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Retraction Notice

"Population and Trophy Quality Trends of Three Gregarious Herbivores in an Insulated Semi-Arid Savanna Ecosystem (Cawston Ranch, Zimbabwe), 1997–2014" by Victor K. Muposhi, Edson Gandiwa, Stanley M. Makuza, Paul Bartels has been retracted after the Journal was notified by management of the Cawston Ranch (Cawston Block) that the data referenced in the article was used without their authorization.

The article was originally removed from publication by former publisher, MongaBay, which did not issue a retraction notice. After SAGE acquired the journal in September 2016 and transitioned the published articles to its platform, the article was reinstated with the Journal's archive. SAGE is issuing this retraction notice for the article based on the above information.

Research Article

RETRACTED: Population and trophy quality trends of three gregarious herbivores in an insulated semi-arid savanna ecosystem (Cawston Ranch, Zimbabwe), 1997-2014

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Abstract

Long term monitoring of population estimates and trophy size trends is needed to ensure that trophy hunting is sustainable. We explored the influence of trophy hunting on population size and trophy quality of impala (*Aepyceros melampus*), greater kudu (*Tragelaphus strepsiceros*) and sable (*Hippotragus niger*) antelopes from 1997 to 2014 in Cawston Ranch, Zimbabwe. Population estimates of the three species showed a cyclical declining trend, albeit statistically not significant for the three species. Hunting pressure had no significant effect on the population estimates of the three species for the period 1997-2014. Impala population declined (-30 %) between 2003 and 2008 possibly due to increased illegal hunting associated with land invasions during this period. Trophy size of all species declined over time, 2004-2014, (impala (-1.3 %), greater kudu (-3.9 %), sable (-2.6 %)) possibly due to diet quality and loss of genetic variability in these populations. However, trophy size for greater kudu and sable were within the minimum score range of the Safari Club International. We recommend research on genetic variability and inbreeding levels of hunted populations in closed ecosystems to inform adaptive management of trophy hunting as a sustainable conservation tool in small isolated parks in Africa.

Key words: Sustainable utilization, trophy hunting, selective harvesting, game ranching, hunting pressure

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Introduction

Several factors are known to influence population dynamics of wild herbivores through bottom-up and top-down control processes [1, 2]. Bottom-up processes regulate abundance of herbivores in ecosystems through primary productivity, i.e., forage quantity and quality [3]. Primary productivity in ecosystems is determined by rainfall, which affects forage availability for wild herbivores [4-7]. Top-down processes such as predation affect the dynamics and persistence of wild herbivores [2, 3]. However, 'human predation' through trophy hunting [8-10] and illegal off-takes [11, 12] is becoming a more important determinant of wildlife population dynamics in most African savanna ecosystems.

Population declines in wild herbivores in most savanna ecosystems, e.g., Mara region of Kenya [13], northern Botswana [14], Zimbabwe [15-17], are a concern for biodiversity conservation. The animal declines are mostly related to illegal hunting, land use changes, droughts, and isolation of protected areas through fencing [18-20]. However, trophy hunting promotes off-takes of a supposedly small proportion of males from a population [21] and the associated impacts on population dynamics of most polygamous ungulates are expected to be minimal [22].

Trophy hunting refers to hunting by paying clients, who select animals with exceptional phenotypic attributes such as horns, tusks, body size, and skull length, usually in the company of a professional hunter [23]. Generally, potential hunting clients select hunting destinations based on the diversity and quality of trophies on offer [24]. In ungulates, trophy quality is attributed to the dimensions and aesthetic appeal of the animals' horns (horn size) [25]. Persistent hunting of preferred species may precipitate a shift towards more smaller horned individuals within hunted populations [26]. However, if motivation for hunting is mainly exceptional trophy quality, a resultant decline in this desirable attribute may reduce hunter satisfaction [27, 28] and loss of income, to the detriment of conservation and the livelihoods of many local communities depending on wildlife resources.

