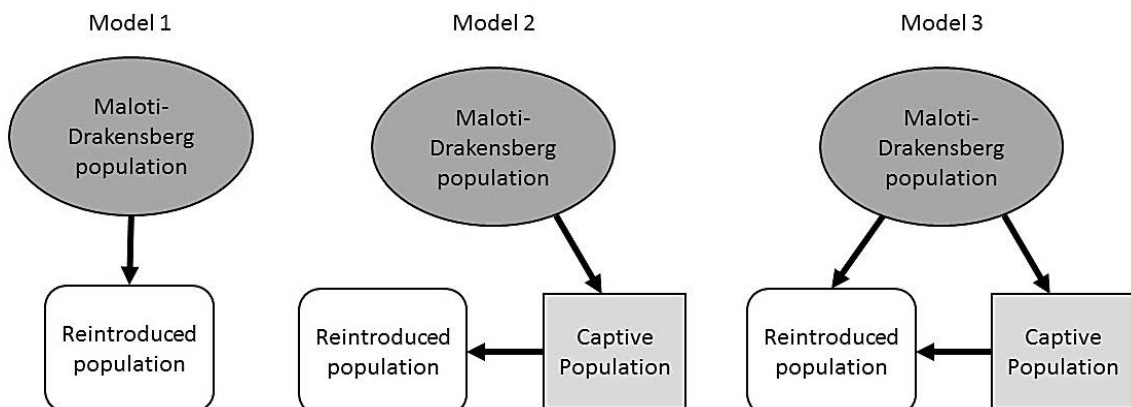


Figure 2: Conceptual representation of the various management strategies, Models 1 to 3, regarding the outcome of individuals reared from harvested eggs. Arrows represent the movement of fledglings among populations.

The demographic rates of the Maloti-Drakensberg population were acquired from Krüger (2014). These were used as baseline rates for the reintroduced population in the Vortex models (Appendix 1). Since it is uncertain what mortality rates the reintroduced population will experience, I modelled various mortality scenarios, differing from the base line by 10 percentage incremental points. These scenarios spanned between a 10% increase in mortality rates up to a 40% decrease in mortality rates. Various productivity scenarios differing by 5% increments and spanning between a 10% decrease in productivity and a 15% increase in productivity, were also modelled. Productivity however showed to have little effect on the outcome of the models and was excluded from further analysis, which corresponds to results found by Krüger (2014). Baseline productivity rates were therefore used in all scenarios in the reported results. Demographic rates used for the captive population were acquired from the captive population in European zoos between 1973 and 1993 for the Alpine reintroduction (Bustamante 1996; Appendix 1).

The aim of this reintroduction is to establish a viable reintroduced population as has been achieved by the alpine reintroduction project. Within 20 years, nine breeding pairs had been established in the Alps and further releases were deemed unnecessary (Schaub *et al.* 2009). I used a more conservative aim of establishing at least 10 pairs after 30 years. Assuming that most adults breed this would equate to at least 20 adults being present in the population. Using the age distribution of 59.6% of bearded vultures being adults and 40.4% non-adults, as calculated by Krüger (2014) through vehicle and foot surveys, I calculated that the total population size needed to be 34 individuals in order to include 20 adults. The extinction definition of my model was consequently set as having fewer than 34 individuals. The probability of extinction produced by Vortex thus provided an estimate for the probability of not reaching this aim, the probability of failure.

Each year for a period of five years, the four available fledglings from wild harvested eggs were supplemented to either the reintroduced population, the captive population or distributed equally among both, depending on the management model being run (Figure 2, Appendix 1). I assumed that the captive programme would only be able to support around 20 individuals as only one facility, the African Bird of Prey Sanctuary, Pietermaritzburg (KwaZulu-Natal), is currently invested in the reintroduction project. The project could however grow to include multiple facilities. The Vortex harvesting and supplementation function was used to simulate the transfer of individuals from the captive population to the reintroduced population. Harvesting was set to occur only when the captive population reached 23 individuals as this resulted in the captive population stabilising at around 20 individuals, the estimated full capacity of the current breeding

facility (Figure 12). At this threshold individuals in their first year were harvested from the captive population and the same number of individuals as was harvested was supplemented to the reintroduced population. This happened each time the captive population was harvested. The average time it took for the captive population to reach a harvestable size under Models 2 and 3 was calculated.

Probabilities of extinction and the mean size of extant populations across the various mortality rates were then compared between management models. The management model that scored the highest with regard to these variables was then rerun with a 50 year time frame and supplementation from the captive population stopping at 30 years. This was done to assess the long term viability of the reintroduced population.

Intervention in the form of reintroduction and captive breeding can be useful even if the reintroduced population is not self-sustaining, because it may provide time to address the causes behind the declines. To assess this possibility, the highest scoring model was run again at baseline demographic rates, along with the Maloti-Drakensberg population (Appendix 1). All individuals in the captive population were removed after harvesting and supplementation had ceased at year 30. Meta-population estimates at year 50 thus only included individuals in both the Maloti-Drakensberg and the reintroduced population. Results from this simulation were compared with a run that included only the Maloti-Drakensberg population and thus provided an estimation of the value of a reintroduction if mortality rates in the current wild population do not improve and are no better in the reintroduction site.

3. RESULTS

3.1 MaxEnt model

Model outputs from MaxEnt consistently highlighted the same areas as suitable regardless of the combination of environmental variables run (Figure 3). Elevation and presence of cliffs had the highest predictive contribution with a 45.7% and 35.1% contribution respectively (Table 2). Terrain ruggedness had a very low additive predictive contribution of 0.1%. Climate's predictive contribution was also significantly less than elevation or cliff presence, with the highest variable, mean temperature in the driest quarter, only reaching a 5% predictive contribution (Table 2).

Table 2: Analysis of variable contributions showing the percentage predictive contribution of each variable for the MaxEnt model.

Variable	Percent contribution	Permutation importance
Elevation	45.7	7.2
Cliff presence	35.1	51
BIO9: Mean Temperature of driest quarter	5	5.7
BIO3: Isothermality	2.1	3.3
BIO2 : Mean Diurnal Range	2.1	0.7
BIO12: Annual Precipitation	1.4	4.9
BIO16: Precipitation of Wettest Quarter	1.1	2.1
BIO15: Precipitation Seasonality	1.1	5.4
BIO4: Temperature Seasonality	1.1	1
BIO1: Annual Mean Temperature	1	1.5
BIO10: Mean Temperature of Warmest Quarter	1	0.6
BIO11: Mean Temperature of Coldest Quarter	0.6	0.7
BIO8: Mean Temperature of Wettest Quarter	0.5	0
BIO7: Temperature Annual Range	0.5	4.3
BIO5: Max Temperature of Warmest Month	0.4	1
BIO13: Precipitation of Wettest Month	0.4	0.8
BIO17: Precipitation of Driest Quarter	0.4	0.8
BIO6: Min Temperature of Coldest Month	0.2	5.8
Terrain ruggedness	0.1	1.1
BIO18: Precipitation of Warmest Quarter	0.1	1.4
BIO19: Precipitation of Coldest Quarter	0	0.4
BIO14: Precipitation of Driest Month	0	0.4

