Jobs, game meat and profits: The benefits of wildlife ranching on marginal lands in South Africa

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Highlights

- 80% of private wildlife properties in South Africa conduct consumptive land uses
- Intensive breeding covers 5% of the total land area of private wildlife properties
- Profitability is highly variable between properties, with an average ROI of 0.068
- Wildlife properties employ more people per unit area than livestock farms
- The private wildlife sector is a financially sustainable land use option on marginal land

Abstract

The private wildlife sector in South Africa must demonstrate value in the face of political pressures for economic growth, job creation and food security. Through structured survey questionnaires of landowners and managers from 276 private wildlife ranches, we describe patterns of wildlife-based land uses (WBLUs), estimate their financial and social contributions and compare these with livestock farming. We show that 46% of surveyed properties combined wildlife with livestock, 86% conducted two or more WBLUs and 80% conducted consumptive use activities. Intensive breeding was conducted on 46% of properties and covered 5.1% of their total land area. Revenues were higher on wildlife only properties than livestock farms, but we were unable to compare the profitability of wildlife and livestock due to data gaps for livestock. Profits from WBLUs were highly variable, while mean return on investment (ROI) was 0.068. Wildlife properties employed more people per unit area than livestock farms, properties conducting ecotourism employed more than twice as many people as non-ecotourism properties, and biltong hunting properties employed 50% fewer people than non-biltong hunting properties. Mean game meat production on wildlife only properties was 4.07 kg/ha, while the top producers harvested game meat at a level comparable with some extensive livestock farms. We suggest that the financial and social benefits of wildlife ranching on marginal land make this a viable land use, but that the contributions towards biodiversity conservation need to be quantified. The South African model could be a suitable option for other African countries seeking sustainable land use alternatives.

Keywords: Wildlife-based land use; Ecotourism; Hunting; Breeding; Revenue; Profit, Employment; Game meat

1. Introduction

To safeguard ecosystems necessary for conserving biodiversity, the Convention on Biological Diversity set a target (Aichi Biodiversity Target 11) for countries to conserve 17% of terrestrial areas through effective area-based conservation measures by 2020 (Secretariat of the Convention on Biological Diversity, 2014). Many African countries, including South Africa, have not met this target (Battistella et al., 2019), and may not be able to do so through the expansion of state owned protected areas (PAs). There is growing recognition, however, that some private land effectively conserves biodiversity (Clements et al., 2018), and might contribute towards this target, or towards Aichi Target 7 that aims to conserve biodiversity through sustainable management of areas under agriculture. Unlike state PAs, private land conservation needs to be financially self-sustaining and, unless alternative funds are available to subsidise conservation activities, this requires income generation from biodiversity resources. Legislation in southern African countries like South Africa, Namibia and Zimbabwe facilitates this by giving private landowners user rights over wildlife and by allowing them to generate revenues through sustainable use activities (Bond et al., 2004; Child, 2009).

Wildlife ranching is a form of private land enterprise that uses wildlife-based land uses (WBLU) as a means to generate profits. These WBLUs may include consumptive activities such as trophy hunting, meat ('biltong') hunting and culling, or non-consumptive activities such as selling live animals, breeding and ecotourism (see Table S1, online supplementary materials 1 for definitions of these activities). In South Africa, the wildlife ranching sector has grown rapidly in recent years on marginal lands formerly used for livestock production, and now encompasses an area of 17–20.5 million hectares (NAMC, 2006; Taylor et al., 2016), equivalent to 14–17% of the country's land surface area, which is larger than the state protected area network (Department of Environmental Affairs, 2019). The Game Theft Act (No. 105 of 1991), which gave private landowners ownership of wildlife under conditions of adequate fencing, coupled with post-apartheid changes to agricultural policies that reduced the amount of support provided to farmers by the government (van Zyl et al., 2001; Vink, 2004), were likely key drivers of this growth (Bothma et al., 2009; Snijders, 2012).

More recently, the South African government implemented a national 'Biodiversity Economy Strategy' (BES) (Department of Environmental Affairs, 2016), an initiative aimed at growing economic activities that depend on biodiversity for their core business (Department of Environmental Affairs, 2016). The wildlife ranching sector is well placed to contribute towards and benefit from the BES, which will probably influence land use policy for the foreseeable future given the official support from the President of the Republic (https://www.gov.za/node/782067).

The conversion of agricultural land to wildlife ranching does not have universal support, however, with some questioning the social impacts of deagrarianisation (Snijders, 2012; Spierenburg and Brooks, 2014), the motivation for land use conversions (Fairhead et al., 2012; Spierenburg and Brooks, 2014) or the acceptability of certain commercial consumptive uses of wildlife (Macdonald et al., 2016b). Moreover, by occupying ~20% of South Africa's marginal agricultural land, wildlife ranching may pose an opportunity cost to the economy.

Determining whether wildlife ranching is a productive and acceptable use of land in South Africa requires knowledge of its combined financial, social and conservation impacts, but none of these factors are sufficiently well understood at present to allow such an assessment. From a financial perspective, there is evidence that wildlife can generate considerable revenues through ecotourism (Muir et al., 2011; Sims-Castley et al., 2004), trophy hunting (Lindsey et al., 2007; Saayman et al., 2018) and biltong hunting (Van der Merwe et al., 2014), but these findings do not include profitability estimates. Potential profitability has been estimated using enterprise budget models (Chiyangwa, 2018; Cloete et al., 2015; Cloete and Spies, 2013), whereby expected returns on investment (ROI) vary from 2 to 7%, but actual profit data are limited. Musengezi (2010) showed that the profitability among a small sample of wildlife ranches in Limpopo Province was highly variable, while Clements et al. (2016) found that land use choices had a big influence on the profitability of private protected areas (PPAs) and wildlife ranches in the Eastern and Western Cape provinces. From a social perspective, there are few recent data on employment opportunities and food production, while there are limited documented impacts on conservation. Exceptions include Gallo et al. (2009) who found that PPAs contributed to the conservation of otherwise underrepresented biomes, and Shumba et al. (2020) who showed that PPAs, including ones with informal status such as wildlife ranches, can be effective in conserving natural land cover and biodiversity intactness. Additionally, Clements et al. (2018) showed that PPAs could facilitate the persistence of large mammals.

