

Research Article

Wildlife as insurance against rainfall fluctuations in a semi-arid savanna setting of southeastern Zimbabwe.

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Abstract

This paper presents modeling approaches for wildlife conservation in a semi-arid savanna setting where there are frequent occurrences of drought. The model was used to test the extent to which wildlife income offers opportunities to reduce fluctuations in income as a result of variations in annual rainfall. For the application of the model the wildlife and agro-pastoral systems of southeastern Zimbabwe were simulated. Results show that wildlife income has the potential to compensate for some of the losses in expected income from livestock during droughts. However, wildlife income becomes second best to irrigated agriculture in stabilizing income in areas that show highly fluctuating rainfall. Possible reasons for this include high costs of exploiting the wildlife resource, and the small fraction of wildlife revenues received by households and communities. In order to search for sustainable solutions in areas such as the southeastern low veld of Zimbabwe, it is also important to be aware that the current human population and livestock densities are far above current sustainable levels. Our results therefore suggest that current and future efforts to conserve biodiversity are doomed to fail if there are no efforts made to decongest areas surrounding parks of high densities of human and herbivore populations, and to let local households earn more revenues from wildlife.

Key words: Wildlife; agro-pastoral; local people; expected income; fluctuating rainfall.

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Introduction

Over the last few decades, establishment of protected areas has constituted the principal system supporting conservation strategies [1]. Protected areas are needed in order to safeguard biological diversity [2]. In developing countries, however, land for establishing parks has often directly displaced rural communities and curtailed their access to natural resources that they traditionally used to depend on [3, 4]. Therefore game parks coexist with people in tightly coupled, fractious and uneasy relationships [5], causing conflict since establishment of parks has alienated the wildlife from the people and frequently transformed wildlife from a valuable commodity into a threat and a nuisance to the local people [4,6, 7]. Furthermore, the benefits or profits of having a park next door for local people in most of the African countries are not equitably distributed over the countryside. It is known that when people are taxed (either physically or financially) and do not benefit, they see a burden. For these and other reasons, protected areas, especially in Africa, have often operated against the economic interests of local communities, and persistent poaching pressure has led to a growing recognition that this 'fences and fines' approach has in many cases failed to achieve its objective of preserving wildlife [6-8].

To address these conflicts between protected areas and local communities, government agencies and non-governmental organizations joined forces in the 1980s and 1990s to develop community-based wildlife programs aimed at providing benefits to affected communities [9]. Community-based wildlife programs (e.g., Communal Areas Management Programme for Indigenous Resources-CAMPFIRE, in Zimbabwe), together with ecotourism approaches, assume that once local communities derive benefits from natural resources in their local area, their livelihoods will be improved and this will motivate them to promote conservation [10, 11] because harvestable wildlife gives the habitats a value to the people and enables a living without clearing the land for agriculture. Particularly due to rapid human population growth, the underlying conflict between wildlife conservation and people (rural development) is over conversion of land from wildlife conservation to agriculture [4].

Wildlife benefits can be a more stable source of income than agriculture. In many of the arid and semi-arid environments, rural farmers' production activities are characterized by uncertainty due to unpredictable climatic conditions [12, 13]. Under such conditions wildlife utilization becomes a highly competitive form of land use [14] and could diversify and consequently reduce drought risk. Therefore, rainfall variability seems to provide one of the strongest justifications for adopting natural resource-based land uses like wildlife as an alternative, sustainable strategy for social, economic and ecological improvement. In the case of CAMPFIRE in Zimbabwe, placing wildlife in the realm of economics and land use rather than conservation provided an important opportunity to complement conventional and subsistence agricultural practice in the communal lands of the country [14]. However, as indicated by one of the reviewers, most benefits of CAMPFIRE end up at the village/community level (water supply, schools, clinics, roads etc.) and not on the individual level. So people forgo potential individual income (e.g. from hunting) but receive individually very little in exchange. CAMPFIRE is touted as textbook CBNRM, but the truth is that individuals in these communities only receive individual benefits of USD 1 – 3 per year, i.e. virtually nothing.

A complicating factor is that livestock and wildlife often share the same diseases such as sleeping sickness (Trypanosomiasis) or Nagana, rinderpest, foot and mouth, and bovine tuberculosis (TB), to name just a few. According to Heitkönig and Prins [15] wildlife are generally immune to indigenous diseases while livestock, by and large, are not. Although wildlife still act as a maintenance host for many of the diseases in livestock [16], the reverse is also possible. For example, in southern Africa fences have been established to reduce the likelihood of cattle contracting foot-and-mouth disease from wildlife [17, 18], but bovine Tuberculosis entered the buffalo population in Kruger National Park through contamination by cattle [19]. Therefore, in a bid to improve local people's welfare it is important to manage the contagious diseases by veterinary control or by keeping wildlife and livestock systems separate, meaning that the spatial dimension in the allocation of resources becomes important as well.

Like biodiversity in general, wildlife use can be treated as a resource allocation problem, where scarce resources such as land are allocated over different competing uses. Thus for any ecosystem there is an 'optimal level' of wildlife, which depends not only on the bio-geophysical characteristics of that system, but also on the preferences of people who depend on that system, on the technology available to them, and on the variability of the natural and economic environments in which they work [20]. The same applies to agro-pastoral systems, particularly when it comes to optimal livestock stocking densities in savanna rangelands. Therefore, for sustainable management of savanna rangelands it is important to define what we mean by sustainability.

Following the Brundtland report [21], sustainable development aims to guarantee inter- and intra-generational fairness concerning the use of natural resources. In this context we can distinguish economic sustainability and ecological sustainability. According to Pezzey [22] economic sustainability typically means that resources should be managed in such a way that the well-being of their users does not decline over time. Ecological sustainability means preserving ecological resilience over time, or ensuring that the flow of some ecological services does not decline over time [23, 24]. Efficient allocation of resources to local people exploiting different sources of income, including wildlife income, is therefore important. An allocation of resources is said to satisfy the efficiency criterion if the net benefits from the use of those resources are maximized by that allocation [25]. Management options that provide optimized benefits under conditions of highly fluctuating rainfall are therefore needed.