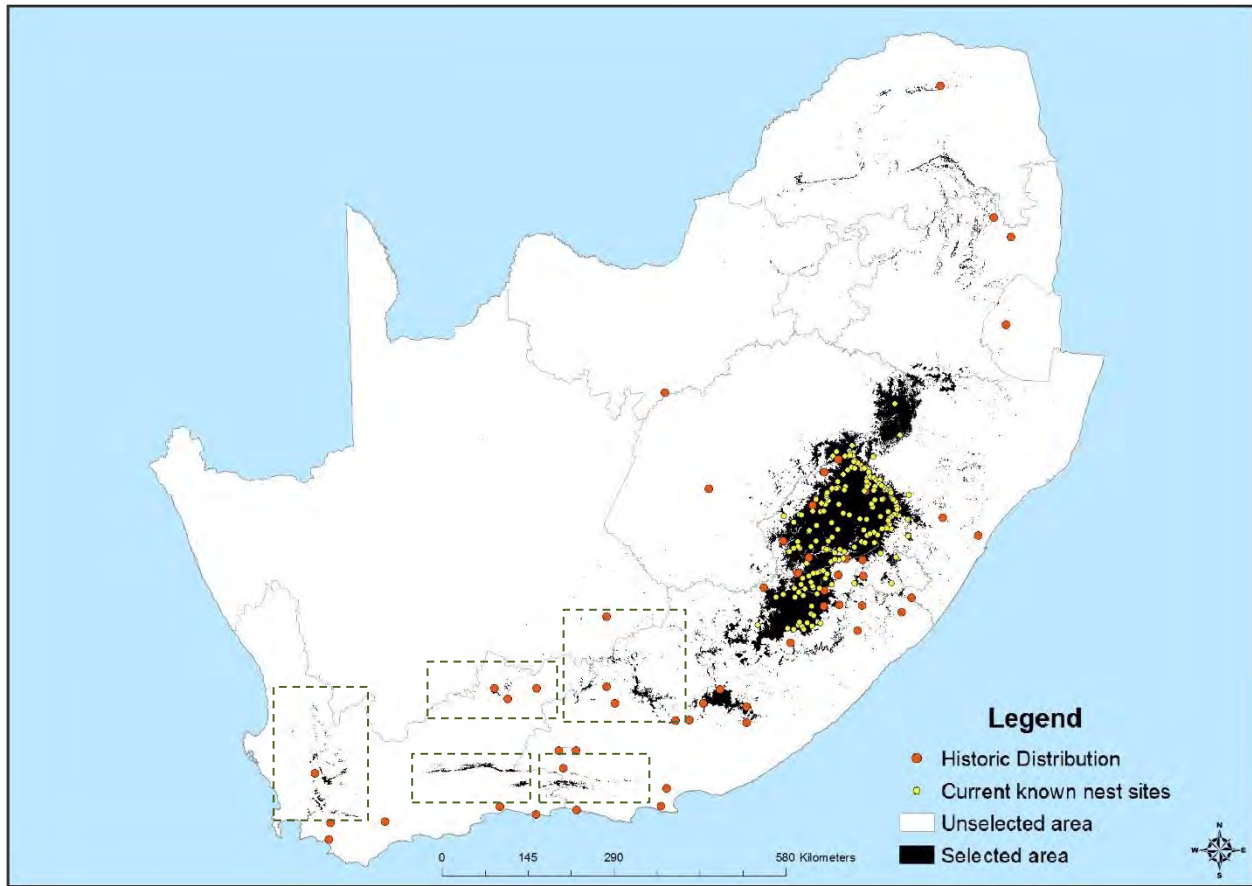


Figure 3: Results from MaxEnt model built with the environmental variables cliff presence, elevation, climate and terrain ruggedness, based on current known nest sites. Black shading indicates areas with >1% probability of bearded vulture nest site occurrence. Squared off areas indicate areas that were consistently chosen across all models run and are more than 100 km away from the current bearded vulture range and within the bearded vulture historic range.

3.2 Parameter weightings

Results from the AHP indicate that the proportion of the site comprising human settlement was considered the most important parameter for bearded vulture suitability, both for the reintroduction site and release site selection (Table 3). Power line density (Figure 4; Appendix 2) was also ranked high, second for reintroduction sites and third for release sites. Windfarms (Appendix 2) were considered less important than power lines and ranked third for reintroduction sites and fourth for release sites. Protected areas (Appendix 3) were weighted as important as protected areas for reintroduction sites and second highest in release site identification. Livestock density was only relevant for reintroduction sites and ranked fourth. Unsuitable habitat (Appendix 4) received the lowest weighting in both analyses.

The consistency ratio (CR) of both analyses was below the 0.1 threshold, indicating that pairwise comparisons were done consistently.

Table 3: Resulting weightings of each parameter calculated from pairwise comparisons, done during an Analytical Hierarchy Process (AHP) in collaboration with bearded vulture expert Dr. Sonja Krüger, and the consistency ratio (CR) of these pairwise comparisons.

	Unsuitable habitat	Livestock density	Wind farms	Protected area	Power lines	Human settlement	CR
Reintroduction Site Weightings	0.0378	0.0395	0.0937	0.0937	0.1982	0.5371	0.061
Release site Weightings (10 km)	0.0338	N/A	0.0650	0.3144	0.1267	0.4602	0.078

3.3 Reintroduction site ranking

Five potential reintroduction sites were identified based on outputs of the MaxEnt model and the locations of cliffs (Figure 3). Once the cliff sites were buffered by 10 km, the size of these sites ranged between approximately 7100 km² and 18200 km² (Table 4). The shapes of these sites varied considerably and the sites spanned three provinces: the Northern, Western and Eastern Cape (Figure 4, 5 and 6).

Site 5 was identified as the most suitable site for the bearded vulture reintroduction with a suitability score of 0.362 (Table 4). This is despite having the lowest percentage area protected at less than 5%, and the second highest percentage area of wind farms (Table 4; Appendix 2 and 3). This site is the largest site at 18175 km² and consequently also contains the largest number of potential breeding territories (which were not including in the scoring system).

Site 1 ranked second and was the only site that had no windfarms in the area (Table 4; Appendix 2). This site however contained the second highest percentage of unsuitable habitat, approximately 23%, as well as the second highest power line densities (1.5 km/km²), although this was significantly lower than the worst site (Table 4; Appendix 2 and 4). Site 1 also ranked second in livestock densities but at 55 animals/km² was almost half that of Site 5. Site 1 was about 56% smaller than Site 5 but was much better protected with approximately 37% of the area consisting of protected area (Table 4; Appendix 3).

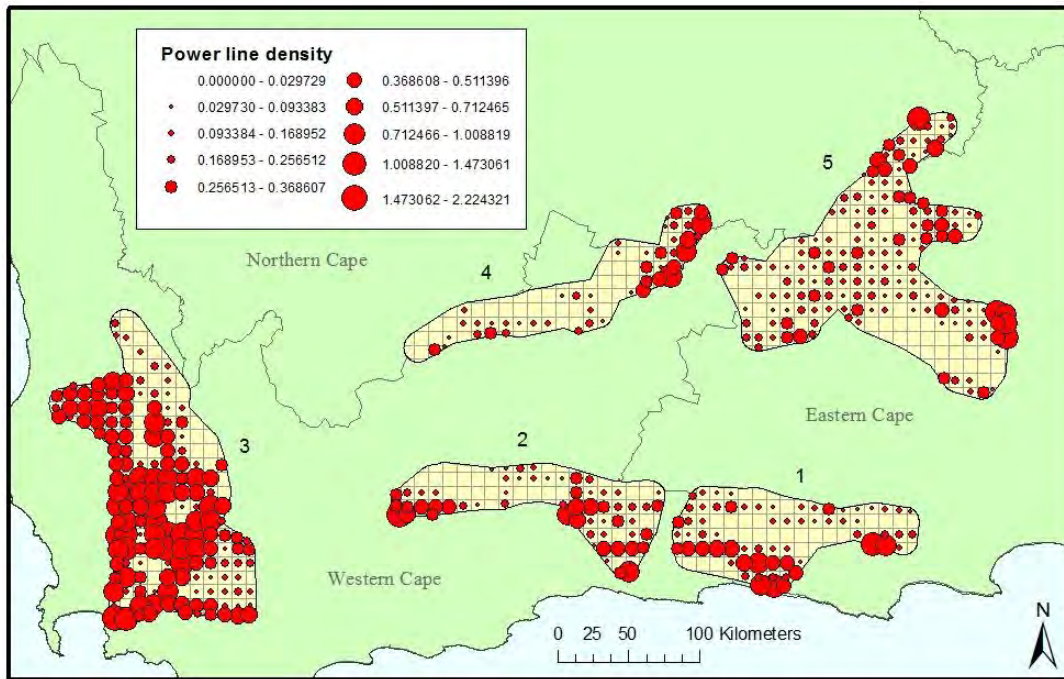


Figure 4: Power line densities (kilometres of power line/ km²) for each 10 x 10 km grid cells in each potential bearded vulture reintroduction site (1-5).