In this paper we estimate the financial and social contributions of wildlife ranching in South Africa and describe the patterns of land use on such ranches, an aspect of the sector that has not been formally documented. Although our focus is on South Africa, there are broader implications for how private land may play a role in mitigating losses of biodiversity in other countries. Given the immense scale of human impacts on global biodiversity (Balvanera and Pfaff, 2019), there should be no dispute that, where possible, private land be used to help nations reach their conservation targets. How private land may be used to achieve this, however, does not have consensus, with disagreements around the issues raised above. In order to find common ground, we must first agree on what forms of land use are financially sustainable, which land uses make the biggest social contributions, and whether these land uses are effective conservation tools. If we can answer these questions, we will be better able to objectively debate the question of what land uses are acceptable and whether other countries should consider adopting the South African wildlife ranching model.

2. Methods

2.1. Participant sampling

There is currently no national list of private wildlife properties in South Africa (Musengezi, 2010; Snijders, 2012; Taylor et al., 2016), so we constructed a database from the following sources: 1) lists of exempt properties (properties with provincial permits confirming adequate fencing) from North West, Free State and KwaZulu-Natal provinces; 2) lists of wildlife rancher members from five Limpopo Wildlife Ranching South Africa chambers; 3) a list of wildlife ranchers from the Eastern Cape, Limpopo and Mpumalanga held by the Endangered Wildlife Trust; 4) internet searches using terms such as wildlife ranch and game farm; and 5) information taken directly from roadside signs. We augmented our list by asking survey participants for contact details of other landowners.

We compiled a database of 1540 wildlife landowners/managers from which we contacted 427 telephonically to request their participation between September 2014 and January 2017. Although we selected randomly from this list and did not have prior knowledge of WBLU activities, we did not have access to the contact details of all possible landowners, which precluded true random sampling. On first contact, we explained the purpose of the surveys and confirmed that the results would be anonymous. To qualify, a property had to generate revenue from wildlife, either as a primary or secondary source of revenue. We considered all types of commercial wildlife utilisation (see Table S1, online supplementary materials 1), including part-time wildlife ranchers and mixed farmers (properties combining a mix of wildlife, livestock and/or crops).

Out of the 427 people contacted, 99 were unwilling to participate, making our non-response rate 23%. We also excluded 52 willing participants because they did not use wildlife commercially. In total, we completed 276 interviews for properties spread between eight of the nine South African provinces (excluding Gauteng) (Fig. 1). Out of the 276 completed surveys, we conducted 124 face-to-face interviews, 140 telephonic interviews and received 12 via email. The email option was only used when requested by the participant.

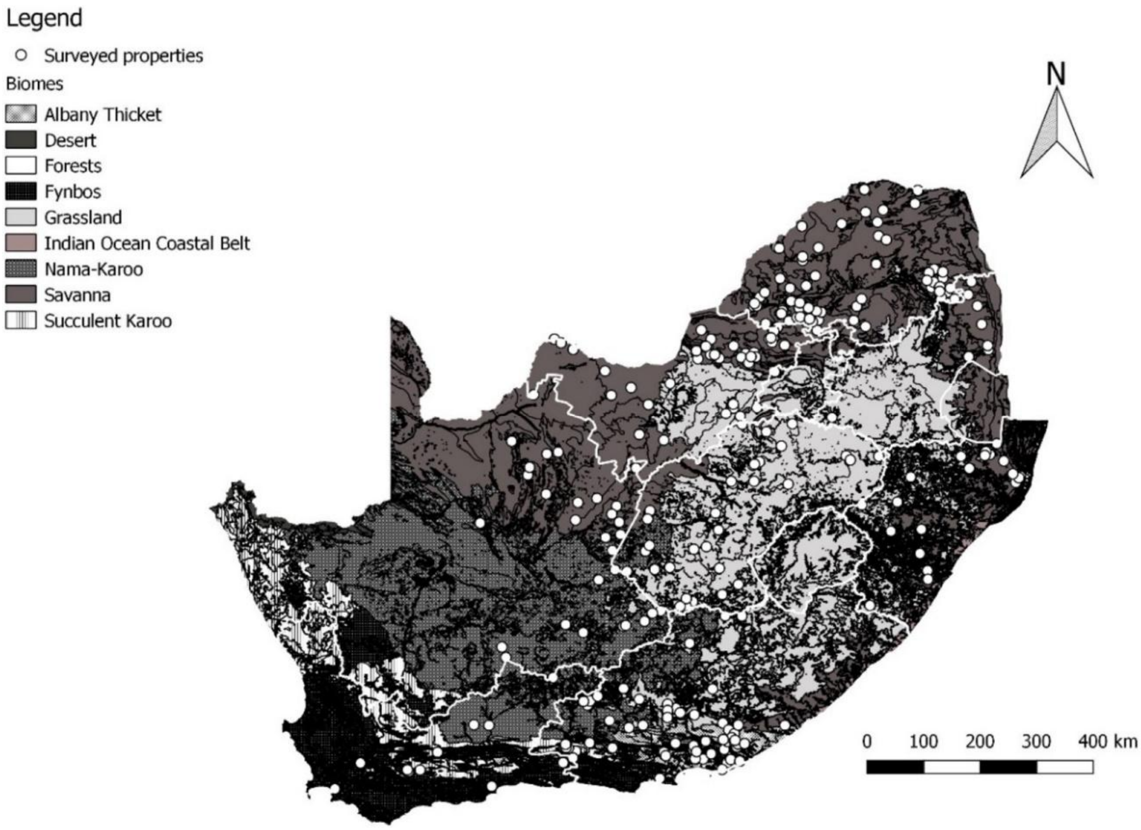

Fig. 1. Locations of surveyed properties across different biomes in South Africa (n = 276).