Recent work by Hein [26, 27] shows the importance of modeling ecosystem management options in systems that show complex dynamics like lakes, coastal estuaries, forests and rangelands. Complex dynamics include irreversible, non-linear and/or stochastic responses of the ecosystem to human and/or ecological factors [28]. Additionally, Schulz and Skonhofs [3], Skonhofs and Solstad [29], and Skonhofs [4] deal with the conflict over land use between wildlife conservation and rural development in developing countries through modeling studies. These studies are some of many attempts to use ecological-economic models to analyze management strategies of rangelands. Further, these studies are intended to provide guidance on how to maximize income from either wildlife or livestock keeping while maintaining the natural resource basis. As far as we know, no attempts have yet been made to formally analyze management of rangelands when local people have an option of exploiting both wildlife and livestock on a sustainable basis. In addition, few studies have looked at the potential of wildlife to reduce the impacts of rainfall fluctuations on income, instead focusing on crop income, private transfers (remittances) and livestock income as buffer against drought [30, 31].

The objective of this paper is to analyze whether wildlife income enables local people to reduce fluctuations in income caused by variations in annual rainfall. Research questions associated with this objective are: (1) What are the potential income levels (expected income and lowest income) given the different land uses and how do they respond to annual rainfall fluctuations? (2) What levels of livestock and people can the system support? (3) How can an increase in area under wildlife conservation (e.g., a national park) affect the land use allocation? The research questions are addressed by a bio-economic model that maximizes income from different sources (wildlife, livestock, irrigation farming and dry land crop cultivation), such that in low rainfall years people can still obtain sufficient income. The model maximizes expected income over eight years (consisting of different yearly combinations of good, average and bad rainfall status). The eight-year rainfall sequences are referred to as rain sequences in order to mimic rainfall fluctuations. Because we are interested in long-term sustainability of the system, the model is also used to provide baselines in terms of herbivore and human populations that can be supported in the agro-pastoral system of southeastern low veld of Zimbabwe. Finally, in systems exhibiting highly fluctuating rainfall, people can improve their welfare by exploiting a combination of wildlife and agricultural activities (livestock and cropping) to reduce fluctuations in their annual welfare. Exploiting different sources of income requires efficient allocation of resources. The most prominent resource is

land, which varies spatially in quality, and ecological resources require spatial connectivity. Because the spatial dimension is important in this allocation, we will show how an increase in the size of the park affects land use allocation to livestock, irrigation and dry land crop cultivation, and what it means to people's welfare.

Methods

The case study area

This study focuses on Gonarezhou National Park in southeastern low veld of Zimbabwe (Fig. 1). This is the second largest national park in Zimbabwe, where there are major conflicts of interest between several stakeholders on best land-use options and natural resource conservation strategies. Local communities rely on agro-pastoral activities, mainly livestock, for their livelihoods, while other stakeholders believe that wildlife use and tourism are much better in this area. This conflict has been compounded by new initiatives in Southern Africa to increase the area under conservation while improving livelihoods in the form of 'Transfrontier' parks. The study area forms part of one of these transboundary initiatives called the Great Limpopo Transfrontier Conservation Area (GLTFCA), joining Gonarezhou National Park to Kruger Park in South Africa and Limpopo National Park in Mozambique. The area outside the park includes five wards (3,078 km² in total) in Chiredzi district: Chikombedzi (ward 11: 358 km²), Gonakudzingwa (ward 12: 306 km²), Pahlela/Makanani (ward 13: 648 km²), Sengwe (ward 14: 813 km²) and Malipati (ward 15: 953 km²). Wards are sub-district units of local administration covering 150 to 1,000 km². These study wards are part of the Sengwe communal lands. Sengwe, Sangwe and Matibi 2 are the three main communal lands surrounding Gonarezhou National Park. The case study area is characterized by low rainfall, shallow soils with low agricultural potential, and high temperatures (☉ 39°C in summer). Annual rainfall ranges between 300 to 600 mm. The average rainfall recorded for this area based on 21-year rainfall data (from 1988 to 2008) from Mabalauta section of Gonarezhou National Park was 511 mm. Effective rainfall occurs from October to April, followed by a long dry season.

