African wildlife and people: finding solutions where equilibrium models fail

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To
Sibongile, Tadiwa, Tanaka
and my parents
…..you are all a constant source
of inspiration, much love.

Contents

Abstract

The equilibrium and non-equilibrium paradigms establish the ecological basis for the rangeland debate. However, despite extensive research on rangeland dynamics the debate still remains unresolved. Additionally, concern about the natural environment, particularly wildlife conservation has continued to grow in importance in these rangelands. This thesis adopts a bioeconomic approach in analysing the implications of non-equilibrium dynamics for the efficient and sustainable management of wildlife and livestock in dryland grazing systems of southeastern Zimbabwe. The thesis focusses on the role of abiotic and biotic factors in determining plant species composition in order to understand possible sources of disturbance for dry ecosystems to shift from one state to the other. Findings showed soil pH and rainfall as the main determinants of vegetation composition in this semi-arid system. However, explained variation in this study was low, suggesting that other important variables might have been missed, or that the ecosystem is responding dynamically to changes not easily captured in environmental variables. Secondly, the relevance of non-equilibrium theory to the rangeland system of southeastern lowveld of Zimbabwe was studied by testing the presence of crashes and studying factors that explain annual changes in livestock numbers. Additionally the implications of non-equilibrium dynamics for herd dynamics were studied by analysing the effect of drought on cattle age and sex categories and their recovery from drought. Results indicated the presence of crashes, lags and thresholds with rainfall having an overriding effect on annual changes in livestock numbers. Immigration of livestock was important during dry years whereas NDVI became an important variable during wetter years. The impact of drought was greater on juvenile bulls and calves and these two groups recovered faster from drought than the other age categories. Drought had similar effect on males and females. From an economic point of view, the question was addressed of how risks can be minimized, especially of household income in order to improve human welfare. The role of wildlife income in reducing fluctuations in household income due to rainfall fluctuations was assessed. The addition of wildlife as an asset to rural farmers' portfolio of assets showed that wildlife can be used as a hedge asset to offset risk from agricultural production without compromising on return. However, the power of diversification using wildlife was limited since revenues from agriculture and wildlife assets were positively correlated. This implies that wildlife income could reduce fluctuations in household incomes only to a limited extent. Subsequently, modelling approaches were used to simulate the agricultural and wildlife systems of southeastern Zimbabwe and the models were used to test the extent to which wildlife income offers insurance value to local people. Findings showed that if wildlife area is increased, and in the absence of irrigated agriculture, this would result in a decline in expected income and an increase in the lowest income which people get during dry years. This suggests that wildlife income has potential to offer insurance to local people during droughts to compensate for losses in expected income from livestock. However, because risk is the major determinant of starvation and systems breakdown, while addition of wildlife plots may decrease people's income, we conclude that wildlife buffers that income better against droughts, thus increasing people's safety. Overall, findings from this research contribute towards an understanding of how people may live in a system that shows non-equilibrium dynamics.

Key words: non-equilibrium, equilibrium, rangeland dynamics, southeastern Zimbabwe, wildlife, livestock, disturbance, drought, risks, crashes, household income, insurance, diversification, herbivore densities, vegetation production and composition, lags, thresholds.

Chapter 1

General Introduction

Background

Humankind developed in southern Africa, and agriculture has been practiced there while climates changed. Both wild animals and civilizations have been coping with droughts for millennia. There is, however, some evidence from southern Africa that the Mapungubwe society in the Limpopo valley, South Africa disappeared and that animals went extinct due to drought (O'Connor and Kiker 2004), and impacts on individual lives have been often dramatic. This study pays particular attention to arid and semi-arid savannas (traditionally called drylands) in Africa. In these arid and semi-arid savannas, droughts are common features, sometimes lasting for years. Furthermore, low and variable rainfall discourages crop production hence people rely mainly on the natural resource base, earning their living from agriculture, fishing, forestry and hunting.

Across the savannas of Africa, grasslands are being changed into cultivation due to increasing human population, at the expense of decreasing wildlife populations (Prins and Grootenhuis 2000). African savannas however, still contain pockets of wilderness surviving as protected areas, but even there, species richness of large mammals is decreasing (Prins and Grootenhuis 2000). The inevitable result is the loss of most of the wild plants and animals that occupy these natural habitats, at the same time threatening the well-being of the inhabitants of these savannas (Wright and Boorse 2010). Hence, to facilitate the management of arid and semi-arid savannas for both biological conservation and sustainable use (improving human welfare), an improved understanding of the complex dynamics of these savannas is critical. Savannas have been intensively studied but are nevertheless not well understood and despite the initiatives already undertaken, calls for further commitment to the cause of biodiversity are often made. Savannas are ecosystems characterized by the co-existence of woody species (trees and shrubs) and grasses (Scholes and Archer 1997). More than half the surface area of the African and Australian continents, about 45% of South America and 10% of India and Southeast Asia are covered by tropical (and subtropical) savannas (Scholes and Archer 1997). Savannas support a large proportion of the world's human population and most of its rangeland, livestock and wild herbivore biomass (Scholes and Archer 1997).

History of equilibrium paradigm

Ecology has long been shaped by the ideas that stress the use of resources and the competition for those resources, and by the assumption that populations and communities typically exist under equilibrium conditions in habitats saturated with both individuals and species (Rohde 2005). In addition, this notion of ecosystems at equilibrium regards ecosystems as stable environments in which species interact constantly in well-balanced predator-prey and competitive relationships, leading to the popular idea of “the balance of nature”. The idea of balance of nature was given its first name “*oeconomia naturae*” in 1749 by Carl Linnaeus and has been a background assumption in ecology for centuries (Egerton 1973; Botkin 1990; Pimm 1991; Wu and Loucks 1995). The main belief of this idea was that nature could be understood in terms of the balance of destructive and conservative forces and that nature would maintain a permanence of structure and function if left undisturbed (Botkin 1980; McIntosh 1985; Wu and Loucks 1995). The modern derivatives such as equilibrium, steady-state, stability and homeostasis, are central concepts of the classical equilibrium paradigm (McIntosh 1985; DeAngelis and Waterhouse 1987; Botkin 1990) which dominated ecological thought during the 1960s and 1970s (Wu and Loucks 1995). To illustrate this we borrow from Scheffer (2009) an example of a herbivore population that has reached the carrying capacity of the environment. Carrying capacity refers to the maximum possible stocking of herbivores that rangeland can support on a long-term sustainable basis (de Leeuw and Tothill 1993). The herbivore population is thought to be at equilibrium density as a result of a balance between birth and death rates (Figure 1.1). If a certain proportion of the population is killed by an adverse event, there would be more resources for the survivors. The result will be an increase in birth rate and/or a reduced death rate leading to the population growing back to the equilibrium density. However, if densities surpass the carrying capacity, reduced birth and/or increased death rate will push the population density back to the equilibrium. An equilibrium point or state refers to particular system state at which all the factors or processes leading to change are being resisted or balanced (Wu and Loucks 1995). It is this assumption of equilibrium that, ‘erroneously’, provided the basis for rangeland management and pastoral development in dryland Africa up to the 1990’s (Scoones 1993).

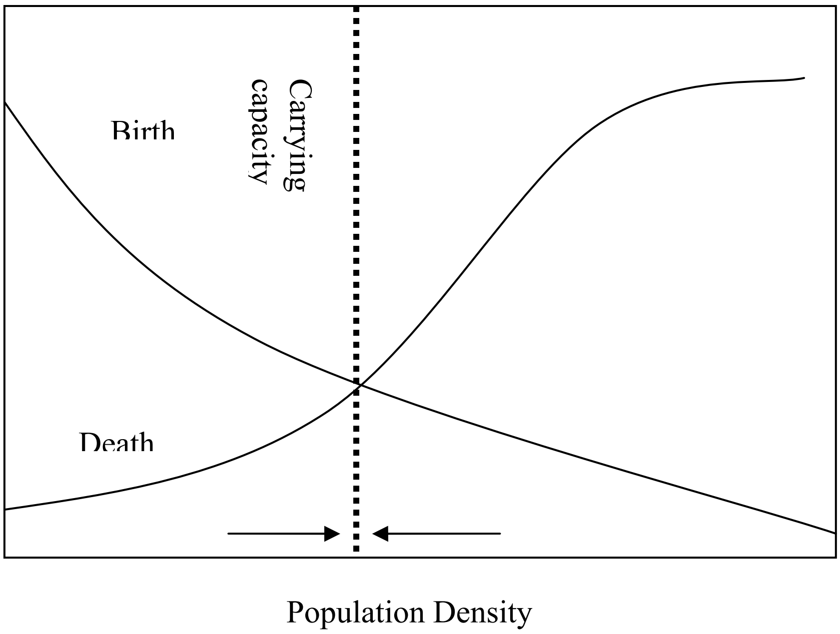


Figure 1.1: The concept of stable dynamic equilibrium illustrated for the case of a hypothetical population that settles at a density that corresponds to the carrying capacity of the environment. This is the net result of per capita birth and death rates. Adapted from Scheffer (2009).

Equilibrium dynamics in dryland savannas

Rangelands have been seen as equilibrium systems driven by abiotic events, as described by Clements' (1916) plant succession theory (Smet and Ward 2005). These systems develop towards an equilibrium along a series of successive stages starting at the pioneer stage and eventually reaching a climax stage determined by the constraints of the environment (Smet and Ward 2005). The major argument being that herbivore populations are tightly coupled to the availability of forage and can thereby impact adversely on vegetation (movement towards pioneer stage) when densities of herbivores exceed forage production (Sullivan and Rohde 2002). In other words, herbivore numbers are controlled through the availability of forage and the availability of forage is controlled by animal numbers, a pattern of negative feedback which produces a stable equilibrium between animal and plant populations (Behnke and Scoones 1993). Hence, there is a notion that there is an ecological carrying capacity for livestock that is determined by the availability and quality of vegetation at equilibrium (Smet and Ward 2005). Traditional range management in southern Africa has been long in the grips of this old and possibly out-dated thinking, and partly still is, as indicated by concepts like 'increaser species' and 'decreaser species' found in many veld management books (e.g., Tainton *et al*. 1999).

The non- equilibrium paradigm

A more recent understanding of ecosystems, however, shows that ecosystems operate in dynamic, changing ways (Wright and Boorse 2010). Fluctuations in the environment and all kinds of smaller or larger perturbations prevent equilibrium (Scheffer 2009). Ehrlich and Birch (1967) had already promoted the idea that populations as well as their environments change constantly, and that the idea of a balance of nature could be misleading (Wu and Loucks 1995). Holling (1973) pointed out that the equilibrium-centred view is static and cannot account for the commonly observed transient behaviour of ecological systems. Thus the general view about ecosystems is that they are composed of disorder, diversity, instability and non-linearity (Murphy 1996).

Beginning in the mid-1970s, ecological studies increasingly showed that ecosystems often change in much more complex ways than previously assumed (Holling 1973; Ludwig *et al.* 1978; Westoby *et al.* 1989; Rietkerk *et al*. 1996; Scheffer *et al.* 2001). Complex ecosystem dynamics comprise irreversible, non-linear and/or stochastic responses of the ecosystem to human and/or ecological factors (Hein 2005). However, the most common type of complex dynamics is chaos (Scheffer 2009). Although it incorporates elements of chance, chaos is not random disorder. Rather, chaos is defined as aperiodic long-term behaviour in a deterministic system that exhibits sensitive dependence on initial conditions (Strogatz 1994). The emergence of chaos theory has made scientists acutely aware of the complex dynamics and unpredictability of nonlinear systems (Wu and Loucks 1995). The implications are that the long-term behaviour of a chaotic system is fundamentally unpredictable due to its sensitivity to initial conditions and even if we know exactly the rules that govern the system, the final result remains unpredictable, because we can never precisely determine the current state (Scheffer 2009). With the equilibrium view of systems, ecosystems maintain their stability by means of negative feedback, a mechanism that, like a thermostat, takes corrective action to discourage deviation and preserve a steady state. Thus, while negative feedback regulates, positive feedback amplifies deviations, working to destabilize existing states and introduce new patterns (Murphy 1996).

Whereas these processes likely contribute to ecosystem functioning, their relative weights and interactions remain evasive (Roy and Chattopadhyay 2007). In summary, environmental fluctuations pushing the system around can be a cause of complex dynamics. However, a combination of the dynamics generated by intrinsic mechanisms and the external factors can be a major cause of complex dynamics (Roy and Chattopadhyay 2007; Scheffer 2009).

Non-equilibrium dynamics in dryland savannas

For rangelands in dryland Africa, extreme and unpredictable variability in rainfall are considered to confer non-equilibrium dynamics by continually disrupting the tight consumer-resource relations otherwise considered to pull a system towards equilibrium (Rietkerk *et al.* 1996; Sullivan and Rohde 2002). Ellis and Swift (1988) argued that vegetation in African pastoral ecosystems is not controlled by livestock density but rather by abiotic events such as drought. It was noted that arid and semi-arid areas are characterized by high spatio-temporal variability in precipitation with low predictability; hence the systems become non-equilibrium (Noy-Meir 1973; Smet and Ward 2005). In other words, animal populations in these systems spend most of the time recovering from the previous drought, and rarely reach densities at which density-dependent mechanisms act to moderate the animal populations (Derry and Boone 2010). The term "non-equilibrium" is normally used as a broad term to mean "not at equilibrium" rather than implying that density-dependent processes are not important (Gillson *et al.* 2005). Though environmental stochasticity can affect population size (DeAngelis and Waterhouse 1987), it is only a very small subset of non-equilibrium theories, termed "disequilibrium", that assert that environmental variability can completely override the effects of biotic interactions (Gillson *et al.* 2005). This disequilibrium viewpoint is extreme, however, and most authors now agree that both density-dependent and environmental variables affect population size (Wu and Loucks 1995; Gillson *et al.* 2005). It is now argued that the system dynamics of rangelands are better described as a continuum between equilibrium and non-equilibrium (Wiens 1984) where everything in between is in disequilibrium, and where the position along this gradient is determined by the strength of coupling between the plants and animals (Derry and Boone 2010).

Uncertainty and risk linked with non-equilibrium dryland ecosystems

It is widely recognized that a high level of uncertainty typifies the lives of rural farmers in developing countries (Ellis *et al.* 1993). Non-equilibrium dynamics bring additional uncertainty and risk to the system. On one hand, uncertainty can be defined as existence of more than one possibility, i.e., the "true" outcome, state, or result, cannot be known (Perman *et al.* 2003). Moreover, typically not all possible outcomes are equally desirable (Muchapondwa 2003), and the outcome of uncertain events can make the difference between survival and starvation (Ellis *et al.* 1993). Output uncertainty becomes the dominant type of uncertainty under these circumstances due to varying weather conditions. On the other hand, risk is defined as a state of uncertainty where some of the possibilities involve a loss, catastrophe, or other undesirable outcome (Perman *et al.* 2003). Meanwhile, the definition of economics is the study of the allocation of limited resources across unlimited wants (McEachern 2000). That is people would like to have it all, but there is not enough land, labour or capital (traditional economic resources) to do so. The economist usually measures the success of any such allocation by an efficiency criterion: resources going to their highest valued use (Elliott 2005). That is, is land, labour, capital and time being put towards the goods and outcomes that people most highly value? If so, it is argued that the economy is working well. If not, people must consider a redistribution of those resources and time to the creation of different, more highly valued goods and outcomes (Elliott 2005). Furthermore, because people live in a world where non-renewable resources are essential inputs to production, then people have to consider sustainable development. I prefer the definition of sustainable development by Asheim (1994) who defines sustainability as a requirement to our generation to manage the resource base such that the quality of life we ensure ourselves can potentially be shared by all future generations. However, attempts to understand efficient and sustainable ways to improve biodiversity and human welfare in systems showing non-equilibrium dynamics have been rare. The behaviour of non-equilibrium systems is characterised as more dynamic and less predictable than equilibrium systems. Therefore, I believe that non-equilibrium dynamics in dryland ecosystems present a different kind of management problem for both livestock and wildlife systems, since their management has been dictated by the equilibrium assumption. Additionally, loss of biodiversity is regarded today as one of the great unsolved environmental problems (Swanson 1991; Figge 2004). The decline in biodiversity as an ecological problem has attracted increased public interest in recent years (Figge 2004). Faced with this biodiversity crisis, the challenge is to find ways to respond in a flexible way to deal with uncertainty and surprises brought about by non-equilibrium dynamics.

In sub-Saharan rangelands, high levels of biodiversity still exist; therefore income from wildlife utilization can potentially complement agro-pastoral incomes for local people in communal systems that show high fluctuations in annual rainfall. The emergence of portfolio theory (Markowitz 1952, 1959) helps in providing a good theoretical framework. The basic idea of portfolio theory is that an investor can reduce risks by investing in a portfolio of assets (stocks or bonds) rather than by gambling on a single asset (Koellner and Schmitz 2006). Like agricultural production, wildlife conservation is characterised by uncertainty, but the sources of risk in wildlife conservation are not the same as those to which agricultural production is subjected and the impacts on revenues may differ substantially among the two sources of income (Muchapondwa and Sterner Forthcoming).

Study objectives and approach

In this thesis I use a bioeconomic approach in analyzing the implications of non-equilibrium dynamics for the efficient and sustainable management of wildlife and livestock in dryland grazing systems. But first, a brief description of the socio-economic system of the southeastern lowveld. The people in the southeastern lowveld of Zimbabwe are culturally described as Shangaan. The Shangaan culture is recognizable since about the late 18th and early 19th century. Livelihoods of the Shangaan people depended on riverbank cultivation, fishing and hunting (Andersson and Cumming 2013). However, fishing was then restricted and cultivation close to rivers prohibited due to colonial rule (Bannerman 1978). In the southeastern lowveld of Zimbabwe, 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 with the enactment of the Land Apportionment Act of 1930 (Giller *et al.* 2013). Even post-independence there was a continued movement of people, particularly the Shona, escaping the crowded communal lands in search of more extensive grazing and arable lands. Small grains such as sorghum and millet still are the major crops grown, as are groundnuts, pumpkins, watermelons and sweet potatoes. Maize is increasing in importance since its introduction by Shona and Ndebele settlers in the 1950s (Wolmer 2007). A large range of plants, fruits, nuts, roots and tubers were and are gathered for food, medicine, construction, firewood and beer brewing. These become particularly important during drought years. The most important include marula (*Sclerocarya birrea*), ilala palm (*Hyphaene pertesiana*) and baobab (*Adansonia digitata*). Extensive livestock husbandry is practiced in this area and the indigenous breeds are most suitable for the production of meat off the rich natural grazing.

The spirits of dead ancestors are part and parcel of land and life in the lowveld just as elsewhere in Zimbabwe and much of sub-Saharan Africa. The ancestral spirits are respected when people start cultivating their fields, when fruits or crops ripe and are ready to be eaten or when a family is about to migrate to a new home. They enjoy drinking beer and beating drums and dancing for their ancestors. However, missionaries, returning labour migrants and urban evangelists have contributed to the lowveld population's long exposure to Christianity. Churches have challenged the power of ancestral spirits but for many Christians this has resulted only in a change in the relative authority of spirit mediums rather than a total loss of belief (Wolmer 2007). Like any other people on Earth they want to be happy, healthy, wealthy and wise, and they are thus concerned about anything that is putting health, happiness and wealth at risk. Erratic rainfall is the greatest threat to the people of the lowveld's welfare. Frequent droughts mean that as many as one year in four can be a year of hunger and so people depend mainly on livestock and wildlife and both depend on vegetation and its productivity. From an ecological point of view, the role of abiotic and biotic factors in determining plant species composition is poorly quantified. This leads to my first research question:

1. What are the important biotic and abiotic factors explaining vegetation variables such as grass and woody species composition, production and basal cover?

The expectation would be (1) that the vegetation composition in areas with high densities of large herbivores contrasts most strongly with areas where herbivore density is low; (2) that high densities of livestock and other large grazing herbivores foremost influences the herbaceous composition, rather than the woody plant composition; and (3) that in livestock-rich areas outside the conservation zone, soil contrasts partially explain plant community contrasts among sampled areas.

Further, I analyse herbivore dynamics over time and relate these to environmental factors. The expectation would be that cattle dynamics would show a boom and bust pattern which is also expected to be explained by non-cyclic environmental factors, namely rainfall. This leads to my following research question:

2. Is there evidence of non-equilibrium dynamics, and what are the impacts of such dynamics on cattle herd dynamics?

From an economic point of view, I address the question of how risks of fluctuations in household income can be managed in order to improve human welfare. I expect that in systems exhibiting non-equilibrium dynamics people can improve their welfare by exploiting a combination of wildlife and agricultural activities (livestock and cropping) in their attempts to reduce fluctuations in their annual welfare. This would be possible if the risks in wildlife and agro-pastoral systems are sufficiently different. Exploiting different sources of income requires efficient allocation of resources. The most prominent resource is land and land varies spatially in quality, and ecological resources require spatial connectivity. Therefore the spatial dimension is important in this allocation. Further, though I will not provide empirical evidence for this, I would predict that local people would value their wildlife and would take measures leading to conservation of biodiversity. This leads to the following research question:

3. To what extent can wildlife income buffer rural households' incomes against fluctuations in rainfall?

Finally I study how different scenarios for land allocation will change income levels, based on the following research question:

4. From a theoretical and practical perspective, can wildlife income have an insurance value to local people?

Study area

The study was conducted in southeastern lowveld of Zimbabwe (Figure 1.2), which includes five wards[1] (3 077 km^2 in total) in Chiredzi district namely Chikombedzi (ward 11: 358 km^2), Gonakudzingwa (ward 12: 306 km^2), Pahlela/Makanani (ward 13:

[1] A ward is a sub-district administrative unit comprising an average of six villages, though settlement in these is not consolidated.

648 km²), Sengwe (ward 14: 813 km²) and Malipati (ward 15: 953 km²). The area lies close to Gonarezhou National Park (ward 22). Wards 11, 13, 14 and 15 are under communal tenure while ward 12 is a small scale commercial area divided into 43 farms, each with a mean size of 700ha. The southeastern lowveld is characterized by low rainfall, shallow soils with low agricultural potential and high temperatures. Annual rainfall ranges between 300 to 600mm. Effective rainfall occurs from October to April, followed by a long dry season.

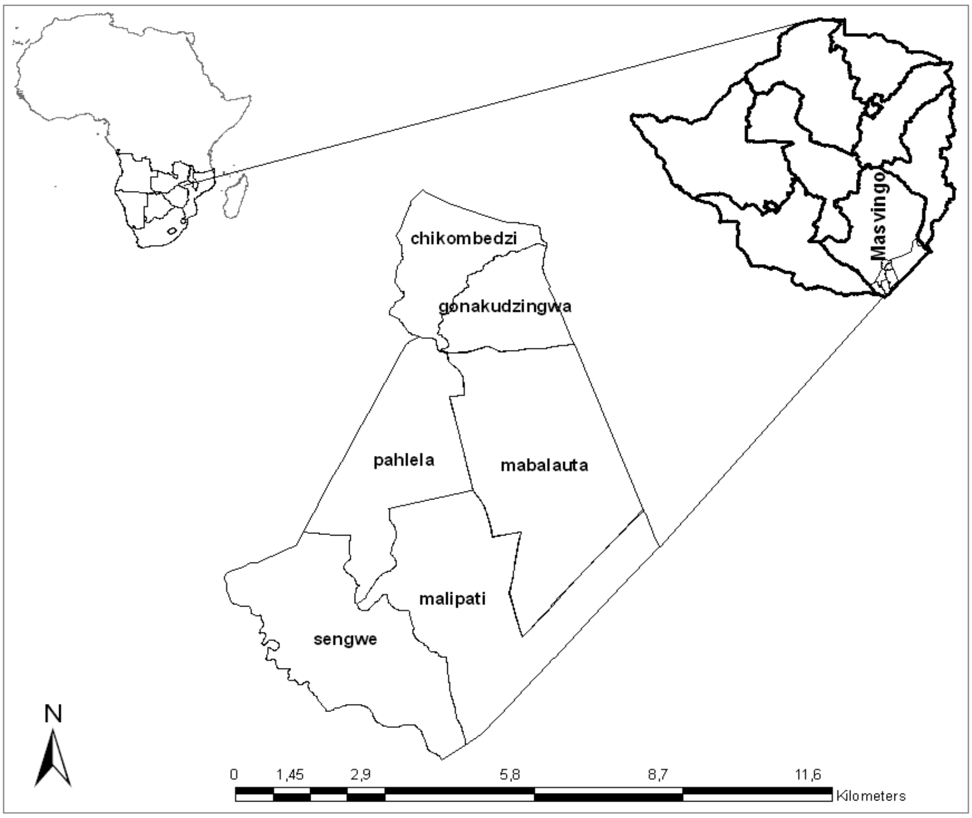


Figure 1.2. A map indicating the location of the study area in the southeastern part of Zimbabwe on the edges of Gonarezhou National Park in Masvingo Province. The insert shows the layout of a portion of Gonarezhou National Park (Mabaluta), small scale commercial area (Gonakudzingwa) and the four wards (Chikombedzi, Pahlela, Malipati and Sengwe) of the Sengwe communal lands.

Thesis outline

This thesis is organized in two sections. The first section (Chapters 2 and 3) analyses the behaviour of the system through the abiotic and biotic relationships in south eastern lowveld of Zimbabwe. Such an analysis will help in establishing the strength of the coupling between herbivores and their resource. The section contributes to the current debate by rangeland ecologists concerning the dynamics of rangelands especially in semi-arid areas. In particular, I quantify the effects of rainfall and soil characteristics vs. land use and herbivore densities on the rangeland dynamics. In chapter 2, I use primary data collected through biotic and abiotic sampling to investigate the relative effects of land use, rainfall, soil characteristics and herbivore densities on grass and woody species composition, above ground herbaceous biomass production and basal cover. Chapter 3 uses long term data on cattle, rainfall and vegetation (as indicated by NDVI), to study the dynamics in cattle numbers in the southeastern lowveld of Zimbabwe. In this chapter I highlight the important factors that explain the dynamics in cattle numbers over time and how observations depend on the spatial and temporal scale of analysis. Hence in this chapter I refer the debate on what scale scientists can make conclusions about the system and also on what level do scientists have an understanding of the system?

After investigating the behaviour of the system (Chapters 2 and 3), and recognizing that drought is an ever recurring part of rangeland dynamics in arid and semi-arid regions. Furthermore, drought was found to have enormous impacts on extensive livestock production by reducing outputs as well as by generating short-term capital destruction at the farm for example when animals die. Since rural people rely on animal husbandry for their livelihood, productive losses become a social problem. Few efforts however, have been made to understand the adaptive capacity of households to cope with drought through changing from on-farm to off-farm approaches (Easdale and Rosso 2010). The second section (Chapters 4 and 5) focuses on ways for people residing in these areas to minimize risk due to complex dynamics of these ecosystems. This is premised on the fact that in sub-Saharan rangelands, high levels of biodiversity are commonly juxtaposed with chronic poverty and underdevelopment leading to frequent conflicts over natural resources (Homewood 2004). These conflicts centre on contested access to wild land, resulting in clashes between wildlife conservation interests and rural livelihoods. Further economic literature suggests that forbidding the use of wildlife products can simply diminish economic value, that is make the resource less valuable or even valueless from an economic point of view (Pearce 1994). Therefore, this section analyses the potential of wildlife income in reducing household income variation and how the two can be incorporated in the landscape (spatial planning) as the two need to be separated to prevent disease transmission.

In chapter 4, I explore approaches that incorporate wildlife. I argue that by having access to crown land and to economic exploitation of wildlife and or other 'wild' resources, people can improve their livelihoods. The main question is "under which conditions can access to crown land enhance livelihoods without deteriorating wildlife?" Hence, under non-equilibrium grazing systems, the objectives would be to reduce costs and optimize income. I use primary data collected through household surveys and wildlife revenues remitted from a local communal areas management programme for indigenous resources (CAMPFIRE) to understand the different costs and benefits related to wildlife and livestock. Furthermore, I assess the degree to which income from the agro-pastoral system fluctuates with variations in annual rainfall, and to what extent income from wildlife would reduce these fluctuations in household income. I borrow from Markowitz's (1952, 1959) analysis related to financial securities, a theoretical framework based on portfolio theory. I use this framework to investigate whether risk management through diversification into wildlife conservation could help farmers deal with risks related to drought.

In chapter 5, I use a land use modelling approach to manage the SEL of Zimbabwe when a trade-off exists between wildlife conservation and economic development. What levels of wildlife, livestock and people are sustainably attainable within the natural boundaries of southeastern lowveld of Zimbabwe? What are the possible income levels from each land use and how constant are they against annual rainfall fluctuations? Given that wildlife conservation needs to be separate from agro-pastoral system, how can the two co-exist in southeastern lowveld of Zimbabwe? In Chapter 5 I contribute to an understanding of how people can balance conservation against the developmental objective in systems showing non-equilibrium dynamics. Similar models have been formulated for East African (Schulz and Skonhoft 1996; Skonhoft 2007) and Zimbabwean (Muchapondwa 2003) cases. I extend the literature by incorporating issues of spatial land allocation and uncertainty in annual rainfall.

In Chapter 6, I have been grasping with the problem "how can people live in a system that is showing non-equilibrium dynamics?" I come to the conclusion that Malthus is right. It is only through industrialisation that people can meet their aspirations. That way, people would reduce reliance on primary production and subsequently lower dependence by local people on natural resources, thereby increasing chances of sustainable management of natural resources. I also provide suggestions for further research.

Chapter 2

The effect of land use on vegetation composition in southeastern lowveld of Zimbabwe

Xavier Poshiwa; Ignas M. A. Heitkönig; Craig Morris; Kevin P. Kirkman; Ekko C. van Ierland and Herbert H.T. Prins.

Abstract

Arid and semi-arid systems support diverse and non-equilibrium dynamic ecosystems which sustain land use activities like nature conservation, commercial and communal farming, and animal husbandry. These may gradually alter vegetation in terms of species composition and basal cover, increase the relative availability of annual grass species, and decrease the relative availability of perennial grass species, but their impact is poorly quantified. The role of abiotic and biotic factors in determining plant species composition was studied and three expectations were tested. The predictions were (1) that the vegetation composition in areas with high densities of large herbivores contrasts most strongly with areas where herbivore density is low; (2) that high densities of livestock and other large grazing herbivores would foremost influence the herbaceous composition, rather than the woody plant composition; and (3) that in livestock-rich areas outside the conservation zone, soil contrasts would partially explain plant community contrasts. Species composition of both woody and herbaceous vegetation layers were sampled in a small-scale commercial, a conservation (park) and several communal areas. Grass standing crop was measured and environmental variables on soil fertility, rainfall, and herbivore density were quantified in the southeastern lowveld of Zimbabwe. Peak standing biomass measured in exclosures was similar across all areas at about 1500 kg DM ha^{-1}, but grazing intensity was significantly larger in the communal areas only. Co-correspondence analysis (CoCA) revealed that the woody and the herbaceous plant communities did not relate strongly to one another (cross correlation between all CoCA axes: $r < 0.77$; permutation tests for all axes $P = 0.18$). As predicted vegetation composition contrasted most strongly between the conservation area and neighbouring rangeland (Multiresponse permutation procedures, $P_{Bonferroni} < 0.0033$). Rangelands outside the conservation area were characterized by significantly higher soil fertility parameters than the conservation areas, by 3.6 to 28 times higher grazing herbivore pressure, and by higher rainfall. However, in contrast to our prediction, not herbivore density, but abiotic variables explained the strongest contrasts among sampling sites, for both the woody and the grass communities. Constrained analysis on the grass community revealed that permutation tests on all axes were significant at $P = 0.002$, with pH and rainfall together explaining 5.7% of the adjusted variation (False Discovery Rate, $P_{adj} = 0.004$ and 0.030, respectively). Constrained analysis on the woody plants revealed that permutation tests on all axes were significant at $P = 0.002$, with rainfall, NDVI and phosphate (P_2O_5) together explaining 5.6% of the adjusted variation (False Discovery Rate, $P_{adj} = 0.004$, 0.01, and 0.08, respectively). Lastly, the third prediction on soil contrasts in livestock-rich areas was not supported, since neither the abiotic nor the biotic factors measured in this study clearly explained plant community contrasts in human used areas. It was concluded that differences in plant species composition across the study area is mainly explained by abiotic factors like rainfall and soil, and only to a smaller extent by grazing intensity. Because explained

variation in this study was low, it can be suggested that other factors such as periodic droughts, rather than local contrasts in abiotic and biotic variables, determine vegetation communities in this semi-arid ecosystem.

Key words: Savanna, land use, nature conservation, communal, small scale commercial, rainfall, stocking density.

Introduction

Savannas are ecosystems characterized by the co-existence of woody species (trees and shrubs) and grasses (Scholes and Archer 1997). They occupy a fifth of the earth's land surface and support a large proportion of the world's human population and most of its rangeland, livestock and wild herbivore biomass (Scholes and Archer 1997). The composition of savanna rangelands is driven by a combination of bottom-up environmental factors, particularly plant available moisture and nutrients, and top-down "disturbance" processes such as fire and grazing (Furley 2004; Bond and Keeley 2005; Scott *et al*. 2009). Work by Higgins *et al.* (2002), Wessels *et al.* (2011) and Fisher *et al.* (2012) showed a clear impact of wood extraction on woody vegetation structure in communally used lowveld savannas of South Africa.

Considerable debate still surrounds the relative importance of each of these factors as determinants of the grass-tree ratio (Sankaran *et al.* 2008; Scott *et al.* 2009). While early studies emphasized the importance of edaphic and environmental controls on plant species distribution and spatial variation in vegetation composition, recent studies have documented the importance of both natural and anthropogenic disturbances in this respect (Motzkin *et al*. 2002; Gillson and Willis 2004; Aguiar and Ferreira 2005; Urbieta *et al*. 2008; Cochrane and Barber 2009). The wide range of environmental, faunal and anthropogenic conditions in savannas have frustrated attempts to reach consensus on the relative importance of these factors and how the vegetation structure of savannas are controlled (Bond 2008). At a regional scale vegetation structure (i.e., grass/tree ratio) and species composition in savannas is largely determined by precipitation (Sankaran *et al*. 2004), whereas at the nested landscape-scale vegetation structure and composition is more prominently determined by geologic substrate, topography, fire and herbivory (Witkowski and O'Connor 1996; Bond and Keeley 2005; Sankaran *et al.* 2008; Asner *et al.* 2009). The multiple factors involved interact in a complex manner, to which humans also interact, using land for pasture, agriculture, extracting fuel and timber, and causing alterations in tree-grass ratios (Bucini and Hanan 2007). Land use intensity and associated vegetation disturbance thus contribute to the composition and structure of an ecosystem in any given place (Foster *et al*. 2003), but it is not clear whether environmental or grazing factors dominate the effect on the vegetation structure at landscape level (Tessema *et al.* 2011).

Belskey *et al.* (1993) and Abule *et al.* (2007) demonstrated that intensive grazing would alter the herbaceous species composition, both outside and under tree canopies, towards less palatable and often annual grass species. Herbivores may affect the regeneration of woody species (Prins and Van der Jeugd 1993; O'Kane *et al.* 2012) and this may lead to converging vegetation composition under moderate and heavy grazing over time (Allred *et al*. 2012). On shorter time scales, the woody vegetation structure is most strongly affected by cutting of firewood and woody plant removal for agricultural fields (Fisher *et al.* 2012).

Rangeland state variables such as plant species diversity, abundance, composition and standing biomass can be used as indicators of rangeland degradation. Rangeland degradation is defined in various ways, such as a decrease in plant diversity, plant height, vegetation cover and plant productivity (Han *et al.* 2008; Ho and Azadi, 2010), or is characterized by dramatic declines in perennial grass cover and substantial increases in woody shrub density attributed to intense grazing by domestic livestock (Belsky, 1995; Valone *et al.* 2001; Vetter, 2005). Further characteristics are a reduction in palatable plant species, an increase in undesirable and unpalatable plants, and depletion of soil quality and nutrients (Mekuria *et al*. 2007). Extreme degradation takes decades of recovery (Searle *et al*. 2009).

A clear understanding of the 'state of health' of rangelands has important implications for natural resource management and conservation, and also for maintaining and enhancing the short as well as long-term socio-economic benefits. The effects of land use on woody and herbaceous vegetation composition were studied. The following predictions were tested (1) that the vegetation composition in areas with high densities of large herbivores would contrast most strongly with areas where herbivore density is low; (2) that high densities of livestock and other large grazing herbivores would foremost influence the herbaceous composition, rather than the woody plant composition; and (3) that in livestock-rich areas outside the conservation area, soil contrasts would partially explain plant community contrasts.

Materials and Methods

Study Area

The study was conducted in south eastern lowveld of Zimbabwe (Figure 2.1), which falls within the Greater Limpopo Transfrontier Park and Conservation area (GLTP/CA). The study was conducted within five wards in Chiredzi district. A ward is an administrative unit at sub-district level, comprising an average of six villages, though settlement in these is not consolidated. In this area, livestock is partly kept on communal natural rangelands, a common means of subsistence for rural families (Sellen 2003), and veterinary services and administration are conducted at this level. Key feature in these communal systems is that rangeland used for grazing is held and administered as common property resource. Four wards were under communal tenure: Chikombedzi (358 km^2, abbreviated as 1C), Pahlela/Makanani (648 km^2, 1P), Sengwe (813 km^2, 1S) and Malipati (953 km^2, 1M). One ward was a small scale commercial farming area, Gonakudzingwa (306 km^2, abbreviated as 2comG). These wards lie close to the Mabalauta section of Gonarezhou National Park (1,060 km^2, assigned to nature conservation, abbreviated as 3park). Most rainfall occurs from October to April, and mean annual rainfall is approximately 511 mm, with a coefficient of variation (CV) for inter-annual rainfall of 51% (this thesis Chapter 3). The southeastern lowveld of Zimbabwe is characterized by low rainfall, shallow soils with low agricultural potential and high temperatures. According to a study by (Zisadza 2008) on land cover changes in the same study area, the percentage area under cultivation and settlement in 2007 was 26%.