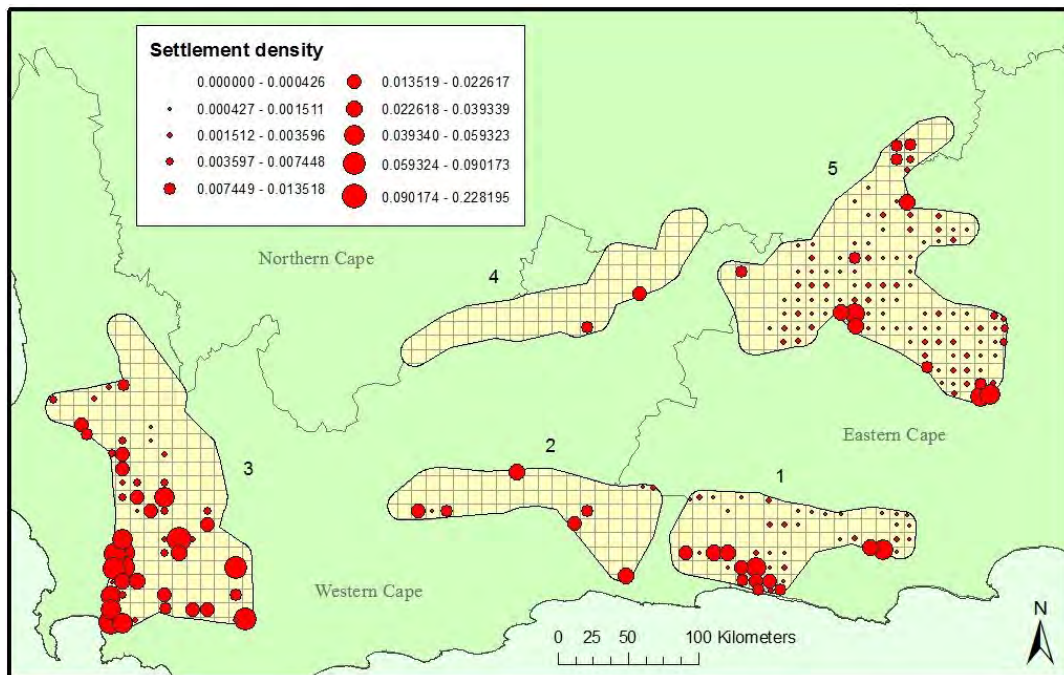


Figure 5: Human settlement density (Settlement km²/ km²) for 10 x 10 km grid cells in each potential bearded vulture reintroduction site (1-5).

Site 2 was ranked in third place, and had a high percentage of protected area (Table 4; Appendix 3). It scored the second lowest in almost all other parameters including size, number of potential

breeding territories, livestock density and the percentage area consisting of wind farms and human settlements.

Site 4 had the lowest proportion of area containing human settlements or unsuitable habitat and the lowest power line density, but ranked fourth (Table 4; Appendix 2 and 4). This was due to a large concentration of windfarm developments in the northern corner of this site, leading to the highest percentage area of windfarm developments (Table 4; Appendix 2). This site also had the second lowest percentage of protected area (Table 4; Appendix 3). This was the smallest site with only 18 estimated potential breeding territories.

Site 3 had the lowest score and was identified as the worst site for a bearded vulture reintroduction (Table 4). Although this was the largest site and had the highest percentage area that was protected it also had the highest power line densities, percentage area human settlement, percentage area unsuitable habitat and the lowest livestock density (Table 4; Appendix 2, 3 and 4).

Table 4: Attributes of each identified potential reintroduction site (Windfarms (WF), Protected Area (PA), Power Lines (PL), Human Settlement (HS), Unsuitable Habitat (UH), Potential Breeding Territories (PBT), Livestock Density (LD), as well as the calculated suitability score and resulting rank of each site.

Site ID	Site Size (km ²)	WF (%)	PA (%)	PL (km/km ²)	HS (%)	UH (%)	LD (#/km ²)	SCORE	RANK	PBT
1	8077.84	0	37.41	1.50	0.31	22.69	54.79	0.213	2	27
2	7278.69	0.15	32.47	1.47	0.10	17.25	29.68	0.152	3	25
3	16523.52	0.41	40.80	3.94	0.80	30.60	4.92	-0.191	5	55
4	7083.98	7.39	11.47	1.10	0.02	3.63	32.54	0.093	4	18
5	18174.73	0.97	4.48	1.31	0.2	7.20	99.71	0.362	1	62

3.4 Release site selection

Suitability scores for 10 x 10 km cells varied between -0.041 and 0.002 (Figure 6). Due to high weighting of protected areas there was a close association between high scoring cells and the presence of protected areas (Figure 6; Appendix 3). Within Site 5 only three areas had high suitability scores (Figure 6), each corresponding to a protected area: the Agter Sneeu Berg Nature

Reserve, the Mountain Zebra National Park and the Compasberg Protected Environment (Appendix 3).

Site 1 had more wide spread highly suitable release sites, corresponding to its approximately 33% higher proportion area protected (Table 4; Appendix 3). This area consists of several Cape Floral Region Protected Areas such as the Garden Route and Baviaanskloof.

Sharp changes in the suitability scores of cells in Site 3 were observed (Figure 6). All other sites exhibited much more gradual changes in release site suitability. Site 3 also contained some of the least suitable sites for release (Figure 6).

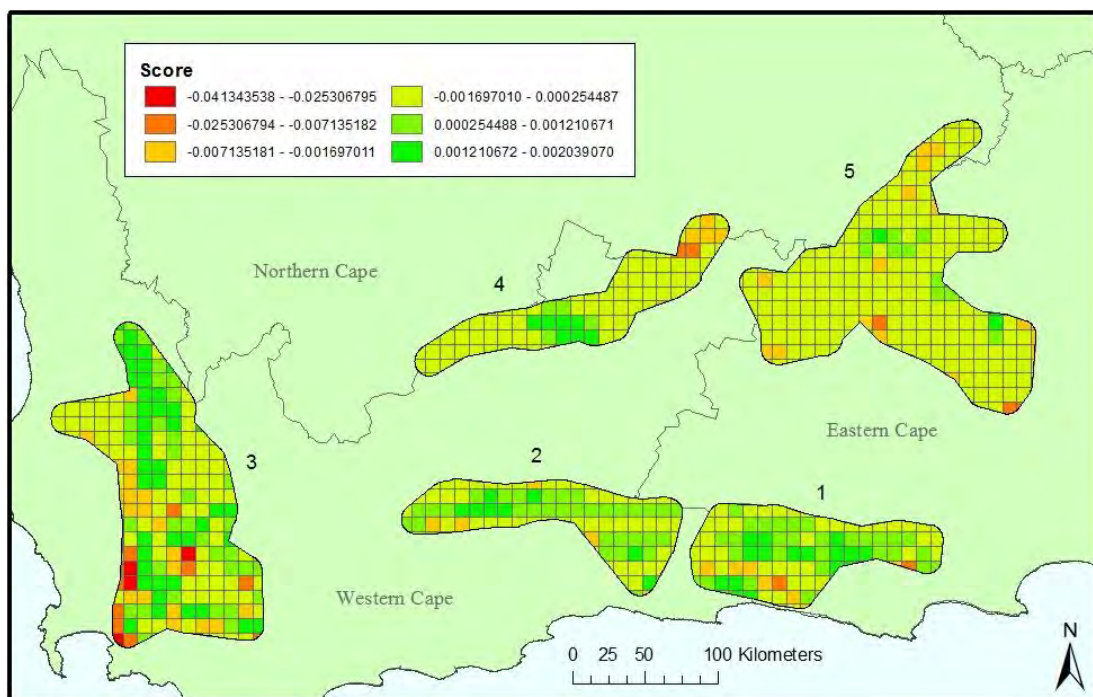


Figure 6: Suitability scores calculated for 10 x 10 km grid cells across all identified potential bearded vulture reintroduction sites (1-5). Greens are the most suitable and reds are the least suitable.

3.5 Population modelling

Model 1 (direct reintroduction) resulted in very high probabilities of extinction (<34 individuals) across all mortality rates for the reintroduced population, ranging from 95.7 to 78.3%, indicating that this management strategy is unlikely to succeed (Figure 7). In total only 20 individuals were supplemented to the reintroduced population over the 30 year simulation period under this management model, resulting in an average of 0.66 ± 1.52 individuals supplemented per year. In the few populations that did survive, the average extant population size of the reintroduced population ranged between 86.79 ± 57.06 and 132.8 ± 75.98 (Figure 8).

Model 2 (captive breeding followed by reintroduction) had the lowest probability of failure ranging between 49.8 and 25.5% (Figure 7). This model resulted in the most fledglings being supplemented on average, 4 ± 4.86 per year, bringing the average total of supplemented birds over the 30 year simulation to 112 ± 44.42 . Model 2 also resulted in the highest extant population sizes ranging between 91.85 ± 68.31 and 161.4 ± 107.73 (Figure 8).

Model 3's (simultaneous captive breeding and reintroduction) probability of failure was similar to Model 2 and ranged between 53.8 and 27.6% (Figure 7). This model resulted in an average of 2.95 ± 4.12 fledglings supplemented per year and an average total of 88.55 ± 36.47 over the entire simulation period. The average extant population size ranged between 78.97 ± 45.83 and 111.54 ± 68.00 (Figure 8).