Although the significance of trophy hunting in Africa is well documented, very little attention has been given to trends in trophy quality in African wildlife species other than lions (*Panthera leo*) [25, 29-32]. In Zimbabwe, a decline in the trophy quality of three hunted gregarious herbivores, impala (*Aepyceros melampus*), greater kudu (*Tragelaphus strepsiceros*) and sable (*Hippotragus niger*) in Matetsi hunting area has been noticed [30]. However, considering the significance and contribution of private conservation areas or game farms for wildlife conservation in southern Africa and Zimbabwe in particular [33], the long-term trends in trophy quality and the resultant population dynamics of hunted species in such closed and insularized ecosystems have received little attention in literature. With increasing pressure on most protected areas in African savannas through land use change and habitat loss [34], private wildlife areas, which are mostly closed and insular environments, may become relics of wildlife conservation in most disturbed ecosystems [35]. However, the increased insularization of wildlife ecosystems through land use changes and habitat loss [36, 37] may cause demographic changes [8, 38] and possible loss of desirable phenotypic traits in wild herbivores [26, 39].

The sustainability of trophy hunting as a conservation tool lies in the long term monitoring of harvested populations [40-42] as well through the use of ecological theory, i.e., maximum sustainable yield or off-take (MSY) [43-45]. If sustainable hunting practices are applied, animal populations are expected to be stable, given that mostly males would be harvested. However, populations may decline due to human disturbances associated with economic decline and extent of illegal hunting [46]. To determine the effect of trophy hunting on population dynamics and phenotypic traits in closed ecosystems [41, 42], we analyzed data on population trends of three gregarious commonly hunted wild ungulates over an 18 year period, which includes a period of land invasions and reforms (2003-2008) in Zimbabwe. We further explored the trophy size trends (11 years) for these three species using a case study from an insularized semi-arid savanna ecosystem in Zimbabwe. We expected that population estimates and trophy quality of hunted ungulates would decline over time due to hunting pressure. We hypothesized that: (1) population estimates of hunted wild ungulates would decline over time and, (2) trophy quality of hunted ungulates would change over time in a closed and insularized ecosystem.

Methods

Study area

The study was conducted in Cawston Ranch, which is 128 km^2 in extent, located in western Zimbabwe (Fig. 1). The ranch was established in 1989 through the conversion of a livestock farm into a game ranch, where trophy hunting has been practiced for more than two decades using ecological and sustainable principles. The ranch is surrounded by resettlement schemes practicing mostly subsistence agriculture and livestock ranching. However, during the period 2003-2008, there was a wave of disturbances in the ranch due to invasion by settlers as part of the agrarian reforms in Zimbabwe, but normalcy was obtained thereafter. The mean monthly rainfall recorded for the period between 1997 and 2014 was 48.11 mm, with no rainfall having been recorded for July and August. Average annual rainfall for the period 1997-2014 was 577.27 mm. The mean monthly temperatures ranges were 23.5°C and 32.4 °C for July and October respectively.

