Sustainable management of semi-arid African savannas under environmental and political change

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Summary

Drylands cover about 40% of the earth's land surface and provide the basis for the livelihoods of 38% of the global human population. Worldwide, these ecosystems are prone to heavy degradation. Increasing levels of dryland degradation result a strong decline of ecosystem services such as the recharge of groundwater resources, the protection of soils from erosion or the provision of grass biomass. In addition, in highly variable semi-arid environments changing future environmental conditions will potentially have severe consequences for productivity and ecosystem dynamics. Hence, global efforts have to be made to understand the particular causes and consequences of dryland degradation and to promote sustainable management options for semi-arid and arid ecosystems in a changing world.

In my thesis I particularly address the problem of semi-arid savanna degradation which mostly occurs in form of woody plant encroachment. At this, I aim at finding viable sustainable management strategies and improving the general understanding of semi-arid savanna vegetation dynamics under conditions of extensive livestock production. Moreover, the influence of external forces, i.e. environmental change and land reform, on the use of savanna vegetation and on the ecosystem response to this land use is assessed. Based on this I identify conditions and strategies that facilitate a sustainable use of semi-arid savanna rangelands in a changing world.

Simulation models provide the opportunity to predict system dynamics under potential future conditions, to test alternative management scenarios and to integrate the knowledge from different disciplines. Thus, I extended an eco-hydrological model to simulate rangeland vegetation dynamics for a typical semi-arid savanna in eastern Namibia. In particular, I identified the response of semi-arid savanna vegetation to different land use strategies (including fire management) also with regard to different predicted precipitation, temperature and CO_2 regimes.

Not only environmental but also economic and political constraints like e.g. land reform programmes are shaping rangeland management strategies. Hence, I aimed at understanding the effects of the ongoing process of land reform in southern Africa on land use and the semi-arid savanna vegetation. Therefore, I developed and implemented an agent-based ecological-economic modelling tool for interactive role plays with land users. This tool was applied in an interdisciplinary empirical study to identify general patterns of management decisions and the between-farm cooperation of land reform beneficiaries in eastern Namibia.

The eco-hydrological simulations revealed that the future dynamics of semi-arid savanna vegetation strongly depend on the respective climate change scenario. In particular, I found that the capacity of the system to sustain domestic livestock production will strongly depend on changes in the amount and temporal distribution of precipitation. In addition, my simulations revealed that shrub encroachment will become less likely under future climatic conditions although positive effects of CO_2 on woody plant growth and transpiration have been considered. While earlier studies predicted a further increase in shrub encroachment due to increased levels of atmospheric CO_2, my contrary finding is based on the negative impacts of temperature increase on the drought sensitive seedling germination and establishment of woody plant species.

Further simulation experiments revealed that prescribed fires are an efficient tool for semi-arid rangeland management, since they suppress woody plant seedling establishment. The strategies tested have increased the long term productivity of the savanna in terms of livestock production and decreased the risk for shrub encroachment (i.e. savanna degradation). This finding refutes the views promoted by existing studies, which state that fires are of minor importance for the vegetation dynamics of semi-arid and arid savannas. Again, the difference in predictions is related to the bottleneck at the seedling establishment stage of woody plants, which has not been sufficiently considered in earlier studies.

The ecological-economic role plays with Namibian land reform beneficiaries showed that the farmers made their decisions with regard to herd size adjustments according to economic but not according to environmental variables. Hence, they do not manage opportunistically by tracking grass biomass availability but rather apply conservative management strategies with low stocking rates. This implies that under the given circumstances the management of these farmers will not *per se* cause (or further worsen) the problem of savanna degradation and shrub encroachment due to overgrazing. However, as my results indicate that this management strategy is rather based on high financial pressure, it is not an indicator for successful rangeland management. Rather, farmers struggle hard to make any positive revenue from their farming business and the success of the Namibian land reform is currently disputable. The role-plays also revealed that cooperation between farmers is difficult even though obligatory due to the often small farm sizes. I thus propose that cooperation needs to be facilitated to improve the success of land reform beneficiaries.

Concluding from the overall results of the eco-hydrological simulation experiments I emphasize the utmost importance of woody plant demography, i.e. the bottleneck at the seedling stage for a thorough understanding of semi-arid savanna vegetation dynamics and sustainable rangeland management. In this context I especially promote the application of fires to prevent shrub encroachment. Furthermore, and derived from the overall results of my thesis, I strongly recommend the application of conservative management strategies. The main reasons for this suggestion are that such precautious and non-opportunistic strategies buffer against uncertainty, both financially and ecologically. Further they facilitate the application of fires as grass fuel is available and are finally also in agreement with the needs of emerging commercial farmers, who are financially not enabled to tolerate highly variable income and erratic costs of herd adjustment.

