RESEARCH ARTICLE

Fenced in: wildlife fencing intensification in southwest Limpopo, South Africa

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Received 17 May 2023. To authors for revision 11 July 2023. Accepted 8 January 2024

Wildlife farming, or game ranching, collectively termed here as the wildlife sector, has changed over the past few decades in South Africa, leading to growth in conservation and green economy, but resulting in an increase in wildlife fencing and intensive wildlife management practices. To describe spatial changes in the wildlife sector, the fences, camps, and intensive farm portions of wildlife properties in southwest Limpopo, South Africa, were mapped using remote sensing data from 2007, 2012 and 2017. A ground-truth study in 2019 confirmed the accuracy of the remotely-sensed maps. The number of camps smaller than 100 ha significantly increased, especially very small camps <50 ha, from 2007 to 2017. Over the ten-year period, farm portions became proportionately more intensive. This intensification resulted in an increased number of intensive farm portions within protected areas, critical biodiversity areas, and ecological support areas. An increase in total length of fences and number of camps, and a decrease in the mean size of camps were subsequently observed. These changes coincided with the rise of intensive breeding in the wildlife industry market between 2012 and 2017, after which this part of the sector declined. The increase in fencing and intensive management practices in southwest Limpopo could have implications for biodiversity conservation and planning. The remotely-sensed maps may help ensure sustainable wildlife management practices by aiding conservation policy frameworks and monitoring infrastructure developments.

Keywords: fence, fragmentation, intensive management, Limpopo, remote sensing, wildlife breeding.

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 (Reilly, Sutherland & Harley, 2003; NAMC, 2006; Taylor, Lindsey & Davies-Mostert, 2015). 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, Holden & Collinson, 1999) where privately owned South African livestock and crop farms were transformed into wildlife ranches (Kamuti, 2014). This trend is coupled with a 40-fold increase in the number of wildlife in both private and public conservation areas from the early 1960s (van Hoven, 2015). Wildlife ranching in particular was formally recognized as an agricultural activity by the Department of Agricultural Development in 1987 (van Hoven, 2015). Recent reports suggest that there are now approximately 9000 wildlife ranches in South Africa, covering an area between 170 000 km^2 and 200 000 km^2, and containing 16 to 20 million wild animals (NAMC, 2006; Taylor *et al*., 2015). Privately owned wildlife ranches are particularly important as these are reported to protect almost twice as much land than state terrestrial protected areas (Taylor *et al*., 2015; Skowno *et al*., 2019). Wildlife ranches generally produce an income through either hunting, game meat sales, live animal sales, game viewing, or a combination of these (Luxmoore, 1985; Child *et al*., 2012), which ultimately contribute to South African ecotourism,

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ISSN 2410-8200 [Online only] — DOI: https://doi.org/10.3957/056.054.0001

economy and conservation (Child *et al*., 2012; Taylor *et al*., 2015).

Despite the benefits of the growing wildlife sector, there are several ranching practices that conflict with conservation principles (Cousins, Sadler & Evans, 2010). These include: introducing and breeding exotic and extralimital species (Luxmoore, 1985; Cousins, Sadler & Evans, 2008), selectively breeding colour variants (rare colour phenotypes) and hybrids (Lindsey, Romañach & Davies-Mostert, 2009; Cousins *et al*., 2010; Taylor *et al*., 2015), removing predators to protect valuable game (Cousins *et al*., 2010), and altering landscapes by modifying vegetation structure (Sims-Castley *et al*., 2005) and constructing artificial waterholes (Child, 2013). Such ranching practices usually fall under intensive farming, which can be defined as an agricultural system where animals are maintained in small, fenced areas with food and water in order to harvest by-products (Carruthers, 2008). The profits earned by intensively bred animals can be used to invest in the wildlife enterprise and to motivate landowners to remain in the wildlife sector (Taylor *et al*., 2020). However, it is proposed that intensive wildlife management can have negative consequences, such as the loss of genetic variation, semi-domestication, hybridization, landscape homogenization, killing or removal of predators, enhancement of pathogen spread, habitat degradation, and increased fragmentation (Nel, 2015; Desmet, Nel & Pillay, 2017; Selier *et al*., 2018).

The wildlife sector is inevitably accompanied by fencing, as it is required by landowners to obtain a 'Certificate of Adequate Enclosure' (CAE) to legally own game on their property in South Africa (Government Gazette, 1991; Child *et al*., 2012; Lindsey *et al*., 2012; Nel, 2015; Taylor *et al*., 2015). Fenced wildlife areas 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 from predation, establishing ownership and authority over game, controlling access, reducing human–wildlife conflict, decreasing the spread of disease, and establishing boundaries (Hayward & Kerley, 2009; Gadd, 2012; Lindsey *et al*., 2012). Fence-lines also cause mortality in wildlife, inhibit migration and dispersal, restrict recolonization, increase risk of inbreeding, limit evolutionary potential, alter animal behaviour, exclude wildlife from vital resources and conspecifics, cause local extinction, and lead to habitat degradation (Hayward & Kerley, 2009; Gadd, 2012; Lindsey *et al*., 2012; Selier *et al*., 2018). 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). Yet, there is a deficiency in the literature regarding the potential ecological, economic, and social impacts of fencing (Lindsey *et al*., 2012).

Our study aimed to describe changes in the wildlife sector by quantifying the extent of and changes in fence-lines as well as in the number and size of camps within a wildlife area over a ten-year period in the southwestern Limpopo Province. We manually mapped fences and camps of the wildlife sector and inferred the management system (on a spectrum from intensive to extensive) using remote sensing data. A ground-truth study was also conducted to evaluate the accuracy of the fence and intensive camp maps. It was decided to focus on the Limpopo province in South Africa because it is a popular hunting destination (Warren, 2011), has the greatest number of exempt farms with a CAE than any other province (Taylor *et al*., 2015), and contains 45% of the registered wildlife ranchers in South Africa (Wildlife Ranching South Africa; Schepers, Matthews & van Niekerk, 2018).