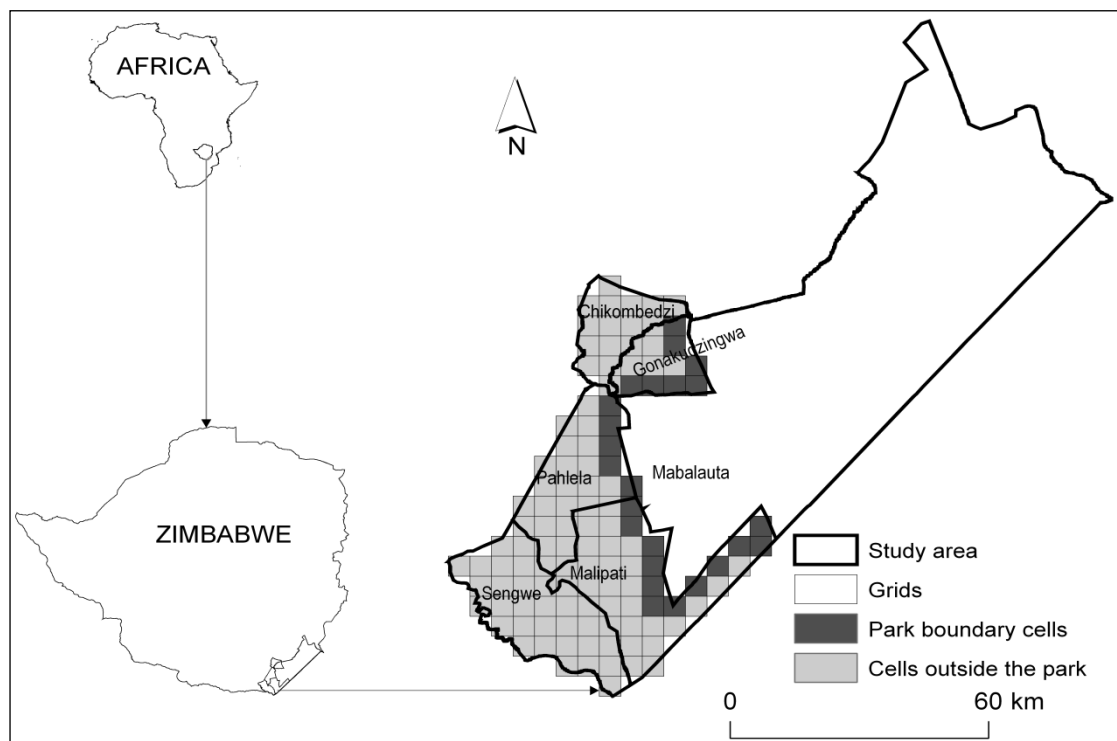


Fig. 1. South-eastern lowveld study area in Masvingo Province of Zimbabwe

The people in southeastern Zimbabwe are culturally described as Shangaan. Historically they were hunter-gatherers, not pastoralists; cattle and cropping are recent developments to their way of living. There is also a significant population of Ndebele and Shona people who came to the area after being displaced by land alienation for white farms. The human population in the communal areas grew more than tenfold during 1920-2000 [37, 38], with people surviving on less than US\$1 per day [37]. Densities of 29 people per km² have been reported for southeastern Zimbabwe compared to 3 people per km² and 2 people per km² for Botswana and Namibia [15], respectively. Extensive livestock husbandry is practiced in this area and small grains such as sorghum and millet are the major crops grown. However, maize is increasing in importance since its introduction by Shona and Ndebele settlers in the 1950s.

Model structure and specifications

Consider an area or ecosystem of fixed size divided into two sub-areas; a protected area (park) and an area outside the park [32]. In the area outside the park, a community of local people use land for agricultural production, i.e., livestock keeping, irrigation farming and dry land crop cultivation. There are four land use types considered in this model: wildlife, which is fixed in the park, and livestock, irrigation and dry land cropping, which are located in the area outside the park. The model maximizes eight-year income considering different rainfall probabilities and different proportions of land allocation, which we refer to as expected income. The eight years consist of different yearly combinations of good, average and bad rainfall status, referred to as rain sequence r . These sequences mimic different scenarios of rainfall fluctuations. A rain sequence is a sequence of 8 years, in each of which rainfall can be either wet, average, or bad. Rainfall status is denoted by α . In a 'good' year the area receives enough rainfall for agricultural activities; in an 'average' year the area receives average to below average (moderate) amounts of rainfall; in a 'bad' year the area receives too little or no rain, not enough to support agricultural activities. We assume that rainfall in a given year is independent of the rainfall of the previous or the next year. Given the rather low success rate in predicting annual rainfall, this is acceptable, although there might be cyclical forms of annual rainfall data. Therefore, each status has a probability of occurrence which we refer to as the rain probability, denoted as π_α , i.e., the probability of having a good, average or a bad year. For instance, given that probability π_α (good) = 0.35, probability π_α (average) = 0.45, probability π_α (bad) = 0.20, then a sequence consisting of eight 'good' years would have probability of rain sequence x_r as follows:

$$x_r(\text{wwwwwwww}) = 0.35^8 \quad (1)$$

where w denotes a 'good' rainfall year, while a sequence consisting of a 'bad' year followed by seven 'good' years will be as follows:

$$x_r(\text{bwwwwwww}) = 0.20 \times 0.35^7 \quad (2)$$

where b denotes a 'bad' rainfall year and w a 'good' rainfall year. The number of sequences equals 3^8 , and rain sequences are denoted by r , i.e. the vector of all possible rain sequences for y years. Hence the model maximizes expected income as follows:

$$\max \left\{ \sum_r \sum_y \frac{x_r I_{ry}}{(1+\rho)^y} \right\} \quad (3)$$

Where I_{ry} denotes discounted income in a rain sequence r in each year y ; x_r denotes the probability of rain sequence r ; and ρ denotes the discount rate.

Income I_{ry} is a function of the spatial allocation of land use types, the benefits of each land use type depending on location and rainfall, and the proportion of income that is received by the local people:

$$I_{ry} = \sum_c \sum_l A_{lc} b_{lcr} \sigma_l \quad \forall r, y \quad (4)$$

where A_{lc} denotes the total area (km^2) of land use l in plot c ; b_{lcr} denotes the benefits of land use l in plot c , in rainfall sequencer and in year y ; σ_l denotes the proportion of the allocation of the land use l that goes to local people as such that for instance $\sigma_{\text{irrigation}} = 1$ if all revenues of irrigation go to the local people, but $0 < \sigma_{\text{wildlife}} < 1$, if only a share of the revenues are received by the local people. A_{lc} is constrained by plot size a_c :

$$\sum_l A_{lc} \leq a_c \quad \forall c \quad (5)$$

Land use benefits (b_{lcr}) are equal to the revenues that are calculated as price per unit of output multiplied by the output per km^2 , minus the costs of producing a unit of output and damage costs due to predation or disease as follows:

$$b_{lcr} = p_l \theta_{lcr} - \varphi_l - h_{lcr} + tN_{lcr} - \mu M_{lcr} \quad \forall l, c, r, y \quad (6)$$

where p_l denotes the price of a unit of output for land use l in US\$; θ_{lcr} denotes the maximum potential output from plot c , in land use l , in rainfall sequencer, in year y (livestock units per km^2 or kg dry matter per km^2), one livestock unit is defined to be equivalent to an animal weighing 450 kg live mass; φ_l denotes the cost of producing a unit of output for land use l (US\$); h_{lcr} denotes the damage costs (US\$) as a result of wildlife predation in plot c , for land use l in rainfall sequencer in year y ; t denotes the price (US\$) for livestock sold; N_{lcr} denotes the number of livestock sold in plot c , for livestock land use, in rainfall sequence r in year y ; μ denotes the cost of purchasing a livestock unit (US\$); and M_{lcr} denotes the numbers of livestock units bought in plot c for land use livestock, in rainfall sequence r , in year y .

The cost of producing a unit of output for land use l (US\$) φ_l consists of fixed cost f_l and v_l variable costs of each land use as follows:

$$\varphi_l = v_l + f_l \quad \forall l \quad (7)$$

where v_l denotes variable costs per unit of output for each land use l and f_l denotes fixed costs per unit of output for each land use l .

