Assessing the extent of and changes in the wildlife sector in Limpopo province, South Africa

by

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Summary

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The wildlife sector has grown rapidly over the past few decades and is considered a valuable asset for South African ecotourism, economy and conservation. However, there has been an increasing concern around its conservation efficacy, particularly with the industry becoming more intensive regarding its animal production. The growth of the wildlife sector, especially intensive breeding practices, has proliferated the use of fencing. Fences establish boundaries and protect wildlife, but may also cause mortality, inhibit animal movement, and can ultimately lead to landscape fragmentation which has been shown to have adverse effects on wildlife and the environment.

To infer spatial changes in the wildlife sector across a ten year time frame, I used remote sensing procedures to manually map and quantify the changes in fences and camps (fenced areas) of wildlife properties based on satellite images of south-west Limpopo during 2007, 2012 and 2017. Results show an increase in intensive wildlife properties, total length of fences, and total number of camps from 2007 to 2017. The mean area of camps decreased over the ten year time period, accompanied by an overall increase in the number of smaller camps (≤200 ha) and a general decrease in larger camps (≥500 ha). Furthermore, the areas covered by smaller camps (≤200 ha) increased whilst the areas covered by larger camps decreased (≥500 ha) over the entire time period. The biggest changes in the wildlife sector occurred between 2012 and 2017, which suggest that the changes may be occurring progressively more and should therefore be urgently addressed.

As fence maps would be very beneficial to wildlife researchers and managers, I pursued an alternative method to 'automate' the mapping of fences through image classification. Two image classification methods were used, namely Support Vector Machine (SVM) and Random Forest (RF), to classify the satellite images of the wildlife sector in south-west Limpopo. The fence area obtained from the classified images did not however correspond with the manual fence map, due to the high variability in accuracy values, specifically overall accuracy and kappa index. The SVM and RF methods were statistically identical in accuracy values. Furthermore, it was found that some landscape characteristics, such as percentage elevation and presence of water, correlated with the overall accuracy of certain classified images. Therefore, image classification methods have the potential to map fences of the wildlife sector, and needs to be improved for future use.

The extent of increase in intensive wildlife production and the rise of fences are disconcerting trends that may have detrimental consequences to wildlife and their environment. It is vital to increase research efforts to assess the extent and effects of fencing, and inform landowners of fence impacts in South Africa so as to mitigate the ecological effects of fencing. Remote sensing and image classification methods can be used to map the

full extent of fences in the wildlife sector. Ultimately, the reduction and regulation of intensive wildlife management practices and fencing may significantly aid in conserving South African wildlife.

Declaration

I, Cecilia Prinsloo, declare that the thesis, which I hereby submit for the degree Master of Science in Wildlife Management at the University of Pretoria, is my own work and has not previously been submitted by me for a degree at this or any other tertiary institution.

The thesis consists of five chapters written in the format of the Journal of Applied Ecology, three of which are compiled as if submitted to the Journal. Therefore, repetition between chapters was unavoidable.

Signature: ..……………………………...

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Lastly, I thank my husband, Mathew Wagner, and my family for their support and motivation. Their encouraging words drove this study from start to finish.

In addition, this thesis does not merely provide insights into the wildlife sector, fencing, and potential fence mapping methods. With this thesis I hope to motivate people involved in the wildlife sector to combat the increasing fragmentation of wildlife areas through fencing. Implementing regulations on intensive breeding practices and fencing, and encouraging the formation of conservancies will ultimately be the most significant and effective way to sustain South African wildlife.

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Chapter 1:

Prelude to Evaluating Changes in the Wildlife Sector

1.1 Introduction

The South African wildlife ranching sector has grown over the past few decades, and is very profitable and sustainable despite being a relatively young economic sector (Grossman, Holden, & Collinson, 1999; Reilly, Sutherland, & Harley, 2003; The National Agricultural Marketing Council, 2006; van der Waal & Dekker, 2000). Furthermore, wildlife ranching practices have become increasingly intensive (Pitman et al., 2017). Fencing, which is legally required to own wildlife in South Africa (Child et al., 2012; Cousins et al., 2010; Lindsey et al., 2012; van Hoven, 2015), may therefore be increasing as well.

Intensive wildlife practices and fencing have benefits, but also many negative effects, both of which will be discussed throughout this thesis. Our limited knowledge of the consequences of intensive wildlife practices and fencing is, however, surpassed by our very limited knowledge of the extent of and changes in the growing South African wildlife sector (Bothma & von Bach, 2010; Taylor et al., 2015). Fence maps can be used to quantify and infer changes in the wildlife sector, particularly regarding its degree of intensive management and fencing. According to current literature, the only African research to map fences is that of Desmet and Pillay (2016) who mapped a wildlife area in Limpopo province, South Africa, and that of Løvschal et al. (2017) who mapped the Greater Mara ecosystem in Kenya. Furthermore, the only study to endeavour to quantify intensive breeding in South Africa is that of Taylor et al. (2015).

The manual mapping of fences through remote sensing methods is relatively effective and widely applicable, however limited in scope (Desmet & Pillay, 2016; Løvschal et al., 2017). Image classification methods may be able to automate fence maps and encompass larger spatial scales, but has yet to be tried and tested. As there are many known and unknown consequences to fencing, it is imperative to expand research efforts and methods on the extent and effects of fencing in the wildlife sector (Lindsey et al., 2012). Subsequently, the negative effects can be circumvented and wildlife management can be improved.

1.2 Research Questions

1. What are the extent of and changes in fences within an area comprising of the wildlife sector over a 10 year time period?
2. What are the extent of and changes in fenced camps within an area comprising of the wildlife sector over a 10 year time period?
3. What are the extent of and changes in properties that show characteristics of intensive wildlife management practices within an area comprising of the wildlife sector over a 10 year time period?
4. How accurately can image classification methods, specifically Support Vector Machines (SVM) and Random Forest (RF), automate the mapping of fences?

Answering the first three questions is vital to understanding the extent of and changes in the growing wildlife sector in South Africa. An increase in fences, fenced camps, and intensive wildlife management properties may signify a trend which has far reaching ecological and economic consequences. The importance of the last question is stemmed with improving methods for mapping fences which would benefit both researchers and managers.

1.3 Aims

- To assess the extent of and changes in the fences and camps within the wildlife sector in Limpopo, South Africa.
- To automate the mapping of fences through image classification methods.

1.4 Objectives

- To manually map fences and fenced camps within the wildlife sector in south-west Limpopo, South Africa, in three time intervals spanning 10 years.
- To classify fences in satellite imagery of the wildlife sector through image classification methods, namely Support Vector Machine (SVM) and Random Forest (RF) methods.

1.5 Thesis outline

This introduction servers as the first chapter of my thesis for my MSc Wildlife Management degree. This thesis comprises of a total of five chapters, including this chapter, of which three have been prepared as peer-reviewed journal articles. Because of this, some contents will be repeated in other chapters. Chapter 2 is an extensive literature review which provides a background and elaborates on the growing wildlife sector, intensive wildlife management, and fencing. Chapter 3 focuses on assessing the extent of and changes in the wildlife sector by manually mapping and quantifying fences based on remote sensing data. Chapter 4 focuses on automating the mapping of fences through image classification methods. Lastly, Chapter 5 provides concluding remarks and recommendations for future research.

Chapter 2:

The History and Effects of the Wildlife Sector and Fencing in South Africa

2.1 Introduction

South Africa is known for its captivating wildlife, beautiful landscapes, and complex history. The wildlife industry is a valuable asset for South African ecotourism, economy and conservation (Child, Musengezi, Parent, & Child, 2012; Taylor, Lindsey, & Davies-Mostert, 2015; van der Merwe & Saayman, 2003). Over the past few decades, the South African landscape has rapidly been transformed through the transition of livestock and crop farming to game farming due to the productivity advantages and legal changes in ownership of wildlife (Child, 2013; Kamuti, 2014; Lindsey, Romañach, & Davies-Mostert, 2009). Increases in the number of properties associated with the wildlife sector has been accompanied by an increase in fencing, which have positive and negative consequences that have not yet been thoroughly studied (Lindsey, Masterson, Beck, & Romañach, 2012). Indeed, there is limited knowledge on the true scale and scope of wildlife ranching across the entire South Africa (Taylor et al., 2015). This chapter investigates and summarises some of the aspects of the wildlife sector in South Africa and specifically the fencing that accompanies it.

This literature review focuses on the wildlife sector in South Africa from the 1900s to present. Several topics are be covered including a land-use perspective of the wildlife sector, the past and present of the wildlife sector, intensive wildlife production, the pros and cons of fencing, fragmentation caused by fencing, and the financial and managerial implications of fencing. The terms ‘game’ and ‘wildlife’ as well as ‘farming’ and ‘ranching’ are used interchangeably. However, there is a slight differentiation between the terms ‘wildlife ranching’ and ‘game farming’, where wildlife ranching refers to larger areas (>5000 ha) with infrequent extensive management interventions, whilst game farming refers to smaller areas (<5000 ha) with constant intensive management interventions (see Bothma, 2010; Taylor et al., 2015). For this thesis it was decided to use ‘wildlife production’ and ‘wildlife sector’ as collective terms. The goal of this review is to lay out the positive and negative ecological, economic and management features of the wildlife sector and its associated fencing.

2.2 The Wildlife Sector from a Land-use Perspective

Game ranching can be defined as a type of land-use (Carruthers, 2008), which is connected to economic development, population growth, technology, and environmental change (Houghton, 1994). Land-uses can greatly alter and modify natural landscapes through human use and management practices (Foley et al., 2005). For example, poor farming practices, overgrazing and land clearance can lead to erosion and land degradation (Tizora, Le Roux, Mans, & Cooper, 2016). Land-use change can occur in various forms, such as changes in area and intensity of use (Houghton, 1994). Increased land-use intensity may result in decreased species diversity, especially mammal diversity, due to habitat simplification (Maitima et al., 2009). Land-use changes generally affect biodiversity, which in turn affect ecosystem functioning and services (Martínez et al., 2009), thereby emphasising the importance of maintaining biodiversity in order to conserve ecosystems. A study by Wessels et al. (2003) revealed that there is significant overlap between priority biodiversity conservation areas (where animal and plant richness and rarity are considered), and transformed and arable land in the northern parts of South Africa. In fact, future land-use changes will likely decrease species' representation in both protected areas and priority areas, especially when combined with climate change (Smith et al., 2016). Land-use has had and will have lasting effects on the environment. Historical land-use and land-use changes, from 100 to 3000 years ago, have been shown to have long-term legacy effects on soil properties, plant species diversity, and bird communities (Culbert et al., 2017; Dupouey, Dambrine, Laffite, & Moares, 2002; Foster et al., 2003; Lindborg & Eriksson, 2004). Furthermore, the legacies of past land-use modify vegetation structure and composition, which strongly influences landscape susceptibility and ecosystem response to fire, wind, and mass movement (Foster et al., 2003).

The impact of land-use on biodiversity depends on the extent of transformation, where a natural asset is lost to another form of land cover, and fragmentation, which is the pattern of natural assets remaining after transformation (O'Connor & Kuyler, 2009). Moreover, different land-uses have different degrees of impact on the landscape. O'Connor and Kuyler (2009) revealed that conservation estates, livestock ranches, game ranches, and tourism or recreation type land-uses have the least impact on biodiversity integrity, in terms of landscape composition, structure and functioning, as compared to other land-uses such as urban settlement. Livestock and game ranching have limited negative impacts and are less damaging to biodiversity due to the large proportion of remaining natural assets which support the persistence of indigenous biota (Luxmoore, 1985; O'Connor & Kuyler, 2009). Conversely, urban development leads to isolated fragments of natural assets which disrupt ecosystem functioning and result in a loss of habitat and species and an increase in alien plants (O'Connor & Kuyler, 2009). However, livestock farming can also have detrimental effects to the environment if not sustainably managed. For instance, the increase in livestock farming and reduction of wildlife due to a growing human population has led to the degradation and desertification of Kenyan rangelands in the past (Hopcraft, 2000). Furthermore, O'Connor and Kuyler (2009) added that though tourism properties were considered 'biodiversity-friendly' there remains biodiversity impacts through; transformation from infrastructure development, increased fragmentation, reduced burning and grazing, increased human waste and disturbance, and alien plant threats. Therefore, a land-use which is theoretically considered as having 'natural areas', such as the wildlife sector, may have underlying factors that could essentially make it artificial and detrimental to the broader ecosystem.

Despite the rapid expansion of the wildlife sector and its ecological and economic importance, the wildlife sector is poorly researched particularly with regards to its impact on biodiversity and the wildlife economy (Child, 2009). It is thus essential to interpret the wildlife sector as a land-use and inspect its artificial components, specifically management practices and fencing.

2.3 The Past and Present of the Wildlife Sector in South Africa

In South Africa during the 1920s and 1930s national parks and other forms of game reserves were managed without any ecological foundation, though they were highly influenced by veterinarians and agriculturalists who at the time were the country's most authoritative field scientists (Carruthers, 2008). By the mid-1960s the aesthetic value of wildlife was well recognised, however the commercial prospects were still being questioned (Carruthers, 2008) as financial organisations did not recognise wildlife as economic properties (Snijders, 2012). Nevertheless, the term 'Conservation Ethic' emerged in the 1960s which highlighted the potential of African ungulates, both from an aesthetic and conservation perspective, and from a meat production standpoint (Grossman, Holden, & Collinson, 1999). Subsequently, over the past few decades, South African livestock and crop farmers on privately owned land have transformed into game farmers (Kamuti, 2014). The transformation occurred gradually on most farms by lowering cattle stocking rates, removing cattle fences, reintroducing valuable game species, and changing the environmental management to improve wildlife productivity (Child et al., 2012). The reason behind this change may have been leisure-based at first, but has extended to include profit seeking, conservation objectives, and the increasing comprehension that game farming is more sustainable than conventional livestock and crop farming (Bothma, Suich, & Spenceley, 2009). In comparison with cattle farming, wildlife ranching is often more productive and profitable, is less labour intensive, experience less theft and diseases, provides more job opportunities, uses less water, promotes nature conservation, and is regarded as the best farming option in arid regions (ABSA Group Economic Research, 2002; Child, 2013; Smith & Wilson, 2002; The National Agricultural Marketing Council, 2006). Since the 1980s the shift in land-use, specifically where wildlife was used commercially on private land, has been a rapidly growing trend in South Africa (Grossman, Holden, & Collinson, 1999). In fact, the wildlife sector has been renowned as the fastest expanding agricultural activity in South Africa in the last three decades (The National Agricultural Marketing Council, 2006). For instance, the number of wildlife ranches with exemption permits (which allows the landowner to hunt and sell certain wildlife on his property) grew by an average of 7.4% per year from 1987 to 2005 (Taylor et al., 2015). This increase may have been due to game ranching officially becoming a form of land-use in South Africa after the 1980s (Carruthers, 2008), and legislative changes that transformed wildlife into assets for landowners (Lindsey, Romañach, & Davies-Mostert, 2009).

Wildlife ranching was formally recognised as an agricultural activity by the Department of Agricultural Development of South Africa in 1987 (van Hoven, 2015), and the requirement for legal ownership of game was officially implemented in 1991 through the Game Theft Act, Act No. 105 (Nel, 2015) which required landowners to obtain a ‘Certificate of Adequate Enclosure’ (CAE), also called ‘Certificate of Sufficient Enclosure’ (CSE), ‘Exemption Permit’, or ‘Game Farm Permit’ depending on the province (Snijders, 2012; Taylor et al., 2015). Before 1991 all wildlife in South Africa was *res nullius* and belonged to the state (Bothma et al., 2009). Therefore, the Game Theft Act aimed to regulate ownership of game and combat wildlife theft, illegal hunting and capture through fencing (Bothma et al., 2009). The survival of the animals and their habitats on the fenced property thus relies on the landowner's decisions regarding the land-use management and whether there is sufficient incentive to protect the animals (Child & Chitsike, 2000). An exemption permit allows the landowner to hunt, capture, sell and trade any wild animal authorised on the permit for approximately three years depending on the province (Snijders, 2012; Taylor et al., 2015), which ultimately gives the landowner incentive to ensure the survival of the wildlife they own. Subsequently, van Hoven (2015) reported a 40-fold increase in the number of wild animals in both private and public conservation areas from the early 1960s. By the year 2000 there were approximately 5000 fenced wildlife farms and 4000 mixed wildlife and livestock farms that covered more than 13% of South Africa, compared to the 5.8% land that officially declared conservation areas covered at the time (ABSA Group Economic Research, 2002). More recent reports suggest that there are now over 9000 wildlife ranches in South Africa containing 16 to 20 million wild animals (Taylor et al., 2015). As a result, wildlife ranches are now reportedly protecting approximately 16.8% land (3.8% increase in ±5 years), as compared to the 6% land (0.2% increase in ±5 years) that government protected areas protect (Bothma & von Bach, 2010; Child et al., 2012; Cousins, Sadler, & Evans, 2008; The National Agricultural Marketing Council, 2006). Game ranching has been regarded to be beneficial to conservation by increasing the area of land used for wildlife production, maintaining natural areas, and improving several endangered species populations through reintroductions (Cousins et al., 2008; Lindsey et al., 2009). Private land often conserves biodiversity at a lower cost to the taxpayer than state-protected areas which are seemingly deemed as superior as they are automatically given a higher status according to International Union for Conservation of Nature (IUCN) categories (Child, 2009). Private landowners also conserve biota and habitats that are not well represented in formal protected areas, and neighbouring properties increase the range of effective conservation management (Higginbottom & King, 2006). In 2006 the private wildlife sector covered ±20.5 million ha and contained about three to four times more animals than in the ±7.5 million ha government protected areas in South Africa (The National Agricultural Marketing Council, 2006). Smith (1981) argued that as opposed to common property resources, private ownership such as game ranches can successfully preserve wildlife even though the animals are raised for profit and hunted, because private property owners are driven by self-interest to protect their resource. Yet, there is still a lack in literature concerning the role that private wildlife ranching plays in economy and conservation (Bond et al., 2004; Cousins et al., 2008; Langholz & Kerley, 2006; Taylor et al., 2015).

Game farming generates income through selling game meat, hunting, live animal sales, and game viewing (Child et al., 2012; Luxmoore, 1985). Since the 2000s game ranching has been renowned as a rapidly expanding enterprise of South Africa (Reilly, Sutherland, & Harley, 2003; van der Waal & Dekker, 2000) with an increase of 5% to 20% in income per year in the 2000s (Child et al., 2012). Van der Merwe and Saayman (2003) estimated that trophy and biltong (dried, cured meat) hunting are the biggest form of income for game farms, while live game sales and the breeding of rare species are the second biggest generator of income. Game ranching trends, in terms of income, are accelerating at present with high-value game species (>R10 000 per animal, e.g. sable antelope *Hippotragus niger*) being auctioned at the highest prices (Pitman et al., 2017). In South Africa there are several methods for trading animals, namely private sales, public live auctions, public catalogue auctions, tender systems, and electronic auctions (Ebedes, 1994; The National Agricultural Marketing Council, 2006). Live game auctions in South Africa date back to 1975, from the first live animal auction at Hoedspruit which generated R20 362 from 128 animals (du Toit, 2010a) - equal to R713 647 in 2014 based on the average annual inflation rate. Profits produced by live game auctions seemed to have greatly increased since the first auction. In 2005 live game auctions contributed R93.5 million to the wildlife industry (Cloete, Taljaard, & Grové, 2007) which is equal to R152.7 million in 2014, and in 2014 it generated R4.3 billion from approximately 225 200 animals (Taylor et al., 2015). During 2014 around 130 186 animals were hunted for trophies which generated an estimate of R1.96 billion, 277 027 animals were hunted for biltong which generated an estimate R650 million, and 176 969 animals were culled for game meat which generated R349.7 million for the South African economy (Taylor et al., 2015). A study by Saayman et al. (2011) discovered that biltong hunting contributed an estimated R6 billion to real Gross Domestic Product (GDP) and could create approximately 140 000 jobs due to the growth in the wildlife industry in South Africa. However, the source and structure of income of wildlife producers differs widely between South African provinces and is affected by distance from cities, habitat type, and the wildlife present on the property (Bothma et al., 2009). Studies that attempt to estimate the economic impact of the wildlife sector are unfortunately scarce. In fact, the comparative contribution of wildlife-related tourism and nature-based tourism on private land is unknown regardless of the economic importance of tourism (Bothma et al., 2009).

Whether the economic or conservation value of the wildlife sector is being prioritised is up for debate, as a detailed assessment of the social, economic and conservation value of the wildlife ranching industry is lacking (Taylor et al., 2015). Wildlife ranches mostly have limited conservation value due to their commercial nature (Cousins et al., 2008), which contribute to several ranching practices conflicting with conservation principles (Cousins, Sadler, & Evans, 2010). However, it is being increasingly acknowledged that conservation activities are mostly driven by proprietorship and price, which make them ecologically, economically and socio-politically sustainable (Child, 2013). The expansion of wildlife in southern Africa was greatly influenced by wildlife owners who increased the price of wildlife and introduced a variety of wildlife products to the markets (Child, 2000). Unfortunately, game auction prices have not correlated with species contribution to either phylogenetic or functional diversity for the past 20 years, thereby questioning the contribution of the game market to biodiversity conservation (Dalerum & Miranda, 2016). Nevertheless, in order for wildlife to survive it has been argued that wildlife should have some socio-economic benefit to the human population. Game viewing, sport hunting, game meat and game ranching have essentially been introduced in order to give wildlife value so that people overall may be more willing to participate in conservation (Eltringham, 1994).

The wildlife industry in South Africa is generally driven by tourist and hunter preferences which result in an artificial representation of species and an increase in the introduction of exotic (species not native to the subregion) and extra-limital species (native species translocated to regions outside their natural historical range) (Cousins et al., 2008; Luxmoore, 1985). By introducing exotic and extra-limital species, wildlife operations ultimately choose short-term commercial benefits above long-term conservation policies (Castley, Boshoff, & Kerley, 2001). The introduction of such species may increase the biodiversity of the area per se, however it is deemed as artificial (anthropogenic) biodiversity (Angermeier, 1994). Over the long term artificial biodiversity is of much less value than native (naturally evolved) diversity, especially in terms of societal value or ecological function (Angermeier, 1994). Introducing non-indigenous and extra-limital species threatens indigenous biodiversity through hybridisation with native species, resource competition, modification of functional relationships, niche displacement, homogenisation of native biota, introduction of foreign parasites and pathogens, changing vegetation structure and composition, facilitation of non-indigenous plants, and alteration of ecosystem functioning (Castley et al., 2001; Spear & Chown, 2009; Taylor et al., 2015). A study by Taylor et al. (2015) on 249 South African wildlife ranches in 2014 reported that 16% of the ranches had non-indigenous species, and 88% of ranches had at least one extra-limital species according to the Department of Environmental Affairs (DEA) distribution maps. Ranchers have also generated colour variants (rare colour phenotypes) of some species through genetic manipulation (Cousins et al., 2010; Taylor et al., 2015), and even created new varieties of hunting trophies through cross-breeding species (Cousins et al., 2010; Lindsey et al., 2009). Nonetheless, du Toit (2010a) highlight the negative impacts of selective breeding of colour variants and hybrids while emphasising the benefits of maintaining genetically viable populations.

The profits gained from selling colour variants may however be needed for some wildlife ranchers to remain financially sustainable under economic pressures (Taylor et al., 2015). Colour variants were very profitable, with a golden wildebeest *Connochaetes taurinus* bull selling for US$35 700 (143 times more than normal), and a black impala *Aepyceros melampus* selling for US$22 000 (220 times more than normal) in 2013 (van Hoven, 2015). The prices of colour variants rose until its peak in 2014 (e.g. white impala sold for R8.2 million), but was shortly followed by a downward spiral where it collapsed in 2016 (e.g. white impala sold for R48 333) due to a combination of strong opposition toward hunting of bred animals, drought, and there being no viable market for colour variants (Thomas, 2017). The breeding of colour variants may be profitable, but threaten biodiversity through genetic mixing, biotic homogenization, potentially increasing detrimental genetic traits, distorting natural selection, and reducing adaptive capacity (Taylor et al., 2015). There are also interspecific consequences of colour variant production, such as the persecution of predators to safeguard colour variants (Taylor et al., 2015). In addition to breeding colour variants, wildlife ranches are also prone to introduce invasive alien species, convert veld through burning and cutting, and trap or hunt predators in order to protect valuable game (Cousins et al., 2010). Private ranches have also introduced extra-limital species, and impeded conservation objectives by stocking high numbers of high-value species and modifying vegetation structure (Sims-Castley, Kerley, Geach, & Langholz, 2005) in order to improve tourist wildlife-viewing, which ultimately results in negative impacts on ecosystem function and biodiversity (Bothma et al., 2009). The need to 'display' animals for aesthetic purposes has led to habitat manipulation, for example construction of artificial waterholes (Child, 2013) which had unforeseen consequences, such as the decline in roan antelope populations in the Kruger National Park, South Africa (Harrington et al., 1999). Habitat manipulation, increased reserve isolation and reduced predation essentially create imbalanced species composition which suppress ecological values (Child, 2013). Other management interventions include removing competition with focal species (pests and predators), manipulating the food supply, rotational burning, fertilization, irrigation, introducing food species or directly providing food, providing additional water or mineral supplements, and creating nesting or breeding sites (Luxmoore & Swanson, 1992).

Most management interventions are implemented by wildlife ranchers themselves, based on their own knowledge and experience, books, wildlife ranching magazines, or trial and error (Pienaar, Rubino, Saayman, & van der Merwe, 2017). A study by Pienaar et al. (2017) observed that only 32% of the ranchers they interviewed appointed a manager with an ecological background, received guidance from the provincial department of environment or nature conservation, or used ecological assessments to establish a management plan. Furthermore, there are often insufficient resources available for professional conservation management and planning (Cousins et al., 2008), which may lead to unethical activities such as 'canned' hunting (releasing captive reared animals in small enclosures to be shot) and 'put and take' hunting (buying and releasing animals on the farm immediately before being shot) (Lindsey, Alexander, Frank, Mathieson, & Romañach, 2006; Lindsey et al., 2009). Inadequate management is not the only concern regarding animal welfare. For example, game ranchers and conservationists at auctions have made the following complaints: some animals have been kept in inadequate holdings, pens are at times too small and are overcrowded, animals are on occasion under stress and not even tranquilized, and sometimes even injured and sick animals are offered for sale (Ebedes, 1994). The Animal Protection Act (No. 71 of 1962) is the primary source of welfare protection for most animals, domestic or wild, however the wordings of the offences contain subjective exceptions and contradictions (Agjee et al., 2018). For instance the word “unnecessary” is used in conjunction with “overloading”, “ill-treatment” and “neglect”, which create loopholes based on the owner’s personal views, ability to care for the animal, and circumstances in which the animal is used (Agjee et al., 2018).

Nevertheless, there is little information on the growth of the wildlife sector in South Africa and its ecological impact with which to guide policy (Bothma & von Bach, 2010; Lindsey et al., 2009; Taylor et al., 2015). One of the issues is that private game farming juxtaposes the wildlife and agricultural sectors, both of which have to be sustainably managed to provide various needs (Kamuti, 2014). Together, conservation and agriculture form the foundation of game farming which thus requires unique management practices to guarantee sustainable land-use (Smith & Wilson, 2002). However, there is a jurisdictional divide between the Department of Environmental Affairs (DEA) and the Department of Agriculture, Forestry and Fisheries (DAFF) (Agjee et al., 2018), with DAFF associating with domesticated animals and agribusiness and the DEA associating with wildlife and biodiversity conservation (Snijders, 2012; Taylor et al., 2015). Yet, in 2016 DAFF added twelve wildlife species to the list of tame and domesticated animals regulated under its Animal Improvement Act (No. 62 of 1998), thereby allowing for the genetic manipulation and cross-breeding of the wildlife species (Naude, 2016). As a result, the act ultimately conflicts with conservation principles that protect indigenous wildlife and their genetic integrity (Naude, 2016). This may further proliferate the intensification of wildlife ranching, which is ultimately causing more and more wild animals to become increasingly domestic, causing them to lose some of their conservation value in essence (Taylor et al., 2015). The intensification of the wildlife sector may generally have far-reaching effects on wildlife and the environment which may have not yet been extensively investigated.