MATERIAL AND METHODS

Study area

The study area is in the southwest corner of the Limpopo province, South Africa, and covers an area of approximately 12 883 km^2 (Fig. 1). Limpopo province consists mostly of savanna vegetation and is bound on the north and northeast by the Limpopo River (Reyers, 2004). South African protected areas (PA) overlapped with 19.6% of the study area (South Africa Protected Areas Database, 2021), while critical biodiversity areas (CBA) and ecological support areas (ESA; SANBI, 2017) overlapped 44.0% and 12.0% of the study area, respectively. A PA is defined as an area protected by law to primarily conserve biodiversity (Skowno *et al*., 2019), whereas a CBA is an area that must meet biodiversity targets by persisting in an adequate ecological state, and an ESA is an area that must meet biodiversity targets not met in CBAs by preserving its ecological processes (SANBI, 2017). The Thabazimbi district of the

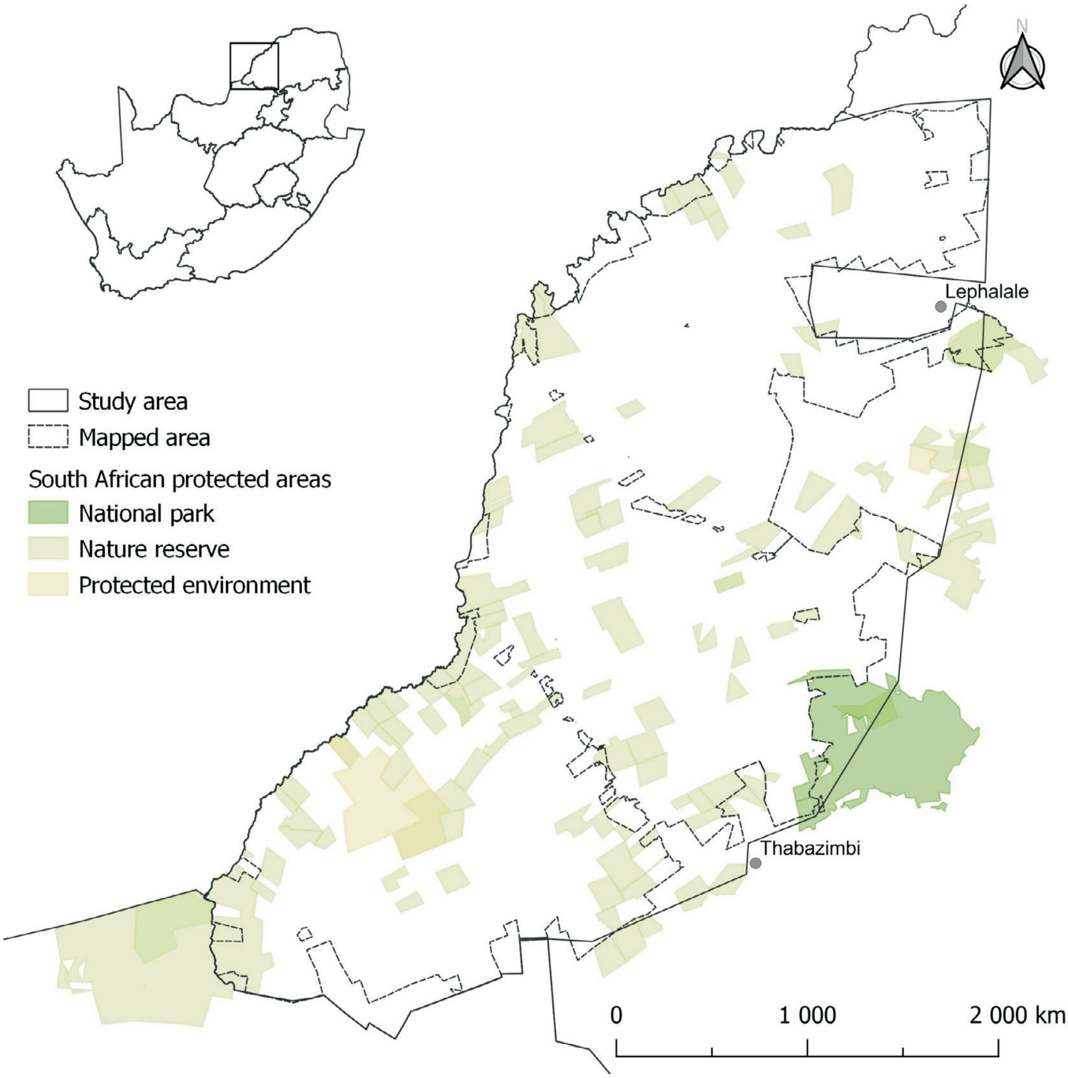

Fig. 1. Study area in the southwest corner of Limpopo province, South Africa, with 104 protected areas (one national park, two protected areas, and 101 nature reserves), where fences and camps were mapped using remote sensing data.

Limpopo province contained mostly privately owned properties that conduct wildlife and cattle ranching (Wilson, 2006), as well as intensive breeding (Desmet *et al.*, 2017). The study area acts as an ecological corridor (LEDET, 2016) with a matrix of protected areas, wildlife ranches, and agricultural farms, and contains priority conservation areas identified for wild dog (*Lycaon pictus*) and cheetah (*Acinonyx jubatus*) (Jackson *et al.*, 2016; Selier *et al.*, 2018).

The methodology used in our study follows a similar methodology by Desmet *et al.* (2017). However, we endeavoured to quantify the change in fences and camps of a larger section of the wildlife sector of Limpopo, and in doing so infer the extent of and changes in the wildlife sector at a broader extent.