Wildlife land use is restricted to the area inside the park, hence all plots that were part of the park **P** would likewise have wildlife as a land use. Whereas in the area outside the park the allocation of land use to a plot was also determined by the distance of the plot from the park boundary because the nuisance effects of wildlife on crop and livestock q_{lc} (probability of predation and crop raiding taken together, here named 'predation') depend on distance d_c to the park boundary as follows:

$$q_{lc} = \max \{0, \beta_l - \tau_l d_c\} \quad \forall l, c \quad (8)$$

where β_i is the base probability of predation, τ_i the marginal probability of predation and d_c denotes the distance between plot c and the park.

The (Euclidian) distance measure used is the straight line distance between the center of each park boundary plot to the center of the other plot c outside the park. We have two distance measures: d_c that denotes the distance between plot c and the original park boundary; and δ_c that denotes the distance between plot c and the new park boundary. These two distance measures allow the shifting of the park boundary towards the communal areas, mimicking the creation of a buffer zone. The original park boundary changes whenever the park increases, therefore the need for another distance measure that allows for the redefinition of the new park boundary. The second distance parameter δ_c is initially equal to d_c , but is later on changed as \mathbf{P} is redefined, because changing the park boundary affects the distance to the park. The model allocates land to irrigated agriculture based on whether the plot c is close to a water source, i.e., a river. The calculation of distance of a particular plot c from the river also followed the same principle explained above, where \mathbf{R} is the set of plots in the river. The distance of a plot from the river determined the allocation of land use to irrigation. The model calculates distance d_c in the following way:

$$d_c = \min_{c' \in \mathbf{P}} \left\{ \sqrt{(x_c - x_{c'})^2 + (y_c - y_{c'})^2} \right\} \quad \forall c \tag{9}$$

where c' is any other plot than c ; \mathbf{P} is the set of plots in the park; x_c is the x-coordinate of plot c ; and y_c is the y-coordinate of plot c .

Furthermore, in this model we also shift the park boundary towards surrounding communal areas, mimicking creation of a buffer zone. We use the second distance δ_c that allows for recalculation of equation 9 whenever we increase the park. Let's say we move the park boundary by a distance b , i.e., we say that every plot closer than b to the original park is converted to wildlife, hence we update plots' membership of \mathbf{P} using:

$$c \begin{cases} \in \mathbf{P} & \text{if } d_c < b \\ \notin \mathbf{P} & \text{if } d_c \geq b \end{cases} \tag{10}$$

The model calculates the maximum potential output denoted by ϑ_{dry} considering the carrying capacity of the plot k_c . Carrying capacity refers to the maximum possible stocking of herbivores that a rangeland can support on a long term sustainable basis [33]. Similar to Hein [27] the model is based on the assumption that not drinking water, but grass biomass is the limiting factor for livestock grazing in southeastern Zimbabwe. The reason is that a large number of boreholes have been constructed in the area, so drinking water for livestock is now generally also available even in the dry season; it takes a severe drought for the boreholes to run dry. Therefore, the maximum potential outputs ϑ_{dry} are a function of carrying capacity of the plot, the biomass demand per livestock unit (in the case of wildlife and livestock), the grain coefficient (for irrigation and dry land), the sum of rainfall scenarios and the rainfall coefficient. We assumed a fixed share of biomass produced to be grain, which we refer to as the grain coefficient. We use the grain coefficient to calculate the grain yields for irrigation and dry land crop cultivation. Hence, maximum output is calculated as follows:

$$\theta_{dry} = \frac{k_c (1 - o_l)^{z_c} \gamma_1}{\lambda_l} \sum_a v_{ary} \mu_{al} \quad \forall c, l, r, y \tag{11}$$

where k_c denotes the carrying capacity of plot c (tonnes dry matter per km²); o_l denotes the fraction of biomass production lost per unit of distance from the river for land use l ; z_c denotes the distance of plot c from the river; γ_l denotes the grain coefficient for land use l ; λ_l denotes the amount of biomass required to feed an animal for land use l , assuming that an animal requires feed amounting to an equivalent of 2.5 % to 3 % of its body weight per day; u_{ary} is a binary coefficient that denotes whether year y has rainfall status a in a rain sequence r ; μ_{al} denotes the rainfall coefficient of a rainfall status a for land use l . This factor indicates the relative impact of rainfall status (good, average and bad) on the different land uses.

Damages h_{clry} in this model are a function of the costs of predation and disease, their respective probabilities of occurrence in a plot and the maximum potential output from the plot as follows;

$$h_{clry} = \eta_l (1 - (1 - q_{lc})(1 - \varpi_l)) \theta_{clry} \quad \forall c, l, r, y \quad (12)$$

where η_l denotes the cost of predation or disease (US\$) for land use l ; q_{lc} denotes the probability of predation for land use l in plot c . ϖ_l denotes the probability of disease for land use l .

Herd dynamics

The model assumes that for irrigation and dry land cultivation, income for a particular year depends only on the current rainfall status in a rainfall sequence in the actual year, whereas for wildlife and livestock, yearly income depends on previous year output or the stock of animals that were there in the previous year and the rainfall status of the current year. Herd dynamics in a given plot depend on the size of the stock, the amount of rainfall, and a fixed growth rate:

$$S_{clry} = (1 + g_l)(1 - q_{lc})(1 - \varpi_l) S_{clr, y-1} \quad (13)$$

where S_{clry} denotes the size of the stock in plot c for land use l in rainfall sequence r in year y ; g_l denotes the growth rate of livestock and wildlife for land use l ; and $S_{clr, y-1}$ denotes the size of the stock of livestock and wildlife in plot c , for land use l in rainfall sequence r in year $y-1$. By assuming a fixed growth rate of the livestock population, we assume that there is no density dependency, and no immigration or emigration, but this we have tackled as follows:

For changes in the stock of livestock we assume buying and selling of the stock depending on what the system can accommodate in that particular year as defined in Equation (6). In 'good' years the reproduction of the stock leads to surpluses, hence farmers sell extra stock to the market at a price adjusted for the transactions cost:

$$N_{clry} = \max \{0, S_{clr, y-1} (1 + g_l)(1 - q_{lc})(1 - \varpi_l) - k_{clr, y-1}\} \quad \forall c, l, r, y \quad (14)$$

In a 'bad' year we assume farmers maintain only as many animals as can be supported by the plot in that year and animals that cannot be fed are sold to the market. However, in some cases, 'good' years may come after an 'average' or 'bad' year, in which case the stock of livestock available will be less than what the system can carry that year. Therefore we assume in such years farmers buy livestock M_{clry} as follows:

$$M_{clry} = \min \{ \psi, \max \{0, k_{clr, y-1} - S_{clr, y-1} (1 + g_l)(1 - q_{lc})(1 - \varpi_l)\} \} \quad \forall c, l, r, y \quad (15)$$

where ψ denotes the maximum number of animals that farmers can afford to buy. Furthermore, we assume that income from livestock consists of selling milk and selling live animals to the market. We assume no market failure or limitations in acceptance by the market or delivery to the market.