The results indicate that Model 2 was the best management strategy (however, other considerations may actually make Model 3 management strategy preferable – see discussion). If the mortality rate is 40% lower at the reintroduction site and the Maloti-Drakensberg population continues to decline there is a possibility that the reintroduced population may grow to be larger than the Maloti-Drakensberg population within 30 years (Figure 9). This is only true under the Model 2 management scenario.

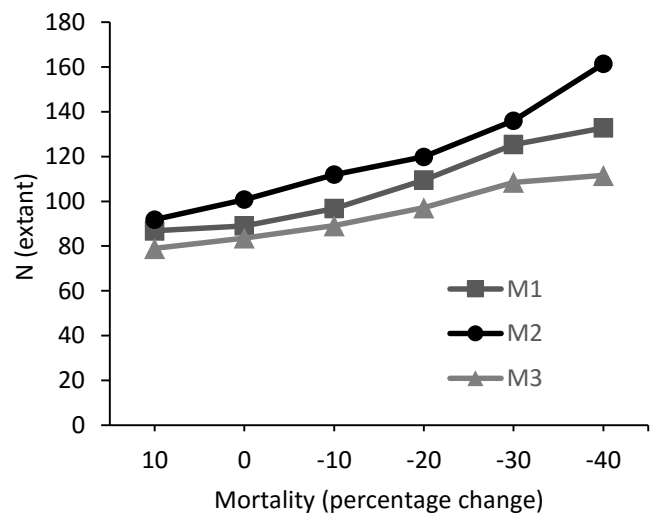
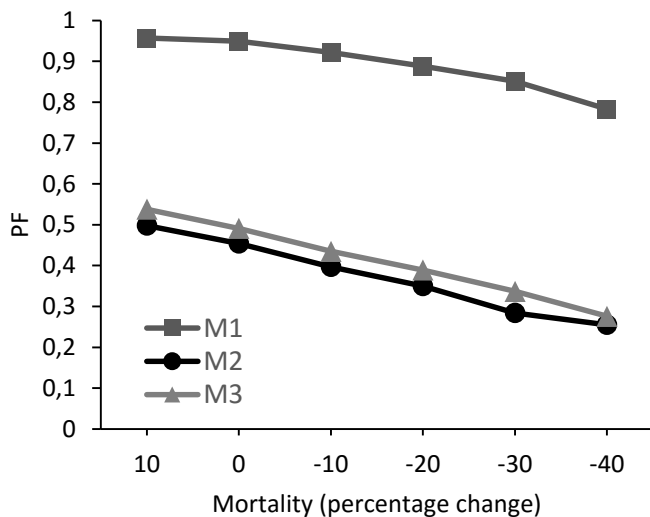


Figure 7: Probability of failure (<34 individuals, PF) within 30 years for a reintroduced bearded vulture population in relation to 10 percentage point incremental changes in the baseline mortality rates (0) for three different management models (M1-M3).

Figure 8: Mean population size (N(extant)) after 30 years for a reintroduced bearded vulture population in relation to 10 percentage point incremental changes in the baseline mortality (0) for three different management models (M1-M3).

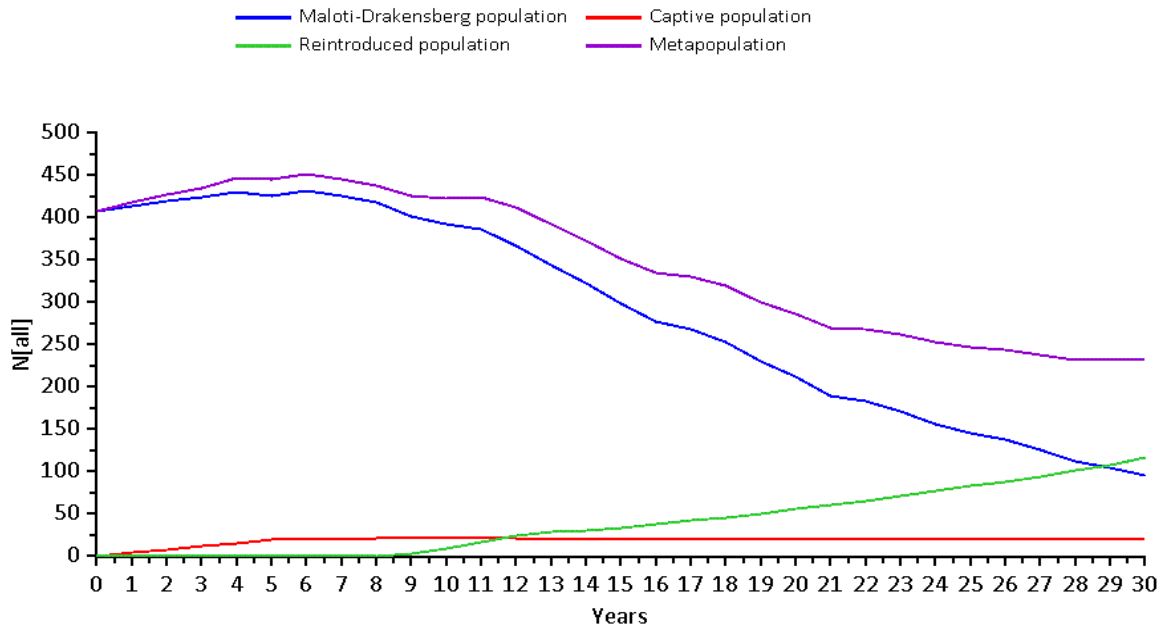


Figure 9: Mean population size (N(all)) of each population per year for all iterations of a bearded vulture reintroduction project simulation with the reintroduced population run at a 40% reduction of baseline mortality rates with the Model 2 management scenario (captive breeding followed by a reintroduction) implemented.

Simulations of Model 2 over 50 years resulted in much higher probabilities of extinction when compared with the 30 year simulations, ranging between 86.9% for the highest mortality rate scenario and 50.5% for the lowest (Figure 10). The average extant population size ranged between 156.95 and 250.81 (Figure 10). The reintroduced population only had a 19.5% chance of having more than 34 individuals in the population 20 years after the supplementation has ceased if mortality rates are similar to those of the current range (Figure 10). This indicates that the population would be highly dependent on the continual flow of birds from the captive population and would unlikely be self-sustaining if left alone. If mortality rates can, however, be reduced by 40% then this probability increases to 49.5%, indicating that under such circumstances the establishment of a viable reintroduced population is more probable.

My models suggest that the southern African bearded vulture population, which currently consists of only the Maloti-Drakensberg population, has a 92.5% probability of extinction (one sex remains) over the next 50 years. A reintroduction initiative will lower the probability of this extinction to 62.6%, even if similar mortalities are experienced at the reintroduction site (Figure 11).

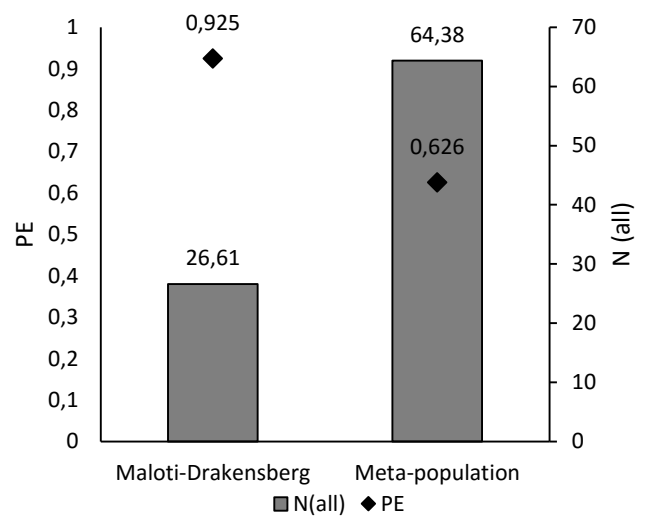
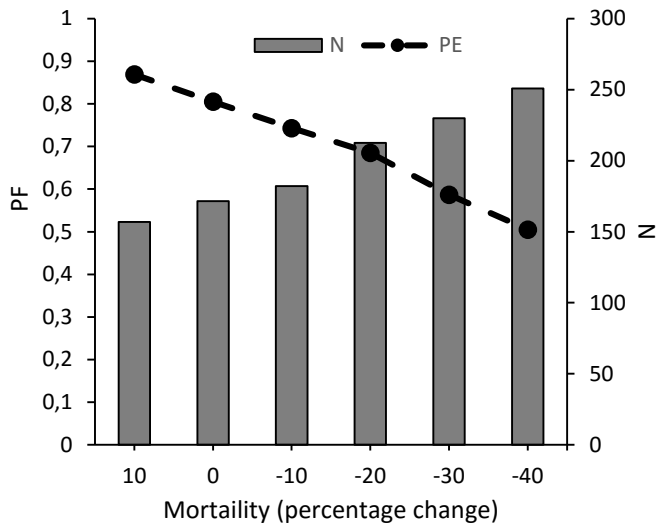


Figure 10: Probability of failure (<34 individuals, PF) and extant population size (N) for a reintroduced bearded vulture population after 50 years, across mortality rates varied by 10 percentage point increments from baseline demographic parameters for the Model 2 management scenario.