2.2. Survey questionnaire

We obtained all the data for our analyses using structured, pre-tested questionnaire surveys of private landowners or managers. Our questionnaire was based on similar research assessing WBLUs in Namibia and Zambia (Lindsey et al., 2013a, Lindsey et al., 2013b) (online supplementary materials 2). Many participants were reluctant to answer sensitive questions around finances and, in these cases, we present results from subsets of the data. Additionally, due to the length of the questionnaire, some willing landowners did not have enough time to complete the entire survey, so we developed a short version of the questionnaire to obtain basic information on land uses. Overall, we conducted 193 long questionnaires and 83 short questionnaires.

2.3. Financial and social measurements

We measured direct use values – those derived from the actual use of an environmental resource, including consumptive and non-consumptive uses, and involve the production and/or consumption of marketable products that can be measured through income and spending (Barnes, 2013). We did not measure indirect uses, option, bequest or non-use values. Following Barnes (2013), the direct use values for each wildlife operation were:

1) Total annual revenues for the most recently completed financial year. These were provided by the participants if they knew what their revenues were (based on their income statements) and were willing to divulge them. With mixed farming we asked participants to specify total revenues for all activities and revenues from wildlife only. Values are in United States Dollars (USD) using exchange rates of USD 1: ZAR 12.7 in 2015 and USD 1: ZAR 13.7 in 2016. 2015 revenues were adjusted to 2016 values using average annual consumer price indices from (https://www.bls.gov/data/inflation_calculator.htm; accessed 31 March 2019).

2) Total annual running costs (defined here as all the direct expenses associated with running a property excluding capital expenses and loan repayments). Some participants were not willing to provide these in addition to revenue data, which meant that data on profits were limited. With mixed farming it was difficult to separate total running costs for all activities (livestock plus wildlife) from wildlife only running costs because some costs are shared between livestock and wildlife, so we used total costs only.

3) Total operating profit. This was calculated as the difference between the total annual revenue and the total annual running costs.

4) Perceived relative profitability. We asked participants to list the order of profitability for different WBLUs on properties with multiple land uses.

5) Return on investment (ROI). This was estimated by dividing operating profit by the sum of property and wildlife values (which we used as a proxy for capital investment following Clements et al. (2016)). Property values were obtained directly from participants, who either had formal evaluations done or were comparing the values of neighbouring properties. Some of these estimates may have been slightly outdated, thus introducing some uncertainty. Wildlife values were estimated by multiplying species counts provided by participants by average species prices. All participants for which ROI was estimated had conducted wildlife counts in the last two years, but counting methods varied between aerial, drive and walk counts, which have variable accuracy. Small antelopes, warthogs and bushpigs were excluded because they are hard to count. Prices for species in extensive areas were obtained from members of the Wildlife Translocation Association (WTA), while prices for intensively bred species were taken from average 2015 auction prices adjusted to 2016 prices using average annual consumer price indices (Table S2, online supplementary materials).
6) Annual revenues from individual WBLUs. For extractive land uses, these were estimated using offtake data provided by participants and comprised numbers of each species hunted or sold multiplied by their prices during the year of offtake. For hunted species, average prices were estimated using data obtained directly from the participants and from participant websites where these existed (Table S2, online supplementary materials). We separated live sales data into animals from extensive areas (where average prices were obtained from WTA members) and animals from intensive breeding (using 2015 auction prices adjusted to 2016 prices). Ecotourism revenues were estimated by subtracting extractive revenue (if these occurred) from total revenue.

7) Game meat production. Game meat is produced by trophy hunting, biltong hunting and culling, and we estimated the total mass of game meat produced per property from these activities using species off-take data. With trophy hunting, game meat is a by-product that is either kept by the landowner or taken by the trophy operator. Trophy animals are normally adult males, and an average of 16% of meat is lost due to body shot placement (M. Van der Merwe, 2012) (Eq. (1)). With biltong hunting, the carcass is taken by the hunting clients, who shoot adult animals of both sexes. We used the mean adult body mass (kg) of both sexes and assumed a meat loss of 16% due to shot placement (Eq. (2)). With culling we assumed that animals of all ages and both sexes are equally likely to be shot and multiplied the mean adult body mass of both sexes by 0.75 to account for the inclusion of young animals (Eq. (3)). We assumed no loss of meat because head shots are used. For all equations, body masses and dressing percentages (the percentage of body mass that ends up as carcass) were taken from Bothma et al. (2010) (Table S2, online supplementary materials). We asked survey participants to indicate how they used the meat and the prices obtained.

$$\begin{aligned}&\text{Total carcass mass obtained from trophy hunting}\\&= \text{Number of animals trophy hunted} \times \text{Mean adult male body mass} \times \text{Dressing}\\&\% \times 0.84\end{aligned} \tag{1}$$

$$\begin{aligned}&\text{Total carcass mass obtained from biltong hunting}\\&= \text{Number of animals biltong hunted} \times \text{Mean adult body mass}\\&\ (\text{both sexes})\times\text{Dressing}\% \times 0.84\end{aligned} \tag{2}$$

$$\begin{aligned}&\text{Total carcass mass obtained from culling}\\&= \text{Number of animals culled} \times \text{Mean adult body mass (both sexes)}\times\text{Dressing}\\&\% \times 0.75\end{aligned} \tag{3}$$

8) Local livelihood value. We measured this as the contribution to local households in terms of numbers of permanent jobs supported, salaries and provision of game meat for rations. We did not separate out skilled vs unskilled staff.

2.4. Sector wide estimates and comparisons with the livestock sector

Where we extrapolated our results for the entire private wildlife sector, we assumed a total area of 20 million hectares, based on estimates from NAMC (2006) and Taylor et al. (2016). These are imprecise estimates limited by our imperfect knowledge of how many wildlife properties there are, how big they are, and in which biomes they occur.