Chapter 0

General Introduction

0.1 Motivation

Drylands cover about 40% of the earth's land surface and provide the basis for the livelihoods of about 38% of the total global population (Reynolds *et al.* 2007). Globally, these ecosystems are prone to heavy degradation caused by unsustainable land use practices and environmental change (Reynolds *et al.* 2007). Increasing levels of savanna degradation result in a strong decline in ecosystem services such as recharge of groundwater resources, the protection of soils from erosion or the provision of grass biomass for extensive livestock production, which is the main form of land use (Sankaran *et al.* 2005; Scholes 2009). Hence, global efforts have to be made to understand the causes and consequences of this degradation problem and to identify and promote sustainable management options for semi-arid and arid ecosystems in a changing world (UNCCD 1994; Lehmann 2010).

Degradation of semi-arid savanna vegetation mainly occurs in form of a loss of total vegetation cover or as an increase of woody vegetation at the cost of perennial grasses, the so-called shrub encroachment (Skarpe 1990; Jeltsch, Weber & Grimm 2000; Buitenwerf *et al.* 2012). The reasons for this degradation are seen in several interacting (mostly anthropogenic) factors on the local, regional and global scale. First, at the local scale the most frequently mentioned reason for semi-arid savanna degradation is the management of land users. In particular, non-adapted rangeland management causing high grazing pressure (e.g. Walker *et al.* 1981; Skarpe 1991; Graz 2008; Ward & Esler 2011) and the suppression of fires (e.g. van Langevelde *et al.* 2003; Joubert, Rothauge & Smit 2008; Rohde & Hoffman 2012; Joubert, Smit & Hoffman 2012b) are seen as the most important drivers. Second, on the regional scale and especially in Africa, the growing human population exhibits an increasing pressure on the ecosystem (Reynolds *et al.* 2007; Marchant 2010). Especially in southern Africa, this goes along with redistributive land reform programs resulting from political changes during the second half of the last century strongly influence the frame conditions of land use (Adams 1993; Clover & Eriksen 2009). And third, global change in form of climate change and rising atmospheric CO_2 levels, but also changes in global markets and the rising demand in food products are increasingly considered as drivers of savanna rangeland degradation (Reynolds *et al.* 2007; Tietjen *et al.* 2010; Wigley, Bond & Hoffman 2010; Kgope, Bond & Midgley 2010).

Hence, a sustainable and holistic solution that tackles the degradation challenge requires the consideration of ecological, socio-economic and political aspects of the problem and thus requires inter- and transdisciplinary research approaches (Cousins *et al.* 2007; Marchant 2010). However, even in the disciplinary (i.e. solely ecological or economic) debate we do not find a consistent theory of semi-arid savanna ecology or rangeland science.

In the following I will therefore introduce the main concepts of semi-arid rangeland management from an ecological (0.1.1) and a more applied and economic perspective (0.1.2) respectively. Thereafter I will introduce key aspects of global and regional change in southern Africa (0.1.3) followed by a very brief summary of the state of the art of simulation models with regard to the dynamics of semi-arid savannas under environmental change (0.1.4).

0.1.1 Ecological perspective

The basic question of savanna ecology, also referred to as the ‘savanna question’ is how the ecosystem is maintained in a state where scattered trees and shrubs coexist within a matrix of perennial grasses without either grasses or woody plants dominating the system exclusively (e.g. Jeltsch, Weber & Grimm 2000). There has been an extensive debate amongst ecologists trying to get to the bottom of the question to what extent (semi-arid) savannas are equilibrium or non-equilibrium systems (Jeltsch, Weber & Grimm 2000; Briske, Fuhlendorf & Smeins 2003; Vetter 2005). In other words, it remains unclear whether the coexistence of grasses and trees is mediated by niche differentiation and competition, or by stochastic environmental conditions and demographic bottlenecks (Sankaran, Ratnam & Hanan 2004; Graz 2008). However, a growing body of literature suggests that both stochastic and deterministic processes act on the system simultaneously (Joubert, Rothauge & Smit 2008; Meyer, Wiegand & Ward 2009). While there is strong evidence for competition and niche differentiation between tree seedlings and perennial grasses (e.g. Kulmatiski *et al.* 2010; Kambatuku, Cramer & Ward 2011; Kambatuku, Cramer & Ward 2012), and consequently grazing induced promotion of tree growth (Kraaij & Ward 2006; Ward & Esler 2011), there is also strong evidence for the importance of climatic variability for the mortality of perennial grasses or the rarely occurring recruitment of woody plants in semi-arid savannas (Joubert, Rothauge & Smit 2008; Buitenwerf, Swemmer & Peel 2011; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012a). Although these findings could be used to argue for either equilibrium or non-equilibrium dynamics respectively, they allow for a general conclusion: Woody plant populations in semi-arid savannas are strongly limited by a demographic bottleneck during the recruitment from the seedling or sapling stage to the stage of reproductive shrubs or trees. Hereby, both, climatic variability and competition play an important role.

