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Commodity crops in biodiversity-rich production landscapes: Friends or foes? The example of cotton in the Mid Zambezi Valley, Zimbabwe

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ABSTRACT

The production and trade of agricultural commodities is a major driver of the loss of tropical biodiversity. In the Mid Zambezi Valley, Zimbabwe, which is home to many emblematic African mammals, cotton production has historically been a major driver of land cover change. The collapse of cotton production in Zimbabwe over the last decade provides a unique opportunity to understand the linkages between the profitability of cotton and land-use changes in this multifunctional landscape. By re-visiting 141 households that had been surveyed in 2007 and combining this panel survey data with a land cover analysis and secondary data, we demonstrate that the decreasing profitability of cotton led to a shift from cotton farming (mean area per farm decreasing from 1.79 ± 2.05 ha in 2007 to 0.72 ± 0.90 ha in 2020) to livestock farming (mean number of cattle and goats per farm increasing several-fold between 2007 and 2020), resulting in drastic land cover changes. Indeed, open vegetation (including crops, fallows and grazing areas) expanded from 10 to 20% of the total land cover area between 2007 and 2020. Populations of wildlife species have declined drastically during this period, although this cannot be attributed solely to the observed changes in land cover. However, increasing human-wildlife conflicts are likely to threaten the long-term coexistence of people and wildlife in the area. We argue that commodity crops can be an opportunity for nature conservation, not only a threat, and that conservation needs to support a 'living income' for people coexisting with wildlife.

1. Introduction

Protected areas are critical for biodiversity conservation (Geldmann et al., 2013), but not enough, as they only cover a small fraction of the range of most species (Rodrigues et al., 2004). For instance, 80% of the land area that is of priority for the conservation of mammals is occupied by people and has lost some of its natural vegetation to agriculture (Ceballos et al., 2005). Recognition of the key role of buffer zones and corridors around and between protected areas led to the establishment of many Trans Frontier Conservation Areas (TFCAs) in southern Africa (Andersson et al., 2013; Munthali, 2007). Such TFCAs are central to conservation of the emblematic large African wildlife species which tend to have large ranges. This raises the central question as to what land uses are suitable for buffer zones and corridors to deliver benefits to both people and wildlife.

The global demand for agricultural commodities – to meet both the needs of national and global markets – is a major driver of land-use change and tropical deforestation, overruling the effect of local population growth (Defries et al., 2010). The major global expansion in cropland over the past decades occurred in developing countries; a trend that is projected to continue (Gibbs et al., 2010; Green et al., 2005; Phalan et al., 2013; Tilman et al., 2001). In fact, agricultural expansion in developing countries is partly fuelled by displacement of agricultural activities from developed countries, where the area of cropland has decreased (Lambin and Meyfroidt, 2011). Developed countries increasingly rely on import of tropical commodities, which is the economic equivalent of exporting ecological impact (Lenzen et al., 2012; Meyfroidt et al., 2010). For example, Europe gained 13 million hectares of forest between 1990 and 2014, but its imports of agricultural commodities were responsible for the conversion of 11 million hectares of forest into cropland outside of Europe, mainly in Brazil and Indonesia (Fuchs et al., 2020).

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Land-use change and deforestation are associated with expansion of oil palm in Indonesia and Malaysia (Vijay et al., 2016), soya production in the Brazilian Amazon and the Cerrado (Lima et al., 2019), and cotton production in African savannas (Baudron et al., 2009, 2011). In addition, the production of agricultural commodities consumes other resources. Whilst cotton in Africa is largely rainfed, most cotton in the world is irrigated: Chapagain et al. (2006) estimated that production of cotton needed for a single pair of jeans required >10 m^3 of water. In addition, although cotton cultivation represents only 2.4% of the global cropland it consumes 11% of the pesticides used worldwide (Kooistra and Termorshuizen, 2006). Thus, the production and trade of agricultural commodities – including palm oil, soya and cotton – has become a major concern of conservation organizations (e.g., WWF, 2012; https://tradehub.earth). Conservationists have recommended that land-use change and deforestation in the tropics should be addressed by focusing efforts on avoiding the expansion of export-oriented agricultural commodities, by increasing yield in already cleared land (e.g., Defries et al., 2010), or more often by encouraging the production of food crops targeting the local market and embracing low-input agriculture based on principles of agroecology (see e.g., Altieri and Toledo, 2011; Fischer et al., 2017). New commitments were made by 141 countries to halt deforestation and land degradation and to promote sustainable land-use at the COP26 in Glasgow in September 2021 (https://ukcop26.org/glasgow-leaders-declaration-on-forests-and-land-use/).

In the Mid Zambezi Valley of Zimbabwe, an area of great importance for the conservation of many emblematic African mammals, cotton production was found to be a major driver in deforestation, with likely negative impact on wildlife habitat (Baudron et al., 2011). Would a reduction in cotton production in this area slow down land cover changes, with potential benefits for wildlife? Or, as predicted by Baudron (2011), would this lead to a shift to less labour-intensive crops and expansion of farmland? Cotton production in Zimbabwe has collapsed over the last decade (Fig. 1A), largely due to the fact that the national cotton market is now dominated by Government-owned companies that repeatedly failed to pay farmers for their produce on time or at the agreed price (https://www.theindependent.co.zw/2021/09/03/inside-the-cotton-value-chain/). We set out to investigate the consequences of the collapse of the cotton sector in Zimbabwe on land cover changes, taking the Mid Zambezi Valley as study case. By re-visiting 141 households that had been surveyed in 2007 we could establish the shifts in farming practices that took place. Combined with key informant interviews, a land cover analysis, and secondary data, including time series monitoring of wildlife populations, this provided a unique opportunity to understand the consequences of changes from intensive to extensive farming on the capacity of a multifunctional landscape to conserve biodiversity.

2. Materials and methods

2.1. Study area

The study area covers five wards (Ward 2, Ward 3, Ward 9, Ward 12, Ward 17) of Mbire District located in northern Zimbabwe between 30°00 and 31°45 longitude East and 16°00 and 16°30 latitude South (Fig. 2). Wards are administrative sub-divisions of districts which comprise about 10–15 villages in the case of Mbire District. Human population is still relatively sparse and there is no land scarcity: in 2012 a total of 26,448 people lived in these five wards that cover a total area of 1400 km^2 (ZimStat, 2012). It is part of the Mid-Zambezi Valley, which is formed by the former floodplains of the Zambezi River between the Victoria Falls and Cabora Bassa Lake, at an average elevation of 400 m above sea level. The natural land cover is deciduous dry savannah, dominated by mopane trees (*Colophospermum mopane* (J.Kirk ex J. Léonard)). To the West, the study area abuts a complex of protected areas formed by Mana Pools National Park, Sapi Safari Area and Chewore Safari Area, and designated as a World Heritage since 1984 (Fig. 2). Although the study area is communal land (i.e., state-owned land designated for small-scale family farming), it hosts significant wildlife populations (Gaidet et al., 2003), and is characterized by high biodiversity: over 40 large mammal, 200 bird and 700 plant species have been recorded (Coid et al., 2001). Mbire was one of the first districts in Zimbabwe to implement the world-renowned Communal Area Management Programme for Indigenous Resources (CAMPFIRE) in 1989 (Taylor, 2009). CAMPFIRE enables local communities, through their local government, to manage wildlife as an economic asset for rural development, and is considered one of the earliest programmes worldwide expressing the 'new conservation approach' (Hulme and Murphree, 1999).

The Mid-Zambezi Valley has a dry tropical climate, with low and very variable annual rainfall (on average between 450 and 650 mm $year^{-1}$) and a mean annual temperature of about 25 °C. Two seasons are clearly defined: a rainy season from December to March and a long dry season from April to November. Cotton, sorghum, maize and groundnut are the major crops cultivated. The study area exhibits a gradient of agricultural intensification running from north-west-west to south-east-east (Baudron et al., 2011). Along this gradient, two geographic zones can been distinguished: 1) at the western end of the gradient, a sparsely populated zone, where tsetse fly remains abundant (Chikowore et al., 2017), large wild mammals are numerous, cattle are very limited in

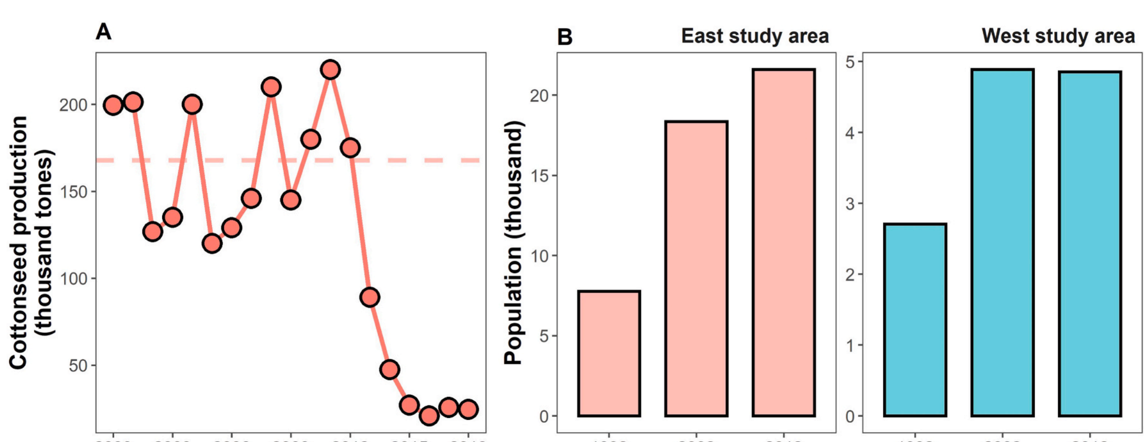

Fig. 1. (A) Change in total cottonseed produced in Zimbabwe from 2000 to 2018 (data source: www.faostat.org/faostat; the dashed line represents mean annual production over the period 2000–2011: 167.7 t $year^{-1}$); and (B) population in 'East study area' (corresponding to Wards 3, 9, 12 and 17 in 2020, Wards 3 and 9 in 2002, and Ward 3 in 1992), and 'West study area' (corresponding to Ward 2) in 1992, 2002, and 2012 (data source: Central Statistical Office, Harare).