2.4 The Growing Intensive Wildlife Production Sector

In southern Africa, commercial wildlife production has only developed in the past 50 years, mostly due to increasingly smaller farming units, often on agriculturally marginal land that had lost its economic viability (Bothma & van Rooyen, 2005). Ideally there should be a balance between the commercial and conservation values of the wildlife sector, however intensive wildlife production seems to be ‘tipping the scale’ by prioritising financial profits. There are two main forms of wildlife production systems, namely extensive and intensive (Cousins et al., 2008, 2010). In order to breed high-value sought after game species, wildlife ranching practices have become more intensive over the last decade (Pitman et al., 2017). Intensive wildlife farming can be referred to as an agricultural system where wildlife is maintained in relatively small, enclosed areas with food and water in order to harvest by-products (Carruthers, 2008). Similarly, intensive livestock farms (also known as concentrated animal feeding operations) raise animals in confinement at high stocking density, using economies of scale, modern machinery, and biotechnology (Ilea, 2009).

While researching ‘intensive wildlife production’ for this literature review, it was found that there is in fact very little literature on the topic. Thus a systematic quantitative approach (see Pickering et al., 2014) was used to quantify the literature on intensive wildlife production. The following keywords were selected for the systematic quantitative literature review method using Google Scholar: “intensive” in combination with “wildlife breeding”, “game breeding”, “wildlife production”, “game production”, “wildlife farm”, “game farm”, “wildlife ranch”, “game ranch”, “wildlife management”, and “game management”. Through this keyword search only sixteen articles and four books were found to be relevant to intensive wildlife production specifically (see Appendix: Table A1). Several other references were found coincidentally through searching for other references for this literature review.

Many of the terms linked with intensive wildlife management were used interchangeably despite many authors defining them differently. In some cases ‘game’ and ‘wildlife’ were used interchangeably, despite the term ‘game’ being reserved for animals that are hunted for meat or sport (Carruthers, 2008; Taylor et al., 2015). In particular, ‘wildlife ranching’ refers to larger areas (>5000 ha), whereas ‘game farming’ refers to smaller areas (<5000 ha) (Bothma, 2010). ‘Game farming’ can also be defined as a production mode where game animals are managed, handled and slaughtered in ways alike to traditional livestock (Drew, 1989). ‘Game ranching’ is also said to be where captive herds are kept under extensive conditions with minimal interchange between wild populations while the habitat is managed by providing water points and having burning regimes, in addition to species composition being manipulated through selective culling and the introduction of livestock bought from other farms or caught in nature reserves (Luxmoore & Swanson, 1992). In short, ‘game ranching’ infers the commercially-orientated stocking and use of game (Grossman et al., 1999). ‘Intensive game or wildlife farming’ is where wildlife is kept in small, fenced areas with food and water (Carruthers, 2008), which is very similar to captive breeding. ‘Captive breeding’ is where animals are separated from their wild population, which results in separate gene pools, by removing them to an artificial enclosure, fencing in an extensive area, or closely controlling a free-ranging herd (Luxmoore & Swanson, 1992). ‘Intensive breeding’ is defined as the confinement of wild species (usually high in value such as sable antelope) in small to medium sized fenced camps where they are protected from predators and provided with food, water and veterinary care (Taylor et al., 2015). The main purpose of intensive breeding is to produce 'superior' animals for live game sales or trophy hunting by manipulating breeding conditions to select animals for desirable traits, such as long horns, large body size or multiplication of a colour variant (Taylor et al., 2015).

'Intensive breeding' and 'captive breeding' were also used interchangeably, even for animals that were not strictly "captive" or "confined" securely. Captive breeding can be used directly to obtain animal products through intensive breeding, for example, ostrich *Struthio camelus* and crocodile *Crocodylus niloticus* production in southern Africa (Barnes, 2001, 1998), deer (cervid) farms in Europe (Hoffman & Wiklund, 2006), turtle farms in China (Shi, Fan, Yin, & Yuan, 2004) and intensive and semi-intensive production systems for collared peccary *Tayassu tajacu* and capybara *Hydrochaeris hydrochaeris* in Brazil (Nogueira-Filho & Nogueira, 2004). Captive or intensive breeding can also be used indirectly to obtain animal products through restocking game for hunting purposes, which can result in genetic homogenization and hybridisation (Barbanera et al., 2010; Díaz-Fernández, Arroyo, Casas, Martinez-Haro, & Viñuela, 2013; Yilmaz & Tepeli, 2009). Example species include the chukar partridge *Alectoris chukar* (Yilmaz & Tepeli, 2009) and the red-legged partridge *Alectoris rufa* (Barbanera et al., 2010; Díaz-Fernández et al., 2013). Nogueira and Nogueira-Filho (2011) claim that the wildlife farming of collared peccary may be an alternative to unsustainable hunting and deforestation if the program could integrate the community and the government. Captive breeding may aid conservation by increasing the populations of exploited species, diverting trade away from wildlife populations, and conserving natural habitats (Luxmoore & Swanson, 1992). In certain cases intensive breeding may be necessary to support a population, for instance intensive management has increased endangered roan antelope populations in small conservation areas in South Africa (Dorgeloh, van Hoven, &

Rethman, 1996). Reintroduction through captive-bred animals has often been a method used for the conservation of endangered species (Caughley, 1994; Wilson & Price, 1994); for example, the brush-tailed bettong *Bettongia penicillata*, in Australia (Delroy, Earl, Radbone, Robinson, & Hewett, 1986). However the uncertainty and political consequences concerning the removal of threatened species from wild source populations, and the complexities and costs of reintroducing of captive-bred animals suggest that it may be an unreliable method to reduce the threat of extinction (Ginsberg, 1994).

Captive breeding may also be destructive to pre-existing vegetation and does not contribute significantly to the conservation of natural habitats (Luxmoore & Swanson, 1992). Furthermore, after a period of time captive-bred animals are no longer fit for reintroduction or interbreeding with their wild counterparts (Luxmoore & Swanson, 1992). Unscientific intensive captive breeding programmes can also cause inbreeding (the mating between close relatives) of rare game species, and when released into wild populations can cause genetic pollution, extinction of subspecies, and the spread of disease and parasites (Cousins et al., 2010). Indeed, the number of species that has truly been saved from extinction by captive breeding programmes is relatively small, mostly due to low reintroduction success (Magin, Johnson, Groombridge, Jenkins, & Smith, 1994). It is also possible to use captive sources as an alternative supply to an over-exploited wild population by diverting some trade and reducing the pressure on the wild population (Luxmoore & Swanson, 1992). Similarly, wildlife farming could potentially be used to combat rhinoceros poaching in southern Africa with the added incentives that the captive rhinoceros populations may be used to restock depleted wild populations, offer a genetic safety net, and provide revenues to fund the conservation of the wild populations (Bulte & Damania, 2005). However, as in the example of rhinoceros farming, intensive breeding may in turn facilitate the laundering of illegal output from the wild (as was observed in green pythons *Morelia virdis* in Indonesia (Lyons &

Natusch, 2011)), reduce the stigma associated with certain wildlife commodities, inflate the demand and prices, result in more intense hunting pressure, and even remove the incentive to protect the wild populations (Bulte & Damania, 2005). However, it should be added that the factors that drive poaching, especially rhinoceros poaching, goes beyond that of wildlife management and conservation, and in particular beyond the scope of this study.

Intensive wildlife production has a number of negative consequences. Semi-intensive multispecies productions systems, such as game farms, often manipulate wildlife species composition and sometimes introduce alien species (Bond et al., 2004). Intensively bred species populations on game farms are often not self-sustaining due to being provided with plenty food, water and veterinary care (Taylor et al., 2015). Although management intensity can have a positive correlation with herbivore density, stocking rate and community heterogeneity, it can also have a negative correlation with primary productivity and reintroduction success (Child, Peel, Smit, & Sutherland, 2013). Intensive and selective breeding can result in loss of rare alleles, genetic fitness and adaptability; changes in behaviour and morphology; enhancement of pathogen spread and susceptibility; homogenisation; semi-domestication; reduced overall welfare; increased predator persecution; and erosion of habitat integrity (Nel, 2015). Other risks include inbreeding or outbreeding depression, hybridisation, development of resistant parasite strains, habitat loss, and fragmentation (Desmet & Pillay, 2016). There is also the growing possibility of domestication of some wildlife populations. In fact, attempts at wildlife domestication continue and have been increasing over the past century worldwide (Luxmoore & Swanson, 1992). For example, musk rat *Ondatra zibethicus* have been bred in captivity for their fur in North America since 1910, and moose *Alces alces* for their meat and milk in Eastern Europe since 1949 (Luxmoore & Swanson, 1992).

The conversion of natural habitat to specialised production, such as intensive breeding, is driven by the economic law of specialisation and the productivity gains that it promises (Luxmoore & Swanson, 1992). In South Africa some wildlife ranchers have shifted their focus to intensive breeding to remain financially sustainable (Taylor et al., 2015) or increase profits as in the case of colour variants (Thomas, 2017; van Hoven, 2015). The desire to make profits has led to unnatural breeding methods where animals are bred (and killed) under human-dependent conditions, and can therefore not be classified as wildlife (The National Agricultural Marketing Council, 2006). Indeed, Nel (2015) stated that certain animal populations cannot strictly be classified as 'wildlife' as they do not originate from 'free-roaming wild animals' and cannot survive in the wilderness, procreate naturally, defend themselves, flee successfully from predation, or select and survive off food from their natural habitat. An interesting quote from Child and Chitsike (2000) describes the issue very well: "After all, there is nothing wild about intensively farmed crocodiles and ostriches that are bred in captivity and managed much like battery raised chicken or pen fed livestock". Conversely, Luxmoore and Swanson (1992) distinguish domestication and captive breeding from ranching and wild harvest by highlighting the difference in the interaction of the exploited animal with the wild population which is what keeps the ecosystem intact.

Few studies have attempted to quantify the intensive wildlife production sector, particularly in South Africa. The most comprehensive study of intensive wildlife production was by Taylor et al. (2015) which reported that 38% of 251 wildlife properties in South Africa bred high-value species (defined as 'uncommon herbivore species with high monetary value', e.g. sable antelope), which included 14 different species, and 23% of the properties bred colour variants (defined as 'wild animals expressing a rare colour phenotype', e.g. black impala) in camps. The study also observed that the mean camp size was 111 ha and the median camp size was 50 ha on 112 wildlife properties that conducted intensive breeding

(Taylor et al., 2015). Furthermore, the provinces that had the highest percentage of wildlife ranches conducting intensive breeding surveyed by Taylor et al. (2015) were; North West (78%), Northern Cape (65%), and Limpopo (51%). The following are a few examples of intensive wildlife production in South Africa. In North West a popular wildlife ranch of 500 ha has bred and sold high-value game species, colour variants, and predators by: using six breeding camps (sized 25 ha to 50 ha) with waterholes, using a Tickoff system for parasite control, following a strict inoculation programme, and supplying animals with additional feed in winter (Coleman, 2015). A farmer in North West specialises in breeding bushbuck *Tragelaphus scriptus*, which he states can thrive in 0.2 ha to 0.5 ha 'mini camps' if provided with supplementary feed and lick in winter (Nel, 2017). A family in Free State is breeding buffalo *Syncerus caffer* on their 1500 ha farm which is divided into five camps – with electrified 2 m fences and 2.4 m, 22-strand Bonnox fences – which have watering points, high biosecurity, and additional lucerne and maize stover during droughts and in winter (Coleman, 2017).

The intensive wildlife breeding sector has been expanding rapidly recently, and the conversion from extensive to intensive farming is presented by the spread of fences which fragment the landscape into smaller camps (Desmet & Pillay, 2016). According to Luxmoore and Swanson (1992) specialised production, such as intensive breeding, is based on homogeneity, intervention and segregation. Segregation is necessary for optimal individual investment and is clarified by the designation of individual property rights, which is applied through fencing (Luxmoore & Swanson, 1992). Intensive wildlife management practices are not possible without fenced camps, resulting in the proliferation of fencing which has its own consequences. Therefore to improve our understanding of the effects of intensive breeding, the effects of fencing should be investigated.

2.5 The Positive and Negative Effects of Fencing

Fencing is widely used for various purposes around the world. As described in Hayward and Somers (2012); Australia and New Zealand have used fencing for conservation especially to protect native species from introduced 'pest' species, while India has only recently used fencing to address human-wildlife conflict, and East Africa has rarely used any fencing. South Africa and Namibia opted for game fencing to devolve the rights to wildlife based on properties, whereas Zimbabwe used a collective action solution where residents control environmental externalities through 'bottom-up', community-based conservation governance (Child, 2009). Fencing is a legal requirement for ranchers to own wildlife in South Africa (Child et al., 2012; Cousins et al., 2010; Lindsey et al., 2012; van Hoven, 2015) due to the Game Theft Act which states that a landowner has absolute rights to game that have been properly fenced on a landowner's property (Nel, 2015). Wildlife-proof fences first appeared on private land in South Africa in the 1950s and allowed for the use of wildlife (Hearne & McKenzie, 2000). Fencing is increasing exponentially across Africa due to a growing human population, moving from pastoralism to agriculture, altering land ownership policies, sub-division of large portions of land into smaller privately owned fragments, and rising human-wildlife conflict (Gadd, 2012). In the year 2000 there was an estimated 4000 ranches in South Africa that had game fences extending over almost 80 000 km^2, as compared to less than 10 000 km^2 in 1979 (Hearne & McKenzie, 2000). Despite the prevalence of fencing in southern Africa, the literature lacks a comprehensive review of the potentially significant ecological, financial and social impacts of fencing as a wildlife management tool (Lindsey et al., 2012).

Game fences have many benefits for the farmer, including the protection of game populations from over-exploitation, ownership of game, and more authority concerning the hunting, capturing and selling of game (Grobler & van der Bank, 1994). In short, fencing confines livestock and wildlife, controls access, reduces human-wildlife conflict, allows animal utilisation, decreases the spread of disease, establishes boundaries, and improves security (Boone & Hobbs, 2004; Gadd, 2012; Hayward & Kerley, 2009; Hoare, 1992; Lindsey et al., 2012; Taylor & Martin, 1987). Illegal extraction of natural resources, poaching and other criminal activities are restricted by fencing as well (Lindsey et al., 2012). Furthermore, the areas adjacent to fences are usually cleared to form a four to ten metre open patch of ground which enables fence patrols to scan for human and animal footprints, and also aids in controlling fires by acting as a firebreak (Lindsey et al., 2012). Predator-proof fences have been used to protect threatened species and locally rare ungulates, reintroduce species (with release paddocks or 'bomas'), display native species for educational and ecotourism purposes, and serve dual educational and research functions (Dickman, 2012; Lindsey et al., 2012). Fences are not only used for enclosures but for exclosures as well, for example fences can exclude large herbivores from camps, staff housing, infrastructure, threatened plant species, and sensitive habitats (Lindsey et al., 2012; Maschinski, Frye, Rutman, & Rutmant, 1997; Slotow, 2012). Fenced areas are essential for scientific research, such as the herbivore exclosures in the Kruger National Park which offer a means to better understand herbivore impacts on ecological processes (Botha & Siebert, 2014). Fencing also excludes animals from roads, and therefore reduces wildlife-vehicle collisions (Clevenger, Chruszcz, & Gunson, 2001). Over the past century expensive veterinary cordon fences have been erected across southern Africa to control animal diseases (Gadd, 2012; Lindsey et al., 2012), such as foot and mouth disease, African swine fever, malignant catarrhal fever, rinderpest, canine distemper, and bovine tuberculosis (Bengis, Kock, & Fischer, 2002). However, after extensive areas of southern Africa were fenced to control foot and mouth disease in the 1980s, questions were raised over the technical and economic merits of a fencing solution (Owen & Owen, 1980; Taylor & Martin, 1987). In fact, the erection of veterinary fences in the Okavango Basin, Botswana was never approved by the Department of Wildlife and National Parks, the tourism industry, nor local and international conservation groups, due to its detrimental effects on migratory wildlife species and not having undergone any Environmental Impact Assessments (Mbaiwa & Darkoh, 1999). Indeed the effectiveness of fencing in disease management is low (du Toit, 2010b), and most fences were later evidenced to be unnecessary, ineffective, or economically unjustified (Child, 2013).

There are various negative consequences to fences due to them being constructed for political rather than ecological reasons (Gadd, 2012). In summary, fences cause animal collision and entanglement (which may result in death), block migration and dispersal, limit recolonisation, stop gene flow, cause inbreeding, restrict evolutionary potential, negate behavioural advantages, modify predator behaviour, allow overstocking, prevent animals from accessing key resources, lead to local extinction, and cause habitat degradation (Boone & Hobbs, 2004; Child et al., 2012; du Toit, 2010a; Gadd, 2012; Hayward & Kerley, 2009; Kozakiewicz, 1993; Lindsey et al., 2012; Mbaiwa & Mbaiwa, 2006; Taylor & Martin, 1987). Medium and large ungulate and sub-ungulate carcasses have been found along, entangled in, or trapped between fences (Gadd, 2012). A study in North America witnessed that juvenile pronghorn *Antilocapra americana*, mule deer *Odocoileus hemionus*, and elk *Cervus elaphus* in particular were eight times more likely to die in fence-related accidents than adults (Harrington, 2005). Beck (2008) observed animals from 33 species that were killed as a direct result of electric fencing infrastructure in South African conservation areas and livestock farms. The most frequently killed reptile species were; leopard tortoises *Stigmochelys pardalis*, rock monitors *Varanus albigularis*, southern African python *Python natalensis*, Lobatse hinged tortoise *Kinexys lobatsian*, and mammal species were; pangolin *Smutsia temminckii* and porcupine *Hystrix africaeaustralis* (Beck, 2008). Fences also impact birds through snagging (impaling a body part), snaring (trapping a foot or leg), snarling (trapping the entire body), impact injuries, electrocution, and causing a barrier effect for flightless birds (BirdLife South Africa, 2018). A study by BirdLife South Africa in 2013 revealed that 36 bird species, of which eleven were considered threatened, were killed as a result of fencing (BirdLife South Africa, 2018). The most conspicuous consequence of fence construction over decades has been mass mortality events involving migratory ungulates such as springbok *Antidorcas marsupialis*, hartebeest *Alcelaphus buselaphus* and blue wildebeest (Gadd, 2012; Owen & Owen, 1980). Indeed, many fences cut through habitat types and wilderness areas, and do not accommodate for animal movement regarding seasonal migration, wet season range expansion, or natal dispersal (Gadd, 2012). By blocking animal movement fencing may solve human-wildlife conflict locally, but may cause conflicts to shift elsewhere and ultimately decrease local connectivity (Osipova et al., 2018). Without dispersal local extinctions will occur in even larger areas and remain longer in these areas (Fahrig & Merriam, 1994). Populations may become isolated preventing the natural mixing of populations and interrupting gene flow, which may result in inbreeding and increase the occurrence and impacts of founder effects (the loss of genetic variation due to a new population being established by very few individuals) and genetic drift (the decrease in the number of different alleles in a population), and ultimately decrease heterozygosity (having two different alleles at a locus) and fitness (Caughley, 1994; Lindsey et al., 2012). The isolation of populations and the prevention of immigration is believed to be fundamental causes in the long-term decline of some mammals in southern Africa (Gadd, 2012).

There are various other consequences of fencing that have not yet been studied in depth. For example, pest-proof fences in New Zealand may cause changes in community composition, prevent individual dispersal and disrupt mutualistic species interactions (Burns, Innes, & Day, 2012). Climate change may aggravate fence effects as animal movements would try to counter changing rainfall patterns and resource distribution (Gadd, 2012). The impact of fencing on migratory populations and species can also lead to changes in community composition and animal behaviour (Gadd, 2012). For instance, fencing specifically affects the spatial and social behaviour of reintroduced species (Hayward, 2012). In addition, studies on elephants *Loxodonta africana* show that fences constrain their movement, increase their densities (Mackey, Page, Duffy, & Slotow, 2006; Slotow, Garaï, Reilly, Page, & Carr, 2005), and may cause them to bunch up against the fence (Loarie, van Aarde, & Pimm, 2009). Large mammals, such as elephants, have however been shown to be capable of learning about fences and devising means to overcome the obstacles (Slotow, 2012). Predators frequently escape fences (Taylor et al., 2015) and may also use fences to their advantage (Gadd, 2012), for example wild dogs *Lycaon pictus* have learned to charge animals against fences when hunting (van Dyk & Slotow, 2003).

There are also social consequences regarding the wildlife sector and fencing in South Africa, such as the establishment of government protected areas which had often led to forced removals and resource dispossession of African communities during the Apartheid era (The National Agricultural Marketing Council, 2006). Furthermore, due to land conversions, game-fencing policies, and wildlife practices, people who dwell on game farms may be denied access to grazing land and livestock (Snijders, 2012). However, the social implications of fencing is an entire subject on its own and is unfortunately not covered in this literature review (see Brooks et al., 2011) for more information on the matter).

2.6 Fragmentation via Fencing

Fences and roads fragment natural habitats into isolated remnants (Lovejoy, Bierregaard, Brown, Emmons, & van der Voort, 1984) and are one of the many and major causes of African protected areas becoming more remote (Newmark, 2008). Linear infrastructure, such as roads, affect local soils and hydrology, is a source of chemical pollutants, cause edge effects (which result in changes in species abundance and community composition), cause mortalities through vehicle road-kill, create barriers to faunal movements, and are consequently avoided by most species (Laurance, Goosem, & Laurance, 2009). Fences may also be considered detrimental linear infrastructure. Unlike roads fences are vertical barriers that are typically unregulated and mostly constructed and maintained by private landowners (Jakes, Jones, Paige, Seidler, & Huijser, 2018). The rise of fences has augmented the fragmentation of ecosystems globally (Jakes et al., 2018). For instance, the Greater Mara ecosystem in Kenya is being fragmented due to a rapid increase in fencing caused by land privatization, increased livestock densities, and a growing human population (Løvschal et al., 2017). Game fencing in particular has fragmented large regions of private land into small parcels across southern Africa due to its increasingly important role in aiding wildlife management, preventing disease spread, and reducing human-wildlife conflict (Gadd, 2012; Lindsey et al., 2012). Owen-Smith (1983) accounts that "all wildlife reserves are destined to become ecological islands in a sea of man-modified landscapes". Fences, especially non-porous such as predator proof fences, may function as hard boundaries fragmenting the landscape into small, disconnected patches and thus be subject to the predictions of island biogeography theory (Gadd, 2012; MacArthur & Wilson, 1967), along with the species-area relationship, which together predict that as reserves become increasingly isolated, more and more species will be lost as reserve area declines (Newmark, 2008). The island biogeography theory (MacArthur & Wilson, 1967) is one of the earliest theories to predict extinction and immigration rates and patterns of species in isolated fragments (Haddad et al., 2015). A collective study on several forest fragmentation experiments covering five continents and 35 years revealed that fragmentation reduced plant and animal species richness, changed community composition, and degraded core ecosystem functions (Haddad et al., 2015). Habitat fragmentation essentially reduces the overall area of a habitat, increases the isolation of habitat patches and the amount of edge habitats (Kozakiewicz, 1993), as well as cause the subdivision of the remaining vegetation, and the introduction of alternative land-use forms to substitute the lost vegetation (Bennett & Saunders, 2011). Fences play a role in habitat conversion, degradation, and fragmentation (Desmet & Pillay, 2016; Ferguson & Hanks, 2012) and could lead to temporary or permanent habitat loss for many large mammals (Reid et al., 2017). Fences fragment a landscape by reducing carrying capacity (the capability of habitats to support populations) and restricting animal movement (Ben-Shahar, 1993; Boone & Hobbs, 2004) between habitat patches which are crucial to sustain populations in heterogeneous landscapes (Kozakiewicz, 1993). Furthermore, fragmentation causes heterogeneity to decrease within each fragment (Gadd, 2012), resulting in small isolated populations with lower levels of heterozygosity than larger populations (Saunders, Hobbs, & Margules, 1991).

Fenced reserves are thus prone to problems associated with small populations, namely becoming more vulnerable to environmental, demographic and genetic stochasticity (Caughley, 1994; Gadd, 2012; Lindsey et al., 2012; MacArthur & Wilson, 1967). Fragments are therefore very sensitive to external fluctuations, causing the smallest change in local management practices to drive a fragmented ecosystem in a different direction (Laurance et al., 2011). Events such as windstorms and droughts strongly influence the ecology of fragments (Laurance et al., 2011), which make South African farm fragments even more fragile due to the country’s semi-arid climate and susceptibility to drought. Indeed, arid and semi-arid ecosystems are especially sensitive to fragmentation which impacts vegetation heterogeneity which in turn influences population dynamics (Hobbs, Reid, Galvin, & Ellis, 2008). Ultimately, the smaller the area the more ecosystems dynamics are driven by external factors, such as radiation and water fluxes, rather than internal factors, such as population dynamics (Saunders et al., 1991). Small and isolated populations are fragile and inclined to extinction, and the loss of keystone species alters the species interactions causing further local extinctions (see Terborgh et al., 2001). Some species may even be absent from small fragments due to the area being smaller than the minimum area needed for a self-sustaining population (Bennett & Saunders, 2011). For example, by increasing predator-proof fencing to protect valuable antelopes, South African wildlife ranchers are inadvertently reducing the habitat availability of free-ranging populations of species, such as wild dog and cheetah *Acinonyx jubatus* (Lindsey et al., 2012; Taylor et al., 2015).

Biodiversity can only truly exist in large ecosystems where the complete variety of wildlife occur and cannot properly exist in small fenced-off, predator-free, intensive wildlife production units (du Toit, 2010a). Small fenced camps that are intensively managed are inherently overstocked with high wildlife densities which may result in overgrazing, depressed primary production, decreased carrying capacity, and consequently habitat degradation and population crashes (ABSA Group Economic Research, 2002; Boone & Hobbs, 2004; Gadd, 2012; Hayward & Kerley, 2009; Lindsey et al., 2012; Owen-Smith, 1983). Single small habitat patches cannot fulfil all requirements of a species and cannot support a stable and viable population, forcing animals to adapt behaviourally (Kozakiewicz, 1993). Furthermore, small fenced ranches prevent natural dispersion, emigration, juvenile dispersal, and the immigration of new individuals that would create diversity in local gene pools (Cousins et al., 2008; for example Hunter et al., 2007; Lehmann et al., 2008). A population that is confined by a fence can therefore no longer profit from the total gene pool of the species, and populations on South African game ranches for example are often extremely small (<10 individuals), resulting in the risk of losing genetic diversity (Grobler & van der Bank, 1994). Many wildlife ranches are in fact too small to support large enough predator populations to ensure genetically fit populations (Bothma et al., 2009), therefore it is particularly necessary for ranchers to manage population numbers and predator-prey ratios (Cousins et al., 2008).

Overall, the spatial effects caused by fragmentation can result in a decrease in species richness, changes in species composition and distribution, and can also affect interspecific interactions (Kozakiewicz, 1993). Landscape spatial structure, such as habitat patch size, shape and quality, has also been shown to influence population survival and abundance (Fahrig & Merriam, 1994) and is deemed vital to a collection of ecological functions (Reid et al., 2017). Indeed, the size and shape of a wildlife ranch influence its economics, management, utilization, and ecological integrity (du Toit & van Rooyen, 2010). Macdonald and Brooks (1983) suggest that small reserves (less than 1000 km^2) require special management interventions as they are subjected to a single climatic regime, have a smaller range of habitat types, contain fewer mammalian predators, are only able to support small populations of large mammals, and cover only a portion of migratory area of some mobile mammal species. Ultimately, the smaller the area the more intensively it has to be managed (Bothma, 2010).