In the specification of the model we had to make a number of simplifying assumptions, e.g. in reality wildlife is a quite fluid resource, which is highly variable between places, years and seasons. In the model we have focused on average annual revenues from wildlife.

Scenarios

The model calculates the increase in park as defined in Equations (9) and (10) with b increasing from zero to 13.5 km in steps of 1.5 km for each scenario. In other words, Scenario 1 shows no shift; in Scenario 2 the park increases by 1.5 km; in Scenario 3 by 3 km; in Scenario 4 by 4.5 km; in Scenario 5 by 6 km; in Scenario 6 by 7.5 km; in Scenario 7 by 9 km; in Scenario 8 by 10.5 km; in Scenario 9 by 12 km; and in Scenario 10 by 13.5 km. In each scenario the model allocates plots to different land uses and calculates expected income given the 3⁸ different rain sequences. In this study probabilities of drought indicate the impact of drought since in semi-arid areas annual rainfall varies markedly between years. Hence for example, a probability of drought value of 0.3 means that the area has received 30 % of the potential annual rainfall. Running the model with different probabilities of drought was also used as a sensitivity analysis to see if the model behaves differently when the value of a parameter is changed.

In the case study area the boundaries of the park and the buffer zones are well known [17, 58]. Wildlife sometimes migrate from the park to surrounding areas, but wildlife densities are of course much higher in the park than outside. We consider that shifting the park boundary will accordingly affect the presence of wildlife in the relevant cells.

Data

The parameters in equation (8) were derived through regression analysis on the basis of predation data by Kuvawoga [34]. Carrying capacity K_c (tonnes dry matter per km²) of the area was taken from an analysis of potential productivity data for much of the area surrounding southeastern low veld by Pachavo and Murwira [35], who found values ranging between 2 to 4 tonnes dry matter per hectare. We used an average figure of 3 tonnes dry matter per hectare in this study. The amount of plant biomass λ_i required to feed one livestock unit during one year was estimated using the energy requirements per livestock unit, i.e., 2.5% (expressed as dry weight) of animal mass per day [36]. Rain coefficients μ_{ryl} and probabilities of occurrence of different rain status π_a were estimated using long term rainfall data from Meteorological Department of Zimbabwe, records for Buffalo Range (1950 to 2008). Prices p_i , costs φ_i per unit of output for each land use were estimated based on averages prices, variable v_i and fixed costs fl for land uses in southeastern low veld of Zimbabwe. These were established through a two-tier longitudinal survey done in October 2008 and July 2009. The survey data were also augmented by secondary data from Extension and Veterinary Departments of Zimbabwe. Local people's share of revenues from wildlife σ_i were estimated from Rural District Council CAMPFIRE records (1996 to 2009). See Appendix 1 for data summary.

Results

We applied the model to calculate expected income and lowest income for an increasing park size, as a result of shifting the park boundary. Such an extension of course will affect the land use in the plots outside the park area. In this study we present two possibilities: one where model income is allowed to be negative in any year and the other where model income has a lower bound of zero. Without the lower bound of zero, lowest income becomes negative and irrigated agriculture is not allocated as shown in Figures 2a to 2c. Under this possibility, irrigated agriculture is not allowed because expected income is maximized by livestock, whereas with the addition of a lower bound of zero, meaning that income should be positive in any year, the lowest income becomes zero (Figure 3). Under this possibility, the model

needs irrigated agriculture to buffer incomes in bad years. The model shows that after suffering eight years of consecutive droughts, local people lose so much that the lowest income becomes negative.

The situation only improves with the addition of wildlife plots, resulting in lowest income increasing with addition of wildlife (Figs. 2a to 2c). Figures 2a to 2c show the relationship between expected income and lowest income with increasing park size at different probabilities of drought. In this study probabilities of drought are proportional to the impact of drought. In this regard the impact of drought increases as the probability of drought increases. Figures 2a to 2c show a general trend where with an increase of wildlife plots, expected income stays the same until an increase in park extension of 4.5 km. Thereafter it starts to decline. As drought probability increases: from 0.20 to 0.60 (Figures 2a to 2c) expected income follows a similar declining trend. However, lowest income increases with the extension of the park and decreases with an increase in probability of drought (Figures 2a to 2c). Both expected income and lowest income are higher under a lower drought probability (Figure 2a to c). The declining trend in expected income is also evident even with the addition of a lower bound of zero, which resulted in allocation of irrigated agriculture (Figure 3). Figure 3 also shows larger expected income under a lower drought probability, with the expected income declining as drought probability increases (from 0.20 to 0.60).

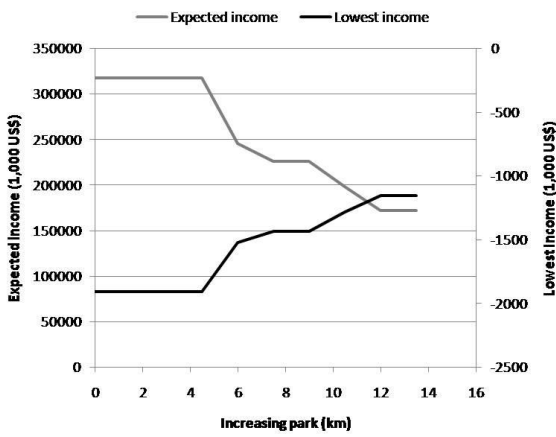


Fig. 2a. Relationship between expected, lowest income and increasing park when probability of drought is 0.20.