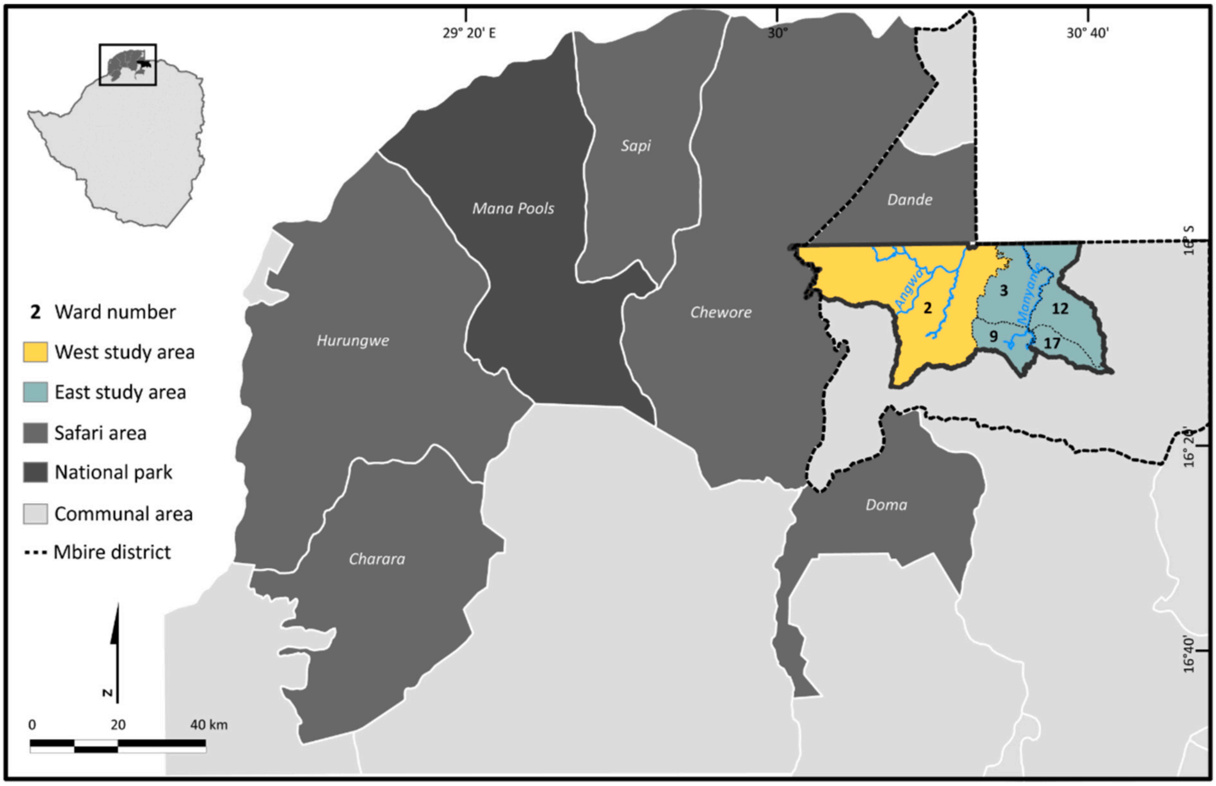

Fig. 2. Location of the study area.

number and cotton is cultivated on relatively small areas, corresponding to Ward 2 and referred to as the 'West study area' below; and 2) at the eastern end of the gradient, a more densely-settled zone where tsetse fly has been eradicated, large wild mammals are few, the cattle population is comparatively large, and cotton is cultivated on large areas, corresponding to Wards 3, 9, 12 and 17 and referred to hereon as the 'East study area' (Fig. 2A).

2.2. Farm survey

Between mid-September and mid-October 2020, the head of the same farming households who had been interviewed in 2007 by Baudron et al. (2011) was re-interviewed using a similar questionnaire. These households had been sampled systematically on three transects along the agricultural intensification gradient described above in 2007 by the same team of enumerators. Each household which had at least one cultivated field on one of the transects had been interviewed in 2007, although some of these households were located several kilometres away from the corresponding fields. From the sample of 176 farms surveyed in 2007, 141 farms were re-surveyed in 2020 (23 households had emigrated from the area, the head of 8 households had died between 2007 and 2020 and these farms had been abandoned, the head of one household in the 2007 database had married the head of another household already included in the 2020 survey, the head of one household was in jail, and the heads of four households could not be found in the area for several weeks). For 88 of the farming households interviewed in 2020, the head was the same as in 2007. For the remaining 53, the head of the household in 2020 was often a son (26) or the wife (19) of the head of the household in 2007. From the 141 farms surveyed, 55 were located in the 'West study area', and 86 in the 'East study area'. The questionnaire addressed size and composition of the household, production capital (e.g., land, equipment), crop and livestock management, income generating and food producing activities, food security, and interactions with wildlife.

2.3. Development of a land-use data base

Three cloud-free and haze-free Landsat satellite images with a spatial resolution of 30 m × 30 m were analysed: one Landsat 5 Thematic Mapper (TM) satellite image from May 2006 and two Landsat 8 Operational Land Imager (OLI) satellite images from April 2014 and April 2020. Three supervised classifications of land cover using a maximum likelihood classifier from a three-channel composition (Bands 4, 3, 2 for the Landsat 5 TM and Bands 5, 4, 3 for Landsat 8 images) were performed, using Envi 5.6 software (www.exelisvis.co.uk). We digitized manually 105 polygons from the 2006 image, 149 from the 2014 image, and 102 from the 2020 image. The 2006 classification was validated by 152 GPS field observations, the 2014 classification by 89 pseudo-ground-truthed points from Google Earth image, and the 2020 classification by 54 polygons that were generated from a Sentinel-2 image (April 2020) at high spatial resolution (10 m).

Supervised classifications were validated from the calculation of confusion matrices and Kappa Index of Agreement coefficients. The 'n-Dimensional visualizer' tool was used to control the absence of confusion between the different regions of interest, i.e., training sites of the land cover classes. The absence of confusion of the pixels allocated within each class was validated for the three classifications using a separability coefficient. The three overall classifications accuracy were 94.0%, 95.4% and 95.0% for 2006, 2014 and 2020 respectively with corresponding Kappa Indices of 0.89, 0.93 and 0.93. The pair comparisons of the land cover classes gave a separability coefficient between 1.87 and 2.00, corresponding to an absence of confusion of the pixels allocated within each class (Girard and Girard, 1999). Five classes of land cover were identified from the supervised classifications: woodland (dense and well-preserved wooded environment), scrub (sparse shrub environment), open vegetation (crops, fallow, and grazing areas), bare ground and water.

2.4. Secondary data: human and wildlife populations

Secondary data related to population were extracted from reports issued by the Zimbabwe National Statistics Agency (ZimStats). In addition, to explore possible effects of land cover changes on wildlife populations in the study area around the period of investigation, we used data of the aerial census conducted by the African Wildlife Foundation (AWF) for the year 2003 (Dunham, 2004), and data of the aerial census conducted by the Great Elephant Census in 2014 (Dunham et al., 2015). Both aerial censuses used similar strata, as well as the same general survey methodology, with sampling done around August–September in parallel transects with a fixed-wing aircraft at a speed of about 160 km h^{-1} and 300 ft. above ground (Dunham et al., 2015). The strata of 'Dande' and 'Chisunga' matched closely the 'East study area' and the 'West study area', respectively. We also calculated estimated numbers for the study area (aggregate of the strata 'Dande' and 'Chisunga'), for the neighbouring protected areas (aggregate of the strata 'Mana Pools National Park', 'Hurungwe Safari Area', 'Sapi Safari Area', 'Chewore Safari Area', 'Charara Safari Area', and 'Doma Safari Area'), and for the sum of the study area and neighbouring protected areas (aggregate of all the strata above), using the method described in Dunham et al. (2015).

2.5. Key informant interviews

To gain an understanding of changes that took place in the area between 2007 and 2020, two sessions of key informant interviews were conducted in October 2021, one in the 'East study area' and one in the 'West study area'. During each session, a representative sample of farmers (in terms of gender, age, etc) who had been interviewed were invited: 14 and 22 attended the key informant interviews in the 'East study area' and the 'West study area', respectively. The interviews focused on the following themes, using open questions and care not to push or guide the conversation: (1) major changes between 2007 and 2020 (including changes in farming systems, in sources of income and food, in land-use and in use of natural resources, including wildlife); (2) perceived causes of these changes; and (3) labour productivity of cotton, cereal, and livestock farming.

2.6. Statistical analysis

When testing for differences between years, medians of quantitative data were compared using Kruskal–Wallis (non-parametric) tests. For the qualitative data, proportions were compared using Chi-square tests. To ensure that observed changes between the two periods were not simply due to a shift along the 'farm development cycle' (family farms tending to pass through different stages – i.e., establishment, growth, maturity, decline, and dissolution – correlated to the age of the head of the household; Chayanov, 1921), we tested the association between observed changes and age of the head of the household in 2020. For quantitative variables, correlations between % change and age of the head of the household in 2020 were tested using Kendall's tau coefficient. For qualitative data, the same assessment was done by testing correlations between change (from 0 to 1, 0 to 0 or 1 to 1, and 1 to 0) and age of the household using generalised linear models with Poisson distribution (3 levels). All analyses were carried out using R (Version 4.1.0). Student tests were used to compare wildlife population sizes estimated by aerial census in 2003 and 2014.

3. Results

3.1. Description of the farming systems in 2020 based on survey results

The mean farm size reported in the 2020 survey was 4.30 ± 2.83 ha, with a mean cultivated area per farm of 2.09 ± 1.50 ha, and a mean uncultivated area per farm of 2.21 ± 1.56 ha (1.29 ± 1.24 ha of fallow, and 0.92 ± 1.24 ha of uncleared land). The mean area per farm of cotton was 0.72 ± 0.90 ha, the mean area per farm of cereals was 1.09 ± 0.84 ha (0.47 ± 0.49 ha of maize and 0.61 ± 0.72 ha of sorghum), and the mean area per farm of other crops was 0.28 ± 0.55 ha (0.15 ± 0.43 ha of sesame and 0.13 ± 0.29 ha of groundnut) (Table 1).

Mean yields were 347.2 ± 284.4 kg ha^{-1} for cotton, 252.3 ± 305.8 kg ha^{-1} for maize, 505.3 ± 435.1 kg ha^{-1} for sorghum, 174.3 ± 208.4 kg ha^{-1} for sesame and 408.3 ± 370.8 kg ha^{-1} for groundnut. These poor yields are explained (in part) by the fact that the 2019–20 season was marked by a long dry spell (over a month) and an early end of the season (around mid-March) (MLAWRR, 2020). For instance, Baudron et al. (2012b) found the mean on-farm yields of cotton and sorghum in the same area to be 722.3 ± 540.8 kg ha^{-1} and 984.6 ± 707.5 kg ha^{-1}, respectively, during three seasons from 2007 to 2010. The mean rate of fertilizer application during the 2019–20 was reported as 99.3 ± 106.0 kg ha^{-1}, 52.9 ± 81.8 kg ha^{-1}, 12.2 ± 42.9 kg ha^{-1}, 4.5 ± 15.2 kg ha^{-1}, and 1.7 ± 10.2 kg ha^{-1} for cotton, maize, sorghum, sesame and groundnut, respectively (Annex 1A).

The mean number of cattle per farm was 6.82 ± 12.86, with 6.41 ± 12.67 indigenous cattle (including 0.91 ± 2.20 oxen) and 0.40 ± 2.14 improved cattle (Table 1). The mean number of goats per farm was 10.06 ± 11.55, with 10.01 ± 11.48 indigenous goats and 0.05 ± 0.51 improved goats. The mean numbers of sheep and donkeys per farm were 1.71 ± 6.54 and 0.17 ± 0.68, respectively. The mean numbers of chicken and guinea fowl per farm were 8.24 ± 9.12 and 1.64 ± 4.35, respectively. The majority (> 50%) of farming households used dip tanks and home vaccines (Annex 1B). However, only a minority (< 50%)

Table 1
Characteristics of the farming households surveyed in 2007 and 2020 ($n = 141$). Means are provided (with standard deviations after the signs '±') for quantitative variables, and proportions (%) for qualitative data. Change (%) between 2007 and 2020 are also given, as well as Chi^2 and P-value for Kruskal–Wallis tests (comparison of medians for quantitative data) and Chi-square tests (comparison of proportions for qualitative tests). Significant effects (P-value < 0.05) are shown in bold.