2.7 Costs and Management of Fencing

Ownership of wildlife is made clear with fencing, however fencing is expensive and cause landscape fragmentation (Child, 2009). During the 2000s basic fencing in South Africa had been estimated to cost more than R30 000 per km (van Rooyen, du Toit, & van Rooyen, 2010), and electrical fencing had cost as much as R0.2 million to R2.2 million in total, depending on the size and ecological region of the ranch (ABSA Group Economic Research, 2002). Fence maintenance also proves to be expensive, costing approximately US$32 000 per year for a reserve with 100 km of fencing (Masterson unpublished data, as referenced by

Lindsey et al., 2012). Hayward and Kerley (2009) conclude that the costs of fencing outweigh the benefits; however Slotow (2012) disagrees suggesting that the costs cannot be equally weighed against the benefits. Moreover, East et al. (2012) argue that the high cost of constructing, maintaining and patrolling a wildlife-proof perimeter fences around the Serengeti National Park would surpass the cost of crop damage or livestock predation caused by wildlife.

The impact of fencing depends on the characteristics of the fence, for example wire fencing is designed to restrict either the movement of mega-herbivores, medium-sizes wild ungulates, or predators (Lindsey et al., 2012). The height and security level of fences are also dependent on the species present which are grouped as ‘Class 1’ species (large animals such as kudu *Tragelaphus strepsiceros* and giraffe *Giraffa camelopardalis*) which require 2.4 m tall fences, and ‘Class 2’ species (medium-sized animals such as sable antelope and bushbuck) which require 1.4 m tall fences (Snijders, 2012). The effectiveness of a fence depends on whether it is completely operational (Slotow, 2012) and thus hinges on its maintenance and management (Massey, King, & Foufopoulos, 2014). Fence maintenance is affected by natural factors, such as flooding, animals causing damage, theft, and vandalism (Grant, 2008). Furthermore, the choice of an effective fence depends on the nature of the terrain, the type and availability of material, and the available finances (van Rooyen et al., 2010). Ideally fences should be able to persist for a long period of time (Dickman, 2012), though on average the expected life span of fences is 20 to 30 years depending on the climate, quality of materials and fire frequency (van Rooyen et al., 2010). Fences should also be as permeable as possible, preventing only movement of the target species, so as to allow non-target species to move freely (Slotow, 2012). Studies on such fences have been conducted by South African National Parks (SANParks), for instance the twelve year study on large herbivore exclosures in the Kruger National Park (Botha & Siebert, 2014). The SANParks study excluded elephant and giraffes from small areas with fencing, allowing all other herbivores access, which resulted in increases in herbaceous biomass but decreases in species richness and diversity after only five years (Botha & Siebert, 2014). The study also revealed the challenges of fence maintenance, as animals managed to break the fences during the drier periods of the year resulting in the need to chase the animals out using a helicopter and then facing expensive fence repairs (Botha & Siebert, 2014). Animal break-outs are inevitable as fencing an area securely is challenged by the extent of the fence, damage to the fence caused by animals, uneven and remote terrain, and rivers and streams entering the area (Symonds & Swemmer, 2013).

Fences are therefore not easily nor readily monitored and maintained. Gadd (2012) conducted a meta-analysis on 34 reports on the effects of veterinary cordon fences and pointed out that most were hindered by political pressures as fence construction, maintenance and assessment were often rushed or done without professional consultation. Furthermore, there is a general lack of research and literature on fencing, especially regarding its ecological impacts (Jakes et al., 2018; Lindsey et al., 2012). Large gaps exist in empirical research on wildlife-fence interactions and fence systems, with most studies being limited in scope and focusing on single species at local spatial scales (Jakes et al., 2018). Research is also lacking in how often mammals attempt to cross fences, individual behaviour before and after fence construction (Gadd, 2012), how animals perceive, physically negotiate, and habituate to fences, and the cumulative stress from crossing multiple fences (Jakes et al., 2018). More research is especially needed on indirect effects, such as how fencing influences vegetation communities, soil properties, nutrient cycling, and ecosystems in general (Jakes et al., 2018). There is also a deficiency in understanding the influence of human and social capital on landscape connectivity, even though it is recognised that ecological connectivity can be improved with social connectivity (Knight & Cowling, 2012). Several European countries belonging to the Infra Eco Network Europe (IENE) have realised that habitat connectivity is essential for sustaining animal movement across a landscape, and have taken the initiative to plan and construct their infrastructure in such a way as to accommodate wildlife (Bekker & Iuell, 2003). Fortunately, some South African wildlife ranchers have amended their connectivity with one another by combining their properties into conservancies (Bothma, 2010), which are reported to have long-term benefits (Lindsey et al., 2009) such as being more ecologically viable (Bothma, 2010). Results from Cousins et al. (2008), shows that a number of interviewees suggested fence removal and thus the aggregation of properties in order to lessen the impact of ranching and intensive management on wildlife. However, fencing is increasing in southern and eastern Africa (Desmet & Pillay, 2016; Løvschal et al., 2017), despite the call to removes fences. It may be due to the wildlife sector community lacking information on the benefits of forming conservancies (Lindsey et al. 2009), and there being no legal requirements for or regulation of internal fences of game farms at present (Desmet & Pillay, 2016). Furthermore, there are no formal national guidelines for the design of electrified game fences in South Africa (Beck, 2008).

2.8 Conclusion

By reviewing the wildlife sector as a land-use, exploring the past and present of the wildlife sector, highlighting the impacts of intensive wildlife production and the existing knowledge gaps on the topic, and examining the effects and implications of fencing, it is clear that further in-depth research is needed to improve wildlife management overall. The relationship between science and wildlife ranching management should be improved as the wildlife sector is largely tourist-driven – which encourages habitat manipulation and the introduction of extra-limital and exotic species – and often has inadequate professional conservation management and planning (Cousins et al., 2008).

Numerous recommendations and plans aimed at improving wildlife management and reducing the effects of fencing and fragmentation have been made over the years (Beck, 2008; Bekker & Iuell, 2003; BirdLife South Africa, 2018; Gadd, 2012; Paige, 2008; Taylor et al., 2015). Taylor et al. (2015) recommend the following objectives: provincial nature conservation departments need to monitor and manage all permitting requirements through a centralised, electronic permitting system; the impacts of wildlife ranching management practices on biodiversity need to be clarified; consensus on owning extra-limital and non-indigenous species should be established; more research on the occurrence and transmission of diseases in wildlife species is needed; the decision whether wildlife ranching should remain the mandate of DEA or move under DAFF should be made; and larger conservation areas should be created through joining wildlife ranches. Gadd (2012) suggests the following: an accurate, centralised, spatially explicit database of all fences (containing fence types and conditions) to understand the current fence network and prioritise future construction and removals; frequent monitoring of fences to better understand mortality patterns and wildlife movements; and removal of fences or installation of cattle grids where disease-control trade-offs are reasonable. Beck (2008) recommends altering fencing designs to reduce wildlife mortalities, such as increasing the height of the bottom electrified strand and increasing the distance that the lowest electrified strand is offset from the main fence. To minimise the impact of fences on birds, BirdLife South Africa (2018) suggests the following mitigations: remove all non-essential fences; replace the top two barbed strands with smooth wire; routinely re-tension loose wires; increase spacing between strands to a minimum of 30 cm; make fences more visible with markers; and reduce the barrier-effect during livestock rotation. Other countries have taken steps to minimise the impact of fencing and fragmentation, such as the previously mentioned European countries that plan and construct their infrastructure to allow wildlife movement (Bekker & Iuell, 2003). For example, the United States of America has integrated wildlife friendly fencing, which allow for safe passage, increased fence visibility, healthier habitats, and improved access to resources (Paige, 2008).

The extent and effects of fencing should be researched extensively with appropriate data collection tools. By improving our understanding of the degree and implications of fencing and fragmentation in South Africa, the above recommendations may be applied with more support and certainty. Alternative management strategies that consider practical and financial constraints should also be brought forward and implemented with proper monitoring programmes.

Chapter 3:

Assessing Changes in Fencing of the Wildlife Sector in South-West Limpopo through Remote Sensing

3.1 Introduction

The wildlife sector is an important economic and ecological asset of South Africa that quickly expanded as an agricultural enterprise over the past few decades (Grossman et al., 1999; Reilly et al., 2003; The National Agricultural Marketing Council, 2006; van der Waal & Dekker, 2000). The term 'wildlife sector' is used here as a collective term for 'wildlife- or game farming or -ranching'. Since the 1980s there has been a rapidly growing trend (Grossman et al., 1999) where privately owned South African livestock and crop farms have been transformed into game farms (Kamuti, 2014). In South Africa, the Department of Agricultural Development formally recognised wildlife ranching as an agricultural activity in 1987 (van Hoven, 2015), and in 1991 the Game Theft Act, Act No. 105, was implemented (Nel, 2015) which required landowners to obtain a 'Certificate of Adequate Enclosure' (CAE) to legally own game on their property (Snijders, 2012; Taylor et al., 2015).

Game farms generate income through selling meat and live animals, and providing hunting and game viewing opportunities (Child et al., 2012; Luxmoore, 1985), of which profits increased from 5% to 20% per year in the 2000s (Child et al., 2012). Taylor et al. (2015) estimated the total revenues generated in 2014 by live game sales in South Africa to be R4.33 billion, trophy hunting R1.96 billion, biltong hunting R650 million, and game meat R349.7 million. Furthermore, there has been a 40-fold increase in the number of wildlife in both private and public conservation areas from the early 1960s (van Hoven, 2015), with reports suggesting that there are over 9000 wildlife ranches in South Africa, covering an area of more than 200 000 km^2 and containing 16 to 20 million wild animals (Taylor et al., 2015). Privately owned wildlife ranches are particularly important as they have been reported to protect almost three times more land than government protected areas (Bothma & von Bach, 2010; Child et al., 2012; Cousins et al., 2008; The National Agricultural Marketing Council, 2006), and contribute to South African ecotourism, economy and conservation (Child et al., 2012; Taylor et al., 2015; van der Merwe & Saayman, 2003). However, despite the benefits of the growing wildlife sector, its true extent and ecological impacts are mostly unknown and under-researched, which may consequently disparage some of the benefits (Bothma & von Bach, 2010; Child, 2009; Lindsey et al., 2009; Taylor et al., 2015).

It has been argued that wildlife ranches essentially have limited conservation value due to their commercial nature (Cousins et al., 2008) and several ranching practices are in conflict with conservation principles (Cousins et al., 2010). Due to the industry being driven by tourist and hunter preferences, wildlife farmers have introduced exotic and extra-limital species (Cousins et al., 2008; Luxmoore, 1985), generated colour variants (rare colour phenotypes) and hybrids through selective breeding (Cousins et al., 2010; Lindsey et al., 2009; Taylor et al., 2015), removed predators to protect valuable game (Cousins et al., 2010), and altered landscapes by modifying vegetation structure (Sims-Castley et al., 2005) and constructing artificial waterholes (Child, 2013). All of which can have negative consequences to wildlife and their environment. Wildlife ranching practices have become more intensive in order to breed high-value popular game species (>R10 000 per animal, e.g. sable antelope *Hippotragus niger*) (Pitman et al., 2017). Intensive wildlife farming can be defined as an agricultural system where wildlife is maintained in fairly small, fenced areas with food and water in order to harvest by-products (Carruthers, 2008). Intensive wildlife practices include harvesting animal products, for example ostrich *Struthio camelus* and crocodile *Crocodylus niloticus* farms (Barnes 1998, 2001), as well as breeding and protecting threatened species, such as roan antelope *Hippotragus equinus* (Dorgeloh et al., 1996) and black rhinoceros *Diceros bicornis* (Bulte & Damania, 2005). Furthermore, intensive breeding usually produce 'superior' animals for live game sales or trophy hunting by manipulating breeding conditions to select animals for desirable traits, such as long horns, large body size or multiplication of a colour variant (Taylor et al., 2015). Intensive wildlife practices can result in a number of genetic, behavioural, and ecological consequences. For instance, it can lead to the loss of genetic fitness, hybridisation, homogenisation, behavioural changes, semi-domestication, enhancement of pathogen spread and susceptibility, habitat loss, and augmentation of fragmentation by increasing the use of small fenced camps (Child, 2013; Desmet & Pillay, 2016; Nel, 2015).

The wildlife sector is accompanied by fencing which is legally required to own wildlife in South Africa (Child et al., 2012; Cousins et al., 2010; Lindsey et al., 2012; van Hoven, 2015) due to the Game Theft Act (Nel, 2015). Fences had extended over almost 80 000 km^2 in South Africa by the year 2000 (Hearne & McKenzie, 2000) and are increasing exponentially across Africa (Gadd, 2012). Fences benefit wildlife farmers and wildlife management in general by protecting game populations, establishing ownership and authority over game, controlling access, reducing human-wildlife conflict, decreasing the spread of disease, and establishing boundaries (Boone & Hobbs, 2004; Gadd, 2012; Grobler & van der Bank, 1994; Hayward & Kerley, 2009; Hoare, 1992; Lindsey et al., 2012; Taylor & Martin, 1987). Additionally, fencing may hinder poaching, act as a firebreak, be used to reintroduce species (with ‘bomas’), serve scientific research purposes, and exclude animals from infrastructure, threatened plant species and roads (Botha & Siebert, 2014; Clevenger et al., 2001; Lindsey et al., 2012; Maschinski et al., 1997; Slotow, 2012).

The presence of fences unfortunately also cause mortality in wildlife, inhibit migration and dispersal, restrict recolonization, stop gene flow, cause inbreeding, limit evolutionary potential, negate behavioural strategies, alter predator behaviour, permit overstocking, exclude wildlife from vital resources, cause local extinction, and lead to habitat degradation (Boone & Hobbs, 2004; Child et al., 2012; du Toit, 2010a; Gadd, 2012; Hayward & Kerley, 2009; Kozakiewicz, 1993; Lindsey et al., 2012; Mbaiwa & Mbaiwa, 2006; Taylor & Martin, 1987). The rise of fences has amplified the fragmentation of ecosystems worldwide (Jakes et al., 2018), particularly in southern Africa due to its important role in wildlife management (Lindsey et al., 2012). Fragmentation divides the landscape into small, disconnected patches which are vulnerable to environmental, demographic and genetic stochasticity (Caughley, 1994; Gadd, 2012; Lindsey et al., 2012; MacArthur & Wilson, 1967). Fragmented, fenced properties are often too small to support stable populations (Kozakiewicz, 1993). This may result in some species, such as large predators, to be completely absent from the fenced area (Bennett & Saunders, 2011). Ultimately, biodiversity mostly cannot persist in small fenced-off, predator-free, intensive wildlife production units (du Toit, 2010a). Fencing does not only have ecological consequences, but financial costs as well. The installation and maintenance of fencing can be very expensive, with electrical fencing installation costing R0.2 million to R2.2 million, depending on the size and biome of the ranch (ABSA Group Economic Research, 2002), and fence maintenance costing approximately R221 000 per year for a reserve with 100 km of fencing (Masterson unpublished data, as referenced by Lindsey et al., 2012). Despite all of the above, there is a deficiency in literature regarding the potential ecological, economical, and social impacts of fencing (Lindsey et al., 2012).

This study aims to infer changes in the wildlife sector, by quantifying the extent of and changes in fences, camps, and intensive wildlife management practices within a wildlife area over a 10 year time period. To do so required manually mapping fences and camps of the wildlife sector with remote sensing data. Satellite remote sensing data is cost-effective, multi-spectral and multi-temporal, and can be transformed into valuable information that can be used for understanding and monitoring land-use patterns and processes (Weng, 2002). It was decided to focus on the Limpopo province in South Africa because it is a popular hunting destination (Warren, 2011), and had the most number of farms with exemption permits (3366), or exempt farms, covering the largest area (5 499 519 ha) than any other province in

2014 (Taylor et al., 2015). The methodology used in this study was greatly influenced by a similar study by Desmet and Pillay (2016) who considered the manual mapping method to produce relatively accurate results given the use of high resolution satellite imagery, and the capability of removing potential misclassifications, such as power lines. However, this study endeavoured to quantify fences and camps using satellite images of a larger section of the wildlife sector of Limpopo, and in doing so infer the extent of and changes in the wildlife sector.

3.2 Materials and Methods

3.2.1 Study Area

The study area is located in the south-west corner of the Limpopo province, South Africa (Figure 3.1). Limpopo is located in north-eastern South Africa, covers 122 305 km^2, consisting mostly of savanna vegetation, and is bound on the north and northeast by the Limpopo river (Reyers, 2004). The south-west corner of Limpopo is part of the Bushveld plateau region and contains a little of the Waterberg mountain range (Reyers, 2004). Interestingly, of the 75 farms in Limpopo surveyed by Taylor et al. (2015), 51% of them conducted ecotourism, 51% practised intensive breeding, 79% had live game sales, 49% managed trophy hunting, 52% organised biltong hunting, and 36% handled both trophy and biltong hunting. Furthermore, the largest percentage of hunters generally hunt in Limpopo due to the province generally harbouring the most preferred hunted species, namely blesbok *Damaliscus pygargus phillipsi*, impala *Aepyceros melampus*, kudu *Tragelaphus strepsiceros*, and blue wildebeest *Connochaetes taurinus* (van der Merwe, Saayman, & Rossouw, 2015; Warren, 2011).

The study area consists of the ecological corridor between Marakele National Park, Atherstone Nature Reserve, Madikwe Nature Reserve, Hans Strijdom Nature Reserve and D'Nyala Nature Reserve in the Waterberg District. A study by Meadows and Hoffman (2002) provided a moderate degradation index for the Thabazimbi and Laphalale local municipal areas in the Waterberg district, which the entire study area consists of, highlighting poor soil and vegetation conditions in the study area. The Thabazimbi district contains mostly privately owned properties that conduct wildlife and cattle ranching (Wilson, 2006), as well as intensive breeding (Desmet & Pillay, 2016). The study site covers an area of approximately 15 080 km^2 and will hereto be referred to as the Limpopo corner.

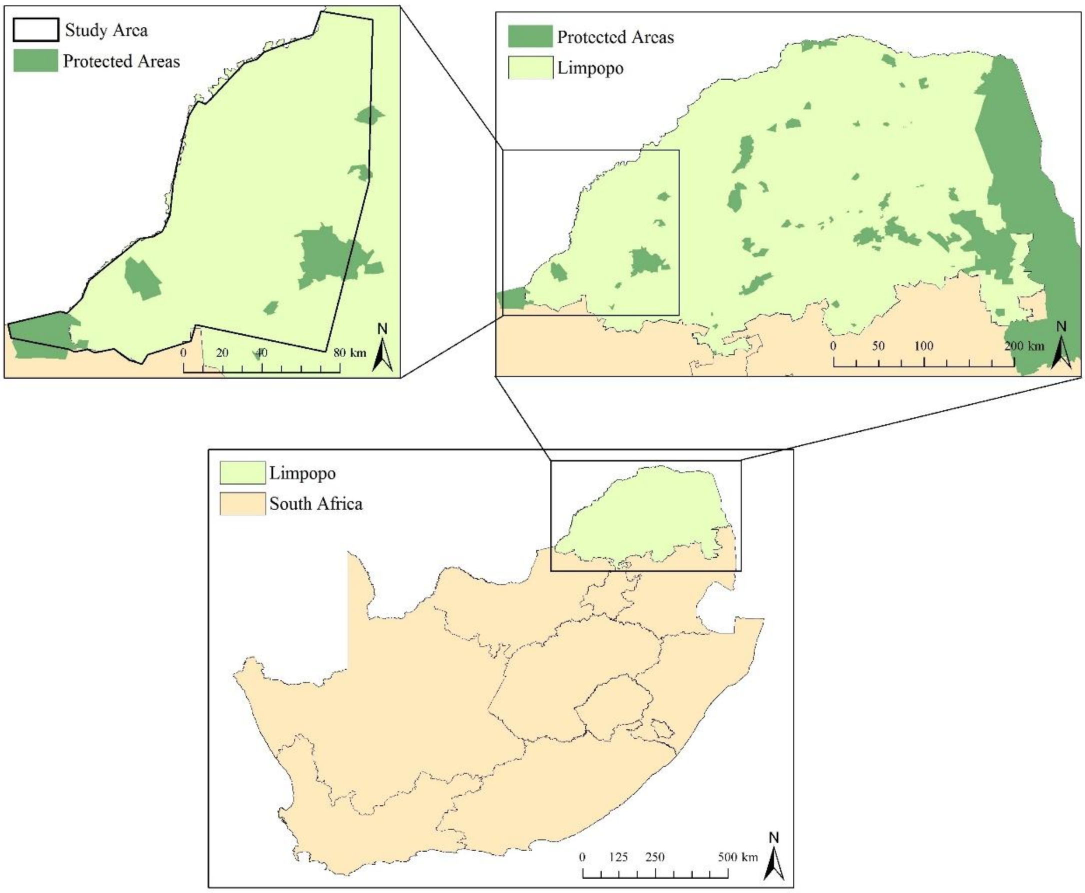

Figure 3.1. Geographical location of the study area in the south-west corner of Limpopo province, South Africa.

3.2.2 Remote Sensing Data

To assess the extent of and changes in the wildlife sector, fences and camps were observed through remote sensing products. The best remotely sensed data are high or moderate resolution images, such as Satellite Pour I'Observation de la Terre (SPOT) (Magee, 2011; Tehrany et al., 2014), which are essential for producing land-cover or land-use information (Topaloğlu et al., 2016). SPOT 5 imagery was obtained for 2007 and 2012 from the South African National Space Agency (SANSA) (https://www.sansa.org.za/), and Sentinel-2 imagery was obtained for 2017 from the European Space Agency (ESA) (https://www.esa.int/ESA). The online databases were filtered so that only satellite images captured between 1 February and 31 May in each respective study year could be obtained, resulting in SPOT 5 images captured between 15 February and 2 May, and Sentinel-2 images captured on 5 April and 5 May (see Appendix: Table A2). The SPOT 5 satellite was operational between 2002 and 2015, had three multispectral bands (green, red and near infrared) of 10 m pixels, and the image swaths were 60 km x 60 km (Boggs, 2010). The Sentinel-2a and Sentinel-2b satellites were launched in 2015, which have 13 multispectral bands with a spatial resolution of 10 m (Topaloğlu, Sertel, & Musaoğlu, 2016), and 110 km x 110 km image swaths, allowing comparisons with the discontinued SPOT 5.

Cadastral data of the Limpopo province were acquired from the Chief Surveyor-General (CSG) office (Pretoria, Gauteng) and were used as a reference for mapping the wildlife sector properties. Land cover data of South Africa were provided by GeoTerraImage (GTI) Pty Ltd. (©GEOTERRAIMAGE – "Southern African Land Cover 2015"), and spatial data of protected areas were obtained from South African National Biodiversity Institute (SANBI) (South African National Parks. Archived NSBA Terrestrial Protected Areas [vector geospatial dataset] 2004. Available from the Biodiversity GIS website, downloaded on 25 Oct 2017).

3.2.3 Data Processing

Fences were manually mapped, using ArcGIS Desktop 10.5 (ESRI, 2016), for 2007 and 2012 with SPOT 5 imagery, and for 2017 with Sentinel-2 imagery. Fence-lines were manually drawn with the help of cadastral and land cover data. Fences were defined as straight lines that were aligned with or ran parallel to Farm Portions and Parent Farms (information obtained from CSG cadastral data). Fence-lines were not mapped in mountainous areas, nor croplands, industrial areas, and urban areas based on the GTI land cover data. Camps were also manually mapped in ArcGIS 10.5 for each study year and were defined as areas that were completely enclosed by fencing and did not comprise of more than a third of cropland.

To quantify changes in the intensive wildlife sector, the number of farm portions (CSG cadastral data) that contained small camps (≤100 ha and ≤50 ha in size) and had irregular fencing patterns (resulting in camps with different shapes and sizes) were counted for each study year. According to Taylor et al. (2015) the mean camp size for intensive breeding properties was 111 ha, while the median camp size was 50 ha. Therefore, it was decided to count both farm portions with ≤100 ha camps and with ≤50 ha camps, hereto referred to as 'medium intensive' and 'highly intensive' camps. The number of intensive farms portions that contained visible waterholes were also counted through Google Earth Pro (Google Inc., 2018), due to waterholes being a characteristic of intensive breeding practices (Taylor et al., 2015). The total lengths of fences (km) and the total number of camps in 2007, 2012, and 2017 were estimated to identify any changes over time. The total, mean, standard deviation (SD), median and range of area (ha) covered by camps in each study year were estimated. To investigate whether there had been a change in camp sizes the camp data were categorised into different camp sizes (<10 ha, 10-50 ha, 50-100 ha, 100-200 ha, 200-500 ha, 500-1000 ha, 1000-1500 ha, >1500 ha), and the number and total area of camps per camp size category were measured. Camp size categories were mostly selected based on the categories used by Desmet and Pillay (2016), as well as the size definitions of intensive breeding camps made by Taylor et al. (2015). Furthermore, a one-way analysis of variance (ANOVA) ($p < 0.05$) compared the following individually to the time intervals (2007, 2012, 2017): fence-length, number of camps, number of camps per size category, mean and median camp area, number of intensive farm portions and the number of intensive farms portions with waterholes (containing medium and highly intensive camps). All statistical analyses were conducted in R version 3.3.3 (R Core Team, 2014) using RStudio version 1.0.136 (RStudio, 2017).

3.3 Results

Based on cadastral data there were 4271 farm portions in the Limpopo corner, covering an area of approximately 1.5 x 10^{11} ha. There was a decrease in extensive and an increase in intensive farm portions, and an increase in intensive farm portions that contained waterholes from 2007 to 2017. The number of farm portions containing medium intensive camps (≤100 ha) increased by 10.7% between 2007 and 2012, by 34.0% between 2012 and 2017, and in total by 48.3% over the entire time period (Table 3.1). The area covered by medium intensive farm portions was 425 806.9 ha in 2007, 466 383.3 ha in 2012, and 607 708.5 ha in 2017. Medium intensive farm portions that contained waterholes increased by 79.0% between 2007 and 2012, by 53.0% between 2012 and 2017, and in total by 173.7%

over the entire time period (Table 3.1). The number of farm portions containing highly intensive camps (≤50 ha) increased by 16.9% between 2007 and 2012, by 22.5% between 2012 and 2017, and in total by 43.1% over the time period (Table 3.2). The area covered by highly intensive farm portions was 224 977.5 ha in 2007, 260 700.9 ha in 2012, and 338 281.4 ha in 2017. Highly intensive farm portions that contained waterholes increased by 93.8% between 2007 and 2012, by 45.2% between 2012 and 2017, and in total by 181.3% over the entire time period (Table 3.2). Overall the biggest increase in the number of farm portions that had intensive wildlife practices and waterholes occurred between 2012 and 2017.

Table 3.1. The number and percentage of farm portions that applied extensive and medium intensive practices (≤100 ha camps), and medium intensive farm portions that contained waterholes in the wildlife sector in south-west Limpopo in 2007, 2012, and 2017.

	No. and % of specific farm portions		
Year	**Extensive**	**Medium intensive**	**Medium intensive with waterholes**
2007	3832 (89.7%)	439 (10.3%)	19 (4.3%)
2012	3785 (88.6%)	486 (11.4%)	34 (7.0%)
2017	3620 (84.8%)	651 (15.2%)	52 (8.0%)

Table 3.2. The number and percentage of farm portions that applied extensive and highly intensive practices (≤50 ha camps), and highly intensive farm portions that contained waterholes in the wildlife sector in south-west Limpopo in 2007, 2012, and 2017.

	No. and % of specific farm portions

Year	Extensive	Highly intensive	Highly intensive with waterholes
2007	4016 (94.0%)	255 (6.0%)	16 (6.3%)
2012	3973 (93.0%)	298 (7.0%)	31 (10.4%)
2017	3906 (91.5%)	365 (8.5%)	45 (12.3%)

There was an increase in the total length of fences (km) in the Limpopo corner from 2007 to 2017 (Table 3.3). From 2007 to 2012, fences totalling 919.7 km were erected, while from 2012 to 2017, fences totalling 1768.3 km were erected, making the latter time period the one with the largest increase in fence length. In sum, 2688.0 km fence was erected between 2007 and 2017, in other words the total fence length increased by 15.9% overall.