Figure 11: Probability of extinction (PE), defined as one sex remaining, and average population size (N(all)) of all iterations for the Maloti-Drakensberg population and the Meta-population using baseline demographic rates and the Model 2 management scenario.

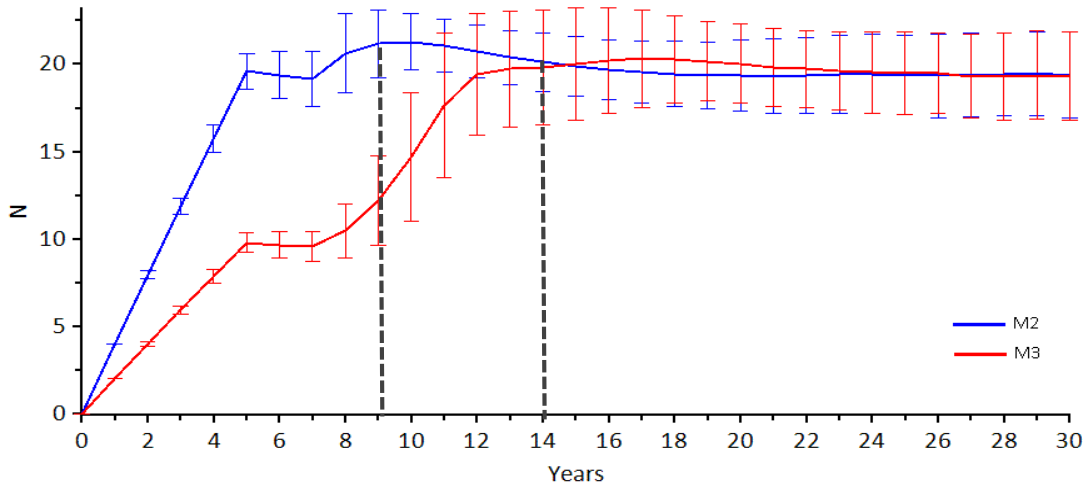


Figure 12: Mean population size for a captive population of bearded vultures modelled for a captive population prioritizing management strategy, Model 2 (M2) and a simultaneous captive breeding and reintroduction management model, Model 3 (M3). The amount of years taken for the captive population to reach the predetermined harvestable size of 23 individuals is indicated by the dashed line.

Model 2 took the shortest time, 9.04 ± 0.91 years to build up the captive population to the harvesting threshold (Figure 12). Model 3 took an average time of 13.85 ± 3.18 years before the first harvest occurred (Figure 12).

4. DISCUSSION

Due to the current decline and probable extinction of the southern African bearded vulture population, coupled with the emerging threat of windfarms developments within their range (Reid *et al.* 2015), the reintroduction of this species within its South African historic range is being considered. This population is to function as an insurance population against such extinction events as have occurred across Europe (Hiraldo 1979; Mingozi and Estève 1997). In this study I identified the most suitable candidate sites for such a reintroduction as well as providing some insight into which release strategy would be most efficient in establishing a reintroduced population. This provides critical information to the team that are implementing the Biodiversity Management Plan for bearded vultures (South Africa 2014), thereby enabling this team to more effectively conserve this species.

4.1 Reintroduction site selection

Results from this study indicate that both the two top ranked sites for the bearded vulture reintroduction are located mostly within the Eastern Cape. The top ranking site, Site 5, was the largest and had an estimated 62 potential breeding territories (Table 4). Consequently, if a viable population is established here then it would contribute a significant proportion of the bearded vultures found in southern Africa. A larger population would also be less vulnerable to stochastic environmental factors (Reed 2005).

Potential breeding territory estimates are however conservative as bearded vultures have been known to have overlapping territories (Krüger 2014). On the other hand, although all cliff sites were identified, not all cliff sites would necessarily have nesting structures (pot holes or large ledges with overhangs). Arguably however, these could be artificially provided as is often done in the conservation of raptor species which are nest-site limited (Katzner *et al.* 2005, Smallwood and Collopy 2009; Liébana *et al.* 2013). In the case of cliff nesting raptors the logistics would, however, be considerably more challenging. Nest site availability is an aspect that was not assessed in this study and a higher cliff prevalence could provide another parameter for habitat suitability as this might correlate with a higher number of potential nest sites. The presence of

nesting structures (e.g. potholes or suitable ledges with an overhang) at the identified sites should thus be verified on the ground before site selection. Visual comparison of the two top sites indicates that Site 1 has a higher density of cliffs. In Site 5 cliffs are more isolated and spread sparsely across the entire landscape. The site that was ranked the lowest, Site 3, seemed to have the highest prevalence of cliffs.

Site 3 ironically also had the highest percentage of protected area, as well as the highest percentage of unsuitable habitat, human settlements and power line density. The sharp changes in release site suitability in Site 3 illustrates why this site is the worst site for reintroduction as suitable areas such as protected areas occur adjacent to very unsuitable areas (Figure 6). As there is no way of keeping bearded vultures within the confines of protected areas they are likely to be exposed to increased mortality rates that they may well experience at these sites.

This situation highlights the major challenge faced in the conservation of far ranging avian species as no protected areas are large enough to ensure that individuals do not come into contact with threats outside these areas. Fortress conservation (Brockington 2002; Büscher 2016) is thus insufficient to protect such species and a large focus should be placed on mitigating the threats in surrounding areas. Poisoning, as discussed, is one of the major threats to bearded vultures and is commonly practiced by farmers wishing to reduce livestock losses through predator control (Ogada *et al.* 2012a). Livestock densities in Site 5 were significantly higher compared to any other site, indicating that intensive farming activities occur in this area. Community engagement and awareness campaigns as well as strong enforcement of environmental laws is thus imperative for the success of the reintroduction as well as for addressing the Maloti-Drakensberg population decline (Krüger *et al.* 2006).

Although Site 5 was identified as the most suitable site for reintroduction the results from this study should be carefully scrutinized by authorities managing the reintroduction based on their capacity to address the various parameters assessed. Site 4, for example, seemed highly suitable except for a high concentration of windfarms and power lines in the northern tip of the site (Appendix 3; Figure 4). If vultures could be persuaded, through supplementary feeding for example, to stay out of this area, then the suitability of this site will greatly improve. Being the smallest site however means that reducing the effective size of the site could leave the site too small to be a viable candidate. The effects of power lines are also most likely under estimated because transmission and distribution networks are continuously expanding (Krüger 2014). This may account for the increase in estimated mortalities attributed to collisions in South Africa, from an estimated 10% (Brown 1991) to 21% (Krüger 2014). In comparison 18% of mortalities are

attributed to collisions in Europe (Margalida *et al.* 2008). Many collision mortalities are also never recorded since carcasses are easily missed due to inaccessible terrain, landing a distance away from power lines and being scavenged by other animals (Schutgens *et al.* 2014; Krüger 2014).

Nonetheless, the way in which landscape features are distributed throughout the landscape can influence the severity of their ecological effect (Kie *et al.* 2002; Peters *et al.* 2012). Site 5 does not have many human settlements when compared to the other sites (Table 4), but the settlements that are present are spread relatively evenly across the entire site (Figure 5). The widespread nature of these settlements means that poisoning events may be more widespread compared to other sites, such as Sites 2 and 4, where settlements are concentrated into certain patches, leaving large areas free of settlement (Figure 5). If settlements are concentrated in a certain area they could more easily be avoided by bearded vultures.

The threat of poisoning and the reintroduction itself could be much better managed within protected areas and they were consequently considered a very important variable in the identification of release sites within reintroduction sites. Within Site 5, three areas were highlighted as potential release sites (Figure 6). Each of these areas corresponds to a protected area (Appendix 3). Two of these protected areas, the Agter Sneeuberg Private Nature Reserve and the Mountain Zebra National Park, occur on the edges of the reintroduction site. Compasberg Protected Environment, the most northern high scoring site, is better situated in the centre of the reintroduction site (Figure 6). This situation would be more beneficial for dispersing bearded vultures. This site specifically should thus be assessed in more detail to ground truth the suitability of this site for reintroduction. The area is part of the Sneeuberge mountain range and the Compasberg mountain peak is the highest peak west of the Maloti-Drakensberg range at 2502 m above sea level.