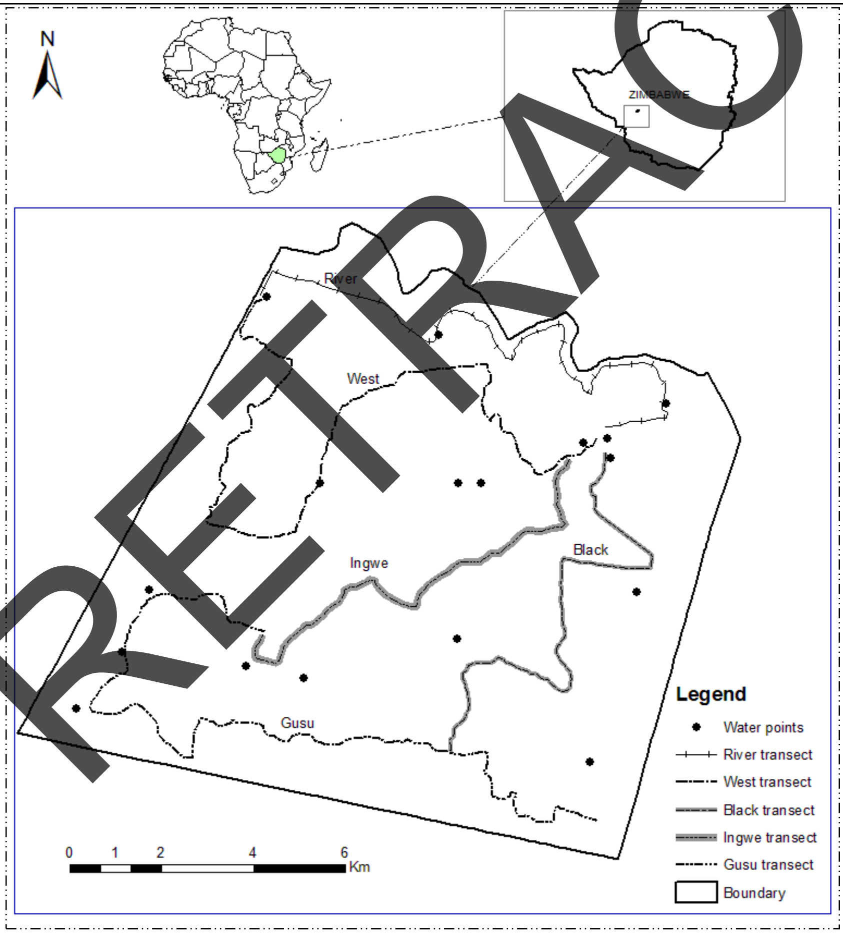

Fig 1: Study area map of showing transects and location of water points in Cawston Ranch, Zimbabwe.

Cawston Ranch is endowed with a number of wildlife species, including impala, greater kudu, sable, giraffe (*Giraffa camelopardalis*), zebra (*Equus quagga*), bushbuck (*Tragelaphus scriptus*), wildebeest (*Connochaetes taurinus*), tsessebe (*Damaliscus lunatus*), eland (*Taurotragus oryx*) and waterbuck (*Kobus ellipsiprymnus*). The ranch's vegetation is generally categorised into about 9 types, namely: *Colophospermum mopane–Acacia–Combretum* woodland*, C. mopane-Acacia–Terminalia* open shrubland*, C. mopane–Acacia* woodland*, Acacia-Terminalia* woodland*, open Combretum* woodland*,* open *Terminalia sericea* woodland*, Acacia karroo,* Teakwoodland and open grassland.

Data collection

Animal population data

The abundance of wildlife species at Cawston Ranch has been consistently monitored annually, using road strip counts following the principles of line transects and distance sampling theory [47]. Cawston Ranch has well established roads in all the common habitats, i.e., vegetation types used as transects for long term wildlife species monitoring. We used secondary data obtained from Cawston Ranch records of annual population estimates for the period 1997-2012 during the dry season between September and November. These transects were consistently surveyed each year using the same sampling techniques with a survey team consisting of one driver, two counters and one recorder. To supplement the secondary data, we conducted road strip counts for the years 2013 and 2014 using the established five roads-routes as transects (see Fig. 1), as with the previous Cawston Ranch surveys. This consistency in the survey approaches made it possible to compare changes in abundance for different time periods.

Each transect was replicated six times during each survey period. The average sampling effort for each transect in the survey design used for the period 1997-2014 is shown in Table 1. For each sighting, the following were recorded: (a) species, (b) group size, (c) distance from vehicle to the centre of the group using a rangefinder, (d) angle of the group sighting at its centre relative to the direction the vehicle was travelling, and (e) distance covered on each transect. Transects were surveyed in the morning (07:00-10:00 hrs) and late afternoon (15:00-18:00 hrs) [48]. We studied population estimates, animal density, and group sizes of three gregarious and most commonly hunted species at Cawston Ranch, impala, greater kudu and sable antelope for the period 1997-2014. To establish the population estimates and animal density for each species, we used the Distance 6.0 release 2 software [49].

Table 1. Average sampling effort for each transect for the period 1997-2014 in Cawston Ranch, Zimbabwe.

Transect	Total distance (km)	Average distance (km)	Average time (hrs)	Average speed (km/h)
Gusu route	111.40 ± 22.20	15.91 ± 0.12	10.00 ± 1.41	10.97 ± 0.19
Black route	88.55 ± 15.49	12.65 ± 0.65	7.50 ± 0.71	10.75 ± 0.24
West route	143.40 ± 27.29	20.49 ± 1.46	14.50 ± 2.21	9.63 ± 0.41
Ingwe route	85.85 ± 17.04	12.26 ± 0.63	9.00 ± 1.41	9.65 ± 0.42
River route	104.65 ± 20.72	14.95 ± 0.17	12.00 ± 1.41	8.87 ± 0.72