Remote sensing data

To assess the extent of and changes in the wildlife sector, we identified fences and camps through satellite imagery and remote sensing products. Satellite Pour l'Observation de la Terre (SPOT) 5 imagery was obtained for 2007 and 2012 from the South African National Space Agency (SANSA) (https://www.sansa.org.za/). Sentinel-2 imagery was obtained for 2017 and 2019 from the European Space Agency (ESA) (https://www.esa.int/ESA). SPOT 5 images from between 15 February and 2 May, and Sentinel-2 images from 5 April and 5 May were used for this study (Table S1). The SPOT 5 satellite was operational between 2002 and 2015, with three multispectral bands (green, red and near infrared) at 10 m resolution, and the image swaths were 60 km × 60 km (Boggs, 2010). The Sentinel-2a satellite was launched in 2015, which has 13 multispectral bands with a spatial resolution of 10 m (Topaloğlu, Sertel & Musaoğlu, 2016), and 110 km × 110 km image swaths, allowing comparisons with the discontinued SPOT 5 (*e.g.* Lopes *et al*., 2017). The above data were used as two separate datasets, one decadal time series (2007, 2012, and 2017) used for the remote sensing assessment to assess changes in the extent of fencing, and one data set collected in 2019 used to verify the remote sensing assessment with ground-truth data of certain areas in the defined study area.

Cadastral data of the Limpopo province were acquired from the Chief Surveyor-General (CSG, of the Department of Rural Developments and Land Reform, South Africa; version 1.2) and were used as a reference for mapping the wildlife sector properties. Land cover data of South Africa were obtained from GeoTerraImage (GTI) Pty Ltd. (©GEOTERRAIMAGE – 'Southern African Land Cover 2015' version 4.1), and Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) Global Digital Elevation Model (GDEM) images were retrieved from the online EarthExplorer tool (https://earthexplorer.usgs.gov), which are products of the Ministry of Economy, Trade, and Industry (METI) of Japan and the United States National Aeronautics and Space Administration (NASA). The land cover and elevation data aided in identifying which areas were ideal for mapping wildlife fences and to omit areas where wildlife fencing would be absent (such as residential areas).

To broaden the context of the extent to which the changes in fencing could make an impact, we incorporated two data sources related to biodiversity and conservation; spatial data of protected areas (South Africa Protected Areas Database, 2021 by the Department of Forestry, Fisheries and the Environment) and critical biodiversity areas (CBA; SANBI, 2017). These areas represent a proxy of the associated conservation value of the vast wildlife sector in the study area.

Data processing

Fences were manually mapped in ArcGIS Desktop 10.5 (ESRI, 2016), for 2007 and 2012 with SPOT 5 imagery, and for 2017 and 2019 with Sentinel-2 imagery, as well as cadastral and land cover data. Fences were defined as straight lines adjoining with other lines that were often perpendicular or parallel to 'farm portions' (cadastral units that define land boundaries, CSG version 1.2), and which did not taper off or lose shape. Fence-lines were not mapped in farm portions with more than a third agricultural area, nor in mountainous areas (areas >1200m asl, inferred from ASTER elevation data), industrial areas, residential areas, dams and rivers (inferred from GTI land cover data and satellite imagery). Camps were mapped and defined as areas that were completely enclosed by a fence-line, as per above definition, within a farm portion. Therefore, wildlife farms consist of one or more farm portions which may or may not be delineated with fences and contain fenced camps. The farm portions, in which fences and camps were mapped per above criteria, remained fixed over the study period to ensure consistent comparisons between time frames.

A ground-truth study was conducted in 2019 to evaluate the accuracy of the remotely-sensed fence-line and camp classification maps. We verified the occurrence of fences and roads while driving a public road through the study area (Appendix 1). Any additional observations, such as wildlife (including extralimital species and colour variants), and fencing condition and type, were also noted. A confusion matrix, which measures the performance of a classification model, was produced to calculate the accuracy of the remotely-sensed fence map. The intensive farm portions defined during the ground-truth study (based on the presence of intensive camps or intensively bred species) were matched with those of the 2019 remotely-sensed intensive farm portions map (described below) to infer the map accuracy.

To investigate whether there had been a change in camp sizes the camp data were categorized 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 of camps per camp size category were counted. Camp size categories were mostly selected based on the categories used by Desmet *et al*. (2017), as well as the size definitions of intensive breeding camps made by Taylor *et al*. (2015). A chi-square test of independence was used to evaluate whether there were significant differences in the number of camps within the different size categories among the years 2007, 2012 and 2017.

Camps sized <50 ha, referred to as 'highly intensive', and 50–100 ha, classified as 'medium intensive', were extracted and the number, and mean and standard deviation (± S.D.) area measured to reflect changes in the intensive wildlife sector. Only farm portions with two or more camps and/or with two or more intensive camps in 2007, 2012, and 2017, contributed to this assessment.

Farm portions were classified as either 'extensive' (contains none or one >100 ha camp), 'semi-extensive' (contains two or more >100 ha camps), or 'intensive' (contains two or more <50 ha or 50–100 ha camps) – the number and proportions of which were then calculated. Chi-square tests of independence were used to test whether the proportions of 'extensive', 'semi-extensive', and 'intensive' farm portions changed significantly over the time period.

Fence length (km), total number of camps, and camp sizes (ha), as well as the mean ± S.D. of camp sizes, were measured or counted for 2007, 2012, and 2017 (see Desmet *et al.*, 2017). A Kruskal Wallis test was used to test whether camp sizes differed between timeframes. The number of farm portions and intensive farm portions that overlapped with the boundaries of protected areas (PA; South Africa Protected Areas Database, 2021, see the 'National Environmental Management: Protected Areas Act 57 of 2003' for definitions), critical biodiversity areas (CBA) and ecological support areas (ESA; SANBI, 2017) were counted with the 'overlap analysis' tool in QGIS, version 3.22.11 (QGIS Development Team, 2022). We assessed whether the number of intensive farm portions within PAs, CBAs, and ESAs changed significantly over time through Chi-square tests of independence.

All statistics were performed in R version 4.2.3 (R Core Team, 2023) using RStudio version 2023.03.0+386 (RStudio Team, 2023), and illustrated with the ggplot2 package (version 3.4.3; Wickham, 2016).