Table 3.3. The total length of fences for 2007, 2012, and 2017, and the percentage increase in fence length between these time periods, in the wildlife sector in south-west Limpopo.

Year	Total length of fences (km)	Increase between time periods (%)
2007	16 949.1	
2012	17 868.8	5.4
2017	19 637.1	9.9

The total number of camps increased from 5788 in 2007, to 6384 in 2012, to 7976 in 2017, resulting in increases of 10.3% between 2007 and 2012, 24.9% between 2012 and 2017, and 37.8% between 2007 and 2017. The mean and median area of camps decreased respectively from 171.4 ha and 105.8 ha in 2007, to 155.6 ha and 99.7 ha in 2012, to 124.9 ha and 80.8 ha in 2017 (Figure 3.2). In other words, the mean area (ha) decreased by 9.2% between 2007 and 2012, 19.7% between 2012 and 2017, and 27.1% between 2007 and 2017.

The median area (ha) decreased by 5.7% between 2007 and 2012, 19.0% between 2012 and 2017, and 23.6% between 2007 and 2017. The largest percentages changes regarding number and area of camps occurred between 2012 and 2017. Furthermore, the standard deviation (SD) values were noticeably high (SD_{2007} = 244.9, SD_{2012} = 203.1, SD_{2017} = 168.2), yet showed a decrease over the 10 year period.

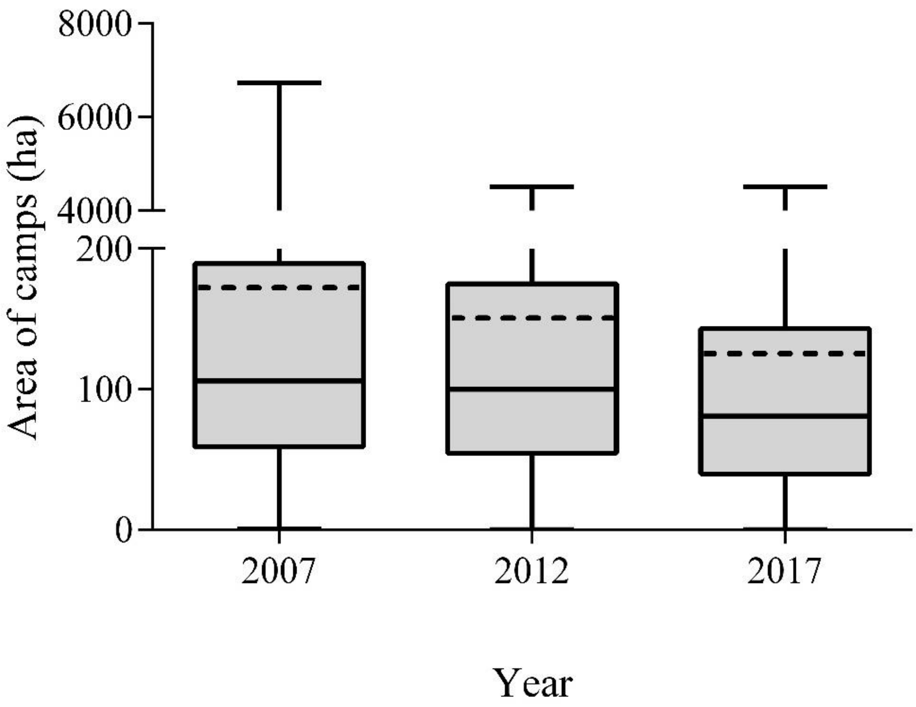

Figure 3.2 Box-and-whisker plots of the area of camps (ha) in the wildlife sector in south-west Limpopo in 2007, 2012, and 2017. Plots contain minimums, first quartiles, medians, means (dashed line), third quartiles, and maximums.

Further investigation discovered that the number of camps smaller than 200 ha increased, whereas the number of camps larger than 500 ha decreased from 2007 to 2017 (Table 3.4). The camp size category 200-500 ha is an exception as the total number of camps increased from 2007 to 2012, but decreased from 2012 to 2017. Over the entire 10 year period (2007-2017), the largest change in number of camps were the <10 ha category with a 202.5% increase, and the 10-50 ha category with a 113.4% increase. Moreover, the number of >1500 ha camps decreased by 45.8% and 1000-1500 ha camps decreased by 28.6% from 2007 to 2017. Most of the camp size changes occurred between 2012 and 2017. For example, the number of <10 ha camps increased by 20.0% from 2007 to 2012 and by 152.1% from 2012 to 2017, and the number of 10-50 ha camps increased by 25.1% from 2007 to 2012 and by 70.6% from 2012 to 2017.

Table 3.4. The total number of camps for each camp size category (<10 ha, 10-50 ha, 50-100 ha, 100-200 ha, 200-500 ha, 500-1000 ha, 1000-1500 ha, >1500 ha) in 2007, 2012, and 2017 in the wildlife sector in south-west Limpopo.

	Camp size category (ha)							
Year	**<10**	**10-50**	**50-100**	**100-200**	**200-500**	**500-1000**	**1000-1500**	**>1500**
2007	120	1021	1606	1697	997	274	49	24
2012	144	1277	1776	1856	1032	237	44	18
2017	363	2179	2226	1963	1011	186	35	13

In conjunction with the number of camps, there was an overall increase in area (ha) covered by camps smaller than 200 ha especially for the smaller camp size category (<10 ha), whereas there was a general decrease in area (ha) covered by camps larger than 500 ha from 2007 to 2017 (Figure 3.3).

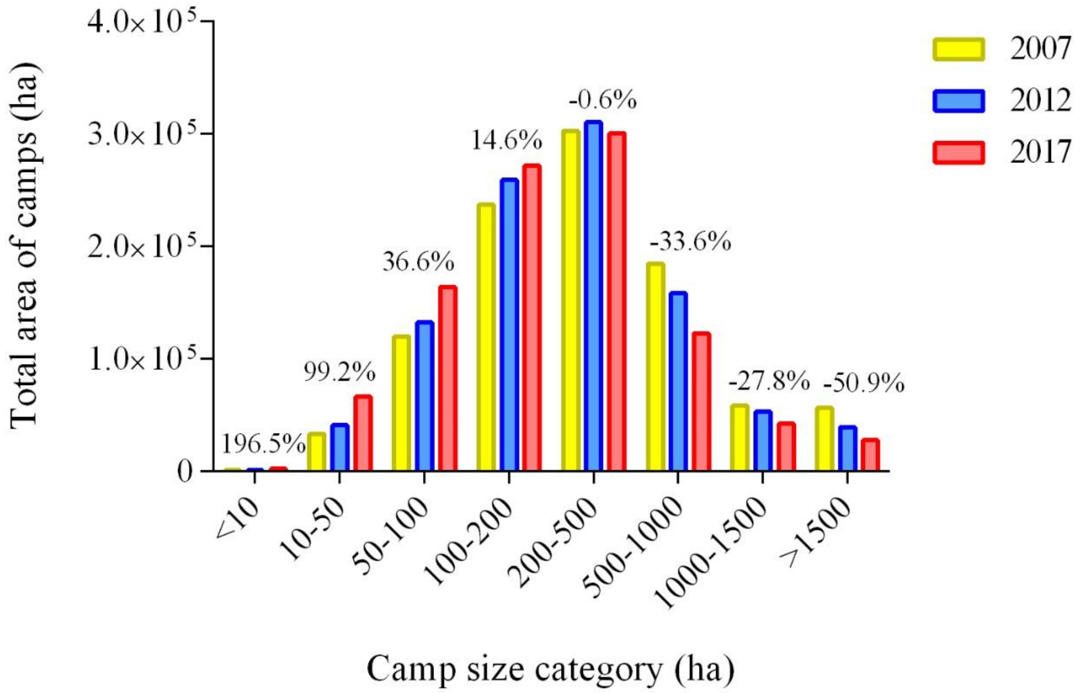

Figure 3.3. The total area of camps for each camp size category (<10 ha, 10-50 ha, 50-100 ha, 100-200 ha, 200-500 ha, 500-1000 ha, 1000-1500 ha, >1500 ha) in 2007, 2012, and 2017, as well as the total percentage change (%) in area from 2007 to 2017, in the wildlife sector in south-west Limpopo.

When fence-length, number of camps, number of camps per size category (all except >1500 ha), mean and median camp area, and number of medium and highly intensive farm portions were compared between 2007, 2012 and 2017 through ANOVA, there were no significant differences ($p > 0.05$) (see Appendix: Table A3). There was a significant difference ($p < 0.05$) between number of camps in the >1500 ha category between years ($F = 363$, $p < 0.03$), as well as in number of medium intensive farms with waterholes ($F = 363$, $p < 0.03$) and highly intensive farms with waterholes ($F = 2523$, $p < 0.01$) between years.

Visual representations of fence increases, camp size decreases, and where these changes occurred from 2007 to 2017, are illustrated in Figure 3.4.

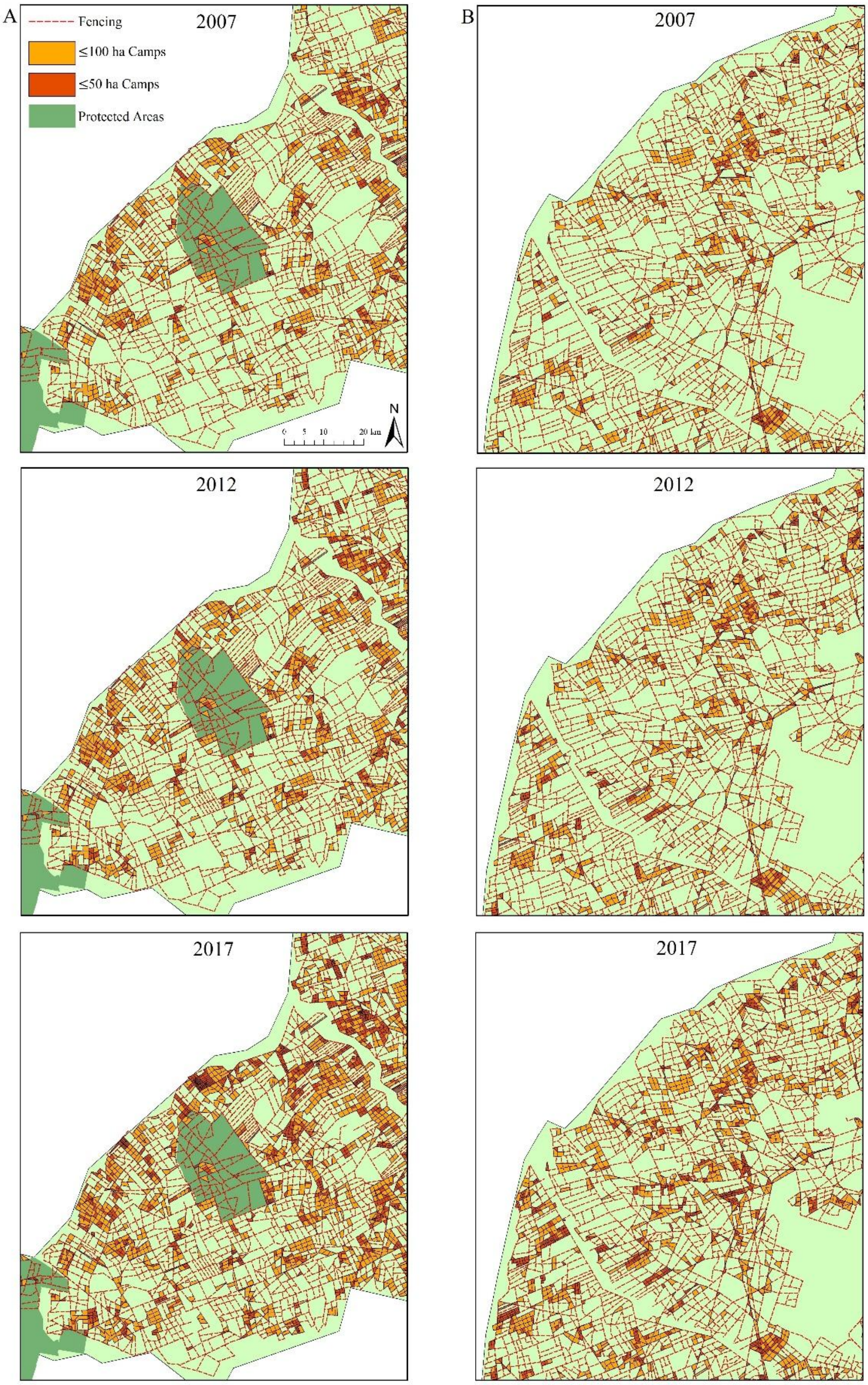

Figure 3.4. Maps depicting the southern (A) and northern (B) regions of the wildlife sector in south-west Limpopo, with manually mapped fences and camps (≤100 ha and ≤50 ha) from 2007, 2012, and 2017, and the protected areas in the regions.

3.4 Discussion

This study attempted to quantify the extent of and changes in fences, camps, and intensive wildlife management practices within the wildlife sector in south-west Limpopo from 2007 to 2017. The fence maps produced for the study area for 2007, 2012, and 2017, indicated increases in intensive wildlife practices, fences, and camps. Intensive wildlife practices have grown from 2007 to 2017 in the wildlife sector in south-west Limpopo through an increase in farm portions containing small camps (≤100 ha and ≤50 ha). The growth in intensive wildlife practices is moreover supported by the increase in intensive farm portions containing waterholes, which is a characteristic of intensive breeding practices (Taylor et al., 2015). Furthermore, increases in fence-length and number of camps, as well as decreases in mean and median camp sizes were observed over the ten year time period in the study area. The decreasing trend in camp sizes of the wildlife sector in Limpopo corner that resulted in a mean of 125 ha and a median of 81 ha in 2017 are especially concerning. If the trend continues, camp sizes would eventually become closer to the mean (111 ha) and median (50 ha) of intensive breeding camp sizes observed by Taylor et al. (2015). It should also be considered that the standard deviations (SD) of camp sizes were high, yet decreased from 2007 to 2017 signifying a slight ‘aggregation’ of the similar sizes particularly within small camps. In addition, the number and area of camps smaller than 200 ha increased, while the number and area of camps larger than 500 ha decreased over the entire time period. The changes in the wildlife sector also seem to be occurring progressively in recent years. Compared to the smaller changes between 2007 and 2012 (e.g. 5.4% increase in fence-length and 10.3% increase in number of camps), the biggest changes occurred between 2012 and 2017 (e.g. 9.9% increase in fence-length and 24.9% increase in number of camps). Even though the changes over the time period are mostly not significant ($p > 0.05$), they may increase in frequency and extent in the near future due to growing intensive wildlife management practices (Pitman et al., 2017), and increase in fencing across southern Africa due to land-use changes and sub-division of land into smaller privately owned fragments (Gadd, 2012).

The manual mapping of fences is a valid method based on high resolution satellite imagery (Desmet & Pillay, 2016) that can be applied to other wildlife sector areas. The method is only limited by the lack of 'ground truth' data, being unable to differentiate different fence types, and potential false positives of removed fences that have similar visual signatures as existing fences (Desmet & Pillay, 2016). Yet, even where fences were removed there may still be long lasting effects, as past land use have been shown to cause legacy effects where ecological impacts persist for centuries and even millennia (Culbert et al., 2017; Dupouey et al., 2002; Foster et al., 2003; Lindborg & Eriksson, 2004). Ultimately, the number of intensive wildlife properties, the total fence-lengths, and the number and sizes of camps in this study are estimates. The values may be influenced by 'fence legacies', resulting in over-estimates, and small fences or extremely intensive camps (for example the 0.2 – 0.5 ha mini bushbuck camps (Nel, 2017)) that are not easily observed, resulting in under-estimates.

The results of this study are nevertheless supported by a corresponding South African study (Desmet & Pillay, 2016), and echoed by a Kenyan study (Løvschal et al., 2017). Similar results were observed by Desmet and Pillay (2016) who manually mapped fences of 208 properties in south-west Limpopo, where intensive breeding properties and total fence length increased the most from 2010 to 2015 than from 2006 to 2010. Likewise, Løvschal et al. (2017) observed an increase in fenced areas in the Greater Mara ecosystem in Kenya, where fences slowly progressed from 1985 to 2014 and thereafter swiftly expanded from 2014 to 2016 In summary, fence expansion seemingly accelerated from 2010 in the study by Desmet and Pillay (2016), from 2014 in the study by Løvschal et al. (2017), and from 2012 in our own study, indicating a certain land use change concerning wildlife in certain African areas around this time. More research would however be needed to ascertain the rapidity, range and reason of the increase in fencing in the wildlife sector. According to current literature, the only study to endeavour to quantify the intensive wildlife sector in the whole of South Africa is that of Taylor et al. (2015) which reported that of 251 wildlife properties in South Africa; 38% bred high-value species, 23% bred colour variants in camps, 16% had non-indigenous species, and 88% had at least one extra-limital species (Taylor et al., 2015). Therefore, there is a definite need for more research on the increase in fencing and intensive management practices in the South African wildlife sector, especially as it may lead to long-lasting detrimental effects on the wildlife and environment.

There are numerous factors that may contribute to the increase in intensive breeding and fencing. Intensive breeding has made wildlife farmers financially more sustainable (Taylor et al., 2015) and more profitable, as illustrated by the colour variant market demand – though it had recently fallen (Thomas, 2017). Intensive breeding practices may further increase due to the recent changes in the Animal Improvement Act (No. 62 of 1998) of South Africa which permits genetic manipulation and cross-breeding of certain wildlife species (Naude, 2016). Ultimately, intensive breeding practices and fencing have several adverse consequences to wildlife and the environment (Gadd, 2012; Hayward & Kerley, 2009; Lindsey et al., 2012; Nel, 2015; Taylor & Martin, 1987). From a broader perspective, the increase in intensive breeding practises and fencing may also cause fragmentation (Desmet & Pillay, 2016; Ferguson & Hanks, 2012; Gadd, 2012; Jakes et al., 2018; Lindsey et al., 2012) which negatively effects wildlife and the entire landscape on a broader spatial and temporal scale. Fragmented ecosystems are also very fragile and may therefore become altered due to relatively slight changes in local management practices (Laurance et al., 2011). Another concern is that the Limpopo corner has been classified with a moderate degradation index

(Meadows & Hoffman, 2002), which suggests that increased fencing may aggravate degradation levels even further. Moreover, as large areas without internal subdivision and human influence are preferred by hunters and tourists (Damm, 2005), the increase in intensive breeding and fencing may negatively affect their views on the wildlife sector to the detriment of its economy.

In order to counteract the detrimental effects of increased intensive breeding, fencing, and fragmentation it would be essential to: increase research efforts on the wildlife sector and fencing, improve the communication between wildlife researchers and the wildlife sector community, enhance the regulation of wildlife management practices and fencing, and form more conservancies through fence removal. There are several gaps in research that should be filled, such as the growth and ecological effects of the wildlife sector (Bothma & von Bach, 2010; Lindsey et al., 2009; Taylor et al., 2015), and the ecological, financial and social impacts of fencing (Jakes et al., 2018; Lindsey et al., 2012). Communication between wildlife researchers and the wildlife sector community should also be strengthened, as the wildlife sector is largely tourist-driven (Cousins et al., 2010) and often lacks professional planning, management and maintenance regarding conservation and fencing (Cousins et al., 2008; Gadd, 2012; Pienaar et al., 2017). Another important resolution would be to regulate fencing, since there are no legal requirements or formal guidelines for fence construction and management in South Africa (Beck, 2008; Desmet & Pillay, 2016). Currently, there are only minimum requirements for fencing where different class species require different fence heights, for example class 1 species such as kudu require a fence height of at least 2.4 m (Brown, Gildenhuys, Hignett, & van Deventer, 2014). Ultimately, the removal of unnecessary fences would be ideal, as in the case with conservancies where internal fences are removed to form larger wildlife farms that are cooperatively managed (Bond et al., 2004). Conservancies have several economic and ecological benefits such as; supporting market demands (Damm, 2005), lower management costs, sustaining whole ecosystems, and increased ecological resilience (Lindsey et al., 2012, 2009). The wildlife sector community is generally uninformed of these benefits (Lindsey et al., 2009), thereby calling for improved distribution of information to the wildlife sector community.

The effects of fencing and fragmentation can be reduced in numerous ways by improving our understanding of it and by refining wildlife management plans and applications (Beck, 2008; Bekker & Iuell, 2003; BirdLife South Africa, 2018; Gadd, 2012; Paige, 2008; Taylor et al., 2015). Taylor et al. (2015) recommend: clarifying the impacts of wildlife management practices on biodiversity; reaching a consensus on the ownership of extra-limital and non-indigenous species; and creating larger conservation areas by joining wildlife ranches. Gadd (2012) suggests: creating an accurate, centralised, spatially explicit database of all fences to understand the current fence network and prioritise future construction and removals; frequent monitoring of fences to better understand mortality patterns and wildlife movements; and removing fences or installing of cattle grids where disease-control trade-offs are reasonable. Altering fencing designs to be more wildlife friendly is also of importance and may result in healthier habitats, improved access to resources (Paige, 2008), and reduced animal mortalities and injuries (Beck, 2008; BirdLife South Africa, 2018). Alternative fencing design include: increasing the height of the bottom electrified strand, increasing the distance that the lowest electrified strand is offset from the main fence, replacing the top two barbed strands with smooth wire; routinely re-tensioning loose wires; increasing the spacing between strands; and making fences more visible with markers (Beck, 2008; BirdLife South Africa, 2018). In addition, the growing intensive wildlife sector should be inspected and regulated with ecological principles in mind.

3.5 Conclusion

This study has documented increases in intensive wildlife properties, total fence-length, and number of fenced camps, as well as a decrease in camp sizes over a ten year period within the wildlife sector in south-west Limpopo. The trends are concerning as they are occurring rapidly and highlight the increasing threat of fragmentation which would have dire effects on the greater landscape. The observed trends may therefore firstly motivate researchers to further investigate the growing intensive wildlife sector, and the proliferation, extent and effects of fencing. It would particularly be beneficial to further monitor the south-west Limpopo wildlife sector to determine whether the trends continue or increase in rate. Furthermore, more studies are needed to confirm whether the pattern is occurring in other parts of the wildlife sector across South Africa. Secondly, the results may be used as a foundation to inform the wildlife sector community of the current trends and the potential consequences thereof. By providing information on the ecological and financial consequences of fencing more available, wildlife farmers may cooperate with researchers and support the implementation of possible solutions. Lastly, the patterns shown in the study may prompt resolution and action, such as creating conservancies (Taylor et al., 2015), producing a fence database (Gadd, 2012), and employing wildlife-friendly fencing (Beck, 2008; Paige, 2008; BirdLife South Africa, 2018). Alternative, practical management strategies should be proposed and implemented with proper monitoring programmes. Ultimately, reducing intensive breeding practices, creating conservancies and implementing fencing regulations would be the most significant contribution for sustainable wildlife management.

Chapter 4:

Image Classification as a Method to Map Fences of the Wildlife Sector in South-West Limpopo

4.1 Introduction

The wildlife sector in South Africa has expanded rapidly as an agricultural enterprise over the past few decades (Grossman et al., 1999; Reilly et al., 2003; The National Agricultural Marketing Council, 2006; van der Waal & Dekker, 2000), with recent reports indicating that there are over 9000 wildlife ranches that contain 16 to 20 million wild animals (Taylor et al., 2015). The term ‘wildlife sector’ is used here as a collective term for ‘game’ or ‘wildlife’ ‘farming’ or ‘ranching’. The wildlife sector proves to be important for conservation, especially privately owned wildlife ranches which have been reported to protect almost three times more land than government protected areas (Bothma & von Bach, 2010; Child et al., 2012; Cousins et al., 2008; The National Agricultural Marketing Council, 2006). Furthermore, game farms generate income through selling meat, hunting, live animal sales, and game viewing (Child et al., 2012; Luxmoore, 1985), and is considered to be an important economic aspect of South Africa (Saayman et al., 2011). It has been debated that wildlife ranches have limited conservation value due to their commercial nature (Cousins et al., 2008), however a detailed assessment of the social, economic and conservation value of the wildlife ranching industry is lacking (Taylor et al., 2015). Several ranching practices are in conflict with conservation principles (Cousins et al., 2010), such as introducing exotic and extra-limital species (Cousins et al., 2008; Luxmoore, 1985), selectively breeding colour variants and hybrids (Cousins et al., 2010; Lindsey et al., 2009; Taylor et al., 2015), and trapping or hunting predators to protect valuable game (Cousins et al., 2010). Wildlife farmers also tend to modify vegetation structure (Sims-Castley et al., 2005), for example by constructing artificial waterholes, which ultimately affects habitat (Child, 2013). Furthermore, wildlife ranching practices have become more intensive due to the high demand for high-value popular game species (>R 10 000 per animal, e.g. sable antelope *Hippotragus niger*) (Pitman et al., 2017). Intensive game farming can be defined as an agricultural system where wildlife is maintained in fairly small, fenced areas with food and water in order to harvest by-products (Carruthers, 2008). Intensive wildlife practices can be used to harvest animal products (Barnes, 2001, 1998), breed and protect endangered species (see Bulte & Damania, 2005; Dorgeloh et al., 1996), or produce 'superior' animals for live game sales or trophy hunting through selective breeding (Taylor et al., 2015). Intensive wildlife practices can result in a number of genetic, behavioural, and ecological consequences. For instance, it can lead to the loss of genetic fitness, hybridisation, homogenisation, behavioural changes, semi-domestication, enhancement of pathogen spread and susceptibility, habitat loss, and fragmentation (Child, 2013; Desmet & Pillay, 2016; Nel, 2015).

The wildlife sector is accompanied by kilometres of fences, due to fencing being a legal requirement to own wildlife in South Africa (Child et al., 2012; Cousins et al., 2010; Lindsey et al., 2012; van Hoven, 2015) on account of the Game Theft Act, Act No. 105 (Nel, 2015) which obligates landowners to acquire a 'Certificate of Adequate Enclosure' (CAE) (Snijders, 2012; Taylor et al., 2015). Fences essentially form boundaries which establish game ownership, protect wildlife, decrease human-wildlife conflict, and reduce the spread of disease (Boone & Hobbs, 2004; Gadd, 2012; Grobler & van der Bank, 1994; Hayward & Kerley, 2009; Hoare, 1992; Lindsey et al., 2012; Taylor & Martin, 1987). In addition, fencing may be used to restrict poaching, act as a firebreak, and reintroduce species with 'bomas' (Lindsey et al., 2012). Fencing can also be used to exclude animals from roads to mitigate wildlife-vehicle collisions (Clevenger et al., 2001). Wildlife can also be excluded from certain areas through fencing to protect infrastructure or threatened plant species (Maschinski et al., 1997; Slotow, 2012), or to serve scientific research purposes (Botha & Siebert, 2014). Unfortunately, fences also obstruct migration and dispersal, limit recolonization, and stop gene flow, which lead to inbreeding where populations are not managed appropriately (Boone & Hobbs, 2004; du Toit, 2010a; Gadd, 2012; Hayward & Kerley, 2009; Kozakiewicz, 1993; Lindsey et al., 2012; Mbaiwa & Mbaiwa, 2006; Taylor & Martin, 1987). Fences can furthermore lead to direct mortality in wildlife, restrict evolutionary potential, negate behavioural advantages, change predator behaviour, and lead to habitat degradation (Boone & Hobbs, 2004; Child et al., 2012; Gadd, 2012; Hayward & Kerley, 2009; Lindsey et al., 2012; Mbaiwa & Mbaiwa, 2006; Taylor & Martin, 1987). Fencing is also associated with fragmentation which divides the landscape into small, isolated patches that are vulnerable to environmental, demographic and genetic stochasticity (Caughley, 1994; Gadd, 2012; Lindsey et al., 2012; MacArthur & Wilson, 1967). Fragmented wildlife areas are also often too small to support stable populations (Bennett & Saunders, 2011; Kozakiewicz, 1993). The expansion of fences has augmented the fragmentation of ecosystems globally (Jakes et al., 2018), especially in southern Africa where fencing has increased exponentially due to land-use changes and land sub-division (Gadd 2012). Despite the prevalence of fencing in southern Africa, there is little information on the ecological, financial and social effects of fencing as a wildlife management tool (Lindsey et al., 2012). There is also a lack of empirical research on wildlife-fence interactions and fence systems (Jakes et al., 2018). Therefore it is imperative to bolster research efforts regarding the extent and effects of fencing in the wildlife sector.