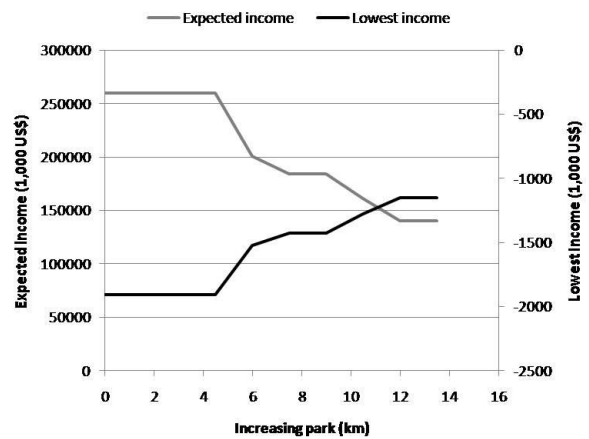


Fig. 2b. Relationship between expected, lowest income and increasing park when probability of drought is 0.40.

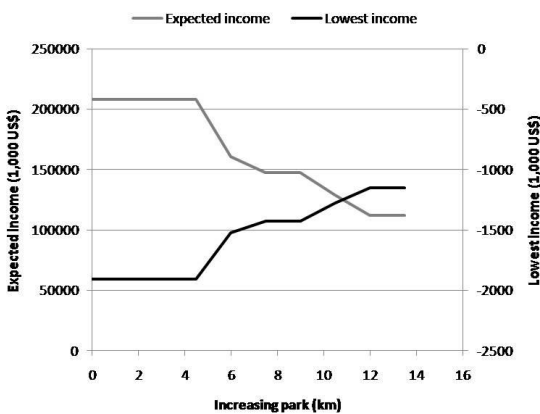


Fig. 2c. Relationship between expected, lowest income and increasing park when probability of drought is 0.60.

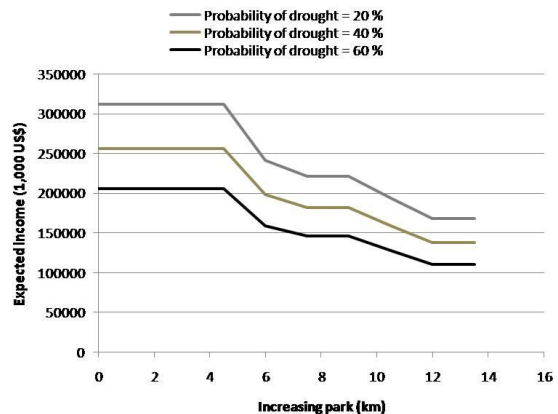


Fig. 3. Relationship between expected income and increasing park when probability of drought is 0.20, 0.40 and 0.60. In these model runs income in any year must be positive, which causes the lowest income to be zero.

We also evaluated how an extension of the park affects land use allocation, given the two possibilities: one without and the other with the addition of a lower bound of zero. Figure 4a shows the proportions of land uses as park area increases. At an extension of the park shifting the boundary by a distance below 4.5 km, more land is allocated to livestock than wildlife, whereas an increase in the park by more than 4.5 km leads to more land being allocated to wildlife (Figure 4a). The increase in park beyond 4.5 km, however, leads to a decrease in expected and an increase in lowest income as shown in Figure 2a to c. Addition of a lower bound of zero results in wildlife substituting for some but not all irrigated agriculture (Figure 4b). However, irrigated agriculture shifts to some few plots previously used for livestock (Figure 4b). Results also showed that an increase in impact of drought did not result in a different land use allocation (Figure 4a and b).

We also evaluated how many persons and herbivores could be sustained in southeastern low veld of Zimbabwe. Model results show 8,510 livestock units as herbivore numbers that can be optimally managed under this system outside the park. This is based on the size of the study area, its potential productivity and the feed requirements per livestock unit per year. Whereas in 2008, dip tank records show that there were 39,200 mature cattle. This implies that the number of cattle is 5 times higher than what is sustainable. Based on total area outside the park considered in this study (3,077 km²), the area can sustain a total of 770 households. This is based on the fact that a household requires 4 km² to cover its needs for arable and grazing land according to an estimate by Cumming [37]. However, the human population estimate for wards 11 to 15 was 36,986 people and 6,485 households in year 2000 [38]. This indicates that the system carries eight times more people than what is sustainable.

Discussion

This paper presents a modeling approach to the analysis of whether wildlife has an insurance value to local people during years when rainfall is low, with a special reference to sub-Saharan Africa. This work was motivated by the assertion that describes biodiversity as “the wealth of the poor” [39] and hence we expected wildlife to cushion local people against income fluctuations due to drought.

In the area considered in this study we find that wildlife provides local people with insurance against rainfall fluctuations when local people do not engage in irrigated agriculture. As shown in Figure 2a to c, lowest income increases as expected income declines with addition of wildlife plots, suggesting that there is potential to reduce the negative impacts of droughts using wildlife income. In an analysis of possible land use options at the interface of livestock and wildlife in rural communities near Kruger National Park, Chaminuka [40] showed that introduction of wildlife and tourism-based land uses can substantially increase the benefits derived from the land. This may mean that wildlife income may be substantial to insure local people against drought under the South African set up. These results are in agreement with findings from this study; however, the negative lowest incomes in this study may be due to the fact that households and communities in Zimbabwe only get a small fraction of the wildlife revenues. Under the current situation of economic hardship, rural district councils may use a bigger fraction from wildlife revenues for their own activities at the expense of communities.

However, model results seem to be affected strongly by the profitability of irrigated agriculture, where the allocation of irrigated agriculture just prevents income to be negative (Figure 3). The finding that expected income declines with addition of wildlife plots shows that agro-pastoral activities (livestock keeping and crop cultivation) will remain important in the study area because as more plots are added to wildlife, they take more land that provides substantial income to people, especially livestock and potential irrigable land for cropping. The expectation was that wildlife offers a more stable, albeit generally lower income, especially during dry years. Apparently it can only do so in the absence of irrigated agriculture, which suggests that wildlife is second to irrigation for reducing fluctuations in local income due to rainfall variability, likely because of the very high variable and fixed costs of exploiting the wildlife resource. Such costs severely reduce the net income from wildlife so that it becomes worthwhile for rural households to

focus on agro-pastoral activities to cope with drought, leading to a decrease in overall expected income in years with low rainfall.