RESULTS

Camp size changes and intensification

The remote sensing assessment identified a total of 1600 farm portions, encompassing an area of 10 259 km^2, that met the criteria to map wildlife fencing and camps for the 2007, 2012, 2017, and 2019 (see definitions of fences and camps under Methods – Data Processing).

Chi-square tests indicated that the number of camps per camp size categories <10 ha (χ^2 (3) = 1209.2, $P < 0.001$), 10–50 ha (χ^2 (3) = 1638.8, $P <$ 0.001), 50–100 ha (χ^2 (3) = 118.8, $P < 0.001$), and 500–1000 ha (χ^2 (3) = 11.3, $P = 0.01$), differed over the study period. The number of camps smaller than 200 ha increased (including <10 ha, 10–50 ha, 50–100 ha, 100–200 ha), whereas camps larger than 200 ha decreased (including 200–500 ha, 500–1000 ha, and >1500 ha) between 2007 and 2017 (Fig. 2). In particular, the total number of highly intensive camps (<50 ha) increased by 246.6%, whereas the number of medium intensive camps (50–100 ha) increased by 36.2%.

Intensive camps and farm portions

The chi-square tests support the observed intensification, as 'extensive' farm portions (χ^2 (2) = 13.5, $P = 0.001$) and semi-extensive farm portions (χ^2 (2) = 21.7, $P < 0.001$) decreased, and 'intensive' farm portions increased (χ^2 (2) = 65.5, $P < 0.001$) over the study period. Of the 1600 farm portions mapped, 910 farm portions had no change in the number of camps, 676 farm portions had an increase in the number of camps, and 14 had a decrease in the number of camps between 2007 and 2017 (Appendix 2). The total number of farm portions containing two or more camps increased only slightly, from 927 in 2007 to 1055 in 2017. However, the number of intensive farm portions, *i.e.* farm portions with two or more highly or medium intensive camps increased, resulting in farm portions becoming proportionally more intensive (Fig. 3). The overall mean number of intensive camps within intensive farm portions increased from 3.0 in 2007 to 4.7 in 2017), encompassing of an increase in the mean number of highly intensive camps, from 2.7 in 2007 to 6.5 in 2017 and a slight decrease in that of medium intensive camps, from 3.3 in 2007 to 3.0 in 2017.

Changes in fences and camps

The total fence length increased by 20.3% between 2007 and 2017, with the largest increase occurring between 2012 and 2017 (Table 1). Camp sizes differed significantly between 2007, 2012, and 2017 (Kruskal Wallis: $H_{(2)} = 1119$, $P <$ 0.001). The total number of camps increased by 62.1%, and the mean camp size correspond-

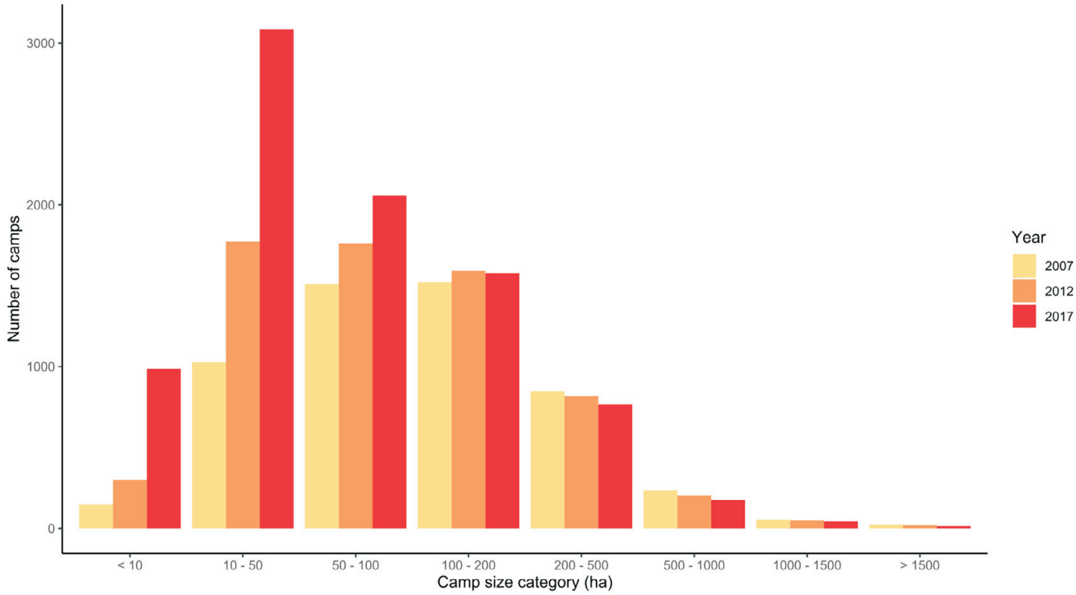

Fig. 2. The number of camps per camp size category (<10 ha, 10–50 ha, 50–100 ha, 100–200 ha, 200–500 ha, 500–1000 ha, and >1500 ha) within the study area in 2007, 2012, and 2017.

Table 1. Length of fences, number of camps, camp size (mean ± standard deviation), and the number of farm portions with intensive camps in important biodiversity areas (protected areas, critical biodiversity areas, and ecological support areas) within the study area in 2007, 2012, and 2017.

	2007	2012	2017
Length of fences mapped in study area:			
Total fence length (km)	14 976.2	16 134.4	18 015.3
Increase between years (%)		7.7	11.7
Number of camps and camp sizes:			
Number of camps	5368	6519	8704
Camp size mean ± S.D. (ha)	164.1 ± 217.6	135.6 ± 192.0	101.6 ± 160.6
Number of farm portions with intensive camps in important biodiversity areas:			
Protected areas	180	228	291
Critical biodiversity areas	303	386	500
Ecological support areas	221	280	358

ingly decreased and standard deviation (±S.D.) increased over the study period (Table 1).