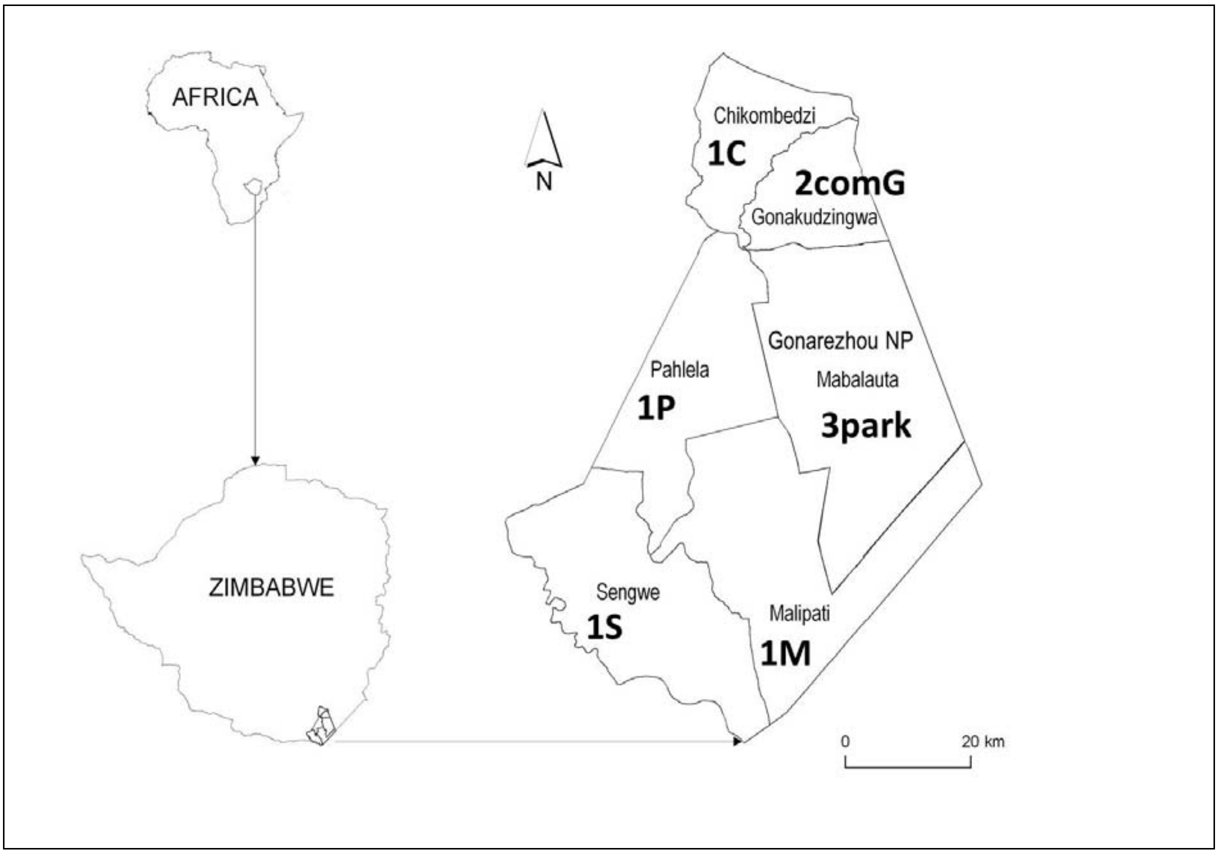


Figure 2.1: Southeastern lowveld study area in Masvingo Province of Zimbabwe, with names of land use areas (wards), and their codes as used in this chapter. 1S, 1P, 1C, and 1M are communal, livestock-rich areas; 2comG is a small scale commercial livestock-rich area, and 3park is a conservation area with game species.

Vegetation sampling

A vegetation survey was conducted at the end of March/ early April 2008. A stratified random sampling procedure was used to generate points for sampling in Arcview 3.2 GIS. The study area was stratified into different land cover types using the 1996 land cover classification map by the forestry commisison of Zimbabwe. Fifty sampling plots in each of the six wards were generated, 300 in total, ensuring that the plots covered all major land cover classes (including woodland, wooded grassland, and grassland). The size of the woody layer sampling plots was 30 x 30 m. In these 30 x 30 m plots, all woody (tree and shrub) species were identified and individuals counted to determine the abundance of the different species. For the herbaceous layer, five 1 m x 1 m quadrats within these 30 x 30 m plots were sampled to visually estimate the percentage grass species cover. Percentage cover was then averaged for the five quadrats within each 30 x 30 m plot.

Herbivore Densities and Rainfall

In order to help explain variation in vegetation composition across the study area, grazing large herbivore densities were converted to metabolic body weights ($W^{0.75}$ in kg) (Prins 1992). Livestock population size in 2008 and composition (cattle age class and respective weight) for each ward were obtained from the Chiredzi District veterinary department, from which average metabolic weights per age class were calculated. Dip tank data was used as a reliable source of data since it is compulsory and also enforced in Zimbabwe for farmers to have their cattle dipped as part of a highly controlled cattle husbandry system nation-wide. Adult body mass was 385 kg. This amounted to determining the metabolic body weight (MW) of an average cow as:

$$MW = (0.835 * \text{adult body mass})^{0.75} \qquad [1]$$

Stocking densities under the communal set up range from 7.3 to 20.0 kg MW/ha. Unlike communal lands, small scale commercial farms (2comG) operate under a single manager, whose primary objective is to optimize animal production in relation to input costs using rotational grazing, and stocking densities are considerably lower (2.6 kg MW/ha). The park supports a wide diversity of game animals in a continuous grazing system with minimal intrusion by external forces. Wildlife densities were calculated from aerial census figures of herbivores in the Mabalauta section of the Gonarezhou National Park done jointly by the Department of National Parks and Wildlife Management and WWF-SARPO in October 2007 (Dunham 2007). The longitudes and latitudes of transects used were converted to UTM coordinates in ILWIS 3.3. Game species' adult body weight estimates were derived from Skinner and Chimimba (2006). Since information on the population age structure of game species was not available, a similar age structure as in cattle was assumed, and the metabolic body weight (MW) of an average individual was calculated as above, for each game species separately. Only grazers were included here, i.e., buffalo (*Syncerus caffer*, n = 178), zebra (*Equus quagga burchellii*, n = 59), wildebeest (*Connochaetes taurinus*, n = 1), and cattle (*Bos Taurus*, n = 33) as well as 50% of the metabolic biomass of mixed feeders, i.e., elephant (*Loxodonta africana*, n = 229), impala

(*Aepyceros melampus*, n = 98) and warthog (*Phacochoerus africanus*, n = 7), amounting to a grazer stocking density in the park of 0.7 kg MW/ha.

Rainfall data was derived from the National Oceanic and Aeronautics Administration a Very High Resolution Spectroradiometer (NOAA AVHRR) 8 km resolution images of October 2007 through March 2008. The processing of NOAA images involved using Erdas Imagine for BIL format of images. Layer stacking was done to form one image, followed by projecting the image from Albers Equal area to UTM for Zone 36. Overlaying of points on the image was done allowing spectral profiles to be taken and exported to Excel 2003 for analysis.

Biomass production and soil sampling

Above ground herbaceous biomass standing crop of the study area was measured during the wet season of 2008/2009. Random points were generated in Arcview 3.2 GIS as described in the previous section. The area was stratified according to soil type and six points in the park and six points in each of the four communal areas were randomly selected. Four sampling points were generated for the small-scale commercial area due to its relatively small size. At each site a 5 x 5 m^2 exclosure plot (grazing excluded) and an adjacent 5 x 5 m^2 control plot (grazing allowed) were established at the start of the rainy season in December 2008. Herbaceous sampling to determine species composition and basal cover was done following the methods explained in the previous section and measurements to estimate above ground biomass were done monthly up to the end of the rainy season (January through April). The biomass within each quadrat was clipped using shears and the fresh weight taken ('t Mannetje, 2000). The samples were taken to the laboratory for oven drying at 60 ^{0}C for 72 h to determine the dry matter content. From these samples the herbaceous biomass was calculated as follows:

$$\text{Dry weight (g)}/25\text{m}^2 = \text{g/m}^2 \text{ X } 10 \text{ to get an estimate in kg/ha} \qquad [2]$$

At each site, soil samples from 5 points were combined into a single composite soil sample from a depth of up to 20 cm using an auger, and analysed for soil composition. Soil pH ($CaCl_2$) (Anderson & Ingrams, 1993), mineral nitrogen (initial and after incubation) using the incubation technique (Saunder *et al*. 1957), available phosphorus (P) was extracted by the Resin method (Anderson & Ingrams, 1993) and determined spectroscopically. Atomic emission was used to read potassium (K), while calcium (Ca) and magnesium (Mg) were determined by extracting with neutral normal ammonium acetate and read on the atomic absorption spectrophotometer (AAS) (Summer & Miller 1996) at Chemistry and soil research laboratories in Harare.

Statistical Analyses

Both log-transformed data on woody plant density, and untransformed data on grass cover were subjected to co-correspondence analysis (CoCA), correspondence analysis (CA), and canonical correspondence analysis (CCA) in Canoco5 (Ter Braak and Smilauer 2012). CoCA assesses to what extent the woody plant community aligns with the grass community. CA summarizes differences and similarities among the sites analysed, in terms of their plant species composition. Since we were interested in testing hypotheses on plant species composition in relation to animal impact, CA was applied to sites where both herbaceous and woody plants were available (n = 128), and the analysis done separately for grasses and for woody plants, the resulting ordination graphs were inspected after the analysis for patterns to reveal which land use types overlapped most in the ordination results.

Subsequently, multiresponse permutation procedures (MRPP) in PC-ORD (version 4.25; McCune and Mefford 1999) were used to test for statistical differences in grass and in woody species composition separately between land use areas, to help interpret the CA output. Euclidian distances were used in calculating fifteen pairwise comparisons between land use areas, with test statistic T to compare the observed intragroup average distances with the average distances that would have resulted from all the other possible combinations of the data, and applied the Bonferroni correction (alpha = 0.05/15) to account for increased Type I error.

A Canonical Correspondence Analysis on log (x+1) frequency data for woody species, and cover for grass species was carried out, with rare species down weighted, to quantify to what extent environmental variables rainfall, stocking density, and each of the soil parameters accounted for variation in species composition. Environmental variables with variance inflation factor greater than 10 were removed to prevent strong collinearity effects (Zuur 2010). In the forward selection process, the Monte

Carlo permutation test was used to estimate the additional marginal effect of each additional environmental variable, corrected for the False Discovery Rate (Canoco5, see Ter Braak and Smilauer 2012).

Analysis of variance (ANOVA) in SPSS (v.15) was used to test for differences in above ground peak herbaceous biomass with land use, followed by Fischer's LSD *post-hoc* test. A Kruskal-Wallis test was used to test for differences in basal cover of the grass layer with land use, and for environmental variables related to land use type, followed by Scheffé multiple comparison *post-hoc* tests.

Results

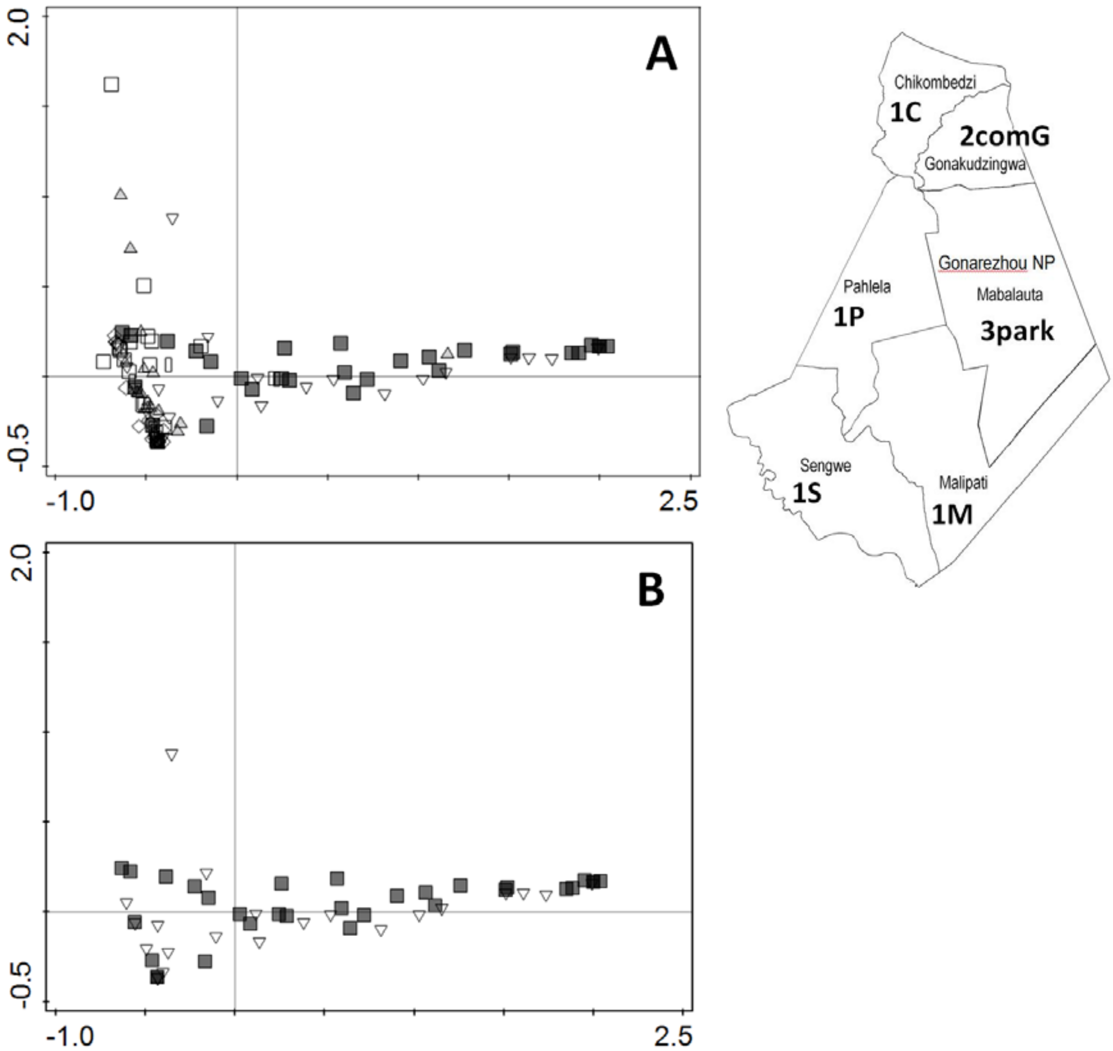
Species composition

Of the 25 recorded grass species, the most abundant and widespread grass species in the study area were *Urochloa mosambicensis*, which accounted for 40% of the sum of all basal herbaceous cover, and *Aristida congesta* (24%). 'Sedges' (15%) were locally abundant in the park and in the neighbouring, western communal area of Pahlela. All other grass species contributed less than 5% to the sum of all basal cover. Of the 66 woody species identified in the study area, *Colophospermum mopane* was found to be common in all areas, making up 53% of the summed woody plant abundances. *Combretum apiculatum* was ranked second (13%), and *Acacia nigrescens*, *Grewia* spp, *Dichrostachys cinerea* and *Androstachys johnsonii* each contributed 4 to 6%, all other species less than 3% each.

Co-correspondence analysis (CoCA) revealed that the woody and the herbaceous plant communities did not relate strongly to one another: cross correlation coefficients between all CoCA axes ranged from 0.70 to 0.77, and permutation tests for all axes showed no significance correspondence (trace = 0.855, P = 0.18).

Grass species community

The correspondence analysis-derived ordination graphs of the sites, labelled by ward, show particular overlap among study sites associated with wards or the park (Fig 2.4A). The grass community of the park (3park in Fig 2.4) overlaps strongly with that of communal area Pahlela (1P; Fig 2.4B), and both respond quite strongly to the gradient underlying the first ordination axis. The community of the small scale commercial farms (2comG) overlaps considerably with that of communal area Sengwe (1S, Fig 2.4C), particularly along the second ordination axis. The remaining communal areas Chikombedzi (1C) and Malipati (1M) also show considerable overlap, but are less responsive to underlying environmental gradients (Fig 2.4D).



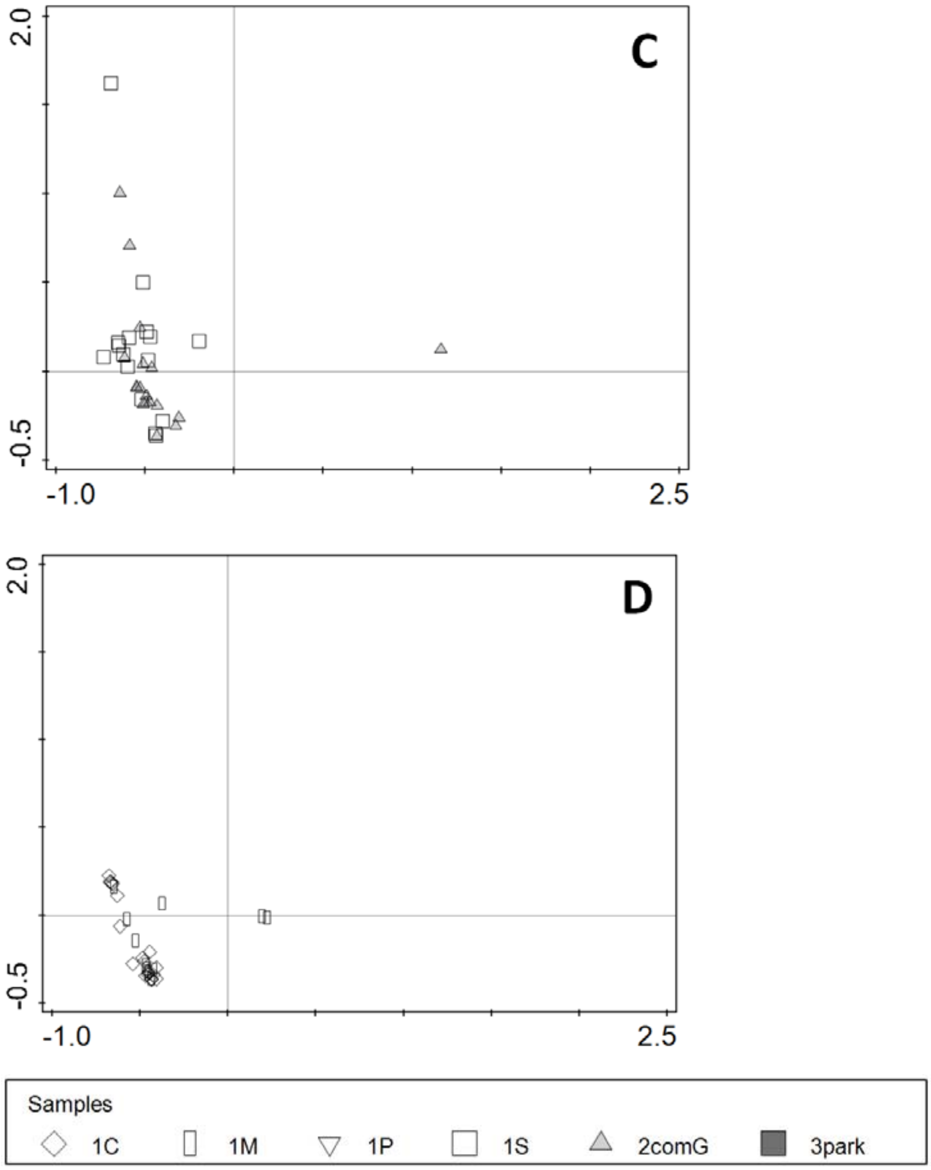


Figure 2.2. Graphs of ordination results following correspondence analysis for grasses in plots (samples) from all wards (**A**); from Pahlela (1P) and the park (3park) which largely overlap along the first ordination axis (**B**); the small scale commercial area Gonakudzingwa (2comG) and communal area Sengwe (1S) show considerable overlap along the second ordination axis (**C**); the communal areas Chikombedzi (1C) and Malapati (1M) show little differentiation in ordination (**D**).

Land use effects on grass species community

Table 2.1 on the pairwise MRPP results provides statistical support to the ordination results of grasses in Figure 2.2A-D, and as reported above. The park grass community differs highly significantly ($P_{Bonferroni} < 0.0001$) from all other land use areas, but not ($P_{Bonferroni} = 0.0322$) from the Pahlela communal area (Fig 2.2B). Both of these areas share many of the same grass species hardly found elsewhere, including *Eragrostis curvula*, *Digitaria milanjiana*, *Perotis patens*, and *Rhynchelytrum repens*. Most of the grass communities outside the park overlap to some extent, but the small scale commercial area Gonakudzingwa overlaps least with the directly neighbouring communal areas of Pahlela ($P_{Bonferroni} < 0.0033$) and Chikombedzi ($P_{Bonferroni} < 0.0033$), and more with the distant communal areas of Sengwe ($P_{Bonferroni} = 0.0092$, Fig. 2.2C) and Malipati ($P_{Bonferroni} = 0.1556$). Lastly, Malipati communal rangelands show more overlap with the distant land use area of Chikombedzi ($P_{Bonferroni} = 0.0415$; Fig 2.2D) than with neighbouring Sengwe ($P_{Bonferroni} = 0.0045$). Hence, directly neighbouring land use areas tend to differ more in grass species composition than distant areas. Inspection of the grass sample data suggest that this can be attributed to subtle differences in uncommon species, rather than in clear contrasts in commonly available species.

Table 2.1. Multiresponse permutation procedures (MRPP) indicating pairwise contrasts in grass communities between the six land use areas. The Bonferroni correction was applied based on 15 comparisons, achieving significance at $P < 0.0033$. Communal areas: 1S = Sengwe, 1C = Chikombedzi, 1M = Malipati, 1P = Pahlela; 2comG = small scale commercial farming area Gonakudzingwa; 3park = the Mabalauta section of Gonarezhou National Park.

	1S	1C	1M	1P	2comG	3park
1S	-	T= -4.375 P= 0.0034	T= -4.314 P= 0.0045	T= -4.871 P= 0.0013	T= -3.437 P= 0.0092	T= -9.030 P< 0.0001
1C		-	T= -2.122 P= 0.0415	T= -5.764 P= 0.0003	T= -5.886 P= 0.0005	T= -15.71 P< 0.0001
1M			-	T= -2.727 P= 0.0201	T= -0.885 P= 0.1556	T= -6.981 P< 0.0001
1P				-	T= -4.986 P= 0.0012	T= -2.374 P= 0.0322
2comG					-	T= -9.332 P< 0.0001
3park						-

Woody species community

The correspondence analysis-derived ordination graphs of the sites, labelled by ward, show considerable overlap among study sites (Fig 2.3A). The woody plant community of the park (3park in Fig 2.5) encompasses almost all others. Those from the small scale commercial area Gonakudzingwa (2comG) overlap strongly with those from communal areas Malapati (1M) and Chikombedzi (1C; Fig 2.3B) and with Sengwe (1S; Fig 2.3C). Samples from the park encompass those of the communal area Pahlela (1), and of the small scale commercial area Gonakudzingwa (2comG; Fig 2.3D). The environmental gradients underlying the ordination results appear weak in discriminating among communal and small scale commercial livestock areas.

Land use effects

The extensive overlap in woody plant communities across the study area (Fig 2.3) is supported by the multiresponse permutation procedures, which show only four land use areas differing significantly from one another (Table 2.2). In particular, the woody plant community in the park only differed significantly from the communal area of Chikombedzi ($P_{Bonferroni} = 0.0010$; Table 2.2), but not from any other area ($P_{Bonferroni} > 0.0033$; Table 2.2). The woody plant community in the small scale commercial area only differed significantly from that of communal area Pahlela ($P_{Bonferroni} = 0.0028$; Table 2.2). Among the communal areas, Sengwe differed highly significantly from Chikombedzi ($P_{Bonferroni} < 0.0001$) and from Pahlela ($P_{Bonferroni} = 0.0005$), but not from Malipati ($P_{Bonferroni} = 0.1672$), suggesting considerable overlap in woody species composition between Chikombedzi and Malipati.

The contrast between the park and Chikombedzi is partly attributed to the commonly available *Combretum imberbe*, *C. hereroense*, *Grewia* spp, *Dichrostachys cinerea, Acacia nigrescens* and *A. nilotica* in Chikombedzi (not sampled in the park), whereas *C. molle* was available in the park but was not encountered in Chikombedzi. Sengwe lacked *Combretum imberbe* and *C. hereroense*, whereas Pahlela contained *Pseudolachnostylis maprouneifolia*, *A. polyacantha*, and *C. fragrans*, absent from Chikombedzi. *Sclerocarya birrea* was widely available outside, but not inside the park.

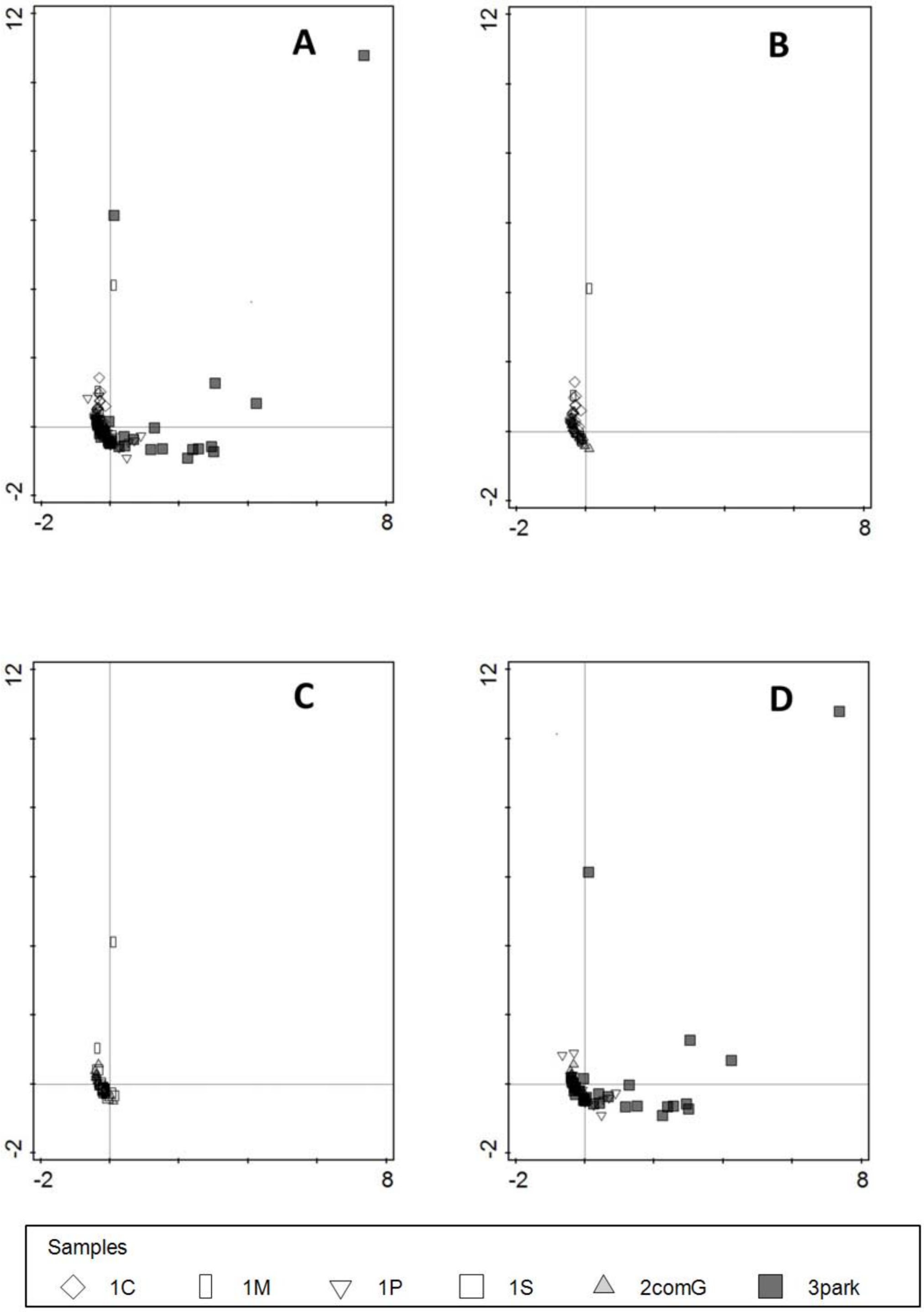


Figure 2.3. Graphs of ordination results following correspondence analysis for woody species in plots (samples) from all wards **(A)**; those from small scale commercial area Gonakudzingwa (2comG) show little differentiation and overlap strongly with those from communal areas Malapati (1M) and Chikombedzi (1C) **(B)** but also with Sengwe (1S) **(C)**; samples from the park (3park) encompass those of the communal area Pahlela (1), and of the small scale commercial area Gonakudzingwa (2comG) **(D).**

Table 2.2. Multiresponse permutation procedures (MRPP) indicating pairwise contrasts in woody plant communities between the six land use areas. The Bonferroni correction was applied based on 15 comparisons, suggesting significance at *P < 0.0033*. Area codes are listed in Table 1.

	1S	1C	1M	1P	2comG	3park
1S	-	T= -8.707	T= -0.851	T= -6.423	T= -0.991	T= -2.387
		P< 0.0001	P= 0.1672	P= 0.0005	P= 0.1414	P= 0.0347
1C		-	T= -1.322	T= -3.769	T= -3.226	T= -6.583
			P= 0.1013	P= 0.0059	P= 0.0120	P= 0.0010
1M			-	T= -0.307	T= -0.568	T= -1.255
				P= 0.2924	P= 0.2060	P= 0.1017
1P				-	T= -4.653	T= -4.579
					P= 0.0028	P= 0.0046
2comG					-	T= -0.138
						P= 0.3139
3park						-

Grass and woody species composition explained by environmental variables

The environmental variables included in the Canonical Correspondence Analysis (CCA) explained a limited (5.7%) but significant part of the variation grass community variation across the sites (pseudo-$F_{499\ \mathrm{permutations}}$ = 1.7, P_{adj} = *0.002*). The CCA forward selection procedure showed that only pH (pseudo-F = 7.0, P_{adj} = *0.004*) and rainfall (pseudo-F = 2.6, P_{adj} = *0.03*) significantly contributed to the explained variation, together for 54.1% of all environmental variables. Grazing density and land use type contributed insignificantly to the remaining variation in this analysis *(P_{adj} >> 0.25)*. This analysis mainly distinguished sampling sites in park (average pH = 4.6 and rain = 599 mm) from those elsewhere (pH = 6.6 - 6.8, and rain = 465 - 478 mm).

Woody plant community variation was also significantly explained by the set of environmental variables (CCA, pseudo-$F_{499\ \mathrm{permutations}}$ =1.7, P_{adj} = *0.002*), and again to a limited extent (5.6%). The CCA forward selection procedure showed that rainfall (pseudo-F = 7.3, $P_{adj} = 0.004$) and NDVI (pseudo-F = 2.3, $P_{adj} = 0.01$) significantly contributed to the explained variation, together for 54.1% of all environmental variables. Phosphate (P_2O_5, pseudo-F = 2.6, $P_{adj} = 0.08$) contributed marginally to explaining the remaining variation. Again, grazing density and other environmental variables contributed insignificantly to the remaining variation ($P_{adj} >> 0.25$). This analysis also mainly distinguished sampling sites in park (average rain = 599 mm, average P_2O_5 = 48 ppm) from those elsewhere (average rain = 465 - 478 mm, average P_2O_5 = 62 - 65 ppm); NDVI differentiated the communal area of Chikombedzi (NDVI = 0.48) form other land use areas (NDVI = 0.51 - 0.53).

Excluding the conservation area, neither the grass community nor the woody species community of the livestock areas was significantly explained by the environmental variables (adjusted explained variation = 3.4% and 4.0% respectively, $P_{adj} > 0.05$), so extraction of an environmental variable by means of a forward selection procedure was not appropriate.

Soil samples showed marked contrasts, mostly between the conservation area and the intensively used livestock areas. Only nitrogen values were higher in the park, but other minerals and pH were significantly lower in the conservation area. The small scale commercial area of Gonakudzingwa showed values intermediate between the communal areas and the conservation area (Table 2.3).

Table 2.3. Median values of soil variables for each land use type, and the level of significant difference (bottom row) as per Kruskal-Wallis test for each variable across the land use types. The Scheffé test, based on means, was applied for the multiple comparisons. SSC = small scale commercial tenure; Comm = communal tenure; Park = Mabalauta section of Gonarezhou National Park. With the exception of soil N, most mineral values are higher outside the park.

LUT	pH	Nppm	lncNppm	P_2O_5	Mg	Ca	K
SSC	6.6^a	29.0^a	41.0^a	62.0^a	3.37^a	26.8^a	0.64^a
Comm	6.8^a	26.0^a	52.0^b	64.5^a	8.98^a	33.3^a	0.75^b
Park	4.6^b	49.0^b	55.0^b	48.0^b	1.68^b	2.2^b	0.31^a
P	*<0.001*	*<0.001*	*=0.001*	*<0.001*	*<0.001*	*<0.001*	*<0.001*

Herbaceous Biomass production and Basal cover

There were no significant *($P > 0.05$)* differences in herbaceous peak standing biomass (about 1500 kg DM ha^{-1}) from plots where grazing was excluded between the three management systems. Instead, communal area peak standing biomass (690 kg DM ha^{-1}) was significantly ($P < 0.05$) lower than peak standing biomass from the small scale commercial area and the park (both similar at *ca*. 1150 kg DM ha^{-1}) (Fig 2.4). Hence, peak standing biomass produced was similar between land uses, but grazing intensity was significantly larger in the communal areas. Basal cover ranged widely in communal areas (0-40%), less so in the park (1-32%) and least (0-20%) in the small scale commercial area. Median basal cover values range from 11% in communal areas to 18% in the park. The Kruskal-Wallis test showed significant differences in basal cover with land use (Chi-square = 8.58, d.f. = 2, $P < 0.05$; Fig 2.5).

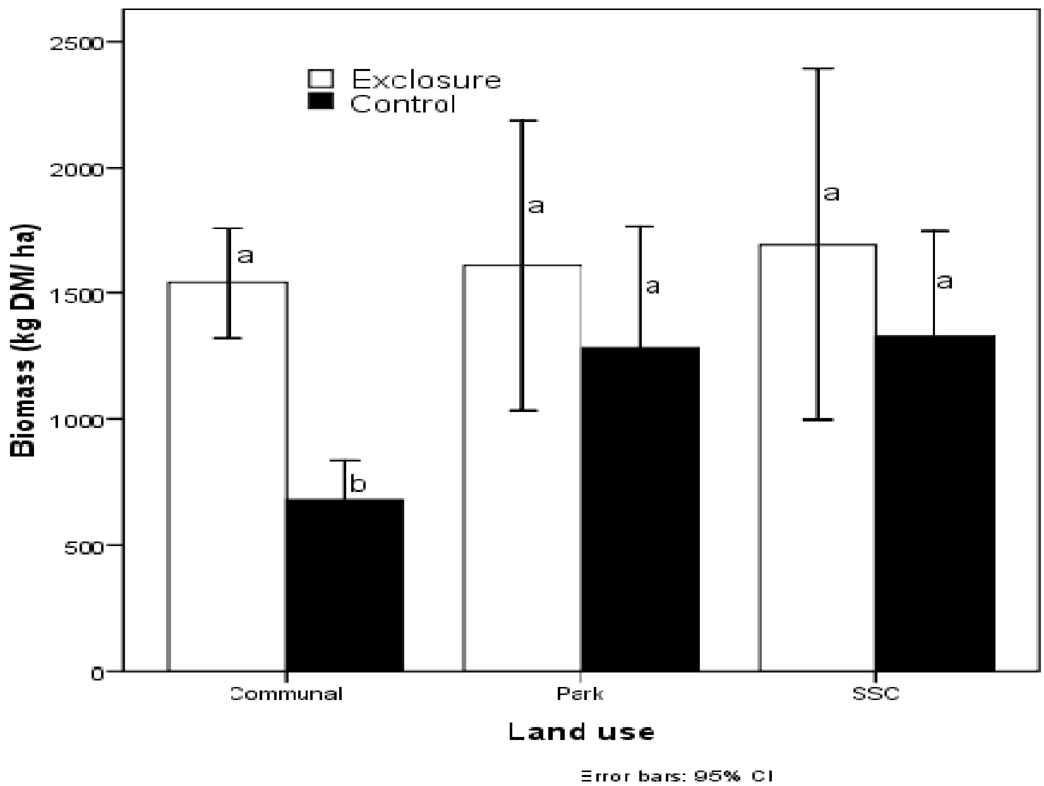


Figure 2.4: Peak above ground grass biomass (kgDM/ha) in communal, park and small-scale commercial areas (SCC) at the end of the 2008-2009 rainy season in exclosures (open bars) and neighbouring control plots (black bars). Error bars denote 95% confidence intervals. Bars with the same letter are not significantly different (LSD test, $P > 0.05$). In communal lands, grass cropping was more intense than elsewhere.