4.2 Dispersal between populations

Only sites that were at least 100 km away from the current bearded vulture range were considered as the aim of the project was to establish a separate insurance population. As Site 5 is the closest site to the current range, reintroduction here has the highest probability of leading to dispersal to and from the Maloti-Drakensberg population. Bearded vultures have been known to range up to 870 km (Margalida *et al.* 2013).

Dispersal between the populations could be both beneficial and disadvantageous and authorities should carefully evaluate if this situation is preferred. Inbreeding and the genetic

integrity is, for instance, always a concern in small populations and natural movement of individuals between the reintroduced and Maloti-Drakensberg population would lead to increased gene flow between these populations.

Large suitable patches between Site 5 and the current Maloti-Drakensberg range were identified by the MaxEnt model (Figure 3). If Site 5 were to be chosen for the reintroduction then the abandoned territories between these sites (Krüger *et al.* 2015) may be re-occupied, a scenario which is more likely if individuals move between these populations.

On the other hand the current bearded vulture range may function as a sink for dispersing sub-adults and thus compromise the viability of the reintroduced population

4.3 Release strategies

My simulations showed that releasing birds into the wild without the subsequent supporting releases from a captive population has a very high probability of failing and Model 1 is thus not a viable management strategy (Figure 7). Making use of a captive breeding programme to supplement the reintroduced population, such as in Models 2 and 3, greatly increases the probability of success for the reintroduction. Probabilities of success were larger than 50% under most mortality scenarios (Figure 7). Although Models 2 and 3 had similar probabilities of extinction (<34 individuals) across all mortality rates, those of Model 2 were consistently slightly lower. Model 2 also had higher resulting population sizes than Model 3 (Figure 8). Model 2 was the only model in which the reintroduced population reached a larger population size than the current Maloti-Drakensberg population within 30 years (Figure 9). This situation would only potentially arise in Sites 5 or 3 as all other candidate reintroduction sites are too small to support such a large population.

My results consequently indicate that prioritizing the building of a captive population with regard to the use of fledglings coming from harvested eggs is the most efficient management strategy in terms of producing the strongest reintroduced population. The efficiency of this strategy is due to the high amount of individuals being produced from the captive population for supplementation, equating to approximately four per year over the entire 30 year period.

The benefit of management Model 3 is that releases start earlier, in the first year of the reintroduction project, and more gradually than in Model 2. These early releases of smaller numbers of birds can be used to test the suitability of the reintroduction sites. If high mortalities are incurred then the reintroduction site can be changed more readily compared to Model 2 where

larger groups of birds are released only once the captive population has reached full capacity. Alternatively, such problems can be highlighted early and mitigation measures (e.g. education or enforcement) can be implemented. If large numbers of birds were released into an inadequate site then it would be a significant waste in resources as each released individual incurs a large financial cost (Frey 1998). Evidence, however, shows that the productivity in breeding facilities can differ significantly and deciding which management model to implement will also be a function of the performance of the South African breeding programme (data from the Bearded Vulture European Endangered Species Programme (EEP)). Additionally, other evidence shows that breeding success and survival of wild fledged birds can be much higher than that of reintroduced birds (Evans *et al.* 2009). This therefore, also favours the strategy used in Model 3 since wild fledged birds will start appearing and contributing early to the reintroduced population with this approach.

Determining that Model 1 is not feasible, illustrates how modelling can be used to test managing strategies without incurring the cost of implementation. In the 1970s, for example, there was an attempt to reintroduce bearded vultures into the French Alps using birds from Afghanistan. Because of the irregular supply of birds, high mortality rates and the demonstrated success of captive breeding facilities, the reintroduction was abandoned in favour of a captive breeding project (Walter 1979).

High mortality rates, such as those experienced in the Maloti-Drakensberg population, poses a major threat to the success of the proposed reintroduction. My results nonetheless indicate that even if the same levels of mortality are experienced at the reintroduction site, a reintroduction project will still be a valuable initiative. This is regardless of the high probability that the reintroduced population would decrease to fewer than 34 individuals 20 years after supplementation has ceased (Figure 10). Such an initiative would lead to an approximately 30% decrease in the probability of extinction of the species from southern Africa in the next 50 years and a 242% larger population (Figure 11). Building up a captive population may also be considered regardless of the commencement of a reintroduction. When a species faces extinction artificial conservation actions such as captive breeding or supplementary feeding, to sustain the population until environmental conditions return to a favourable state for a self-sustaining population, are justifiable actions (Anderson *et al.* 2015).

If the reintroduced population is to be viable after a 50 year period then the mortality rates at the reintroduced site should be at least 40% lower than currently experienced by the Maloti-Drakensberg population. This provides a probability of success for the project of about 50% (Figure 10). Supplementation from a captive population buys at least 30 years to achieve this

mortality reduction, as shown by the relatively low probabilities of extinction in simulations where a captive breeding model was used (Figure 7).

These results are similar to those found for the Maloti-Drakensberg population by Krüger (2014), who found that a positive population growth rate was only achieved if mortality could be decreased by 40%, productivity increased by 15% and the population was supplemented by four individuals each year. My results have shown that a captive population would be able to supplement the required four individuals each year. The required decrease in mortality and increase in productivity may potentially be achieved simply through the release of birds at a site with lower anthropogenic pressures. This highlights the importance of identifying the most suitable site for the reintroduction. The next step, to provide some indication of potential mortality rates at reintroduction sites, would be to ground truth parameters such as the prevalence of poisoning, or at least the attitudes of the different stakeholders.

4.4 The IUCN assumptions:

4.4.1 Suitability of the species for reintroduction

Being obligate scavengers that feed mostly on bone, bearded vultures are prime candidates for reintroduction as they are unlikely to compete with, nor predate on other species. Factors which are often of concern in reintroductions as they have strong ecological effects and have the potential to extirpate other species or restructure the entire ecosystem (Krefting 1969; Mittelbach *et al.* 1995; Berger and Gese 2007). The fact that the bearded vulture has already been successfully released into the European Alps also lends credence to the suitability of the species for reintroduction. Due to this reintroduction and those currently underway in Spain and France (Frey and Llopis 2014), the South African reintroduction will have a wealth of knowledge acquired over 30 years of reintroduction efforts to draw on.

4.4.2 Historical evidence of occurrence and anthropogenic extinction

It is a requirement for a reintroduction, by definition, that the species being reintroduced was once a member of the local fauna and flora (O'Toole *et al.* 2002), the IUCN has, however, recently made provisions for the introduction of species outside their historic range if climate change plays a significant role in the availability of suitable habitat (IUCN 2013). Recent research, however, suggests that climate change has not played a major role in the current range loss of bearded vultures in southern Africa (Krüger *et al.* 2015). There are multiple records indicating that bearded

vultures once occurred in mountainous areas between the Western Cape and its current distribution (Boshoff *et al.* 1978; Boshoff *et al.* 1983; Brooke 1984). All the candidate reintroduction sites are thus within the historic range (Figure 1 and 3).

Bearded vultures had already lost 38% of their former breeding range by 1991 (Brown 1991). Large raptors have historically been persecuted due to the perception that these animals predate livestock. Brown (1991) showed that some farmers still held this faulty perception in the 1980s, partly due to misinterpretation of the common name of the species (Lammergeier). It is thus highly likely that direct persecution played a part in the extirpation of bearded vultures. It is assumed that this perception has changed in the past few decades following a lot of awareness and education conducted by Brown and the change in name of the species. Hiltunen (2008) does not mention persecution in her thesis after interviewing lots of farmers, and direct persecution was not one of the mortality factors mentioned by Krüger (2014). However, South Africa has a long history of using poison to control predator numbers and as this is currently the main cause of mortalities it is very likely that this played a significant role in its extirpation as well. Natural recolonization is unlikely considering the declining population, shrinking range, and the fact that it has not happened to date.

4.4.3 Rectification of the causes of extinction:

According to the IUCN (2013) guidelines attempting a reintroduction only makes sense if the original causes of extinction have been addressed. Steps were taken in this study to identify areas where the major causes of mortality to bearded vultures, poisoning and power line collisions, are expected to be least severe. As the reintroduction site would be smaller than the current range, and not cross political boundaries, the management of the population and the enforcement of conservation regulations might be more effective. Public awareness campaigns, the novelty of the species in the area and the dedicated monitoring of released individuals (e.g. GPS tracking of individuals) should also help discourage unlawful behaviour as perpetrators would more easily be discovered.