Changes related to protected areas and important biodiversity areas

Of the 1600 mapped farm portions, 14.8% fall into a protected area (PA) classification, 73% critical biodiversity area (CBA) classification and 49% ecological support area (ESA) classification, some of which overlap with each other. The numbers of intensive farm portions within PAs increased during the study period (χ^2 (2) = 26.6, $P < 0.001$), with intensive farm portions increasing by 62% (Table 1). Intensive farm portions also increased extensively in the CBAs (χ^2 (2) = 49.4, $P < 0.001$) and ESAs (χ^2 (2) = 33.0, $P < 0.001$) during the study period (Table 2), representing an increase of 65% and 62%, respectively, in ten years.

Accuracy of remotely-sensed maps

The 2019 ground-truth study assessed 92 farm portions with a wide array of different fence

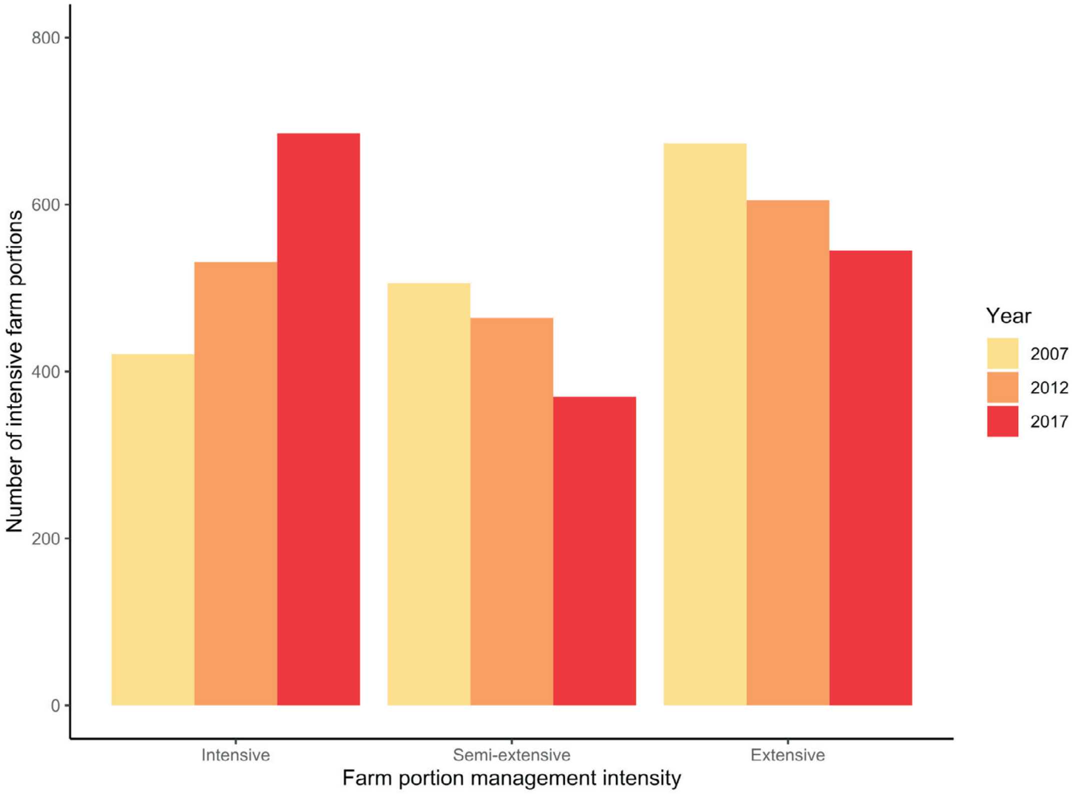

Fig. 3. Total proportions of farm portions (%) of different levels of management intensity (extensive, semi-extensive, intensive) within the study area in 2007, 2012, and 2017. Total farm portions = 1600.

Table 2. Confusion matrix depicting accuracy values for the fence map, including producer's accuracy (false negatives), user's accuracy (false positives), and overall accuracy, where a point is classified as either 'Fence' or 'No fence' (such as vegetation or road).

		Predicted			
		'Fence'	'No fence'	User's accuracy:	
Actual:	'Fence'	257	16	0.941	Sensitivity
	'No fence'	35	30	0.462	Specificity
	Producer's accuracy:	Precision	Negative predictive value	Overall accuracy:	
		0.880	0.652	0.859	

types in different conditions. The confusion matrix revealed an overall accuracy of 85.9% (Table 2). Of these 92 farm portions assessed during the ground-truth study, 49 farm portions were classified as intensive management through either the presence of intensive camps or intensively bred species. Here, 45 out of the 49 intensive farm portions (91.8%) were also classified as intensive through remote sensing.

DISCUSSION

We quantified the extent of and changes in fences and camps within the wildlife sector in southwest Limpopo in 2007, 2012 and 2017. The remote sensing component indicated an increase in the number of camps smaller than 100 ha – especially highly intensive camps (<50 ha) which increased by ~250% – and a decrease in camps sized 500–1000 ha from 2007 to 2017. Our results showed that extensive and semi-extensive farm portions decreased and intensive farm portions increased. Correspondingly, the mean number of intensive camps in intensive farm portions increased, especially that of highly intensive camps. Our results further showed an increase in total fence length and total number of camps over the ten-year period, and the mean camp size subsequently decreased to ~100 ha in 2017 (Table 1). The number of intensive farm portions within protected areas, critical biodiversity areas, and ecological support areas also increased, which may have implications for biodiversity conservation. The increase in fencing and intensive camps observed in the study area is likely to impact the wildlife, habitats, and biodiversity of the area (Lindsey *et al.*, 2012; Desmet *et al.*, 2017; Selier *et al.*, 2018).