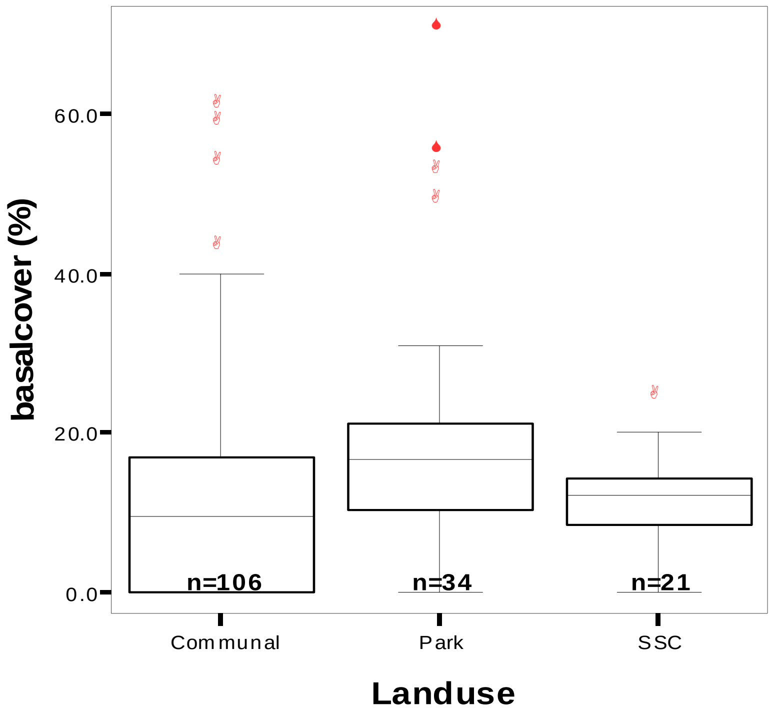


Figure 2.5: Boxplot of the effect of land use on herbaceous basal cover (%) in the south eastern lowveld of Zimbabwe. Boxed areas encompass 50% of the data, the bars cover close to the remaining 50% for each land use type, excluding large values above each of the bars. Communal = communal areas; SSC = small scale commercial area.

Discussion

Grasses

The analyses of the field data show that the grass species composition of the conservation area is different from that of most other land use types (Fig 2.2, Table 2.1), in agreement with the first hypothesis that the vegetation composition in areas with high densities of large herbivores would contrast most strongly with areas where herbivore density is low. The finding that there was considerable overlap between the park and one of the communal areas, Pahlela (ordination: Fig 2.2B, MRPP: Table 2.1), was not expected, but this can be attributed at least in part to the availability of 'sedges' in both areas, which are hardly available elsewhere. What was also not expected was the same low number of species that constituted the majority of grass basal cover, and, with the exception of the sedges, all land use types share these species: *Urochloa mosambicensis* and *Aristida congesta.* This commonality may well have attributed to the similarities in peak standing crop of about 1500 kgDM/ha in grazing-protected exclosures across the land use types. They are both perennial species, drought tolerant, both withstand a high grazing pressure (van Oudtshoorn 1999), and both may also occur in eroded areas. The lower median basal cover in communal rangelands (Fig 2.4) can be attributed to the larger grazing pressure there (Fig 2.3). Even the area with the largest grazing pressure (Chikombedzi, 19.7 kg MW/ha) has *Urochloa mosambicensis* and *Aristida congesta* as the most dominant grass species. The finding that perennial grasses constituted the majority of grass cover, regardless of grazing intensity, was not expected either, and this finding contrasts with findings by Abule *et al*. (2007) and Tessema *et al.* (2011) in Ethiopia, and by Brinkmann *et al*. (2009) in Oman. Abule *et al*. (2007) reported grazing pressures of 1.8 – 14.8 kg MW/ha (recalculated from their data), which is well within the range of values in this study (0.7 – 19.7 kg MW/ha), with comparable average rainfall (550 mm), yet their study area showed mostly annual species under high grazing pressure. Hence, grazing pressure *per sé* clearly does not constitute an explanation for the dominant cover of perennial grasses in our study area, and it would appear that there are no signs of widespread overgrazing in the study area.

Woody plants

In line with hypothesis 2, our results show that the herbaceous composition does not mirror the woody species composition in our study area (CoCA: trace = 0.855, P = 0.18). The woody plant composition was expected to be more uniform across the study area than the herbaceous species composition, and that is supported by our results (ordination of sampling plots: Fig 2.3, MRPP: Table 2.2). Botanically, the park, rich in broad/leaved deciduous woody species and poor in *Acacia* species contrasts most strongly with the communal area of Chikombedzi - which contains *Acacia* species as well as *Dichrostachys cinerea*, an indicator of long-term disturbance. Yet, it is not clear whether these contrasts in woody species are indicative of intensive wood extraction activities outside the park, *sensu* Fisher *et al*. (2012). Indeed, savannas provide a number of ecosystem services to society, and most of the trees have some use value to rural communities, providing resources such as fuel wood, edible fruits, construction timber, medicine or some cultural significance (Higgins *et al.* 1999; Shackleton 2000). Trees being conserved within the communal areas, even in cultivated fields, provide fruit, shade and other goods, such as marula (*Sclerocarya* *birrea*) (Shackleton *et al.* 2002; Shackleton *et al.* 2003). However, soil fertility indices are strikingly different between the conservation area and the livestock rangelands (Table 2.3), with the conservation area being markedly acidic and poor in most minerals, except nitrogen. This makes the soil variables potentially as powerful in explaining woody plant contrasts as human extraction of wood. Many studies have reported the importance of nutrients and rainfall (Fynn and O'Connor 2000; Snyman 2002; Van Der Waal *et al.* 2009) and the effect of consistently high stocking levels (Du Toit and Cumming 1999; Skarpe 2000) on vegetation production and species composition in savanna rangelands. However, the fact that inherent herbaceous biomass production was similar in control plots (where grazing was excluded) suggests that the edaphic and environmental condition for plant growth was similar in the study site.

Environmental variables

Results from the constrained analyses (CCA) clearly show that the environmental variables measured in the study area have a small (< 6%), but significant ($P_{adj} = 0.002$) role to play in explaining both the grass and the woody species composition. The forward selection procedures to identify the variables best explaining grass community ordination, identify pH and rainfall, but not grazing density nor land use type. For woody species, rainfall and NDVI, but again not grazing density and land use type best explain the constrained ordination results. Assuming that land use type (communal, small scale commercial, and conservation) are indicative of decreasing human impact on the vegetation, this means that rainfall and soil override grazing intensity and human impact as drivers of landscape scale variation in botanical composition. In this study area, this can be explained by the different geological formations underlying the livestock area and the conservation area. In a recent study on the vegetation of mopane-dominated Malilangwe Wildlife Reserve in Zimbabwe, a previously commercial ranching area about 100 km NE of our study area, vegetation types were also mainly separated by soil variables (Clegg and O’Connor 2012). In line with our results, Clegg and O’Connor (2012) also show higher soil N values to be associated with lower soil pH, whereas other minerals were associated with higher pH values. We found an even weaker influence of land use type - and hence human activities - in the non-conservation area on woody species composition than on grass species composition. So, if cutting for firewood has played a role in the communal and small scale commercial areas, the effects appear strongly overshadowed by other environmental factors. Thus, in contrast to studies by Higgins *et al.* (2002), Wessels *et al.* (2011) and Fisher *et al*. (2012), who found marked effects of wood harvesting on the vegetation structure, it can be concluded that communal management in southeastern lowveld of Zimbabwe may have had some, but certainly not an overriding effect on the composition of woody plant communities. These findings clearly mean that we have to reject our second hypothesis; not grazer density but abiotic factors like rainfall and soil fertility indices foremost influence herbaceous and woody composition. To further support findings in this study, studies done in the Kruger and Limpopo National Parks (Gillson and Ekblom 2009; Ekblom and Gillson 2010a and Ekblom and Gillson 2010b) found very little variation in woody cover and in the herbaceous populations, indicating a relatively stable grass dominated system. These studies concluded that, despite high herbivore densities and intensive agriculture, there was no evidence of deforestation and changes in local vegetation. It was suggested that it is primarily the abundance of megaherbivores that may lead to changes in vegetation and changes in riverine forests were primarily influenced by climate.

Outside the conservation area, among the livestock areas only, the set of environmental variables did not produce significant canonical axes, hence none of the environmental variables was instrumental in explaining most of the variation in herbaceous or woody community composition, despite the 7.6 times higher herbivore density in communal Chikombedzi compared to small scale commercial Gonakudzingwa. This result means that we have to reject our third hypothesis too, so neither the soil contrasts, nor the biotic factors measured in this study clearly explain plant community contrasts within livestock-rich areas. It is thus unlikely that depletion of soil quality and nutrients takes place in this system (Mekuria *et al.* 2007).

Our multivariate analyses have pointed at a limited number of abiotic variables to explain community contrasts, but in all cases the amount of variation to be explained was below 6%. Although explained variation in ecological data sets is often rather low, typically < 15%, the values obtained in this study are on the low end of the range. This in itself is not worrying, but it gives room to the suggestion that the analysis may have missed important variables, or that the ecosystem is responding dynamically to changes not easily captured in environmental variables (*cf.* Desta and Coppock 2002). In either case, the Southeastern lowveld of Zimbabwe does not show clear signs of system collapse due to overgrazing, in contrast to what is often reported in the literature (Fisher *et al.* 2012; Alldred *et al.* 2012). Zisadza (2008) showed that the area under cultivation has increased from 11% in 1972 to 26% in 2007. These figures coupled with high fluctuations in rainfall makes southeastern lowveld of Zimbabwe rangelands vulnerable because soils of continually overgrazed areas contribute towards the tendency to form crusts which reduce water infiltration (Tefera *et al.* 2008). Fynn & O'Connor (2000) stated that vegetation cover and species composition decline when grazing is heavy and sustained, and improve with increased precipitation and reduced grazing pressure. Periodic droughts followed by recovering rainfall may well mask local contrasts in grazing pressure. Such changes reflect 'ecological resilience' (Berkes *et al.* 1998). The hypothesis by Archer *et al.* (1996), however, states that savanna ecosystems are resilient to disturbance, but can be pushed beyond their resilience limits into new states by intense disturbances. The fact that, in this study, vegetation sampling was conducted in only one year, i.e. the 2008/2009 rainy season, may have masked the partial contribution of various system drivers or variables over time. In line with resilience thinking, rainfall variation over years may drive changes in grazing pressure, a slow variable inducing changes in grass species composition, possibly in interaction with soil variables which may also act as slow variables. Repeated sampling of the botanical composition over time, concurrent with soil and grazing pressure measurements, and subjected to time series analyses, could quantify the role of grazing pressure.

Conclusion

Our study at the landscape scale has demonstrated that abiotic (rainfall and soil) variables play an important role in shaping the herbaceous and the woody plant composition in the semi-arid study area of SE Zimbabwe, despite 30-fold differences in grazing intensity across the land use types. The study has also shown significant differences in production between land uses, which can be attributed to different grazing pressures. Therefore, the study area appears to show signs of equilibrium dynamics, although the dominance of non-equilibrium dynamics is statistically evident.

Chapter 3

Rainfall, primary production and cattle density relationships in southeastern lowveld of Zimbabwe

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Abstract

Debate still exists in rangelands and ecology regarding the sources and types of dynamic behaviour driving rangeland systems. The equilibrium model perspective stresses the importance of biotic feedbacks such as stocking density, whereas the non-equilibrium model perspective stresses stochastic abiotic factors, such as drought, as primary factors determining vegetation and herbivore dynamics. The objective of this study is to investigate the relevance of equilibrium and non-equilibrium theory to the rangeland system of southeastern lowveld of Zimbabwe. We used 17-year cattle density, rainfall, as well as primary production data that we estimated using a satellite based normalized difference vegetation index (NDVI). Firstly, we tested the presence of non-equilibrium by fitting a step function. Secondly, the importance of factors such as rainfall, NDVI, sales, slaughters and migration of cattle, in explaining annual changes in cattle numbers (delta) were investigated using a regression tree model. Finally, non-parametric (Kruskal-Wallis, and Mann-Whitney U) and parametric (T-test) tests were used to investigate the implications of non-equilibrium dynamics on herd dynamics by studying the effect of drought on cattle age and sex categories and their recovery. Results show the existence of thresholds set by rainfall and NDVI in explaining variation in annual changes in the numbers of cattle (delta). Rainfall, NDVI and immigration of cattle were important factors in explaining changes in cattle numbers. The impact of drought was high on juvenile bulls and calves and the same categories had higher recovery rates compared to other age classes. Males and females were not different in their response to drought and the rates they recover. These results support the perspective of southeastern lowveld being a non-equilibrium grazing system. We recommend that management of such systems should put more emphasis on saving the young animals as they are the ones that are vulnerable to such shocks.

Keywords: Rangelands, equilibrium, drought, non-equilibrium dynamics, annual cattle change (delta) and NDVI.

Introduction

In recent years there has been debate concerning the degree of feedback between livestock and vegetation in rangeland systems (Vetter 2005; Bennett and Barrett 2007). This debate arose because of the dissatisfaction with the Clementsian-based procedure (range model) for range condition and trend analysis (Briske *et al.* 2003), that it is an ineffective, over-simplification of vegetation dynamics on many rangelands (Noy-Meir 1973; Laycock 1989; Smith 1989; Westoby *et al.* 1989). The concern is that application of the range model may contribute to mismanagement and degradation of some rangeland ecosystems (Ellis and Swift, 1988; Mentis *et al.* 1989; Walker 1993a; Briske *et al.* 2003). Therefore, state and transition models were specifically developed to overcome the limitations associated with the range model for evaluation of vegetation dynamics in variable rangeland environments (Westoby *et al.* 1989, Rietkerk *et al.* 1996; Briske *et al.* 2003). Consequently, rangelands in semi-arid environments have been described as ecosystems with more than one state and transitions from one state to another, often occurring under influence of disturbances such as grazing or fires (Rietkerk *et al.* 1996; Van Langevelde *et al.* 2003).

Grazing systems, covering about half of the terrestrial surface, tend to be either equilibrial or non-equilibrial in nature, largely depending on the environmental stochasticity (Scoones 1995). The equilibrium model perspective stresses the importance of biotic feedbacks between herbivores and their resource, while the non-equilibrium model perspective stresses stochastic abiotic factors as the primary drivers of vegetation and herbivore dynamics. Furthermore, the range and state-and-transition models are conceptually related to the equilibrium and non-equilibrium paradigms, respectively (Briske *et al.* 2003). In semi-arid and arid tropical systems, environmental stochasticity is rather high, making the systems essentially non-equilibrial in nature, suggesting that feedback between livestock and vegetation is absent or at least severely attenuated for much of the time (Ellis and Swift 1988; Behnke and Scoones 1993; Niamir-Fuller 1998). In southern Africa, range and livestock management however, has been built around the concept of range condition class and the practices of determining carrying capacities and manipulating livestock numbers and grazing seasons to influence range condition (Ellis *et al*. 1993). This management approach is derived from the equilibrium or climax concept of

Clementsian succession (Clements 1916; Stoddart 1975). In equilibrium grazing systems, the physical conditions supporting plant growth are relatively unvarying, consumption by herbivores controls plant biomass and the availability of feed ultimately regulates the growth of the herbivore population. In other words, this concept is based on the view that herbivore numbers are regulated through the availability of forage and that the availability of forage is controlled by animal numbers; in a model this leads to a negative feedback loop which then automatically results in a stable equilibrium between animal and plant populations if the time steps in the model are small enough. Traditionally, the equilibrium to which such a biological model tends is named the 'carrying capacity'. These traditional carrying capacity models are elegant in their simplicity, fit psychological needs for believing in 'balance of nature' and are widely embraced especially in the temperate part of the world. They have been exported from Europe to former colonies, and have been adopted there even though such models have little relevance for biotic and abiotic conditions there. Especially, the erratic and variable rainfall in many pastoral areas of Africa poses a fundamental challenge to this conventional notion of carrying capacity in range management (Ellis and Swift 1988). This realization has caused a shift towards models that embrace non-equilibrium dynamics in ecosystems. Under non-equilibrium dynamics, herbivore populations are controlled by abiotic factors such as precipitation and the frequency of drought, so that their populations in a given year are not closely related to their populations in the previous year, i.e., they tend to be density-independent. Simulations by Boone and Wang (2007) suggest indeed that annual precipitation and its variability cannot be directly linked to dynamics of ungulates within arid and semi-arid African systems, and real data show the same (e.g., Drent & Prins 1987). Derry and Boone (2010) showed that the degree of disequilibrium and coupling between animals and plants may be related to the degree of rainfall variability as measured by the coefficient of variation. However, only a few studies in rangelands have empirically tested the non-equilibrium hypothesis, particularly for south Turkana in Kenya (Ellis and Swift 1988), Borana in semi-arid Ethiopia (Desta and Coppock 2002) and the wetter areas of semi-arid Zimbabwe (Scoones 1993). We extend this work by studying the impacts of non-equilibrium dynamics on cattle numbers, and sex and age categories, and their recovery after droughts. Given that people in semi-arid and arid rangelands rely on livestock, they tend to be food insecure in years of low rainfall. Hence understanding how livestock populations react to stress brought about by non-equilibrium dynamics is crucial to address future challenges related to people's welfare.

In this study, we investigated the relevance of the concepts of equilibrium and non-equilibrium theory to the southeastern lowveld of Zimbabwe system. In southeastern lowveld, extensive livestock production from natural rangeland areas is an important livelihood strategy. We used 17-year datasets on livestock density, rainfall and green biomass (vegetation). Satellite based normalized difference vegetation index (NDVI) data was used to estimate green biomass (vegetation). Because NDVI is correlated with green biomass and therefore provides an estimate of the abundance of vegetation (Tucker 1978; Pettorelli *et al*. 2005). The expectation was that the more vegetation there is in area (as measured by NDVI), the higher the chances of having high forage for livestock. Specifically, the presence of non-equilibrium dynamics was tested by analysing the relationship between cattle population change with total annual rainfall and average NDVI. Secondly, the study investigated the factors that explained the changes in cattle growth rates over time. Finally, we studied how different age classes and sex of cattle are affected by and how they recover from drought. The practical relevance of this study is that it helps identify the kind of intervention needed in managing (semi-) arid rangeland and wildlife systems. More so, faced with the current challenges of climate change, it is essential to understand the dynamics in semi-arid and arid rangelands in order to design strategies for dealing with the impacts of climate change in these drought-sensitive areas.

Materials and methods

Study area

The study was conducted in southeastern lowveld of Zimbabwe (Figure 3.1), which includes five wards (3 078 km^2 in total) in Chiredzi district namely Chikombedzi (ward 11: 358 km^2), Gonakudzingwa (ward 12: 306 km^2), Pahlela/Makanani (ward 13: 648 km^2), Sengwe (ward 14: 813 km^2) and Malipati (ward 15: 953 km^2). A ward is a sub-district administrative unit comprising an average of six villages, though settlement in these is not consolidated. The area lies close to Gonarezhou National Park (ward 22). Wards 11, 13, 14 and 15 are under communal tenure while ward 12 is a small scale commercial area divided into 43 farms, each with a mean size of 7 km^2. The southeastern lowveld is characterized by low rainfall, shallow soils with low agricultural potential and high temperatures. Annual rainfall ranges between 300 to 600 mm. Effective rainfall occurs from October to April, followed by a long dry season.

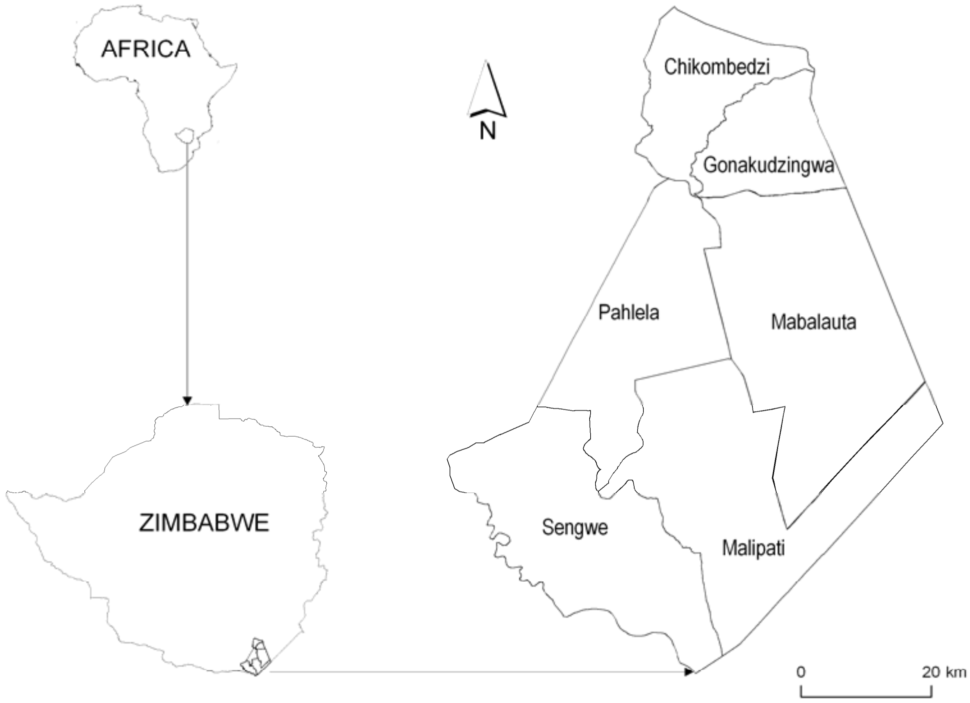


Figure 3.1: South eastern Zimbabwe

Normalized Difference Vegetation Index (NDVI) and Rainfall Data

NDVI is a useful indicator of vegetation cover (Rasmussen 1998; de Fries 2000; Murwira 2003), vegetation condition (Ottichilo *et al*. 2000), and grass greenness (Verlinden and Masogo 1997). NDVI is a useful tool in areas where the green band is not getting saturated like in the southeastern lowveld of Zimbabwe which has a mean annual rainfall of 511 mm. Above the upper threshold of approximately 800 mm (Prince *et al*. 2007) or 1200 mm (Davenport and Nicholson 1993; Nicholson and Farrar 1994), the index saturates and NDVI increases only very slowly with increasing rainfall or it becomes constant (Nicholson and Farrar 1994). The amount of green vegetation was estimated from NDVI derived from the National Oceanic and Aeronautics Administration “A Very High Resolution Spectroradiometer” (NOAA

AVHRR) 8 km resolution images of 1991 to 2007. In this study NDVI (with values ranging from 0 to 255) was used as a stand-in for amount of green vegetation.

Rainfall data was also derived from NOAA satellite images of 1996 to 2007, while those for 1991 to 1995 were taken from Gonarezhou National Park (Mabalauta station) rainfall records due to unavailability of good NOAA images for this period. Rainfall variability analysis and distribution (using number of rain days) was done using 21-year (1988 through 2008) rainfall data from Mabalauta station. Image-derived rainfall data were used to match it with NDVI data at actual location. The processing of NOAA images for both NDVI and rainfall involved using Erdas Imagine (v.8.7) for BIL format of images. Layer stacking was done to form one image, followed by projecting the image from Albers Equal area to UTM for Zone 36. Overlaying of points on the image was done allowing spectral profiles to be taken and exported to Excel 2003 for analysis.

Cattle changes and densities

Cattle data were collected from dip tank livestock records for the period 1991 through 2008, from the District offices of the Department of Veterinary Services in Chiredzi. Dip tank data was used as a reliable source of data since it is compulsory and also enforced in Zimbabwe for farmers to have their cattle dipped as part of a highly controlled cattle husbandry system nation-wide. The cattle data collected included monthly cattle numbers, quarterly cattle numbers by age class and numbers of cattle deaths, birth, slaughters, sold, movement in and out for each year. The location of dip tanks within the study wards was taken using a GPS, and later converted into map locations in GIS. Using the spatial extent of each ward which was generated in GIS, cattle densities (1991 through 2008) within each ward were calculated. The change in cattle populations (δ) was calculated as the log (N_t / N_{t-1}).

Statistical analysis

A step function was fitted to data from individual wards where cattle population change (Δ_{cattle}) was plotted as a function of average NDVI or total annual rainfall and their lags in R v2.11.0 (Team 2010). Hein *et al*. (2011) reported that the relation between rainfall and net primary production may be distorted by a lag in the response of vegetation to changes in rainfall, so that net primary production in a specific year is partly affected by rainfall in previous years. Therefore, lag effects may also be evident between rainfall, NDVI and changes in cattle numbers, hence the reason for use of lags in this study. A step function was used to investigate the existence of a threshold either as explained by NDVI or rainfall. The presence of a threshold was used as evidence for non-equilibrium dynamics. Analysis of variance in R v2.11.0 (Team 2010) was used to test the significance of this threshold. Autocorrelation (acf) function in R v2.11.0 (Team 2010) was used for multiple time series plots. The plots were used to check whether changes in NDVI, rainfall and Δ_{cattle} were correlated, particularly to identify lags. Non-parametric smoothers are excellent at showing the humped relationship between Δ_{cattle} and the explanatory factors and at highlighting the possibility of a threshold (Crawley 2007). A LOESS smoother in R v2.11.0 (Team 2010) was used in highlighting non-linear patterns in the data from individual wards. Data from 1996 to 2007 was used for this part of analysis since satellite derived rainfall was available starting 1996 due to unavailability of good NOAA images before this period.

A regression tree model in R v2.11.0 (Team 2010) was used to investigate important factors that determined changes in cattle numbers (Δ_{cattle}). The explanatory factors included NDVI, rainfall and their lags, slaughters, sales and movements in and out of the area. This part of analysis was done using data from 7 dip tanks in Chikombedzi and 1 dip tank in Gonakudzingwa small scale commercial farms. The data from other dip tanks in other wards were complete in total numbers and had some missing values for herd compositional data. The same data from the eight dip tanks was used to test for the impacts of drought on different age categories (adult bulls, adult cows, adult oxen, juvenile bulls, steers, heifers and calves) and sex (males and females). The impact of drought was tested using the relative decline of each age or sex category calculated by taking the log of numbers at the end of a severe drought (1993) divided by the numbers before the drought (1991). The test for recovery of the different age categories and sex was also calculated based on the log of numbers in the year when the cattle numbers reached their peak (e.g., in 2005 for Gonakudzingwa), divided by the numbers in a year when the animals started to recover from drought (1994). A Non-parametric, Kruskal-Wallis test in PASW Statistic v 17.0. (SPSS 2009) was used for testing for the relative decline and recovery amongst the age class categories, whereas a non-parametric, Mann-Whitney U test in PASW Statistic v 17.0. (SPSS

2009) was used for testing the impact of drought on sex categories of the animals. T-test (with equal variance) in PASW Statistic v 17.0. (SPSS 2009) was used for testing whether recovery was different between males and females because the data followed a normal distribution.

Results

Rainfall

Analysis of a 21-year (1988 to 2008) rainfall dataset measured from Mabalauta shows a rainfall average of 511 mm with a coefficient of variation (CV) for inter-annual rainfall of 51%. The number of years that the area received rainfall which was below average was 13 in the 21 year period (Figure 3.2a). Furthermore, the mean number of rain days was 37 (CV= 0.41). Lowest number of rain days of 16 and 17, were recorded for years 1992 and 2005, respectively. A rain day was defined to have occurred when daily rainfall of at least 0.3 mm was recorded. In contrast 1999 and 2000 had the total annual rainfall distributed over a long period, with 75 and 57 rain days, respectively. Drought is usually defined as a deficit of rainfall in respect to the long-term mean, affecting a large area for one or several seasons or years, that drastically reduces primary production in natural ecosystems and rain fed agriculture (Le Houerou 1996). In this study we objectively define drought as the mean rainfall minus one standard deviation or less (Prins 1996 p. 13). Therefore, given that the mean annual rainfall was 511 mm minus 262 which is the standard deviation, any rainfall year with rainfall below 249 mm is classified as drought. The combined effects of annual rainfall and number of rain days helped in tracking the occurrence and severity of droughts. Severe droughts were experienced in years where both the amount of rainfall and number of rain days were far below 249 mm, for example in 1992 (Figure 2b). Year 2002 was just a bad year and cannot be classified as drought as rainfall above 249 mm was received in that year. Though cattle death exceeded birth in 2002 (Figure 3.3), cattle densities still remained high, meaning that animals still had forage at their disposal. Years 1990, 1994, 1995 and the period 2006 to 2008 were relatively dry years since both the amount and distribution of rainfall was low (Figure 2b). Locals classified these relatively dry years as drought perhaps because of the high densities of cattle found in the area, hence it may point to the fact that farmers in this area may be farming to the limits of what is possible. Some years had below average rainfall which was distributed over a long period in the season, hence the years were better for vegetation for example, 1997, 1998 and 2001.

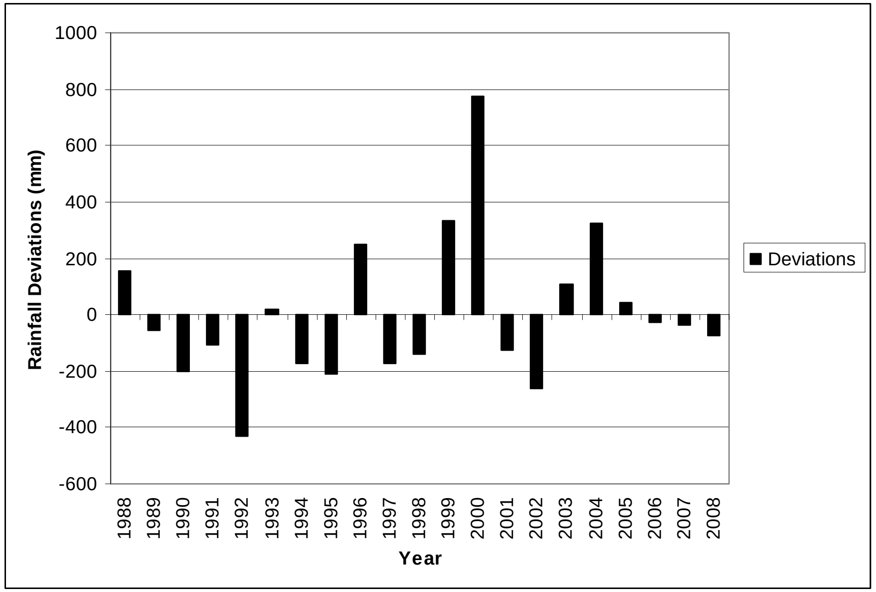


Figure 2a: 21 Year annual rainfall deviations from the mean of 511mm.

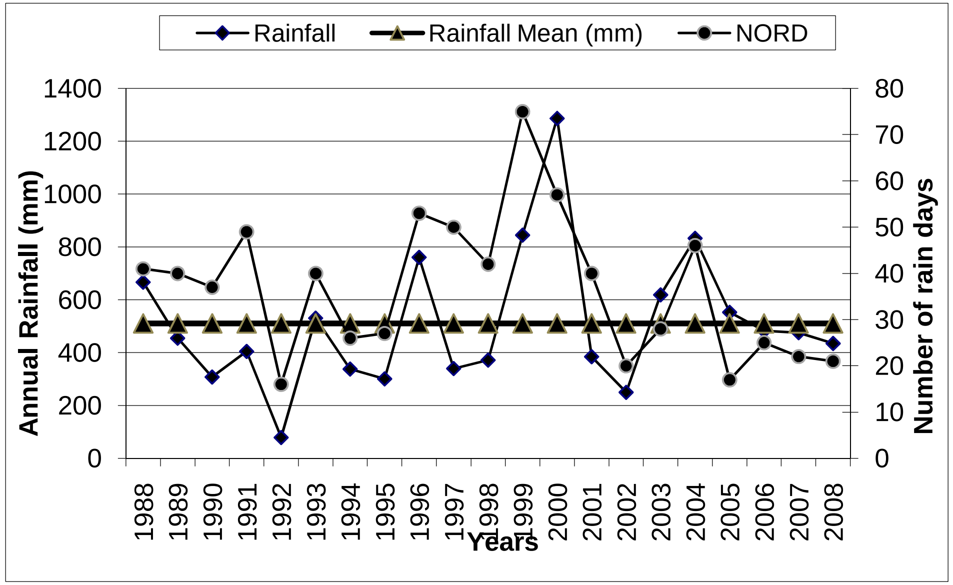


Figure 2b: Fluctuations in total annual rainfall and number of rainfall days (NORD) over a 21 year period using rainfall recorded from Mabalauta section of Gonarezhou National Park.

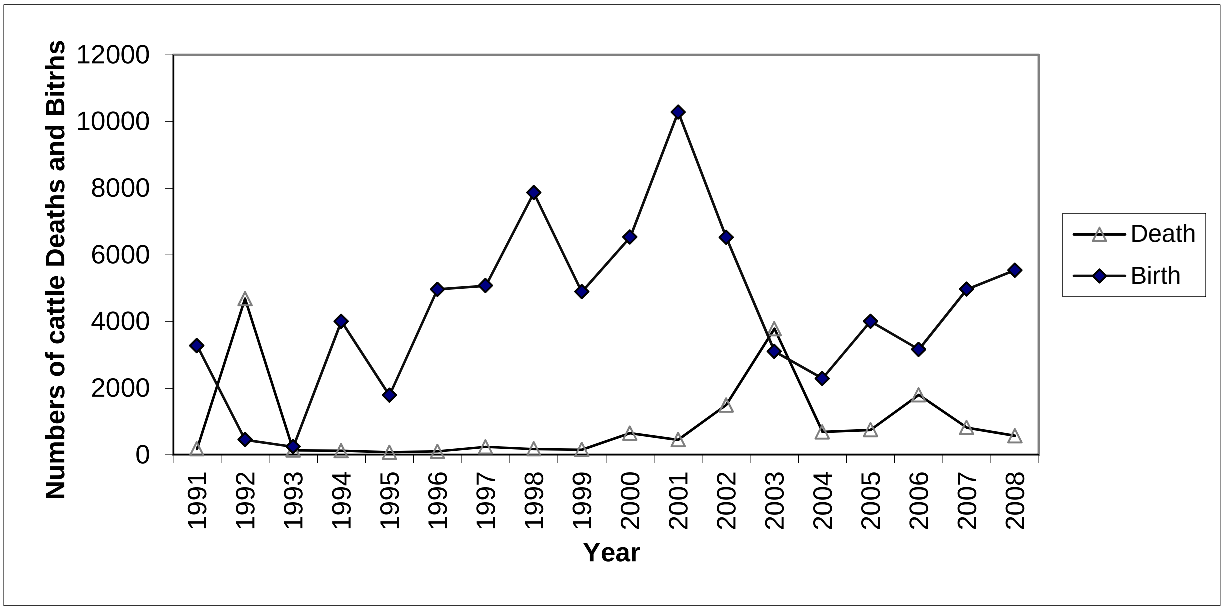


Figure 3.3: Total cattle births and deaths from 1991 to 2008 for cattle populations in wards 11 to 15 of south eastern lowveld of Zimbabwe.

Cattle

Visible inspection of Figures 3.2 and 3.3 shows that the trends of cattle mortalities from data from the whole area (Figure 3.3) followed rainfall fluctuations described above. Peaks in cattle mortalities were evident in 1992, 2003 and 2006. It is also apparent that the increases in cattle mortalities started in 2002 and 2005 for the 2003 and 2006 peaks respectively, coinciding with periods below average amount of rainfall and/or an unfavourable distribution. Concurrently, fewer calves were born during these periods (Figure 3.3). Cattle densities gradually increased after a major crash in 1992 with data from all the areas showing a similar trend for the period 1993 to 2001 as highlighted by the double arrow. Major crashes in cattle densities were evident in 1992 and 2005 especially for Gonakudzingwa (Figure 4 b) and Pahlela (Figure 3.4c); while in Malipati (Figure 4 d) the crash had occurred due to the 2002 drop in rainfall. However, in some areas a year delay in cattle density response to changes in rainfall was apparent for example in Gonakudzingwa and Pahlela (Figure 3.4 b and c respectively) where there was a decrease in cattle density in 2006 after the poorly distributed rainfall in 2005 season.