For comparative purposes, we assumed that the most likely alternative land use was livestock farming because most wildlife properties were previously under livestock and because most potential agricultural land in South Africa is only suitable for livestock (Kotze and Rose, 2015). As we did not estimate socio-economic impacts of livestock directly, we obtained statistics from the agricultural literature (Department of Agriculture, Forestry and Fisheries, 2017; Meissner et al., 2013; Statistics South Africa, 2018), which has limitations. While national revenues and meat production from agricultural activities are estimated annually (e.g. Department of Agriculture, Forestry and Fisheries, 2017) these statistics do not all differentiate between livestock types or the manner in which they are farmed (e.g. intensive vs. extensive production). They also do not differentiate between different regions of the country. We could not find recent profitability data for livestock farming, while there is a history of poor quality employment data in the agricultural sector (Liebenberg and Kirsten, 2013; Sandrey et al., 2011). Different methods for estimating employment have been used and the types of employees included in the estimates (permanent vs temporary) are not well defined. There are also no accurate estimates for the total area of land used for commercial livestock, nor for the distribution of different livestock practices, making unit area calculations for revenues and offtakes imprecise.

We estimated hypothetical meat offtakes for extensive cattle production on a per hectare basis using average long-term grazing capacities for four biomes (Tainton, 1999), a large stock unit (LSU) equivalent of 450 kg live weight, an average live offtake weight of 225 kg from every one LSU, and a dressing percentages of 55% (Eq. (4). See Table S3, online supplementary materials.

$$\text{Average meat offtake per ha for extensive cattle farms} \\ = \text{Long} - \text{term grazing capacity (LSU / ha)} \times \text{Average live offtake weight} \\ (225 \text{ kg}) \times \text{Dressing}\% \tag{4}$$

2.5. Data analysis

We were interested in which land use types supported the most jobs. To evaluate this, we conducted a multiple regression analysis using R software (v 3.5.1). The log link function for quasi-Poisson regression was used because the response variable, number of jobs, was over-dispersed. Predictor variables included the land use types (ecotourism, trophy hunting, biltong hunting, culling, live sales from extensive areas, live sales from intensive areas, and live sales from both intensive and extensive areas), as well as three property-linked factors included as control variables because we thought they might influence job numbers. These were: 1) property size; 2) biome; and 3) whether a property was a mixed farm or wildlife only. We conducted a model comparison procedure whereby we first ran a full model with all predictor variables, then removed non-significant covariates from the control variables. We then compared the full and reduced models using F-tests. Although we were also interested in which land use types were most profitable, the sample size for profits was too small for rigorous analysis. Because revenues without expenses have limited use (high expenses might negate high revenues), we ran a Spearman correlation test to assess the relationship between revenues and profits on all properties for which profit data were obtained. A high correlation between the two would suggest that revenues may be used as a proxy for profit. See Table S4, online supplementary materials for individual property data.

3. Results

3.1. Patterns of WBLUs

Combinations of commercial wildlife use and various forms of agriculture were common among our sampled properties, with these mixed farms comprising 46% of those surveyed (Fig. 2a). Only 14% of properties specialised in one WBLU, while most conducted two or more (86%) (Fig. 2b). The mean number of WBLUs per property was 3.0 (±1.4). The most commonly practised land use was the sale of live game, while the least common activity was culling for game meat (Fig. 2c). Overall, 80% of properties conducted some form of consumptive WBLU.