Further recent findings also indicate that fires might play a central role for semi-arid savanna vegetation dynamics (Midgley, Lawes & Chamaille-Jammes 2010; Joubert, Smit & Hoffman 2012b). Fire (see Fig. 0.1) was so far mainly considered to be important for mesic savanna vegetation dynamics, since grass biomass, serving as fire fuel, was repeatedly found to be too low to efficiently control woody plant encroachment in semi-arid and arid systems (Sankaran *et al.* 2005; Higgins *et al.* 2007a). However, fires

Figure 0.1: Veld fire in a semi-arid Namibian savanna. Photo: Larkin Powell

might further tighten the abovementioned bottleneck of woody plants when their occurrence coincides with rare recruitment events of woody plants (Harrington 1991; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012b).

0.1.2 Rangeland management perspective

From a more applied and economic perspective, the question of sustainable semi-arid rangeland management is also under debate. Here, supporters of ´new` and ´classic range science` provide different solutions to the question of how to manage savanna rangelands in a sustainable manner (Illius, Derry & Gordon 1998; Illius & O'Connor 2000; Cowling 2000; Briske, Fuhlendorf & Smeins 2003). While ´classic rangeland science` is founded on the assumption of an equilibrium savanna (see above), the so-called ´new rangeland science` is based on the opposite assumption of a system that is not at equilibrium, but driven by external forces like variable and unpredictable precipitation and fires. In the former case, a conservative stocking regime, with low and stable livestock densities is recommended, so that the competitive balance between trees and grasses is not disturbed. ´New range science` in contrast, recommends tracking forage biomass availability opportunistically. This approach is based on the assumption that perennial grass mortality is not mainly caused by grazing, but by climatic variability. Thus, not using peak biomass availability for livestock production during good years represents a lost opportunity for rangeland managers (Behnke & Scoones 1993). Numerous studies, however, clearly doubt the findings of the ´new rangeland science` and rather promote conservative stocking rates, not only referring to ecological reasons like e.g. building up of fuel biomass for fire or recovery of grasses in good years, but also to economic arguments such as buffering the implications of drought, lower transaction costs for herd size adjustments or risk aversion (Cowling 2000; Sandford & Scoones 2006; Quaas *et al.* 2007; Borner *et al.* 2007; Muller, Frank & Wissel 2007). Nevertheless, although strongly under debate, opportunistic management strategies, have been promoted in the context of southern African land reforms (Cowling 2002) and in the context of communal range management. It remains unclear, to what extent such management strategies are suitable and viable for rangeland management in semi-arid environments from an ecological as well as from an economic perspective.

0.1.3 Semi-arid rangelands under global environmental and regional political change

Consequently, finding a viable solution for sustainable use of semi-arid savannas remains a challenging task, all the more so since many of the abovementioned factors (e.g. variability in rainfall, grazing intensity) might change considerably due to global change (UNCCD 1994; Reynolds *et al.* 2007; Marchant 2010). For semi-arid regions, especially in southern Africa, but also elsewhere, climate change predictions include a number of changes (see Fig. 0.2): while an increase of mean temperature seems inevitable, model predictions regarding the amount of precipitation and its intra- and interannual distribution vary a lot (IPCC 2007; Scheiter & Higgins 2009). Both, temperature increase and changes in precipitation pattern can lead to significant changes in water availability, which is the key driver of semi-arid savanna dynamics (Graz 2008; Tietjen *et al.* 2010). Further, rising levels of atmospheric CO_2 have been suggested to potentially influence the vegetation dynamics of semi-arid savannas by favouring woody plant species with a C_3 carbon pathway over savanna grasses, which predominantly are of the C_4 type (Polley 1997; Bond, Midgley & Woodward 2003; Tietjen *et al.* 2010; Buitenwerf *et al.* 2012).