However, before reintroductions begin, there should be further investigation of reintroduction sites. Arguably the most important action would be to acquire some measure of the prevalence of the use of poison in the area. As illegal activities such as poisoning, are socially sensitive it can be hard to acquire estimates that mirror reality. Techniques have, however, been developed that produce useful estimates of such behaviours (St John *et al.* 2012). People's

estimates on the proportion of their peers involved in sensitive activities, can for example, be useful indicators of involvement in illicit behaviours, as people tend to assume that others behave like themselves (St John *et al.* 2012). I suggest that such techniques are implemented to survey potential reintroduction sites and provide both an estimate of local land user's attitudes towards the reintroduced species as well as an estimate of the potential level of poisoning that can be expected in the area. This could also usefully be compared with stakeholders within the current range.

4.4.4 Effects on donor population:

During ex situ conservation efforts, the effects on the donor population and animal welfare are always of concern. Margalida *et al.* (2015), for example, showed that the removal of clutches, chicks or fledglings from the Pyrenean bearded vulture population led to a decline of 77% in all 57 analysed scenarios. They however concluded that given the fact that this species usually lays two eggs and commits subicide, second eggs could be harvested with minimal effect on the wild population. Two eggs have already been successfully harvested in this way from the Maloti-Drakensberg population and both eggs produced fledglings, which are currently in captivity together with a previously confiscated adult (S. Kruger, pers comm.).

There is also a risk that continuous disturbance at nest sites to remove the second eggs may lead to them being abandoned, but annual disturbances are hoped to be too infrequent to cause this. The planned reintroduction would thus in all likelihood have little effect on the Maloti-Drakensberg population.

4.5 Development of good practice

The success of the project will depend on the attitudes of local communities towards bearded vultures. Wide scale consultation with potential stakeholders is therefore important prior to the start of releases (O'Rourke 2014). This is especially important in areas where a species has been extinct in an area for a long time, such as in this case, and local communities have lost any connection to the species and might harbour negative attitudes to the reintroduction (IUCN 2013). On the other hand, old prejudices and superstitions that led to the persecution of the species may also have died out.

Stakeholders or interested and affected parties consist of rural subsistence farmers, commercial farmers, game farmers, tourism enterprises, the general public, conservation non-governmental organisations (NGOs) and environmental authorities. Vulture awareness campaigns should be implemented among these groups prior to the reintroduction to hopefully ensure positive attitudes through highlighting the ecosystem services that these birds provide, the effects of poisoning on the population and the potential monetary gain from ecotourism and cost-effective organic waste disposal. Such initiatives should hopefully reduce the mortality rates that reintroduced birds may experience.

Local communities currently foster a feeling of ownership over birds within their area and there has already been some discontent from villages about people accessing bearded vulture nests sites without chieftain approval (Krüger, S. pers. comm.). Such situations are however thought to be easily avoidable if the correct procedures are followed. It is therefore important to incorporate a representative from the local community into the reintroduction initiative who knows local customs and is already trusted by the community.

4.6 Measuring success:

From the alpine reintroduction project it is clear that it would take many years before the success of the project can be judged. The first successful breeding attempt in the alpine population only occurred 11 years after releases had started (Schaub *et al.* 2009). More immediate performance indicators such as the mortality rates of the released young birds should, however, give some indication of the early success of the project. The ultimate measure of success of this project will be the establishment of 10 breeding pairs within a 30 year time frame and the persistence of this population into the future.

4.7 Limitations and recommendations for further studies

While this study has brought insight into reintroduction site selection and management strategies, it differs from a true feasibility analysis. Various data limitations were encountered. To start with measurements of reintroduction site conditions could not be compared with those in the existing Maloti-Drakensberg population as I did not have access to relevant data from Lesotho (in which the core of the bearded vulture range resides). Consequently it could not be shown whether conditions at reintroduction sites would be similar or better.

Geology was also not considered because of a lack of sufficient data. Geology can play a very important role in the distribution of bearded vultures. Hirzel *et al.* (2004) found that geology was the number one driver of habitat selection by bearded vultures during the settling phase followed by ibex, *Capra ibex*, biomass. This was proposed to be because of the higher suitability of limestone for producing thermals as well as scree slopes with appropriate rock exposure for ossuaries, which can be very important during the breeding season for the preparation of food for chicks (Hirzel *et al.* 2004). Some geology types may also be more prone to form nesting structures such as potholes, small caves or ledges under overhangs on cliffs which are required by bearded vultures to breed (Hiraldo *et al.* 1979).

My assessment of food availability was based on coarse data of livestock numbers per province. Information on wild ungulate numbers was unavailable and my measure of food availability was consequently incomplete. Sites with a high proportion of protected area especially could have higher food availability than anticipated. Food availability was however not considered of major importance as supplementary feeding sites may provide suitable food availability as well as economic benefits to local livestock farmers. A thorough assessment of food availability before reintroduction is nonetheless advisable.

Cost analyses were also not done and there may be variance in the costs incurred at each site, due to staff availability for example. These cost consideration were also not applied to the different management scenarios. These considerations are also important in terms of future monitoring, as required by the IUCN.

4.8 Conclusion

This study has identified five candidate areas for a bearded vulture reintroduction and provided a qualitative assessment of each. Further assessment is however required to validate assumptions made in this study and broaden the scope of the parameters analysed. Specifically, an assessment of stakeholder attitudes and the likely prevalence of poisoning is needed. In terms of release strategies it is clear that a captive breeding programme is imperative for the success of the reintroduction and is a prudent measure considering the current decline of the Maloti-Drakensberg population.

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APPENDICES

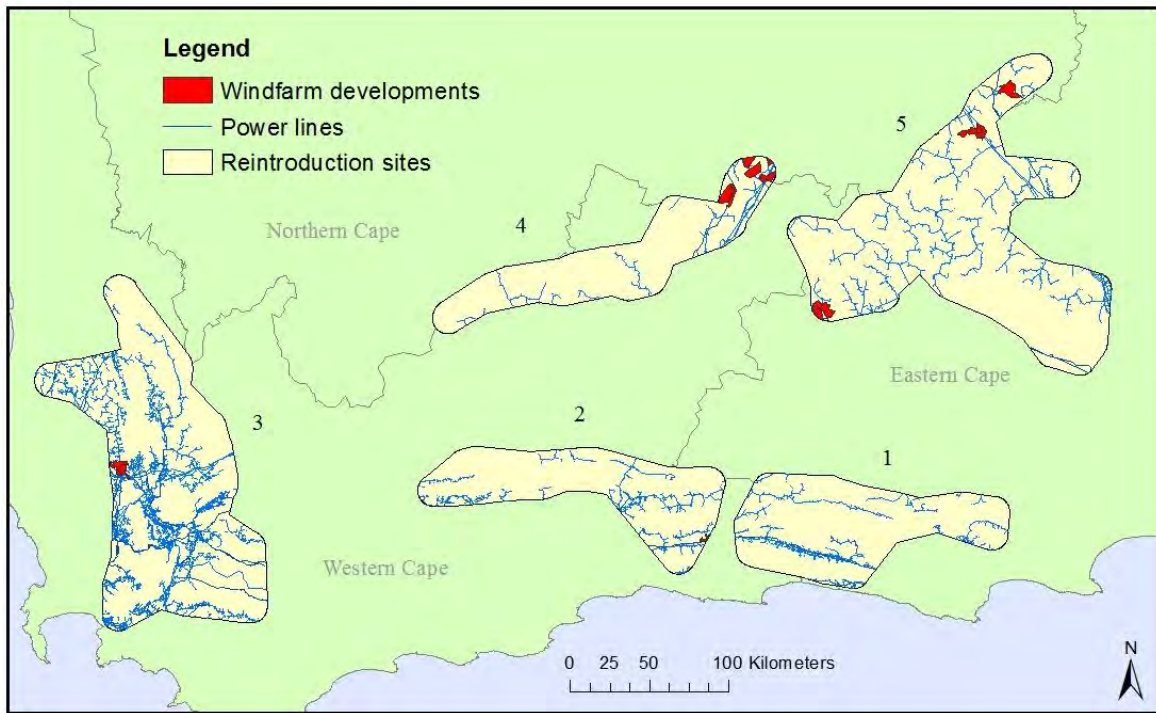
Appendix 1: Baseline input parameters for Vortex models.