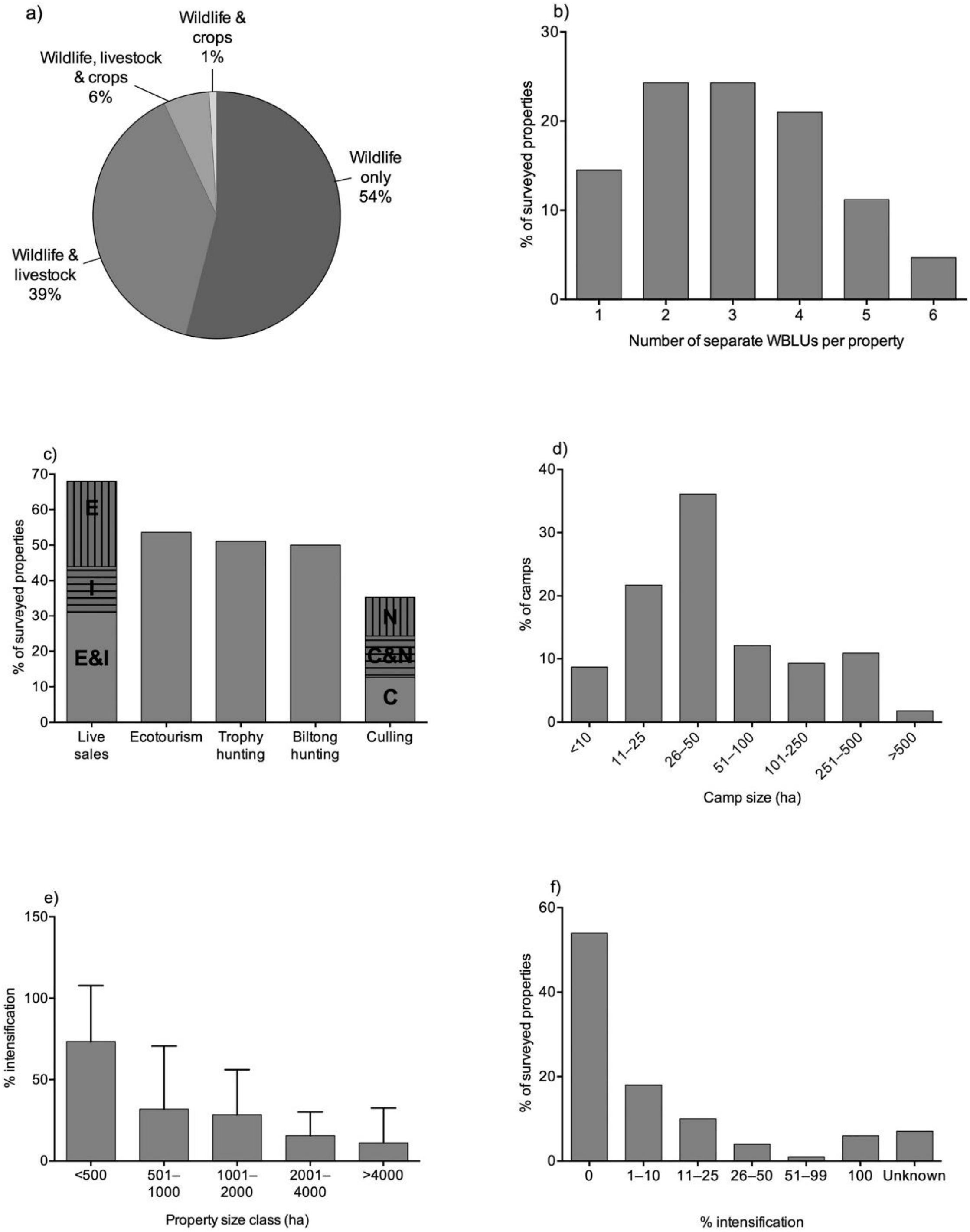

Fig. 2. Wildlife-based land use (WBLU) practises on 276 private commercial wildlife properties: a) percentage occurrence of mixed farmers and properties keeping wildlife exclusively; b) number of WBLU practises per property; c) types of land use per property (note that intensive breeding falls under lives sales here and that the total percentage is >100% because most properties conduct more than one land use). E = live sales from extensive areas; I = live sales from intensive breeding; E & I = live sales from both extensive and intensive areas; C = commercial culling; N = non-commercial culling; C & N = both commercial and non-commercial culling; d) size distribution of intensive breeding camps (n = 803 camps, 109 properties); e) percentage intensification vs property size (n = 109 properties grouped into five size classes) (where percentage intensification = total intensive area / total property area × 100); f) percentage of surveyed wildlife properties vs percentage intensification (n = 276 properties).

Forty-six percent (46%) of properties conducted intensive breeding activities. Mean camp size was 85 ± 132 ha (median = 40 ha) (n = 109) (Fig. 2d), while the total area of camps was 68,300 ha. This equated to 10.4% of the total area of the surveyed properties on which intensive breeding occurred (656,815 ha) and 5.1% of all surveyed properties (1,333,687 ha). Many properties had multiple camps of different sizes, and we used the total area of all camps on an individual property to calculate the percentage intensification for that property (percentage intensification = [total intensive area of property/total property area] × 100).

Considering only the properties with intensive breeding, there was a statistically significant trend for greater intensification on small properties (property size <500 ha: 73% ± 34) than large properties (property size cluster >4000 ha: 11% ± 21) ($H = 40.75_{(4)}$, $P < 0.001$) (Fig. 2e). There was considerable variation in percentage intensification between properties, and 14% were completely subdivided into breeding camps (i.e. they had no extensive area) (Fig. 2f). There was significant overlap between intensive and selective breeding, with 96% of intensive breeders controlling which animals breed in camps (n = 193). Extrapolating our sample results to the entire private wildlife sector (20 million hectares) yielded an estimate of ~1 million hectares of land under some form of intensive breeding camps.

3.2. Financial value

The median total revenue from WBLUs on wildlife only properties was USD 104.3/ha (IQ range: 48.6–455.9) (mean: USD 539.9 ± 1064.5) (n = 34), while on mixed farms median total revenue for all activities was USD 51.0/ha (IQ range: 7.0–192.0) (mean: USD 233.4 ± 478.5) (n = 27) (Fig. 3a; Table 1). The median profit from WBLUs on wildlife only properties was USD 45.8/ha (IQ range: 10.3–229.7) (n = 21), while on mixed farms median profit for all activities was USD 6.5/ha (IQ range: 3.0–40.5) (n = 12) (Fig. 3b). Four (12%) properties made a loss during the financial year preceding the surveys, while one (3%) broke even. Three made profits of >USD 1000/ha, and all these included live sales from intensive breeding. When properties conducting intensive breeding were excluded, 79% of wildlife only properties (n = 14) and 86% of mixed farms (n = 7) were profitable, although the median profits were low (USD 27.2 and USD 5.0 respectively). The median ROI for wildlife only properties was 0.068 (IQ range: 0.014–0.15) (n = 17) but was not calculated for mixed farms (Fig. 3c).

Based on estimated revenues from 139 complete extractive offtake datasets, live sales from intensive breeding on wildlife only properties generated the highest average revenues (median USD 155.9/ha, IQ range: 39.9–488.2), while culling generated the lowest (median = USD 2.9/ha, IQ range: 0.6–5.4) (Fig. 3d). Trophy hunting and live sales from intensive breeding were both the primary revenue earners on 27% of properties, while game meat sales (from culling) was the primary earner on only 3% (Fig. 3e). A Spearman correlation test showed a strong positive correlation between revenues and profits ($r_s = 0.77$, n = 33, $p < 0.001$), suggesting that revenues can be used as a proxy for profits in our sample.

Based on the perspectives of the participants, live sales from intensive breeding was the most profitable activity on 58% of properties on which it occurred with other activities, followed by ecotourism at 40% of properties on which it occurred with other activities. Trophy hunting and live sales from extensive areas followed at 32%, biltong hunting at 23% and culling at just 6%. Of the mixed farmers who provided an opinion (n = 80), 57% thought that WBLUs were more profitable than livestock, while 43% thought the livestock was more profitable.