Trophy size data

We collected data on annual off-takes for the three species for the period 1997-2014. Data on off-takes were used to calculate hunting pressure, which we considered to be the number of hunted or cropped individuals (off-take) for a particular year divided by the corresponding population estimate for that year [30]. Secondary data on trophy size were obtained from management records at Cawston

Ranch. Although we looked at information for the period 1997-2014, we used only data for eleven years, i.e., 2004-2014, because data from other years were not complete. Data collected for this study on trophy quality were based on horn length, all measured using the Safari Club International (SCI) system for (a) simple horned species, e.g., impala and sable antelope, and (b) spiral-horned species, e.g., greater kudu (http://www.scirecordbook.org/docs/methods).

Data Analysis

We performed explanatory data analysis on population estimates, animal density, hunting pressure and trophy quality to test whether the normality assumptions were being met, using Shapiro Wilk test and Levene's test for normality and equality of variance respectively. All data on explanatory variables, i.e., population estimates, animal density, group size, hunting pressure and trophy quality were found to conform to the normality assumptions. Prior to conducting the regression analysis, we plotted normal probability plots and studentized residuals to check that they did not violate the normality.

We further grouped data on population estimate and hunting pressure into three time intervals: (a) period before the land invasions, 1997-2002, (b) peak land invasions and agrarian reform, 2003-2008, and (c) post land invasions, 2009-2014, in Zimbabwe. Three main analyses were conducted: (1) regression analysis to establish trends of population estimates for the period 1997-2014 and trophy quality for 2004-2014; (2) one-way analysis of variance (ANOVA) to determine whether there were differences in (i) group size, animal density and population estimates for the three species, and (ii) population estimates and hunting pressure for the three time intervals; and (3) one sample *t*-test to ascertain whether there was a difference in the observed trophy quality, and (i) the known standard trophy sizes for impala: 26.4, greater kudu: 52 and sable: 41.88 inches respectively [32], and (ii) the SCI minimum score for impala: 52, greater kudu: 121 and sable 96 inches (http://www.scirecordbook.org). Bonferonni post-hoc tests were further computed for significant differences observed in the population parameters between the three species and time intervals. All statistical analyses were conducted in IBM SPSS 20 software package (IMB, New York, USA) at 5 % level of significance.

Results

Population parameters and trends

There were significant differences in the group size ($F_{2, 51}$ = 54.25, p = 0.000), animal density ($F_{2, 52}$ = 37.45, p = 0.000) and population estimates ($F_{2, 52}$ = 57.16, p = 0.000) of the three wild herbivores. However, Bonferonni multi comparisons tests showed no significant differences in animal density (p = 0.452) and population estimates (p = 0.301) for sable and greater kudu (Table 2). The population trends for the three species had a seemingly declining cyclic fashion mode (Fig. 2a-c), although this change was not statistically significant: impala (R^2 = 0.01, (β ± SE) = 7.83 ± 26.39, *t* = 0.30, p = 0.771), kudu (R^2 = 0.12, (β ± SE) = -12.44 ± 8.57, *t* = -1.45, p = 0.166) and sable (R^2 = 0.01, (β ± SE) = -4.35 ± 11.31, *t* = -0.38, p = 0.706). No significant differences were recorded in the population estimates for sable ($F_{2, 15}$ = 2.71, p = 0.099) and greater kudu ($F_{2, 15}$ = 3.07, p = 0.076) for the three time periods. However, impala population differed significantly ($F_{2, 15}$ = 4.56, p = 0.028) for the three time periods (Fig. 3a-c). Bonferonni post-hoc tests showed that impala population estimates were significantly different for the periods 2003-2008 and 2009-2014 (p = 0.031), as they dropped by 30 % in relation to the average population for the period 1997-2002.

Table 2. Mean (± SD) group size, animal density and population estimates for impala, greater kudu and sable antelope for the period 1997-2014 in Cawston Ranch, Zimbabwe.