Further scenarios of change, in particular in southern Africa, arise from the growing human population and political changes of the last century (Marchant 2010). Redistributive land reforms change the conditions for rangeland management throughout southern Africa (Adams 1993; Cousins 2007; Clover & Eriksen 2009; Marchant 2010). The corresponding changes in cultural, social and economic background of new land users can clearly influence the actual land use strategies that are applied (Cousins *et al.* 2007; Prediger, Vollan & Frolich 2011). The main farming goals of these land reform beneficiaries might not predominantly be economic revenues, but also social status, insurance against risks, residential issues or subsistence farming (Cousins *et al.* 2007; Allsopp *et al.* 2007; Werner & Odendaal 2010). However, to my knowledge principal management pattern of land reform beneficiaries, though likely different from those of previous users, have so far not been scientifically assessed in the peer reviewed literature.

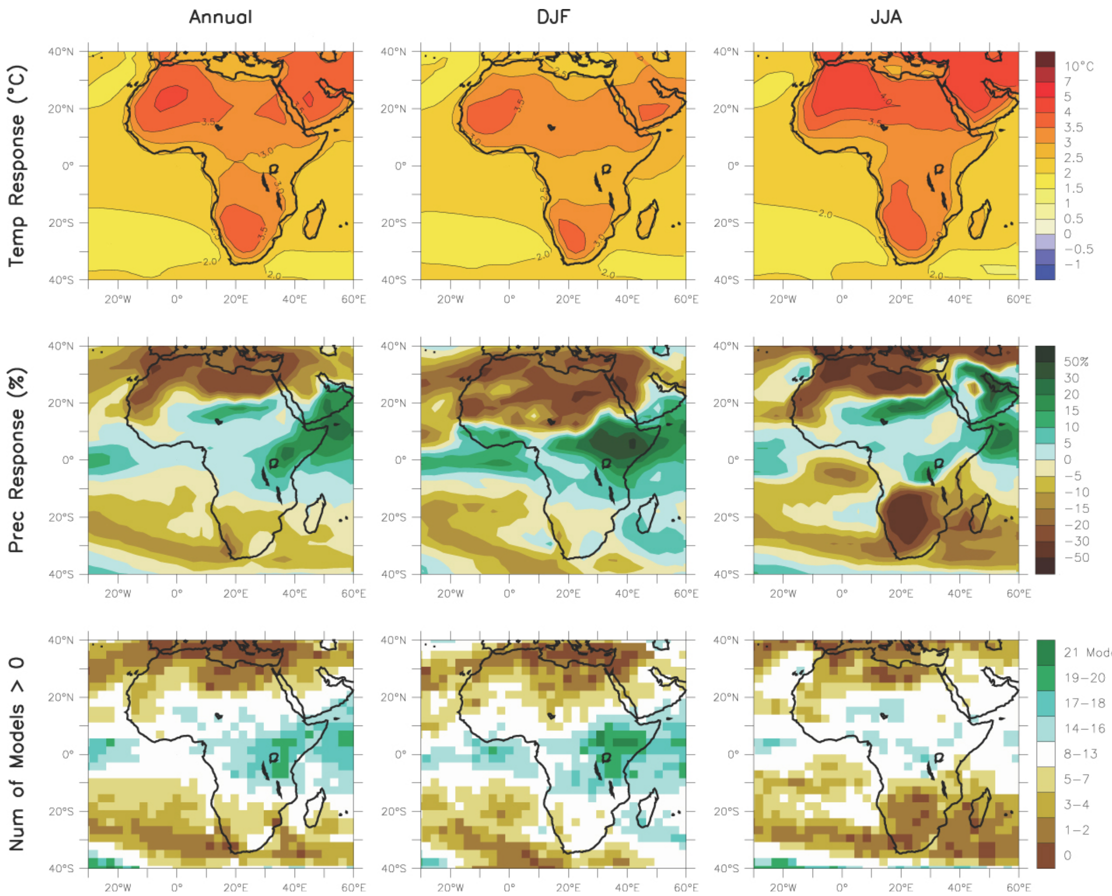

Figure 0.2: Temperature and precipitation changes over Africa from the A1B simulations (IPCC 2007). Top row: Annual mean, DJF and JJA temperature change between 1980 to 1999 and 2080 to 2099, averaged over 21 models. Middle row: same as top, but for fractional change in precipitation. Bottom row: number of models out of 21 that project increases in precipitation.
Source IPCC: Christensen *et al.*(2007)