Indicator	2007	2020	% change	Chi^2	P-value
Age of the head of the household (yr)	46.05 ± 15.74	51.65 ± 15.83	**12.2**	**7.954**	**0.005**
Household size (no.)	6.23 ± 3.22	6.68 ± 3.00	7.2	2.595	0.107
Farm area (ha)	5.25 ± 3.48	4.30 ± 2.28	**−18.1**	**4.3951**	**0.036**
Cultivated area (ha)	3.47 ± 2.72	2.23 ± 1.69	**−35.7**	**24.726**	**< 0.001**
Cotton area (ha)	1.79 ± 2.05	0.72 ± 0.90	**−59.8**	**38.677**	**< 0.001**
Cereal area (ha)	1.27 ± 0.77	1.09 ± 0.84	**−14.2**	**6.962**	**0.008**
Other crops area (ha)	0.22 ± 0.31	0.28 ± 0.55	**27.3**	**5.868**	**0.015**
Non-cropped area (ha)	1.96 ± 2.18	2.21 ± 1.56	**12.8**	**7.089**	**0.008**
Cattle number	1.48 ± 6.15	6.82 ± 12.86	**360.8**	**63.614**	**< 0.001**
Small ruminant number	2.33 ± 4.83	11.77 ± 14.88	**405.2**	**90.178**	**< 0.001**
Food security (months/yr)	7.85 ± 3.55	5.61 ± 2.87	**−28.5**	**26.290**	**< 0.001**
% owning ≥1 pair draught animals	23.40	26.24	12.1	0.171	0.679
% owning ≥2 pairs draught animals	10.64	7.80	−26.7	0.381	0.537
% growing cotton	84.40	58.16	**−31.1**	**22.448**	**< 0.001**
% planning to clear land	37.59	55.32	**47.2**	**8.212**	**0.004**
% guarding fields	65.71	77.30	**17.6**	**4.434**	**0.035**
% crops destroyed by elephant	39.00	56.74	**45.5**	**8.185**	**0.004**
% crops destroyed by buffalo	19.86	6.38	**−67.9**	**10.079**	**0.002**

of them used other improved livestock management approaches. Few farmers produced fodder (6.4%), used concentrate (2.1%) or used improved animal breeds (7.8% for cattle and 1.4% for goats). 14.2% of the interviewed households had experienced death of their own cattle due to African trypanosomosis (disease caused by blood protozoan parasites from the genus *Trypanosoma* and vectored by several species of hematophagous tsetse flies from the genera *Glossina*) in the past years (13.9% in the 'East study area', and 23.5% in the 'West study area'). Cattle and small ruminants were sold to middlemen purchasing animals for the slaughterhouses of the capital city Harare. No other livestock product was marketed to a significant level.

During key informant interviews, a typical farming household composed of five people was said to require more than four hectares of cotton to cover all its cash needs if it had no other source of income. In comparison, less than two hectares of sorghum would be required, and about three heads of cattle. Cotton was said to be twice as labour intensive when compared with sorghum: the same typical household composed of five people would be able to manage two hectares of cotton with its labour only (and without the use of herbicides) but four hectares of sorghum. One person was said to be able to herd up to 50 head of cattle alone.

3.2. Changes in farming systems

We only present results for the whole study area. Results of the changes in each zone are provided in Supplementary Materials (Annexes 2 and 3) and referred to when main differences were observed between the two zones.

According to survey data, the mean total farm area decreased by 18.1% and the mean cultivated area by 35.7% between 2007 and 2020 (Table 1). Over the same period, the proportion of farms growing cotton fell by 31.1%, and the mean cotton area per farm decreased by 59.8% (Table 1, Fig. 3A). The decrease in the mean cotton area was particularly large for the 'East study area' (65.2%, Annexes 2 and 3). During the same period, the mean cereal area per farm decreased significantly (Table 1), but modestly (−14.2%, Fig. 3B). The mean area of other crops per farm (groundnut and sesame) increased by 27.3% – with sesame cultivated in 2020 but not in 2007 – and the mean non-cropped area per farm (fallow and uncleared land) increased by 12.8% (Table 1). The increase in the mean non-cropped area per farm was particularly strong in the 'East study area' (35.7%, Annex 2). The proportion of households planning to clear land increased by 47.2%. This increase was particularly strong in the 'East study area' (95.2%, Annex 2). Only 9 farms out of the 141 interviewed had shifted their location since 2007.

Livestock numbers increased significantly from 2007 to 2020: by 361% for cattle and 405% for small ruminants (sheep and goats) according to survey data (Table 1, Fig. 3C and D). The increase in the mean number of cattle per farm was particularly high in the 'West study area' (547%, Annexes 2 and 3) and the increase in the mean number of small ruminants per farm in the 'East study area' (892%, Annexes 2 and 3). However, the proportions of farms owning at least one pair of draft animals, and owning at least two pairs of draught animals, did not change significantly between 2007 and 2020. A significant increase in cattle number was also detected for the period 2003–2014 by aerial census (Fig. 4A), while the population of small ruminants did not change significantly (Fig. 4B).

Based on the key informant interviews, the total area on which cotton was farmed was estimated to have fallen by 50–60% between 2007 and 2020 in the 'East study area' and by '80-90% in the 'West study area'. During the same period, the number of cattle tripled in the 'East study area' and increased more than six-fold in the 'West study area'. The number of small ruminants increased more than four-fold in the

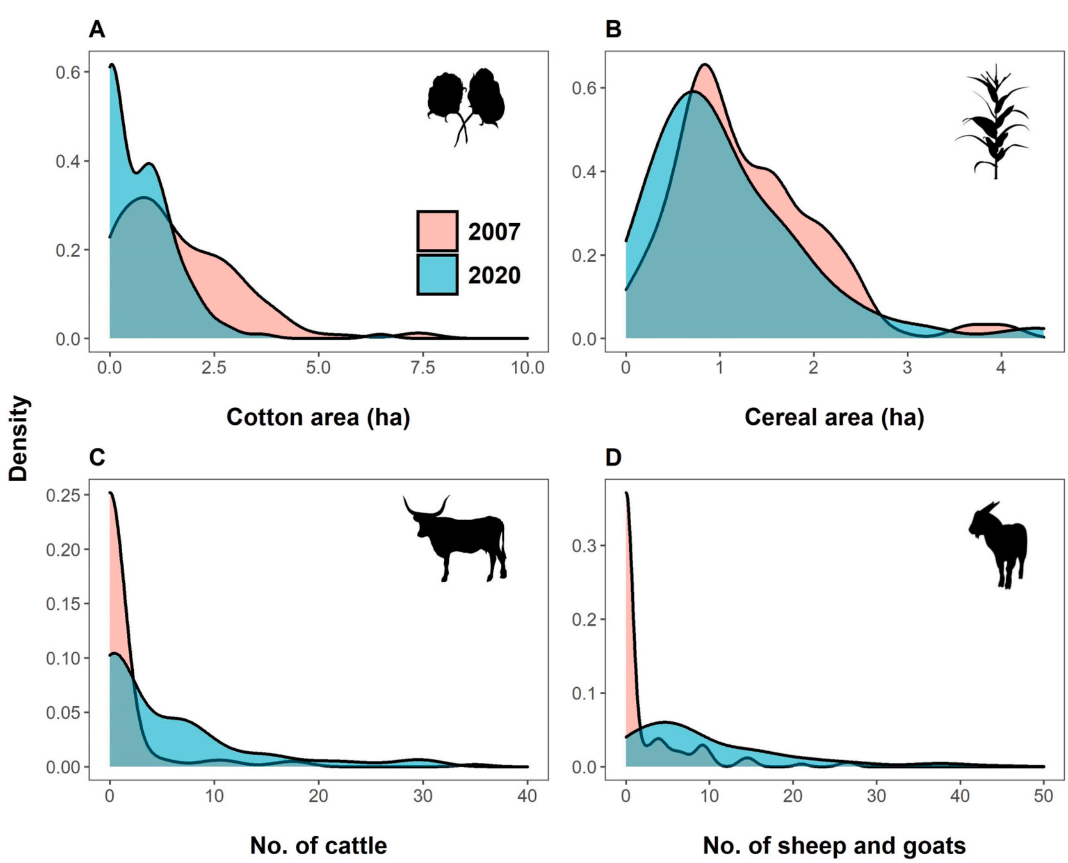

Fig. 3. Density plots of (A) the cotton area per farm, (B) the cereal (maize and sorghum) area per farm, (C) the number of cattle per farm, and (D) the number of small ruminants (sheep and goats) per farm in 2007 and 2020 ($n = 141$, same farms during both years).

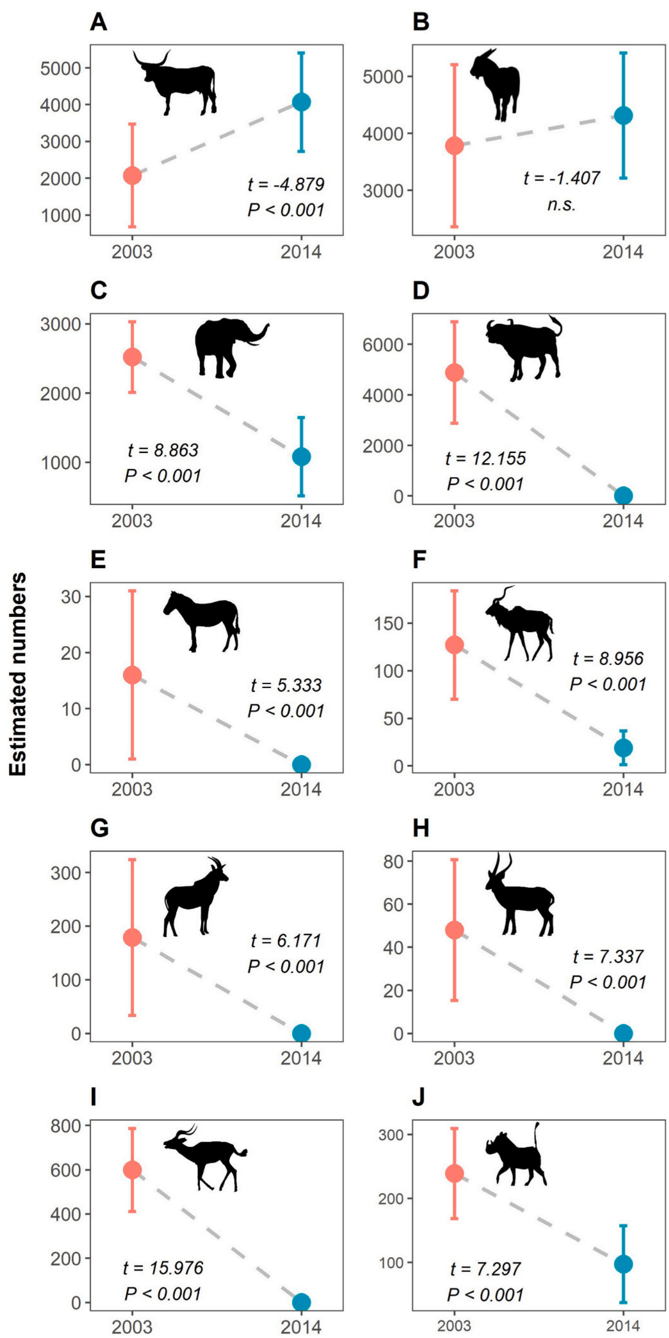

Fig. 4. Changes between 2003 and 2014 in the estimated numbers of (A) cattle, (B) small ruminants (sheep and goats), (C) elephant (*Loxodonta africana*), and (D) buffalo (*Syncerus caffer caffer*), (E) zebra (*Equus quagga*), (F) greater kudu (*Tragelaphus strepsiceros*), (G) sable antelope (*Hippotragus niger*), (H) waterbuck (*Kobus ellipsiprymnus*), (I) impala (*Aepyceros melampus*), and (J) warthog (*Phacochoerus africanus*) in the study area (data source: Dunham, 2004, Dunham et al. 2014; vertical lines represent standard deviations; statistics of Student's test comparing estimated numbers in 2003 and 2014 are also provided for each species).