Variable	Impala	Greater kudu	Sable
Group size	7.32 ± 0.87^a	3.31 ± 0.61^b	8.76 ± 2.61^c
Density (animals/km^2)	12.08 ± 4.80^a	3.68 ± 1.52^b	5.19 ± 1.89^b
Population estimate	1628.83 ± 515.06^a	472.83 ± 198.72^b	666.72 ± 242.62^b

Different superscripts in the same row indicate significant differences ($p < 0.05$)

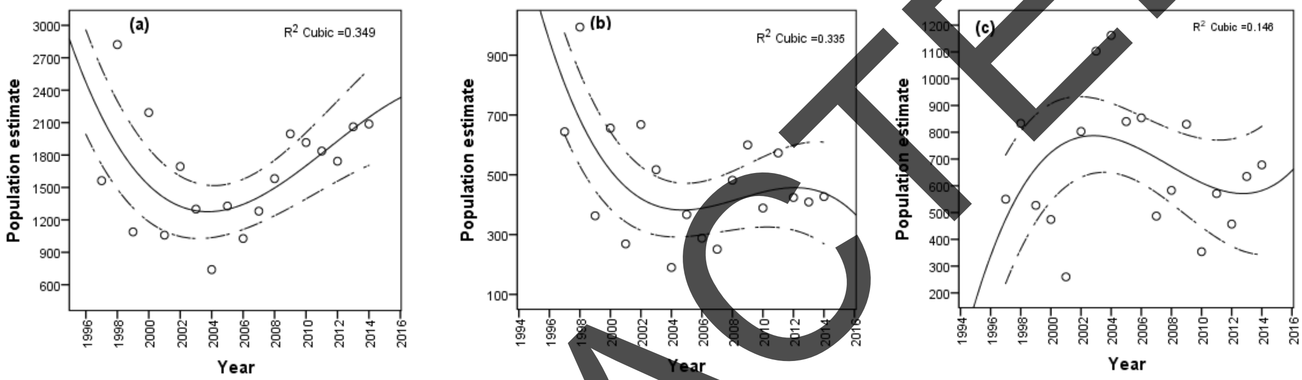

Fig 2. Relationship between population estimates and observation year for(a) impala, (b) greater kudu,(c) sable antelope for the period 1997-2014 in Cawston Ranch, Zimbabwe. Solid line denotes a cubic relationship. Broken dotted line indicates the 95 % confidence interval of population mean.

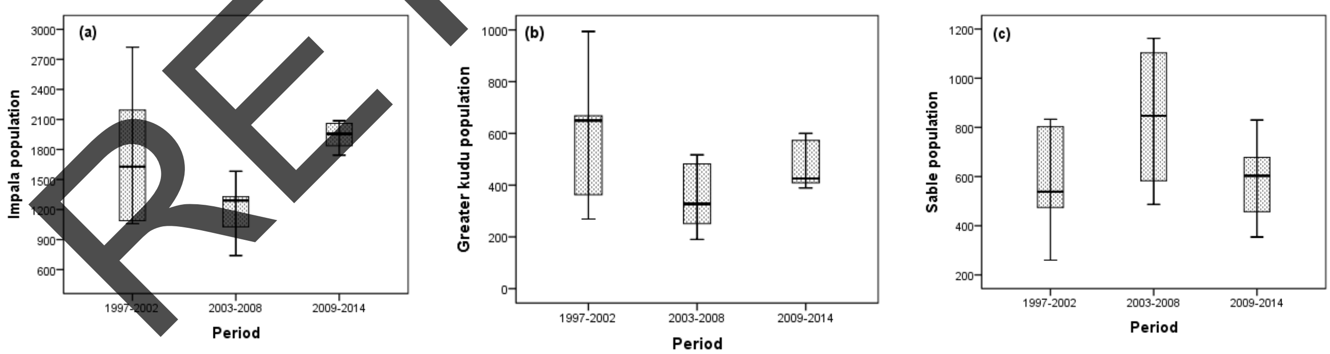

Fig 3. Mean population for impala, greater kudu and sable for the three periods, 1997-2003, 2004-2008 and 2009-2014 in Cawston Ranch, Zimbabwe. Error bars indicate ± SE.

Effect of trophy hunting pressure on population estimates

The trophy hunting pressure on the three species for the period 1997-2004 fluctuated annually, inversely with population estimates of impala, greater kudu and sable antelope. The highest trophy hunting pressure upon impala was during the period 2003-2008, as opposed to greater kudu and sable antelope (Fig. 4). However, there was no significant effect of hunting pressure on the population estimates of the three species, except for the group size of greater kudu ($p = 0.033$, Table 3).