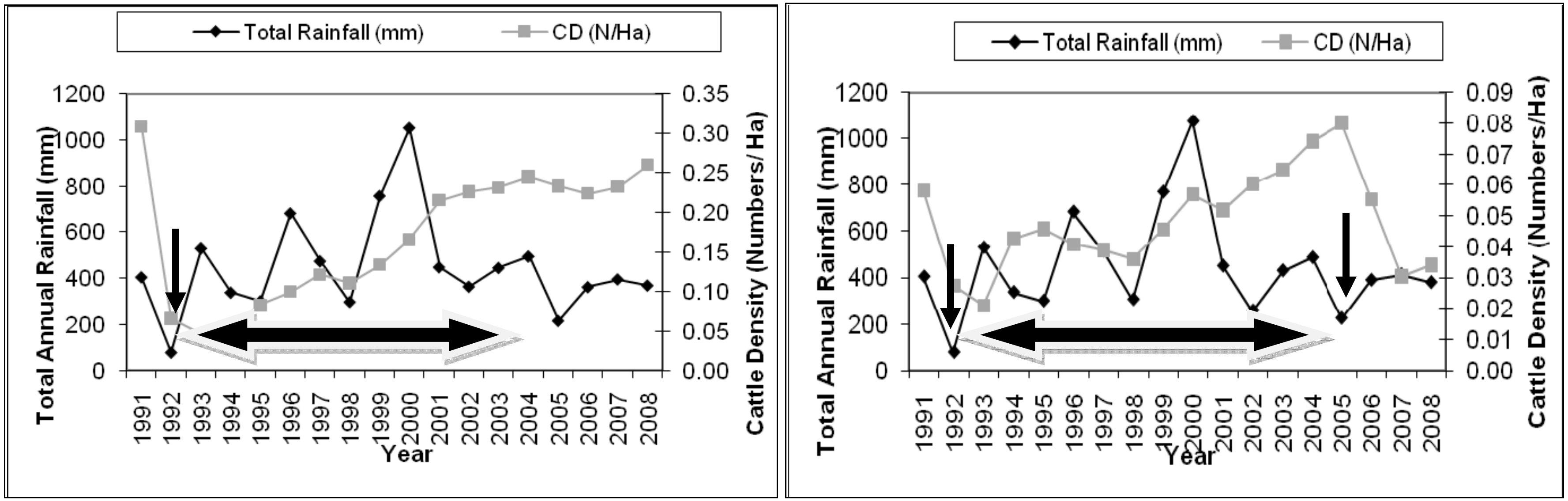
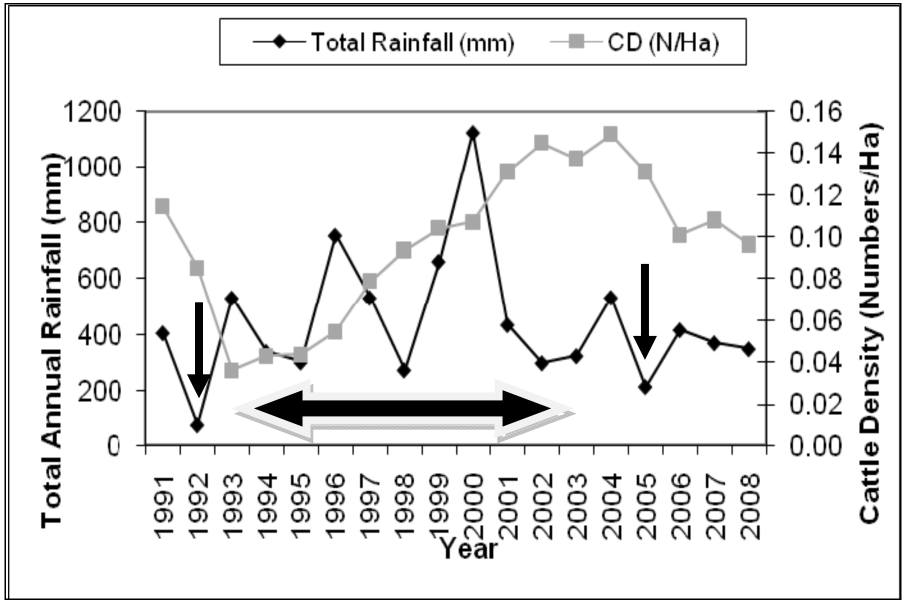


Figure 3.4 a: Total annual rainfall and cattle densities from 1991 to 2008 for cattle populations in Chikombedzi. **Figure 3.4 b:** Total annual rainfall and cattle densities from 1991 to 2008 for cattle populations in Gonakudzingwa [Arrow pointing down = years with a combination of low rainfall and low rain days (crashes in cattle densities), double arrow = Period of gradual increase in cattle density].



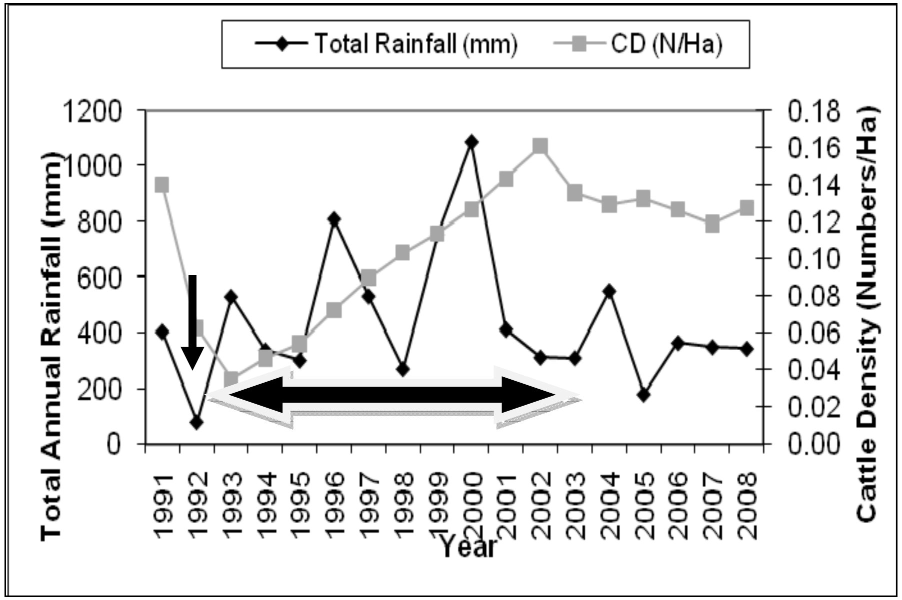


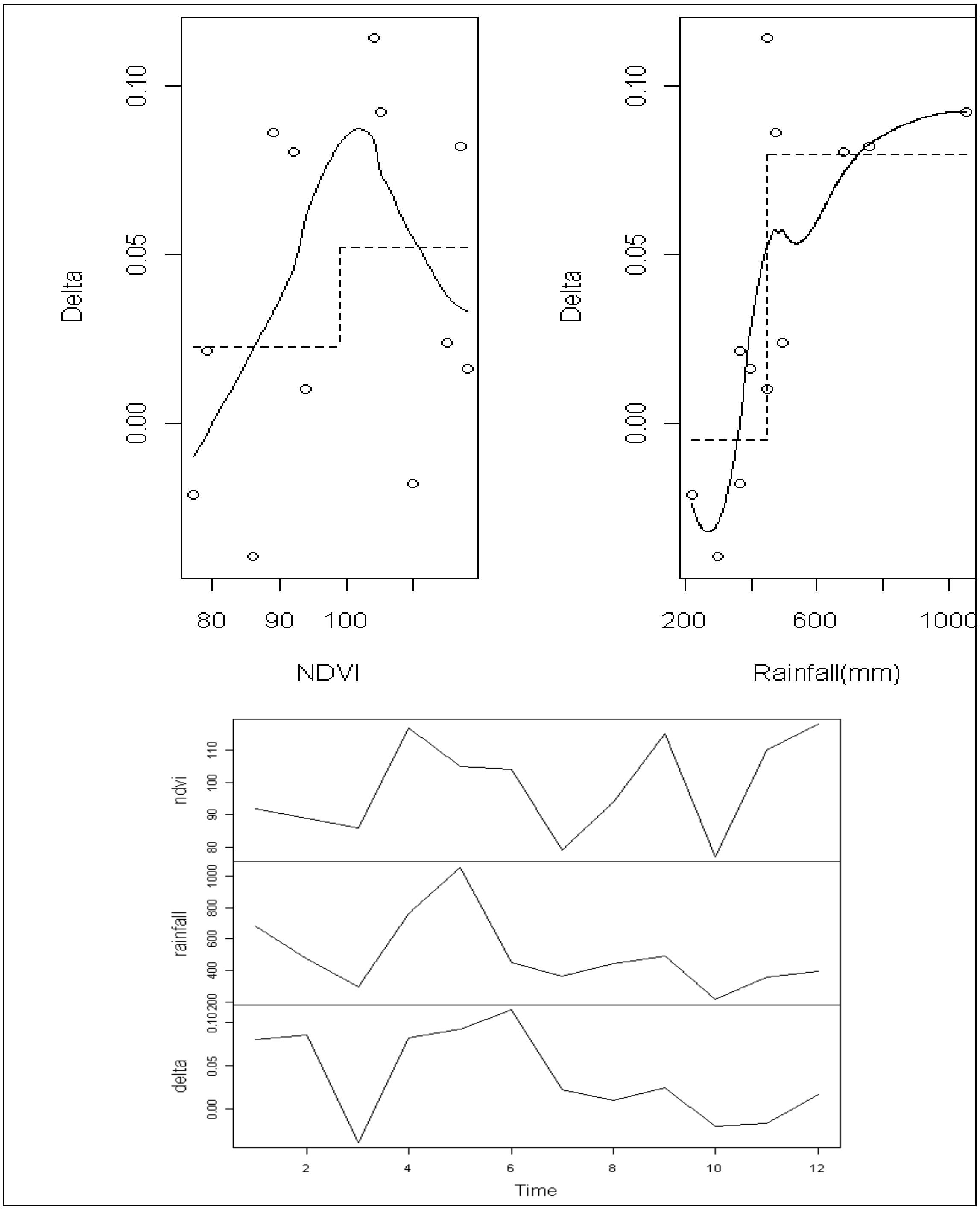
Figure 3.4 c: Total annual rainfall and cattle densities from 1991 to 2008 for cattle populations in Pahlela. **Figure 3.4 d:** Total annual rainfall and cattle densities from 1991 to 2008 for cattle populations in Malipati. [Arrow pointing down = years with a combination of low rainfall and low rain days (crashes in cattle densities), double arrow = Period of gradual increase in cattle density].

Step function

A step function involves estimation of three parameters: two averages and a threshold. When the two averages are significantly different from each other, it shows the existence of a threshold (Crawley 2007). In this study a negative Δ_{cattle} below the threshold and a positive Δ_{cattle} above the threshold were expected. Meaning that below a certain threshold set by either rainfall or NDVI, population was expected to be declining while above the threshold a population increase was expected due to availability of forage. Results from individual wards showed the presence of a threshold using rainfall as an explanatory factor for Chikombedzi (Threshold, $F_{1, 10}$ = 28.88, $P = 0.00031$) and Sengwe (Threshold, $F_{1, 10}$ = 10.26, $P = 0.0094$) (Figures 5a and 5c). The presence of a threshold was confirmed in Pahlela (Threshold, $F_{1, 10}$ = 5.59, $P = 0.0397$) and Malipati (Threshold, $F_{1, 10}$ = 18.05, $P = 0.0017$) using NDVI as an explanatory factor (Figure 3.5b and 3.5d). In Gonakudzingwa, rainfall had a better fit as an explanatory variable (Threshold, $F_{1, 10}$ = 3.8154, $P = 0.07951$), than NDVI (Threshold, $F_{1, 10}$ = 2.9482, $P = 0.1167$), however it was not significant (Figure 3.5e).

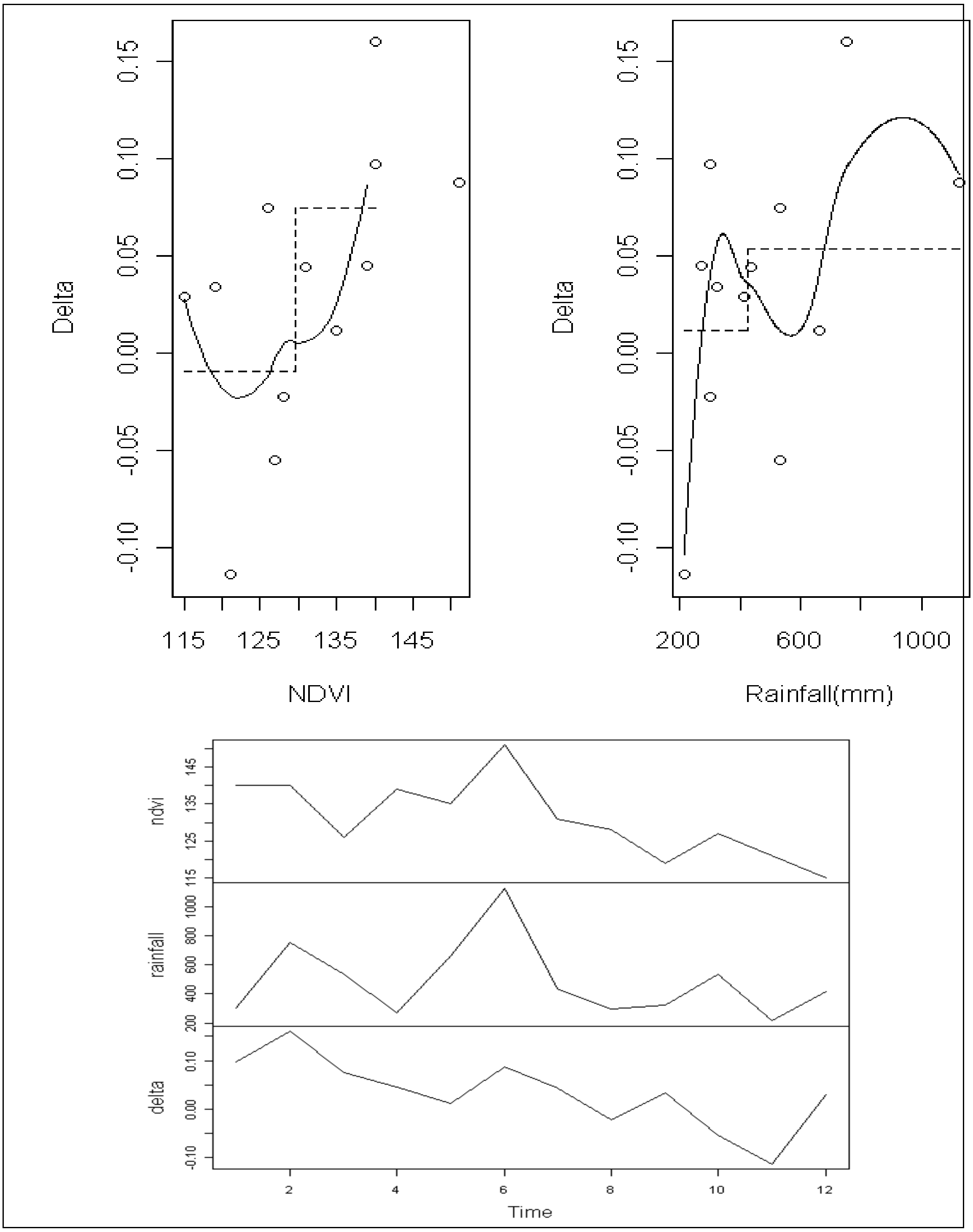
Results from a regression tree model to investigate factors that determine changes in cattle populations showed that the dataset was first split by rainfall with a mean value of 423 mm (Figure 3.6). This indicates the overriding effect of rainfall in changes in numbers of cattle over years. The node corresponding to the lower part of rainfall (below 423) was further split by movement in (buying in) of cattle into the area. Indicating that the major changes in cattle numbers during years below the rainfall mean were explained by buying in of animals. Above the rainfall mean, NDVI (3 year lag) significantly ($P < 0.05$) explained changes in cattle numbers. Indicating that in wetter years changes in cattle numbers were best explained by vegetation.

Figure 3.5a: Cattle population change, Delta = Δ_{cattle} = log (N_{t+1}/N_t) as a function of NDVI (left) and rainfall (right) for Chikombedzi. Solid lines show a loess smooth curve fit (Residual error = 0.054 for NDVI and 0.039 for rainfall) and the broken lines show a step function fit (Threshold, $F_{1, 10}$ = 0.9474, N.S for NDVI and Threshold, $F_{1, 10}$ = 28.88, P < 0.001 for rainfall***). Insert showing multi-series plot of NDVI, rainfall and Δ_{cattle} (= delta).



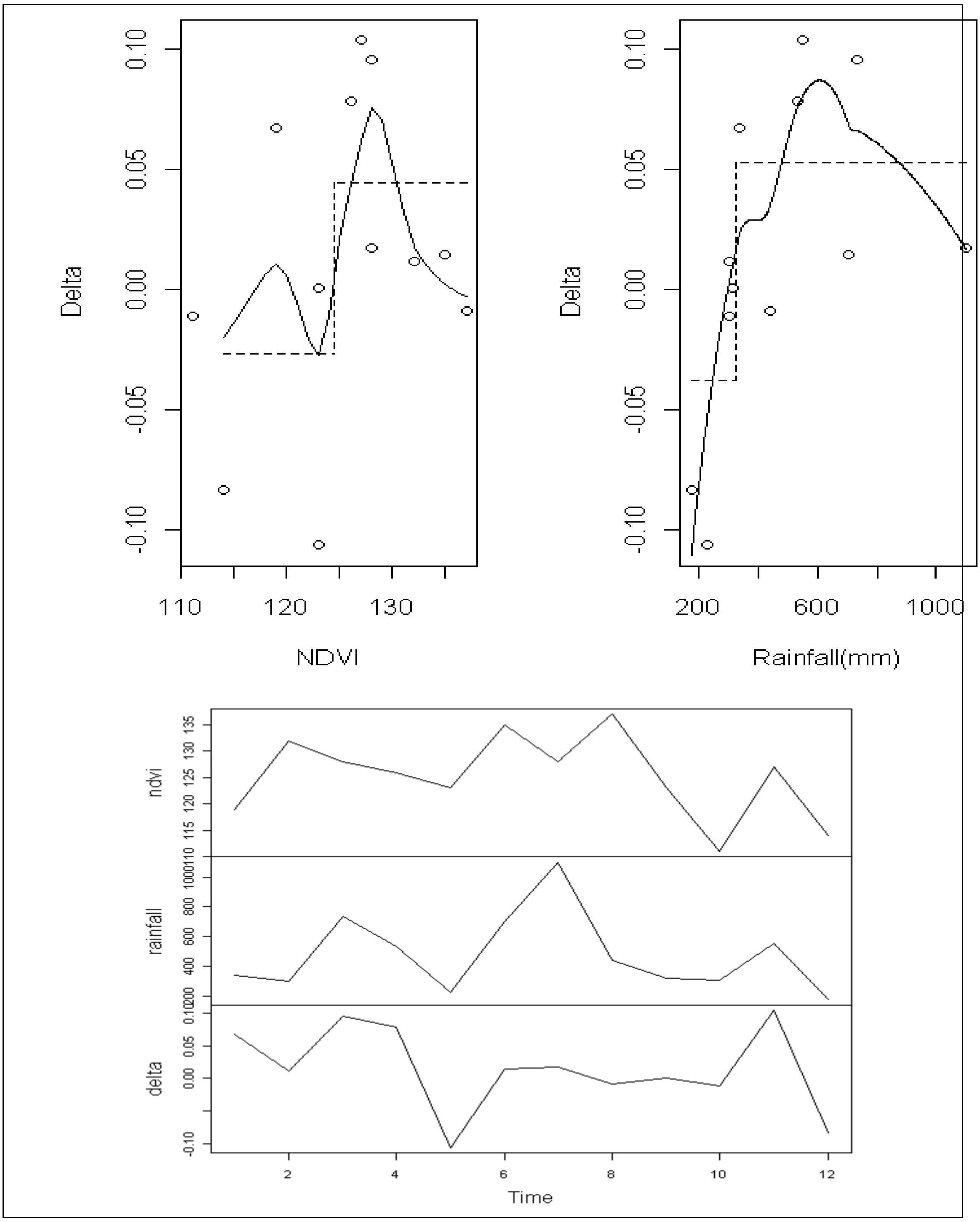
*** Shows significance at P < *0.0001*

Figure 3.5b: Cattle population change, Delta = Δ_{cattle} = log (N_{t+1}/N_t) as a function of NDVI (left) and rainfall (right) for Pahlela. Solid lines show a loess smooth curve fit (Residual error = 0.06883 for NDVI and 0.06338 for rainfall) and the broken lines show a step function fit (Threshold, $F_{1, 10}$ = 5.59, P < 0.05 for NDVI* and Threshold, $F_{1, 10}$ = 1.0233, N.S for rainfall). Insert showing multi-series plot of NDVI (2 year lag), rainfall (1 year lag) and Δ_{cattle} (= delta).



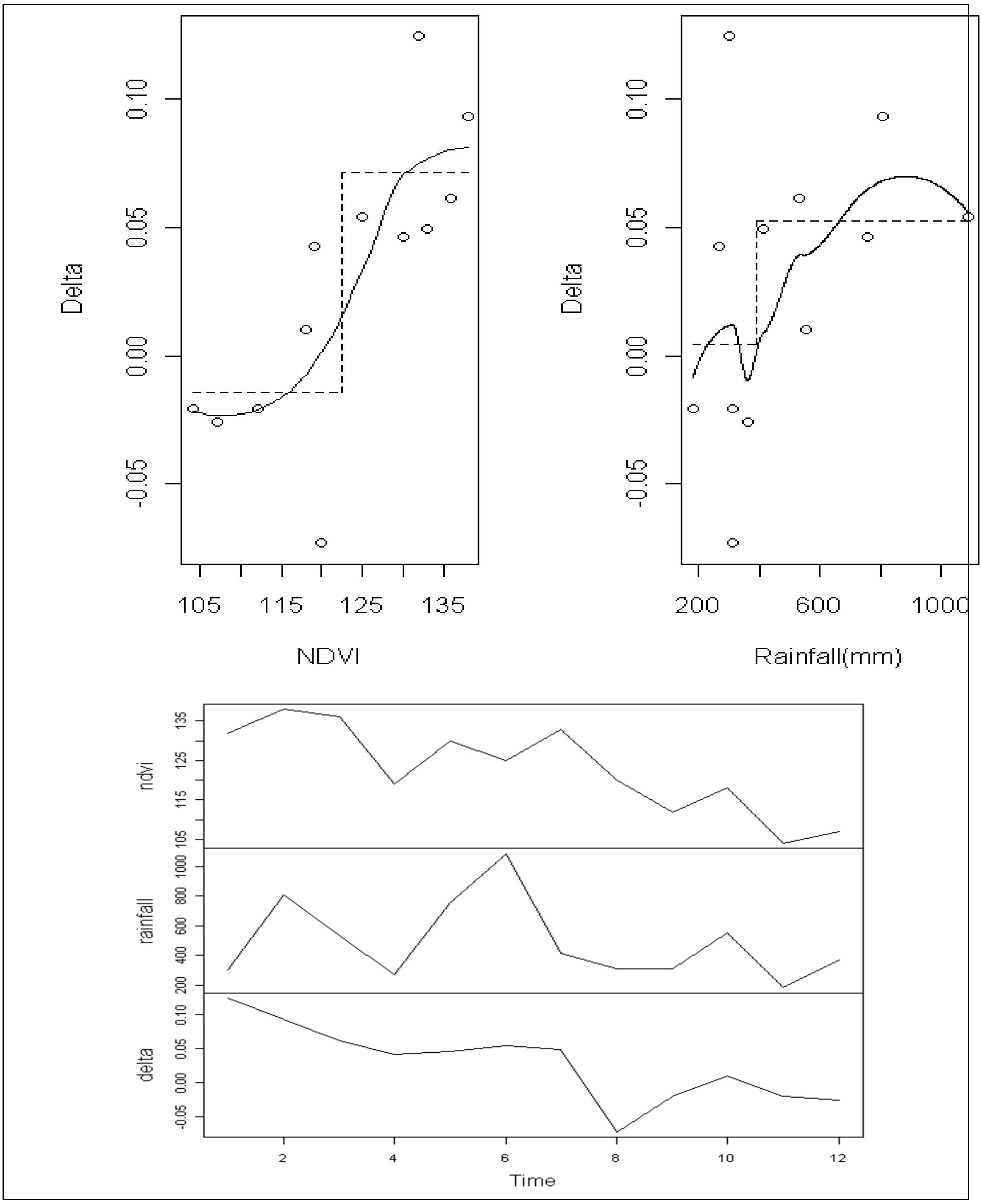
* Shows significance at $P < 0.05$

Figure 3.5c: Cattle population change, Delta = Δ_{cattle} = log (N_{t+1}/N_t) as a function of NDVI (left) and rainfall (right) for Sengwe. Solid lines show a loess smooth curve fit (Residual error = 0.06264 for NDVI and 0.04474 for rainfall) and the broken lines show a step function fit (Threshold, $F_{1, 10}$ = 4.5971, P < 0.1 for NDVI and Threshold, $F_{1, 10}$ = 10.258, P < 0.01 for rainfall**). Insert showing multi-series plot of NDVI (2 year lag), rainfall (2 year lag) and Δ_{cattle} (= delta).



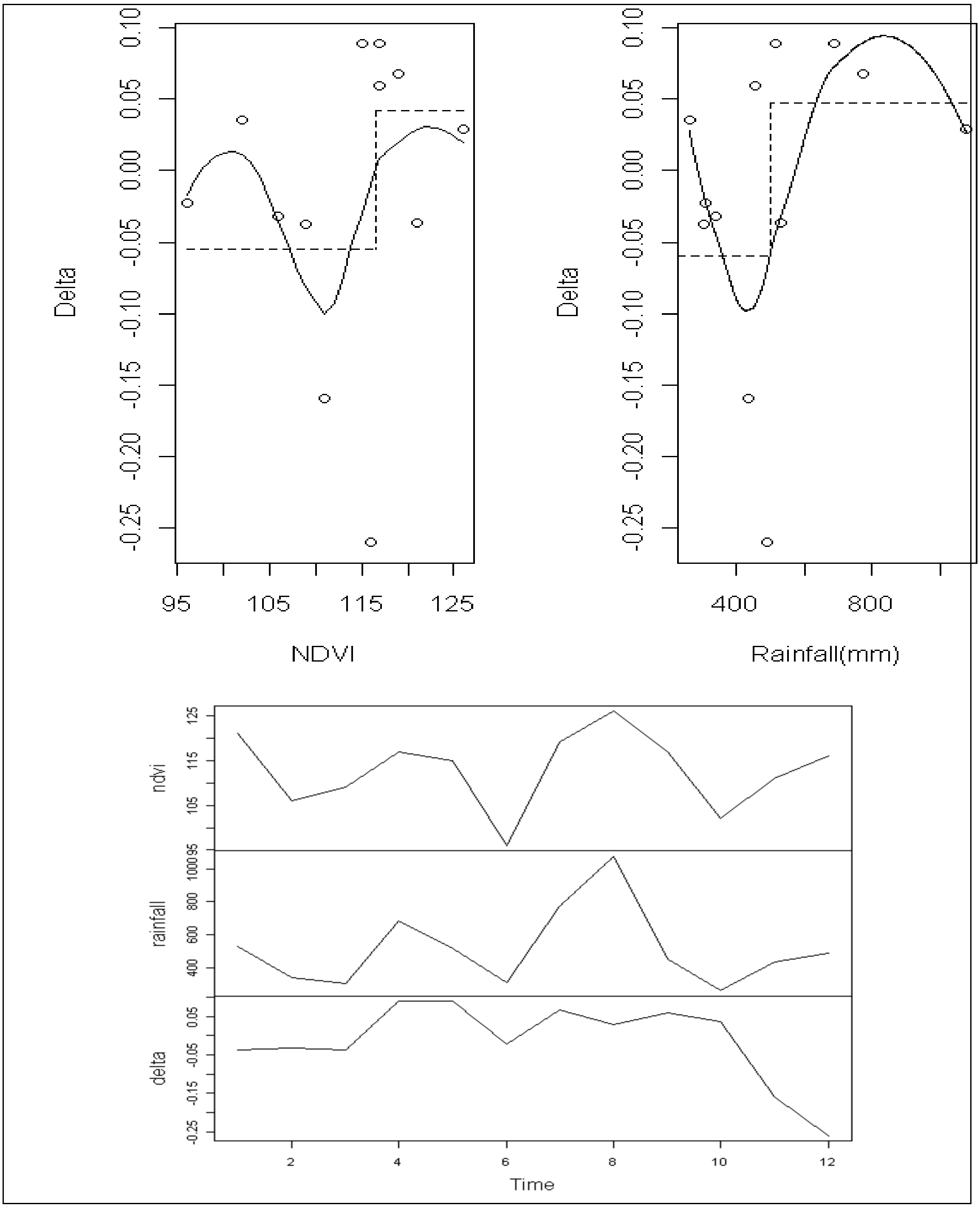
** Shows significance at *P < 0.001*

Figure 3.5d: Cattle population change, Delta = log (N_{t+1}/N_t) as a function of NDVI (left) and rainfall (right) for Malipati. Solid lines show a loess smooth curve fit (Residual error = 0.0433 for NDVI and 0.06813 for rainfall) and the broken lines show a step function fit (Threshold, $F_{1,\ 10}$ = 18.046, P < 0.01 for NDVI** and Threshold, $F_{1, 10}$ = 2.4879, P = 0.1458 for rainfall). Insert showing multi-series plot of NDVI (1 year lag), rainfall (1 year lag) and Δ_{cattle} (= delta).



** Shows significance at $P < 0.001$

Figure 3.5e: Cattle population change, Delta = log (N_{t+1}/N_t) as a function of NDVI (left) and rainfall (right) for Gonakudzingwa. Solid lines show a loess smooth curve fit (Residual error = 0.1295 for NDVI and 0.1197 for rainfall) and the broken lines show a step function fit (Threshold, $F_{1, 10}$ = 2.9482, N.S for NDVI and Threshold, $F_{1, 10}$ = 3.8154, P < 0.1 for rainfall). Insert showing multi-series plot of NDVI (3 year lag), rainfall (3 year lag) and Δ_{cattle} (= delta).



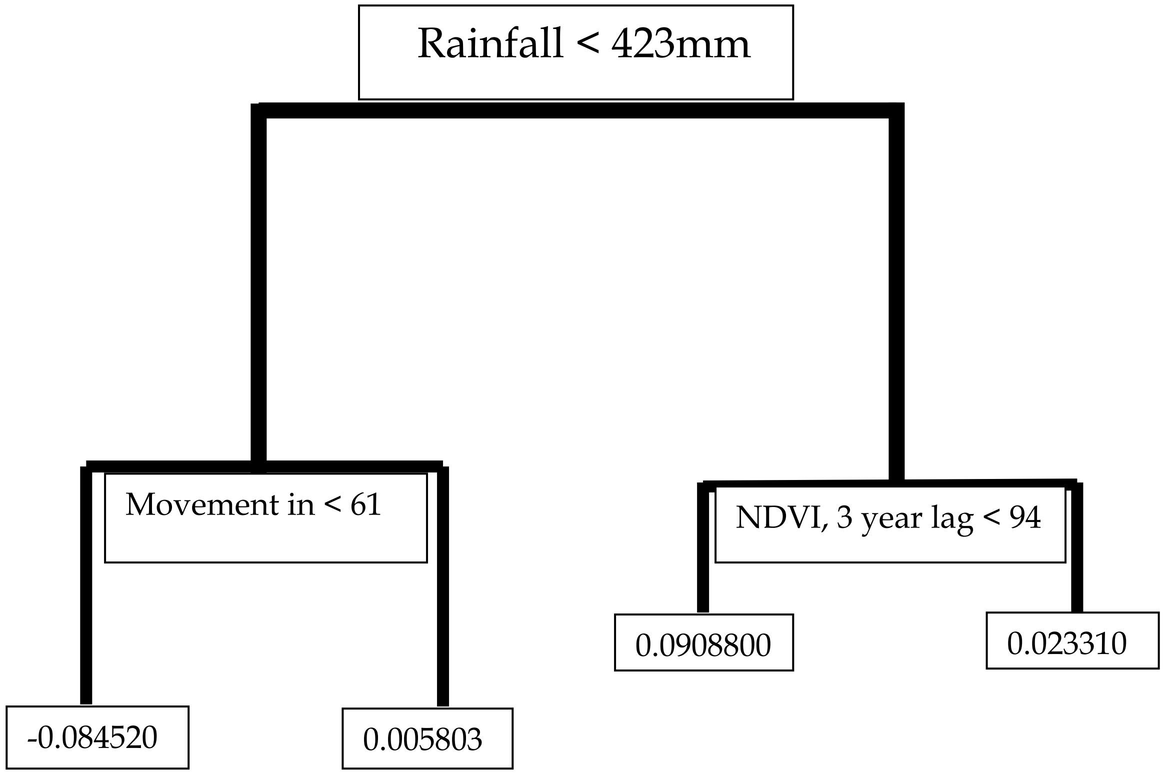


Figure 3.6: Regression tree showing variables that explain the greatest amount of the deviance in Δ_{cattle} (delta: changes in cattle numbers) in southeastern Zimbabwe.

Cattle age categories and sex response to drought and their recovery

Based on data from the 8 dip tanks, we found that the rates of decline differ significantly (Chi-Square = 38.7, d.f. = 7, $P < 0.001$) between cattle age categories (Figure 3.7a). Calves and juvenile bulls had the highest decline while heifers were the least affected by drought. The effects of drought however, were the same for males and females (U = 23, N = 7, $P > 0.05$) (Figure 3.7b). Recovery of different age groups from drought was significantly different (Chi-Square = 25, d.f. = 7, $P < 0.001$), with calves showing a higher recovery rate and adult oxen and heifers showing a low recovery rate (Figure 3.8a). Recovery of males from drought was similar to that of females (t-test, t = 1.42, d.f. = 12, $P > 0.05$) (Figure 3.8b).

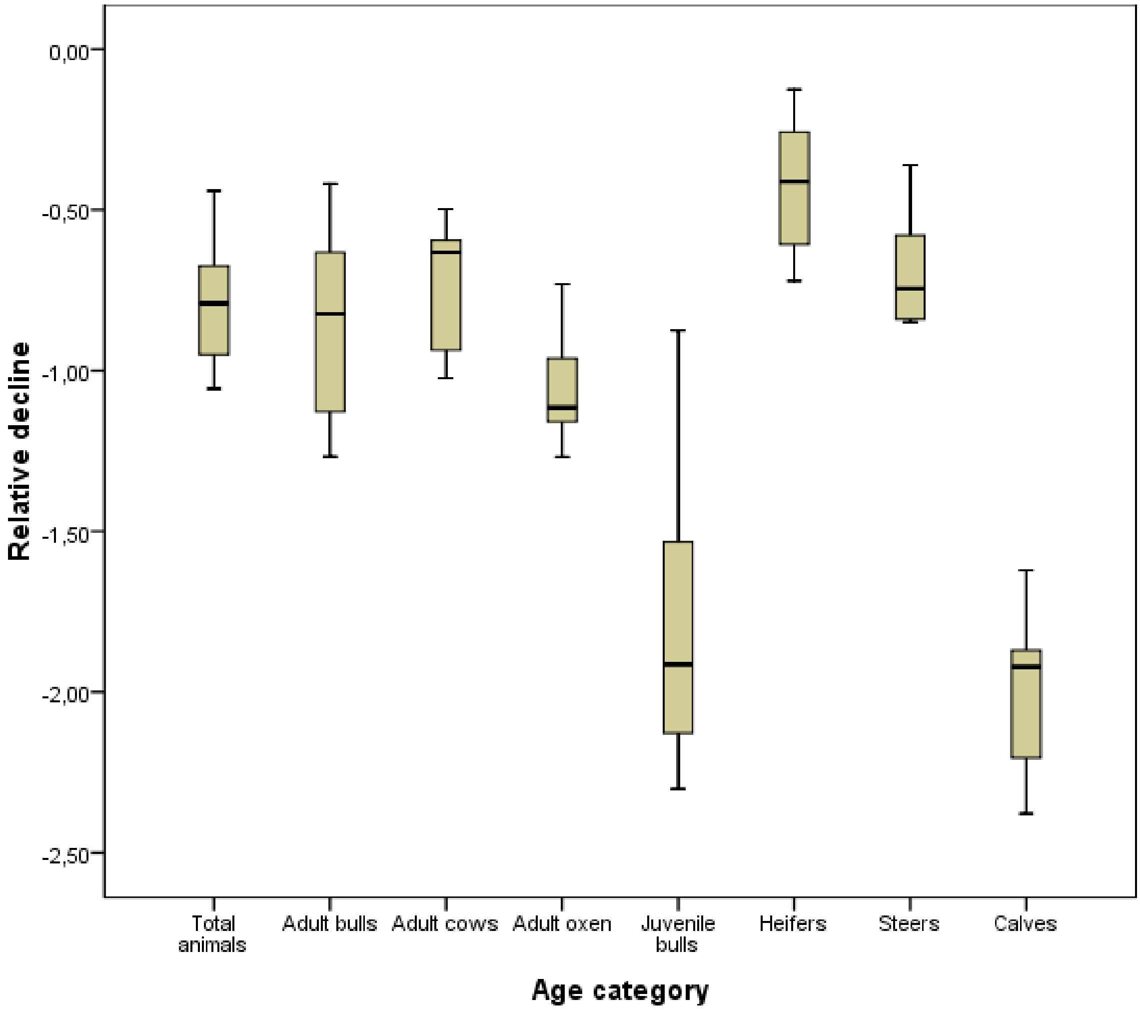


Figure 3.7a: Differences in relative decline in cattle numbers by their age categories due to the 1991/92 drought for Chikombedzi and Gonakudzingwa diptanks.