The increase in fencing and number of camps, as well as decrease in camp sizes evident in our results, are indicative that intensification of the wildlife sector has occurred. The decrease in mean size of all the camps in study area (~100 ha) is particularly perturbing, as it is less than the mean size of intensive camps (111 ha) surveyed by Taylor *et al.* (2017). Moreover, the intensification of farm portions in our study area comprised more highly intensive camps than medium intensive camps indicating a high density of fences per area, which may lead to habitat degradation, decreased primary production, and subsequently reduced carrying capacity (Gadd, 2012). The extent of extensive farm portions is also decreasing, as it is being replaced by intensive farm portions. The above changes were larger between 2012 and 2017 compared to 2007 and 2012. Similar results were observed by Desmet *et al.* (2017) in southwest Limpopo, and Løvschal *et al.* (2017) in the Greater Mara ecosystem in Kenya. These observations coincide with the expansion of the wildlife industry where the total revenue of South African wildlife auctions increased exponentially in 2012, reached its peak in 2015, and plummeted with the fallen market demand in 2017 (Appendix 3; Cloete, van der Merwe & Saayman, 2015; Thomas, 2017; AWA, 2021). The wildlife market demand has recovered slightly since 2021 (AWA, 2021), and may be influenced by recent legal changes. On 3 March 2023, the high court reversed the recent amendments to the Animal Improvement Act (No. 62 of 1998) of South Africa, which would have permitted the genetic manipulation and cross-breeding of certain wildlife species (Hanks, 2020; Somers *et al.*, 2020; Bega, 2023). However, despite the decreased market demand, wildlife ranchers appeared to have kept their fenced camps as observed by our 2019 ground truth study, which presented >85% accuracy for the fence and intensive farm portion maps (Table 2).

The associated adverse effects of intensive wildlife systems on wildlife and their habitat have been highlighted by Nel (2015), Desmet *et al.*, (2017), and Selier *et al.* (2018), and the increase in intensive wildlife management practices and impermeable fencing has been observed in the period which this study investigated. Subsequently, the study area may experience landscape fragmentation and associated impacts (Nel, 2015; Desmet *et al.*, 2017; Selier *et al.*, 2018). High intensity farming in the study area could be detrimental to the environment, as the area has already been classified with a moderate degradation index (Meadows & Hoffman, 2002). Furthermore, 685 out of the 1600 farm portions were intensive in 2017 (Fig. 3) – of which 291 overlapped with PAs, 500 with CBAs, and 358 with ESAs (Table 2). PAs should only be in areas of environmental conservation (SANBI, 2017). CBAs should preferably be maintained in natural or near-natural ecological conditions and ESAs in at least semi-natural ecological conditions, which include areas of environmental conservation, extensive agriculture, low impact tourism, or open space (SANBI, 2017). Ideally, only ESA2 should have intensive agricultural areas (SANBI, 2017), which is only 84 of the 358 intensive farm portions that overlapped with ESAs in the study area. The high number of intensive farm portions and fences may therefore negatively impact present and future biodiversity conservation planning in the study area.

The effects of fencing and fragmentation can be reduced by improving our understanding of fence impacts and by refining wildlife management plans and applications (Paige, 2008; Gadd, 2012; Taylor *et al.*, 2015). Currently, there are no legal requirements or formal guidelines for fence construction and management in South Africa, only minimum requirements, such as different class species requiring different fence heights (Brown *et al.*, 2014). The regulation of fencing is therefore necessary, as emphasized by Beck (2008) and Desmet *et al.* (2017). Gadd (2012) suggests: creating a spatial database of all fences; frequently monitoring fences; and removing fences or installing cattle grids where disease-control trade-offs are reasonable. Fence removal should be encouraged, as in the case with conservancies where internal fences are removed to form larger wildlife farms that are cooperatively managed (Lindsey *et al.*, 2009, 2012; Taylor *et al.*, 2015). Alternatively, wildlife-friendly fence designs could also be implemented which may result in healthier habitats, improved access to resources (Paige, 2008), and reduced animal mortalities and injuries (see Beck, 2008; BirdLife South Africa, 2018 for examples). The acceptance and application of the above suggestions are likely to be a slow process, as wildlife ranchers rely heavily on fencing for wildlife management (Lindsey *et al.* 2012).

The true extent and ecological impacts of the growing wildlife sector are not well documented (Lindsey *et al.*, 2009; Bothma & von Bach, 2010; Taylor *et al.*, 2015). This calls for more research, monitoring and mitigation measures that can be supported by remotely-sensed maps. Though Selier *et al.* (2019) recommends locating new intensive breeding ranches outside of sensitive areas, such as CBAs and PAs, it would be particularly challenging to do in the study area as the maps show the area consist largely of CBAs and partially of ESAs, and PAs (see Study area). The focus should be to ensure sustainable wildlife management practices, that can contribute to wildlife conservation in a holistic manner (Child *et al.*, 2019), throughout the continuum of intensive to extensive wildlife ranches and protected areas. Wildlife ranching is an economic activity that profits from healthy wildlife and their habitats and can have far-reaching ecological implications should best practice not be implemented (Somers *et al.*, 2020). Therefore, the benefits of implementing sustainable land-use practices within the wildlife sector, hinge on strong ecological and conservation principles that ensures long-term ecological integrity balanced with optimal financial gains.

Remotely-sensed maps, such as fence and intensive camp maps, could be useful for monitoring the wildlife sector and helping to certify that wildlife management practices are done sustainably. The current policy of the Limpopo Department of Economic Development, Environment and Tourism (LEDET), is to only consider exemption for game ranchers from applying for individual land-use and wildlife utilization permits if the farm is more than 400 ha excluding internal fences erected for breeding camps (LEDET, 2016; Selier *et al.*, 2018). However, it is difficult to monitor the compliance to this policy due to lack of resources, which currently enables permits to be renewed without site visitation (Selier *et al.*, 2018). The remotely-sensed maps produced in this study prove to be accurate and reliable, and could be used alongside conservation policy frameworks such as the 400-ha exemption policy of LEDET. Our method could also be applied to other environmental and agricultural practices to assess and monitor infrastructure developments over space and time, such as the subdivision of land through fencing.