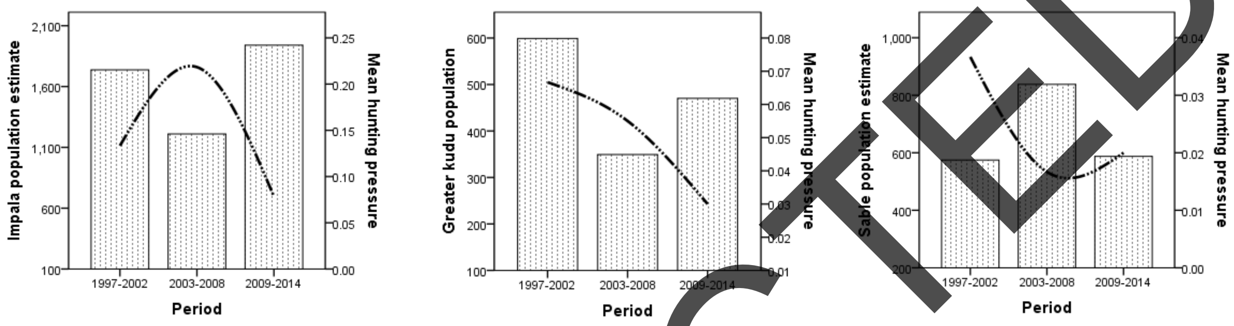

Fig 4. Mean population estimates of impala, greater kudu, and sable antelope in relation to hunting pressure (dotted broken line) across the three time periods in Cawston Ranch, Zimbabwe.

Table 3. Parameter estimates showing effect of hunting pressure on the group size and population estimate for impala, greater kudu and sable antelope in Cawston Ranch for the period 1997-2014. Significant values ($p < 0.05$) are indicated in bold.

Species	Group size			Population estimate		
	β ± SE	*t*	p-value	β ± SE	*t*	p-value
Impala	-0.002 ± 0.001	-1.45	0.167	-0.947 ± 0.926	-1.02	0.323
Greater kudu	-0.013 ± 0.006	-2.35	**0.033**	2.142 ± 2.193	0.98	0.344
Sable	-0.088 ± 0.043	-2.04	0.060	0.731 ± 4.534	0.16	0.874

Trophy size trends

The observed average trophy size for impala (mean ± standard deviation, SD: 20.31 ± 0.59) was significantly below the expected standard trophy size ($t_{10} = -34.22$, $p = 0.000$). There was a notable decline (1.3 %) in the impala trophy size trend over the 11 year period ($R^2 = 0.53$, β ± SE = -0.13 ± 0.04, $t = -3.19$, $p = 0.011$, Fig 5a). Although the observed mean SCI minimum score for impala (51.33 ± 1.43) was not significantly different ($t_{10} = -1.54$, $p = 0.154$) from the standard minimum SCI score of 52 inches, there was a significant decline ($R^2 = 0.56$, $p = 0.008$) in the SCI minimum score during the period 2004-2014 (Table 4). There was a significant difference ($t_{10} = -4.52$, $p = 0.001$) in the observed greater kudu mean trophy size (50.35 ± 1.21) and the expected standard trophy size of 52 inches. Nonetheless, the notably declining trend (3.9 %) in trophy size for 2004-2014 was not significant ($R^2 = 0.30$, β ± SE: (-0.20 ± 0.04), $t_{10} = -1.94$, $p = 0.084$, Fig 5b). The observed SCI minimum score (121.93 ± 2.73) for 2004-2014 was not significantly ($t_{10} = 1.13$, $p = 0.286$) different from the SCI minimum score of 121 (Table 4).