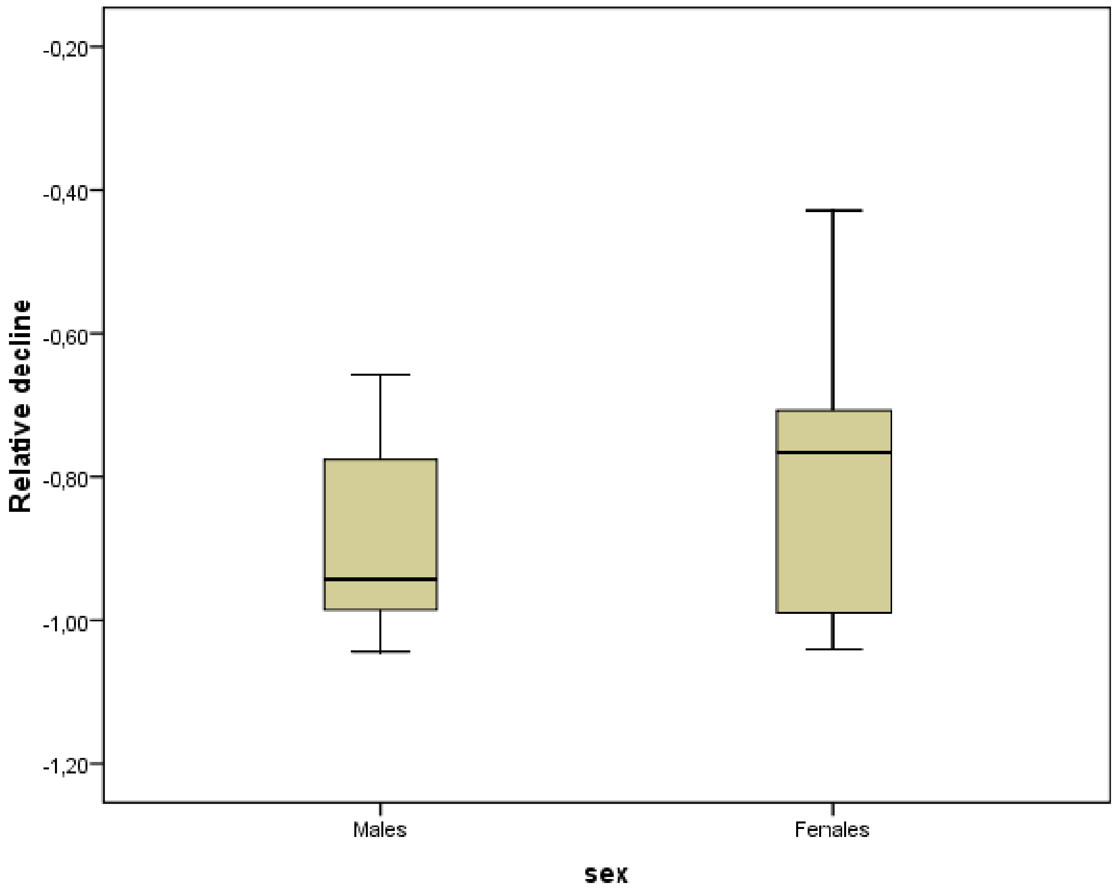


Figure 3.7b: Relative decline in cattle numbers by sex due to the 1991/92 drought for Chikombedzi and Gonakudzingwa diptanks.

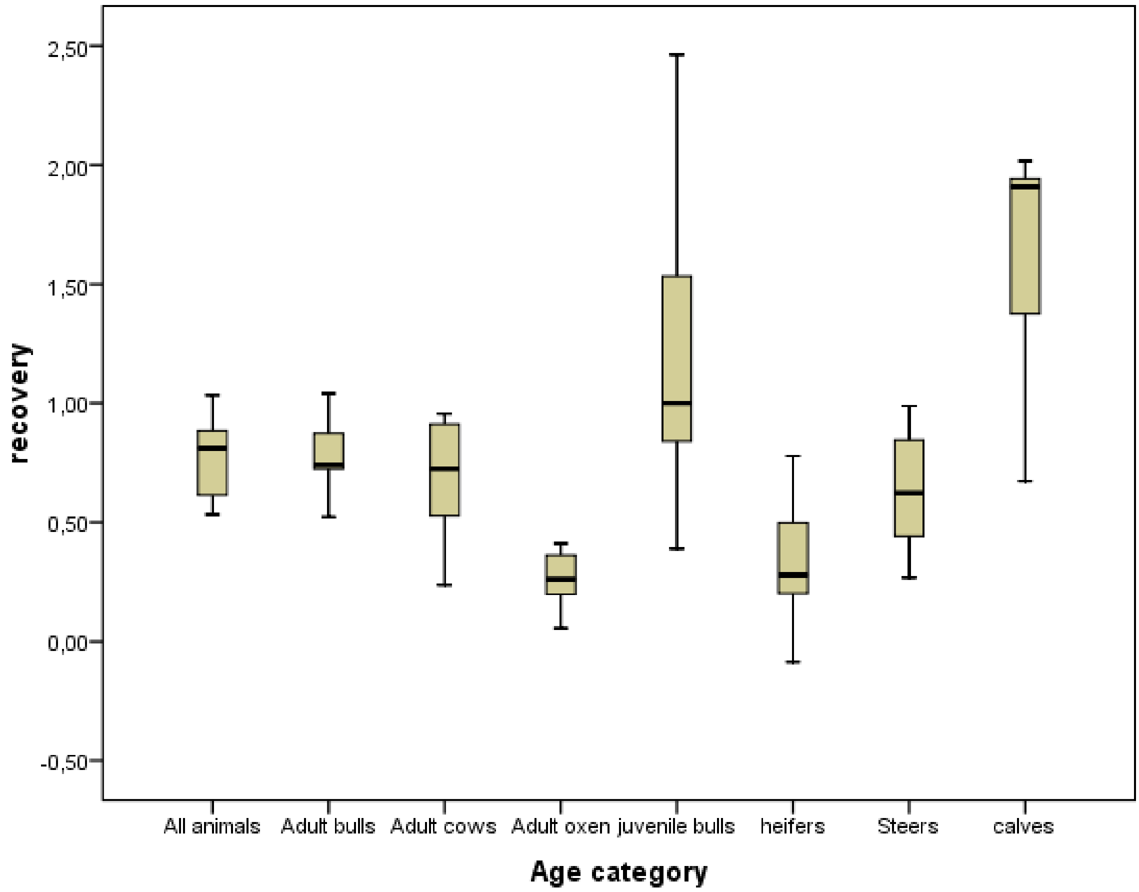


Figure 3.8a: Differences in recovery in cattle numbers by their age categories after the 1991/92 drought for Chikombedzi and Gonakudzingwa diptanks.

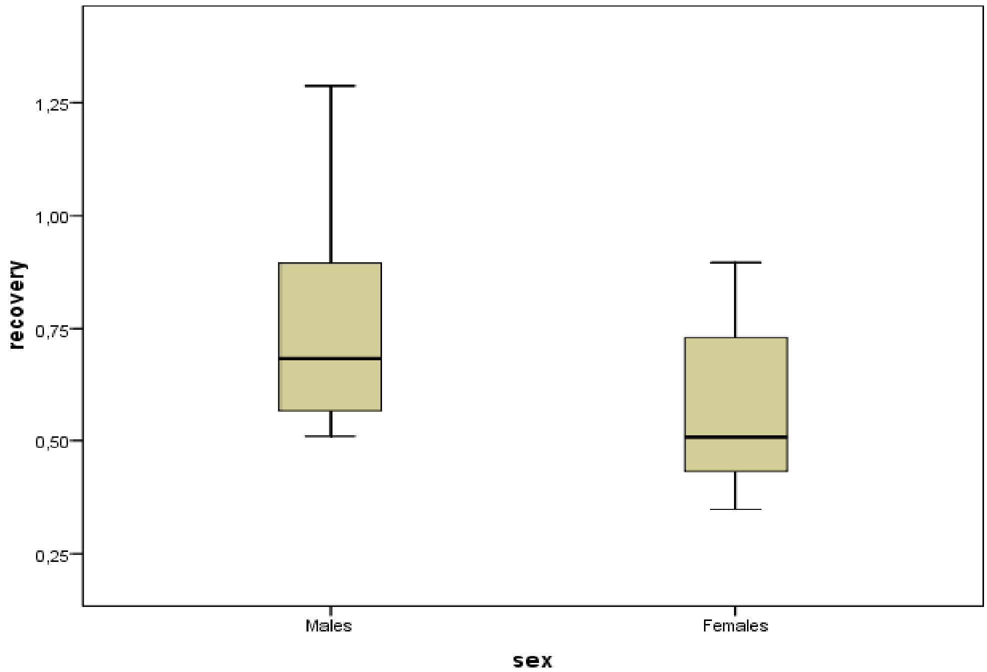


Figure 3.8b: Recovery in cattle numbers by sex after the 1991/92 drought for Chikombedzi and Gonakudzingwa diptanks.

Discussion

In this study, we showed that in areas with highly variable rainfall, like the southeastern lowveld of Zimbabwe, non-equilibrium dynamics are present as evidenced by the existence of thresholds established using rainfall and vegetation (NDVI) as explanatory variables. Non-equilibrium is any situation where species densities do not remain constant over time at each spatial location; hence the presence of thresholds supports this view. Results have also shown that rainfall was the overriding factor, whereas NDVI and inward movement of animals also account for greater variation in population changes (Delta) of cattle. Furthermore, the effects of these rainfall fluctuations through drought affect juveniles more than mature animals whereas the effects are homogenous between sexes. This study also established that recovery from drought was heterogeneous among cattle age classes and homogenous among sexes, with juveniles showing higher recovery rates than other age class categories. These results suggest that the southeastern lowveld rangeland system is driven primarily by variable rainfall which results in highly variable and unpredictable primary production. These findings agree with Campbell *et al*. (2006), who stated that forage production is closely correlated with annual rainfall in semi-arid and arid systems.

Analysis of the rainfall data over a 21-year period show that rainfall is unpredictable across the year and from one year to the next as evidenced by high CVs for inter-annual rainfall variation and for the number of rain days (51 and 41%, respectively). These values exceed the 30% threshold where a system becomes dominated by variability more than by average conditions reported by Caughley (1987). As reported by Ellis (1995) CVs around 30-33% may occur if positive or negative departures from the mean are frequent, but not too large, or large but not too frequent. Thus, CVs in this study would suggest that departures from the mean are both frequent and large. Similarly high variability in rainfall has been reported from studies done in South Turkana region in Kenya (Ellis and Swift 1988), Lake Manyara National Park in Tanzania (Prins & Loth 1988), and rangelands of Southern Ethiopia (Angassa and Oba 2007) from which non-equilibrium dynamics were suggested.

Further, trends of cattle mortality, birth and cattle densities closely followed fluctuations in annual rainfall and its distribution. Angassa and Oba (2007) also reported similar trends in cattle mortalities in a study in southern Ethiopia. This suggests that erratic rainfall leads to swings of available forage which likewise becomes variable and also in turn leads to swings in cattle densities and mortalities. Similar findings were reported in Australian rangelands where swings in Kangaroo (*Macropus rufus*) density were generated and maintained by swings in pasture biomass that were influenced by swings in rainfall (Caughley 1987; Ellis *et al*. 1993).

In addition, our results show that green vegetation as measured by average NDVI can significantly be explained by total annual rainfall in majority of the wards (Figure 3.5) and was significant in establishing thresholds to split the growth rates of cattle populations (Delta) in two wards. These two wards (Pahlela and Malipati) had high tree cover compared to areas (Chikombedzi and Sengwe) where rainfall was significant, that are mainly covered by annual grass species. This suggests that in latter areas cattle changes were sensitive to annual fluctuations in rainfall via the direct impact of rainfall on annual forage availability. Whereas in the former areas, lag effects of NDVI were evident as tree cover would take time to respond to fluctuations in annual rainfall. This suggests that, although rainfall in southeastern lowveld of Zimbabwe may be unpredictable, the response of plants to rainfall events was predictable. This indicates that rainfall effect on forage production dominates the grazing effect, meaning that feedback between forage production and grazing is not evident. It was suggested that in grazing systems with very high climatic variability, forage availability varies to such a great degree with rainfall that herbivore population dynamics are driven by rainfall via its direct effect on forage availability in any given year (Wiens 1977; Ellis and Swift 1988; Vetter 2005). Therefore in such systems density-dependent interactions such as competition for resources play a minor role in controlling populations.

Regression tree analysis showed that rainfall, NDVI (with a 3 year lag) and inward movement of animals were significant factors in explaining cattle changes. The mechanism possible is that below the rainfall thresholds (Figure 3.6) established above, the main factor that explains cattle changes is the buying in of animals. This factor is also the main means of recovery after a devastating drought like the 1991/1992 drought. NDVI becomes important above the rainfall threshold, again proving the importance of rainfall in this system. This observation agrees with Sullivan and Rohde (2002) and Derry and Boone (2010), who stated that the dynamics of all living systems intrinsically are non-equilibrial, although predictable and tightly coupled interactions and dynamics might be exhibited at certain scales of observation. Hence, at certain times, for example, during a series of low to medium rainfall years, or for key resources like dry season grazing, these fluctuations can be mediated by density dependent effects (Illius and O'Connor 1999; Gillson and Hoffman 2007). Therefore, we acknowledge that disequilibrium in its strict sense of a system dominated by environmental variation can occur only in extremely dry environments and / or in exceptionally prolonged drought periods, when there is literally no primary productivity for which animals can compete (Sullivan and Rohde 2002; Gillson and Hoffman 2007).

Overall, our results show the importance of rainfall fluctuations in southeastern lowveld grazing system which makes non-equilibrium dynamics likely. However, other authors advocate a less stringent division between equilibrium and non-equilibrium dynamics. For instance (Buttolph and Coppock 2004; Zemmrich 2007) found that both non-equilibrium and equilibrium forces appear to operate on small, wet meadow subsystems nested within an extensive dry alpine system. They state that stable and low rates of above-ground net primary production (ANPP) in wet meadows are largely shaped by the cold climate, a non-equilibrium factor. Changes in plant species composition and livestock productivity, however, support equilibrium theory. The authors concluded that the relatively small wet meadow patches may operate as equilibrium subsystems within a much larger, non-equilibrium landscape, in line with the findings of Briske *et al.* (2003). The latter argue that equilibrium and non-equilibrium ecosystems should not be distinguished on the basis of unique processes or functions, but rather by the evaluation of system dynamics at various temporal and spatial scales. They argue that ecosystems may express both equilibrium and non-equilibrium dynamics.

This study demonstrates that the southeastern lowveld grazing system is a non-equilibrium system. We maintain that, while livestock clearly require forage, the availability of forage is driven by (or coupled more strongly with) overriding abiotic factors, like rainfall. It is these abiotic constraints on primary productivity that drive animal populations and thereby weaken any deterministic coupling between plants and animals (Sullivan 2002). The impacts of these dynamics as evidenced by impacts of drought are on the survival of juveniles. Results from this study agree with the view that juveniles and yearling males are more susceptible to harsh environmental conditions than even breeding females (Clutton-Brock 1991). In a similar study in Northern Kenya, Oba (2001) reported breeding females and calves as the age groups most affected by multiple droughts. In this study however, juvenile bulls and calves were the main age categories to suffer most from effects of drought. Males invest more in growth and less in body reserves (Focardi *et al*. 2008), leading to juvenile bulls being more susceptible to drought. The other reason that makes calves more susceptible to drought is the deliberate culling of calves to save lactating cows as reported by Oba (2001). Survival of males and females were reported to differ only for yearlings in wild boar (Focardi *et al*. 2008), in agreement with no differences between sexes reported in this study. The recovery from drought also showed that calves had high recovery rates compared to other age categories. Supporting the view that with good rainfall, recovery improves calving rates therefore a higher number of calves and juveniles. Since calves and juveniles are the common denominator in responding and recovering from droughts, we argue that efforts in mitigating impacts of non-equilibrium dynamics should revolve around saving the young.

We can conclude that southeastern lowveld is characterized by a highly risky and uncertain system, so management of such grazing system should put more emphasis on saving the young cattle, because our study has shown that these animals are particularly vulnerable.

Acknowledgements

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Chapter 4

Reducing rural households annual income fluctuations due to rainfall variation through diversification of wildlife use: portfolio theory in a case study of southeastern Zimbabwe

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Abstract

Annual rural incomes in Southern Africa show large rainfall-induced fluctuations. Variable rainfall has serious implications for agro-pastoral activities (crop cultivation and livestock keeping), whereas wildlife and tourism are less affected. The aim of this paper is to investigate the role of wildlife income in reducing rainfall-induced fluctuations in households' annual incomes. We analyse costs and benefits from agro-pastoral systems in southeastern Zimbabwe by means of a two-tier longitudinal survey and wildlife benefits through analysis of wildlife revenues. We use the portfolio theory framework to investigate whether wildlife conservation has the potential for farmers to reduce risk associated with agricultural production. Results show that even though wildlife income is small, it tends to be less volatile than income from the agro-pastoral system. Furthermore, the addition of wildlife as an asset to the rural farmers' portfolio of assets showed that wildlife can be used as a hedge asset to offset risk from agricultural production without compromising on return. The potential of diversification using wildlife is, however, limited since agriculture and wildlife assets are positively correlated. We conclude that revenues from wildlife have some potential to reduce annual household income fluctuations, but only to a limited extent. We argue that if wildlife is organized on a more commercial basis, a more substantial role can be played by wildlife income in reducing variations in rural households' incomes.

Key words: Southeastern Zimbabwe; droughts; portfolio theory; assets; risk.

Introduction

Most rural households in Sub-Saharan rangelands depend on agro-pastoral land-use activities for their livelihood, combining small scale farming with livestock keeping, or they specialize in herding (pastoralists) or crop cultivation (Homewood 2004). These households are vulnerable to a wide variety of shocks such as droughts, floods, illness, or localized insect infestation (Owens *et al.* 2003). Such shocks may impose utility losses on households, and reduce the capacity of households or individuals to generate income, mainly because local insurance schemes are absent and monetary savings are too small to act as buffers. Climate-related natural events like droughts are principal sources of risk in savannas. Drought is considered to describe a situation of limited rainfall that is substantially below what has been established to be a "normal" value for the area concerned, leading to adverse consequences on human welfare (Pandey 2007) or loss of physical condition or even mortality among livestock and wildlife. Droughts may induce short-term coping tactics like producing and selling charcoal, thus damaging the resource base and endangering long-term livelihood security (Eriksen and Watson 2009). Income fluctuations due to droughts tend to lead to consumption instability or even to starvation (Kinsey *et al.* 1998). However, income from wildlife utilization often has potential to reduce these fluctuations in income. In sub-Saharan rangelands, high levels of biodiversity still exist, and because wildlife species have evolved with the savanna vegetation (Bouchenak-Khelladi *et al.* 2009), they may be better adapted to annual rainfall fluctuations than domestic livestock species.

The 'sustainable use' of wildlife, as opposed to its outright preservation through command and control policies, has a clear economic rationale (Pearce and Moran 1994; Child 1996; Mbaiwa 2005), because human appropriation of the land for food supply, infrastructure and other economic developments competes with wildlife (Prins 1992). Wildlife needs to be of economic value to local people in order to compete with other land uses. Stripped of its economic value, wildlife cannot compete with other land uses because the competition is too heavily tilted against it (Pearce and Moran 1994) and the potential for a conservation relationship between wildlife and local communities is removed. Wildlife is often considered to be a nuisance in terms of disease, crop and livestock predation, and even a danger to human life (Prins 2000). Taking economic value away removes added value from wildlife in the form of trophies or for the support of tourism and recreation that make wildlife exploitation economically more attractive than livestock exploitation in a market economy (Prins and Grootenhuis 2000). For example, sustainable use of wildlife more than doubled the land allocated to wildlife in southern Africa by the year 2000 compared to the late 1980s (Cumming and Bond 1991; Hearne and Mckenzie 2000), because it has a comparative economic advantage in these environments (Child and Chitsike 2000).

Despite claims that African wildlife can generate greater profits than cattle, the relative profitability of extensive cattle and wildlife has not been well established for semi-arid savannas with limited diversity of wildlife (Gambiza *et al.* 2010), especially outside of protected areas. For southeastern Zimbabwe, which receives unreliable annual rainfall below 600mm, Child reported that wildlife alone provides more profit than either cattle or a combination of cattle and wildlife (Child 2009). Economic analysis of community wildlife-use initiatives in Namibia and Botswana have shown that conservancy investments in Namibia and wildlife resources in Botswana are economically efficient and contribute positively to national economic well-being (Barnes 2001; Barnes *et al.* 2001; Barnes *et al*. 2002; Barnes and Jones 2009). Additionally, data from South Africa confirm that switching to wildlife increased employment five times, the total wage bill 30 times, created numerous upstream and downstream economic multipliers and doubled land values (Child 2009; Langholz and Kerley 2006). Wildlife is therefore an important and growing source of income throughout southern Africa under a commercial or ranch set up.

Very few attempts have been made to understand the extent to which wildlife income can complement income in rural households. Most rural Africans live on communal lands, where they are often politically disempowered and administratively alienated from the wild resources upon which they depend (Child and Barnes 2010). Radeny *et al.* (2007) investigated livelihood choices and income diversification strategies in a traditionally Masai pastoral area of southern Kenya, finding that diversification through cropping was a weak option, with many households not getting a harvest even in a 'good rainfall year'. Instead, households that received wildlife use-related income found it to be a more lucrative option compared to cropping. This implies that wildlife income can potentially complement agro-pastoral incomes for local people in communal systems that show high fluctuations in annual rainfall. The theoretical framework of this paper is based on portfolio theory (Markowitz 1952; Markowitz 1959). Markowitz's original analysis related to financial securities (Figge 2004), but in this study, under the CAMPFIRE philosophy, rural farmers have an opportunity to acquire income from wildlife conservation as an additional asset. Like agricultural production, wildlife conservation is characterized by uncertainty, but the sources of risk in wildlife conservation are not the same as those to which agricultural production is subjected and the impacts on revenues may differ substantially among the two sources of income (Muchapondwa and Sterner Forthcoming). This paper builds on a study by Muchapondwa (Forthcoming) who focused on the theoretical arguments for risk management in agricultural production, by incorporating a more detailed empirical investigation.

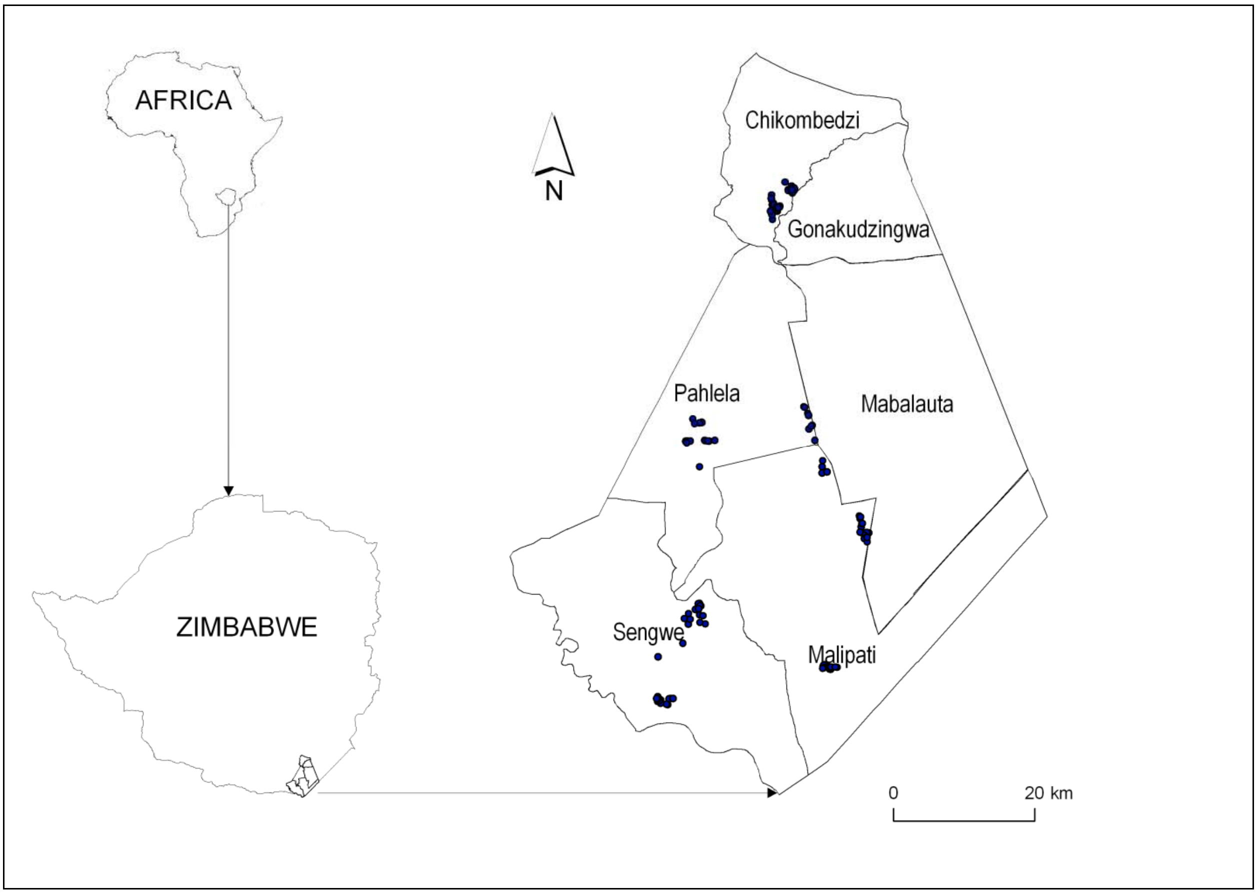
In this paper we study how wildlife income can reduce fluctuations in household incomes due to variability in rainfall in a typical savanna system, such as southeastern Zimbabwe. Our main research questions are formulated as follows: (1) What are the costs and benefits associated with agro-pastoral and wildlife systems in southeastern Zimbabwe? (2) How does income from agro-pastoral and wildlife systems vary with fluctuations in rainfall? and (3) To what extent does wildlife income reduce rainfall-induced fluctuations in household incomes?

Methods

Study Area

We focus on the case study area in southeastern Zimbabwe, where wards are sub-district units of local administration covering 150 to 1,000 km^2. The research was conducted in four wards (Chikombedzi, Pahlela, Sengwe and Malipati) within southeastern Zimbabwe (Figure 4.1), which are part of the Sengwe communal lands. Sengwe, Sangwe and Matibi 2 are the three main communal lands surrounding Gonarezhou National Park (the second largest national park in Zimbabwe). We did not consider Gonakudzingwa in our analysis since the area is under private ownership and the focus of our study is on wildlife benefits under communal set up. The case study area is characterized by low rainfall, shallow soils with low agricultural potential and high temperatures (about 39^oC in summer). Annual rainfall ranges between 300 to 600mm. The average rainfall recorded for this area based on 21 year rainfall data (from 1988 till 2008) from Mabalauta section of Gonarezhou National Park was 511 mm. Effective rainfall occurs from October to April, followed by a long dry season.

Figure 4.1: Study Area (dots indicate positions of sampled villages).



General framework

To apply the portfolio analysis we need to measure the economic or financial advantages of various activities and their volatility. This requires an economic analysis that focuses on the cost and benefits of particular production units of the activity in question using actual market prices (financial benefits), non-market values or opportunity costs (economic benefits) to value inputs, factors of production, and output (Murindagomo 1997). However, comparing peasant agro-pastoral systems by the value of their products is complicated by the fact that many intermediate products and services have no real market (Behnke 1985) and hence no observable market price. We included costs of crop protection in the field and costs of storage. We did not include costs of fertilizer because farmers in the study area do not use fertilisers as soil fertility is not a limiting factor. Labour costs have not been included because the opportunity cost of labour in the region is about zero due to a lack of other productive opportunities. Some factors cannot be easily assessed quantitatively. For instance, the role of livestock in the marriage contract and ceremonial activities cannot be assessed in terms of a quantitative comparison, but should not be ignored either (Scoones 1992). To deal with these complications we adopt the replacement cost method by Scoones (1992), which attempts to value production according to local economic criteria. The economic assessment uses a wide definition of productivity to include both off take (milk, meat) and live animal sales, while services provision (transport, draught) was taken to be an intermediary product.

Valuing wildlife using market prices is to some extent possible in southeastern Zimbabwe communal areas. Under the auspices of the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE), communities have created institutions which allow some hunting activities under strict conditions, making it possible for villagers to gain revenues from hunting. This is achieved through the use of services provided by safari operators, who sell hunting quota. In order to obtain information on the direct benefits from wildlife to the local communities, we have assessed the CAMPFIRE revenues given to the communities in two villages: Mutombo and Hlarweni, close to Malipati Safari Area (Figure 4.1). Under the CAMPFIRE programme (Child and Chitsike 2000) the state contracts out hunting concessions to safari operators for an agreed and renewable period. The safari operator buys the right to bring sport hunters and eco-tourists to their concession areas to hunt a set quota of animals, or to track, observe and photograph wildlife. Proceeds from these activities are given to the Rural District Councils, who then make payments to the communities after retaining a levy (38-46%) and also subtracting a percentage which goes to the CAMPFIRE association at national level as a levy (3-4%). The safari operator pays an annual fee (in hard currency) for the concession (about 30% of the total quota revenues) plus a trophy fee for each animal shot from an annual quota. The quota is the number of animals that annually can be hunted.

In southeastern Zimbabwe, the Department of Agriculture and Extension Services (AGRITEX) assesses crop production twice (mid-season and end of crop growing season) every year. The Veterinary Department also keeps records of cattle dipped per dip tank every two weeks in the dry season and every week in the wet season. These data together with the survey help us in analysing whether household income fluctuates with fluctuations in rainfall from one season to the next.

Data Collection

The research was done using both primary and secondary data sources. Primary sources of data involved a two-tier longitudinal survey of 144 households. The first survey was done in October 2008 during which the area had received below average rainfall (435 mm), i.e., after the 2007/2008 cropping season; and the second in July 2009,when the area had received above average rainfall (681 mm), after the 2008/2009 cropping season. A detailed description of the data collection is given in Appendix 1.

Data Analysis

From the two-tier longitudinal survey, descriptive statistics were used to explore household livelihood strategies and household financial indicators in PASW Statistic v17.0. Kruskal-Wallis tests (SPSS 2009) were used to investigate differences between villages. The survey data allowed calculating costs and benefits from the agro-pastoral system. CAMPFIRE records allowed calculating returns from wildlife systems. To calculate the potential contribution of each system to local people's livelihoods, a detailed comparative economic analysis of the two systems (agro-pastoral and wildlife) was done. This comparative economic analysis involved comparing tangible and intangible benefits and costs from the two production systems. For comparison we calculated the returns for each production system by subtracting total costs from gross benefits. For those tangible benefits and costs that do not have a market or thin market, shadow pricing was employed to express the underlying marginal opportunity cost of goods, services and factors of production.

Calculation of returns per household from wildlife system based on CAMPFIRE revenues was done using three scenarios. The first scenario ('Current scenario') shows communities getting 57% of the revenues, Rural District council (RDC) taking a levy of 39% of revenues, with another 4% going to the National CAMPFIRE association; this represents the current status. The second scenario ('1997 Scenario') shows the revenues which communities would get if the 2008 revenues were to be shared using the 1997 model when communities were getting 78% of the revenues, the RDC taking 20% and the CAMPFIRE association taking 2% of the revenues. The third scenario ('Market Scenario') was calculated assuming the distribution model of the 1997 scenario but based on market prices (data from safari operators) for the animals on the quota, assuming that the costs for hunting (i.e., fuel, food for clients, ammunition, labour, ivory registration) do not exceed 30% of total wildlife earnings.

A step function was fitted to data from individual wards where cattle population change (Δ_{cattle}) was plotted as a function of average NDVI or total annual rainfall and their lags in R v2.11.0 (Team 2010). This was done in order to test whether income from the agro-pastoral system varies with fluctuations in rainfall, particularly for analysis of livestock. We focused mainly on non-linear relationships between total rainfall and cattle population changes recorded at Pahlela and Malipati dip tanks, where Mutombo and Hlarweni villages dip their cattle respectively. Linear regression was used to estimate the relationship between seasonal rainfall (October to May) and average grain (maize and sorghum) yield from Mutombo and Hlarweni in PASW Statistic v 17.0. We also analysed the potential wildlife revenues based on the price of a species and the respective quota using the 2004 to 2009 quota levels allocated to Malipati safari area. This was done to investigate the response of wildlife revenues to changes in rainfall.

Finally, we investigated whether wildlife conservation is a useful asset for peasants to offset exposure to risk associated with agricultural production. First, we analysed the returns and risks of wildlife and agro-pastoral on their own. Secondly, we analysed a portfolio that includes both wildlife and agro-pastoral activities as elements or securities. Historical rainfall data, i.e., from 1988 to 2008 allowed calculation of probabilities of having a bad year (a year with below average rainfall) and a good year. In this study we objectively define drought as the mean rainfall minus one standard deviation or less following Prins (1996). There was a single drought (1991/92) during this period (Figure 4.2).

In order to match the analysis to the data from the two-tier survey, probabilities of a year with rainfall below the mean (bad year) and one in which rainfall was above the mean (good year) were considered as the two states of rainfall (Figure 4.2). The returns given the two states of rainfall were taken from the returns (mean for the two villages) reported in Tables 1 and 2. Since Table 2 gives wildlife returns for a bad year, potential wildlife returns from 2009 based on species on quota for that year were considered.

The data allowed for calculation of the expected outcome (returns) and the risk attached to the respective elements and the diversified portfolio, i.e., one which includes both wildlife and agro-pastoral activities as assets for the local people. This was done through calculation of expected returns, variances, standard deviations, coefficient of variation (CV), covariance and correlation coefficient for the two assets independently and combined (See appendix 4.2 for the calculations).

Results

Household socio-economic and agro-pastoral characteristics

Appendix 4.3 shows the main household and agro-pastoral characteristics for the eight villages in four wards in southeastern Zimbabwe. Statistically significant ($P < 0.05$) differences between villages for the numbers of cattle (chi-square = 24.004, d.f. = 7, $P < 0.001$), cattle sold (chi-square = 24.800, d.f. = 7, $P < 0.001$), number of donkeys (chi-square = 21.730, d.f. = 7, $P < 0.01$), number of work spans (chi-square = 21.297, d.f. = 7, $P < 0.01$), size of home field areas (chi-square = 31.120, d.f. = 7, $P < 0.0001$), maize and sorghum outputs (chi-square = 58,001, d.f. = 7, $P < 0.0001$) were found.

Mutombo, Hlarweni and Mandamwari are located within 20 km radius of the park boundary and they had lower numbers of cattle and donkeys, and also lower crop yields compared to the other villages. Furthermore, results show that villages that are found close to the park boundary had their food security category classified as transitory, meaning that households got food for seven to ten months in a year, implying a feed gap of between two to five months in a year.

Costs and Benefits of the agro-pastoral and wildlife systems

Returns from agro-pastoral systems were higher in both Mutombo and Hlarweni compared to returns from the wildlife system under the CAMPFIRE program (Tables 4.1 and 4.2). Further, it was observed that the annual household returns from the two systems were of similar magnitude for the two villages (US$299 in 2008 and US$1,177 in 2009 for Mutombo and US$446 in 2008 and US$1,081 in 2009 for Hlarweni from agro-pastoral vs. $56 for the two villages from wildlife). Returns from agro-pastoral activities were far much lower in 2008 when the area received below average annual rainfall. Returns from wildlife increased to US$177 under the market scenario.

Table 4.1 shows that households were getting a significant income from remittances, surpassing net benefits from agro-pastoral activities in a year with below average rainfall (2008), while the remittances were lower in a year classified as good rainfall year (2009).

Table 4.1: Gross benefits and costs (US $) and remittances (US $)per household for the years 2008 and 2009 from agro-pastoral activities for Mutombo (in Ward 13) and Hlarweni (in Ward 15) villages living close to Gonarezhou National park.

Village		Mutombo		Hlarweni	
Gross Benefits		2008	2009	2008	2009
Livestock	Meat plus Live animal sales	59	18	124	44
	Milk	217	831	265	226
Cropping					
	Maize + Sorghum	30	369	68	906
Total Benefits		306	1,218	457	1,176
Costs					
Livestock	Veterinary	0	0	0	0
	Dip Maintenance	4	4	4	4
Cropping	Crop and grain protection	3	36.9	6.8	90.6
Total Costs		7	40.9	10.8	94.6
Return		299	1,177	446	1,081
Remittances		432	384	621	352

Table 4.2: Returns per household in US$ for 2008 from wildlife system based on CAMPFIRE revenues generated from Malipati Safari and Malipati communal area quota under three scenarios. The first Scenario indicates the current distribution of revenues where communities get 57%, while the second scenario assumes that communities get 78% of the revenues (no remittances to Park) as used to happen in 1997. The third shows calculations done based on Market prices for the species on quota (see further the text).

		Scenarios		
	Revenue categories	Current Scenario	1997 Scenario	Market Scenario
1	Revenues from Malipati safari offtake	89,903	89,903	153,377
2	Remitted to Park (as owners of Land)	53,590	0	0
3	Revenues from Malipati Communal Area off take	56,493	56,493	56,493
4	Total revenues accrued at RDC (Trophy + Concession Fee) (1 - 2 + 3)	92,806	146,396	209,870
5	Levy (CAMPFIRE Association) (4 and 2 % of 4)	3,712	2,928	4,197
6	Rural District Council (RDC) (39 and 20 % of 4)	36,194	29,279	41,974
7	Community (57 and 78% of 4)	52,899	114,189	163,699
	Cost categories			
8	Livestock Predation	618	618	618
9	Crop Damage	936	936	936
10	Total costs (8 + 9)	1,554	1,554	1,554
11	Return (7 – 10)	51,345	114,189	162,145
12	Number of beneficiary households	915	915	915
13	Return / Household (11/12) (US$)	56	123	177

Note: Malipati Safari Area belongs to Gonarezhou National Park, but was leased to communities for CAMPFIRE activities hence some of the revenues go back to the owners of the land. This arrangement is different with other CAMPFIRE areas owned by the state through the RDC like Malipati communal area, no revenues would go to Park, and all will go to RDC on behalf of communities.