'East study area' and more than nine-fold in the 'West study area'. The decrease in area under cotton was said to have been caused by a decrease in cotton profitability: whereas the cash need of a family of five could be covered by 1–2 ha of cotton in 2007, more than four ha of cotton were needed in 2020. The increase in livestock was said to have primarily been the result of the decreasing profitability of cotton, as well as new investments in the area (particularly the establishment of new dip tanks, and development programs supporting livestock farming in the district). A shift from maize to sorghum, groundnut and cowpea was also reported between 2007 and 2020, caused primarily by climate change according to key informants, but also driven by new marketing opportunities (for example, the national Grain Marketing Board began to purchase sorghum in 2020). Maize was said to now be confined to riverbank cultivation.

The mean age of the head of the household increased from 46.1 ± 15.7 in 2007 to 51.6 ± 15.8 in 2020, but the size of the household did not change significantly (Table 1). None of the changes above were correlated with the age of the head of the household in 2020, except household size and food security, both having decreased significantly with increasing age of the head of the household between 2007 and 2020 (Tables 2, 3). This implies that the changes in farm structure and farm performance observed in the panel survey data are not due to a shift along the 'farm development cycle' during the 13 years separating the two observations, but to other factors.

According to survey data, cotton was the main source of income for the majority of farmers in 2007 (74.5%) (Fig. 5A). In 2020, no income source was so prominent and livestock had become the main source of income for the majority of farms (30.5%), followed by cotton (for 28.4% of farms) and casual labour (for 22.0% of farms). From being the main source of food for 67.4% of farms in 2007, the importance of the households' own production declined to hold true for 51.8% of farms in 2020 (Fig. 5B). Food aid was the second most important source of food at the time of both surveys (main source of food for 17.7% of farms in 2007 and 19.2% of farms in 2020), and food purchase using income was the third most important source of food (main source of food for 13.5% of farms in 2007 and 18.4% of farms in 2020).

3.3. Changes in population and in land cover

According to census data, the population of the study area increased from 10,469 inhabitants in 1992 to 23,248 inhabitants in 2002, and 26,448 in 2012 (Fig. 1B). Thus, population increase declined from a rate of 8.30% per year between 1992 and 2002, to a rate of 1.30% per year between 2002 and 2012. The rate for the 2002–2012 period even became negative (−0.07% per year) in the 'West study area' (while remaining positive – at 1.64% per year – in the 'East study area').

From 2006 to 2014, the open vegetation land cover class in the study area increased from 14,115 ha in 2006 (10% of the total land cover area) to 25,641 ha in 2014 (18% of the total land cover area) and 28,595 ha in 2020 (20% of the total land cover area) (Fig. 6, Table 4). Thus, from a value of 7.75% per year during the period 2006–2014, the rate of open vegetation expansion slowed to a rate of 1.83% per year during the period 2014–2020. Open vegetation expansion between 2006 and 2020 was mainly at the expense of scrub area, which decreased from 91,824 ha in 2006 to 85,756 ha in 2014 and 74,869 ha in 2020. By comparison, the woodland area remained relatively stable: 26,610 ha in 2006,

Table 2
Summary of the correlations between age of the head of the household in 2020 and changes in quantitative variables (% change between 2007 and 2020), tested using Kendall's tau coefficient. Significant effects (*P*-value < 0.05) are shown in bold.

Indicator	tau	z-value	*P*-value
Household size (no.)	**−0.139**	**−2.383**	**0.017**
Farm area (ha)	−0.062	−1.068	0.285
Cultivated area (ha)	−0.071	−1.236	0.216
Cotton area (ha)	−0.115	−1.789	0.074
Cereal area (ha)	−0.101	−1.717	0.086
Other crops area (ha)	−0.018	−0.238	0.812
Non-cropped area (ha)	0.023	0.390	0.697
Cattle number	−0.065	−0.746	0.455
Small ruminant number	−0.061	−0.894	0.372
Food security (months)	**−0.164**	**−2.779**	**0.005**

Table 3
Summary of the results of generalised linear models with Poisson distribution (3 levels) testing the influence of age of the head of the household in 2020 on changes in proportion data.

Indicator	z-value	*P*-value
% owning ≥1 pair draught animals		
Intercept	4.160	< 0.001
Age in 2020	−0.688	0.491
% owning ≥2 pairs draught animals		
Intercept	3.314	0.001
Age in 2020	−0.009	0.993
% growing cotton		
Intercept	3.085	0.002
Age in 2020	−0.579	0.563
% planning to clear land		
Intercept	3.872	< 0.001
Age in 2020	−0.218	0.827
% guarding fields		
Intercept	4.077	< 0.001
Age in 2020	−0.298	0.765
% crops destroyed by elephant		
Intercept	3.644	< 0.001
Age in 2020	0.350	0.726
% crops destroyed by buffalo		
Intercept	3.678	< 0.001
Age in 2020	−0.749	0.454

24,062 ha in 2014 and 26,755 ha in 2020. Although the net woodland area remained largely unchanged, a shift from East to West is noticeable, with woodland area continuously declining in the 'East study area' (from 8647 ha in 2006 to 6854 ha in 2014, and 5585 ha in 2020), while 'the West study area' regained woodland (from 17,947 and 17,192 in 2006 and 2014 to 21,149 ha in 2020).

3.4. Trends in wildlife population sizes and human-wildlife interactions

From the aerial census data, the numbers of all wild ungulates appear to have decreased significantly between 2003 and 2014: elephant, buffalo, zebra, greater kudu, sable antelope, waterbuck, impala and warthog (Fig. 4). In fact, no buffalo, zebra, sable antelope, waterbuck or impala were detected in the study area during the census of 2014. Most farmers (73.7%) perceived elephant populations to have increased or remained stable over the last 15 years (this perception was particularly strong in the 'West study area'; Annex 4), and buffalo populations were perceived by most (80.1%) to have decreased over the same period (more so in the 'East study area' than the 'West study area'; Annex 4) (Fig. 7). The majority of farmers (> 50%) also perceived the populations of lion and leopard – two other species, in addition to elephant and buffalo, of importance for revenue generation through CAMPFIRE – to have declined over the last 15 years (particularly in the 'East study area'; Annex 4).

CAMPFIRE revenues have continuously declined during the period 2011–2018, with total revenues in 2018 less than half that in 2011 (0.6 vs. 1.3 million USD; Fig. 8). By contrast, participants of the key informant interviews all mentioned that the costs associated with living with wildlife – crop destruction and livestock predation in particular – had increased markedly in the past 15 years. According to survey data, the proportion of farmers guarding their fields increased by 17.6% between 2007 and 2020 (Table 1). The proportion of farmers experiencing significant crop destruction by elephants increased by 45.5%, whereas those experiencing significant crop damage by buffalo fell by 67.9%. The increase in the proportion of farms experiencing crop destruction by elephant was particularly strong in the 'West study area' (72.4%, Annex 2). According to official data collected by the district, on average 4.1 people have been killed and 10.7 people injured by wildlife each year in

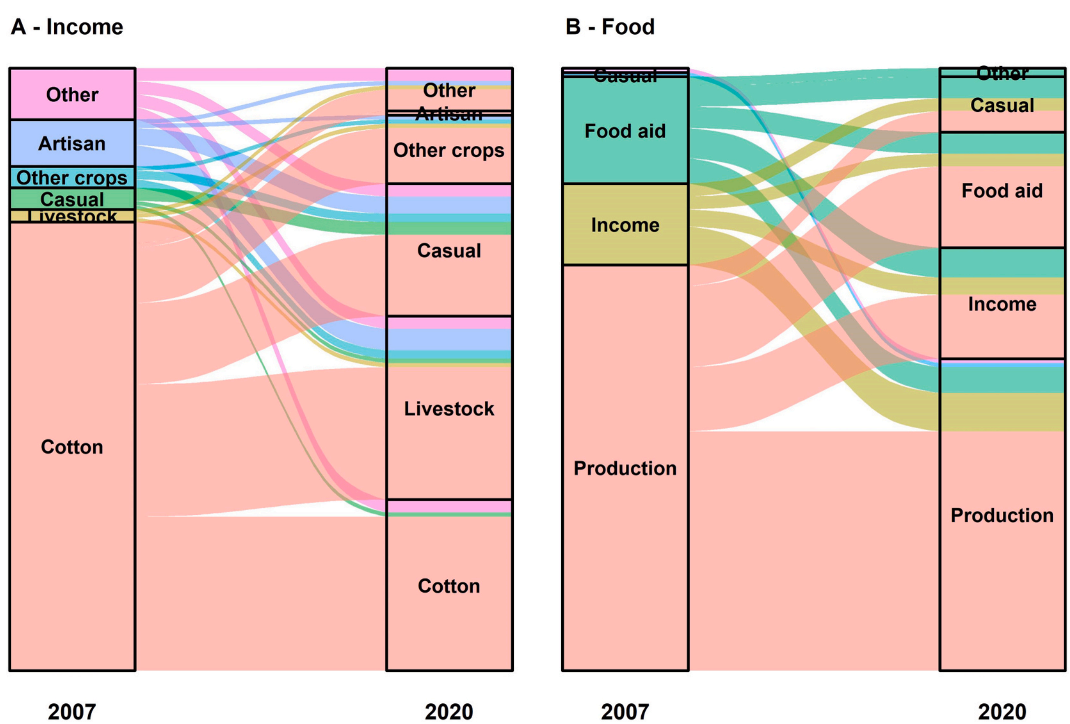

Fig. 5. Alluvial diagrams illustrating the changes in (A) the main source of income and (B) the main source of food for farm surveyed in 2007 and 2020. The height of a block represents the size of the cluster of farms, and the height of a stream field represents the size of the population of farms contained in both clusters connected by the stream field.