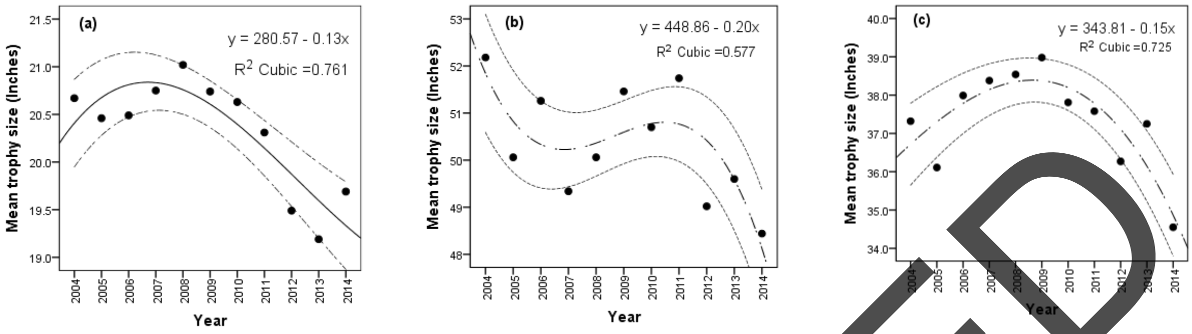

Fig 5: Temporal trend in mean trophy size (with 95 % confidence intervals) for harvested males in Cawston Ranch, Zimbabwe, (a) impala, (b) greater kudu and (c) sable antelopes. Notes: Solid lines indicate significant trends in trophy size over a period 2004-2014. Dotted lines indicate insignificant differences at 5 % level of significance.

Mean sable trophy size (37.34 ± 1.28) was significantly smaller than the expected standard trophy size, 41.88 inches. However, the 2004 -2014 trend in sable trophy size (2.6 %) was not significant (R^2 = 0.16, β ± SE: (-0.15 ± 0.12), t = -1.29, p = 0.229, Fig 5c). The observed SCI minimum score for sable (92.88 ± 3.39) was significantly below the SCI minimum score of 96 inches (t_{10} = -3.06, p = 0.012). Nonetheless, the observed trend in the SCI minimum score was not statistically significant (t_{10} = -1.54, p = 0.157) for the period 2004-2014 (See Table 4).

Table 4. Linear regression model parameter estimates of the observed mean Safari Club International minimum scores for impala, greater kudu and sable for the period 2004 -2014 in Cawston Ranch, Zimbabwe. Significant trends are indicated in bold.

Species	R^2	β	SE	t_{10}	p
Impala	0.56	-0.32	0.10	-3.39	**0.008**
Greater kudu	0.32	-0.46	0.23	-2.04	0.072
Sable	0.21	-0.47	0.30	-1.54	0.157

Discussion

We hypothesized that due to trophy hunting pressure, the population estimates and trophy quality of impala, grater kudu and sable antelope in Cawston Ranch would decline over time. Our results showed a seemingly declining cyclical trend, albeit not statistically significant, on the population estimates of impala, greater kudu and sable. Hunting pressure did not have any significant effect on the population estimates of the three species. However, accounting for the disturbances during period of turmoil (2003-2008) with land reform invasions in Cawston Ranch, only impala showed a significant change in population estimates.

Similar trends where wild herbivores tend to persist without showing a consistent and significant decline have been observed in Gonarezhou National Park, southern Zimbabwe [50, 51], but wild herbivore populations in some countries are declining over time due to over harvesting [52]. Declines in large herbivore populations have also been observed in western Tanzania [53] and the Mara region of Kenya [18]. Generally, documented declines in wild herbivore populations are caused by droughts [54, 55], diseases [56, 57], illegal hunting and overharvesting [46, 58-60] and habitat loss [61]. In Katavi National Park and Rukwa Game Reserve in western Tanzania, illegal hunting was found to be a greater source of mortality than predation, disease, or legal hunting [59]. However, most of these studies were based on large open areas where management interventions are not as targeted as in closed, privately managed game parks such as Cawston Ranch.

Although our results show that hunting pressure associated with trophy hunting influenced greater kudu density, the influence of hunting pressure was not significant for population sizes of the three species. This finding is contrary to the notion that in developing countries hunting quotas are based on inappropriate estimates and not sustainable [62]. However, it is likely that in most private game parks such as Cawston Ranch, trophy hunting and quotas of wild herbivore populations may increase population densities to maximize annual yields, which ultimately promotes biological conservation [63]. It is therefore important to recognize the contribution of these private, isolated and insularized ecosystems as relics of biological conservation in developing countries [33, 64]. We assert that the chances of wildlife population declines in private game ranches are very low, as they are intensively managed with an ecological theory of maximum sustainable yield and harvests [42, 53].