In order to assess the extent of and changes in the growing wildlife sector, fences and camps were manually mapped and quantified in this study based on remotely sensed data (as described in Chapter 3). However, an alternative remote sensing method was pursued in order to ‘automate’ fence mapping by means of image classification. Satellite remote sensing data are cost-effective, multi-spectral and multi-temporal, and can be transformed into valuable information that can be used for producing land-use and land cover datasets (Weng, 2002). The term ‘classification’ is used in the remote sensing community to define the process that allocates single pixels to a set of classes (Camps-Valls, Tuia, Bruzzone, & Benediktsson, 2014) in an image or remotely sensed satellite data. There are various remote sensing classification techniques and algorithms, namely supervised or unsupervised; parametric or non-parametric; hard or soft (fuzzy) classification; and pixel-based or object-based (Duro, Franklin, & Dubé, 2012; Keuchel, Naumann, Heiler, & Siegmund, 2003; Lu & Weng, 2007). Several classification algorithms have been developed for the study of remotely sensed data and land cover classification, such as K-Nearest Neighbour, Support Vector Machines, Decision Trees, Random Forest, Maximum Likelihood, and Neural Networks (Akar & Güngör, 2012; Lu & Weng, 2007; Tehrany, Pradhan, & Jebuv, 2014).

A large assortment of classification algorithms have been used to map land-cover and land-use from remotely sensed data (Rogan et al., 2008). The best remotely sensed data are images of high or moderate spatial resolution, for example Satellite Pour I'Observation de la Terre (SPOT), High Resolution Visible (HRV), IKONOS, Quickbird and Landsat Mapper (TM) / Enhanced Thematic Mapper plus (ETM+) (Magee, 2011; Tehrany et al., 2014). In addition to satellite images, image classification has been successfully applied to Normalised Difference Vegetation Index (NDVI) data as well (for example Lisita et al., 2013; Zheng et al., 2015). The NDVI is a satellite-based vegetation index, ranging from -1 to +1, that derives from the red:near-infrared (RED:NIR) reflectance ratio [NDVI = (NIR – RED) / (NIR + RED)] (Pettorelli et al., 2005). Normalised Difference Vegetation Index measures have been shown to correspond with visual estimates of greenness (Parrini, Macindoe, & Erasmus, 2013), and correlate strongly with aboveground net primary productivity (Pettorelli et al., 2005), rainfall patterns and seasonality (Anyamba & Tucker, 2005; de Jong, de Bruin, de Wit, Schaepman, & Dent, 2011). The NDVI is a useful tool that can be coupled with climate, vegetation and animal distribution at large spatial and temporal scales (Pettorelli et al., 2005), and has been used for many purposes such as assessing land-cover change (Hüttich, Herold, Schmullius, Egorov, & Bartalev, 2007), desertification (Symeonakis & Drake, 2004), and landscape degradation (Holm, Cridland, & Roderick, 2003). For the present study, image classification was performed based on the NDVI derived from both SPOT 5 and Sentinel-2 imagery, which have matching pixel resolutions and timeframes. The SPOT 5 satellite was operational between 2002 and 2015, had three multispectral bands (green, red and near infrared) of 10 m pixels, and the image swaths were 60 km x 60 km (Boggs, 2010). The Sentinel-2a and Sentinel-2b satellites were launched in 2015, which have 13 multispectral bands with a spatial resolution of 10 m (Topaloğlu et al., 2016), and 110 km x 110 km image swaths, allowing comparisons with the discontinued SPOT 5.

For this study two image classification methods were used, namely Support Vector Machine (SVM) and Random Forest (RF). Both methods are based on machine learning, which is the science of computer modelling of learning processes (Huang & Jensen, 1997). Support Vector Machine is a supervised, non-parametric statistical learning technique developed by Vapnik (Vapnik, 1995) that assumes that the feature data are linearly separable (Mountrakis, Im, & Ogole, 2011). A supervised classification method requires the analyst to define small areas (called training sites) on the image, which contain the predictor variables measured in each sampling unit, and to assign prior classes to the sampling units (Cerna & Chytry, 2005). As a non-parametric method, no assumptions about the data are necessary (Lu & Weng, 2007). Support Vector Machines have very high generalization capabilities and thus require only a small number of training samples (Belousov, Verzakov, & von Frese, 2002; Fauvel, Chanussot, & Benediktsson, 2006). As a result, SVM methods have been shown to generally outperform other image classification methods (Bazi & Melgani, 2006; Camps-Valls et al., 2014; Foody & Mathur, 2004; Hermes, Frieauff, Puzicha, & Buhmann, 1999; Huang & Jensen, 1997; Melgani & Bruzzone, 2004; Pal & Mather, 2005). Random Forest was developed by Breiman (Breiman, 2001) and is also recognised to be one of the most proficient classification methods (Akar & Güngör, 2012). The RF classifier is a non-parametric classifier that uses a collection of tree-structured classifiers, or decision trees, that are generated through the Classification and Regression Tree (CART) algorithm (Akar & Güngör, 2012; Breiman, 2001). In standard decision trees each node is split based on the best split among all variables, whereas in a random forest each node of a tree is split based on the best subset of predictors randomly chosen at that node (Liaw & Wiener, 2002). Random Forest is categorised as an ensemble classification method, which is a learning algorithm that constructs a set of classifiers instead of one classifier (Akar & Güngör, 2012). With RF classification, pixels are allocated to each class based on a majority voting rule applied by the group of classification trees (Zhu, Woodcock, Rogan, & Kellndorfer, 2012). The classification accuracy of RF has been shown to be higher (Akar & Güngör, 2012) as well as on par with that of SVM (Pal, 2005). In general, both the classification algorithms, SVM and RF, perform well and are nearly equal to one another in practice (Wieland & Pittore, 2014).

The aim of this chapter is test these methods to automate the mapping of fences in the wildlife sector using image classification. Exploring new methods to mapping fences may expand the understanding of landscape challenges and improve data availability for wildlife researchers and managers. According to current literature, no endeavour has been made by the scientific community as of yet to produce an image classification method to specifically map fences in South Africa. Therefore, this chapter took a ‘trial-and-error’ approach for this novel study, and focused on how accurately image classification methods can automate the mapping of fences.

4.2 Materials and Methods

4.2.1 Study Area

The study area is located in the south-west corner of the Limpopo province, South Africa (Figure 4.1) and is identical to the area described in Chapter 2. Limpopo is located in the north-eastern South Africa, covers 122 305 km^2, consisting mostly of savanna vegetation, and is bound on the north and northeast by the Limpopo river (Reyers, 2004). The south-west corner of Limpopo is part of the Bushveld plateau region and contains a little of the Waterberg mountain range (Reyers, 2004). The province is an ideal area to study the wildlife sector, as it is a popular hunting destination (Warren, 2011) and had the most number of exempt farms (3366) covering the largest area (5 499 519 ha) than any other province in 2014 (Taylor et al., 2015).

The study area chapter consists of the ecological corridor between Marakele National Park, Atherstone Nature Reserve, Madikwe Nature Reserve, Hans Strijdom Nature Reserve and D'Nyala Nature Reserve in the Waterberg District. A study by Meadows and Hoffman (2002) presented a moderate degradation index for the Thabazimbi and Lephalale local municipal areas in the Waterberg district, which the entire study area consists of, emphasising poor soil and vegetation conditions in the study region. The Thabazimbi district contains mostly privately owned properties that conduct wildlife and cattle ranching (Wilson, 2006), as well as intensive breeding (Desmet & Pillay, 2016). The study site covers an area of approximately 15 080 km^2 and will hereto be referred to as the Limpopo corner.

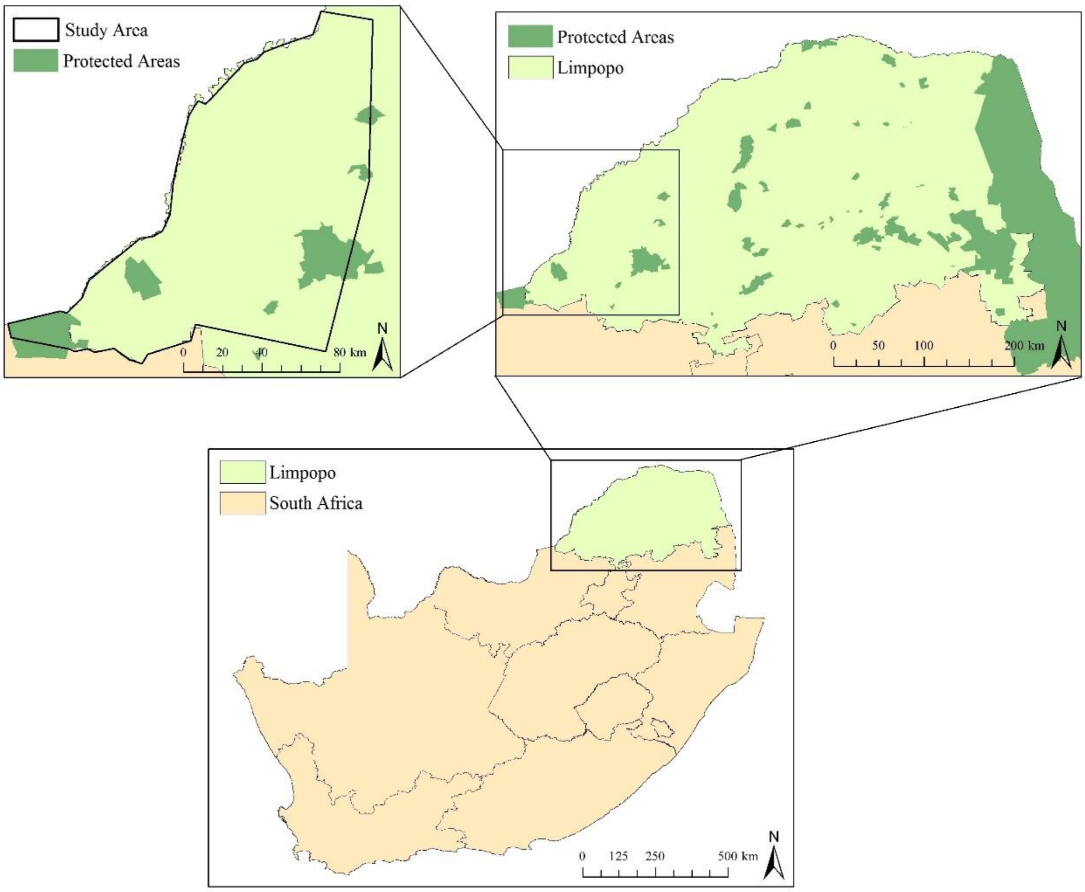

Figure 4.1. Geographical location of the study area in the south-west corner of Limpopo province, South Africa.

4.2.2 Data

The data as described in Chapter 3 was employed in this chapter as well, namely the SPOT 5 imagery provided by the South African National Space Agency (SANSA) (https://www.sansa.org.za/) (15 February to 2 May, 2007 and 2012), the Sentinel-2 imagery provided by the European Space Agency (ESA) (https://www.esa.int/ESA) (5 April to 5 May, 2017) (see Appendix: Table A2), Chief-Surveyor General (CSG) cadastral data (Pretoria, Gauteng office), South African land cover data (©GEOTERRAIMAGE – "Southern African Land Cover 2015"), and protected area spatial data (South African National Parks. Archived

NSBA Terrestrial Protected Areas [vector geospatial dataset] 2004. Available from the Biodiversity GIS website, downloaded on 25 Oct 2017). In addition elevation data was obtained for the study area, namely Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) Global Digital Elevation Model (GDEM) data. A total of four ASTER GDEM images (60 x 60 km) were retrieved from the online EarthExplorer tool, courtesy of the National Aeronautics and Space Administration (NASA) Earth Observing System Data and Information System (EOSDIS) Land Processes Distributed Active Archive Center (LP DAAC), United States Geological Survey (USGS) / Earth Resources Observation and Science (EROS) Center, Sioux Falls, South Dakota [https://earthexplorer.usgs.gov/]. ASTER GDEM is a product of NASA and Ministry of Economy, Trade, and Industry (METI).

4.2.3 Image Classification Methodology

The mapping of fences was automated through two image classification methods, namely Support Vector Machine (SVM) and Random Forest (RF), using ArcGIS Desktop 10.5 (ESRI, 2016). In ArcGIS 10.5, the classifier tools are named ‘Support Vector Machine Classifier’ and ‘Random Trees Classifier’, respectively. ‘Random Trees’ may also be referred to as ‘Random Forests’ (Wieland & Pittore, 2014) which is technically a multitude of random trees. In order not to create confusion, the term Random Forest (RF) will still be used throughout this study. Both methods are categorised as supervised, non-parametric image classification methods based on machine learning.

Firstly the SPOT 5 images (total: 16) and Sentinel-2 images (total: 5) were imported into ArcGIS 10.5 and categorised by their year (2007, 2012 or 2017). Each satellite image was converted into a Normalized Difference Vegetation Index (NDVI) image through the ‘Image Analysis’ tool which is able to define the red and near-infrared bands needed to calculate the NDVI. As described earlier, satellite images for the Limpopo corner were obtained between 15 February and 5 May in 2007, 2012, and 2017, which coincided with the wet to dry season transition. Other studies have used wet season NDVI data, which correlated well with vegetation dynamics, rainfall, and seasonality (Anyamba & Tucker, 2005; de Jong et al., 2011). Therefore, NDVI would indicate both grass and tree cover across the images (inverse of the method used by Mitchard & Flintrop, 2013). A measure of NDVI was included in this study to help distinguish fences from their surroundings as it was expected that NDVI would be low at fence sites due to areas adjacent to fences usually being cleared to form a 4 to 10 m open patch of ground (Lindsey et al., 2012). Normalized Difference Vegetation Index pixel values at fences from this study were mostly <0.1, and where fences appeared in subsequent years the NDVI difference values were mostly <0.2. The NDVI images were segmented through the 'Segment Mean Shift' tool, in the 'Spatial Analyst' toolset, which identifies segments in the imagery by grouping neighbouring pixels together that have similar spectral features. Image segmentation was used as to include spatial information which improves classification in general (Camps-Valls et al., 2014). The segmented NDVI images were then used as training data for the two image classification methods.

The training for the image classification went through a trial-and-error phase. An adequate number of training samples is very important for image classification and may vary based on study context (Lu & Weng, 2007). The number of training samples used in this study varied, ranging from 200 to 500 training samples for each class ('fence' and 'no fence'). Ultimately, the model converged with 300 training samples for each class for the SPOT 5 images, and increasing the number beyond 300 did not greatly increase the quality or accuracy of the image classification results (Wieland & Pittore, 2014). Spatial scale plays an important role in remote sensing studies (Woodcock & Strahler, 1987), therefore 500 training samples for each class were produced for the Sentinel-2 images due to them having larger image swaths (110 km x 110 km) than the SPOT 5 images (60 km x 60 km), and therefore containing more pixels. At first training samples based on the segmented NDVI images were produced for one image per year and then used for the classification of the rest of the images of each year. Yet, the classification quality seemed to have differed between output images. The second attempt involved merging the segmented images into one complete image (±27 600 km^2) and producing training samples from a large portion of the merged area. This was nevertheless unsuccessful. It was then realised that the differences in classification quality between the images may be due to their differences in NDVI values. All the images dated between February and May, however if one had to compare an NDVI image from February with one from May it would differ in NDVI values due to the effect of rainfall on the vegetation. Therefore, it was decided to use the date at which each satellite image was captured to categorise the data. Satellite images were grouped according to their date, and training samples were produced for each image in each study year (2007, 2012 and 2017). In addition, several images comprised of mountainous areas, urban areas, or clouds while others did not, therefore affecting the outcome of the broader scale image classification. For example, if training samples did not include mountains and were applied to an image with mountains, or vice versa, the image classification quality would decrease. The classification of remotely sensed data is certainly challenging due to factors such as landscape complexity (Lu & Weng, 2007).

As a result it was decided to produce training samples for each and every segmented NDVI image in order to incorporate temporal and spatial variables. Ultimately, the training signature files were entered, with the corresponding NDVI segment raster images and NDVI raster images, in the ‘Train Support Vector Machine Classifier’ tool as well as the ‘Train Random Trees Classifier’ tool through the ‘Spatial Analyst’ toolset. The segment attributes, ‘colour’, ‘mean’, and ‘rectangularity’, were selected in the process. Finally, the SVM and RF classifiers were collated respectively with their corresponding raster segments into the ‘Classify Raster’ tool to classify the images into ‘fence’ and ‘no fence’.

4.2.4 Accuracy Assessment

The goal of an accuracy assessment is to quantitatively determine how efficiently pixels were allocated to the correct feature classes in the area of interest (Ismail & Jusoff, 2008). In order to measure the accuracy at which the SVM and RF classifiers classified the images, accuracy assessment was conducted through ArcGIS 10.5. The following metrics were calculated through a confusion matrix to assess image classification; user’s accuracy, producer’s accuracy, overall accuracy, and kappa index. A confusion matrix, also called error matrix, confusion tables, and contingency matrices (Smits, Dellepiane, & Schowengerdt, 1999), is a table of numbers with rows and columns which contain the number of sample units (e.g. pixels) allocated to a particular category relative to the actual, verified category (Congalton, 1991). A user’s accuracy is a measure of commission error based on the row total and indicates the probability that a pixel is correctly classified, whereas a producer’s accuracy is a measure of omission error based on the column total and indicates the probability of a reference pixel being correctly classified (Ahlqvist, Keukelaar, & Oukbir, 2000; Congalton, 1991). Conversely, an overall accuracy is the percentage of correctly classified samples, and a kappa index accounts for all elements of the confusion matrix while excluding agreements that occurred by chance (Thapa & Murayama, 2009). The kappa index value lies between 0 and 1, where 0 signifies agreement due to chance alone and 1 signifies complete agreement between the two data sets (Ismail & Jusoff, 2008). The ideal accuracy target is said to be ≥85% (Anderson, Hardy, Roach, & Witmer, 1976), however the high target has been criticized as harsh and misleading due to its original criteria not being universally applicable (Foody, 2008). Most agree that a 0.55 – 0.70 kappa index is 'good' and a 0.70 – 0.85 kappa index is 'very good' (Monserud, 1990), which indicate that 0.70 could be an 'acceptable' accuracy target.

The accuracy assessment also went through a trial-and-error phase. Confusion matrices were calculated through ArcGIS 10.5 by producing 500 accuracy assessment points per SPOT 5 image and 700 points per Sentinel-2 image, through the 'Create Accuracy Assessment Points' tool, set on Random Stratified sampling. Each accuracy assessment point was 'ground truthed' through remote sensing with the original SPOT 5 or Sentinel-2 image. This specific method however resulted in low and idiosyncratic accuracy values, which may have been due to it being based on large areas, which thus increases the landscape complexity in each image. To enable a finer accuracy assessment it was decided to spatially partition the data by cropping the images with a 30 x 30 km grid of the study area, similar to the segmented sampling designs of Stehman et al. (2003) and Ismail and Jusoff (2008). Confusion matrices were then calculated for each image clip by producing accuracy assessment points for each, the number of which were based on the size of the cropped image (±30 x 30 km = 250, ±22.5 x 22.5 km = 200, ±15 x 15 km = 150, ±7.5 x 7.5 km = 100). This method proved to be more suitable, as accuracy values increased.

4.2.5. Data Analysis

Estimated areas (ha) of 'fence' and 'no fence' were calculated for each classification method and study year by using the 'Zonal Statistics as Table' tool through the 'Spatial Analyst' toolset in ArcGIS 10.5. The mean, standard deviation and range for the overall accuracies and kappa indices were calculated for both the SVM and RF classification methods in R version 3.3.3 (R Core Team, 2014) using RStudio version 1.0.136 (RStudio, 2017). The overall accuracies and kappa indices of the SVM and RF classifications were compared through Mann-Whitney U tests.

The Spearman rank correlation was calculated using the ‘car’ package (Fox & Weisberg, 2011) between overall accuracy and land characteristics (% elevation [<1000 m, 1000-1200 m, >1200 m], % cropland, % urban areas, presence of water) for each image classification method. The elevation data were based on the ASTER GDEM data (courtesy of NASA EOSDIS LP DAAC, USGS/EROS [https://earthexplorer.usgs.gov/]), a product of NASA and METI).

4.3 Results

The total areas (ha) covered by the two classes, ‘fence’ and ‘no fence’, and the percentage ‘fence’ area in the Support Vector Machine (SVM) classified images and Random Forest (RF) classified images of each study year (2007, 2012 and 2017) were inconsistent (Table 4.1 and 4.2).

Table 4.1. Area (ha) covered by ‘no fence’ and ‘fence’ and the percentage area (%) covered by ‘fence’ in the Support Vector Machine classified images of the wildlife sector in south-west Limpopo in 2007, 2012, and 2017.

Year	‘No Fence’ Area (ha)	‘Fence’ Area (ha)	‘Fence’ Area (%)
2007	1 092 282.6	441 808.9	28.8
2012	1 235 878.3	342 007.5	21.7
2017	1 130 931.3	454 527.3	28.7

Table 4.2. Area (ha) covered by 'no fence' and 'fence' and the percentage area (%) covered by 'fence' in the Random Forest classified images of the wildlife sector in south-west Limpopo in 2007, 2012, and 2017.

Year	'No Fence' Area (ha)	'Fence' Area (ha)	'Fence' Area (%)
2007	1 097 405.9	436 650.9	28.5
2012	1 240 601.0	407 668.5	24.7
2017	1 135 888.2	449 570.3	28.4

The mean, standard deviation and range for the overall accuracies (OA) and kappa indices (KI) of both classification methods, SVM (Table 4.3) and RF (Table 4.4), for each study year presented high variability. The total range of overall accuracy was 0.30 – 0.96, and kappa index was -0.06 – 0.56 (Tables 4.3 and 4.4). The most accurate classification method was the SVM method based on the 2012 data (OA = 0.81, KI = 0.20).

Table 4.3. The mean, range and standard deviation (SD) of the overall accuracies and kappa indices of the Support Vector Machine classification of the fences in the wildlife sector in south-west Limpopo in 2007, 2012, and 2017.

	Overall Accuracy		Kappa Index	
Year	Mean (SD)	Range	Mean (SD)	Range
2007	0.716 (0.159)	0.333 – 0.920	0.109 (0.103)	-0.064 – 0.320
2012	0.810 (0.124)	0.387 – 0.953	0.204 (0.125)	-0.045 – 0.548
2017	0.751 (0.188)	0.300 – 0.960	0.144 (0.141)	-0.023 – 0.555

Table 4.4. The mean, range and standard deviation (SD) of the overall accuracies and kappa indices of the Random Forest classification of the fences in the wildlife sector in south-west Limpopo in 2007, 2012, and 2017.

	Overall Accuracy		**Kappa Index**	
Year	**Mean (SD)**	**Range**	**Mean (SD)**	**Range**
2007	0.726 (0.145)	0.433 – 0.940	0.114 (0.089)	0.001 – 0.335
2012	0.801 (0.121)	0.387 – 0.930	0.193 (0.116)	0.000 – 0.440
2017	0.751 (0.149)	0.360 – 0.920	0.133 (0.107)	0.000 – 0.376

A Mann-Whitney test found no evidence for significant differences between the SVM and RF classification methods regarding overall accuracy (U_{2007} = 440.5, p_{2007} = 0.894; U_{2012} = 552.5, p_{2012} = 0.591; U_{2017} = 378.0, p_{2017} = 0.470) and kappa index (U_{2007} = 426.0, p_{2007} = 0.728; U_{2012} = 550.0, p_{2012} = 0.615; U_{2017} = 342.2, p_{2017} = 0.942).

To demonstrate the quality variation of the classified images, due to the broad ranges and high variability of the accuracy values, images with one of the highest (Figure 4.2) as well as the lowest (Figure 4.3) overall accuracies and kappa indices were illustrated.

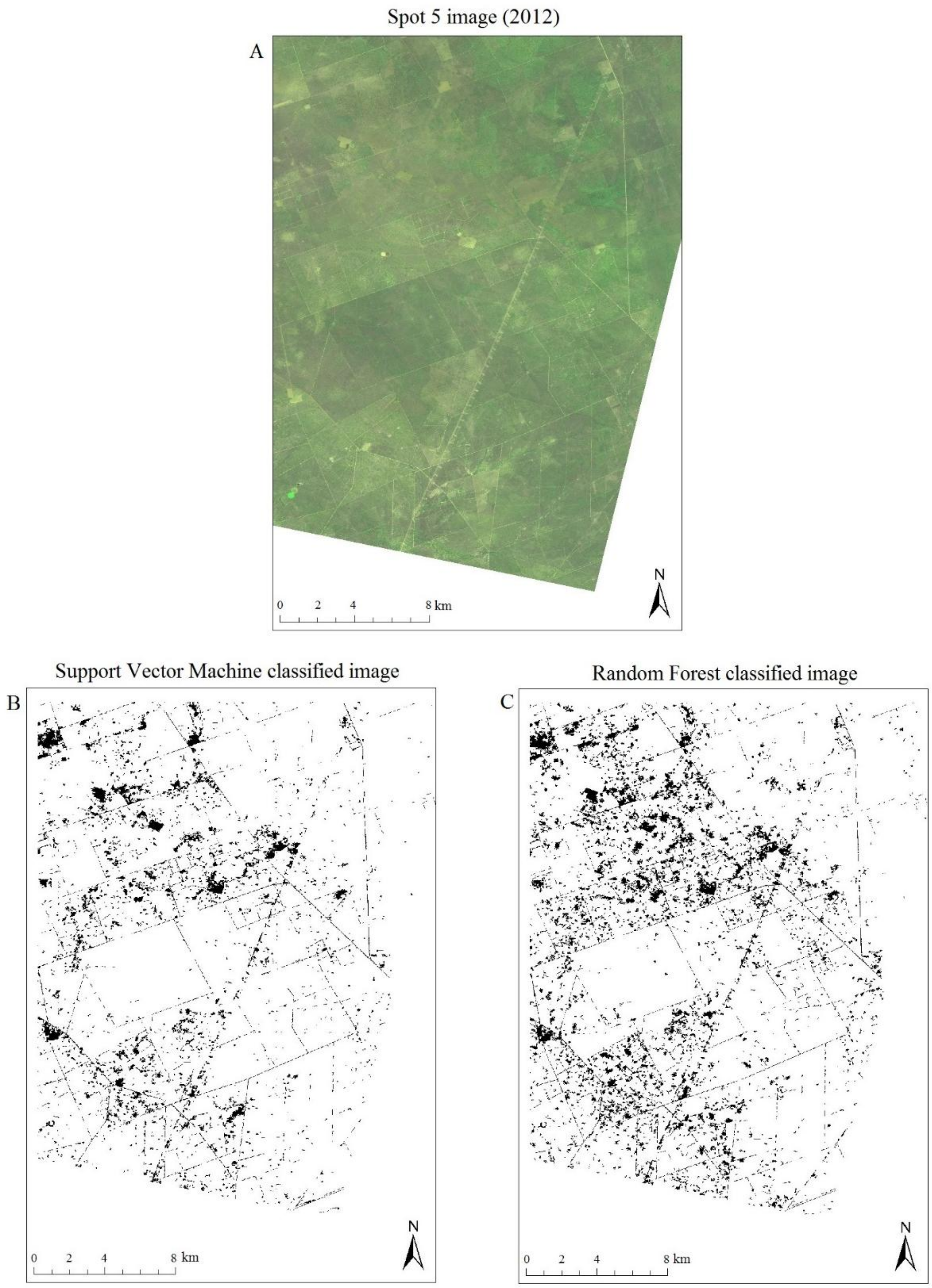

Figure 4.2. High accuracy classified images based on high accuracy values. A: SPOT 5 image, B: Support Vector Machine classified image (OA = 0.953. KI = 0.512), C: Random Forest classified image (OA = 0.930. KI = 0.426). Black = classified as 'fence'. White = classified as 'no fence'. (OA = Overall Accuracy, KI = Kappa Index).