In summary, our research found an increase in fencing, camps, and intensive management within the wildlife sector in an area within the Limpopo Province, South Africa from 2007 to 2017. We suggest that this increase in infrastructure may have detrimental effects on the wildlife sector in the long term. We recommend continued monitoring of infrastructure dynamics using maps as demonstrated in this study. We also reiterate the need for re-evaluation of current policies on fencing requirements and intensive management to ensure an ecologically sustainable and financially profitable wildlife sector.

ACKNOWLEDGEMENTS

We thank P.G. Desmet (ECOSOL GIS) for his assistance and information. We would like to thank the reviewers and editors for the valuable contributions and comments that improved the manuscript. We acknowledge those who provided data for this study, namely South African National Space Agency (SANSA), European Space Agency (ESA), Chief-Surveyor General (CSG), GeoTerraImage (GTI), South African National Biodiversity Institute (SANBI), the Department of Environment, Forestry & Fisheries (DFFE), and United States Geological Survey (USGS).

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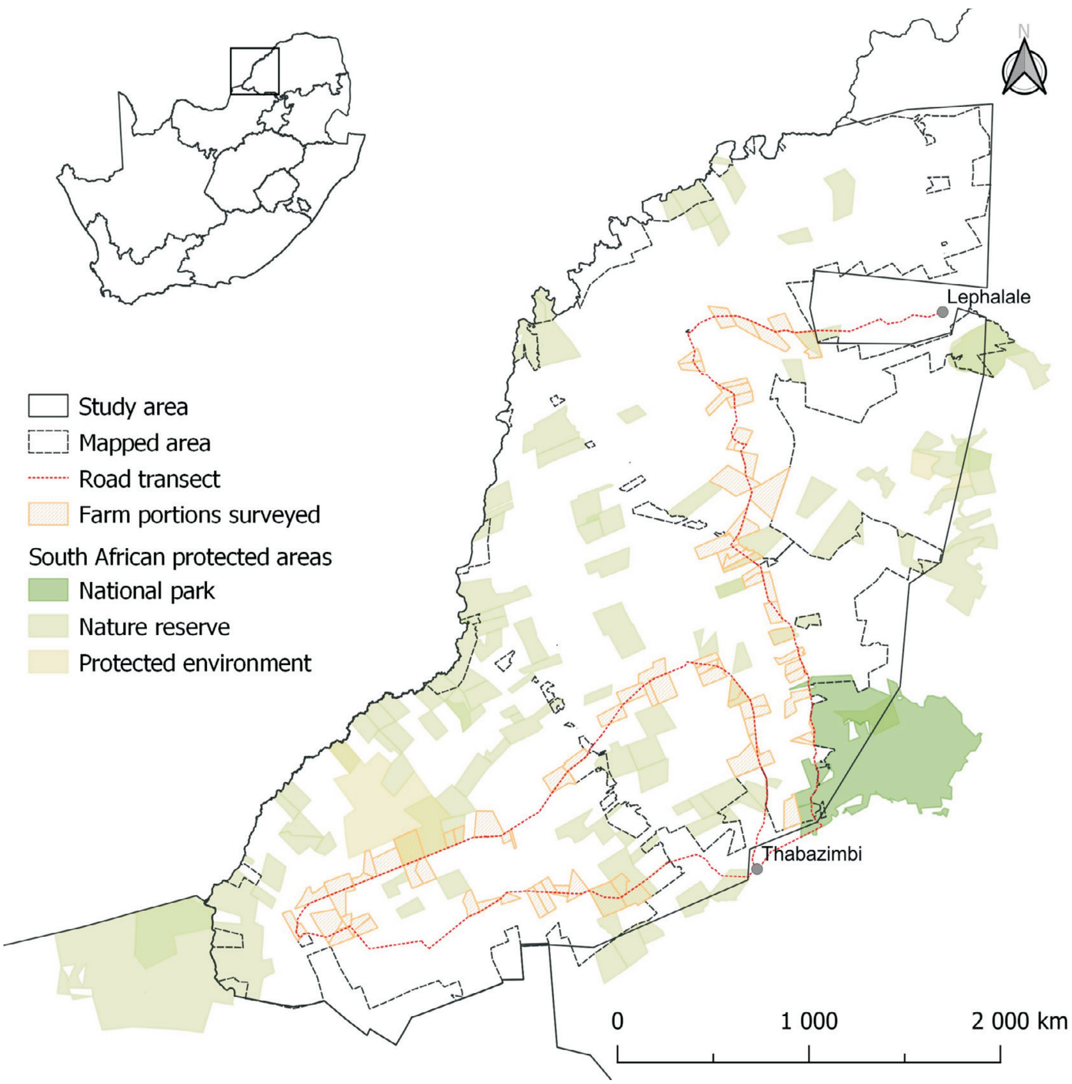
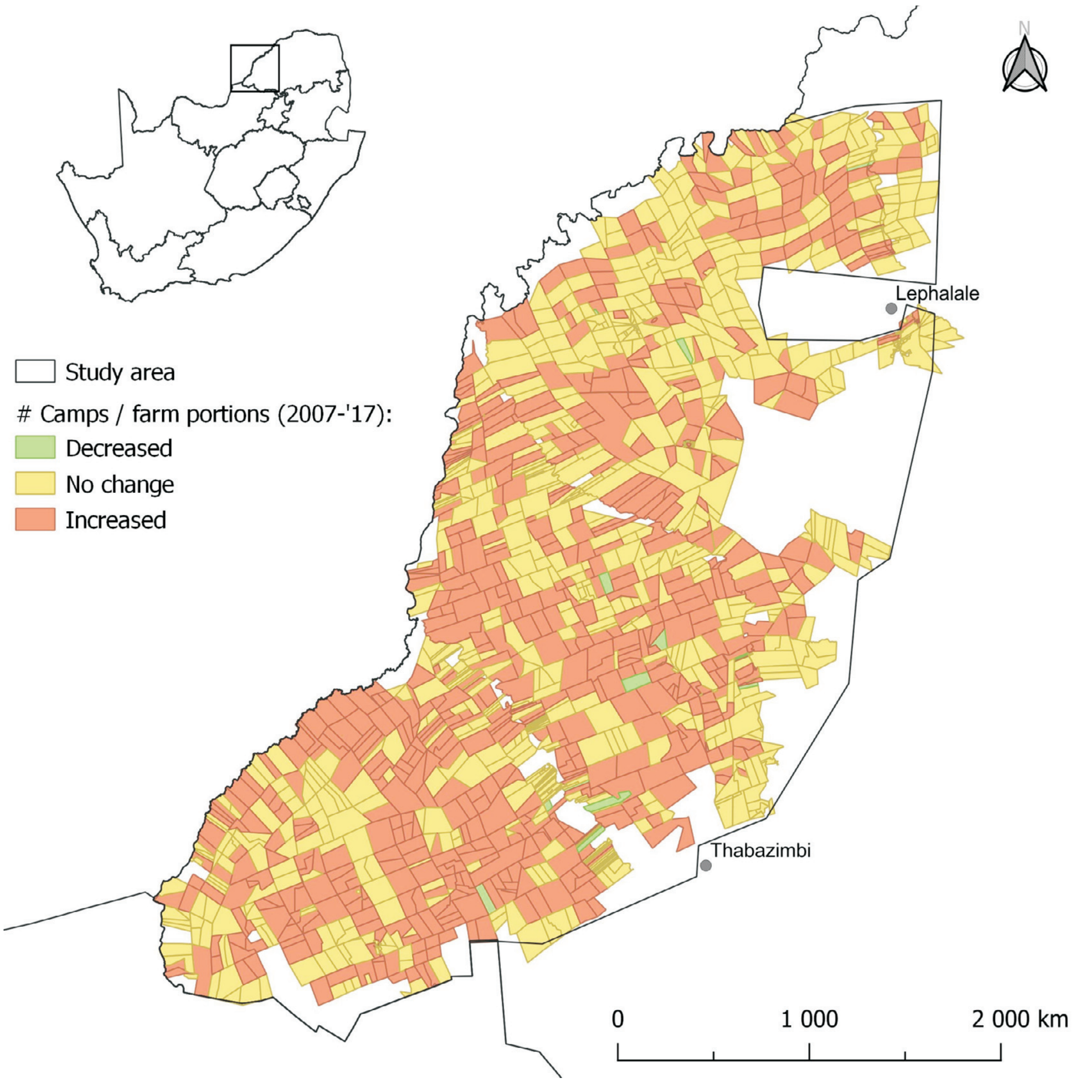
Responsible Editor: F. Dalerum