Parameter	Maloti-Drakensberg population	Captive population	Reintroduced population
Number of iterations	1000	1000	1000
Number of years	30	30	30
Extinction Definition	One sex remains	N<34	N<34
Initial population size (N)	408	0	0
Carrying capacity (K)	1000	100	400
Dispersal	No	No	No
Age Distribution	Stable	Stable	Stable
Inbreeding Depression	No	No	No
Density dependent reproduction	No	No	No
Ev concordance of reproduction and survival	Yes	Yes	Yes
Mating System	Long-term monogamy	Long-term monogamy	Long-term monogamy
Age of first breeding	7	7	7
Maximum age of reproduction	32	32	32
Maximum number of broods per year	4	4	4
Maximum progeny per brood	1	1	1
Sex Ratio at birth	50%	50%	50%
Percent adult females breeding	72% ± 20%	100% ± 5%	72% ± 20%
Distribution of broods: 0 broods	24%	28.14%	24%
1 broods	76%	21.82%	76%
2 broods	0%	46.16%	0%
3 broods	0%	3.48%	0%
4 broods	0%	0.48%	0%
Number of offspring per female per brood	1	1	1

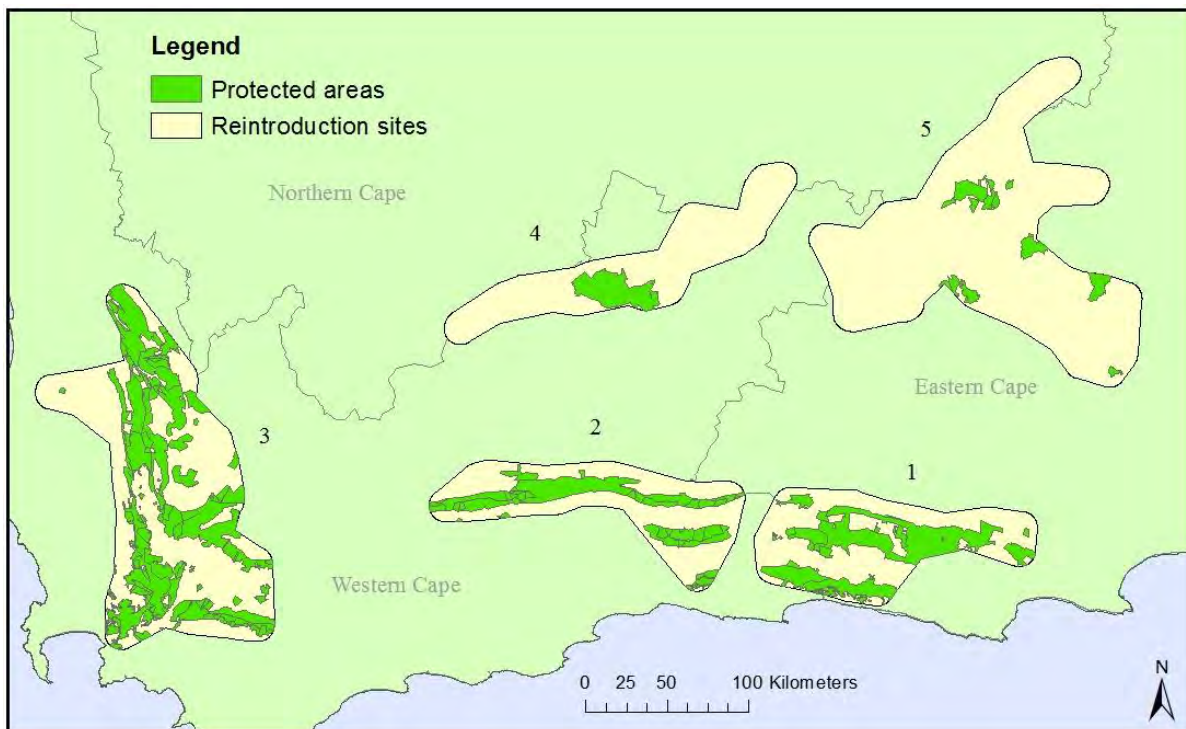
Percentage adult males in breeding pool	98%	100%	98%
Mortality rates: year 0-1	25.8 ± 38.18	7.57 ± 25.06	25.8 ± 38.18
year 1-2	5.4 ± 37.48	1.11 ± 4.03	5.4 ± 37.48
year 2-3	15 ± 42.14	1.11 ± 4.04	15 ± 42.14
year 3-4	25.2 ± 47.05	1.11 ± 4.05	25.2 ± 47.05
year 4-5	22.8 ± 41.57	1.11 ± 4.06	22.8 ± 41.57
year 5-6	0	1.11 ± 4.07	0
year 6-7	8.4 ± 30.7	1.11 ± 4.08	8.4 ± 30.7
year 7+	8.4 ± 30.8	3.33 ± 3.07	8.4 ± 30.8
Number of catastrophes	None	None	None
Harvest	No	Yes	No
Optional criteria		N>=23	
Number of 0-1 year olds harvested		N-23	
Supplementation:	No	Yes	Yes
Model 1		0	4 (1 st five years)
Model 2		4 (1 st five years)	*GS1
Model 3		2(1 st five years)	GS1+2 (1 st five years)

*GS1= number of individuals harvested from the captive population (no time limit).

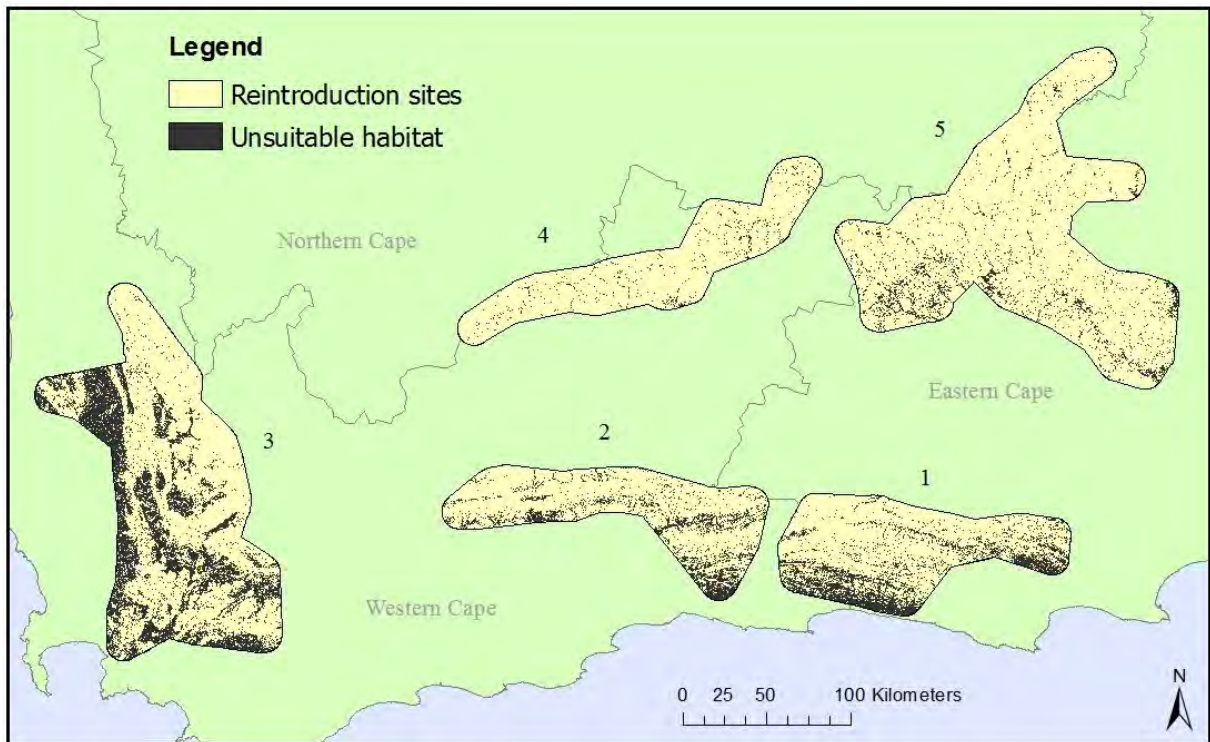
Order of events in the model: Environmental variation; Breeding; Harvest; Update Vars; Supplement; Mortality; Census; Age; Disperse; rCalc; Ktruncation.



Appendix 2: Windfarm developments (proposed, approved and existing), as well as power lines (11 kV, 22 kV and 132 kV) for each reintroduction site (1-5).



Appendix 3: Protected areas per reintroduction site (1-5).



Appendix 4: Unsuitable habitat within each reintroduction site (1-5).