3.3. Local livelihood values

The median number of permanent employees for wildlife only properties was 0.0040/ha (IQ range: 0.0020–0.0107) (mean = 0.0097 ± 0.0146) (n = 145), while for mixed farms it was 0.0031/ha (IQ range: 0.0016–0.0067) (mean = 0.0077 ± 0.0186) (n = 125) (Fig. 4a; Table 1). Participants on mixed farms indicated that it was common for employees involved in husbandry to work with both wildlife and domestic animals, making it difficult to differentiate between them.

Properties conducting ecotourism employed an average of 2.57 (95% CI: 1.80–3.74) times more people than non-ecotourism properties, while live sales on extensive properties employed an average of 1.64 (95% CI: 1.08–2.61) more people (64% increase) than those not conducting live sales (Table 2). Properties conducting biltong hunting employed 50% (95% CI: 25%–68%) fewer people on average than non-biltong hunting properties.

Table 2. Quasi-Poisson model results for number of employees on all surveyed properties. Included are the full regression model equation, selected model, and statistical results from the selected model of predictors of number of employees. Quasi-Poisson GLM results do not provide an AIC value to compare models, so we compared the residual deviances of different models using F-tests.

	Full model equation
Number jobs	~property size (β1) + biome (β2) + mixed status (β3) + ecotourism (β4) + trophy (β5) + biltong (β6) + culling (β7) + live sales intensive (β8) + live sales extensive (β9) + live sales both extensive and intensive (β10)
	Selected model equation
Number jobs	~property size (β1) + ecotourism (β4) + trophy (β5) + biltong (β6) + culling (β7) + live sales intensive (β8) + live sales extensive (β9) + live sales both extensive and intensive (β10)

Results for selected model	Estimate	Std. error	P-value
Intercept	1.990	0.239	<0.001
Property size	0.00002	0.000003	<0.001
Ecotourism	0.945	0.188	<0.001
Trophy	0.144	0.174	0.409
Biltong hunting	−0.692	0.184	<0.001
Culling	0.157	0.162	0.333
Live sales Intensive	0.010	0.368	0.978
Live sales extensive	0.492	0.213	0.022
Live sales both	0.282	0.221	0.204

The median monthly salary for wildlife only properties was USD 313.3 (IQ range: 243.7–430.5) (n = 77), while for mixed farms it was USD 242.8 (IQ range: 203.1–355.5) (n = 82) (Fig. 4b; Table 1).

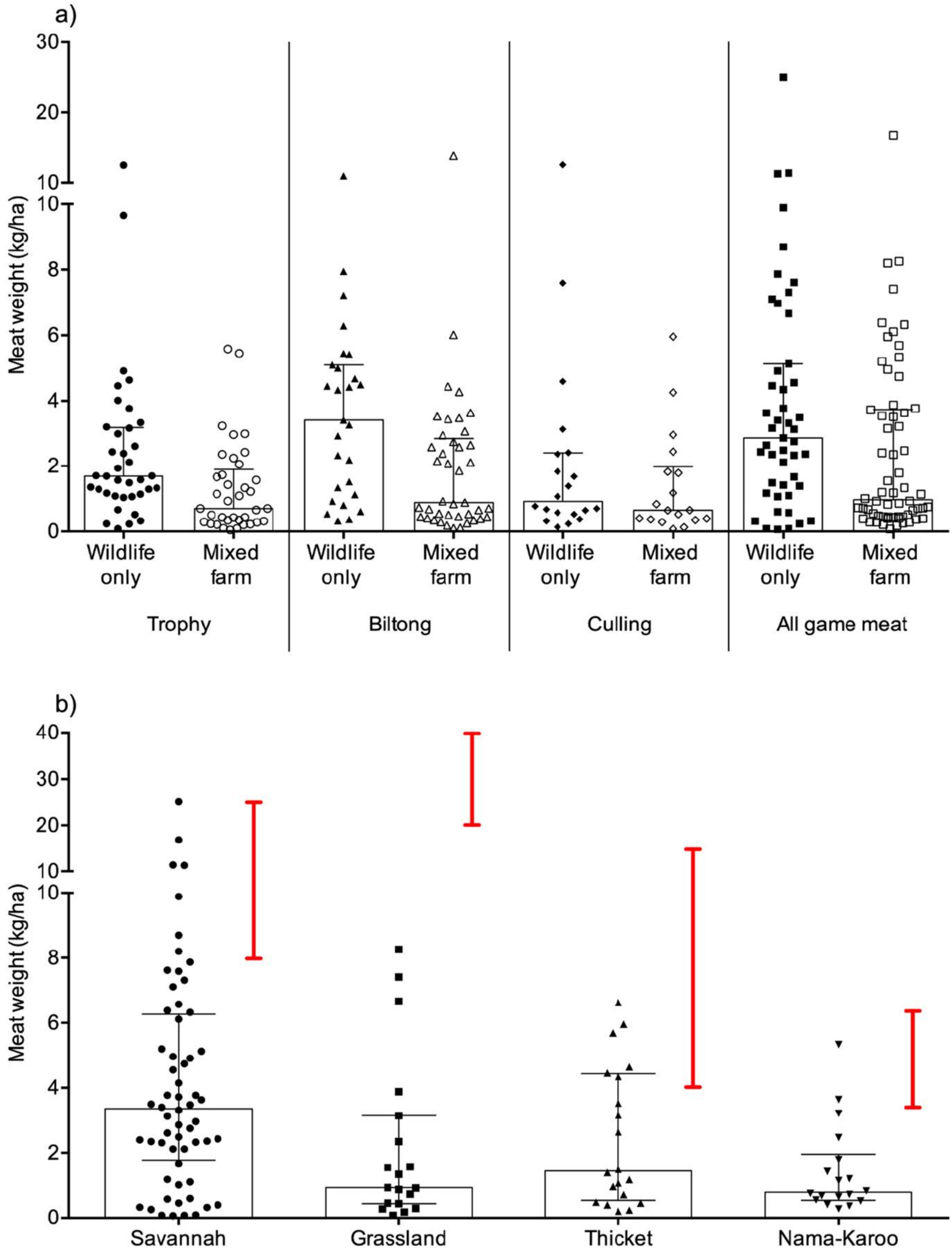

Fig. 5. a) Game meat offtakes (kg/ha) from trophy hunting, biltong hunting, culling and all three consumptive offtakes. Bars and error bars = median and interquartile range. Note that these graphs exclude properties not conducting consumptive use; b) Total game meat offtakes per biome (data points and bar charts) and hypothetical meat production from extensive cattle ranching (red lines showing potential variation).