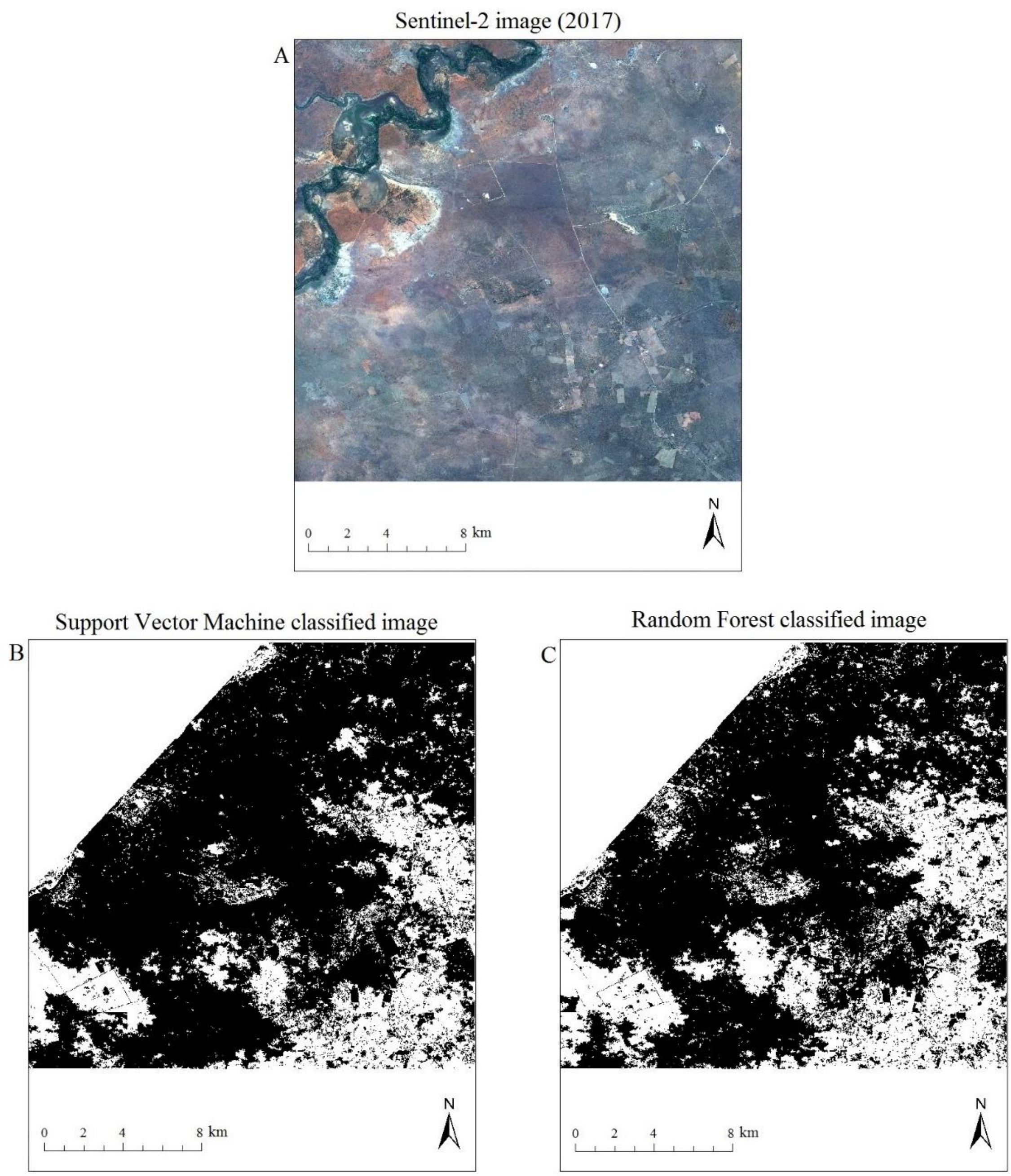

Figure 4.3. Low accuracy classified images based on low accuracy values. A: Sentinel-2 image, B: Support Vector Machine classified image (OA = 0.300. KI = 0.026), C: Random Forest classified image (OA = 0.360. KI = 0.015). Black = classified as 'fence'. White = classified as 'no fence'. (OA = Overall Accuracy, KI = Kappa Index).

The Spearman rank correlation test showed that percentage elevation and presence of water significantly correlated with overall accuracy for the SVM method in 2007 (%

<1000m: r_s = -0.553, p = 0.002; % 1000-1200m: r_s = 0.499, p = 0.005; water present: r_s = -0.630, p < 0.001; see Appendix: Table A4) and the RF method in 2007 (% <1000m: r_s = -0.463, p = 0.010; % 1000-1200m: r_s = 0.439, p = 0.015; see Appendix: Table A5) and 2012 (% <1000m: r_s = -0.476, p = 0.006; % 1000-1200m: r_s = 0.479, p = 0.006; water present: r_s = -0.577, p < 0.001; see Appendix: Table A5)

4.4 Discussion

The automating of the mapping of fences of a large wildlife sector area through image classification methods in this study is a novel approach for monitoring fence expansion in South Africa, presenting experimental results and a foundation for future improvement. Results indicate that the image classification methods, Support Vector Machines (SVM) and Random Forest (RF), could classify fences of the wildlife sector with varying accuracy. Due to the high variability in accuracy values (OA: 0.30 – 0.96, KI: -0.06 – 0.56) obtained from the image classification methods, the results of this chapter do not correlate with that of Chapter 3. Based on the results of Chapter 3, where fence-length increased 16% in total from 2007 to 2017, the present study did not support the expected 'no fence' area decrease, and 'fence' area and percentage increase over the entire time period. In fact, the image classification results indicated that the estimated fence area decreased from 2007 to 2012, and increased from 2012 to 2017. Therefore, the classification methods used in this study to classify fences (hereto referred to as 'fence classification') are not yet reliable due to the wide-ranging accuracy values and low kappa indices, which explains why the classified images varied in accuracy and quality. The SVM and RF classification methods were not statistically significantly different regarding accuracy values (Mann-Whitney: p > 0.05), which infers that the specific methods used may not have been the cause of the wide variety of accuracy values. As the overall accuracy of certain images significantly correlated with percentage elevation and presence of water (Spearman: $p < 0.05$), the cause may have been partially due to landscape complexity which generally affects image classification (Al-doski, Mansor, Zulhaidi, & Shafri, 2013; Lu & Weng, 2007).

This study has revealed the potential of using image classification methods to automate the mapping of fences. Image classification is generally influenced by landscape complexity, the selected remotely sensed data, the classification algorithm used, and the knowledge of the study area (Al-doski et al., 2013; Lu & Weng, 2007). The fence classification methods used in this study have a few limitations, especially as no real-world 'ground truthing' was conducted. Roads may have been misclassified as fences, or legacy effects of removed fences ('fence legacies') may have given false positives due to having similar visual signatures as existing fences (see Desmet & Pillay, 2016). Though 'fence legacies' may be misclassified as fences they may still have lasting effects on the environment which should not be disregarded. For instance, a study by Lindborg and Eriksson (2004) observed that plant species diversity had a stronger relationship with the habitat connectivity and area of a Swedish grassland landscape 50 and 100 years ago than with that of the present. Nevertheless, the current approach to fence classification should be improved when all the above mentioned factors are considered. Despite there being accurate classifiers, the image classification field faces two challenges namely complex statistical image characteristics, such as high pixel dimensionality and noise, and computational problems (large calculations), which require computationally efficient classification techniques due to increasing sensor data (Romero, Gatta, & Camps-valls, 2015). In general, image classification is a developing field which is gradually being developed and improved. New classification algorithms and techniques are continuously emerging (Lu & Weng, 2007). In addition, satellite remote sensing imagery has greatly improved from Landsat 1 (1972) to

WorldView-2 (2009), with an increase in spatial resolution (from 80 m to 0.5 m), an increase in return visit frequency (from 18 days to 1 day), and an increase in number of spectral bands (from four bands to eight or more bands) (Mulla, 2013).

Classification accuracy is the central measure of thematic map quality and fitness which is required by users (Foody, 2008). The accuracy values obtained in this study generally specify that the classification methods are insufficient in classifying fences due to most not meeting the commonly used ≥85% accuracy target (Anderson et al., 1976). The classified images with very low accuracy values (see Figure 4.3) indicate that fence classification in this study area may require improvement, however images with high overall accuracy (OA ≈ 0.9) (see Figure 4.2) indicate that fence classification is plausible – even with a kappa index that is below the general accuracy target (KI ≈ 0.5). Accuracy assessments have been criticised over the years (Foody, 2008; Pontius & Millones, 2011; Smits et al., 1999; Stehman, 1997; Wilkinson, 2005). For instance, the reliability of a confusion matrix is influenced by the subjectivity of classification labels, training samples, and the reference data sampling size and strategy (Smits et al., 1999). The kappa coefficient of agreement is also derived from the confusion matrix and has become a widely used measure of classification accuracy, as it considers the proportion of chance agreement (Foody, 1992). Yet, the use of the kappa coefficient has been questioned and seen as flawed (Pontius & Millones, 2011; Stehman, 1997), since it is often overestimated resulting in an underestimation of classification accuracy (Foody, 1992; Ma & Redmond, 1995). A study by Wilkinson (2005) reviewed 15 years of image classification publications (from 1989 to 2003) and found that during that time there was no apparent improvement in classification performance, and that the mean value of Kappa coefficients across all experiments was 0.656 (s = 0.198). Monserud (1990) listed the different threshold values regarding the degrees of agreement for the kappa statistic, where 0.55 – 0.70 was categorised as ‘good’, 0.7 – 0.85 as ‘very good’ and 0.85 –

0.99 as 'excellent'. Yet, the target accuracy of ≥85% is often used in image classification studies, which has been criticized as inappropriate and biased due to the figure being originally used by Anderson et al. (1976) for mapping broad land-cover classes from Landsat 1 sensor data (Foody, 2008). Ultimately, with regards to remote sensing, the accuracy of a classifier is only a relative accuracy in reality (Smits et al., 1999).

Image classification results are the basis for many environmental and socioeconomic applications (Lu & Weng, 2007), such as land cover and land use mapping (Rogan et al., 2008; Tehrany et al., 2014; Weng, 2002). The two image classification methods used in this study, namely Support Vector Machine (SVM) and Random Forest (RF), show promise in fence classification. Therefore, the aim should be to improve accuracy to utilize the potential of these methods in fence monitoring. Lu and Weng (2007) list the following factors to consider during a classification process: type of remotely sensed data; classification system and number of training samples; data pre-processing; feature extraction and selection; classification method; post-classification processing; and evaluation of classification performance. Based on this list and what was learned from this study, the following recommendations are made for future fence classification.

1. Acquiring high resolution satellite imagery is essential. Even though cleared vegetation surrounding fences help to distinguish fences, a finer resolution than the 10 m resolution of SPOT 5 and Sentinel-2 imagery would be beneficial. In fact, the spatial resolution of satellite imagery has improved over the past few decades, resulting in sub-metre imagery (Mulla, 2013).

2. Satellite imagery should ideally be obtained from the same time of year, preferably in the wet season, when wanting to compare NDVI values of fence pixels between years or

between sites. Fortunately, enhanced satellite technology has resulted in higher return visit frequencies (Mulla, 2013).

3. As image classification is influenced by landscape complexity (Lu & Weng, 2007), landscape characteristics should be considered before producing training samples for the classifiers. Therefore, areas with complex elevation patterns and water bodies should be excluded from the image to simplify training and classification.

4. Training set size should preferably be fixed and optimised for the specific conditions of the classification. Foody and Mathur (2004) reported that classification accuracy increased with increase in training set samples of 15 to 100 pixels per class, whereas Pal and Mather (2003) revealed that classification accuracy increased from 100 to 300 pixels per class. Therefore, training set size should be specified for the particular classification. It should be noted that a small number of training pixels may be adequate for spectrally homogeneous classes, but a large number of pixels are generally necessary for spatially heterogeneous classes in order to extract descriptive training statistics (Chen & Stow, 2002).

5. As roads may occasionally be falsely classified as fences, a database of roads would be beneficial for fence classification. In fact, any supporting data should be added as it has been found that using GIS data as auxiliary information, such as topography, soil, road, and census data, can improve remote sensing image classification performance (Deren, Kaichang, & Deyi, 2000; Lu & Weng, 2007). For instance, Rozenstein and Karnieli (2011) improved land use classification accuracy by up to 10% with ancillary data and GIS techniques.

6. Pre- and post-processing should be considered regarding the remotely sensed data and classification results to improve classification accuracy. Image pre-processing may comprise of geometric rectification or image registration, as well as atmospheric and topographic correction (Lu & Weng, 2007). Post-classification processing techniques may include using a

majority filter to reduce noise, and using ancillary data to modify the classification image (Lu & Weng, 2007).

7. Feature extraction and selection should also be considered in order to improve the discrimination between classes, such as discriminant analysis feature extraction (DAFE) or decision boundary feature extraction (DBFE) (Benediktsson, Pesaresi, & Amason, 2003).

8. The collection of ground truth or reference data is essential to effectively assess the accuracy of the image classification (Congalton, 1991; Smits et al., 1999) and reduce misclassifications due to roads and legacy effects of removed fences (see Foster et al., 2003; Desmet & Pillay, 2016). Hence, fieldwork is necessary to establish exactly where fences are in an area, so that the accuracy assessment of future fence classification methods may be properly measured and the method appropriately adjusted.

Fence classification may be improved by considering the above factors. The ultimate goal would be to automate the mapping of fences in order to assess the extent of fencing in the South African wildlife sector. This data may therefore provide a foundation for future research on the effects of fencing, and may also benefit future wildlife management decisions.

4.5 Conclusion

Image classification methods, such as Support Vector Machine and Random Forest, have the potential to automatically map fences of the wildlife sector in South Africa. The contrasting, low accuracy values produced by the fence classification methods can be improved in future studies by considering the list of recommendations above. Furthermore, new classification methods are constantly being developed (Lu & Weng, 2007), while satellite remote sensing imagery are evolving (Mulla, 2013). Therefore, future endeavours to better fence classification may be reinforced through new classification methods and superior satellite imagery.

The data used for fence classification as well as the fence classification results may open the door for other research opportunities. For instance, landscape degradation due to fencing may be investigated by quantifying the NDVI values in and around fence pixels. However, results would have to be supported with fieldwork, such as testing plant species richness and abundance, and soil quality and erosion. The effects of fencing on animal mortality, movement, behaviour and genetic structure should also be examined with proper field data. Ultimately, fence classification should be applied throughout South Africa to include different biomes, wildlife management practices, and fence types, and therefore to create a fence database. A database of fences in South Africa would be very beneficial and useful to wildlife researchers and managers. An accurate, centralised, spatially explicit fence database was in fact recommended by Gadd (2012) in order to understand the current fence network and prioritise future fence construction and removal. Over time the fence database can be updated to include fence types and conditions to further support wildlife management decisions and research (see Gadd, 2012). Moreover, if the fence database is incorporated with wildlife sector data it can aid with managing exemption permits, researching the impact of wildlife management practices on biodiversity, investigating wildlife disease occurrence and transmission, and monitoring conservancy projects (see Taylor et al., 2015). Ultimately, improving fence classification for future use would be a big leap forward in fence management which would benefit wildlife management overall.

Chapter 5:

The End of the Fence Line

5.1 Summary of findings

This study assessed the extent of and 'spatial' changes in the wildlife sector and its fencing in south-west Limpopo province through: 1) a comprehensive literature review, 2) mapping and quantifying fences and camps, and 3) attempting to automate the mapping of fences through image classification methods. By means of a literature review, Chapter 2 provided insight into the changing wildlife sector and its effects, as well as the implications of using fencing as a wildlife management tool. Over the past few decades there has been a rapid shift in land-use where livestock and crop farmers transformed into wildlife farmers due to productivity advantages and being able to legally own wildlife (Child, 2013; Grossman et al., 1999; Kamuti, 2014; Lindsey et al., 2009). Consequently the wildlife sector grew, resulting in a rise in wildlife farms and wildlife numbers (Taylor et al., 2015; van Hoven, 2015), as well as an increase in game fences (Hearne & McKenzie, 2000). The wildlife sector has generally been driven by tourist and hunter preferences due to their commercial nature (Child et al., 2012; Luxmoore, 1985). During the past decade there has been a marked shift towards intensive breeding of high-value wildlife species (Pitman et al., 2017). As a result, the intensification conveyed associated negative effects on wildlife and the environment (Desmet & Pillay, 2016; Luxmoore & Swanson, 1992; Nel, 2015). The growth of intensive breeding practices ultimately leads to a proliferation of small fenced camps (Taylor et al., 2015) which further increases the impenetrability and extent of fencing over the landscape, resulting in several ecological consequences such as inhibiting migration and causing habitat degradation (Desmet & Pillay, 2016; Gadd, 2012; Hayward & Kerley, 2009; Lindsey et al., 2012; Taylor & Martin, 1987). Increases in fences has moreover amplified the fragmentation of ecosystems globally (Jakes et al., 2018), especially in southern Africa due to the increasingly important role of fencing in aiding wildlife management, preventing disease spread, and reducing human-wildlife conflict (Gadd, 2012; Lindsey et al., 2012). It is therefore essential to study the extent and effects of the wildlife sector and fencing in order to develop sustainable wildlife land-use practices, and uphold its asserted ecological and economic contributions.

Spatial changes of the South African wildlife sector can be measured by mapping its fences due to fencing being one of the foundations of the wildlife sector as a result of the Game Theft Act, Act No. 105 (see Bothma et al., 2009; Nel, 2015). In Chapter 3, the fences and camps of the wildlife sector in south-west Limpopo were manually mapped using satellite imagery (SPOT 5 and Sentinel-2) of 2007, 2012, and 2017. In doing so, the extent of and changes in the wildlife sector could be inferred across a 10 year time frame, regarding the expansion of fences and intensive wildlife management practices. Results showed increases in the number of intensive wildlife properties (farm portions with ≤100 ha and ≤50 ha camps), total fence-length, and number of fenced camps, and a decrease in camp size over the entire time period. However, as there are limitations to solely using remote sensing data, it would be beneficial to confirm the trends by 'ground truthing' the wildlife properties and fences in the study area through fieldwork. Though the trends observed in this study are conservative estimates, they indicate a general increase in intensive breeding practices, fencing, and essentially landscape fragmentation. Based on the observation where smaller changes occurred between 2007 and 2012 and bigger changes occurred between 2012 and 2017, it could be inferred that the changes in the wildlife sector of south-west Limpopo, South Africa are occurring rapidly and progressively. Similar patterns were reported by Desmet and Pillay (2016) where intensive breeding properties and total fence length in Limpopo had higher increases from 2010 to 2015 than from 2006 to 2010. Løvschal et al. (2017) also observed faster fence expansion in the Greater Mara, Kenya between 2014 and 2016 than between 1985 and 2014. Therefore, these trends should be urgently addressed in order to minimize their detrimental effects on the wildlife and environment (Desmet & Pillay, 2016; Gadd, 2012; Hayward & Kerley, 2009; Lindsey et al., 2012; Luxmoore & Swanson, 1992; Nel, 2015; Taylor et al., 2015). The study area in south-west Limpopo has already been classified as moderately degraded Meadows and Hoffman (2002), based on poor soil and vegetation conditions. Ultimately, the trends show that the wildlife sector is becoming increasingly anthropological and decreasingly natural, which may decrease visitations by tourists and hunters who generally prefer large areas without internal subdivisions (Damm, 2005).

This 'rise in fencing' is making it essential to monitor fence expansion and generate fence maps of the wildlife sector in South Africa and any other country where the increase in fencing may pose a threat to wildlife and the environment. A spatially explicit fence database would especially be beneficial to wildlife researchers and managers to better understand the fence network and prioritise fence monitoring and management (Gadd, 2012). Therefore, fence maps will enhance proper wildlife management and conservation. In Chapter 4, the mapping of fences were automated through image classification methods, specifically a Support Vector Machine (SVM) and a Random Forest (RF) model, which could aid in producing a fence database. According to current literature, the mapping of fences through image classification methods, or 'fence classification', is a novel approach and hence required various trial-and-error procedures. The SVM and RF classification methods were applied to a total of 21 segmented, Normalised Difference Vegetation Index (NDVI) images based on satellite images with a 10 m pixel resolution (SPOT 5 and Sentinel-2) captured during the wet to dry season transition (15 February to 5 May) in 2007, 2012, and 2017. The results of Chapter 4 did not correspond with Chapter 3, as the fence area in the classified images did not follow the same increasing trend as fence-length did in the manual fence maps. This may have been due to the fence classification methods producing wide-ranging accuracy values and low kappa indices (overall accuracy range: 0.30 – 0.96, kappa index range: -0.06 – 0.56) which explains the variation in accuracy and quality of classified images. Additionally, when the accuracy values of the SVM and RF methods were compared they were not significantly different (Mann-Whitney: $p > 0.05$) and the overall accuracy of some classified images correlated with landscape characteristics (Spearman: $p < 0.05$), specifically percentage elevation and presence of water. Since this was the first attempt at fence classification using machine learning techniques, it should be noted that the classification methods used were limited by there being no other studies to compare results to. The methods were furthermore limited by a lack of ‘real-world’ ground truthing and possible misclassification of roads or ‘fence legacies’ as existing fences (see Foster et al., 2003; Desmet & Pillay, 2016). In addition, based on the commonly used accuracy target of ≥85% in the remote sensing community (Foody, 2008), fence classification may not yet be practically applicable at landscape scales as undertaken in the present study. Even though there have been disputes on the applicability of the specific target value (Foody, 2008), and the reliability of confusion matrices (Smits et al., 1999) and kappa indices (Pontius & Millones, 2011; Stehman, 1997), the fence classification methods should still be improved and adapted for future use. Image classification is affected by landscape complexity, the particular remotely sensed data, the classification algorithm used, and the knowledge of the study area (Al-doski et al., 2013; Lu & Weng, 2007), and can thus be limited by these factors.

5.2 Recommendations and potential solutions

The results of this study regarding the increase in fencing could serve as an early warning to wildlife researchers and farmers, and highlight the adverse ecological and economic effects linked to the expansion of fencing. It is vital to increase research efforts on the extent and effects of the wildlife sector and fencing, especially as there is a lack of data regarding the growth and ecological effects of the wildlife sector (Bothma & von Bach, 2010; Lindsey et al., 2009; Taylor et al., 2015); wildlife-fence interactions and fence systems (Jakes et al., 2018); and the ecological, financial and social impacts of fencing (Lindsey et al., 2012). The systematic quantitative literature review method used in Chapter 1 found a large gap in literature concerning intensive wildlife production in particular. Hence, further research is required on the growing intensive wildlife sector in South Africa. Furthermore, inconsistent communication regarding public environmental awareness and knowledge should be amended by improving the flow of information, and considering geographic, cultural and social factors (Dalerum, 2014). The communication between wildlife researchers and the wildlife sector community in particular should be improved (see Dalerum, 2014), especially as the wildlife sector is largely tourist-driven and often has inadequate professional conservation management and planning (Cousins et al., 2008; Pienaar et al., 2017). The South African wildlife sector community also generally lacks available information on the ecological consequences of fencing and the benefits of forming conservancies (Lindsey et al., 2009). Strengthening the relationship between researchers, landowners and managers would therefore improve management decision making and planning overall. In addition, management practices, such as intensive breeding, and fencing should be more formally regulated. The intensive wildlife sector should especially be quantified and regulated, as there is currently a lack of research information on the intensive wildlife sector (see Chapter 1). Furthermore, the management and monitoring of fences, and the ecological effects thereof, should be improved as fence construction, maintenance and assessment have often been hastily done or conducted without professional consultation (Gadd, 2012). Fencing should especially be regulated as there are currently only minimum fencing requirements (see Brown et al., 2014) and no legal regulations or formal national guidelines for fences (Beck, 2008; Desmet & Pillay, 2016). Lastly, the formation of conservancies, which have several ecological and financial benefits (Damm, 2005; Lindsey et al., 2012, 2009), would be an essential step forward to connecting wildlife areas as opposed to fragmenting them.

Recommendations have been made which may further improve the management of the wildlife sector and fencing (Beck, 2008; BirdLife South Africa, 2018; Gadd, 2012; Paige, 2008; Taylor et al., 2015). The main suggestions are to clarify the impacts of wildlife management practices, create conservancies (Taylor et al., 2015), monitor and remove fences (Gadd, 2012), and change fencing designs to be more wildlife friendly (Beck, 2008; BirdLife South Africa, 2018; Paige, 2008). Following the example of several European countries, it would also be beneficial to plan infrastructure and roads over the long term in such a way as to accommodate habitat connectivity and animal movement (Bekker & Iuell, 2003).

A spatially explicit fence database would also be beneficial in order to better monitor and manage fences (Gadd, 2012). The fence classification method could be used to help produce a fence database, however the accuracy should firstly be improved. Important aspects based on Lu and Weng (2007) should be considered for the purpose of refining fence classification: a) acquiring high resolution satellite imagery within the same time period; b) excluding landscape complexities from images, such as water bodies; c) using a fixed and optimised training set size (pixels per class); d) incorporating GIS data, such as road data, as auxiliary information; e) adding pre- and post-processing techniques; f) integrating feature extraction and selection methods; and g) collecting ground truth or reference data to improve accuracy assessments. Fortunately, future endeavours to improve fence classification may be reinforced through new classification methods and superior satellite remote sensing imagery which are constantly being developed (Lu & Weng, 2007; Mulla, 2013).

Remote sensing and fence classification can also support other research opportunities such as assessing landscape degradation around fences through NDVI and fieldwork data (see

Holm et al., 2003; Pettorelli et al., 2005; Weng, 2002), or creating long-term research projects on the effects of fencing on wildlife and associated ecological systems. Ultimately, improved fence classification could be applied to the entire South African wildlife sector to create a comprehensive fence database that could aid fence management to allow for informed management decisions (Gadd, 2012). Over time, more data could be integrated with the fence database to further benefit wildlife research and wildlife management.

5.3 Concluding remarks

With this thesis I aim to provide a foundation for further research on fencing, and ultimately allow wildlife management decisions based on quantitative research. Furthermore, with the age of machine learning, remote sensing and image classification constantly evolving, we may observe the advancement of fence classification methods to produce a fence database. The combination of additional research and a fence database may additionally support potential resolutions, such as the removal and regulation of fences, which may ensure sustainable wildlife conservation.

References

ABSA Group Economic Research. (2002). *Game ranch profitability in southern Africa* (2nd ed.). Rivonia: The SA Financial Sector Forum.

Agjee, A., Fourie, M., Loser, N., Reddell, C., Youens, K., Gibbs, A., … Rodwell-van Hasselt, L. (2018). *Fair Game? Improving the well-being of South African wildlife: Review of the legal and practical regulation of the welfare of wild animals in South Africa, 2018*. Retrieved from https://cer.org.za/wp-content/uploads/2018/06/CER-EWT-Regulation-of-Wildlife-Welfare-Report-25-June-2018.pdf

Ahlqvist, O., Keukelaar, J., & Oukbir, K. (2000). Rough classification and accuracy assessment. *International Journal of Geographical Information Science*, *14*(5), 475–496. doi:10.1080/13658810050057605

Akar, Ö., & Güngör, O. (2012). Classification of multispectral images using Random Forest algorithm. *Journal of Geodesy and Geoinformation, 1*(2), 105–112. doi:10.9733/jgg.241212.1

Al-doski, J., Mansor, S. B., Zulhaidi, H., & Shafri, M. (2013). Image classification in remote sensing. *Journal of Environment and Earth Science*, *3*(10), 141–148.