We found the impala population to have declined significantly compared to that of greater kudu and sable during a period of turmoil and land reform (2002-2008), as opposed to periods before and after the peak of the farm invasions in Zimbabwe. This decline could be the result of increased illegal hunting as law enforcement might have lapsed during this time of political turmoil, as was observed in the southeast lowveld of Zimbabwe [65]. We may attribute the decline of impala to the elevated off-take levels during this period, particularly in 2004. Similar observations have been reported in Gonarezhou National Park and surrounding areas during a period of economic decline in Zimbabwe [46]. In the present study, greater kudu and sable populations did not change regardless of the period, possibly due to people preferring impala. This has also been observed in western Tanzania, where most local people are known to hunt small herbivores [59]. We could not get information on illegal hunting activities and patrolling efforts from Cawston Ranch records for this period of turmoil and land reform (2002-2008). In Africa, it is sometimes difficult to get the actual figures on illegal hunting activities and their impacts on wildlife populations [66, 67].

However, drought conditions in this closed ecosystem may have affected its primary productivity and indirectly influenced the population dynamics of the three wild herbivores [68, 69]. It is likely that in insularized and isolated ecosystems, wild herbivores' dispersal ability in times of droughts is restricted, and we would expect population declines for most species [70, 71]. Although rainfall variability influences primary productivity of ecosystems [4, 7] and drinking water availability for wild herbivores [72, 73], due to provisioning at Cawston Ranch, surface water availability to wild herbivores may not be limiting during droughts as has been noted elsewhere [74, 75].

Our study shows that impala trophy SCI scores were within the SCI minimum score limits, although they declined over time. To the best of our knowledge, no previous studies have reported the SCI minimum scores for most species to compare with our findings. Impala trophy size trends declined significantly over time and these are comparable to findings observed in Zimbabwe [30] and South Africa [25]. The contribution of illegal hunting [59] on the declining trophy size trends may be minimal given that subsistence hunters kill indiscriminately and do not select for trophy size [76, 77]. Although trophy sizes for the three ungulates were slightly smaller than expected [32], we note that the habitats and feed quality in Tanzania are different from those found in Zimbabwe. This variation in trophy quality could be attributed to the feed quality variations as influenced by resource allocation towards horn and body growth by these wild ungulates [78, 79]. However, the SCI minimum score for greater kudu and sable in Cawston Ranch did not significantly change over time. This may indicate that the hunting pressure in Cawston Ranch may not necessarily be contributing to the decline in trophy size of some species. We attribute this observation to the use of conservative hunting quotas for animal species by the Cawston Ranch management which does promote ecological and sustainable exploitation principles [43, 80].

a

b

c

Fig 6: Three gregarious antelope species commonly hunted at Cawston Ranch, Zimbabwe; (a) Herd of Impala (*Aepyceros melampus*), (b) Adult male and female greater kudu (*Tragelaphus strepsiceros*), and (c) Adult male and female sable antelope (*Hippotragus niger*). Photo credits: Victor K. Muposhi.

Implications for conservation

The trophy hunting pressure on individuals in Cawston Ranch may be too low to cause a significant decline in the trophy size over time. However, given that the area is closed and dispersal abilities of individuals are limited [70], there seem to be no genetic exchange and inbreeding due to founder effects [81, 82]. The selective nature of trophy hunting, however, may promote the expression of pervasive genes over time, and here we stress the need for active management to prevent genetic erosion [83]. We relate the possible decline in the trophy size to loss of genetic variability, which may result in the expression of undesirable horn size over time, since it is a heritable trait [26, 39, 84]. This challenge could be addressed through introducing breeding males from other areas to reduce inbreeding levels, a management option that Cawston Ranch management are yet to explore. Given that the populations in Cawston Ranch have limited dispersal capabilities due to fencing, introduction of new breeding males would promote genetic diversity in these gregarious and important trophy species (Fig. 6).

Our results show that population dynamics and trophy size trends do play an important role as indicators of management interventions in closed and insularized ecosystems. Thus, with active management and ecologically sound interventions, closed ecosystems may be relics of biodiversity conservation in Africa, where habitat loss and fragmentation are increasing due to demand for agricultural land. We recommend studies on the genetic variability and levels of inbreeding on hunted populations in closed ecosystems, as this selective nature of harvesting may have some influence on the sustainability of trophy hunting as a conservation tool in Africa.

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