The median game meat offtake from properties conducting consumptive use was 2.86 kg/ha (IQ range: 1.39–5.13) on wildlife only properties (n = 47) and 0.97 kg/ha (IQ range: 0.46–3.74) on mixed farms (n = 62) (Fig. 5a; Table 1). When separated by biome, game meat production was highest on Savannah properties. Although the average hypothetical meat production from cattle was higher than the average game meat production in all biomes, the offtakes from the top game meat producers overlapped with the lower end of cattle meat production in all biomes except Grassland (Fig. 5b).

Biltong hunters keep all their own meat, while landowners either use the meat from trophy hunting and culling themselves or have a business agreement with the hunting or culling operator. Most game meat from trophy hunting and culling was sold domestically, either through butcheries (58%) or at guesthouses (8%), while 17% was kept for personal use and 17% was sold to or given to workers (n = 105). Overall, 48% of properties provided game meat to employees as rations.

3.4. Sector wide estimates and comparisons with domestic livestock farming

Comparing the financial and social statistics from properties sampled during this study with livestock data from the literature suggests that properties conducting WBLUs generate higher revenues per ha, employ more people and pay higher wages than livestock farms, while the livestock sector produces more meat per ha (Table 1).

4. Discussion

Diversification is common among private wildlife properties in South Africa and allows for different revenue streams and greater financial stability than specialisation (Bond et al., 2004; Lindsey et al., 2013a). This may prove important for the long-term persistence of the wildlife ranching sector, which is still relatively new in comparison to agricultural, and which is subject to market uncertainties such as volatility in live sale prices (Groenewald, 2019) and possible reductions in future revenue from trophy hunting as a result of global ethical challenges to the practice (Macdonald et al., 2016b).

Almost half of our surveyed properties were mixed farms, which is a legacy of the past whereby most current wildlife properties were originally livestock farms that have not fully converted. As conversion from livestock requires a high capital investment (Chiyangwa, 2018; Cloete et al., 2007; Cloete and Spies, 2013; Sims-Castley et al., 2004), it is sometimes only partially completed or may take several years to accomplish. In Namibia, 70% of commercial farms practice mixed farming, with livestock being the main revenue earner on two-thirds of these properties (Lindsey et al., 2013b), while 85% of fenced wildlife ranches in Zambia are mixed farms (Lindsey et al., 2013a). Commercial wildlife use remains at a disadvantage to other agriculture in Zambia because of policies that limit financial incentives for private landowners to keep wildlife (Lindsey et al., 2013a), and this is reflected by the fact that only half these properties use wildlife commercially.

All combinations of WBLUs occurred, including ecotourism and trophy hunting, with consumptive uses being very widespread. This combination of ecotourism and hunting is not unique to South Africa, with data from communal conservancies in Namibia showing complementary benefits in the same conservancies, where ecotourism provides greater employment and household revenue opportunities, and hunting provides quick cash revenue and more in-kind benefits (e.g. game meat) (Naidoo et al., 2016).

The growing global opposition to trophy hunting (Lindsey et al., 2016; Macdonald et al., 2016b) may affect the private wildlife sector in future. Social media campaigns contribute towards social and political change (Macdonald et al., 2016a) and there have been recent movements to ban the imports of trophies (e.g. https://www.theguardian.com/environment/2019/apr/12/ban-the-import-of-hunting-trophies). Based on our findings that a quarter of properties depend on trophy hunting as their main source of revenue, one-third of multi-use landowners conducting trophy hunting believed it to be their most profitable activity, and that 50% of properties use trophy hunting as a source of income, this might have a substantial impact on their ability to remain sustainable. While trophy hunting did not generate the extremely high revenues seen in some of the intensive breeding and ecotourism operations, it provides a steady revenue stream of USD 20–70/ha, has not yet been subjected to the fluctuating prices of the intensive breeding subsector, and does not require the huge capital investment seen in the high-end ecotourism market (Muir et al., 2011).

Intensive and selective breeding, which occurred on nearly half our surveyed properties, have also been the subject of controversy in South Africa. Early concerns centred around potential hybridisation of species, inbreeding and phenotypic manipulation (Hamman et al., 2003), while more recently recognised risks that have arisen in parallel with the growth of intensive breeding include an increase in the extent of predator-proof fencing used to enclose high-value species (Cloete et al., 2015), which results in reduced ecological connectivity (Woodroffe et al., 2014), and a decreased tolerance for predators (Pitman et al., 2016). Depending on the stocking densities of camps, there is also a risk of habitat degradation.

We have not tested the conservation impacts of intensive breeding here, but rather estimated the extent to which they are practiced across the country. Our extrapolated area of ~1 million hectares under some form of breeding camps is equivalent to half the size of Kruger National Park, which implies large expanses of fencing and reduced connectivity among properties. Not all intensive breeding practices are ecologically harmful, however, as there is large variation in percentage intensification between properties. To assess the impacts of these activities, comparative field research is needed. Given that the sale of intensively bred animals is the primary revenue generator for 27% of properties and the most profitable activity on 58% of properties on which it occurred with other activities, some degree of tolerance for the activity by conservation practitioners is needed to allow these properties to remain under wildlife production. A middle ground might be sought whereby landowners are more tolerant towards predators and construct fences that are not entirely impassable.