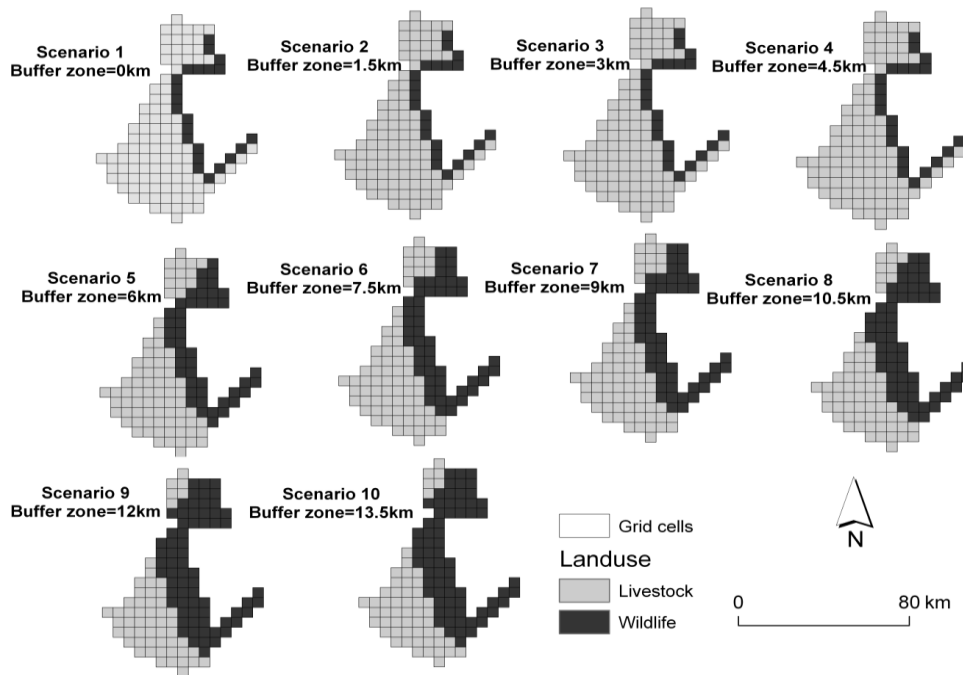


Fig. 4a. Land allocation to land uses: Livestock and Wildlife with an increase in park (Scenario 1 to 10) when probability of drought is 0.20, 0.40 and 0.60.

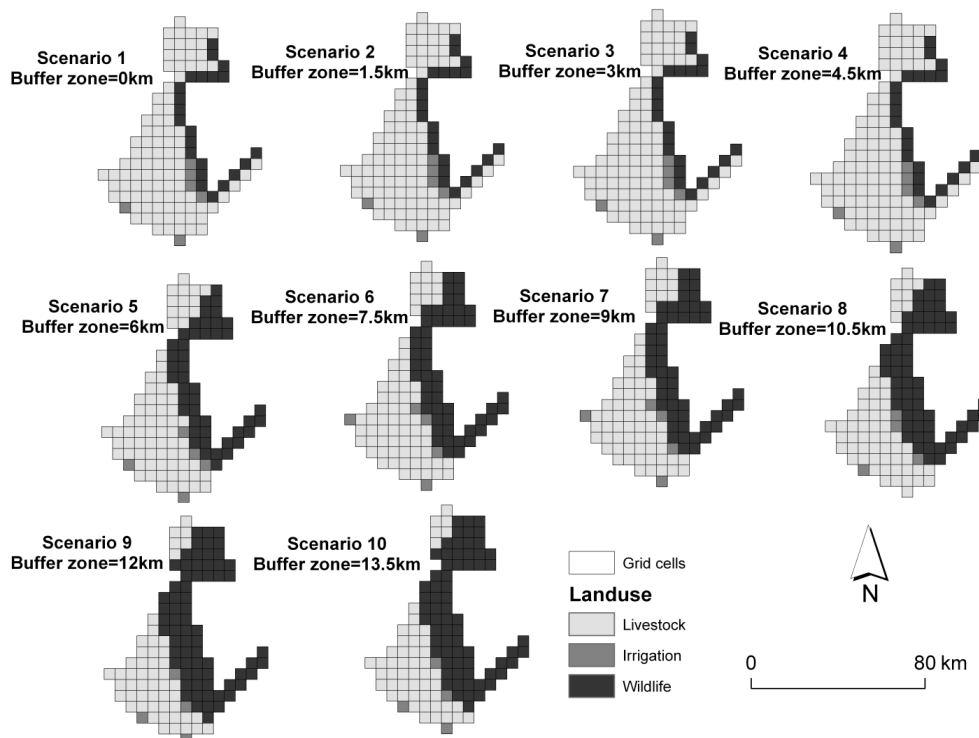


Fig. 4b. Land allocation to land uses: Livestock, Irrigation and Wildlife with an increase in park (Scenario 1 to 10) when probability of drought is 0.40.

This finding is in line with modeling results from Skonhoft [4] who stated that wildlife conservation can work directly against the interests of local people. Johannesen [8] also reported that expansion of protected areas may reduce welfare of the local people. Our findings extend this literature by adding that even with the argument of wildlife being more adapted to semi-arid savannas than introduced livestock [41], income from wildlife can only provide insurance value to local people during dry years in the absence of irrigation. However, because risk is the major determinant of starvation and systems breakdown, while expansion may decrease people's income, we conclude that expansion buffers that income better against droughts, thus increasing people's safety.

In some areas, the rural poor protect themselves from weather-related losses using various structural mitigation measures [42]. Building dams and supplementary irrigation are examples of structural drought mitigation measures. Such measures, however, are not always feasible, reliable, or cost-effective. In sub-Saharan Africa, rural households have a number of drought-aversion strategies which can be referred to as indigenous responses or risk-coping strategies [43, 44]. Food sharing, exploitation of wild resources such as fruits, diversification of food supply, off- and non-farm employment and sales of livestock, poultry and their products, and handicrafts [43-45] are examples of indigenous responses to drought. These strategies are effective for independent risks but ineffective for covariate or systemic risks such as drought, because when many households within the same community face risks that create losses for all, traditional coping mechanisms are likely to fail [46]. Traditional insurance instruments such as crop insurance can be used to cope with the risk of extreme weather events [42]. However, insurance markets are underdeveloped and often non-existent in rural areas of lower income countries due to poor contract enforcement, asymmetric information, high transaction costs and high exposure to spatially covariate risks [42, 44].