Anderson, B. J. R., Hardy, E. E., Roach, J. T., & Witmer, R. E. (1976). *A land use and land cover classification system for use with remote sensor data*. *964*. Washington: United States Government Printing Office.

Angermeier, P. (1994). Does biodiversity include artificial diversity? *Conservation Biology*, *8*(2), 600–602.

Anyamba, A., & Tucker, C. J. (2005). Analysis of Sahelian vegetation dynamics using NOAA-AVHRR NDVI data from 1981-2003. *Journal of Arid Environments*, *63*(3), 596–614. doi:10.1016/j.jaridenv.2005.03.007

Barbanera, F., Pergams, O. R. W., Guerrini, M., Forcina, G., Panayides, P., & Dini, F.

(2010). Genetic consequences of intensive management in game birds. *Biological Conservation*, *143*(5), 1259–1268. doi:10.1016/j.biocon.2010.02.035

Barnes, J. I. (1998). *Wildlife conservation and utilisation as complements to agriculture in southern African development*. No. 27. Windhoek, Namibia: Directorate of Environmental Affairs, Ministry of Environment and Tourism.

Barnes, J. I. (2001). Economic returns and allocation of resources in the wildlife sector in Botswana. *South African Journal of Wildlife Research*, *31*(3&4), 141–153.

Bazi, Y., & Melgani, F. (2006). Toward an optimal SVM classification system for hyperspectral remote sensing images. *IEEE Transactions on Geoscience and Remote Sensing*, *44*(11), 3374–3385. doi:10.1109/TGRS.2006.880628

Beck, A. (2008). *Electric fence induced mortality in South Africa*. MSc thesis. University of Witwatersrand.

Bekker, H., & Iuell, B. (2003). *Habitat fragmentation due to infrastructure*. (C. L. Irwin, P. Garrett, & K. P. McDermott, Eds.), *Proceedings of the International Conference on Ecology and Transportation*. Raleigh, United States: North Carolina State University.

Belousov, A. I., Verzakov, S. A., & von Frese, J. (2002). A flexible classification approach with optimal generalisation performance: support vector machines. *Chemometrics and Intelligent Laboratory Systems*, *64*(1), 15–25. doi:10.1016/S0169-7439(02)00046-1

Ben-Shahar, R. (1993). Does fencing reduce the carrying capacity for populations of large herbivores? *Journal of Tropical Ecology*, *9*(2), 249–253.

Benediktsson, J. A., Pesaresi, M., & Amason, K. (2003). Classification and feature extraction for remote sensing images from urban areas based on morphological transformations. *IEEE Transactions on Geoscience and Remote Sensing*, *41*(9), 1940–1949. doi:10.1109/TGRS.2003.814625

Bengis, R. G., Kock, R. A., & Fischer, J. (2002). Infectious animal diseases: the wildlife/livestock interface. *Revue Scientifique et Technique (International Office of Epizootics)*, *21*(1), 53–65. doi:10.20506/rst.21.1.1322

Bennett, A. F., & Saunders, D. A. (2011). Habitat fragmentation and landscape change. In N. S. Sodhi & P. R. Ehrlich (Eds.), *Conservation biology for all* (pp. 88–106). Oxford: University Press. doi:10.1093/acprof

BirdLife South Africa. (2018). *Fences & birds: Minimising unintended impacts*. Retrieved from https://www.birdlife.org.za/conservation/important-bird-areas/documents-and-downloads

Boggs, G. S. (2010). Assessment of SPOT 5 and QuickBird remotely sensed imagery for mapping tree cover in savannas. *International Journal of Applied Earth Observation and Geoinformation*, *12*(4), 217–224. doi:10.1016/j.jag.2009.11.001

Bond, I., Child, B., de la Harpe, D., Jones, B., Barnes, J., & Anderson, H. (2004). Private land contribution to conservation in South Africa. In B. Child (Ed.), *Parks in transition: Biodiversity, rural development and the bottom line* (pp. 29–62). London: Earthscan.

Boone, R. B., & Hobbs, N. T. (2004). Lines around fragments: effects of fencing on large herbivores. *Journal of Range and Forage Science*, *21*(3), 147–158.

Botha, J., & Siebert, F. (2014). 12 years of exclusion: What have we learnt and where to from here? In *SANPARKS research report 2014* (pp. 42–46). South African National Parks. doi:10.2784/1473

Bothma, J. du P. (2010). Important ecological principles and conservancies. In J. du P. Bothma & J. G. du Toit (Eds.), *Game ranch management* (5th ed., pp. 3–26). Pretoria: Van Schaik.

Bothma, J. du P., Suich, H., & Spenceley, A. (2009). Extensive wildlife production on private land in South Africa. In H. Suich, B. Child, & A. Spenceley (Eds.), *Evolution and innovation in wildlife conservation parks and game ranches to transfrontier* *conservation areas* (pp. 147–162). London: Earthscan.

Bothma, J. du P., & van Rooyen, N. (2005). *Intensive wildlife production in southern Africa* (1st ed.). Pretoria: Van Schaik.

Bothma, J. du P., & von Bach, H. J. S. (2010). Economic aspects of extensive wildlife production in southern Africa. In J. du P. Bothma & J. G. du Toit (Eds.), *Game ranch management* (5th ed., pp. 83–96). Pretoria: Van Schaik.

Breiman, L. (2001). Random Forests. *Machine Learning*, *45*(1), 5–32.

Brooks, S., Spierenburg, M., Van Brakel, L., Kolk, A., & Lukhozi, K. B. (2011). Creating a commodified wilderness: Tourism, private game farming, and “third nature” landscapes in KwaZulu-Natal. *Tijdschrift Voor Economische En Sociale Geografie*, *102*(3), 260–274. doi:10.1111/j.1467-9663.2011.00662.x

Brown, C. J., Gildenhuys, P. S., Hignett, D. L., & Van Deventer, J. D. (2014). *Policy on fencing and enclosure of game, predators and dangerous animals in the Western Cape Province*. *Cape Nature Biodiversity Support Services*. Stellenbosch.

Bulte, E. H., & Damania, R. (2005). An economic assessment of wildlife farming and conservation. *Conservation Biology*, *19*(4), 1222–1233. doi:10.1111/j.1523-1739.2005.00149.x

Burns, B., Innes, J., & Day, T. (2012). The use and potential of pest-proof fencing for ecosystem restoration and fauna conservation in New Zealand. In M. W. Hayward & M. J. Somers (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 65–90). New York: Springer.

Camps-Valls, G., Tuia, D., Bruzzone, L., & Benediktsson, J. A. (2014). Advances in hyperspectral image classification: Earth monitoring with statistical learning methods. *IEEE Signal Processing Magazine*, *31*(1), 45–54. doi:10.1109/MSP.2013.2279179

Carruthers, J. (2008). “Wilding the farm or farming the wild”? The evolution of scientific game ranching in South Africa from the 1960s to the present. *Transactions of the Royal Society of South Africa*, *63*(2), 160–181. doi:10.1080/00359190809519220

Castley, G., Boshoff, A. F., & Kerley, G. I. H. (2001). Compromising South Africa's natural biodiversity - inappropriate herbivore introductions. *South African Journal of Science*, *97*(9–10), 344–348.

Caughley, G. (1994). Directions in conservation biology. *The Journal of Animal Ecology*, *63*(2), 215. doi:10.2307/5542

Cerna, L., & Chytry, M. (2005). Supervised classification of plant communities with artificial neural networks. *Journal of Vegetation Science*, *16*(4), 407–414.

Chen, D., & Stow, D. (2002). The effect of training strategies on supervised classification at different spatial resolutions. *Photogrammetric Engineering & Remote Sensing*, *68*(11), 1155–1161.

Child, B. (2000). Making wildlife pay: Converting wildlife's comparative advantage into real incentives for having wildlife in African savannas, case studies from Zimbabwe and Zambia. In H. H. T. Prins, J. G. Grootenhuis, & T. T. Dolan (Eds.), *Wildlife conservation by sustainable use* (pp. 335–387). Boston: Kluwer Academic Publishers.

Child, B. (2009). Private conservation in southern Africa: practice and emerging principles. In H. Suich, B. Child, & A. Spenceley (Eds.), *Evolution and innovation: parks and game ranches to transfrontier conservation areas* (pp. 103–112). London: Earthscan.

Child, B. (2013). *Parks in transition: Biodiversity, rural development and the bottom line*. London: Earthscan.

Child, B. A., Musengezi, J., Parent, G. D., & Child, G. F. T. (2012). The economics and institutional economics of wildlife on private land in Africa. *Pastoralism: Research, Policy and Practice*, *2*(18), 1–32. doi:10.1186/2041-7136-2-18

Child, G., & Chitsike, L. (2000). "Ownership" of Wildlife. In H. H. T. Prins, J. G.

Grootenhuis, & T. T. Dolan (Eds.), *Wildlife conservation by sustainable use* (pp. 247–265). Boston: Kluwer Academic Publishers.

Child, M. F., Peel, M. J. S., Smit, I. P. J., & Sutherland, W. J. (2013). Quantifying the effects of diverse private protected area management systems on ecosystem properties in a savannah biome, South Africa. *Oryx*, *47*(1), 29–40. doi:10.1017/S0030605312000038

Clevenger, A. P., Chruszcz, B., & Gunson, K. E. (2001). Highway mitigation fencing reduces wildlife-vehicle collisions. *Wildlife Society Bulletin*, *29*(2), 646–653. doi:10.2307/3784191

Cloete, P. C., Taljaard, P. R., & Grové, B. (2007). A comparative economic case study of switching from cattle farming to game ranching in the Northern Cape Province. *South African Journal of Wildlife Research*, *37*(1), 71–78. doi:10.3957/0379-4369-37.1.71

Coleman, A. (2015). Breeding high-value game species in North West. Retrieved October 12, 2018, from https://www.farmersweekly.co.za/animals/game-and-wildlife/breeding-high-value-game-species-in-north-west/

Coleman, A. (2017). Commercial buffalo breeding: ‘It’s not just for the rich’. Retrieved October 12, 2018, from https://www.farmersweekly.co.za/animals/game-and-wildlife/commercial-buffalo-breeding-not-just-rich/

Congalton, R. G. (1991). A review of assessing the accuracy of classification of remotely sensed data. *Remote Sensing of Environment*, *37*(1), 34–46. doi:10.1016/0034-4257(91)90048-B

Cousins, J. A., Sadler, J. P., & Evans, J. (2008). Exploring the role of private wildlife ranching as a conservation tool in South Africa: Stakeholder perspectives. *Ecology and Society*, *13*(2), 1–17.

Cousins, J. A., Sadler, J. P., & Evans, J. (2010). The challenge of regulating private wildlife ranches for conservation in South Africa. *Ecology and Society*, *15*(2), 1–21.

doi:10.5751/ES-03349-150228

Culbert, P. D., Dorresteijn, I., Loos, J., Clayton, M. K., Fischer, J., & Kuemmerle, T. (2017). Legacy effects of past land use on current biodiversity in a low-intensity farming landscape in Transylvania (Romania). *Landscape Ecology*, *32*(2), 429–444. doi:10.1007/s10980-016-0441-3

Dalerum, F. (2014). Identifying the role of conservation biology for solving the environmental crisis. *Ambio*, *43*(7), 839–846. doi:10.1007/s13280-017-0937-3

Dalerum, F., & Miranda, M. (2016). Game auction prices are not related to biodiversity contributions of southern African ungulates and large carnivores. *Scientific Reports*, *6*, 21922. doi:10.1038/srep21922

Damm, G. R. (2005). Hunting in South Africa: Facts, risks, opportunities. *African Indaba*, *3*(4/5), 1–23.

de Jong, R., de Bruin, S., de Wit, A., Schaepman, M. E., & Dent, D. L. (2011). Analysis of monotonic greening and browning trends from global NDVI time-series. *Remote Sensing of Environment*, *115*(2), 692–702. doi:10.1016/j.rse.2010.10.011

Delroy, L. B., Earl, J., Radbone, I., Robinson, A. C., & Hewett, M. (1986). The breeding and re-establishment of the brush-tailed bettong, Bettongia penicillata, in south Australia. *Wildlife Research*, *13*(3), 387–396. doi:10.1071/WR9860387

Deren, L., Kaichang, D., & Deyi, L. (2000). Land use classification of remote sensing image with GIS data based on spatial data mining techniques. *International Archives of Photogrammetry and Remote Sensing*, *33*(B3/1), 238–245. doi:10.1080/01431160903485786

Desmet, P., & Pillay, S. (2016). *The landscape ecological impact of intensive game breeding in South Africa: Mapping the rate of change in game camp fence density at two sites in Limpopo province from 2006 - 2015*. Report for the SA Hunters and Game Conservation

Association by ECOSOL GIS.

Díaz-Fernández, S., Arroyo, B., Casas, F., Martinez-Haro, M., & Viñuela, J. (2013). Effect of game management on wild red-legged partridge abundance. *PLoS One*, *8*(6), 1–10. doi:10.1371/journal.pone.0066671

Dickman, C. R. (2012). Fences or ferals? Benefits and costs of conservation fencing in Australia. In M. W. Hayward & M. J. Somers (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 43–64). New York: Springer.

Dorgeloh, W. G., van Hoven, W., & Rethman, N. F. G. (1996). Population growth of roan antelope under different management systems. *South African Journal of Wildlife Research*, *26*(4), 113–116.

Drew, K. R. (1989). Intensive containment systems: game farming. In R. J. Hudson, K. R. Drew, & L. M. Baskin (Eds.), *Wildlife production systems: Economic utilisation of wild ungulates* (pp. 307–308). Cambridge: Cambridge University Press.

du Toit, J. G. (2010a). Legislation and codes of conduct for animal care, management and use. In J. du P. Bothma & J. G. du Toit (Eds.), *Game ranch management* (5th ed., pp. 759–775). Pretoria: Van Schaik.

du Toit, J. G. (2010b). Preventative disease management and epidemiology. In J. du P. Bothma & J. G. du Toit (Eds.), *Game ranch management* (5th ed., pp. 432–437). Pretoria: Van Schaik.

du Toit, J. G., & van Rooyen, J. (2010). Wildlife ranch acquisition and cattle ranch transformation. In J. du P. Bothma & J. G. du Toit (Eds.), *Game ranch management* (pp. 50–54). Pretoria: Van Schaik.

Dupouey, J. L., Dambrine, E., Laffite, J. D., & Moares, C. (2002). Irreversible impact of past land use on forest soils and biodiversity. *Ecology*, *83*(11), 2978–2984.

Duro, D. C., Franklin, S. E., & Dubé, M. G. (2012). A comparison of pixel-based and object-based image analysis with selected machine learning algorithms for the classification of agricultural landscapes using SPOT-5 HRG imagery. *Remote Sensing of Environment*, *118*, 259–272. doi:10.1016/j.rse.2011.11.020

East, M. L., Nyahongo, J. W., Goller, K. V., & Hofer, H. (2012). Does the vastness of the Serengeti limit human-wildlife conflicts. In M. W. Hayward & M. J. Somers (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 125–152). New York: Springer.

Ebedes, H. (1994). Going, going, gone – an appraisal of game auctions. In W. van Hoven, H. Ebedes, & A. Conroy (Eds.), *Wildlife ranching: a celebration of diversity. Proceedings of the 3rd international wildlife ranching symposium* (pp. 74–80). Pretoria: Promedia.

Eltringham, S. K. (1994). Can wildlife pay its way? *Oryx*, *28*(3), 163–168. doi:10.1017/S0030605300028519

Environmental Systems Research Institute (ESRI). (2016). *ArcGIS Desktop: Release 10.5*. Redlands, CA: Environmental Systems Research Institute.

Fahrig, L., & Merriam, G. (1994). Conservation of fragmented populations. *Conservation Biology*, *8*, 50–59.

Fauvel, M., Chanussot, J., & Benediktsson, J. A. (2006). Evaluation of kernels for multiclass classification of hyperspectral remote sensing data. In *2006 IEEE International Conference on Acoustics Speech and Signal Processing Proceedings* (Vol. 2). doi:10.1109/ICASSP.2006.1660467

Ferguson, K., & Hanks, J. (2012). The effects of protected area and veterinary fencing on wildlife conservation in southern Africa. In S. Stolton & N. Dudley (Eds.), *PARKS. The International Journal of Protected Areas and Conservation* (pp. 49–60). Gland. Switzerland: IUCN WCPA.

Foley, J. A., Defries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., … Snyder, P. K. (2005). Global consequences of land use. *Science*, *309*(5734), 570–574. doi:10.1126/science.1111772

Foody, G. M. (1992). On the compensation for chance agreement in image classification accuracy assessment. *Photogrammetric Engineering and Remote Sensing*, *58*(10), 1459–1460.

Foody, G. M. (2002). Status of land cover classification accuracy assessment. *Remote Sensing of Environment*, *80*(1), 185–201. doi:10.1016/S0034-4257(01)00295-4

Foody, G. M. (2008). Harshness in image classification accuracy assessment. *International Journal of Remote Sensing*, *29*(11), 3137–3158. doi:10.1080/01431160701442120

Foody, G. M., & Mathur, A. (2004). A relative evaluation of multi-class image classification by support vector machines. *IEEE Transactions on Geoscience and Remote Sensing*, *42*(6), 1335–1343.

Foster, D., Swanson, F., Aber, J. D., Burke, I., Brokaw, N., Tilman, D., & Knapp, A. (2003). The importance of land-use legacies to ecology and conservation. *BioScience*, *53*(1), 77–88. doi:10.1641/0006-3568(2003)053[0077:TIOLUL]2.0.CO;2

Fox, J., & Weisberg, S. (2011). An {R} Companion to Applied Regression. Thousand Oaks CA: Sage. Retrieved from http://socserv.socsci.mcmaster.ca/jfox/Books/Companion

Gadd, M. E. (2012). Barriers, the beef industry and unnatural selection: A review of the impact of veterinary fencing on mammals in southern Africa. In M. J. Somers & M. W. Hayward (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 153–186). New York: Springer.

Ginsberg, J. R. (1994). Captive breeding, reintroduction and the conservation of canids. In P. J. S. Olney, G. M. Mace, & A. T. C. Feistner (Eds.), *Creative conservation: Interactive management of wild and captive animals* (1st ed., pp. 365–383). London: Chapman &

Hall.

Google Inc. (2018). *Google Earth Pro*. Retrieved from ttps://www.google.com/earth/desktop/

Grant, C. C. R. (2008). Controlling the distribution of elephants. In R. Scholes & K. Mennell (Eds.), *Elephant management: A scientific assessment of South Africa* (pp. 329–369). Johannesburg: Wits University Press.

Grobler, J. P., & van der Bank, F. H. (1994). Do game fences affect the genetic diversity in commercially utilised game populations? In W. van Hoven, H. Ebedes, & A. Conroy (Eds.), *Wildlife ranching: a celebration of diversity. Proceedings of the 3rd international wildlife ranching symposium* (pp. 238–240). Pretoria: Promedia.

Grossman, D., Holden, P. L., & Collinson, R. F. H. (1999). Veld management on the game ranch. In N. Tainton (Ed.), *Veld management in South Africa* (pp. 261–279). Pietermaritzburg: University of Natal Press.

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., … Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. *Applied Ecology*, *1*(2), 1–9. doi:10.1126/sciadv.1500052

Harrington, J. L. (2005). *Characteristics of ungulate behavior and mortality associated with wire fences*. *All Graduate Theses and Dissertations*. MSc thesis. Utah State University. doi:/10.2193/0091-7648(2006)34[1295:coubam]2.0.co;2

Harrington, R., Owen-Smith, N., Viloen, P. C., Biggs, H. C., Mason, D. R., & Funston, P. (1999). Establishing the cases of the roan antelope decline in the Kruger National Park, South Africa. *Biological Conservation*, *90*(1), 69–78. doi:10.1016/S0006-3207(98)00120-7

Hayward, M. W. (2012). Perspectives on fencing for conservation based on four case studies: Marsupial conservation in Australian forests; bushmeat hunting in South Africa; large predator reintroduction in South Africa; and large mammal conservation in Poland. In

M. J. Somers & M. W. Hayward (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 7–20). New York: Springer.

Hayward, M. W., & Kerley, G. I. H. (2009). Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes? *Biological Conservation, 142*, 1–13. doi:10.1016/j.biocon.2008.09.022

Hayward, M. W., & Somers, M. J. (2012). An introduction to fencing for conservation. In M. J. Somers & M. W. Hayward (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 1–6). New York: Springer.

Hearne, J., & McKenzie, M. (2000). Compelling reasons for game ranching in Maputaland. In H. H. T. Prins, J. G. Grootenhuis, & T. T. Dolan (Eds.), *Wildlife conservation by sustainable use* (pp. 417–438). Dordrecht: Kluwer Academic Publishers.

Hermes, L., Frieauff, D., Puzicha, J., & Buhmann, J. M. (1999). Support vector machines for land usage classification in Landsat TM imagery. *IEEE 1999 International Geoscience and Remote Sensing Symposium*, *1*(3), 348–350. doi:10.1109/IGARSS.1999.773494

Higginbottom, K., & King, N. (2006). *The live trade in free-ranging wildlife within South Africa, and implications for Australia*. A Report for the Rural Industries Research and Development Corporation.

Hoare, R. E. (1992). Present and future use of fencing in the management of larger African mammals. *Environmental Conservation*, *19*(2), 160–164.

Hobbs, N. T., Reid, R. S., Galvin, K. A., & Ellis, J. E. (2008). Fragmentation of arid and semi-arid ecosystems: Implications for people and animals. In K. A. Galvin, R. S. Reid, R. H. Behnke Jr., & N. T. Hobbs (Eds.), *Fragmentation in semi-arid and arid landscapes: Consequences for human and natural Systems* (pp. 25–44). Dordrecht:

Springer.

Hoffman, L. C., & Wiklund, E. (2006). Game and venison - meat for the modern consumer. *Meat Science*, *74*(1), 197–208. doi:10.1016/j.meatsci.2006.04.005

Holm, A. M. R., Cridland, S. W., & Roderick, M. L. (2003). The use of time-integrated NOAA NDVI data and rainfall to assess landscape degradation in the arid shrubland of Western Australia. *Remote Sensing of Environment*, *85*(2), 145–158. doi:10.1016/S0034-4257(02)00199-2

Hopcraft, D. (2000). Wildlife land use and the great experiment. In H. H. T. Prins, J. G. Grootenhuis, & T. T. Dolan (Eds.), *Wildlife conservation by sustainable use* (pp. 267–276). Boston: Kluwer Academic Publishers.

Houghton, R. A. (1994). The worldwide extent of land-use change. *BioScience*, *44*(5), 305–313.

Huang, X., & Jensen, J. R. (1997). A machine-learning approach to automated knowledge-base building for remote sensing image analysis with GIS data. *Photogrammetric Engineering & Remote Sensing*, *63*(10), 1185–1194.

Hunter, L. T. B., Pretorius, K., Carlisle, L. C., Rickelton, M., Walker, C., Slotow, R., & Skinner, J. D. (2007). Restoring lions *Panthera leo* to northern KwaZulu-Natal, South Africa: Short-term biological and technical success but equivocal long-term conservation. *Oryx*, *41*(2), 196–204. doi:10.1017/S003060530700172X

Hüttich, C., Herold, M., Schmullius, C., Egorov, V., & Bartalev, S. A. (2007). Indicators of Northern Eurasia’s land-cover change trends from SPOT-VEGETATION time-series analysis 1998-2005. *International Journal of Remote Sensing*, *28*(18), 4199–4206. doi:10.1080/01431160701442054

Ilea, R. C. (2009). Intensive livestock farming: Global trends, increased environmental concerns, and ethical solutions. *Journal of Agricultural and Environmental Ethics*, *22*(2), 153–167. doi:10.1007/s10806-008-9136-3

Ismail, M. H., & Jusoff, K. (2008). Satellite data classification accuracy assessment based from reference dataset. *International Journal of Computer and Information Science and Engineering*, *2*(3), 96–102.

Jakes, A. F., Jones, P. F., Paige, L. C., Seidler, R. G., & Huijser, M. P. (2018). A fence runs through it: A call for greater attention to the influence of fences on wildlife and ecosystems. *Biological Conservation*, *227*, 310–318. doi:10.1016/j.biocon.2018.09.026

Kamuti, T. (2014). The fractured state in the governance of private game farming: The case of KwaZulu-Natal Province, South Africa. *Journal of Contemporary African Studies*, *32*(2), 190–206. doi:10.1080/02589001.2014.936678

Keuchel, J., Naumann, S., Heiler, M., & Siegmund, A. (2003). Automatic land cover analysis for Tenerife by supervised classification using remotely sensed data. *Remote Sensing of Environment*, *86*(4), 530–541. doi:10.1016/S0034-4257(03)00130-5

Knight, A. T., & Cowling, R. M. (2012). Towards a true ecology: Exploring the implications for conservation of the human and social dimensions of fencing in the subtropical thicket biome, South Africa. In M. W. Hayward & M. J. Somers (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 197–214). New York: Springer.

Kozakiewicz, M. (1993). Habitat isolation and ecological barriers - the effect on small mammal populations and communities. *Acta Theriologica*, *38*(1), 1–30. doi:10.4098/AT.arch.93-1

Langholz, J. A., & Kerley, G. I. H. (2006). Combining conservation and development on private lands: an assessment of ecotourism-based private game reserves in the Eastern Cape. *Centre for African Conservation Ecology (Formerly TERU) Report*, (56), 1–34.

Laurance, W. F., Camargo, J. L. C., Luizão, R. C. C., Laurance, S. G., Pimm, S. L., Bruna, E.

M., … Lovejoy, T. E. (2011). The fate of Amazonian forest fragments: A 32-year investigation. *Biological Conservation*, *144*(1), 56–67. doi:10.1016/j.biocon.2010.09.021

Laurance, W. F., Goosem, M., & Laurance, S. G. W. (2009). Impacts of roads and linear clearing on tropical forests. *Trends in Ecology and Evolution*, *24*(12), 659–679.

Lehmann, M. B., Funston, P. J., Owen, C. R., & Slotow, R. (2008). Feeding behaviour of lions (*Panthera leo*) on a small reserve. *South African Journal of Wildlife Research*, *38*(1), 66–78. doi:10.3957/0379-4369-38.1.66

Liaw, A., & Wiener, M. (2002). Classification and regression by randomForest. *R News*, *2*(3), 18–22. doi:10.1177/154405910408300516

Lindborg, R., & Eriksson, O. (2004). Historical landscape connectivity affects present plant species diversity. *Ecology*, *85*(7), 1840–1845.

Lindsey, P. A., Alexander, R., Frank, L. G., Mathieson, A., & Romañach, S. S. (2006). Potential of trophy hunting to create incentives for wildlife conservation in Africa where alternative wildlife-based land uses may not be viable. *Animal Conservation*, *9*(3), 283–291. doi:10.1111/j.1469-1795.2006.00034.x

Lindsey, P. A., Masterson, C. L., Beck, A. L., & Romanach, S. S. (2012). Ecological, social and financial issues related to fencing as a conservation tool in Africa. In M. W. Hayward & M. J. Somers (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 215–234). New York: Springer.

Lindsey, P. A., Romañach, S. S., & Davies-Mostert, H. T. (2009). The importance of conservancies for enhancing the value of game ranch land for large mammal conservation in southern Africa. *Journal of Zoology*, *277*(2), 99–105. doi:10.1111/j.1469-7998.2008.00529.x

Lisita, A., Sano, E. E., & Durieux, L. (2013). Identifying potential areas of *Cannabis sativa* plantations using object-based image analysis of SPOT-5 satellite data. *International Journal of Remote Sensing*, *34*(15), 5409–5428. doi:10.1080/01431161.2013.790574

Loarie, S. R., van Aarde, R. J., & Pimm, S. L. (2009). Fences and artificial water affect African savannah elephant movement patterns. *Biological Conservation*, *142*(12), 3086–3098. doi:10.1016/j.biocon.2009.08.008

Lovejoy, T., Bierregaard, B., Brown, K., Emmons, L., & van der Voort, M. (1984). Ecosystem decay of Amazon forest fragments. In M. Niteki (Ed.), *Extinctions* (pp. 295–325). Illinois: University of Chicago Press.