An important implication of land reform policies is often a significant change in the average size of individual farms (Adams 1993; Werner & Odendaal 2010). For example in southern Africa, commercial semi-arid rangeland businesses usually manage areas of 5000 ha and more (Buss 2006; Olbrich 2011). In contrast, for example farms allotted according to land reform programmes in Namibia are often even smaller than 1000 ha (Werner & Odendaal 2010). Rangeland management at small farms might cause further pressure on the land, as land users cannot benefit from economies of scale (Tomlinson, Hearne & Alexander 2002). Cooperation between neighbouring farmers might offer a solution to this. However, social and cultural differences, as well as a lack of trust between neighbouring farmers might cause such collaboration to fail or not happen at all (Prediger, Vollan & Frolich 2011). Its success however, is of interest from both, an economic and an ecological perspective as it has great potential to improve overall revenue from rangeland management, and consequently can facilitate the application of ecologically sustainable strategies (Falk 2008; Werner & Odendaal 2010). Therefore, it is important to analyse the circumstances leading to success or failure of cooperation in such a complex social-ecological-system (Prediger, Vollan & Frolich 2011).

0.1.4 Models of semi-arid rangelands

Simulation models provide a very useful tool to address questions of the abovementioned complexity. They allow for an integration of knowledge about processes and parameters derived from different disciplines and the projection of results from short term or small scale studies on dynamics on larger spatial or temporal scales. As mentioned above, scenarios of change affecting semi-arid rangelands include processes on a broad range of scales. Thus, simulation models are ideal instruments for an assessment of alternative strategies for rangeland management under change and their respective long-term implications (Jeltsch *et al.* 1996; Muller, Frank & Wissel 2007; Muller *et al.* 2007).

Despite this potential of modelling approaches, there is a lack of models that allow for an assessment of land use impacts on semi-arid savanna vegetation under environmental change. On the one hand many existing models of semi-arid grazing systems do not consider eco-hydrological processes like infiltration, run-off, evaporation and transpiration, which are necessary for an assessment of environmental change effects (i.e. changes in temperature, inter- and intra-annual precipitation pattern and elevated levels of atmospheric CO_2) on vegetation dynamics (see review of Tietjen & Jeltsch 2007). On the other hand, more recent model based studies incorporating the abovementioned eco-hydrological processes in detail generally lack a thorough assessment of the effect of different grazing intensities or land use strategies on the system dynamics (e.g. Tietjen *et al.* 2010; Higgins & Scheiter 2012).

Further, recent empirical work has underlined the utmost importance of the demographic bottleneck in the early life history of encroaching woody plant species in semi-arid and arid savannas (Joubert, Rothauge & Smit 2008; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012a; Joubert, Smit & Hoffman 2012b). The early recruitment of these species was found to depend on above average water availability in two or even three subsequent years (Joubert, Rothauge & Smit 2008; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012a). Once emerged, first and second year seedlings are furthermore very sensitive to fire (Midgley, Lawes & Chamaille-Jammes 2010; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012b). Though the fact that recruitment is drought sensitive is widely acknowledged in literature (Jeltsch, Weber & Grimm 2000; Schwinning & Sala 2004; Meyer *et al.* 2007b), the extent of this sensitivity is often not considered or underestimated in semi-arid savanna models (e.g. van Langevelde *et al.* 2003; Meyer *et al.* 2007a; de Knegt *et al.* 2008; Tietjen *et al.* 2010; Calabrese *et al.* 2010). Also the strong effect of fire on seedling survival is not included in existing rangeland models (van Langevelde *et al.* 2003; Higgins *et al.* 2007b; de Knegt *et al.* 2008; Tietjen *et al.* 2010) (van Langenvelde 2003, Tietjen et al 2010, Higgins 2007, de Knegt 2008). Notably, many models include fire, but focus on the effect of fire on older saplings and adult trees (van Langevelde *et al.* 2003; Quaas *et al.* 2007; Higgins *et al.* 2007b).

To summarize, to my knowledge no model for the simulation of semi-arid savannas exists, that allows for the simultaneous simulation of climate change effects and land use strategies (see review of existing models byTietjen & Jeltsch 2007) while explicitly accounting for the specific conditions of woody species recruitment.