Faced with such limitations, it is hoped that wildlife could offer insurance to local people against drought, since wildlife income depends on external factors, given that safari hunters and most tourists are usually rich foreigners who cope relatively better with similar sources of risk in their own countries [47]. Otherwise, the only option would be industrialization as suggested by Malthus [48]. China is a good example of how a country can rise from poverty within a generation and become a dominant player on the global scene. China's industrial sector has been impressive, averaging about 12 % per annum over 1985-2005 [49].

Additionally, the higher numbers of herbivores currently in the systems would render other land uses such as wildlife seem unprofitable to local people. Current stocking rates would make livestock production seem more favourable than wildlife; however, if the area is to be managed considering environmental sustainability, then both human and herbivore densities need to be controlled [50]. Many African rangelands are heavily stocked with domestic animals, but also receive low and erratic rainfall. In a dry year, or after a run of dry years, the animals often yield very little output in terms of secondary production and occasionally die in large numbers [51, 52]. In extreme instances, herbivores, when at high density, are 'ambushed' by a drought that cuts the food from under them since droughts are frequent and often severe. Production losses brought about by these crashes bring about anguish and suffering for people that live under these systems. Perhaps with lower numbers of herbivores and people in the area, wildlife revenues could play a significant role in cushioning households against income fluctuations during drought years.

Furthermore, based on levels of benefits established above (Figure 2), the optimal per capita lowest income per year translates to less than \$0.50 per day, whereas the international poverty threshold stands at US\$1.25 per day [53]. These findings agree with the assertion that high numbers of people in sub-Saharan Africa and those surrounding protected areas live in poverty [54, 55]. What is even more worrying is that recent statistics show Sub Saharan Africa worsening in absolute poverty measures, whereas other developing regions show marked improvement [56, 57]. Given the herbivore and human densities in areas

surrounding parks, associated with uncertainty in annual rainfall caused by climate change, we can only expect the welfare of inhabitants to continue to show a spiral decline.

Finally, interest in studying the spatial configuration of land uses in southeastern low veld was driven by the need to understand land use conflicts between conservation and local people. We argue that people need to utilize resources in crown lands, but their exploitation of this resource should not allow mixing of land uses, especially between livestock and wildlife because of disease transfer, hence a spatial configuration that separates these land uses becomes important. The spatial configuration that was found to be optimal in this study is the one where the park increases to between 11 and 12 km. At park increase of 11 to 12 km is where the increasing lowest income meets the decreasing expected income (Figure 2a to c).

Implications for conservation

The results of the model that we present in this paper show that an extension of the park will result in a decline in expected income and an increase in the lowest income which people get during dry years in the absence of irrigated agriculture. Wildlife income has potential to offer insurance to local people during droughts to compensate for losses in expected income from livestock. There was an overall decline in expected income with the addition of wildlife plots with or without irrigated agriculture added to the model. Possible reasons include high costs of exploiting the wildlife resource and a small fraction of wildlife revenues received by households and communities. In order to search for sustainable solutions in areas such as the southeastern low veld of Zimbabwe, it is important to be aware that the current human population and livestock densities are far above sustainable levels. Our results therefore suggest that current and future efforts to conserve biodiversity are doomed to fail if no efforts are made to decongest areas surrounding parks with high densities of human and herbivore populations, and to let local households earn more revenues from wildlife.

These results provide evidence to policy makers that rainfall variability is one of the strongest justifications for adopting wildlife and other natural resource-based land uses as an alternative and sustainable strategy for social and economic betterment [53]. This notion has also promoted recent conservation development paradigms called Transfrontier Conservation Areas (TFCAs) or mega-parks that cross international borders [58]. The rationale is that adding wildlife conservation as a land use could diversify and consequently reduce risk [12]. Results from this study have shown that such initiatives (TFCAs) may improve the livelihoods of those living around them, particularly their ability to cope with drought risk, depending on the profitability of irrigated agriculture. Policy makers should also look into ways of controlling livestock and human densities surrounding parks if the goal is sustainable natural resources management. The general approach taken in this study contributes to an understanding of how people can balance conservation against development objectives in systems that show strong variability in rainfall.

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Appendix 1: Parameter values used in the model for wildlife (W), livestock (L), irrigation (I) and dry land crop cultivation (D).

Parameter	Explanation	Value			
		W	L	I	D
K_c	Biomass production for plot c (tDM/km ²)	300	300	300	300
λ_l	Biomass demand for land use l (tDM /lu /Year)	4.1	4.1	4.5	4.5
γ_l	Grain coefficient for land use l	1	1	1.3	0.65
p_l	Price per unit of output for land use l (US\$/t or US\$/lu)	350	300	265	265
V_l	Variable cost per unit of output for land use l (US\$/t or US\$/lu)	250	50	70	35
f_l	Fixed cost per unit of output for land use l (US\$/t or US\$/lu)	300	100	200	100
σ_l	Local people's share	0.7	1	1	1
η_l	Cost of predation per unit of output for land use l (US\$/t or US\$/lu)	0	300	265	265
o_l	Fraction of biomass production for land use l	0	0	0.20	0.20
ϖ_l	Probability of disease for land use l	0	0.02	0	0
β_l	Base probability of predation for land use l	0	0.15	0.10	0.10
τ_l	Marginal probability of predation for land use l	0	-0.117	-0.093	-0.093
μ_{ryl}	Rainfall coefficient for rainfall sequence r in year y for land use l . Where w denotes Good, a denotes average and b denotes Bad.	$w = 1$ $a = 0.8$ $b = 0.6$	$w = 1$ $a = 0.7$ $b = 0.5$	$w = 1$ $a = 0.7$ $b = 0.4$	$w = 1$ $a = 0.6$ $b = 0.2$
g_l	Growth factor for land use l	0.3	0.2	0	0
π_a	Probability of occurrence for rain status α Where w denotes Good, a denotes average and b denotes bad			$w = 0.35$ $a = 0.40$ $b = 0.25$	
ψ	Maximum number of livestock that a farmer can purchase during a good year after a bad year			5	
ρ	Discount rate			0.05	