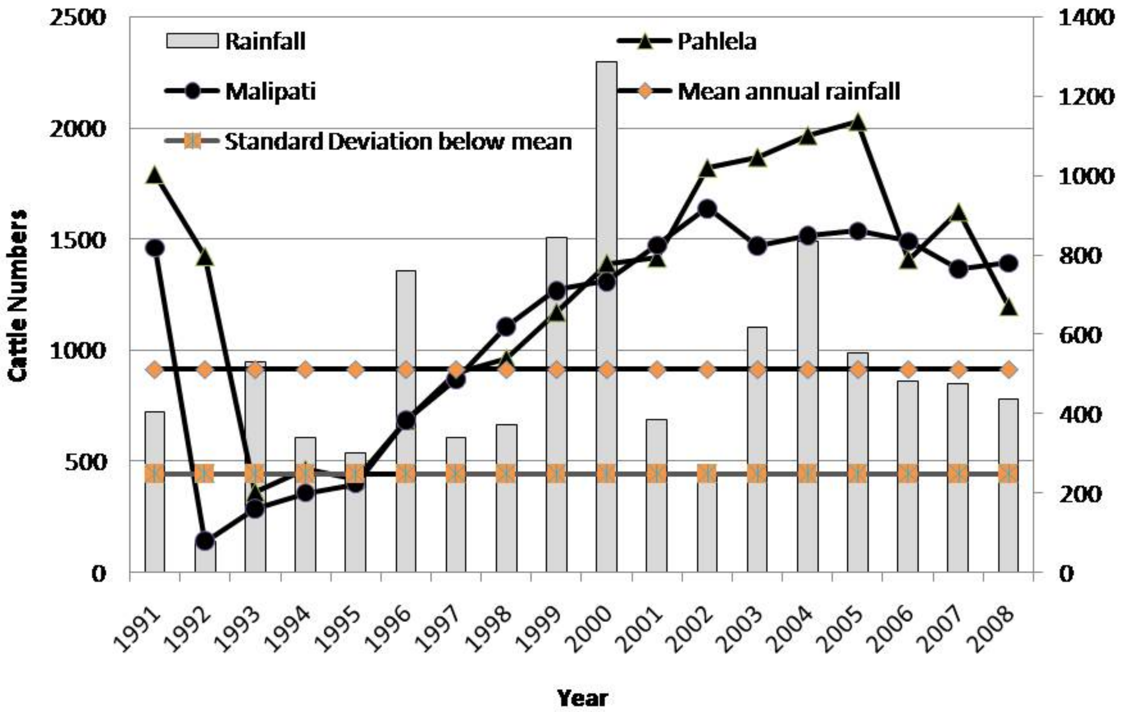
Changes in cattle numbers with variations in annual rainfall

The importance of livestock compared to cropping in southeastern Zimbabwe was shown by the contribution of the two land uses to total benefits from the agro-pastoral system. Table 4.1 shows that in 2008 income from sale of livestock products (meat and milk) and live animals contributes close to 90% and 85% of the total benefits from the agro-pastoral system in Mutombo and Hlarweni villages, respectively.

A step function involves estimation of three parameters: two averages and a threshold. When the two averages are significantly different from each other, it shows the existence of a threshold (Crawley, 2007). The presence of a threshold was confirmed in Pahlela (Threshold, $F_{1, 10} = 5.59$, $P = 0.0397$) and Malipati (Threshold, $F_{1, 10} = 18.05$, $P = 0.0017$) using NDVI as an explanatory factor. However, results from the same study also showed that green vegetation as measured by average NDVI can significantly ($P < 0.05$) be explained by total annual rainfall. This suggests that cattle changes were sensitive to annual fluctuations in rainfall via the direct impact of rainfall on annual forage availability.

Figure 4.2 shows the changes in numbers of cattle recorded at Pahlela and Malipati dip tanks in relation to annual rainfall. After the severe drought of 1991-1992, cattle numbers went down in both areas, as did the numbers of households owning cattle. These numbers dropped by more than 50%: from 112 in 1991 to 52 in 1993 for Pahlela and 109 in 1991 to 54 in 1993 for Malipati. After two consecutive years with rainfall below the mean (1994 to 1995), the numbers of animals started a general increase until 2002 for Malipati and 2005 for Pahlela. Figure 4.2 also shows that the drop in rainfall to below the long-term average (511mm) in 2001 and 2002 and years after 2004 was accompanied by a decline in cattle numbers.

Figure 4.2: Changes in numbers of cattle recorded at Malipati and Pahlela dip tanks with variations in annual rainfall.

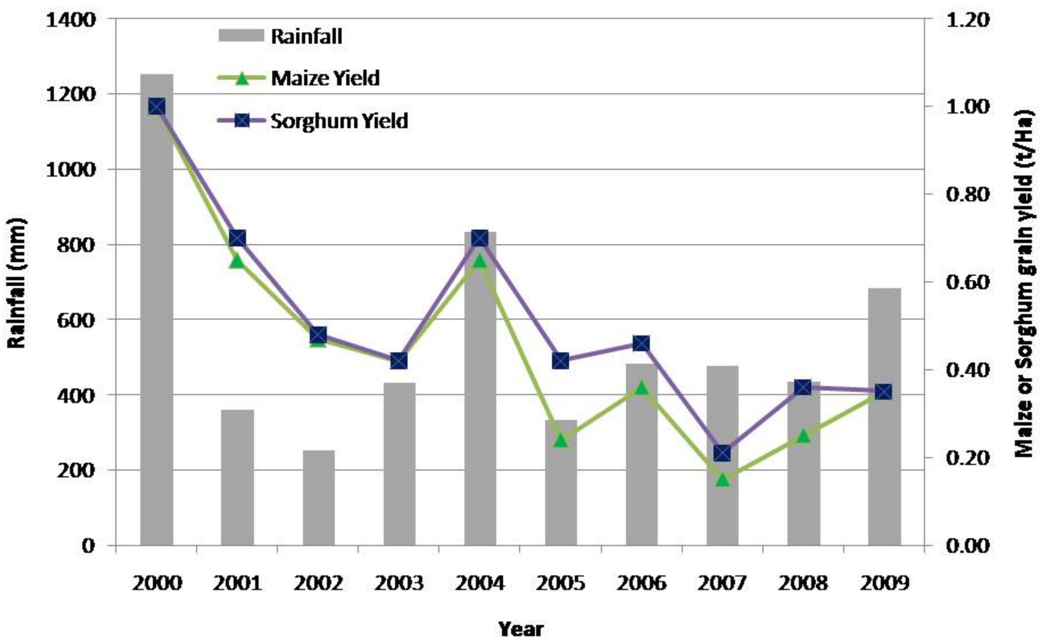


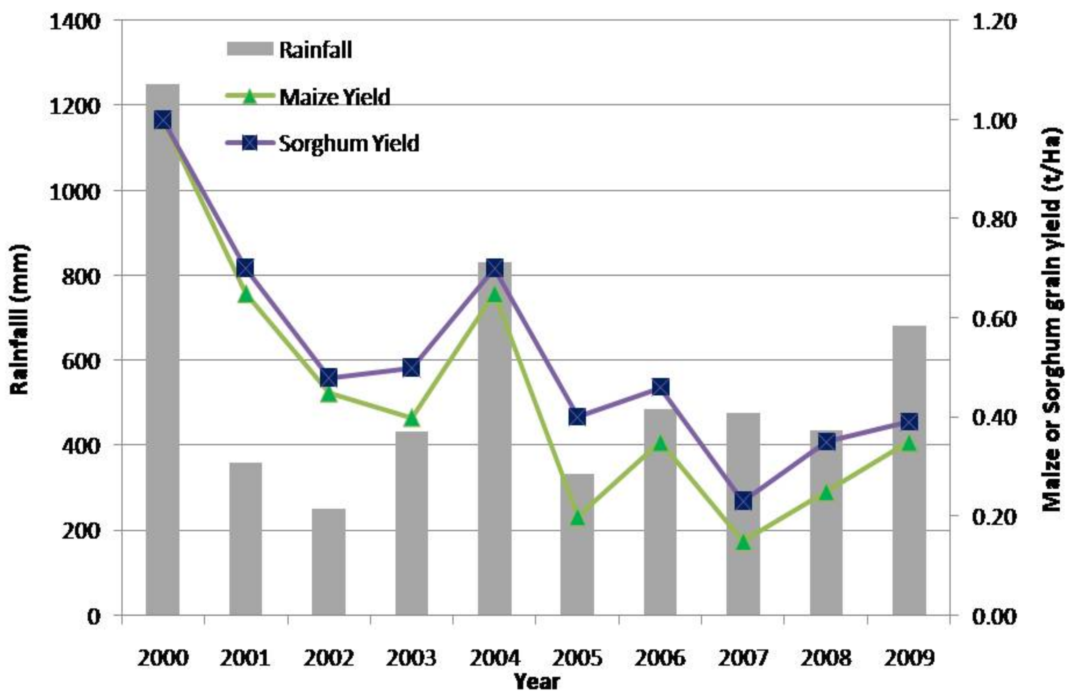
Variations in crop yields with rainfall fluctuations

In this study returns from agro-pastoral systems were calculated based on a 2 year survey, therefore there was need for us to establish if crop yields were varying from year to year due to fluctuations in rainfall using long term data (10 years). Results from a linear regression analysis showed that seasonal rainfall significantly (Adjusted R^2= 0.49, $F_{1,18}$ = 19.5, $P < 0.001$) explained changes in average maize grain yields and sorghum grain yields (Adjusted R^2= 0.49, $F_{1,18}$ = 17.004, $P < 0.001$) that were estimated from period 2000 to 2009 (Figure 4.3). The results generally show that maize and sorghum yields for both Mutombo and Hlarweni (Figure 4.3) decline with a decrease in rainfall. However, the lowest yields for both maize and sorghum in the two villages were not found in lowest rainfall years.

Highest maize and sorghum yields were recorded in year 2000, a year in which the area was hit by cyclone Eline. Two years after the cyclone, the area received the lowest amount of rainfall, therefore we expected lowest yields that year. Perhaps effects of the cyclone, such as raised water table and fertilization (bringing fertile deposits from upstream), caused the yield not to fall to the lowest levels in 2002. Lowest yields were recorded in 2007 due to low amounts of rainfall received in December 2006 and January 2007 resulting in mid season drought.

Figure 4.3. Changes in maize and sorghum yield (t/ha) with changes in seasonal rainfall (mm) from year 2000 to 2009 for Ward 13 (top graph) and Ward 15 (bottom graph) where Mutombo and Hlarweni villages are located.

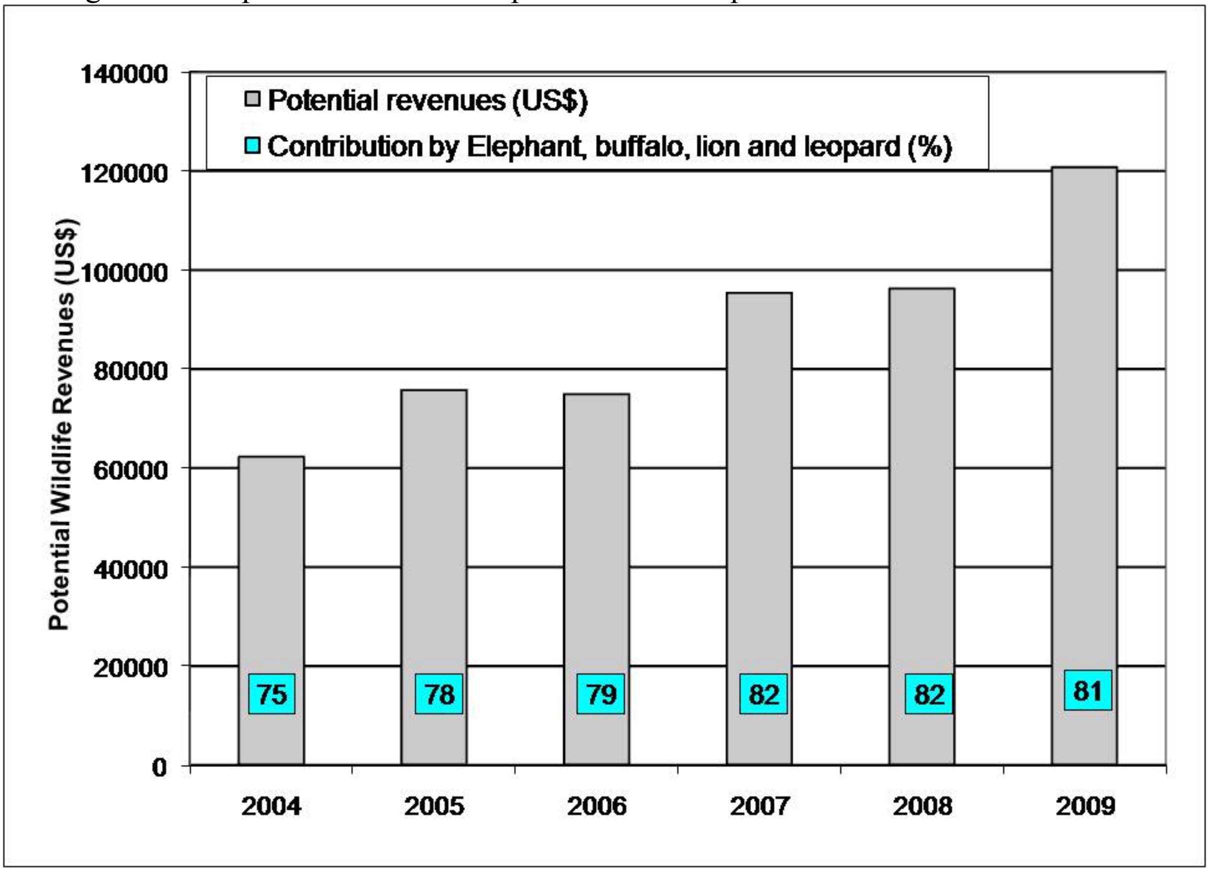




Changes in potential wildlife revenues with fluctuations in rainfall

Based on the returns from the two systems, our study shows that the income from wildlife systems is relatively small compared to the income that can be generated from agro-pastoral systems. Figure 4.4, however, shows an increase in potential revenues using 2004 to 2009 quota levels that were allocated in the Malipati Safari Area for CAMPFIRE activities. These are referred to as potential wildlife revenues because they are calculated based on the number of different species on the quota for that particular year. In many cases not all animals on quota will be killed, the number depends on the preferences of the hunter. Further, the increase in potential wildlife revenues during this period was against a background of a decline (below the long term average of 511 mm) in annual rainfall from 2006 to 2008, suggesting stability of wildlife income.

Figure 4.4: Potential wildlife revenues (US$) and the level of contribution (%) by the big mammal species based on Malipati Safari Area quota from 2004 to 2009.



Wildlife income as strategy for managing and coping with drought risk

By calculating the expected income of individual assets, it can be observed that the expected income from agriculture is higher than that from wildlife (660 vs. 194) (Table 4.3). However by diversifying, i.e., adding wildlife income to agricultural income, especially during bad rainfall years, the diversified portfolio gives a much higher expected income compared to the income from the individual assets. The coefficient of variation shows how risky the undertaking is. It gives a measure of the risk per unit of expected return (income) and it provides a more meaningful basis for comparison when the expected returns (income) on the two alternatives are not the same (Damodaran 1998; Reilly and Brown 1998). It can be observed that agriculture is a risky undertaking compared to wildlife, because the coefficient of variation is 0.56 vs. 0.49 (Table 4.3).

Diversifying using wildlife results in a low coefficient of variation compared to agriculture alone (0.46 vs. 0.56). Therefore, the diversified portfolio results in a higher expected return which is less risky than agriculture alone. The power of diversification can be measured using covariance and correlation. Covariance is a measure of how much two risky assets move in tandem, whereas correlation coefficient (r) is a scale with a value between -1 (perfect negative correlation) and +1 (perfect positive correlation) (Damodaran 1998; Reilly and Brown 1998).

Table 3 shows that revenues from agriculture and wildlife are positively correlated (0.4). An investor would prefer assets with negative correlation to those with positive correlation in order to reduce the risk. A weak correlation in this study already allows for exploiting much risk reduction. In the same sense, rural farmers would prefer a negative correlation between agriculture and wildlife for wildlife to provide farmers with a better hedge asset during bad years. However it is clear that the mixed portfolio is less risky than agriculture alone, because the revenues from wildlife are less volatile.

Table 4.4: Performance in terms of expected incomes and risk attached to different assets (agriculture and wildlife) on their own and as a diversified portfolio.

	Different assets on their own				**Diversified portfolio**
Assets/securities	Agriculture		Wildlife		Agriculture & Wildlife
States of rainfall	Good	Bad	Good	Bad	
Probability	0.38	0.62	0.38	0.62	
Return (US $)	1,129.00	372.50	222.00	177.00	
Expected Income (US $)	660.00		194.00		854.00
Standard Deviation (σ)	367.00		94.60		389.00
Coefficient of variation (CV)	0.56		0.49		0.46
Correlation coefficient (r)					0.40

Discussion

The results demonstrate the role of wildlife income in reducing rainfall-induced fluctuations in household income and the extent to which wildlife income potentially contributes to local people's livelihoods. Analysis of returns from the agro-pastoral system using survey data for 2008 and 2009 has shown that household incomes fluctuate with variations in annual rainfall. Furthermore, our results have established the higher contribution by livestock income, i.e., from sale of livestock products (meat and milk) and live animals compared to cropping. This agrees with findings from other studies that have shown that households keep livestock for the multiple benefits they provide (Shackleton *et al.* 2001; Dovie *et al.* 2006).

Figure 4.2 indicates the fluctuations that take place in cattle income, which also affects household income as drought causes other households to lose their cattle. Rainfall-induced fluctuations in livestock income lead to household income fluctuations in southeastern Zimbabwe from one year to another. For Mutombo village the contribution of livestock income to total agro-pastoral income was high for both years considered bad (2008) or good (2009), while for Hlarweni village the contribution of livestock income to total agro-pastoral income was higher in a bad (2008) year and lower in a good (2009) rainfall year. This may be a reflection of the presence of an irrigation scheme in Hlarweni, where farmers would produce crops rather than livestock in a good year. In areas where there are no irrigation schemes, as in Mutombo village, livestock contribution to household income is significant even in a good rainfall year. The increase in livestock numbers (Figure 4.2) in the area suggests that income from agriculture may be unsustainable.

CAMPFIRE was established in the late 1980s with the aim of integrating biodiversity conservation and rural development (Child and Chitsike 2000; Munthali 2007; Murphree 2009). Specifically it promised to boost household incomes through the commercial use of wildlife resources in communal lands (Cumming 2005). However, our results suggest that returns from CAMPFIRE are small compared to income from the agro-pastoral system, making it unlikely that they make a substantial contribution to livelihoods. Table 4.1 and 4.2 shows that even if communities were given a greater percentage (equal to what they used to get in the 1990s before the economic downturn in Zimbabwe) the returns still remain small (US$123 vs. US$ 299 in 2008 and $1,177 in 2009 for Mutombo, US$ 446 in 2008 and US$ 1,081 in 2009 for Hlarweni). Our results confirm the outcome of an analysis of CAMPFIRE revenues' contribution to household income in nearby Beitbridge district, which clearly showed that CAMPFIRE revenues made a negligible contribution to household income in southeastern Zimbabwe (Cumming 2005). The economic downturn that was experienced in Zimbabwe may explain the low wildlife revenues that households and communities receive.

Table 4.2 shows scenario 2 being much better than scenario 1,perhaps indicating that Rural district council and the Wildlife Authorities were getting a bigger fraction of the wildlife revenues at the expense of rural communities, since the wildlife income was one of the few income sources due to the harsh economic outlook. These findings are consistent with those by Murphree (2009) who stated that the long market chains result in communities receiving only a small and inadequate portion of the net revenues. Additionally, Rural District Councils still retain excessive control, especially revenue retention, resulting in the intended primary beneficiaries being severely disadvantaged (Taylor 2009). Furthermore, these results suggest that if proper pricing of the wildlife resource is done and devolution to communities is completed, as indicated by scenario 3 (Table 4.3), households may realise better incomes from wildlife. The implementation of the market scenario, however, may not be feasible due to challenges that communities may face namely high costs of entering into safari hunting and management, lack of skills and knowledge by communities of the wildlife market chain at both national and international levels.

Finally, we were interested to know the potential contribution of wildlife income to buffer households against income fluctuations caused by variations in annual rainfall. Portfolio theory (Markowitz 1952; 1959) was used to investigate how the addition of wildlife as an asset to the usual activities of agricultural production of rural farmers could be used to diversify and subsequently to reduce risk faced by rural farmers (Koellner and Schmitz 2006). Findings from this study have shown that by exploiting a portfolio that includes wildlife and agriculture, farmers can reduce rainfall-related risk and also improve on the benefits they get (Table 4.3). This is in agreement with the contention that wildlife conservation is potentially a hedge asset against rainfall-related risk, conveniently at the disposal of rural farmers (Muchapondwa and Sterner Forthcoming). Even though wildlife income is small, it has been shown (Table 4.3) that it is less risky than agriculture and it also forms an important hedge asset to rural farmers during years with low rainfall. Thus rural farmers and conservation managers should not look at the development of individual assets, but at the development of the complete portfolio.

The power of diversification can be measured using covariance and correlation (Damodaran 1998; Reilly and Brown 1998). The investor would be better off in terms of risk by combining assets whose returns are inversely related (Reilly and Brown 1998). Under such cases, the risks of the individual elements cancel each other out as a result of the decrease of the return of one asset being offset by the increase of the return of the other asset. The relationship between the variations in return on the two assets is important because it determines the risk of the complete portfolio (Figge 2004). Results have, however, shown a positive correlation coefficient between agriculture and wildlife (Table 4.3). This finding is not surprising as low rainfall affects both agricultural activities and wildlife, particularly availability of forage or browse. The critical point, of course, is that the correlation coefficient is 0.4 only, thus allowing ample scope for compensatory effects to take place because the impacts of rainfall-related risk on the two enterprises differ, with agricultural production being more vulnerable. The coefficient of variation of agriculture shows that it is more risky than wildlife (Table 4.3). Theory predicts that systems with many species can buffer the disturbances better than systems with fewer species, because the probability is greater that some species will be able to maintain a certain level of ecosystem service, even though others may fail to function (Yachi and Loreau 1999; Tilman *et al.* 2005).

The diversification effect does not come to bear, however, if the assets follow a completely parallel variation i.e., when agriculture provides more benefit, wildlife provides more benefit too. Risks will not cancel each other out and thus not be reduced by combining the elements in a portfolio (Figge 2004). Findings from our study did not show perfect positive correlation and a rather low correlation coefficient (of only 0.4), hence diversification can be possible. Under extreme drought, however, all assets of the portfolio will be exposed to the same risk, termed systemic risk, and these types of risks cannot be diversified.

Implications for conservation

We conclude that people in southeastern Zimbabwe earn a substantial part of their household income from an agro-pastoral system compared to a wildlife system, with livestock income being higher than income from cropping. In dry years agro-pastoral income declines due to livestock losses and lower crop yields. These income losses during dry years are compensated by remittances to a large extent and by wildlife income as these revenues are less sensitive to drought.

Revenues from wildlife have some potential to reduce household income fluctuations due to drought, but only to a limited extent. We argue that if wildlife is organised on a more commercial basis as illustrated by the market scenario, then the net revenues could be increased due to a more efficient and equitable exploitation of the resource potential. Therefore a more substantial role can be played in reducing variations in incomes. The current CAMPFIRE approach only contributes to a very limited extent to a stable income for rural households. To our knowledge, this is one of the few studies that empirically tested the applicability of portfolio theory to biodiversity related issues. The portfolio theory framework shows that by exploiting different resources of income, rural farmers can realize a more constant household income than by depending on one resource only, because it is rare for the whole portfolio to be affected by risk. This finding could help efforts to conserve wildlife while also improving welfare of local people.

Acknowledgements

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Appendix 4.1: Detailed description of data collection

The household formed the basic sampling unit in this study. We adopted and used the definition of a household to mean a group of one or more persons living together under the same roof or in several rooms within the same dwelling, and eating from the same pot or making common provision of food and other living arrangements (Kideghesho and Mtoni, 2008). The sampling consisted of two villages in each ward surrounding Mabalauta section of Gonarezhou National park. Survey villages and households were selected through a multi-stage sampling procedure. Eight villages (two from each ward) were selected from the wards, resulting in stratified random sampling based on preliminary data from key informants. Stratification was based on population size, number of households, distance from the park boundary, spatial extent of the village, and most common household livelihood activities in the village. The actual questionnaire surveys involved respondents from a randomly selected sample of 156 households in 2008 (ensuring that more than 30% of the total households in each village were covered) drawn from the village registers. In 2009 the survey covered 144 of the 156 households interviewed in 2008. These 144 households are the same households in 2008 and 2009, in order to capture changes that happen between seasons. Extension workers and village heads helped in visiting and introducing the team of researchers to each respondent and in some cases translating where the respondent preferred speaking in the local language, Shangaan. Household information was gathered on cropping, livestock holdings, numbers of livestock, their classes, age categories, offtake, monetary benefits, and other intangible benefits from livestock as well as the costs incurred in keeping livestock and cropping. The survey also covered crop production interrelationships (draught power, manure and stover from crops), perception of wildlife contribution to household income, and current and past community management systems of animals and natural resources**.** Quantification of livestock and crop predation costs by wildlife was done as part of work reported by Kuvawoga (2008).

Secondary data sources used in this study include dip tank records (1991 to 2008)of livestock numbers, their age categories, and numbers moved in and out of each ward, that were obtained from official statistics by the veterinary department (DVTS 2008). Dip tank counts also showed numbers of animals born, sold and the numbers that died for each particular year. We used dip tank data since cattle dipping is compulsory and also enforced in Zimbabwe as part of a highly controlled cattle husbandry system nation-wide. Data on annual crop yield estimates from southeastern Zimbabwe were obtained from the Department of Agriculture and Extension Services (AGRITEX 2009). The crop estimates were obtained through the rural food security assessments by the Zimbabwe Vulnerability Assessment Committee from 2000 to 2009. The average annual grain yields were estimated at the end of the cropping season by averaging yields for 30 farmers in each ward. Other secondary sources of data included data on actual CAMPFIRE revenues generated, payments made to communities and percentages retained by the Rural District Council (Chiredzi RDC Unpublished). The data were obtained from the Rural District Council records. Rural District Council records were also secondary sources for wildlife animal quotas and the actual offtake for the years 2000 to 2009 for Malipati Safari area (hunting area) and Malipati communal area (Appendix 4.4). The actual offtake would sometimes differ from the quota, particularly for large herbivores like elephants (*Loxodonta africana*), due to problem animal control. Animals not on quota would eventually get killed when they caused crop damage or other problems in surrounding communities. Additionally, Rural District Council records provided information on actual numbers of wildlife animals hunted for trophy by category and their respective revenue values for the same period. These data sources were used to calculate wildlife contribution to household income. Household incomes were calculated for two villages: Mutombo (located in Pahlela) and Hlarweni (located in Malipati) because households from these two wards benefit from CAMPFIRE revenues from Malipati communal and Malipati Safari Area (a 154 km^2 state-owned hunting area under the Department of National Parks and Wildlife Management Authority which has been leased to the community). Further, wildlife data for wildlife animal estimates in the whole park were taken from aerial survey reports (Dunham *et al.* 2007; 2010) that show roughly the densities of wildlife species in the park and the Safari area. For the wildlife densities in the communal area no data were available, but basically in the communal areas the densities are low.

Appendix 4.2: Formulas and calculations
Calculation of Expected income ($E(I)$) of the different assets:

There are two possible outcomes of rainfall: bad and good rainfall years, and two land uses (assets): Agriculture and wildlife. The probabilities refer to different levels of rainfall: p_g = 0.38 for a good year and p_b= 0.62 for a bad year.

Expected Income ($E(I)$) for agriculture and wildlife:

E (I) for agriculture =0 .38 x 1,129 +.62 x 372 = 660;

E (I) for wildlife = 0.38 x 222 + .62 x 177 = 194.

Expected income ($E(I)$) for the diversified portfolio:

E (I) for diversified portfolio (agriculture + wildlife) = E (I) for agriculture + E (I) for wildlife =854

Variance (σ^2) of the expected income:

$$\sigma^2 = P_g[R_g - E(I)]^2 + P_b[R_b - E(I)]^2 \quad \text{.......} \quad (1)$$

Variance (σ^2) for agriculture and wildlife:

Agriculture: σ^2 = .38 $(1{,}129\text{-}660)^2$ + .62 $(372.5\text{-}660)^2$

Wildlife: σ^2 = .38 $(222\text{-}194)^2$ + 0.62 $(177\text{-}194)^2$

Variance σ^2 for diversified portfolio = .38 $((1{,}129+222)\text{-}854)^2$ + .62 $((372.5+177)\text{-}854)^2$

Standard deviation (σ) is calculated as follows:

$$\sigma = \sqrt{\sigma^2} \quad (2)$$

Coefficient of variation (CV) is calculated as below:

$$CV = \frac{\sigma}{E(I)} \quad (3)$$

Covariance of agriculture (a) and wildlife (w) is calculated as follows:

$$Cov_{aw} = \sum p\left[\left(R_a - E(I_a)\right)\left(R_w - E(I_w)\right)\right] \quad (4)$$

Correlation coefficient is calculated as follows:

$$r_{aw} = \frac{Cov_{aw}}{\sigma_a \sigma_w} \quad (5)$$

Appendix 4.3: Mean household and agro pastoral characteristics (standard deviations in parenthesis) for two villages in each ward

Ward	Chikombedzi		Pahlela		Sengwe		Malipati		
Village	Haisa	Ponyoka	Mutombo	Shavani	Chali	Mudhanisi	Hlarweni	Mandamwari	All
N	21	17	16	20	16	16	19	19	144
Household size	10.00 (5.00)	12.00 (8.00)	9.00 (5.00)	7.00 (4.00)	7.00 (2.00)	8.00 (4.00)	7.00 (3.00)	7.00 (2.00)	8.00 (5.00)
Number of cattle**	7.00 (8.00)	8.00 (17.00)	1.00 (2.00)	11.00 (12.00)	10.00 (14.00)	4.00 (5.00)	3.00 (4.00)	1.00 (2.00)	6.00 (10.00)
Number of cattle sold in 2008**	1.10 (2.30)	0.50 (1.30)	0.13 (0.34)	1.30 (1.80)	0.60 (1.00)	0.56 (1.00)	0.40 (0.80)	0.10 (0.20)	0.60 (1.40)
Number of sheep and goats	7.00 (9.00)	6.00 (7.00)	4.10 (7.00)	7.00 (7.00)	4.00 (8.00)	3.00 (4.00)	8.00 (9.20)	4.00 (5.00)	6.00 (7.00)
Number of sheep and goats sold	0.50 (1.20)	0.30 (0.60)	0.50 (1.00)	0.90 (1.20)	0.40 (1.10)	0.40 (0.80)	0.80 (1.60)	0.30 (0.70)	0.50 (1.10)
Number of donkeys**	0.70 (1.30)	0.40 (1.00)	0.20 (0.80)	1.60 (2.00)	0.90 (2.60)	0.13 (0.50)	0.20 (0.70)	0.40 (1.00)	0.60 (1.00)
Number of work span**	1.00 (0.90)	0.90 (1.00)	0.31 (0.60)	1.40 (1.70)	0.90 (0.90)	0.69 (0.70)	0.50 (0.60)	0.30 (0.50)	0.80 (1.00)
Size of outfield arable area (Ha)	3.70 (4.20)	4.00 (3.00)	2.80 (3.50)	3.00 (2.40)	2.00 (1.90)	4.00 (4.60)	2.40 (2.00)	2.00 (1.40)	2.80 (3.10)
Size of home field arable area (Ha)**	2.00 (4.00)	0	0.70 (1.90)	0.10 (0.40)	0	0.40 (0.80)	0.80 (1.50)	0.08 (0.30)	0.50(1.90)
Maize output 2008 (t)***	0.40 (1.00)	0.50 (1.40)	0.04 (0.04)	0.20 (0.20)	0.20 (0.20)	0.08 (0.09)	0.10 (0.10)	0.05 (0.04)	0.19 (0.60)
Sorghum output 2008 (t)***	0.50 (1.30)	0.50 (1.80)	0.06 (0.04)	0.10 (0.10)	0.01 (0.01)	0.10 (0.14)	0.10 (0.14)	0.03 (0.03)	0.20 (0.80)
Maize output 2009 (t)***	4.50 (10.00)	4.70 (13.00)	0.38 (0.30)	2.10 (2.10)	2.20 (2.10)	0.10 (0.10)	1.20 (1.30)	0.62 (0.66)	2.10 (6.20)
Sorghum output 2009 (t)***	5.30 (13.00)	6.10 (19.00)	0.67 (0.50)	1.30 (1.40)	0.05 (0.10)	1.10 (1.40)	1.40 (1.70)	0.43 (0.37)	2.10 (8.30)
Number of cattle sold in 2009	0.50 (1.00)	0.50 (1.50)	0.10 (0.25)	0.60 (0.80)	0.40 (0.70)	0.50 (0.90)	0.20 (0.40)	0.16 (0.50)	0.10 (0.80)
Food security category	enduring	enduring	Transitory	enduring	enduring	transitory	transitory	transitory	

*indicates significant differences between villages at $P < 0.01$ level or better, based on Kruskal-Wallis test.
**indicates significant differences between villages at$P< 0.001$ level or better, based on Kruskal-Wallis test.
***indicates significant differences between villages at$P< 0.0001$ level or better, based on Kruskal-Wallis test.

Appendix 4.4: Wildlife species quota and offtake and their respective prices from Malipati safari area and Malipati communal area.

	Malipati safari Quota 2004/09						Park Price (US$)	RDC Price (US$)	Market Price (US$)	Malipati safari Offtake 2004/08					Malipati Communal Offtake
Species	2004	2005	2006	2007	2008	2009	2004/09	2004/09	2008	2004	2005	2006	2007	2008	2008
Baboon	10	10	10	10	10	10	5	25	300	2	3	0	1	4	10
Buffalo (M)	2	10	10	10	10	10	1,200	1,500	8,000	2	10	10	10	10	10
Bush buck	0	2	2	2	2	2	400	460	1,075	0	2	2	2	2	0
Crocodile	2	2	1	1	1	2	1,000	1,400	3,000	2	2	1	1	1	2
Duiker	0	2	2	2	2	2	90	100	475	0	2	3	2	2	0
Eland	1	1	1	1	1	1	900	1,035	2,750	0	0	0	0	0	4
Elephant	3	3	3	5	5	6	8,500	9,775	18,000	3	7	3	5	5	3
Elephant (TL)	1	1	1	1	1	1	2,000	2,300	5,950	0	0	0	0	0	0
Francolin	25	25	25	25	25	25	4	4.60	5	0	0	0	0	0	0
Guinea Fowl	25	25	25	25	25	25	4	4.60	5	0	0	0	0	2	0
Impala (F)	10	10	10	10	10	10	40	50	100	10	10	3	5	10	0
Impala (M)	25	25	25	25	25	25	80	100	300	25	25	19	25	25	6
Klipspringer	0	1	1	1	1	1	250	300	600	0	0	1	0	1	0
Kudu (F)	1	1	1	1	1	1	300	330	500	2	1	1	1	0	0
Kudu (M)	5	5	5	5	5	5	600	660	1,000	2	4	2	5	5	0
Leopard (M)	3	3	3	3	3	4	2,500	2,800	3,500	3	1	2	2	3	0
Lion	1	1	1	1	1	2	3,000	3,800	6,500	0	0	0	0	1	0
Nyala	0	0	0	1	2	4	700	875	2,850	0	1	0	1	1	0
Pigeons/Doves	25	25	25	25	25	25	4	4.60	5	0	0	0	0	0	50
Hyena	1	1	1	1	1	1	50	62	450	0	1	1	0	2	10
Sand grouse	25	25	25	25	25	25	3	3.45	5	0	0	0	0	0	0
Water buck	3	3	3	3	3	5	850	1,000	2,000	3	3	3	3	3	0
Zebra	2	2	3	3	3	3	550	600	950	2	2	3	3	1	5
Steenbok	0	0	0	0	0	0	0	110	0	0	0	0	0	0	4
Porcupine	0	0	0	0	0	0	0	22	0	0	0	0	0	0	4

Key: Quota shows the number of animals that the safari company was allowed to hunt/kill that year; Offtake are the animals that were actually killed; M: Male; F: Female; TL: Tusk less

Chapter 5

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.

Introduction

Over the last few decades, establishment of protected areas has constituted the principal system supporting conservation strategies (Ruiz-Labourdette *et al.* 2010). Protected areas are needed in order to safeguard biological diversity (McNeely 1994). 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 (Schulz and Skonhoft 1996; Skonhoft 2007). Therefore game parks coexist with people in tightly coupled, fractious and uneasy relationships (Nagendra *et al.* 2010), 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 (Skonhoft 2007; Kiss 1990; Johannesen 2005). 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 (Kiss 1990; Johannesen 2005; Johannesen 2007).

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 (Hulme and Murphree 2001). 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 (Mbaiwa 2005; Stronza and Gordillo 2008) 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 Skonhoft 2007).

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 (Muchapondwa 2003; Stage 2010). Under such conditions wildlife utilization becomes a highly competitive form of land use (Taylor 2009) 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 (Taylor 2009). 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 (2009) 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 (Hudson *et al*. 2002), 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 (Taylor and Martin 1987; Gordon 2009), but bovine Tuberculosis entered the buffalo population in Kruger National Park through contamination by cattle (Renwick *et al.* 2007). 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 (Perrings 2000). 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 (WECD 1987), 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 (1992) 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 (Daily 1997; Higgins *et al.* 2007). 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 (Tietenberg 2000). Management options that provide optimized benefits under conditions of highly fluctuating rainfall are therefore needed.

Recent work by Hein (2005; 2010) 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 (Holling and Gunderson, 2002). Additionally, Schulz and Skonhoft (1996), Skonhoft and Solstad (1998), and Skonhoft (2007) 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 analyse 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 analyse 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 (Fafchamps *et al.* 1998; Owens *et al.* 2003).