0.2 Objectives of the thesis

The overarching goal of my thesis is to improve the understanding of semi-arid savanna vegetation dynamics under conditions of extensive livestock production. Exemplarily for other semi-arid rangelands, I want to assess the influence of external forces and conditions on the land use in southern African semi-arid savannas and on the response of savanna vegetation to this land use. Based on this understanding I want to identify conditions and strategies for sustainable use of semi-arid savanna rangelands in a changing world.

I use simulation modelling to integrate the knowledge and perspectives of ecology, hydrology and economics to meet the requirements coming along with the complexity of this system, which is influenced by strong environmental variability and the dynamics of and feedbacks between soil-water, vegetation and land use. In particular, I want to:

1. Adapt and extend an eco-hydrological model to simulate rangeland dynamics under a broad range of environmental and land use scenarios.
2. Assess rangeland management strategies under climate change based on the explicit consideration of important demographic features of encroaching woody plant species in semi-arid savannas as indicated by recent empirical research.
3. Assess the potential of fire as a tool for semi-arid rangeland management.
4. Develop and implement an agent-based ecological-economic modelling tool for interactive role plays. This tool, which should bridge the gap between ecological and socio-economic research, should be applied in an interdisciplinary empirical study to identify general patterns of management decisions and the between-farm cooperation of land reform beneficiaries in eastern Namibia.

0.3 Structure of the thesis

To achieve the objectives of the thesis I extend an existing eco-hydrological vegetation model by Tietjen *et al.* (2010), which I parameterize and evaluate for a semi-arid southern African savanna site. I have chosen a semi-arid study region in eastern Namibia (see Fig. 3.1) for several reasons: The vegetation found in this region is a typical semi-arid savanna of the Central Kalahari type, the region is known to be very important for livestock production, threatened by shrub encroachment and numerous farms within the region have been redistributed according to the Namibian land reform programme (Mendelsohn *et al.* 2002; Werner & Odendaal 2010). I have chosen the simulation model *EcoHyD* (Tietjen *et al.* 2010) as the basis for my study, because it already included the eco-hydrological processes necessary to simulate effects of climate change and CO_2 on soil-water-vegetation feedbacks and dynamics. However, key amendments of the model are necessary for a successful handling of the central questions of my thesis. These major adjustments are: (i) inclusion of the specific demography of woody vegetation, i.e. the recruitment bottleneck of savanna tree species, according to recent empirical findings (Kraaij & Ward 2006; Joubert, Rothauge & Smit 2008; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012a; Joubert, Smit & Hoffman 2012b), a realistic representation of (ii) livestock herbivory and (iii) biomass production, as these are the processes determining the typical interaction between land user and the ecosystem in extensive rangeland systems. Therefore, (iv) annual grasses should be additionally included in the model as they contribute significantly to available forage biomass under certain conditions, i.e. when rangelands are in poor condition and perennial grass biomass production is very low.

In **chapter 1**[1] of my thesis, I describe the new version of the eco-hydrological model with regard to the abovementioned changes. I validate the model against pattern of shrub cover and biomass production found in the study region. Then, forming the main part of this chapter, the model is used to assess the response of this semi-arid savanna to a range of grazing intensities under a variety of scenarios of environmental change. The climate predictions for the southern African study region but also for other semi-arid regions in the world are ambiguous (IPCC 2007). While an increase in temperature and atmospheric CO_2 levels are generally expected, predictions differ a lot with regard to changes in precipitation. Such changes can occur in the mean annual precipitation received (MAP), which according to different climate predictions will increase, decrease, or remain constant. In addition, some but not all global and regional climate models predict changes in inter- and/or intra-annual distribution of rainfall. In order to assess the respective future conditions of livestock production I thus simulate a range of grazing intensities for all possible combinations of the abovementioned changes in climatic conditions. In this regard, I am especially interested in the response of woody vegetation, which might

[1] Published as Lohmann, D., Tietjen, B., Blaum, N., Joubert, D.F. & Jeltsch, F. (2012) "Shifting thresholds and changing degradation patterns: climate change effects on the simulated long-term response of a semi-arid savanna to grazing". *Journal of Applied Ecology.* **49** pp. 814-823

respond strongly to scenarios of change due to the explicit inclusion of the climate sensitive bottleneck at the seedling stage.