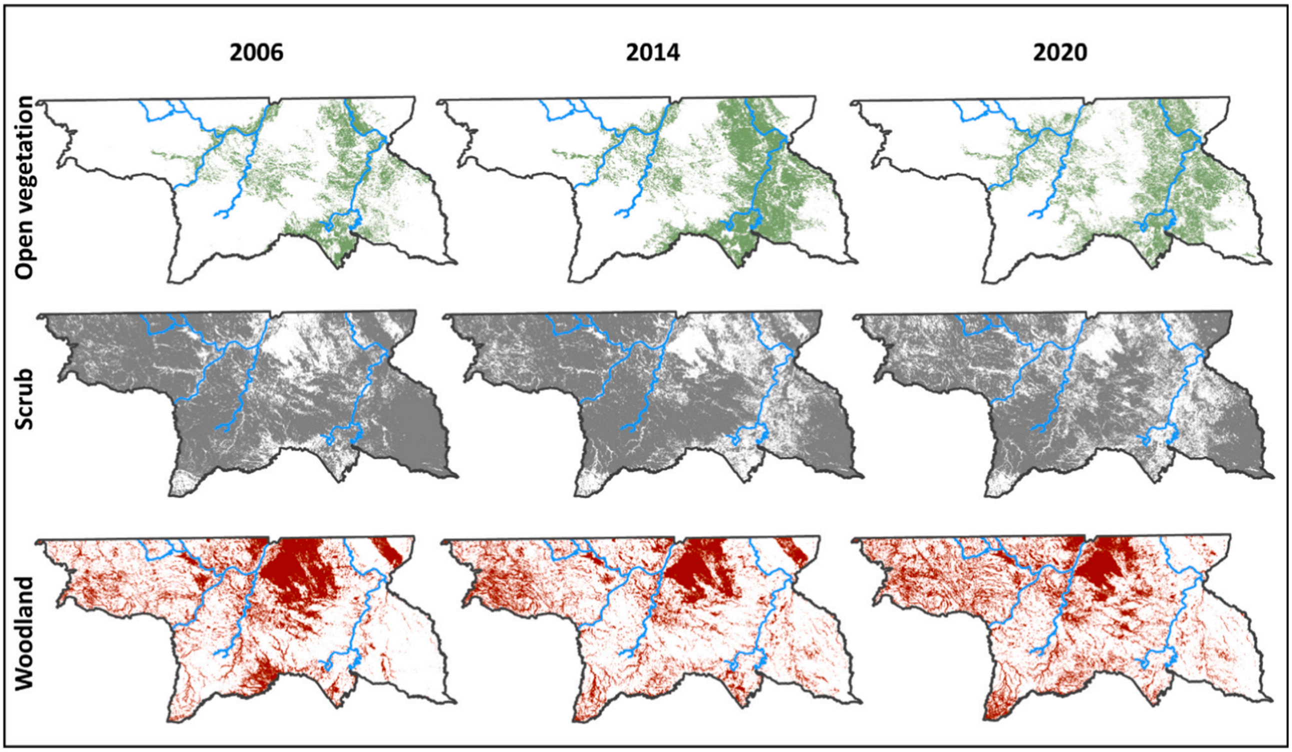

Fig. 6. Open vegetation, scrub, and woodland cover in the study area in 2006, 2014 and 2020.

Table 4
Main land cover areas in 2006, 2014 and 2020.

	2006	2014	2020
Study area			
Open vegetation (ha)	14,115	25,641	28,595
Scrub (ha)	91,824	85,756	74,869
Woodland (ha)	26,610	24,062	26,755
East study area			
Open vegetation (ha)	10,314	20,273	20,293
Scrub (ha)	38,972	31,326	30,229
Woodland (ha)	8647	6854	5585
West study area			
Open vegetation (ha)	3793	5328	8264
Scrub (ha)	52,792	54,377	44,590
Woodland (ha)	17,947	17,192	21,149

the period 2011–2020 (Fig. 9A). Similarly, on average each year during that period, 68.4 cattle and 165.6 small ruminants were killed by wildlife (Fig. 9B), and 30.1 granaries were destroyed (Fig. 9D). Between 2016 and 2018, 639.3 ha of crop per year was destroyed by wildlife (Fig. 9C). Species most often involved in human injuries and deaths were crocodile, elephant, snakes, hippopotamus, and buffalo (Annex 5). Species inflicting significant livestock losses and most often cited by farmers were hyena (89.4%), baboon (41.8%) and civet, genet and other small carnivores (41.1%). However, more farmers experienced significant crop damage from arthropods (millipede, fall armyworm and stemborer in particular) than from megafauna (elephant and bush pig were the species most often cited) (Annex 6).

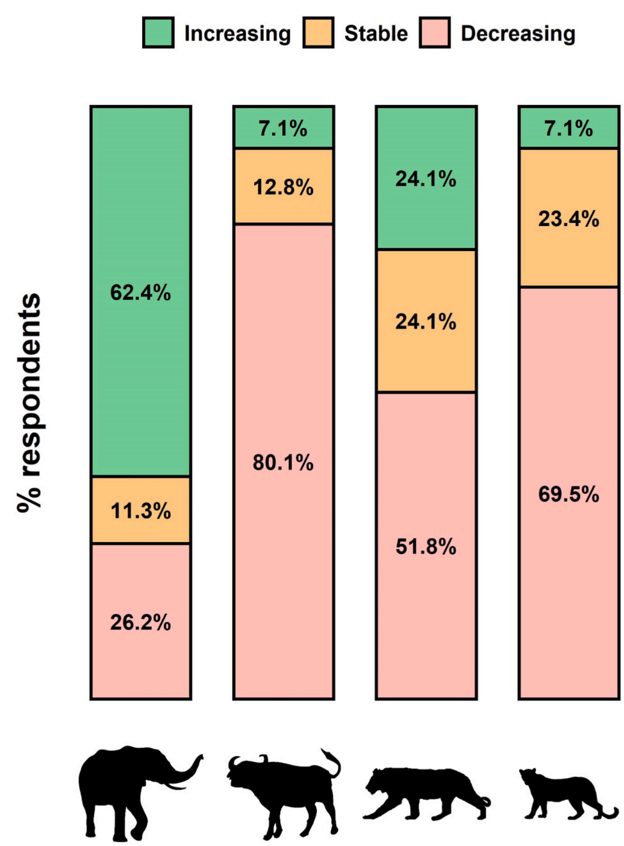

Fig. 7. Proportion of the 141 farmers interviewed perceiving the populations of elephant, buffalo, lion and leopard in the area as increasing, stable or decreasing over the past 15 years.

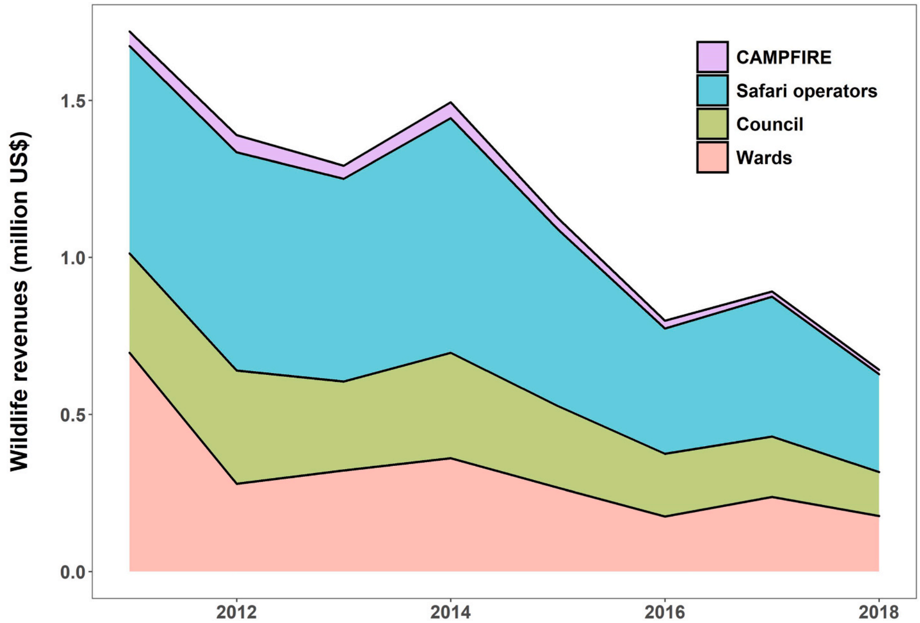

Fig. 8. Wildlife revenues in Mbire District from CAMPFIRE (Communal Area Management Program for Indigenous Resources) for the period 2011 to 2018, broken down by recipients: wards, District Council, safari operators and CAMPFIRE Association (data source: Mbire Rural District Council).

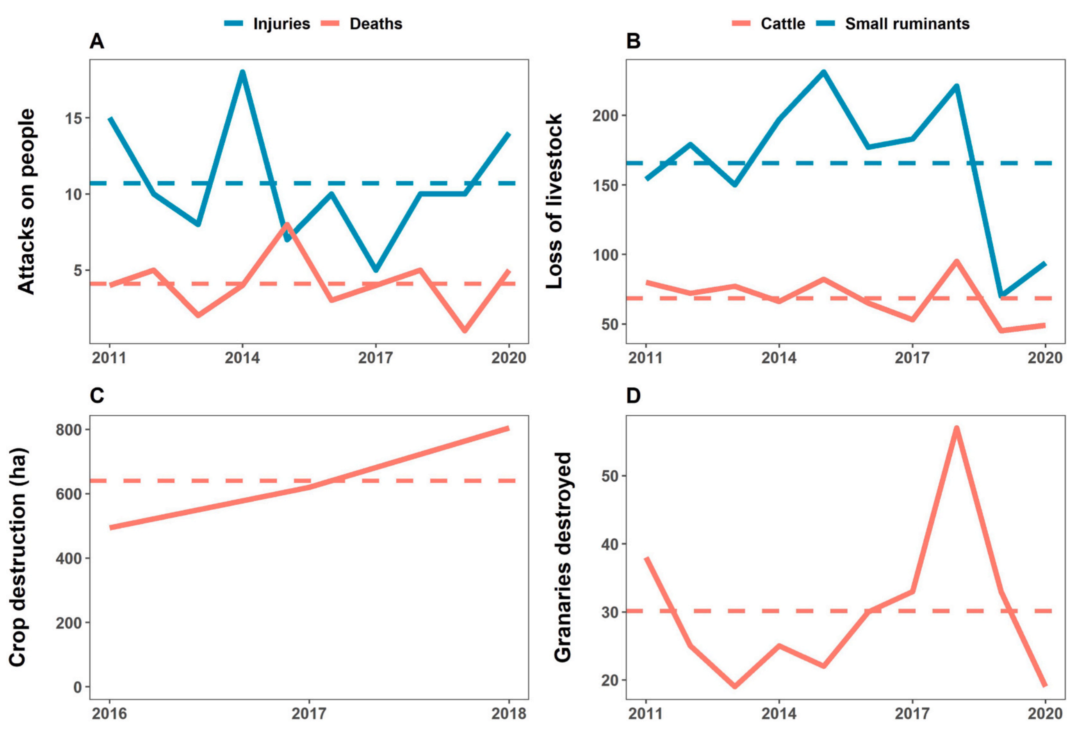

Fig. 9. (A) number of human deaths and injuries caused by wildlife from 2011 to 2020, (B) number of cattle and small ruminants (sheep and goat) lost to predation by wildlife from 2011 to 2020, (C) crop area destroyed by wildlife from 2016 to 2018, and (D) number of granaries destroyed by wildlife from 2011 to 2020. Dashed lines represent mean values for the reported period (data source: Mbire District Council).

4. Discussion

4.1. A shift from cotton to livestock farming fuelled land cover changes

The rates of expansion of open vegetation during the period 2006–2020 in the 'East study area' (4.95%) and the'West study area' (5.72%) were much faster than the rates of population increase during the period 2002–2012 for the same areas (1.64% in the 'East study area' and − 0.07% in the'West study area'), demonstrating that the observed changes in land cover were driven by changes in land-use rather than population (Figs. 1B, 6, Table 4). Land cover changes in the study area were clearly driven by cotton farming during the period 1980–2007 (Baudron et al., 2011). Since 2007 the proportion of farms growing cotton fell by almost a third, while the mean cotton area per farm decreased by close to 60% (Table 1, Fig. 3A). The strongest change in farming system between 2007 and 2020 was the increase in livestock, with the number of cattle and small ruminants per farm increasing roughly five-fold during the period (Table 1, Fig. 3C and D) while the mean cultivated area per household decreased by 35.7%. The increase in livestock numbers estimated by the panel survey was far larger than that estimated by aerial census (Fig. 4A and B), but consistent with the estimates obtained through the key informant interviews. Key informants also stated that the increase in livestock numbers had been primarily driven by the reduction in profitability of cotton: farmers switched their main source of income from cotton to livestock. Other factors (new dip tanks and development programs supporting livestock farming) also played a role. It is also likely that the decline in livestock numbers following droughts (particularly the 2012–13 drought) and outbreaks of tick-borne diseases in the provinces of Zimbabwe that are home to most of the national herd (Masvingo, Midlands, Matebeleland North, Matebeleland South; MLAWRR, 2020) during the period under investigation created demand for livestock from the study area (Bennett et al., 2018).