The objective of this paper is to analyse 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 (Figure 5.1). 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 km^2 in total) in Chiredzi district: Chikombedzi (ward 11: 358 km^2), Gonakudzingwa (ward 12: 306 km^2), Pahlela/Makanani (ward 13: 648 km^2), Sengwe (ward 14: 813 km^2) and Malipati (ward 15: 953 km^2). Wards are sub-district units of local administration covering 150 to 1,000 km^2. 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^{o}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.

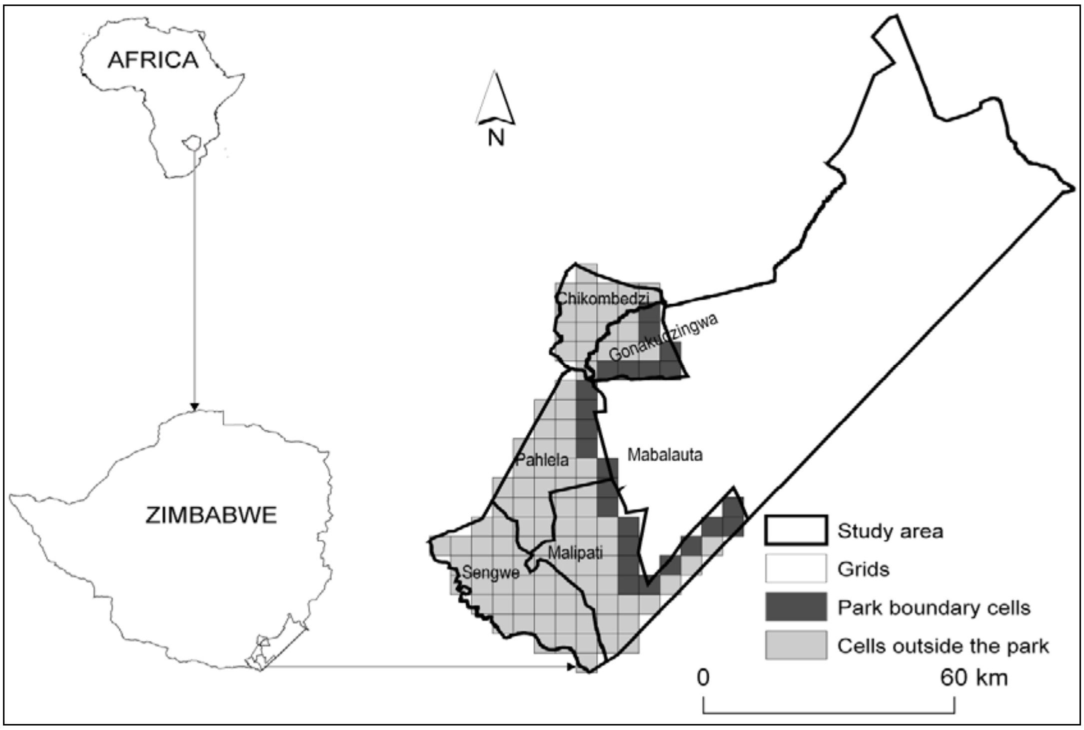

Figure 5.1: South-eastern lowveld study area in Masvingo Province of Zimbabwe.

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 (Cumming 2005; CSO 2002), with people surviving on less than US$1 per day (Cumming 2005). Densities of 29 people per km^2 have been reported for southeastern Zimbabwe compared to 3 people per km^{2}and 2 people per km^{2}for Botswana and Namibia (Heitkönig and Prins 2009), 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 (Mwakiwa 2011). 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 good, 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:

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

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

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

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:

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

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:

$$I_{ry} = \sum_c \sum_l A_{lc} b_{lcry} \sigma_l \quad \forall r, y \tag{4}$$

where A_{lc} denotes the total area (km^2) of land use l in plot c; b_{lcry} denotes the benefits of land use l in plot c, in rainfall sequence r 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:

$$\sum_l A_{lc} \leq a_c \qquad \forall_c \tag{5}$$

Land use benefits (b_{lcry}) 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:

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

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

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

where V_l, denotes variable costs per unit of output for each land use l and fl 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:

$$q_{lc} = \max\{0, \beta_l - \tau_l d_c\} \qquad \forall l, c \tag{8}$$

where β_l is the base probability of predation, τ_l 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 **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_{clry} 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 (de Leeuw and Tothill 1993). Similar to Hein (2010) 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_{clry} 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:

$$\theta_{clry} = \frac{k_c (1 - o_l)^{Z_c} \gamma_1}{\lambda_l} \sum_a \upsilon_{ary} \mu_{al} \qquad \forall c, l, r, y \tag{11}$$

where k_c denotes the carrying capacity of plot c (tonnesdry matter perkm^2); 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; υ_{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;

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

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:

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

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:

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

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:

$$M_{clry} = \min\left\{\psi, \max\left\{0, k_{clr,y-1} - S_{clr,y-1}(1+g_l)(1-q_{lc})(1-\varpi_l)\right\}\right\} \ \forall c,l,r,y$$
(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 3^8 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 (Taylor and Martin 1987; Child 2009). 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 (2008). Carrying capacity K_c (tonnes dry matter perkm^2) 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 (2010), 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_l 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 (Topps and Oliver 1993). 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_l, costs φ_l per unit of output for each land use were estimated based on averages prices, variable v_l 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_l were estimated from Rural District Council CAMPFIRE records (1996 to 2009). See Appendix 5.1 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 5.2a to 5.2c. 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 5.3). 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 (Figures 5.2a to 5.2c). Figures 5.2a to 5.2c 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 5.2a to 5.2c 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 5.2a to 5.2c) 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 5.2a to 5.2c). Both expected income and lowest income are higher under a lower drought probability (Figure 5.2a to 5.2c). 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 5.3). Figure 5.3 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).

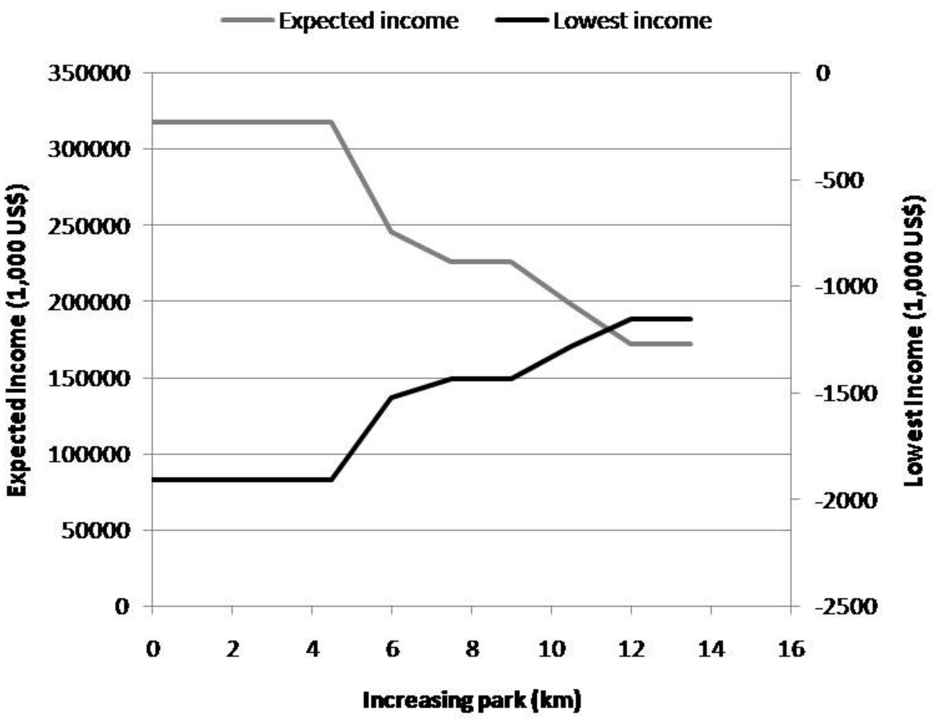

Figure 5.2a: Increasing park leads to a decrease of the expected income (grey line) and an increase in lowest income (black line) when probability of drought is 0.20.

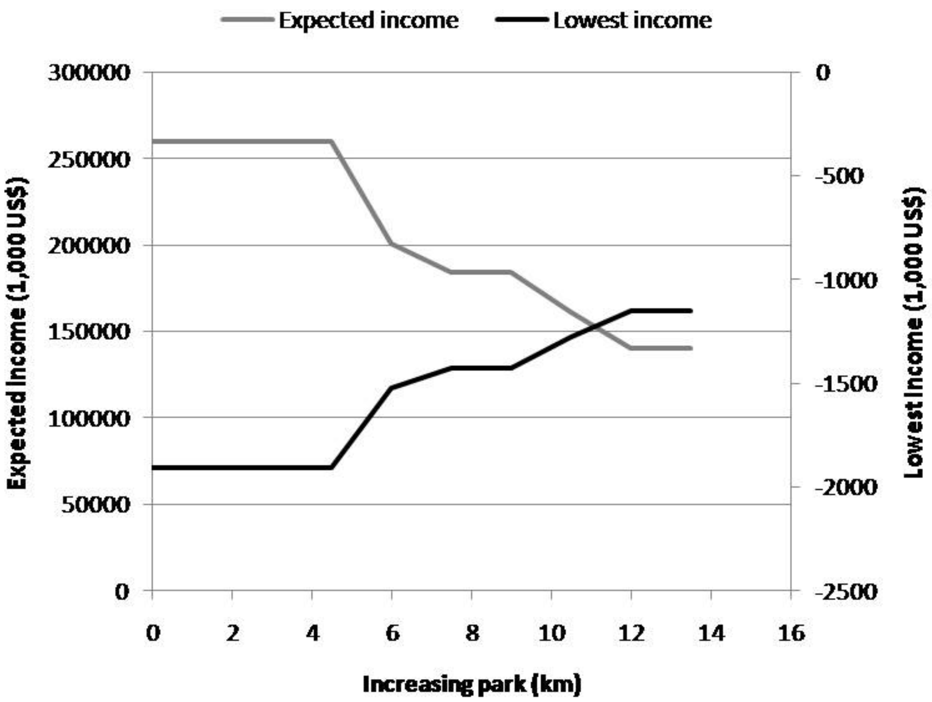

Figure 5.2b: Increasing park leads to a decrease of the expected income (grey line) and an increase in lowest income (black line) when probability of drought is 0.40. Note that as compared to Fig 5.2a the form of the curves does not change but that expected income drops considerably.

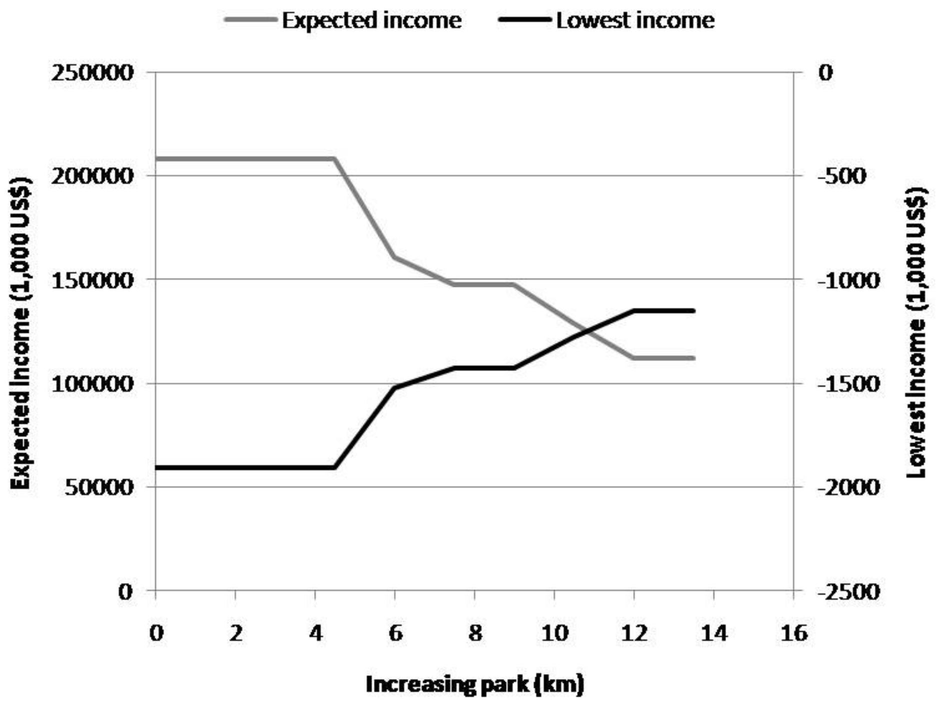

Figure 5.2c: Increasing park leads to a decrease of the expected income (grey line) and an increase in lowest income (black line) when probability of drought is 0.60.

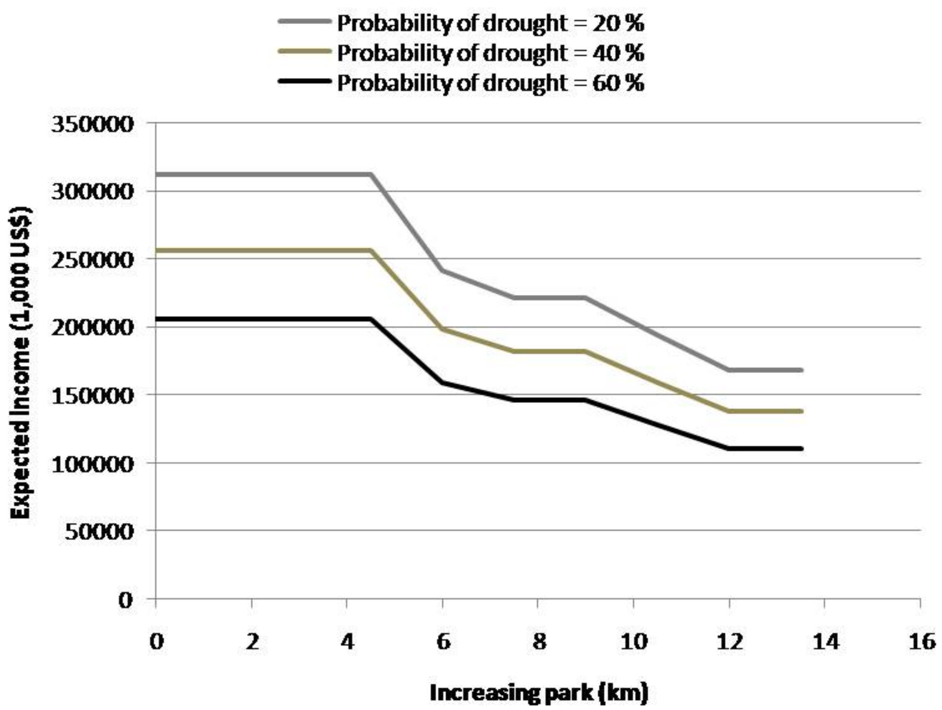

Figure 5.3: Increasing Park leads to a decrease in expected income when probability of drought is 0.20, 0.40 and 0.60. In these model runs income in any year must be positive, which causes the lowest income to be zero.

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 5.4a 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 5.4a). 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 5.2a to 5.2c. Addition of a lower bound of zero results in wildlife substituting for some but not all irrigated agriculture (Figure 5.4b). However, irrigated agriculture shifts to a few plots previously used for livestock (Figure 5.4b). Results also showed that an increase in impact of drought did not result in a different land use allocation (Figure 5.4a and 5.4b).

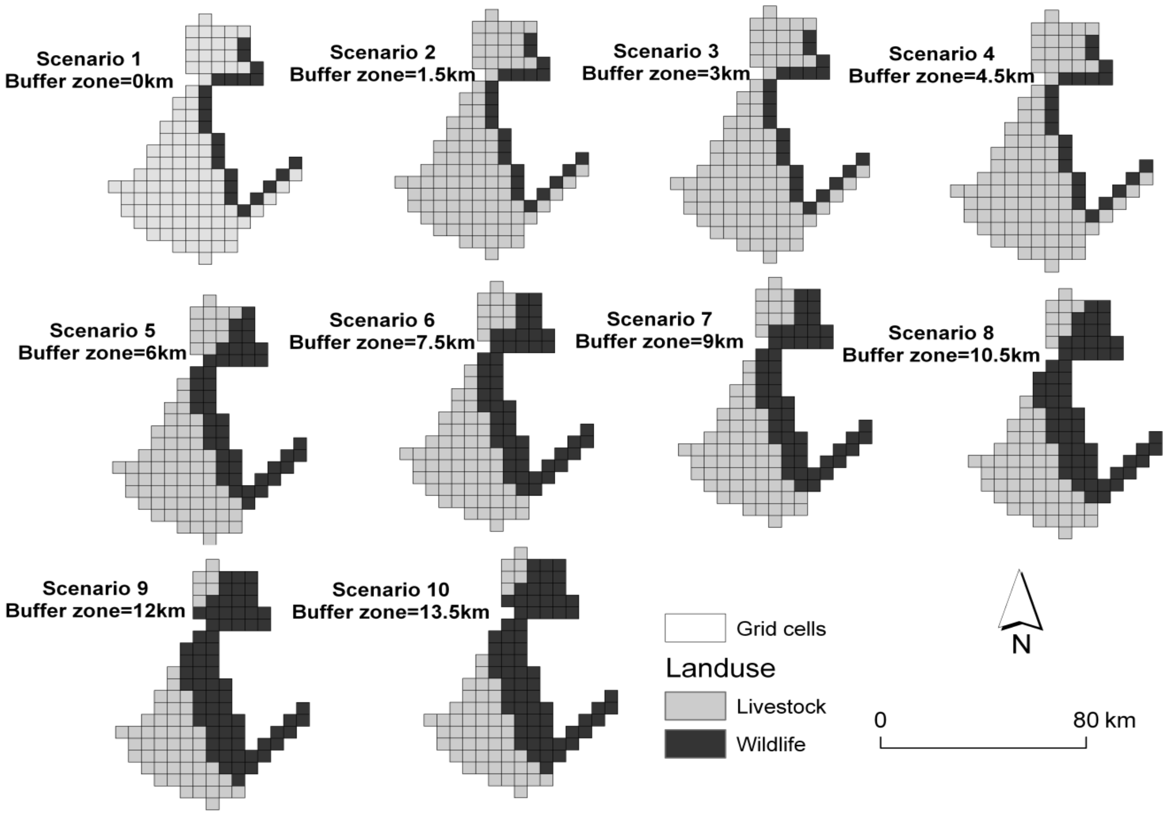

Figure 5.4a: Land allocation to land uses: Livestock and Wildlife with an increase in park (Scenario 1 to 10) when probability of drought is 0.20, 0.40 and 0.60.

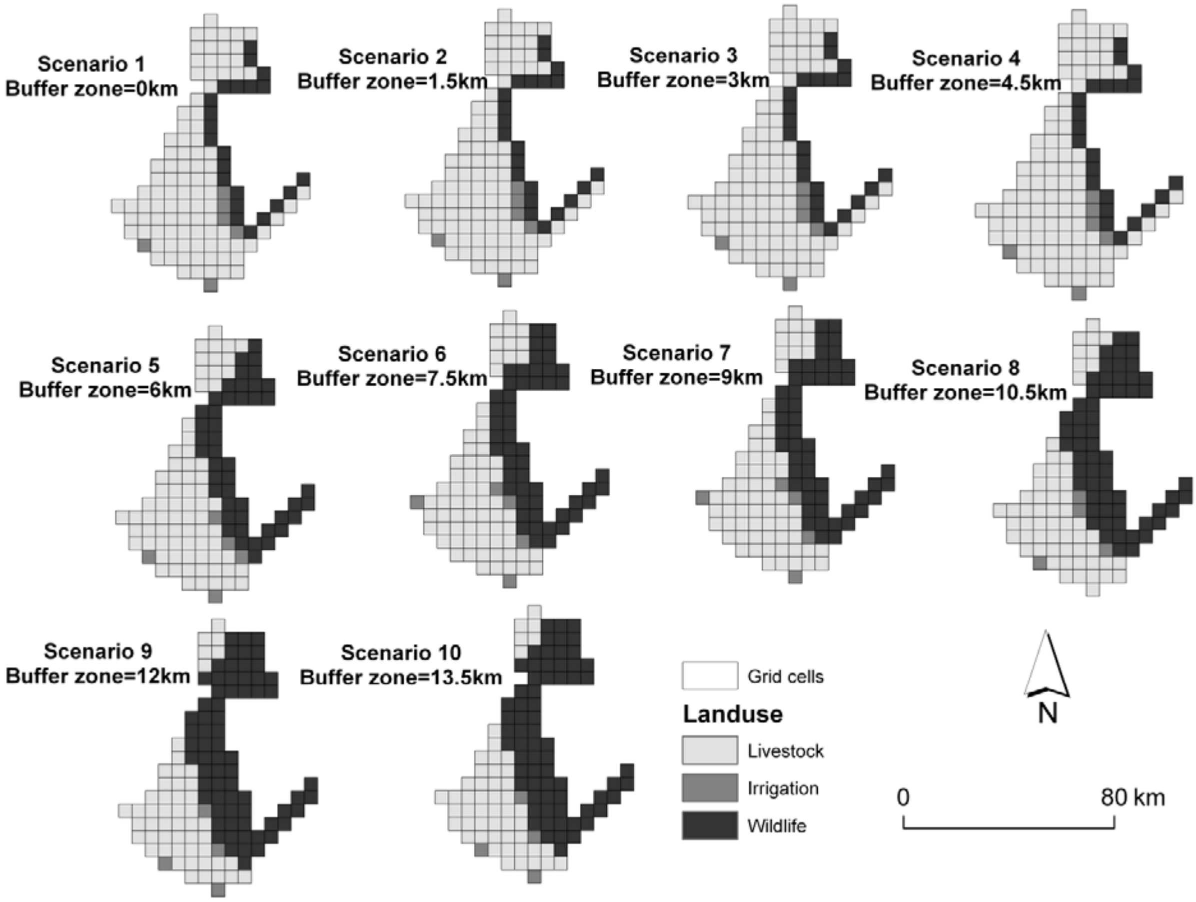

Figure 5.4b: Land allocation to land uses: Livestock, Irrigation and Wildlife with an increase in park (Scenario 1 to 10) when probability of drought is 0.40.

We also evaluated how many persons and herbivores could be sustained in the south eastern 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. Dip tank records show that there were 39,200 mature cattle in 2008. This implies that the actual number of cattle is 5 times higher than what is sustainable. Based on a total area outside the park considered in this study (3,077 km^2), the area can sustain a total of 770 households. This is based on the fact that a household requires 4 km^2 to cover its needs for arable and grazing land according to an estimate by Cumming (2005). However, the human population estimate for wards 11 to 15 was

36,986 people and 6,485 households in the year 2000 (CSO 2002). 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” (WRI 2005) 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 5.2a to 5.2c, 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 (2012) 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 5.3). 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.

This finding is in line with modeling results from Skonhoft (2007) who stated that wildlife conservation can work directly against the interests of local people. Johannesen (2007) 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 (Cumming 2011), 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 the 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 (Barnet and Mahul 2007). 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 (Dercon 2002; Dercon *et al.* 2007). 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 (Dercon 2002; Dercon *et al.* 2007; Cashdan 1985) 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 (Skees *et al*. 2002).Traditional insurance instruments such as crop insurance can be used to cope with the risk of extreme weather events (Barnet and Mahul 2007). 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 (Barnet and Mahul 2007; Dercon *et al.* 2007).

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 (Muchapondwa and Sterner, Forthcoming). Otherwise, the only option would be industrialization as suggested by Malthus (Malthus 1970). 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 (Ravallion 2009).

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 (Prins 1992). 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 (Scoones 1992; Behnke 2000). 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 5.2), 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 (Ravallion *et al.* 2009). These findings agree with the assertion that high numbers of people in sub-Saharan Africa and those surrounding protected areas live in poverty (Balmford and Whitten 2003; Munthali 2007). 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 (Kates and Dasgupta 2007; Chen and Ravallion 2010). 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. A park increase of 11 to 12 km is where the increasing lowest income meets the decreasing expected income (Figure 5.2a to 5.2c).

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 the 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 (Ravallion *et al.*, 2009). This notion has also promoted recent conservation development paradigms called Transfrontier Conservation Areas (TFCAs) or mega-parks that cross international borders (Child 2009). The rationale is that adding wildlife conservation as a land use could diversify and consequently reduce risk (Muchapondwa 2003). 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

Appendix 5.1: 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
fl	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			

Chapter 6

Synthesis

Introduction

Biodiversity is under extreme pressure and adequate policies for biodiversity conservation are urgently needed, particularly in Sub-Saharan rangelands where high levels of biodiversity (Davis *et al.* 1994; Stratterfield *et al.* 1998) are juxtaposed with chronic poverty and underdevelopment (Homewood 2004). More species and more habitats are at risk in these rangelands than elsewhere due to a combination of rapidly rising human population, increasing per capita consumption and the high densities in tropical areas of wildlife and livestock (Balmford 2002). On these rangelands debates continue about whether these systems show equilibrium or non-equilibrium dynamics (Ellis and Swift 1988; Scoones and Behnke 1993; Sullivan 2002; Vetter 2005; Derry 2010; Behnke 2011). A conclusion on this debate has important implications for the science of rangeland ecology, natural resource conservation and management, and policy development on rangelands throughout the world (Ellis and Swift 1988; Briske *et al.* 2003). Limited experimental evaluation of these two broad paradigms has undoubtedly contributed to both the intensity and longevity of this rangeland debate (Brown *et al.* 2001). However, for rangelands in dryland Africa, extreme and unpredictable variability in rainfall are considered to confer non-equilibrium dynamics by continually disrupting the tight consumer-resource relations otherwise considered to pull a system towards equilibrium (Sullivan and Rohde 2002). In these densely populated and often poverty-stricken areas, also other factors play a role to prevent density-dependency mediated equilibria between vegetation and herbivores to arise. These are diseases and civil unrest. In many countries veterinary services are less than adequate, and many diseases cripple the livestock industry or local animal husbandry. Diseases often spill-over to wildlife, playing havoc with animal numbers (Prins & Grootenhuis 2000). Civil unrest leads to little or no control over property. In some areas the demand for bush meat is so large that animal numbers are singularly depressed, thus preventing a situation that even comes close to what could under some world views be considered an equilibrium. In my thesis I concentrate on the vagaries of rainfall because during periods of the falling apart of law and order it is very difficult to collect the necessary data to show non-equilibrium dynamics.

Therefore, the main aim of this thesis was to investigate long-term sustainable and economic efficient management of wildlife and livestock in a dry land savanna system in the context of non-equilibrium dynamics. I presented four studies to meet this aim, divided into two parts: ecological and economics. From the ecological side, I investigated important factors that determined vegetation production and composition in order to understand possible sources of disturbance for dry ecosystems to shift from one state to the other. This was followed by an attempt to answer whether the system was showing non-equilibrium behaviour or not. From the economics side, I studied the potential of wildlife income to buffer and also to provide insurance against fluctuations in household income due to unpredictable rainfall fluctuations. In this chapter, I discuss important points that led to the final conclusion by zooming in on relevant issues of the previous chapters. I then proceed to generalize these findings to a broader context and identify gaps that still need further investigation, and then I conclude with implications for management of (semi-) arid savanna rangelands.

The findings

In the ecological section of this thesis (Chapters 2 and 3), I attempted to understand the ecological relationships as well as the dynamics of the southeastern lowveld of Zimbabwe rangeland system. In Chapter 2, I asked the question: What are the important biotic and abiotic factors explaining vegetation variables such as grass and wood species composition, production and basal cover? I tested the following expectations (1) that the vegetation composition in areas with high densities of large herbivores contrasts most strongly with areas where herbivore density is low; (2) that high densities of livestock and other large grazing herbivores would foremost influence the herbaceous composition, rather than the woody plant composition; and (3) that in livestock-rich areas outside the conservation zone, soil contrasts would partially explain plant community contrasts among sampled areas. Findings showed that differences in plant species composition are explained by abiotic factors like rainfall and soil, and only to a smaller extent by grazing intensity. However, because measured environmental variables could only account for a small percentage of the variation, it can be suggested that other factors such as periodic droughts, rather than local contrasts in abiotic and biotic variables, determine vegetation communities in this semi-arid ecosystem. In Chapter 3 the question was: Is there something like non-equilibrium and what are the impacts of such dynamics on cattle herd dynamics? I studied the relevance of non-equilibrium theory to my study area by testing whether annual changes in cattle numbers showed the presence of crashes and if so, what were the factors best explaining those crashes and what age and sex classes of cattle were most vulnerable to such crashes? I analysed the data with the concept of step functions in mind. Findings showed that crashes in annual cattle numbers were evident and were best explained by rainfall and NDVI and their lags. Immigration i.e., movement in of animals was also an important factor in years when rainfall was below the threshold and so it was a possible source of cattle recovery after a crash together with high calving rates. In years when rainfall was above the rainfall threshold, NDVI explained more variation in annual changes of livestock. Crashes were mainly caused by reductions of calf numbers rather than the other cattle age categories thus explaining why there are legacy effects (lags) in cattle numbers that can only partly be offset by cattle purchases from elsewhere because of poverty or lack of surplus stock elsewhere. These findings make the southeastern lowveld system to be dominated by non-equilibrium dynamics. Indeed, the Ecosystems Approach under the Convention on Biological Diversity (CBD) (CBD 2013), states that "ecosystem processes are often non-linear and the outcome of such processes often show time-lags. The result is discontinuities, leading to surprise and uncertainty". The consequences of such crashes and discontinuities, time-lags and uncertainty, make the system difficult to manage. Hence my main question is how do people cope with this, and what survival strategies have they developed to improve chances of survival and even of wealth protection under such conditions?

People in the lowveld find themselves thus in a situation in which (a) the ecosystem has low productivity, (b) the means to boost productivity are next to absent because private wealth is absent thus preventing the use of irrigation, artificial fertiliser or pesticides, (c) risk sharing with or passing on the burden of insurances to the rest of society outside the local country side has not been developed partly because the rest of society is poor too, (d) local insurance schemes, as many farmer collectives elsewhere have developed already for some 150 years [or longer?] do not form a part of the local culture, see e.g. https://en.wikipedia.org/wiki/Insurance#History_of_insurance. Within these social and cultural constraints, how then can these local people reduce their exposure to risk? My first proposition was that people's welfare is well-served by buffering their income and risk of losing wealth by making use of other sources of income (like remittances or wildlife). My second proposition was that local people's welfare could be improved by making use of resources that fluctuate independently of their agricultural resources, namely, wildlife. This was premised on the common assertion that in semi-arid rangelands (characterised by strong rainfall variability), wild resources have a comparative advantage over their domestic counterparts (Child and Barnes 2010).

The welfare of local people is the issue that I focussed on in my economic section of this thesis (Chapters 4 and 5). Based on the fact that local people in my study site were living adjacent to a nature reserve and because wildlife species have evolved with the savanna vegetation (Prins and Douglas-Hamilton, 1990; Bouchenak-Khelladi *et al.* 1998; Prins and Fritz 2008) wildlife species may be better adapted to annual rainfall fluctuations than domestic livestock hence people may get sufficient income from wildlife. In addition, biodiversity has been described as "the wealth of the poor" (WRI 2005), because it may provide income through hunting and gathering. Culturally this implies that local peasants still function in a pre-Modern economic state; my results are thus of importance outside my own research field for understanding local Iron Age culture too.

In Chapter 4 I asked the question: To what extent can wildlife income buffer rural households' incomes against fluctuations in rainfall? I studied the extent to which wildlife derived income can buffer local households' income against fluctuations due to rainfall. The addition of wildlife as an asset for rural farmers' portfolio of assets showed that wildlife can be used as a hedge asset to offset risk from agricultural production without compromising on return. However, the power of diversification using wildlife is limited because revenues from agriculture and wildlife assets were positively correlated.

However, the correlation is very weak (only 0.4 and the explained variance thus only be 16%) which gives ample scope for buffering. Therefore, revenues from wildlife have potential to reduce household income fluctuations due to drought, but only to a limited extent.

In Chapter 5 the question was: From a theoretical perspective, can wildlife income have an insurance value to local people? I used a modelling approach to study the extent to which wildlife income offers an insurance value to local people against fluctuating annual rainfall. Findings did not support the common assertion that wildlife can offer insurance to local people against income fluctuations due to rainfall fluctuations. The failure by wildlife income to offer insurance value to local people could be explained by high costs of harvesting the wildlife resource and high densities of both human and livestock populations in southeastern lowveld. As corollary I draw the conclusion that wildlife cannot pay its way in these rangelands as long as there are high densities of people, as shown in Chapter 5. Definitely wildlife income becomes insufficient if long-term sustainability of wildlife resources is considered. Moreover, with the high densities of domestic herbivores (Chapters 2 and Chapter 5), the long-term sustainability of the system can be compromised.

On non-equilibrium dynamics

The findings from Chapters 2 and 3, certainly confirm that southeastern lowveld is a non-equilibrium system. The finding that rainfall and soil pH are important determinants of landscape scale variation in botanical composition suggests the presence of non-equilibrium dynamics. Further, the lower explained variation means that either some important variables were missed or more importantly the ecosystem is responding dynamically to changes not easily captured in environmental variables. Furthermore, the finding that rainfall has the overriding effect on changes in cattle numbers, implied evidence of non-equilibrium dynamics. However the fact that NDVI became important above the rainfall threshold also suggests the presence of equilibrium dynamics during wetter years. These findings imply that equilibrium between the animals and the resource only occurs during wetter years. Due to time-lags one would not expect a quick "equilibrium". However, this is made possible by the practice of sharing-out cattle. This practice is also normal with Masai and Samburu in East Africa (Prins personal communication). Sharing-out cattle allows the area to fill up quickly with immigrated animals and once they are there, the system is then properly stocked. However, for the greater part of the time, the equilibrium between the resources and animals is absent. It is now agreed that although density-dependent effects might cause populations to tend toward an equilibrium, it is likely that the position of the equilibrium will change over time and that factors such as rainfall, fire, disease, or human influence deflect populations from a possible equilibrium or "carrying capacity" (Gilson *et al.* 2005). Some authors assert that it is essentially meaningless to talk about a complex adaptive system being in equilibrium: the system can never get there because it is always unfolding, always in transition (Holland 2000; Sullivan and Homewood 2003).

The controversial parts of the equilibrium/non-equilibrium debate is that if rangelands follow the non-equilibrium viewpoint, then herders and their livestock rarely degrade rangelands (Behnke 2011; Sullivan and Homewood 2003). With my findings in Chapter 2, I disagree with such views. The studies reported by Behnke (2011) deal greatly with nomadic people, whereas my study deals with agriculturalists. I reported very high stocking densities in communal areas compared to the small scale commercial farming areas and the nature reserve. In fact people in the communal areas of southeastern Zimbabwe had stocking densities that are only expected in high rainfall areas. Findings from Chapter 2 implicate to a lesser extent grazing intensity as one of the causes of vegetation composition changes. The presence of higher numbers of livestock than what the system could carry is a function of human practices which implies that people play an important role that manifests itself through complex land management and tenure practices. Therefore, even if a system has strong non-equilibrium characteristics, as I showed in Chapter 2, livestock grazing through high stocking densities may have detrimental effects on the system as evidenced by differences in vegetation composition and production, especially during wetter years when there is evidence for coupling between livestock and their resource. The results suggests that the system is always under strain because in wetter years high grazing intensity puts strain on the vegetation, while in drier years the system is strained by absence of moisture. I agree with Vetter (2005) that there is an interaction between rainfall and stocking rate in these systems, with low rainfall exacerbating the effect of high stocking rate and high rainfall mitigating them. Under such circumstances it is difficult to devise a stocking rate which does not result in overgrazing in a year with below average rainfall and underutilization in a good rainfall year.

My findings in Chapter 3 have shown evidence of lags, crashes and thresholds. Thresholds represent boundaries that separate multiple equilibrium states in time and space, and their existence determines that a system is non-equilibrial (Holling 1973; Briske 2003). The presence of thresholds suggests bifurcations and phase changes i.e., below the rainfall threshold immigration becomes important, whereas above the threshold NDVI becomes important in explaining cattle changes. These findings therefore point to chaos theory, which attempts to understand the behaviour of systems that do not unfold in a linearly predictable, conventional cause-and-effect manner over time (Murphy 1996). The trend toward destabilisation in a chaotic system can lead to sudden changes in the system's direction, character, or structure called bifurcations, and at such points the system rearranges itself around a new underlying order, which may be very different from the prior one (Murphy 1996). Thus the system itself is a moving target (Holling 1998), with surprise (CBD 2013), uncertainty and unpredictability emerging from both biotic and abiotic sources and with effect that differ according to scale of observation (Sullivan and Homewood 2003). Meanwhile, at the scale of my study area, different factors explained annual changes in cattle in different wards (Chapter 3). Furthermore, in Chapter 2, rainfall, soil parameters particularly pH were the factors which were significant in explaining variation in vegetation variables. Additionally, my data suggests that other factors which were not measured may also play a role e.g. drought. All these factors make it very difficult to predict the outcome of events from this system. Deterministic chaotic systems are fundamentally unpredictable (Scheffer 2009). Even if we know exactly the rules that govern the system, the final outcome remain unpredictable (Scheffer 2009). According to CBD (2013) it also becomes very difficult for people to react and more so manage unpredictable systems. I guess the essence of science is to find general patterns. I agree with the principle of parsimony in statistics that encourages fitting of a minimum model (Crawley 2007). Fundamentally unpredictable systems can perhaps not be managed as if they are predictable. Predictive science in that case just does not provide the answer but can yield scenarios with higher and lower likelihoods.