In **chapter 2**[2] of my thesis I want to assess the long-term effect of a fire management strategy that aims at narrowing down the recruitment bottleneck of encroaching woody species on semi-arid savanna vegetation dynamics and long-term productivity. In contrast to other models, focussing on fire-driven mortality of saplings and adult plant individuals (van Langevelde *et al.* 2003; Meyer *et al.* 2007a; Higgins *et al.* 2007b), I simulate a strategy of fire application that is targeted on events of tree seedling emergence, as the latter was found to be restricted to series of years of above average soil water availability (Joubert, Smit & Hoffman 2012a). Further, I want to compare different modes of this fire management with regard to their long-term efficiency and applicability in rangeland businesses.

In **chapter 3**[3] I present an interdisciplinary approach for empirical ecological-economic research. I develop and implement an ecological-economic model in cooperation with economists from Marburg University, which is used for simulation-based role plays. The model simulates one or several farms (fixed and running costs, revenues from trading, herd dynamics, animal weight gain) and the vegetation on these farms (mainly grass biomass production). Vegetation dynamics are derived and scaled up to the landscape scale with a state-and-transition approach (Westoby, Walker & Noy-Meir 1989; Popp *et al.* 2009) from the eco-hydrological model (developed in chapter 1) in dependence of vegetation state, precipitation, and grazing intensity. The resulting ecological-economic and agent-based farm model allows for the interactive simulation of several farms and is applied in interactive role-plays with Namibian land reform beneficiaries to test how they adjust their herd management according to environmental and economic variables. I discuss how the farmer's management is determined by these variables and what this implies with regard to rangeland science. I conclude this chapter with a discussion of the causes and consequences of their management decisions against the background of the redistributive land reform in Namibia.

In the general discussion (**chapter 4),** I first discuss the general implications of my findings for savanna ecology and the response of semi-arid savannas to land use. I draw conclusions for semi-arid savanna ecology and management with special regard to the fundamental importance of the woody plant demographic bottleneck, also as distinguished from mesic savannas. Second, I newly introduce findings from a study that was conducted within the research described in chapter 3, within which the contribution of the land reform beneficiaries to collective water infrastructure management was assessed. The implications of these results for the Namibian land reform are discussed followed by a short evaluation of the model-based approach for interactive role plays. I

[2] Submitted to *Journal of Applied Ecology* as Lohmann, D., Tietjen, B., Blaum, N., Joubert, D.F. & Jeltsch, F. "Prescribed fires as a tool for sustainable management of semi-arid savanna rangelands".
[3] Submitted to *Journal of Arid Environments* as Lohmann, D., Falk, T., Geissler, K., Blaum, N. & Jeltsch, F. "Determinants of semi-arid rangeland management in a land reform setting in Namibia".

close my discussion with a short conclusion drawn from this thesis with regard to the management of semi-arid rangelands in general.

Due to the cumulative form of this thesis, parts of it (especially chapters 1-3) are written in first-person plural because they are co-authored and are published in or submitted for publication to peer reviewed journals. However, as I am the lead author of all included publications, I have performed the main work described in these chapters, and the views expressed in this thesis are mine. The work described in chapter 3 was performed in cooperation with Dr. Thomas Falk, by the time of the study member of the Institute for Co-operation in Developing Countries at the Philipps University in Marburg. Dr. Falk was responsible for most of the field work, i.e. the role-plays performed with the model described in chapter 3. Nonetheless, the main work for the study described in chapter 3 was conducted independently by me. From further results of the role plays another manuscript[4] was written. The study described therein deals with the cooperation of the surveyed Namibian land reform beneficiaries with regard to collaborative water infrastructure maintenance. Although this manuscript was predominantly written by Dr. Falk and only co-authored by me it is added as an appendix to my thesis (see Appendix A) as it is based on the model that I developed and the results presented therein are of high relevance for the interpretation of the findings presented in chapter 3.

[4] Manuscript attached to this thesis as Appendix A, submitted to *Ecological Economics* as Falk, T., Lohmann, D., Azebaze N. "The impact of heterogeneity in endowments and norms on common good provision".

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Chapter 1

Shifting thresholds and changing degradation patterns: climate change effects on the simulated long-term response of a semi-arid savanna to grazing[1]

Summary

1. The complex, non-linear response of dryland systems to grazing and climatic variations is a challenge to management of these lands. Predicted climatic changes will impact the desertification of drylands under domestic livestock production. Consequently, there is an urgent need to understand the response of drylands to grazing under climate change

2. We enhanced and parameterized an ecohydrological savanna model to assess the impacts of a range of climate change scenarios on the response of a semi-arid African savanna to grazing. We focused on the effects of temperature and CO_2-level increase in combination with changes in inter- and intra-annual precipitation patterns on the long-term dynamics of three major plant functional types.