We found a large amount of variability in profits between properties, mirroring the findings for PPAs in the Western and Eastern Cape provinces of South Africa (Clements et al., 2016) and wildlife ranches in Limpopo Province (Musengezi, 2010). Our median ROI value of 0.068 for wildlife only properties was at the high end of predicted estimates based on enterprise budgets by Cloete et al. (2015), which varied from 0.02–0.07, but we also found large variability in ROI estimates, indicating that, while financial modelling is an important tool for planning, model averages can mask high variability in reality. We were not able to compare profits between wildlife ranching and livestock farming, and without a more detailed breakdown of profits from livestock that takes livestock type, farming intensity and region into account, such comparisons would not be very useful. As a result, although we have shown that wildlife ranching is a financially sustainable form of land use, we cannot currently state whether wildlife is a better financial option than livestock on marginal land.

Two recent studies that have made such comparisons using enterprise budgets showed that the financial performance of wildlife would be similar or slightly better than livestock. In the

Northern Cape Province, Cloete and Spies (2013) found that wildlife harvesting and biltong hunting were inferior to cattle farming (which had an estimated ROI of 0.032), but trophy hunting and sales of high-value species were superior. This concurs with our finding that revenues derived from biltong hunting and culling were lower than for other activities. They concluded that it was not reasonable to argue that the financial performance of wildlife in general was superior to livestock. In the Karoo, Chiyangwa (2018) estimated an internal rate of return of 5.86% for wildlife compared to 4.02% for sheep farming, making wildlife a better investment over the long-term.

The large profits from the sale of intensively bred animals were a major incentive for landowners to keep their properties under wildlife rather than seek other high-revenue business opportunities or convert back to livestock. In cases where landowners are predisposed to good environmental practice, some of the profits obtained from selling intensively bred animals can be reinvested into the wildlife enterprise. We note, however, that live sale prices of many wildlife species have declined considerably since 2017 (Groenewald, 2019). This will substantially reduce the profits from intensive breeding properties relative to our current findings, which might lead to land use changes over time. However, as we found that most properties without intensive breeding were still profitable, this drop in live sales prices does not mean that the sector will cease to be financially viable if the current reduced prices continue. Intensive breeding will still generate profits, but the relative profitability will shift so that trophy hunting and live sales from extensive properties will become more important.

Our employment estimates only counted permanent employees and did not account for temporary workers or indirect jobs created by related sectors such as game fence construction and game capture. On average they were similar to estimates made by Van der Merwe and Saayman (2003) and Musengezi (2010), but lower than Muir et al. (2011), who found employment densities of 0.012 employees/ha on luxury ecotourism properties in the Eastern Cape.

Overall, our sampled wildlife properties employed more than twice the number of people per unit area than livestock farms. We have already highlighted the limitations of the employment data used for the livestock sector, which includes poor historical monitoring (Sandrey et al., 2011) and no estimates of numbers of jobs per hectare. Given that the livestock estimates include employment across different enterprise types with varying levels of intensification, it is likely that if we considered livestock farming on extensive cattle or sheep ranches only, our wildlife employment figures would be even more favourable.

The extent of game meat production varied considerably among properties. Reasons for this include: a) that management objectives of most landowners are not focussed on game meat; b) a lack of formal regulations for processing game meat in South Africa make it expensive to comply with current meat hygiene legislation (Van der Merwe, 2012); and c) harvesting wildlife requires more effort than livestock because the animals run wild and have to be culled in the field (Hoffman et al., 2004). The first two reasons also contribute towards the low average meat outputs from wildlife properties relative to total national livestock production. Given that ~80% of domestic red meat comes from intensive feedlots (Vink, 2004), while our game meat estimates are primarily based on harvesting animals on extensive areas, our livestock estimates from extensive properties separated by biome provide offtake values that are more comparable. These show that game meat offtakes can match the low end of beef production from extensive cattle ranches under some circumstances. A well-regulated game meat scheme that ensures the efficient but safe processing of wildlife carcasses might incentivise wildlife ranchers to increase the efficiency of their game meat production, making it more competitive with livestock (Van der Merwe, 2012).

5. Conclusions

The ongoing lack of accurate information about how many properties conduct WBLUs, where they are located and how big they are, prevents a precise estimate of the overall financial and social impacts of the sector, which impedes effective planning. This needs to be addressed and could be achieved through the development of a secure, centralised electronic database of all wildlife users.

Evidence accumulated from the current study and others in South Africa (e.g. (Chiyangwa, 2018; Cloete et al., 2015; Cloete and Spies, 2013; Musengezi, 2010) and Namibia (Barnes, 2013; Lindsey et al., 2013b) demonstrate the overall financial sustainability of WBLUs on private land where user rights are clearly defined and where policies provide financial incentives for private landowners to keep wildlife. We have also shown that wildlife ranches support more jobs than livestock on marginal land, although activities like ecotourism do so more than biltong hunting. The international threats to trophy hunting and local volatility in live game prices, however, expose some vulnerability that needs to be monitored.

It is yet to be determined what contribution the private wildlife sector makes to biodiversity conservation in South Africa, so this remains a vital area for future research. Wildlife is likely to become increasingly competitive as climate change takes effect and may additionally provide a coping strategy to help South Africa's agricultural sector deal with reduced and more erratic rainfall. If the sustainable use approach of wildlife ranching does prove to be a legitimate way to conserve biodiversity in addition to the financial and social benefits demonstrated here, we suggest that other African countries consider similar conservation models on their private land.

Credit authorship contribution statement

W. Andrew Taylor:Conceptualization, Methodology, Investigation, Formal analysis, Data curation, Writing - original draft. Peter A. Lindsey:Conceptualization, Methodology, Writing - review & editing. Samantha K. Nicholson:Investigation, Writing - review & editing. Claire Relton:Investigation, Writing - review & editing. Harriet T. Davies-Mostert:Conceptualization, Methodology, Formal analysis, Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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