Both methods for estimation of livestock numbers have limitations: the sample of households included in the panel survey is relatively small, and may not be statistically representative of the total farm population (for instance, farming households that were established between 2007 and 2020 were not surveyed), whilst aerial censuses underestimate population sizes, particularly for small species (e.g., domestic small ruminants), and particularly when multiple species are recorded (Jachmann, 2002). Although the true magnitude of the increase in numbers can be questioned, the number of livestock appears to have increased several-fold in the study area. Key informants indicated that the resulting increased need for grazing land triggered expansion of open vegetation through two mechanisms: first, reduction in tree cover in communal grazing areas and second, expansion of grassy fallows. Herders were reported to cut trees – particularly from highly palatable species such as *Faidherbia albida* or *Kigelia africana* – to feed livestock with their foliage during the dry season. Browsing of seedlings and saplings by livestock also prevents recruitment of these species (Sida et al., 2018). Farmers have also increased their non-cropped areas (fallow and uncleared land, which are both part of the 'open vegetation' land cover class) between 2007 and 2020: by 12.8% in the study area (Table 1), and 35.7% in the 'East study area' according to the survey results (Annex 2). This trend was confirmed by the key informant interviews and the authors' personal observations (the first author has worked in the area from 2002 to 2010 and has visited regularly since, whilst the third and fourth authors have resided in the area throughout the research period). It is also likely that the extent of these areas, which are mainly grassy fallows, was under-estimated during the survey as some fallows (in particular older ones) are no longer considered part of individually managed farms but instead of communal grazing areas. Despite this likely under-estimation, non-cropped areas represented the majority of the farm area reported in 2020 (> 50%, against about 1/3 in 2007; Table 1). The need for grazing appears to have stimulated the clearing of new fields to turn older fields into grassy fallows (authors' personal observations and key informant interviews). This is consistent with the intention of the majority of farmers (> 50%) in 2020 to clear new land the following season (Table 1).

The shift from production of cotton and other crops to livestock production is also demonstrated by the fact that, although cattle numbers increased, there was no change in the proportion of households with one or more spans of draught animals between 2007 and 2020 (Table 1), illustrating that this increase is not due to changes in the availability of traction. Livestock rather than crop production had become the main source of income in 2020 (Fig. 5A). Livestock ownership appears to have become more equitable in 2020 than in 2007, as demonstrated by the 'flatter' density curves in 2020 compared with 2007 (Fig. 3C and D). This is in part due to the fact that cattle are now present west of the river Angwa in the 'West study area' (largest % change from the two zones, Annex 2), where the presence of tsetse fly largely precluded cattle keeping in 2007 (Baudron et al., 2011). Key informants reported a decrease in tsetse fly density in this area, probably driven by loss of natural vegetation which is a suitable habitat for the insect (Chikowore et al., 2017).

Our data does not allow us to test whether the observed loss of natural vegetation and the observed decline in wildlife numbers (for all species; Fig. 4) in the study area is a cause-and-effect relationship. Indeed, wildlife numbers appeared to decline between 2003 and 2014 not only in the study area (both in the 'East study area' and the 'West study area') but also in the whole of Mbire District and in the neighbouring protected areas (Annex 7). As counting methods did not change significantly between periods, this implies that other factors were also driving the decline in wildlife across a larger geographic area in addition to the changes in land cover in the study area. Recurring droughts over the past decade could be one such driver (Kupika et al., 2017). Poaching is likely to have also played a major role, both in the communal land and in protected areas, as suggested by increases in elephant carcass ratios between 2003 and 2014 (Annex 8), personal communications (e.g., with the African Wildlife Foundation local staff), and other published work (e.g., Muboko et al., 2021).

The decline in wildlife numbers, and in particular 'trophy species' (e.g., elephant, buffalo) is in turn responsible – at least partially – for the decline in wildlife revenues (Fig. 8). By contrast the (non-monetary) wildlife costs remain high (Fig. 9), and continue to grow as evidenced by the key informants. Such a situation with high costs with few benefits from wildlife is likely to fuel (or at least not to deter) conversion of natural vegetation into cropland and grazing land. Human-wildlife conflicts remain strong despite a decrease in wildlife densities because of the expansion of human activities (mainly livestock farming) into the natural vegetation (Fig. 6). In addition, key informants highlighted that human-wildlife conflicts are most acute between March and July when crops mature and wildlife numbers increase, whereas the aerial censuses of 2003 and 2014 took place in September and August, respectively. Aerial census data is not able to capture such seasonal movements. The shift from maize to sorghum – which is much more palatable to wildlife – was also mentioned as a cause for increased crop destruction. Finally, the decline in wildlife revenues is also partly due to a change in hunting strategy by safari operators, as mentioned during key informant interviews, with operators relying more on safari areas than in the past due to the decline in numbers of trophy animals in the communal lands during the hunting season.

The patterns of farming livelihoods and environmental changes

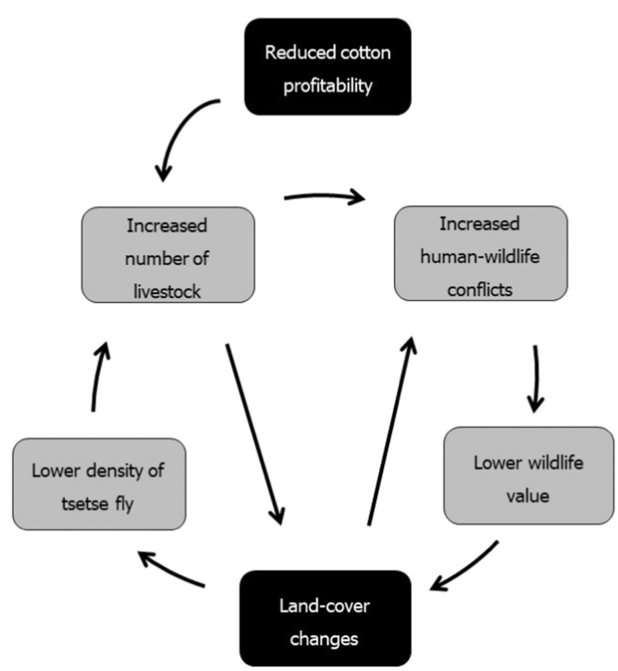

Fig. 10. Conceptual diagram of the mechanisms described in the study linking reduced cotton profitability to land cover changes.

unravelled by this study that link the reduced profitability of cotton to changes in land cover are summarized in Fig. 10. Although the human population density in the region only changed marginally, the numbers of cattle and small ruminants (goats and sheep) have increased several-fold due to a switch from crop production to livestock farming as a major source of income. The increase in livestock numbers in turn created two negative feedback loops. In one, the loss of natural vegetation may have led to a decrease in tsetse fly allowing further expansion of livestock. In the other, increased human-wildlife conflict led to lower value attached to wildlife which in turn may have led to further land cover change, converting natural vegetation into grazing land. Increasing human-wildlife conflicts are likely to threaten the long-term coexistence of people and wildlife in the study area.

4.2. Commodity crops in biodiversity-rich production landscapes: friends or foes?

The Jevons' paradox – which states that improving agricultural productivity would stimulate expansion of that production – is often invoked in nature conservation as a risk associated with investing in interventions to increase the productivity and profitability of crops in biodiversity-rich areas, and especially commodity crops, which tends to have a relatively elastic demand (Angelsen, 1999; Perfecto and Vandermeer, 2010; Rudel et al., 2009). On the other hand, production of commodity crops tends to be labour intensive, and any loss in profitability may drive farmers to shift to less labour-demanding enterprises. In this study, the labour demand of sorghum was found to be about half that of cotton (a typical farming household with five members was said to be able to manage four hectares of sorghum with its own labour, but only two of cotton). The labour demand of livestock production was even less (one person was said to be able to herd up to 50 heads of cattle alone, which is much larger than the herd required to meet the cash needs of a typical farming household in the area). Such shifts can result in land conversion for agriculture as biodiversity-rich areas/agricultural frontiers tend to be sparsely populated, and farming in such areas is thus limited more by labour scarcity than by land in relative terms (Angelsen and Kaimowitz, 2001; Baudron et al., 2012a; Perfecto and Vandermeer, 2010). Although cotton was a major driver of land cover change in the Mid-Zambezi valley in the past (Baudron et al., 2011), Baudron (2011) predicted that a loss of profitability in cotton would lead to a shift to less labour-intensive enterprises, fuelling land cover changes. He recommended that interventions should be explored to maintain the profitability of cotton relative to other enterprises to 'spare land' for nature (and for wildlife in particular).

Although commodity crops may have beneficial impacts on the environment through land-saving, they may also have negative spillover impacts on ecosystems upstream and downstream (and downwind) due to their water and pesticide use (Chapagain et al., 2006; Kooistra and Termorshuizen, 2006). It is thus essential that farmers have access to the appropriate technologies, and correct use of technologies must be enforced by strong regulations, for inputs to be used with maximum efficiency and minimum spillover effects (Baudron et al., 2021). Sustainability standards (e.g. WWF, 2012) can be an important part of the regulatory framework required. Commodity crops also tend to receive more nutrients than food crops, leading to a positive impact on the nutrient balance of the whole farm and reducing soil nutrient mining which is a major cause of land degradation in sub-Saharan Africa (Vitousek et al., 2009). In the study area, much more fertilizer was applied on cotton than on other crops, which received virtually no fertilizer (Annex 1A). The decline of cotton production – compounded by the fact that little manure is applied to the fields, and often only on fields closest to the homestead – is thus likely to have resulted in increasingly negative farm nutrient balances. It could have also reduced the productivity of food crops, which benefit from fertilizer applied to cotton in the rotation (cf. Falconnier et al., 2016). Indeed, food security was found to be correlated to the area planted with cotton in the study area (Baudron et al., 2011), while complementarity between commodity crops and food crops has also been found elsewhere (e.g., Govereh and Jayne, 2003). Although many other factors no doubt came into play (including the general economic crisis in Zimbabwe), the decline of cotton may have partially been responsible for the decline in food security observed between 2007 and 2020 (from a mean of 7.85 to a mean of 5.61 months per year of food adequacy; Table 1).