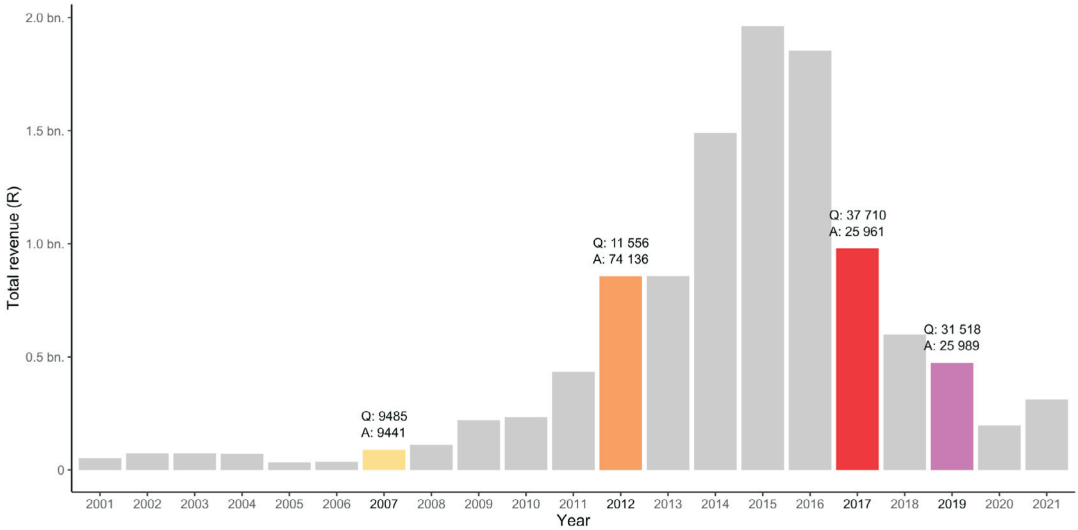
APPENDIX 1. Study area showing the public road transects and farm portions surveyed during the 2019 ground-truth study.



APPENDIX 2. Farm portions where the number of camps either decreased, increased or had no change in the study area during the period of 2007 to 2017.



APPENDIX 3. Total revenue (ZAR) per year, from 2001 to 2021, produced by the South African wildlife industry. Timeframes of study are highlighted (2007, 2012, 2017, and 2019). Q = Quantity of animals sold per year. A = Average pricing per animal per year. Figure modified from African Wildlife Auctions (AWA, 2021).



Supplementary material to:

Cecilia Wagner, Rheinhardt Scholtz, Lizanne Nel & Mark Keith

Fenced in: wildlife fencing intensification in southwest Limpopo, South Africa

African Journal of Wildlife Research **54**: 1–14 (2024)

Table S1. Capture dates of satellite images (SPOT 5 and Sentinel-2) and number of images per date in 2007, 2012, 2017, and 2019.

SPOT 5		Sentinel-2	
2007	2012	2017	2019
13-03 (3)	02-05 (4)	05-04 (3)	16-03 (5)
25-02 (1)	01-04 (1)	05-05 (2)	
18-03 (1)	17-04 (3)		
28-04 (1)			
15-02 (2)			