Løvschal, M., Bøcher, P. K., Pilgaard, J., Amoke, I., Odingo, A., Thuo, A., & Svenning, J. C. (2017a). Fencing bodes a rapid collapse of the unique Greater Mara ecosystem. *Scientific Reports*, *7*(41450). doi:10.1038/srep41450

Lu, D., & Weng, Q. (2007). A survey of image classification methods and techniques for improving classification performance. *International Journal of Remote Sensing*, *28*(5), 823–870. doi:10.1080/01431160600746456

Luxmoore, R. (1985). Game farming in South Africa as a force in conservation. *Oryx*, *19*(4), 225–231. doi:10.1017/S0030605300025667

Luxmoore, R., & Swanson, T. M. (1992). Wildlife and wildland utilization and conservation. In T. M. Swanson & E. B. Barbier (Eds.), *Economics for the wilds: Wildlife, wildlands, diversity and development* (pp. 170–194). London: Earthscan.

Lyons, J. A., & Natusch, D. J. D. (2011). Wildlife laundering through breeding farms: Illegal harvest, population declines and a means of regulating the trade of green pythons (*Morelia viridis*) from Indonesia. *Biological Conservation*, *144*(12), 3073–3081. doi:10.1016/j.biocon.2011.10.002

Ma, Z., & Redmond, R. L. (1995). Tau coefficient for accuracy assessment of classification of remote sensing data. *Photogrametric Engineering and Remote Sensing*, *61*(4), 435–

439.

MacArthur, R. H., & Wilson, E. O. (1967). *The theory of island biogeography*. Princeton: Princeton University Press.

Macdonald, I., & Brooks, P. (1983). Monitoring for the detection of mammal overabundance in small conservation areas. In R. N. Owen-Smith (Ed.), *Management of large mammals in African conservation areas* (pp. 187–200). Pretoria: HAUM.

Mackey, R. L., Page, B. R., Duffy, K. J., & Slotow, R. (2006). Modelling elephant population growth in small, fenced, South African reserves. *South African Journal of Wildlife Research*, *36*(1), 33–43.

Magee, K. S. (2011). *Segmentation, object-oriented applications for remote sensing land cover and land use classification*. PhD thesis. University of Cincinnati.

Magin, C. D., Johnson, T. H., Groombridge, B., Jenkins, M., & Smith, H. (1994). Species extinctions, endangerment and captive breeding. In P. J. S. Olney, G. M. Mace, & A. T. C. Feistner (Eds.), *Creative conservation: Interactive management of wild and captive animals* (1st ed., pp. 3–31). London: Chapman & Hall.

Maitima, J. M., Mugatha, S. M., Reid, R. S., Gachimbi, L. N., Majule, A., Lyaruu, H., … Mugisha, S. (2009). The linkages between land use change, land degradation and biodiversity across East Africa. *African Journal of Environmental Science and Technology*, *3*(10), 310–325. doi:10.5897/AJEST08.173

Martínez, M. L., Pérez-Maqueo, O., Vázquez, G., Castillo-Campos, G., García-Franco, J., Mehltreter, K., … Landgrave, R. (2009). Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. *Forest Ecology and Management*, *258*(9), 1856–1863. doi:10.1016/j.foreco.2009.02.023

Maschinski, J., Frye, R., Rutman, S., & Rutmant, S. (1997). Demography and population viability of an endangered plant species before and after protection from trampling.

Conservation Biology, *11*(4), 990–999.

Massey, A. L., King, A. A., & Foufopoulos, J. (2014). Fencing protected areas: A long-term assessment of the effects of reserve establishment and fencing on African mammalian diversity. *Biological Conservation*, *176*, 162–171. doi:10.1016/j.biocon.2014.05.023

Mbaiwa, J. E., & Darkoh, M. B. K. (1999). Sustainable development and natural resource competition and conflicts in the Okavango Delta, Botswana. *Botswana Notes & Records*, *37*, 40–60.

Mbaiwa, J. E., & Mbaiwa, O. I. (2006). The effects of veterinary fences on wildlife populations in Okavango Delta, Botswana. *International Journal of Wilderness*, *12*(3), 17–24.

Meadows, M. E., & Hoffman, M. T. (2002). The nature, extent and causes of land degradation in South Africa: legacy of the past, lessons for the future? *Area*, *34*(4), 428–437.

Melgani, F., & Bruzzone, L. (2004). Classification of hyperspectral remote sensing images with Support Vector Machines. *IEEE Transactions on Geoscience and Remote Sensing*, *42*(8), 1778–1790.

Mitchard, E. T. A., & Flintrop, C. M. (2013). Woody encroachment and forest degradation in sub-Saharan Africa's woodlands and savannas 1982 – 2006. *Philosophical Transactions of the Royal Society B*, *368*(16245), 1–7.

Monserud, R. A. (1990). Methods for comparing global vegetation maps. *Working Papers. International Institute for Applied Systems Analysis*, 1–37.

Mountrakis, G., Im, J., & Ogole, C. (2011). Support vector machines in remote sensing: A review. *ISPRS Journal of Photogrammetry and Remote Sensing*, *66*(3), 247–259. doi:10.1016/j.isprsjprs.2010.11.001

Mulla, D. J. (2013). Twenty five years of remote sensing in precision agriculture: Key advances and remaining knowledge gaps. *Biosystems Engineering*, *114*(4), 358–371. doi:10.1016/j.biosystemseng.2012.08.009

Naude, M. (2016). SA Hunters concerned about new regulations that allow cross-breeding of wildlife. Retrieved October 10, 2018, from http://www.sahunters.co.za/index.php/hunting-jagsake/item/246-sa-hunters-concerned-about-new-regulations-that-allow-cross-breeding-of-wildlife

Nel, C. (2017). A step-by-step guide to intensive bushbuck breeding. Retrieved October 12, 2018, from https://www.farmersweekly.co.za/animals/game-and-wildlife/step-step-guide-intensive-bushbuck-breeding/

Nel, E. J. (2015). *Risks and impacts associated with intensive and selective breeding of indigenous game for commercial purposes*. Report for the SA Hunters and Game Conservation Association. Pretoria.

Newmark, W. D. (2008). Isolation of African protected areas. *Frontiers in Ecology and the Environment*, *6*(6), 321–328. doi:10.1890/070003

Nogueira-Filho, S. L. G., & Nogueira, S. S. C. (2004). Captive breeding programs as an alternative for wildlife conservation in Brazil. In *People in nature: Wildlife management and conservation in Latin America* (Vol. 2, pp. 171–190). New York: Columbia University Press.

Nogueira, S. S. C., & Nogueira-Filho, S. L. G. (2011). Wildlife farming: An alternative to unsustainable hunting and deforestation in Neotropical forests? *Biodiversity and Conservation*, *20*(7), 1385–1397. doi:10.1007/s10531-011-0047-7

O’Connor, T. G., & Kuyler, P. (2009). Impact of land use on the biodiversity integrity of the moist sub-biome of the grassland biome, South Africa. *Journal of Environmental Management*, *90*(1), 384–395. doi:10.1016/j.jenvman.2007.10.012

Osipova, L., Okello, M. M., Njumbi, S. J., Ngene, S., Western, D., Hayward, M. W., &

Balkenhol, N. (2018). Fencing solves human-wildlife conflict locally but shifts problems elsewhere: A case study using functional connectivity modelling of the African elephant. *Journal of Applied Ecology*, *55*(6), 2673-2684 doi:10.1111/1365-2664.13246

Owen-Smith, R. N. (1983). Management of large mammals in African conservation areas. In *Dispersal and the dynamics of large herbivores in enclosed areas: Implications for management* (pp. 127–144). Pretoria: HAUM.

Owen, M., & Owen, D. (1980). The fences of death. *African Wildlife*, *34*(6), 25–27.

Paige, C. (2008). *A landowner's guide to wildlife friendly fences.* Landowner/Wildlife Resource Program, Montana Fish, Wildlife and Parks, Helena, MT. pp 44. Retrieved from http://fwpiis.mt.gov/content/getItem.aspx?id=34461

Pal, M. (2005). Random forest classifier for remote sensing classification. *International Journal of Remote Sensing*, *26*(1), 217–222.

Pal, M., & Mather, P. M. (2003). An assessment of the effectiveness of decision tree methods for land cover classification. *Remote Sensing of Environment*, *86*(4), 554–565. doi:10.1016/S0034-4257(03)00132-9

Pal, M., & Mather, P. M. (2005). Support vector machines for classification in remote sensing. *International Journal of Remote Sensing*, *26*(5), 1007–1011. doi:10.1080/01431160512331314083

Parrini, F., Macindoe, M., & Erasmus, B. F. N. (2013). A comparison between field-measured NDVI and visual estimates of grass greenness in buffalo-foraging areas. *African Journal of Ecology*, *51*(4), 641–643.

Pettorelli, N., Vik, J. O., Mysterud, A., Gaillard, J. M., Tucker, C. J., & Stenseth, N. C. (2005). Using the satellite-derived NDVI to assess ecological responses to environmental change. *Trends in Ecology and Evolution*, *20*(9), 503–510. doi:10.1016/j.tree.2005.05.011

Pickering, C., Grignon, J., Steven, R., Guitart, D., & Byrne, J. (2014). Publishing not perishing: How research students transition from novice to knowledgeable using systematic quantitative literature reviews. *Studies in Higher Education*, *40*(10), 1756–1769. doi:10.1080/03075079.2014.914907

Pienaar, E. F., Rubino, E. C., Saayman, M., & van der Merwe, P. (2017). Attaining sustainable use on private game ranching lands in South Africa. *Land Use Policy*, *65*, 176–185. doi:10.1016/j.landusepol.2017.04.005

Pitman, R. T., Fattebert, J., Williams, S. T., Williams, K. S., Hill, R. A., Hunter, L. T. B., … Balme, G. A. (2017). The conservation costs of game ranching. *Conservation Letters*, *10*(4), 402–412. doi:10.1111/conl.12276

Pontius, R. G., & Millones, M. (2011). Death to Kappa: Birth of quantity disagreement and allocation disagreement for accuracy assessment. *International Journal of Remote Sensing*, *32*(15), 4407–4429. doi:10.1080/01431161.2011.552923

R Core Team. (2014). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from http://www.r-project.org/

Reid, R. S., Thornton, P. K., Kruska, R. L., Reid, R. S., Thornton, P. K., & Loss, R. L. K. (2017). Loss and fragmentation of habitat for pastoral people and wildlife in east Africa: concepts and issues. *African Journal of Range & Forage Science*, *21*(3), 171–181. doi:10.2989/10220110409485849

Reilly, B. K., Sutherland, E. A., & Harley, V. (2003). The nature and extent of wildlife ranching in Gauteng province, South Africa. *South African Journal of Wildlife Research*, *33*(2), 141–144.

Reyers, B. (2004). Incorporating anthropogenic threats into evaluations of regional biodiversity and prioritisation of conservation areas in the Limpopo Province, South

Africa. *Biological Conservation*, *118*(4), 521–531. doi:10.1016/j.biocon.2003.09.027

Rogan, J., Franklin, J., Stow, D., Miller, J., Woodcock, C., & Roberts, D. (2008). Mapping land-cover modifications over large areas: A comparison of machine learning algorithms. *Remote Sensing of Environment*, *112*(5), 2272–2283. doi:10.1016/j.rse.2007.10.004

Romero, A., Gatta, C., & Camps-valls, G. (2015). Unsupervised deep feature extraction for remote sensing image classification. *IEEE Transactions on Geoscience and Remote Sensing*, *54*(3), 1–14. doi:10.1109/TGRS.2015.2478379.

Rozenstein, O., & Karnieli, A. (2011). Comparison of methods for land-use classification incorporating remote sensing and GIS inputs. *Applied Geography*, *31*(2), 533–544. doi:10.1016/j.apgeog.2010.11.006

RStudio. (2017). *RStudio: Integrated development environment for R*. Boston, MA. Retrieved from http://www.rstudio.org/

Saayman, M., van der Merwe, P., & Rossouw, R. (2011). The impact of hunting for biltong purposes on the SA economy. *Acta Commercii*, *11*(1), 1–12. doi:10.4102/ac.v11i1.143

Saunders, D. A., Hobbs, R. J., & Margules, C. R. (1991). Biological consequences of ecosystem fragmentation: A review. *Conservation Biology*, *5*(1), 18–32. doi:10.1111/j.1523-1739.1991.tb00384.x

Shi, H., Fan, Z., Yin, F., & Yuan, Z. (2004). New data on the trade and captive breeding of turtles in Guangxi Province, South China. *Asiatic Herpetological Research*, *10*, 126–128.

Sims-Castley, R., Kerley, G. I. H., Geach, B., & Langholz, J. (2005). Socio-economic significance of ecotourism-based private game reserves in South Africa's Eastern Cape Province. In P. Goriup & B. Creed (Eds.), *PARKS. The International Journal of Protected Areas and Conservation* (Vol. 15, pp. 6–17). World Commission on Protected

Areas (WCPA).

Slotow, R. (2012). Fencing for purpose: A case study of elephants in South Africa. In M. W. Hayward & M. J. Somers (Eds.), *Fencing for conservation: Restriction of evolutionary potential or a riposte to threatening processes?* (pp. 91–104). New York: Springer.

Slotow, R., Garaï, M. E., Reilly, B. K., Page, B. R., & Carr, R. D. (2005). Population dynamics of elephants re-introduced to small fenced reserves in South Africa. *South African Journal of Wildlife Research*, *35*(1), 23–32.

Smith, A., Schoeman, M. C., Keith, M., Erasmus, B. F. N., Monadjem, A., Moilanen, A., & Di Minin, E. (2016). Synergistic effects of climate and land-use change on representation of African bats in priority conservation areas. *Ecological Indicators*, *69*, 276–283. doi:10.1016/j.ecolind.2016.04.039

Smith, N., & Wilson, S. L. (2002). *Changing land use trends in the thicket biome: pastoralism to game farming*. *Terrestrial Ecology Research Unit, Report No. 38*. Port Elizabeth.

Smith, R. J. (1981). Resolving the tragedy of the commons by creating private property rights in wildlife. *Cato Journal*, *1*(2), 439–468. doi:10.1525/sp.2007.54.1.23.

Smits, P. C., Dellepiane, S. G., & Schowengerdt, R. A. (1999). Quality assessment of image classification algorithms for land-cover mapping: A review and a proposal for a cost-based approach. *International Journal of Remote Sensing*, *20*(8), 1461–1486. doi:10.1080/014311699212560

Snijders, D. (2012). Wild property and its boundaries - on wildlife policy and rural consequences in South Africa. *Journal of Peasant Studies*, *39*(2), 503–520. doi:10.1080/03066150.2012.667406

Spear, D., & Chown, S. L. (2009). Non-indigenous ungulates as a threat to biodiversity. *Journal of Zoology*, *279*(1), 1–17. doi:10.1111/j.1469-7998.2009.00604.x

Stehman, S. V., Wickham, J. D., Smith, J. H., & Yang, L. (2003). Thematic accuracy of the 1992. National land-cover data for the eastern United States: Statistical methodology and regional results. *Remote Sensing of Environment*, *86*, 500–516.

Stehman, S. V. (1997). Selecting and interpreting measure of thematic classification accuracy. *Remote Sensing of Environment*, *62*(1), 77–89.

Swanson, T. M. (1992). Wildlife and wildlands: diversity and development. In T. M. Swanson & E. B. Barbier (Eds.), *Economics for the wilds: Wildlife, wildlands, diversity and development* (pp. 1–14). London: Earthscan.

Symeonakis, E., & Drake, N. (2004). Monitoring desertification and land degradation over sub-Saharan Africa. *International Journal of Remote Sensing*, *25*(3), 573–592. doi:10.1080/0143116031000095998

Symonds, A., & Swemmer, L. (2013). Human-wildlife conflict: Environmental justice and building community support for conservation. In *SANPARKS Research Report 2013* (pp. 33–36).

Taylor, A., Lindsey, P., & Davies-Mostert, H. (2015). *An assessment of the economic, social and conservation value of the wildlife ranching industry and its potential to support the green economy in South Africa*. The Endangered Wildlife Trust, Johannesburg.

Taylor, R. D., & Martin, R. B. (1987). Effects of veterinary fences on wildlife conservation in Zimbabwe. *Environmental Management*, *11*(3), 327–334.

Tehrany, M. S., Pradhan, B., & Jebuv, M. N. (2014). A comparative assessment between object and pixel-based classification approaches for land use/land cover mapping using SPOT 5 imagery. *Geocarto International*, *29*(4), 351–369. doi:10.1080/10106049.2013.768300

Terborgh, J., Lopez, L., Nuñez, P. V, Rao, M., Shahabuddin, G., Orihuela, G., … Balbas, L. (2001). Ecological meltdown in predator-free forest fragments. *Science*, *294*(5548), 1999–2002. doi:10.1126/science.1064397

Thapa, R. B., & Murayama, Y. (2009). Urban mapping, accuracy, & image classification: A comparison of multiple approaches in Tsukuba City, Japan. *Applied Geography*, *29*(1), 135–144. doi:10.1016/j.apgeog.2008.08.001

The National Agricultural Marketing Council. (2006). *Report on the investigation to identify problems for sustainable growth and development in South African wildlife ranching*.

Thomas, S. (2017). Hunting: The colour game is over. *Financialmail*. Retrieved from https://www.businesslive.co.za/fm/features/2017-01-26-colour-game-is-over/

Tizora, P., Le Roux, A., Mans, G., & Cooper, A. (2016). *Land use and land cover change in the Western Cape Province: Quantification of changes & understanding of driving factors*. MSc thesis. University of Pretoria.

Topaloğlu, R. H., Sertel, E., & Musaoğlu, N. (2016). Assessment of classification accuracies of Sentinel-2 and Landsat-8 data for land cover/use mapping. *International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences - ISPRS Archives*, *41*, 1055–1059. doi:10.5194/isprsarchives-XLI-B8-1055-2016

Van der Merwe, P., & Saayman, M. (2003). Determining the economic value of game farm tourism. *Koedoe*, *46*(2), 103–112. doi:10.4102/koedoe.v46i2.59

van der Merwe, P., Saayman, M., & Rossouw, R. (2015). The economic impact of hunting. *Journal of Economic and Financial Sciences*, *8*(1), 223–242.

van der Waal, C., & Dekker, B. (2000). Game ranching in the Northern Province of South Africa. *South African Journal of Wildlife Research*, *30*(4), 151–155.

van Dyk, G., & Slotow, R. (2003). The effects of fences and lions on the ecology of African wild dogs reintroduced to Pilanesberg National Park, South Africa. *African Zoology*, *38*(1), 79–94.

van Hoven, W. (2015). Private game reserves in southern Africa. In R. van der Duim, M.

Lamers, & J. van Wijk (Eds.), *Institutional arrangements for conservation, development and tourism in eastern and southern Africa: A dynamic perspective* (pp. 101–118). Dordrecht: Springer. doi:10.1016/j.annals.2015.05.004

van Rooyen, N., du Toit, J. G., & van Rooyen, J. (2010). Fences. In J. du P. Bothma & J. G. du Toit (Eds.), *Game ranch management* (5th ed., pp. 133–149). Pretoria: Van Schaik.

Vapnik, V. N. (1995). *The nature of statistical learning theory*. New York: Springer.

Warren, R. (2011). *A spending model for biltong hunters*. PhD thesis. North-West University.

Weng, Q. (2002). Land use change analysis in the Zhujiang Delta of China using satellite remote sensing, GIS and stochastic modelling. *Journal of Environmental Management*, *64*(3), 273–284. doi:10.1006/jema.2001.0509

Wessels, K. J., Reyers, B., Jaarsveld, A. S. Van, & Rutherford, M. C. (2003). Identification of potential conflict areas between land transformation and biodiversity conservation in north-eastern South Africa. *Agriculture, Ecosystems and Environment*, *95*(1), 157–178.

Wieland, M., & Pittore, M. (2014). Performance evaluation of machine learning algorithms for urban pattern recognition from multi-spectral satellite images. *Remote Sensing*, *6*(4), 2912–2939. doi:10.3390/rs6042912

Wilkinson, G. G. (2005). Results and implications of a study of fifteen years of satellite image classification experiments. *IEEE Transactions on Geoscience and Remote Sensing*, *43*(3), 433–440. doi:10.1109/TGRS.2004.837325

Wilson, A. C., & Price, M. R. S. (1994). Reintroduction as a reason for captive breeding. In P. J. S. Olney, G. M. Mace, & A. T. C. Feistner (Eds.), *Creative conservation: Interactive management of wild and captive animals* (1st ed., pp. 243–264). London: Chapman & Hall.

Wilson, K. (2006). *Status and distribution of cheetah outside formal conservation areas in the Thabazimbi district, Limpopo province*. MSc thesis. University of Pretoria.

Woodcock, C. E., & Strahler, A. H. (1987). The factor of scale in remote-sensing. *Remote Sensing of Environment*, *21*(3), 311–332. doi:10.1016/0034-4257(87)90015-0

Yilmaz, A., & Tepeli, C. (2009). Breeding performance of a captive Chukar Partridge (*Alectoris chukar*) flock. *Journal of Animal and Veterinary Advances*, *8*(8), 1584–1588.

Zheng, B., Myint, S. W., Thenkabail, P. S., & Aggarwal, R. M. (2015). A support vector machine to identify irrigated crop types using time-series Landsat NDVI data. *International Journal of Applied Earth Observation and Geoinformation*, *34*(1), 103–112. doi:10.1016/j.jag.2014.07.002

Zhu, Z., Woodcock, C. E., Rogan, J., & Kellndorfer, J. (2012). Assessment of spectral, polarimetric, temporal, and spatial dimensions for urban and peri-urban land cover classification using Landsat and SAR data. *Remote Sensing of Environment*, *117*, 72–82. doi:10.1016/j.rse.2011.07.020

Appendix

Table A1. List of articles and books found through the systematic quantitative approach with the specific keywords inputted in Google Scholar, the study area, the number of times it has been cited, and study species.

Article / Book	Keywords (with 'intensive')	Study area	# Cited	Species
Barbanera (2010)	game breeding, game management	Europe	57	Partridge
Barnes (1998)	wildlife production, wildlife farm	Southern Africa	35	Ostrich, crocodile
Barnes (2001)	wildlife production	Botswana	63	Ostrich, crocodile
Bothma and van Rooyen (2005)	wildlife farm	Southern Africa	14	Game
Bulte and Damania (2005)	wildlife breeding, wildlife farm	Netherlands	92	Rhinoceros
Carruthers (2008)	game breeding, game farm, game ranch	South Africa	111	Game
Cousins et al. (2008)	wildlife ranch	South Africa	123	Game
Cousins et al. (2010)	wildlife breeding, wildlife ranch	South Africa	53	Game

Delroy et al. (1986)	wildlife breeding	Australia	59	Bettong
Diaz-Fernandez et al. (2013)	game breeding	Spain	30	Partridge
Hoffman and Wiklund (2006)	game production, game farm	Europe and South Africa	358	Cervids
Hudson (1989)	wildlife production	Worldwide	53	Ungulates
• Drew			5	
Swanson and Barbier (1992)	wildlife breeding, wildlife production	South Africa	152	Game
• Swanson			16	
• Luxmoore and Swanson			19	
Lyons and Natusch (2011)	wildlife breeding	Indonesia	113	Python
Nel (2015)	game breeding	South Africa	3	Game
Noguiera and Noguiera-Filho (2011)	game breeding, wildlife production, wildlife farm	Brazil	64	Peccary
Noguiera-Filho and Noguiera (2004)	wildlife breeding, game breeding, wildlife production	Brazil	43	Peccary, capybara

Olney (1994)	wildlife breeding	Worldwide	104	Wildlife
• Magin et al.			67	
• Wilson and Price			75	
• Ginsberg			51	
Taylor et al. (2015)	wildlife breeding	South Africa	45	Game
Yilmaz and Tepeli (2009)	game breeding	Turkey	7	Partridge
Zeder (2012)	game breeding	Worldwide	148	Wildlife

Table A2. Dates when satellite images (SPOT 5 and Sentinel-2) were captured and number of images per date in 2007, 2012, and 2017.

SPOT 5		Sentinel-2
2007	**2012**	**2017**
13-03 (3)	02-05 (4)	05-04 (3)
25-02 (1)	01-04 (1)	05-05 (2)
18-03 (1)	17-04 (3)	
28-04 (1)		
15-02 (2)		

Table A3. ANOVA comparing various factors to the time intervals (2007, 2012, and 2017), namely fence-length, number of camps, number of camps per size category (<10 ha, 10-50 ha, 50-100 ha, 100-200 ha, 200-500 ha, 500-1000 ha, 1000-1500 ha, >1500 ha), mean and median camp area, number of medium (≤100 ha camps) and highly (≤50 ha) intensive farm portions, and the number of medium and highly intensive farms portions with waterholes. (*: $p < 0.05$)

Factors	R^2	***p*-value**
Fence-length	0.936	0.115
No. of camps	0.871	0.164
<10 ha camps	0.647	0.276
10-50 ha camps	0.812	0.198
50-100 ha camps	0.873	0.162
100-200 ha camps	0.975	0.072
200-500 ha camps	-0.684	0.740
500-1000 ha camps	0.983	0.058
1000-1500 ha camps	0.947	0.104
>1500 ha camps	0.995	0.033*
Mean camp area	0.934	0.117
Median camp area	0.838	0.184
Medium intensive farm portions	0.813	0.198
Medium intensive farm portions with waterholes	0.995	0.033*
Highly intensive farm portions	0.969	0.080
Highly intensive farm portions with waterholes	0.999	0.013*

Table A4. Spearman rank correlation and its *P*-value, between overall accuracy and landscape characteristics (% elevation [<1000 m, 1000-1200 m, >1200 m], % cropland, % urban areas, presence of water) for the Support Vector Machine method for 2007, 2012, and 2017 (*: $p < 0.05$).

	2007		**2012**		**2017**	
Landscape Characteristic	**Spearman rho**	***p*-value**	**Spearman rho**	***p*-value**	**Spearman rho**	***p*-value**
% <1000 m	-0.553	0.002*	-0.124	0.499	-0.253	0.213
% 1000-1200 m	0.499	0.005*	0.138	0.451	0.209	0.305
% >1200 m	0.333	0.072	0.033	0.858	0.351	0.079
% Cropland	-0.079	0.676	0.011	0.954	0.110	0.592
% Urban areas	0.240	0.201	0.247	0.172	0.225	0.270
Water present	-0.630	<0.001*	-0.173	0.343	-0.297	0.140

Table A5. Spearman rank correlation and its *P*-value, between overall accuracy and landscape characteristics (% elevation [<1000 m, 1000-1200 m, >1200 m], % cropland, % urban areas, presence of water) for the Random Forest method for 2007, 2012, and 2017 (*: $p < 0.05$).

	2007		**2012**		**2017**	
Landscape Characteristic	**Spearman rho**	***p*-value**	**Spearman rho**	***p*-value**	**Spearman rho**	***p*-value**
% <1000 m	-0.463	0.010*	-0.476	0.006*	-0.166	0.418
% 1000-1200 m	0.439	0.015*	0.479	0.006*	0.160	0.436
% >1200 m	0.219	0.244	0.091	0.620	0.160	0.436
% Cropland	0.077	0.685	-0.029	0.873	0.076	0.711
% Urban areas	0.092	0.630	0.184	0.315	0.021	0.920
Water present	-0.327	0.078	-0.577	<0.001*	-0.241	0.235