The observed changes in farming appear to have resulted in some positive outcomes for livelihoods (e.g., more equitable distribution of livestock among farming households, Fig. 3; more flexible and diverse cash flow as mentioned during key informant interviews), but the current situation of food insecurity coupled with rapid land cover changes is a lose-lose situation for people and nature, which cautions against lauding smallholder-based agroecological systems (as proposed by e.g., Altieri and Toledo, 2011; Fischer et al., 2017) as a panacea for biodiversity conservation and food security. Conservation interventions should support local production systems that provide a decent living for local households (beyond food security, see Section 4.3 below) while having a low demand for land, water and other resources. Such production systems may include commodity crops, while sustainability standards could be used to ensure negative environmental impacts are minimized (WWF, 2012). In addition, labour-intensive commodity crops will only benefit local livelihoods while slowing agricultural expansion if immigration from other regions is controlled (Angelsen and Kaimowitz, 2001; Scholte, 2003). Intensification alone is also unlikely to lead to reduced agricultural expansion without enforced land-use planning that includes set-aside programmes (Pierce et al., 2005; Rudel et al., 2009). In the case of the study area, considering the recent depopulation and concomitant reforestation in the 'West study area' (Figs. 1B, 6, Table 4), a large portion of that ward could be set aside for conservation and management of wildlife through CAMPFIRE, while agricultural development could focus on the 'East study area'. In this part of the study area, which is most remote from wildlife conservation areas, and in the absence of a profitable and labour-intensive cash crop (cotton or other crop), interventions to stimulate livestock intensification (especially feed management; Annex 1B) would be needed to avoid degradation through over-grazing/over-browsing, given current extensive livestock management. Rangelands could also be managed as multi-species production systems, with livestock coexisting with wildlife – and managed sustainably through e.g., CAMPFIRE (Du Toit and Cumming, 1999) – though disease transmission is no doubt a major barrier to such systems (Caron et al., 2013).

4.3. Rethinking smallholder farming 'on the edge of protected areas'

The land sparing/sharing framework (Phalan, 2018) has become the dominant approach to assess agriculture-biodiversity trade-offs, since the seminal paper of Green et al. (2005). Yet to date, most of the studies using the framework assess agricultural performance based on land productivity/crop yields (Baudron et al., 2021). Labour productivity may be a more relevant metric in agricultural frontiers, as in the study area. The demise of the cotton sector is part of a general economic crisis in Zimbabwe, such that opportunities for alternative off-farm employment that could draw people away from farming in areas of conservation value is extremely limited (see e.g., Giller et al., 2013). Thus, local alternatives are pursued which are extensive in terms of land but give better returns to labour, such as livestock farming.

In addition, people 'living on the edge' of protected areas should not be assumed to be purely subsistence-oriented, as is too often the case. In 2020, production was not the main source of food for about half of the farms surveyed (Fig. 5B). Although the mean food security situation of farms in the study area had deteriorated between 2007 and 2020 (due to a number of factors, not all local; Table 1), most farm households invested in income-generating activities (mainly livestock production), not food production, to guarantee their food security. Globally, few farms are truly subsistence-based and most are connected to markets to some degree (Frelat et al., 2016; Giller et al., 2021), including those found in areas perceived as 'remote'. Conservation interventions need to promote livelihood activities that generate a decent living, not activities merely aiming at food security, in addition to being benign for the environment. Adopting (and adapting) the 'living income' concept, which is increasingly used to evaluate the livelihoods of small-scale producers in commodity chains (van de Ven et al., 2020), could be helpful when comparing different land-use options. If practices that are relatively benign to the environment fail to provide a decent living for local residents, they are likely to adopt practices that are more extractive and damaging to the environment, but that lead to a higher income.

5. Conclusions

Though cotton was the main driver of land cover changes in the Mid-Zambezi Valley in the past (Baudron et al., 2011), a decrease in the profitability of cotton has led to a shift to less-labour intensive livestock farming, accelerating the loss of natural vegetation (Fig. 10), as local farming systems are limited more by labour than by land (Baudron et al., 2012a). This case is probably representative of many sparsely-populated, biodiversity-rich areas or agricultural frontiers, where commodity crops may represent opportunities, not only challenges, for nature conservation. Our results also illustrate the complexity of the interactions between agriculture and biodiversity, calling for further involvement of agricultural scientists in conservation (Baudron et al., 2021). Such challenges are prescient considering that the majority of vertebrates are projected to lose habitat to agriculture by 2050 (Williams et al., 2021). Land productivity expressed as crop yields is the main indicator of agricultural performance used in assessments of the potential for land sparing/land sharing, but is probably not be the most relevant for people residing in biodiversity-rich multifunctional landscapes. For example, our results highlight the importance of labour productivity and income. Other properties of farming that are valued by local residents may include resilience, and contributions to nutrition (Baudron et al., 2021).

Our study also illustrates the harsh reality of people coexisting with wildlife while barely meeting (or failing to meet) the basic human rights for a decent living. Even without considering health care and costs of education, households in the study area were food secured for <6 months per year on average in 2020 (Table 1). At the same time, they suffered major crop damage and harmful conflict with wildlife including loss of life. Biodiversity-rich landscapes tend to be remote rural areas where livelihood opportunities are limited, and 'nature-friendly' ones even more so. Once the profitability of cotton farming declined, residents of the study area took up other income-generating activities (extensive livestock farming) which happened to be more harmful to the environment (land cover change) and wildlife (increased human-wildlife conflicts). But this trend is reversible, and there is actually an indication that land cover changes are plateauing (the rate of farmland expansion decreased from 7.75% per year between 2006 and 2014, to 1.83% per year between 2014 and 2020). For Mbire to remain a landscape where people and wildlife coexist, changes are needed on two fronts. First, agricultural systems are needed that provide a decent living while having a low demand for land, water and other resources. Second wildlife must contribute to local livelihoods and generate benefits that far exceed the costs (which are substantial, Fig. 9). Sustainability standards can be a great tool to achieve the former with commodities traded globally (e.g., cotton), but this is challenging with livestock, which tends to be produced mainly for local and national markets. The most realistic way to achieve incomes from wildlife in the short-term appears to be through CAMPFIRE, although the program depends on trophy-hunting, which faces growing opposition globally (Batavia et al., 2019). Yet wildlife-based alternatives to trophy hunting (e.g., photo-tourism) are unrealistic in most landscapes shared between people and wildlife (Hart et al., 2020). Ecotourism was attempted earlier in the study area for example, but failed. Curbing the current biodiversity crisis requires more pragmatism than ideology, especially when driven from outside. Local communities must be empowered – through good governance and evidence-based recommendations – but ultimately, they will decide how to manage the landscapes and natural resources on which their livelihoods depend.

Author statement

All datasets used for this study are publicly available (https://doi.org/10.7910/DVN/VMVZRD).

CRediT authorship contribution statement

FB conceived the research, analysed the data, and wrote the first draft of the manuscript. LG conducted the spatial analysis. EC, EC and FB conducted the field surveys and collected secondary data. LG and KEG contributed to the writing of the final manuscript.

Declaration of competing interest

The authors declared no potential conflicts of interest with respect to the research, authorship, or publication of this article.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2022.109496.

References

Altieri, M.A., Toledo, V.M., 2011. The agroecological revolution in Latin America: rescuing nature, ensuring food sovereignty and empowering peasants. J. Peasant Stud. 38, 587–612. https://doi.org/10.1080/03066150.2011.582947.

Andersson, J.A., de Garine-Wichatitsky, M., Cumming, D.H.M., Dzingirai, V., Giller, K.E. (Eds.), 2013. Transfrontier conservation areas: People living on the edge. Routledge. London and New York.

Angelsen, A., 1999. Agricultural expansion and deforestation: modelling the impact of population, market forces and property rights. J. Dev. Econ. 58, 185–218. https://doi.org/10.1016/S0304-3878(98)00108-4.

Angelsen, A., Kaimowitz, D., 2001. Agricultural Technologies and Tropical Deforestation. CABI Publi, Wallingford, UK. https://doi.org/10.3362/9781780446097.001.

Batavia, C., Nelson, M.P., Darimont, C.T., Paquet, P.C., Ripple, W.J., Wallach, A.D., 2019. The elephant (head) in the room: a critical look at trophy hunting. Conserv. Lett. 12, 1–6. https://doi.org/10.1111/conl.12565.

Baudron, F., 2011. Agricultural Intensification - Saving Space for Wildlife ? Wageningen University.

Baudron, F., Corbeels, M., Monicat, F., Giller, K.E., 2009. Cotton expansion and biodiversity loss in african savannahs, opportunities and challenges for conservation agriculture: a review paper based on two case studies. Biodivers. Conserv. 18, 2625–2644. https://doi.org/10.1007/s10531-009-9663-x.

Baudron, F., Corbeels, M., Andersson, J.A., Sibanda, M., Giller, K.E., 2011. Delineating the drivers of waning wildlife habitat: the predominance of cotton farming on the fringe of protected areas in the mid-Zambezi Valley, Zimbabwe. Biol. Conserv. 144, 1481–1493. https://doi.org/10.1016/j.biocon.2011.01.017.

Baudron, F., Andersson, J.A., Corbeels, M., Giller, K.E., 2012a. Failing to yield? Ploughs, conservation agriculture and the problem of agricultural intensification: an example from the Zambezi Valley, Zimbabwe. J. Dev. Stud. 48, 393–412. https://doi.org/10.1080/00220388.2011.587509.

Baudron, F., Tittonell, P.A., Corbeels, M., Letourmy, P., Giller, K.E., 2012b. Comparative performance of conservation agriculture and current smallholder farming practices in semi-arid Zimbabwe. F. Crop. Res. 132, 117–128. https://doi.org/10.1016/j.fcr.2011.09.008.

Baudron, F., Govaerts, B., Verhulst, N., McDonald, A., Gérard, B., 2021. Sparing or sharing land? Views from agricultural scientists. Biol. Conserv. 259, 109167 https://doi.org/10.1016/j.biocon.2021.109167.

Bennett, B., Vigne, M., Figuie, M., Chakoma, C., Katic, P., 2018. Beef Value Chain Study in Zimbabwe - Draft Final Report. Greenwich, UK and Montpellier, France.

Caron, A., Miguel, E., Gomo, C., Makaya, P., Pfukenyi, D.M., Foggin, C., Hove, T., De Garine-Wichatitsky, M., 2013. Relationship between burden of infection in ungulate populations and wildlife/livestock interfaces. Epidemiol. Infect. 141, 1522–1535. https://doi.org/10.1017/S0950268813000204.

Ceballos, G., Ehrlich, P.R., Soberón, J., Salazar, I., Fay, J.P., 2005. Global mammal conservation: What must we manage? Science 309, 603–607. https://doi.org/10.1126/science.1114015 (80-.).

Chapagain, A.K., Hoekstra, A.Y., Savenije, H.H.G., Gautam, R., 2006. The water footprint of cotton consumption: an assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. Ecol. Econ. 60, 186–203. https://doi.org/10.1016/j.ecolecon.2005.11.027.

Chayanov, A.V., 1921. The Theory of Peasant Economy. Leipzig, Berlin.

Chikowore, G., Dicko, A.H., Chinwada, P., Zimba, M., Shereni, W., Roger, F., Bouyer, J., Guerrini, L., 2017. A pilot study to delimit tsetse target populations in Zimbabwe. PLoS Negl. Trop. Dis. 11, 1–17. https://doi.org/10.1371/journal.pntd.0005566.

Coid, C., Gaidet-Drapier, N., Moya, C., Poilecot, P., Poulet, D., Renaud, P.C., Ricard, X., Takawira, S., 2001. Les hommes et les animaux dans la moyenne vallée du Zambèze, Zimbabwe, Cirad. ed. Montpellier, France.

Defries, R.S., Rudel, T., Uriarte, M., Hansen, M., 2010. Deforestation driven by urban population growth and agricultural trade in the twenty-first century. Nat. Geosci. 3, 178–181. https://doi.org/10.1038/ngeo756.