On local people's welfare

The poorest people in the World rely disproportionately on the natural resource base, earning their living from agriculture, fishing, forestry and hunting (Wright and Boorse 2010). Thus questions of human development and the alleviation of poverty cannot be separated from issues of environmental management. Findings from my economic section of this thesis (Chapters 4 and 5) point to some of the important factors of the biodiversity crisis. According to Vandermeer (2009) most efforts directed at biodiversity conservation have been failures. Although it is true that some national parks and other biological reserves function well, most are poorly managed or exist only on paper. The small contribution of wildlife income to local people's welfare (Chapter 4 and 5) goes to show the widely shared view that financial rewards generated through integrated conservation and development programmes such as CAMPFIRE have generally been seen as insufficient (Barnes *et al.* 2002; Wolmer 2003; Child 2009). No wonder why opponents of community natural resources management (CBNRM) argue that after more than 20 years of participatory and community centred experiences, the integrative approach has proved to be a failure and that, given the current crisis of biodiversity, new, urgent and efficient actions must be implemented (van Schaik and Rijksen 2002; Roday 2009). Based on findings from this thesis I argue that current (e.g. CAMPFIRE) and even new initiatives proliferating in southern Africa in the name of Transfrontier conservation areas (TFCAs) or mega-parks that cross international borders (Jones 2005; Child 2009), will hardly yield positive results pertaining to local people's welfare, unless we get the numbers of both people and livestock in the system right. Densities of 29 people per km^2, have been reported for Zimbabwe, compared to 3 people per km^2 and 2 people per km^2 for Botswana and Namibia (Heitkönig and Prins 2009), respectively. We need to know that primary production is relatively low in south eastern lowveld compared to other ecosystems due to high aridity (Chapter 2). This set the limits to how many livestock and people can be supported in such areas. In Chapter 5 I reported that the system can accommodate about 7,000 to 10,000 livestock units but there are over 39,000 mature cattle in the area. Heavy stocking rates are thought to be inevitable in communal rangelands because of the problems inherent in communal ownership of a resource where individual benefit is maximised at the expense of the community (Hardin 1968). Human population density in the area was calculated to be 12 people per km^2. With increase in population, the pressure on natural resources in many regions has continued to intensify (Prins 1992; Lusigi 1994; Ayoo 2007). Furthermore, with such densities and given that the area shows non-equilibrium dynamics, then it can only be expected that poverty and declining overall people's welfare is the order of life in these areas. I agree with the claim that the problems in community based policies are the consequence of the lack of real empowerment at local level and of the misunderstanding of social processes in natural resources access and use (Brechin *et al* 2003, Raday 2009). However, I found in my study area that income from sustainable wildlife management will be insufficient given the many beneficiaries it is supposed to cater for. Cumming (2011) reports that interactions between wild animals, domestic livestock and humans have been greatly magnified by rapidly growing human and livestock populations, expanding agriculture, and land use land cover changes over the last century. This explains the ever increasing conflicts around nature conservation areas. Given the undeniable fact that we continue with the biodiversity crisis, even after many well-meaning intelligent and even rich people have become concerned, suggests that something is wrong. If we have a moral or ethical obligation to protect wildlife species, then an important way for people to meet their aspirations economically was suggested by Malthus (1798).

On sustainability and Malthus's views

Concern about the natural environment has continued to grow in importance, and encountering the term 'sustainability' on a regular basis has become an unavoidable feature of academic life (Brander 2007). The Brundtland commission (1987) provided a good working definition of sustainability, by stating it is "development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs". I use the definition of sustainability provided by Asheim (1994) (see Hanley and Atkinson 2003), which is defined as “a requirement to our generation to manage the resource base such that the average quality of life we ensure ourselves can potentially be shared by all future generations”. The second definition does not focus only on future generations, but it also highlights the quality of life of the current. Definitely in Africa the quality of life for the poor needs to be substantially improved first before sharing it with future generations. Faced with this challenge of sustainability on one hand, and the need for people to live a decent life on the other hand, we need to search for solutions that try to provide conservation of biodiversity in combination with rural development. These will however often be conflicting objectives. It has been considered the Holy Grail of development organisations for the last 25 years and seems to be more akin to trying to solve the quadrature of the circle.

Chapters 4 and 5 showed that people in southeastern lowveld live on less than US$ 1.25 per day which is the poverty line according to Chen and Ravallion (2010). Absolute poverty, globally, has been in decline for approximately the past 25 years, yet in Africa it is still increasing (Collier 2007). The percentage of extremely poor fell from 40% to 18%, whereas in Sub-Saharan Africa the numbers increased by 44% between 1981 and 2004 (Kates and Dasgupta 2007). Therefore, Africa’s problem is to break out of an economic stagnation that has persisted for the past three decades (Collier 2007) and this is where Thomas Malthus (1798) comes in. Malthus in his essays, expressed concern about a possible tendency of human population to grow more rapidly than can be accommodated by arable land and other components of the resource base, thus according to Brander (2005) sowing the seeds of their own decline. The power of demography as indicated in Chapter 5 of my thesis, I think is being felt by all of us these days. However, of particular importance in this study, Malthus predicted that the best we could do was to go for higher incomes through advanced stages of commerce and manufacturing. He stated that with higher incomes, the population can enhance its type of subsistence from the simple food needed for survival to a sophisticated mixture of necessaries, comforts and luxuries. According to Rutherford (2007), this would suggest that through industrialization, the problem whereby subsistence grows more slowly than population within any country would be solved. In addition, to further cement Malthus’s ideas, Beckerman (1994) also suggested that the only way to attain a decent environment in most countries is to become rich.

Malthus was heavily criticised mainly because since his time, world population has risen seven fold to more than 7 billion, yet apocalyptic collapses have mostly been prevented by the advent of cheap energy, the rise of science and technology and the green revolution (Stokstad 2005). Boulding (1966) argued that the World is a closed system and that the economy is more, like a spaceship. He pointed out that there are only limited resources and so the spaceship economy needed to be concerned primarily with the stocks (Elliott 2005). Obviously these statements led to the reawakening of Malthusian concern. If we have 'sustainability' at the back of our minds, then the current generation, must be aware that we do not live in isolation, but as a continuing stream of people who have and who will exist (Elliott 2005). Then to go the route of industrialisation and becoming rich as nations is the only way out of poverty for most African nations. I think this will be a way for most people to move into cities and giving a chance for nature to proliferate at the country side.

In addition to Africa having to industrialise in order to curb poverty, other factors that still are subject of debate are the low population densities in Africa relative to other continents and the fact that people farming in Africa have to produce more for the market than just for themselves. I have reported high population densities in the communal areas of southeastern lowveld of Zimbabwe (Chapter 5), but comparing African societies to those of other regions of the world, the continent has the lowest population density of any of the major continents (Green 2012). The role of low population density is in influencing the retarded development of modern state institutions in Africa (Herbst 2000; Robinson 2002). According to Robinson (2002), unlike in Europe, land was and is not scarce in Africa, rather, labour was scarce. Thus in the pre-colonial period, states did not fight over land, but rather people. This explains why to this day most land in Africa is held communally. Economists have long emphasised how Africa's high land-to-labour ratio has led to high labour costs and also due to few cities far removed from each other, transport costs too become very high causing market failures. The overall effect is that the costs per head become enormously high in Africa due to low densities of people while in other continents the costs are low as they are shared by many people due to high population densities. Another fact is that farmers produce for the market, but peasants produce only for themselves. In other words, peasants stay outside the economy and they will never get rich. Peasants do not accumulate wealth, do not turn wealth into capital and do not properly invest, thus peasantry does not lead to development. For industrialisation one need capital, machinery and labourers. That is why the colonial governments focused so much on labour creation (and compulsory drafts from peasants for labour). In southeastern Zimbabwe, the left-over is still visible from the enormous impact on the local economy on remittance money (Chapter 4). Yet, the sad thing is that most remittance money is merely transferred into consumption, and not on capital development.

Future directions

Despite the various caveats, the various chapters in this thesis contribute towards an understanding of how people can live in a system governed by non-equilibrium dynamics. More importantly this thesis contributes to understanding how welfare of people perhaps can be improved in systems that are chaotic in nature. From a methodological standpoint, non-equilibrium theory cautions against uncritical acceptance of traditional statistical analysis. It is therefore imperative that rigorous research in statistical tools that can help analyse non-linear phenomenon be undertaken. This thesis also focused on the understanding of the underlying dynamics of the ecosystem, which includes taking into account the feedback mechanisms between rainfall, vegetation and livestock in a highly fluctuating environment. Further research should include sociology, cattle transfers, power, class and scale, since results on chaotic systems will vary greatly depending on scale, that is, which portion of a phase space the researcher happens to study. The trend toward destabilisation in a chaotic system can lead to sudden changes in the system's direction, character, or structure called bifurcations (Murphy 1996; Van Langevelde *et al.* 2003). Future research can study whether it may be possible to influence the development by making choices if one intervenes at the point when a system is about to bifurcate.

Work on how to deal with drought using a variety of sources of income, ranging from farm assets, livestock, remittances and non-farm income has been extensively covered. My contribution to this literature is by studying diversification using wildlife as an asset using the portfolio theory framework (Markowitz 1952, 1959). Analyses of this problem can be improved by further research using different levels of risk aversion and also the different weights of the different assets. Weights being the proportion of the portfolio invested in each asset (Damodaran 1998; Reilly and Brown 1999). The characteristics of the portfolio, for example how much risk they entail and the expected return from the portfolio, are altered by a change in the weighting of the assets or elements (Figge 2004). In analysing the ability of wildlife to insure local people against drought using modelling approaches, future research can also consider other important factors such as plot connectivity, fencing constraints, land elevation, slope, or habitat patch size. The use of management tools by wildlife managers such as fencing, fire management, closing and construction of artificial water points and population manipulation for sustainable wildlife management given multiple land owners can also be further studied. Initial modelling work was covered by Mwakiwa (2011) and Mwakiwa *et al.* (2013).

Management implications

The equilibrium vs. non-equilibrium debate arose because of the dissatisfaction with the Clementsian-based procedure (range model) for range condition and trend analysis (Briske *et al*. 2003), that it is an ineffective, over-simplification of vegetation dynamics on many rangelands (Noy-Meir 1973; Laycock 1989; Smith 1989; Westoby *et al.* 1989). The concern was that application of the range model may contribute to mismanagement and degradation of some rangeland ecosystems (Ellis and Swift 1988; Mentis *et al.* 1989; Walker 1993a; Briske *et al.* 2003). My contribution is towards finding strategies to sustainably manage rangelands especially in arid and semi-arid zones. I have established that the interaction between rainfall, soil parameters and stocking densities in these rangelands would be critical in explaining the dramatic crashes in cattle numbers that have occurred when there is a drought. The fact that drought effects on herd dynamics are heterogeneous underscores the importance of herd population structure. The effect of population structure on herd dynamics is strongest at high and intermediate initial population sizes and weakest when the population is low (Wallington *et al.* 2005). This is because there is more variation across age classes in survival rates which drive population crashes. These findings imply that it is important for initial herd population sizes to be low for less dramatic crashes during drought years. Further, high stocking rates leads to heavy grazing which is detrimental to the sustainability of livestock production systems and will result in the permanent degradation of the natural environment and an associated collapse of people's livelihoods. Workman (1986) stated that it is a fallacy to blame the profit motive for resource degradation associated with excessive stocking rates, the rationale being that of diminishing economic returns and higher input costs associated with increasing level of stocking. In order to prevent degradation of primary resources it is essential to adopt a long-term view when planning, and to adjust animal numbers accordingly (Teague *et al.* 2009). In the long-term profits are maximised at lower stocking levels than those that maximise livestock production per hectare (and gross revenue) which is the goal of many producers who do not account for production costs, but instead externalise the cost of rangeland degradation (Teague *et al.* 2009). Economically optimal stocking rates were established in Chapter 5 of this thesis and farmers would better maintain a base herd of animals not above this economically optimal stocking rate if they hope to gain economically from the system in the long-term.

Moreover, due to rainfall variability, the establishment of fixed stocking rates for semi-arid rangelands appears unwise (Behnke and Scoones 1993; Illius *et al.* 1999). One common option for maintaining a constant stocking rate as rainfall varies is to provide supplementary feed; however, irreversible vegetation change may occur if animal numbers are held constant when natural resources are scarce (Van de Koppel and Rietkerk 2000). Further, semi-arid rangelands are too dry for planted pastures and improvement of the protective cover and forage value of the range must be achieved by manipulating the natural vegetation (Kelly and Walker 1976). Therefore alternative management strategies for dealing with drought might include increasing or decreasing stocking rate based on the current condition of the pasture, season of the year, and the direction and rate of change in animal body condition. In addition, generally successful, properly timed, calving and breeding seasons that coincide with expected available forage would also help.

I have also contributed to the management of wildlife resources through findings from Chapters 4 and 5. Cumming (2011) reported that the conservation success in these rangelands depends on the extent to which communal farmers - the *de facto* resource managers in communal areas - manage their land in ways that support conservation. He also indicates that farmers will only do so if it is to their benefit and if those benefits outweigh alternative land and resource uses. I, however, showed that wildlife benefits are usually viewed as small mainly because of the direct use values that are normally considered. Other values such as indirect use, option, non-use values such as existence and bequest value (see Chapter 5 for definitions) hardly get considered. Indeed, the value of these ecosystem services may exceed the value of commodities derived from traditionally managed natural resource sectors such as forestry, fisheries and agriculture (Costanza and Folke 1997; Daily 1997). Additionally in semi-arid rangelands where drought risk is very high, diversification using wildlife conservation is a feasible option even if wildlife income is small compared to agro-pastoral income (Chapter 4). If the decisions relate to complete portfolios, it is not the return and risk of each individual element that are of interest, but those of the complete portfolio (Figge 2004). Decision rules, which relate to the individual elements, can then lead to incorrect, economically irrational decisions (Figge 2004). The dilemma though is how to get the numbers of people in the system correct, for wildlife benefits to be visible.

I have also showed in Chapters 4 and 5, the need for policy makers to ensure that revenues intended for communities should be increased, either by increasing shares to communities or by considering an option where communities run their own business, in CAMPFIRE for example. Though the second option could be curtailed by implementation challenges, conservation efforts in semi-arid rangelands can be improved by a further devolution. Cumming (2011) supports this view when he reported that the single greatest weakness of CBNRM is aborted devolution of rights and responsibilities (costs and benefits) to the lowest level of social organisation for common pool resources.

Since resource use in communal areas is regulated by open access property regimes, there are no identified owners, no-one can be excluded from the resource and each person has a right to withdraw resources (Heitkönig and Prins 2009). What is needed is a change in the institution controlling open access areas, which can be achieved by adopting a set of collective choice property rights (Schlager and Ostrom 1992) which include the right of management, but also the right of exclusion and alienation (Heitkönig and Prins 2009). Levies should decrease too, or more clearly result in services to local people.

In conclusion, a property of unstable systems is resilience. As explained by Holling (1973), resilience refers to the ability of major ecosystem processes to remain functional in the presence of exogenous shocks such as drought. Under excessive anthropogenic pressures and the high variation in rainfall, the resilience threshold of the system can be exceeded which can result in land degradation. Hence the loss of utilisable rangeland carries huge economic and social costs. Even though my data does not show clear evidence of land degradation in rangelands for now due to high stocking densities, catastrophic effects like system collapse may result if this continues. Further, semi-arid rangeland ecosystems like those in southeastern Zimbabwe, when viewed as a whole, exhibit non-equilibrium behaviour. At no single point could the future direction of such systems have been predicted from their past history. Therefore the strategies for livestock management are characterised by a close adaptation of the stocking rate to available forage. The best way to deal with these situations, apart from becoming nomadic and abandoning arable agriculture, are (a) share-herding over vast areas, (b) very fast buying and destocking, or (c) trucking of livestock. Trucking is done in North America, Australia and also Mongolia, and it entails transporting livestock within a vast region so that animal numbers are not linked to local climatic vagaries; in Australia this can be over distances over several thousand kilometres (pers. comm. H.H.T. Prins). Share-herding is a system that has been used for a long time in East Africa: owners allot proportions of their herd to different other owners, who then deploy herders to graze herds that are comprised of different owners. Cattle owned by one owner may graze in herds that are separated by hundreds of kilometers (pers. comm. H.H.T. Prins). My data further shows that wildlife utilisation is a highly competitive form of land use in these drier regions, only if economic institutions can reflect the true value of wildlife and if economically optimal densities of human and livestock populations are not exceeded. I have therefore contributed to a better understanding of how people can live in a system that is characterised by non-equilibrium dynamics.

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Summary

Grazing systems, covering about half of the terrestrial surface, tend to be either equilibrial or non-equilibrial in nature, largely depending on the environmental stochasticity. The equilibrium model perspective stresses the importance of biotic feedbacks between herbivores and their resource, while the non-equilibrium model perspective stresses stochastic abiotic factors as the primary drivers of vegetation and herbivore dynamics. In semi-arid and arid tropical systems, environmental stochasticity is rather high, making the systems essentially non-equilibrial in nature, suggesting that feedback between livestock and vegetation is absent or at least severely attenuated for much of the time. In southern Africa, range and livestock management however, has been built around the concept of range condition class and the practices of determining carrying capacities and manipulating livestock numbers and grazing seasons to influence range condition. This management approach is derived from the equilibrium or climax concept of Clementsian succession. The erratic and variable rainfall in many pastoral areas of Africa poses a fundamental challenge to this conventional notion of carrying capacity in range management. This realization has caused a shift towards models that embrace non-equilibrium dynamics in ecosystems. The main concern is that application of the range model may contribute to mismanagement and degradation of some rangeland ecosystems. However, only a few studies in rangelands have empirically tested the non-equilibrium hypothesis leading to the debate on rangeland dynamics remaining unresolved.

Across the savannas of Africa, grasslands are being changed into cultivation due to increasing human population, at the expense of decreasing wildlife populations. African savannas however, still contain pockets of wilderness surviving as protected areas, but even there, species richness of large mammals is decreasing. The inevitable result is the loss of most of the wild plants and animals that occupy these natural habitats, at the same time threatening the well-being of the inhabitants of these savannas. Hence, to facilitate the management of arid and semi-arid savannas for both biological conservation and sustainable use (improving human welfare) an improved understanding of the complex

dynamics of these savannas is critical. Furthermore, it is widely recognized that a high level of uncertainty typifies the lives of rural farmers in developing countries. Non-equilibrium dynamics bring additional uncertainty and risk to the system. However, attempts to understand efficient and sustainable ways to improve biodiversity and human welfare in systems showing non-equilibrium dynamics have been rare. The behaviour of non-equilibrium systems is characterised as more dynamic and less predictable than equilibrium systems. Therefore, non-equilibrium dynamics in dryland ecosystems present a different kind of management problem for both livestock and wildlife systems since their management has been dictated by the equilibrium assumption. Additionally, loss of biodiversity is regarded today as one of the great unsolved environmental problems. Faced with this biodiversity crisis, the challenge is to find ways to respond in a flexible way to deal with uncertainty and surprises brought about by non-equilibrium dynamics.

In this thesis I use a bioeconomic approach in analyzing the implications of non-equilibrium dynamics for the efficient and sustainable management of wildlife and livestock in dryland grazing systems. The study area for this thesis is southeastern lowveld of Zimbabwe.

In chapter 2, I investigate the role of abiotic and biotic factors in determining plant species composition. While early studies emphasized the importance of edaphic and environmental controls on plant species distribution and spatial variation in vegetation composition, recent studies have documented the importance of both natural and anthropogenic disturbances in this respect. At a regional scale vegetation structure (i.e., grass/tree ratio) and species composition in savannas is largely determined by precipitation, whereas at the nested landscape-scale vegetation structure and composition is more prominently determined by geologic substrate, topography, fire and herbivory. Chapter 2, shows that at the landscape scale, abiotic variables such as rainfall and soil fertility override the effect of humans and livestock on the herbaceous and the woody plant composition.

Then, in Chapter 3, I ask the question whether there is something like non-equilibrium and what are the impacts of such dynamics on cattle herd dynamics? I studied the relevance of non-equilibrium theory to my study area by testing whether annual changes in cattle numbers showed the presence of crashes and if so, what were the factors best explaining those crashes and what age and sex classes of cattle were most vulnerable to such crashes? Chapter 3 showed that crashes in annual cattle numbers were evident and were best explained by rainfall and NDVI and their lags. Immigration i.e., movement in of animals was also an important factor in years when rainfall was below the threshold and so it was a possible source of cattle recovery after a crash together with high calving rates. In years when rainfall was above the rainfall threshold, NDVI explained more variation in annual changes of livestock. The impacts of crashes were greater on calves than other cattle age categories thus explaining why there are legacy effects (lags) in cattle numbers that can only partly be offset by cattle purchases from elsewhere because of poverty or lack of surplus stock elsewhere. These findings make the southeastern lowveld system to be dominated by non-equilibrium dynamics.

The welfare of local people is the issue that I focused on in my economic section of this thesis (Chapters 4 and 5). I addressed the question of how risks of fluctuations in household income can be managed in order to improve human welfare. The expectation was that in systems exhibiting non-equilibrium dynamics people can improve their welfare by exploiting a combination of wildlife and agricultural activities (livestock and cropping) in their attempts to reduce fluctuations in their annual welfare. This would be possible if the risks in wildlife and agro-pastoral systems were sufficiently different. Exploiting different sources of income requires efficient allocation of resources. The most prominent resource is land and land varies spatially in quality and ecological resources require spatial connectivity. Therefore the spatial dimension is important in this allocation.

In Chapter 4 I asked the question: To what extent can wildlife income buffer rural households' incomes against fluctuations in rainfall? I studied the extent to which wildlife derived income can buffer local households' income against fluctuations due to rainfall. The addition of wildlife as an asset for rural farmers' portfolio of assets showed that wildlife can be used as a hedge asset to offset risk from agricultural production without compromising on return. However, the power of diversification using wildlife is limited because revenues from agriculture and wildlife assets were positively correlated. However, the correlation was very weak (only 0.4 and the explained variance thus only be 16%) which gives ample scope for buffering. Therefore, revenues from wildlife have potential to reduce household income fluctuations due to drought, but only to a limited extent.

In Chapter 5 the question was: From a theoretical perspective, can wildlife income have an insurance value to local people? I used a modelling approach to study the extent to which wildlife income offers an insurance value to local people against fluctuating annual rainfall. Findings did not support the common assertion that wildlife can offer insurance to local people against income fluctuations due to rainfall fluctuations. The failure by wildlife income to offer insurance value to local people could be explained by high costs of harvesting the wildlife resource and high densities of both human and livestock populations in southeastern lowveld. As corollary I draw the conclusion that wildlife cannot pay its way in these rangelands as long as there are high densities of people as shown in Chapter 5. Definitely wildlife income becomes insufficient if long-term sustainability of wildlife resources is considered.

Chapter 6, finally synthesizes the conclusions that can be drawn from the preceding chapters and puts the issues addressed in a broader context. In summary, this thesis shows evidence of non-equilibrium dynamics in semi-arid grazing systems. Furthermore, the small contribution of wildlife income to local people's welfare goes to show the widely shared view that financial rewards generated through integrated conservation and development programmes such as CAMPFIRE have generally been seen as insufficient. This led me to suggest that if we have a moral or ethical obligation to protect wildlife species, then an important way for people to meet their aspirations economically was suggested by Malthus.

Samenvatting

Begrazingssystemen, die ongeveer de helft van het vasteland bedekken, zijn in de natuur in evenwicht of niet, voornamelijk afhankelijk van de stochasticiteit van de omgeving. Het "equilibriummodel" benadrukt het belang van biotische terugkoppeling tussen herbivoren en hun hulpbronnen, terwijl het "non-equilibriummodel" de nadruk legt op stochastische abiotische factoren als de primaire kracht achter de dynamica van vegetatie en herbivoren. In semi-aride en aride tropische begrazingssystemen is de stochasticiteit van de omgeving vrij hoog, wat deze systemen uit evenwicht brengt. Dit suggereert dat de terugkoppeling tussen vee en vegetatie afwezig is, of tenminste meestal zwak is. Maar in zuidelijk Afrika is het beheer van de veestapel opgebouwd rond het concept van conditie van graslanden, het bepalen van draagkracht en de manipulatie van het aantal dieren en begrazingsseizoenen om de conditie van grasland te beïnvloeden. Deze beheerstrategie is afgeleid van het evenwichts- of climaxconcept uit de zogenaamde Clementsiaanse successietheorie. De onstabiele en variabele regenval in vele veehouderijgebieden in Afrika biedt een fundamentele uitdaging voor dit conventionele denkbeeld van draagkracht in graslandbeheer. Dit besef heeft een verschuiving veroorzaakt naar modellen die non-equilibrium dynamica gebruiken voor het begrijpen van het functioneren van ecosystemen. De belangrijkste zorg is dat de toepassing van graslandconditiescores zou kunnen bijdragen aan een foutief beheer en degradatie van bepaalde grasland ecosystemen. Echter, tot nog toe hebben weinig onderzoeken de non-equilibrium hypothese empirisch getest, waardoor de discussie over rangeland dynamica onopgelost blijft.

Op de Afrikaanse savannes worden graslanden omgezet in landbouwgronden door de toenemende menselijke bevolking, ten koste van het afnemende wild. Afrikaanse savannes bevatten nog steeds wildernisgebieden die blijven bestaan als beschermde gebieden, maar ook dáár neemt de soortenrijkdom aan grote zoogdieren af. Het onvermijdelijke resultaat is het verlies van het merendeel aan wilde planten en dieren in deze natuurlijke habitats, en tegelijkertijd wordt ook het welzijn van mensen die in de savannes leven bedreigd. Dus, om het beheer van aride en semi-aride savannes voor

natuurbescherming en duurzaam gebruik (verbetering van menselijk welzijn) mogelijk te maken, is er een verbeterde kennis nodig van de complexe dynamica van deze savannes. Daarnaast is het algemeen bekend dat de levens van boeren in ontwikkelingslanden gekarakteriseerd worden door een hoge mate van onzekerheid. Non-equilibrium dynamica zorgt voor bijkomende onzekerheid en risico. Pogingen om efficiënte en duurzame manieren te creëren die biodiversiteit en menselijke voorspoed verbeteren in non-equilibriumsystemen zijn echter zeldzaam. Het gedrag van deze systemen wordt gekarakteriseerd als meer dynamisch en minder voorspelbaar dan equilibriumsystemen. Non-equilibriumsystemen in droge gebieden bieden een ander soort uitdaging voor het beheren van vee en ecosystemen, omdat hun beheer gedicteerd wordt door een aanname van evenwicht ("equilibrium"). Daarnaast wordt het verlies van biodiversiteit vandaag de dag gezien als één van de grootste onopgeloste milieuproblemen. Geconfronteerd met deze biodiversiteitscrisis, is de uitdaging om manieren te vinden om op een flexibele wijze om te gaan met onzekerheden en verrassingen van non-equilibrium dynamica.

In dit proefschrift gebruik ik een bio-economische aanpak om de implicaties van non-equilibrium dynamica voor efficiënt en duurzaam beheer van wild en vee in aride begrazingssystemen te analyseren. Het studiegebied voor dit proefschrift is het zuidoostelijke laagland van Zimbabwe.

In hoofdstuk 2 onderzoek ik de rol van abiotische en biotische factoren die de samenstelling van de plantenrijkdom bepalen. Terwijl vroegere onderzoeken de nadruk legden op het belang van bodem en omgevingskenmerken op de plantenverbreiding en ruimtelijke variatie in vegetatiesamenstelling, hebben recente studies aangetoond dat zowel natuurlijke als antropogene verstoring belangrijk zijn. Op het regionale niveau wordt de vegetatiestructuur (de gras/boom ratio) en soortensamenstelling op savannes voornamelijk bepaald door neerslag, terwijl op landschapsschaal vegetatiestructuur en –samenstelling in aanzienlijke mate bepaald worden door het geologische substraat, de topografie, vuur en begrazing. In hoofdstuk 2 toon ik aan dat op de landschapsschaal abiotische variabelen, zoals regenval en bodemvruchtbaarheid, het effect van mensen en vee op kruidachtige en houtachtige plantensamenstelling overtreffen.

Vervolgens stel ik in hoofdstuk 3 de vraag of er zoiets bestaat als “non-equilibrium” en wat de impact is van zulke dynamica op het vee. Ik bestudeerde de relevantie van de non-equilibriumtheorie in mijn studiegebied door te testen of jaarlijkse veranderingen in aantallen vee instortingen (“crashes”) lieten zien en zo ja, wat de factoren waren die zulke instortingen het best verklaarden en op welke leeftijd en geslacht het vee het meest kwetsbaar was voor dit instorten. Hoofdstuk 3 toonde aan dat grote in jaarlijkse vee-aantallen evident waren en verklaard werden door regenval, NDVI en hun na-ijling. Immigratie, dat is de binnenkomst van dieren in het gebied, was ook een belangrijke factor in jaren waarin regenval onder een drempelwaarde viel. Zo was immigratie net als hoge kalvergeboortecijfers mogelijk een bron van herstel van de veestapel na een instorting. In jaren waarin regenval boven die drempelwaarde viel, verklaarde NDVI meer variatie in jaarlijkse veranderingen in de veestapel. Vooral kalveren leden onder de instortingen, wat verklaart waarom er na-ijlingen in veestapelgrootte zijn die slechts gedeeltelijk verklaard kunnen worden door de verkoop van vee van elders vanwege armoede of een gebrek aan extra financiële reserves. Deze bevindingen maken dat het zuidoostelijke laaglandsysteem gedomineerd wordt door non-equilibrium dynamica.

Het welzijn van de lokale bevolking is een kwestie waarop ik focus in mijn economische sectie van dit proefschrift (Hoofdstukken 4 en 5). Ik behandelde de vraag hoe risico’s van schommelingen in het inkomen van gezinnen beheerst kunnen worden met als doel het menselijk welzijn te verbeteren. De verwachting was dat in non-equilibriumsystemen de bevolking haar welzijn kan verbeteren door een combinatie van wildbenutting en landbouwactiviteiten (vee en gewassen) te exploiteren in een poging fluctuaties in jaarlijkse inkomsten te verminderen. Dit zou mogelijk zijn als de risico’s van wildbenutting en agro-pastorale systemen voldoende verschillend waren. Het exploiteren van verschillende bronnen van inkomen vereist een efficiënte allocatie van hulpbronnen. De meest prominente hulpbron is land en land vertoont ruimtelijke verschillen in kwaliteit en ecologische hulpbronnen vereisen ruimtelijke verbindingen. Dat maakt de ruimtelijke component belangrijk in deze verdeling.

In hoofdstuk 4 stelde ik de volgende vraag: tot op welke hoogte kan een inkomen gebaseerd op wild bufferend werken tegen de effecten van variaties in regenval op de rurale inkomens? De toevoeging van wild als een pluspunt voor rurale landbouwers toonde aan dat wild gebruikt kan worden om risico dat bij landbouwproductie hoort te verminderen zonder de opbrengst te verlagen. Maar de kracht van verscheidenheid door het gebruik van wild is beperkt, omdat inkomsten van landbouw en wild positief gecorreleerd zijn. Deze correlatie was echter zeer zwak (slechts 0.4 en met een verklarende variatie van slechts 16%), dus wild geeft een beperkte mogelijkheid tot buffering. Daarom hebben inkomsten van wild het potentieel om de fluctuaties op inkomens van gezinnen door droogte te verkleinen, zij het dan in beperkte mate.

In hoofdstuk 5 staat de volgende vraag centraal: vanuit een theoretisch oogpunt, kan een inkomen gebaseerd op wild een verzekeringswaarde hebben voor de lokale bevolking? Ik gebruikte modellen om de mate te bepalen waarin een inkomen gebaseerd op wild een verzekeringswaarde kan bieden aan de lokale bevolking tegen fluctuerende jaarlijkse regenval. De resultaten leverden geen ondersteuning voor de algemene stelling dat wild een verzekering kan bieden aan de lokale bevolking tegen fluctuaties in inkomen door onregelmatige regenval. Het falen van een inkomen gebaseerd op wild als verzekeringswaarde kan verklaard worden door de hoge kosten van het oogsten van de wilde hulpbronnen en de hoge dichtheden van zowel mensen als vee in het zuidoostelijke laagland. Hieruit trek ik de conclusie dat wild zijn weg niet kan bepalen in deze gebieden zo lang er dermate hoge menselijke dichtheden zijn zoals aangetoond in Hoofdstuk 5. Inkomens gebaseerd op wild zijn ontoereikend als duurzaamheid van deze hulpbronnen in beschouwing wordt genomen.

Hoofdstuk 6, ten slotte, synthetiseert de conclusies die kunnen worden getrokken uit de voorgaande hoofdstukken en plaatst de kwesties in een bredere context. Mijn proefschrift ondersteunt de idee dat non-equilibriumdynamica belangrijk is voor het begrijpen van semi-aride begrazingssystemen. De geringe bijdrage van een inkomen gebaseerd op wildbenutting tot het welzijn van de lokale bevolking toont aan dat de wijdverspreide idee dat financiële beloningen tot stand gebracht door geïntegreerde conservatie en ontwikkelingsprogramma's, zoals CAMPFIRE, meestal ontoereikend zijn. Dit leidde tot mijn suggestie dat als we de morele en ethische verplichting hebben om wilde soorten te beschermen, mensen hun economische ambities moeten treffen volgens de principes van Malthus.

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Curriculum vitae

Xavier Poshiwa was born on 2nd of February 1976 in Kwekwe, Zimbabwe. He graduated in 1997 with a BSc in agriculture from the University of Zimbabwe. He received his MSc in animal science from the same university in 2000. For his MSc thesis he worked on validation of the urinary purine derivative technique for estimating microbial protein supply in ruminants. The very same year (2000), he started his research career at Grasslands Research Station, Marondera, Zimbabwe as a rangeland ecologist. Whilst at Grasslands he took part in various collaborative research programmes "Tropical forage and ley legume technology for sustainable grazing and cropping systems in Southern Africa", funded by ACIAR (2000 to 2005); "The effects of ultra- high Stocking density grazing management procedure on the productivity of veld and livestock – a monitoring exercise" (2000) – Sponsored by Cattle Producers Association of Zimbabwe and "Symbionts in Agro forestry systems: What are the long term impacts of inoculation on the growth of *Calliandra calothyrsus* and its intercrops?", funded by the EU (2001 to 2006). In 2007 he joined the Resource Ecology Group, Wageningen University, through the INREF project "Competing claims on natural resources: overcoming mismatches in resource use through multi-scale perspectives". Xavier's project area involved analyzing competing claims for land, and identifying alternative arrangements for wildlife management and livestock farming in Zimbabwe, in close relation with the local stakeholders. The aim of the project was to design sustainable resource distribution and accessibility systems with adequate opportunities for diverse livestock farming and wildlife use in the southeastern Lowveld of Zimbabwe. In April 2010, Xavier was chosen to be a member of the Review Committee tasked with assessment of the Research and Graduate programme for Faculty of Geo-Information Science and Earth Observation (ITC) of the University of Twente, The Netherlands. In 2012, Xavier left Grasslands to join Great Zimbabwe University as a Lecturer only up to December 2012. He is now working for University of Zimbabwe, Marondera College of Agricultural Sciences and Technology as a Lecturer since January 2013.

List of Publications

Poshiwa, X., Heitkönig, I. M.A., Murwira, A., E. C. van Ierland, and H. H. T Prins. Rainfall, primary production and cattle density relationships in southeastern lowveld of Zimbabwe. *Rangeland Ecology and Management*. *In press*

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PE&RC PhD Training Certificate

With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

Review of literature (4.5 ECTS)
- Redressing asymmetry in resource allocation through co-operation among diverse livestock and wildlife systems in South East Lowveld of Zimbabwe (2007)

Writing of project proposal (4.5 ECTS)
- Redressing asymmetry in resource allocation through co-operation among diverse livestock and wildlife systems in South East Lowveld of Zimbabwe (2007)

Post-graduate courses (7.5 ECTS)
- Land sciences: bringing concepts and theory into practice; PE&RC (2007)
- Art of modelling; PE&RC (2008)
- Analysing farming systems and rural livelihoods in a changing world: vulnerability and adaptation; Wageningen Graduate Schools in collaboration with University of Zimbabwe (2008)

Laboratory training and working visits (4.5 ECTS)
- Geographic Information Systems (GIS) and Earth Observation Science (EOS); ITC, Enschede, the Netherlands (2007)
- Ecological relationships in Savannas of SEL of Zimbabwe; University of Kwazulu Natal, South Africa (2010)

Deficiency, refresh, brush-up courses (2.1 ECTS)
- Advanced environmental economics and policy (2007)
- Ecological methods (2007)
- Theories and models in environmental economics (2008)

Competence strengthening / skills courses (1.5 ECTS)
- Competence assessment; PE&RC (2007)
- Techniques for writing and presenting a scientific paper; SENSE (2010)

PE&RC Annual meetings, seminars and the PE&RC weekend (2.7 ECTS)
- PE&RC Introduction weekend (2007)
- Environmental economics and Resource Ecology seminars (2007-2010)

- Competing Claims meetings and workshops (2007-2010)

Discussion groups / local seminars / other scientific meetings (4.5 ECTS)

- Competing Claims discussion groups (2007-2010)
- Forestry, Nature and Conservation (2007-2010)
- Department of Animal Science and Department of Geography seminars at the University of Zimbabwe (2007-2013)

International symposia, workshops and conferences (9 ECTS)

- Kruger National Park Network Meeting (2010)
- AHEAD-GLTFCA Working Group Meeting (2008-2009)
- IX International Rangeland Congress: Diverse rangelands for a sustainable society; Rosario, Santa Fe-Argentina (2011)
- IV International Wildlife Management Congress (IWMC); Durban (2012)

Supervision of 1 MSc student

- Spatial distribution of wildlife-livestock conflict around Gonarezhou National Park Area

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