Du Toit, J.T., Cumming, D.H.M., 1999. Functional significance of ungulate diversity in African savannas and the ecological implications of the spread of pastoralism. Biodivers. Conserv. 8, 1643–1661. https://doi.org/10.1023/A:1008959721342.

Dunham, K.M., 2004. Aerial Survey of Elephants and Other Large Herbivores in the Zambezi Heartland (Zimbabwe, Mozambique and Zambia): 2003. Kariba, Zimbabwe.

Dunham, K.M., Mackie, C.S., Nyaguse, G., 2015. Aerial Survey of Elephants and other Large Herbivores in the Zambezi Valley (Zimbabwe): 2014. Seattle, USA. Greenwich, UK and Montpellier, France.

Falconnier, G.N., Descheemaeker, K., Mourik, T.A.V., Giller, K.E., 2016. Unravelling the causes of variability in crop yields and treatment responses for better tailoring of options for sustainable intensification in southern Mali. F. Crop. Res. 187, 113–126. https://doi.org/10.1016/j.fcr.2015.12.015.

Fischer, J., Abson, D.J., Bergsten, A., Collier, N.F., Dorresteijn, I., Hanspach, J., Hylander, K., Schultner, J., Senbeta, F., 2017. Reframing the food – biodiversity challenge. Trends Ecol. Evol. 32, 335–345. https://doi.org/10.1016/j.tree.2017.02.009.

Frelat, R., Lopez-Ridaura, S., Giller, K.E., Herrero, M., Douxchamps, S., Djurfeldt, A.A., Erenstein, O., Henderson, B., Kassie, M., Paul, B.K., Rigolot, C., Ritzema, R.S., Rodriguez, D., Van Asten, P.J.A., Van Wijk, M.T., 2016. Drivers of household food availability in sub-saharan Africa based on big data from small farms. Proc. Natl. Acad. Sci. U. S. A. 113, 458–463. https://doi.org/10.1073/pnas.1518384112.

Fuchs, R., Brown, C., Rounsevell, M., 2020. Europe's green Deal offshores environmental damage to other nations. Nature 586, 671–673. https://doi.org/10.1038/d41586-020-02991-1.

Gaidet, N., Fritz, H., Nyahuma, C., 2003. A participatory counting method to monitor populations of large mammals in non-protected areas: a case study of bicycle counts in the Zambezi Valley, Zimbabwe. Biodivers. Conserv. 12, 1571–1585. https://doi.org/10.1023/A:1023646012700.

Geldmann, J., Barnes, M., Coad, L., Craigie, I.D., Hockings, M., Burgess, N.D., 2013. Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. Biol. Conserv. 161, 230–238.

Gibbs, H.K., Ruesch, A.S., Achard, F., Clayton, M.K., Holmgren, P., Ramankutty, N., Foley, J.A., 2010. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proc. Natl. Acad. Sci. U. S. A. 107, 16732–16737. https://doi.org/10.1073/pnas.0910275107.

Giller, K.E., Baudron, F., Matema, S., Milgroom, J., Murungweni, C., Guerbois, C., Twine, W., 2013. Population and livelihoods on the edge. In: Andersson, J.A., de Garine-Wichatitsky, M., Cumming, D.H.M., Dzingirai, V., Giller, K.E. (Eds.), Transfrontier Conservation Areas. People Living on the Edge. London, U.K, pp. 62–88.

Giller, K.E., Delaune, T., Silva, J.V., Descheemaeker, K., van de Ven, G., Schut, A.G.T., van Wijk, M., Hammond, J., Hochman, Z., Taulya, G., Chikowo, R., Narayanan, S., Kishore, A., Bresciani, F., Teixeira, H.M., Andersson, J.A., van Ittersum, M.K., 2021. The future of farming: who will produce our food? Food Secur. 13, 1073–1099. https://doi.org/10.1007/s12571-021-01184-6.

Girard, M.-C., Girard, C.M., 1999. Traitement des données de télédétection, Dunod. ed. Malakoff, France.

Govereh, J., Jayne, T.S., 2003. Cash cropping and food crop productivity: synergies or trade-offs? Agric. Econ. 28, 39–50. https://doi.org/10.1016/S0169-5150(02)00066-X.

Green, R.E., Cornell, S.J., Scharlemann, J.P.W., Balmford, A., 2005. Farming and the fate of wild nature. Science 307, 550–555. https://doi.org/10.1126/science.1106049.

Hart, A.G., Cooney, R., Dickman, A., Hare, D., Jonga, C., Johnson, P.K., Louis, M.P., Lubilo, R., Roe, D., Semcer, C., Somerville, K., 2020. Threats posed to conservation by media misinformation. Conserv. Biol. https://doi.org/10.1111/cobi.13605.

Hulme, D., Murphree, M.W., 1999. Communities, wildlife and the 'new conservation' in Africa. J. Int. Dev. 11, 277–285. https://doi.org/10.1002/(SICI)1099-1328(199903/04)11:2<277::AID-JID582>3.0.CO;2-T.

Jachmann, H., 2002. Comparison of aerial counts with ground counts for large African herbivores. J. Appl. Ecol. 39, 841–852. https://doi.org/10.1046/j.1365-2664.2002.00752.x.

Kooistra, K., Termorshuizen, A., 2006. The sustainability of cotton Consequences for man and environment. In: Science Shop Wageningen University & Research Centre. Report.

Kupika, O.L., Gandiwa, E., Kativu, S., Godwell, N., 2017. Impacts of climate change and climate variability on wildlife resources in Southern Africa: Experience from selected protected areas in Zimbabwe. In: Bülent, Ş., Grillo, O. (Eds.), Selected Studies in Biodiversity.

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. U. S. A. 108, 3465–3472. https://doi.org/10.1073/pnas.1100480108.

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., Geschke, A., 2012. International trade drives biodiversity threats in developing nations. Nature 486, 109–112. https://doi.org/10.1038/nature11145.

Lima, M., da Silva Junior, C.A., Rausch, L., Gibbs, H.K., Johann, J.A., 2019. Demystifying sustainable soy in Brazil. Land Use Policy 82, 349–352. https://doi.org/10.1016/j.landusepol.2018.12.016.

Meyfroidt, P., Rudel, T.K., Lambin, E.F., 2010. Forest transitions, trade, and the global displacement of land use. Proc. Natl. Acad. Sci. U. S. A. 107, 20917–20922. https://doi.org/10.1073/pnas.1014773107.

MLAWRR, 2020. Second Round Crop and Livestock Assessment Report. 2019/2020 Season. Harare, Zimbabwe.

Muboko, N., Dube, P., Mashapa, C., Ngosi, E., Gandiwa, E., 2021. Trophy quality trends and hunting effort of selected big game in Chewore South Safari Area, northern Zimbabwe, 2009–2012. Trop. Ecol. 62, 52–60. https://doi.org/10.1007/s42965-020-00123-4.

Munthali, S.M., 2007. Transfrontier conservation areas: integrating biodiversity and poverty alleviation in southern Africa. Nat. Resour. Forum 31, 51–60.

Perfecto, I., Vandermeer, J., 2010. The agroecological matrix as alternative to the land-sparing/agriculture intensification model. Proc. Natl. Acad. Sci. U. S. A. 107, 5786–5791. https://doi.org/10.1073/pnas.0905455107.

Phalan, B.T., 2018. What have we learned from the land sparing-sharing model? Sustainability 10, 1–24. https://doi.org/10.3390/su10061760.

Phalan, B., Bertzky, M., Butchart, S.H.M., Donald, P.F., Scharlemann, J.P.W., Stattersfield, A.J., Balmford, A., 2013. Crop expansion and conservation priorities in tropical countries. PLoS One 8. https://doi.org/10.1371/journal.pone.0051759.

Pierce, S.M., Cowling, R.M., Knight, A.T., Lombard, A.T., Rouget, M., Wolf, T., 2005. Systematic conservation planning products for land-use planning: interpretation for implementation. Biol. Conserv. 125, 441–458. https://doi.org/10.1016/j.biocon.2005.04.019.

Rodrigues, A.S.L., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., Hoffmann, M., Long, J.S., Marquet, P.A., Pilgrim, J.D., Pressey, R.L., Schipper, J., Sechrest, W., Stuart, S.N., Underhill, L.G., Waller, R.W., Watts, M.E.J., Yan, X., 2004. Effectiveness of the global protected area network in representing species diversity. Nature 428, 640–643. https://doi.org/10.1038/nature02422.

Rudel, T.K., Schneider, L., Uriarte, M., Turner, B.L., DeFries, R.S., Lawrence, D., Geoghegan, J., Hecht, S., Ickowitz, A., Lambin, E.F., Birkenholtz, T., Baptista, S., Grau, H.R., 2009. Agricultural intensification and changes in cultivated areas, 1970–2005. Proc. Natl. Acad. Sci. U. S. A. 106, 20675–20680. https://doi.org/10.1073/pnas.0812540106.

Scholte, P., 2003. Immigration: a potential time bomb under the integration of conservation and development. Ambio 32, 58–64. https://doi.org/10.1579/0044-7447-32.1.58.

Sida, T.S., Baudron, F., Deme, D.A., Tolera, M., Giller, K.E., 2018. Excessive pruning and limited regeneration: are Faidherbia albida parklands heading for extinction in the central Rift Valley of Ethiopia? Land Degrad. Dev. 29, 1623–1633. https://doi.org/10.1002/ldr.2959.

Taylor, R., 2009. Community based natural resource management in Zimbabwe: the experience of CAMPFIRE. Biodivers. Conserv. 18, 2563–2583. https://doi.org/10.1007/s10531-009-9612-8.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D., Swackhamer, D., 2001. Forecasting agriculturally driven global environmental change. Science 292, 281–284. https://doi.org/10.1126/science.1057544.

van de Ven, G.W.J., de Valença, A., Marinus, W., de Jager, I., Descheemaeker, K.K.E., Hekman, W., Mellisse, B.T., Baijukya, F., Omari, M., Giller, K.E., 2020. Living income benchmarking of rural households in low-income countries. Food Secur. https://doi.org/10.1007/s12571-020-01099-8.

Vijay, V., Pimm, S.L., Jenkins, C.N., Smith, S.J., 2016. The impacts of oil palm on recent deforestation and biodiversity loss. PLoS One 11, 1–19. https://doi.org/10.1038/35971.

Vitousek, P.M., Naylor, R.L., Crews, T.E., David, M.B., Drinkwater, L.E., Holland, E., Johnes, P.J., Katzenberger, J., Martinelli, L.A., Matson, P.A., Nziguheba, G., Ojima, D., Palm, C.A., Robertson, G.P., Sanchez, P.A., Townsend, A.R., Zhang, F.S., 2009. Nutrient imbalances in agricultural development. Science (80-.) 324, 1519–1520. https://doi.org/10.1126/science.1170261.

Williams, D.R., Clark, M., Buchanan, G.M., Ficetola, G.F., Rondinini, C., Tilman, D., 2021. Proactive conservation to prevent habitat losses to agricultural expansion. Nat. Sustain. https://doi.org/10.1038/s41893-020-00656-5.

WWF, 2012. The 2050 criteria. In: Guide to Responsible Investment in Agricultural, Forest and Seafood Commodities. Gland, Switzerland.

ZimStat, 2012. Census 2012. Harare, Zimbabwe.