3. We found that the capacity of the savanna to sustain livestock grazing was strongly influenced by climate change. Increased mean annual precipitation and changes in intra-annual precipitation pattern have the potential to slightly increase carrying capacities of the system. In contrast, decreased precipitation, higher inter-annual variation and temperature increase are leading to a severe decline of carrying capacities due to losses of the perennial grass biomass.

4. Semi-arid rangelands will be at lower risk of shrub encroachment and encroachment will be less intense under future climatic conditions. This finding holds in spite of elevated levels of atmospheric CO_2 and irrespective of changes in precipitation pattern, due to the drought sensitivity of germination and establishment of encroaching species.

5. Synthesis and applications. Changes in livestock carrying capacities, both positive and negative, mainly depend on the highly uncertain future rainfall conditions. However, independent of the specific changes, shrub encroachment becomes less likely and in many cases less severe. Thus, managers of semi-arid rangelands should shift their focus from woody vegetation towards perennial grass species as indicators for rangeland degradation. Furthermore, the resulting reduced competition from woody vegetation has the potential to facilitate ecosystem restoration measures such as re-introduction of desirable plant species that are only little promising or infeasible under current climatic conditions. On a global scale, the reductions in standing biomass resulting from altered degradation dynamics of semi-arid rangelands can have negative impacts on carbon sequestration.

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1.1 Introduction

Dry rangelands worldwide are prone to concurrent high levels of human intervention and high climatic variability (Gillson & Hoffman 2007; Reynolds *et al.* 2007). The consequence is often desertification i.e. a decrease in vegetation cover and/or a change of vegetation composition with a subsequent loss of the systems productivity (UNCCD 1994). In many regions, land use intensities are expected to further increase due to global population growth, leading to even higher stress levels in rangeland systems (Reynolds *et al.* 2007; UNCCD 1994). At the same time, water availability is likely to be altered as a result of the predicted temperature increase and changes in precipitation amount and variability (IPCC 2007; Scheiter & Higgins 2009). Thus two of the most important external drivers of dryland ecosystems, climate and land use are likely to change simultaneously. An understanding of the underlying mechanisms of dryland dynamics and the drylands' response to predicted changes is of utmost importance (Tietjen & Jeltsch 2007).

Many studies agree that dryland systems such as semi-arid savannas are not at equilibrium, but are driven by stochasticity and variability in environmental drivers such as precipitation, fire and herbivory (Fensham, Fairfax & Archer 2005; Gillson & Hoffman 2007). In their response to land use, semi-arid savannas can exhibit pronounced thresholds, and consequently, degradation processes are non-linear and difficult to anticipate (Gillson & Hoffman 2007; Jeltsch, Weber & Grimm 2000; Vetter 2005).

Conceptual models addressing the mechanisms behind the coexistence of grasses and woody vegetation have focussed on either competitive interactions between trees and grasses or the demographic bottlenecks related to tree establishment (Sankaran, Ratnam & Hanan 2004). Recently it was suggested, that both mechanisms play an important role on different levels or scales of the system (Meyer, Wiegand & Ward 2009; Sankaran, Ratnam & Hanan 2004). However, both, competitive as well as demographic processes are influenced by the two central drivers of the system, namely land use and climatic variability.

In particular, livestock grazing influences both competitive interactions between grasses and woody species as well as woody species recruitment. Herbivores were found to increase the recruitment success of many encroaching species by enhancing tree seed dispersal and by increasing germination rates after ungulate gut passage of the seeds (Miller 1995; Tews, Schurr & Jeltsch 2004). More importantly, the selective removal of grass biomass leads to reduced competition from grasses, resulting in an increased establishment rate of shrubs (Hiernaux *et al.* 2009; Kraaij & Ward 2006; Ward & Esler 2011).

The influence of variability in precipitation is dependent on the temporal scale that is considered. On an intra-annual scale, the timing of precipitation can be crucial for plant phenology. The size of precipitation events has a strong impact on available soil moisture and related processes such as growth, mortality or establishment (see pulse theory:

Schwinning & Sala 2004): on the one hand, sufficiently large precipitation events are needed in order to obtain deep infiltration and thus less losses through evaporation, on the other hand, intense rainfall events can lead to high water losses through runoff, if the soil texture, structure and roughness do not allow for fast infiltration (Fensham, Fairfax & Archer 2005; Tietjen, Zehe & Jeltsch 2009; Tietjen *et al.* 2010). Variations on the inter-annual scale increase the frequency of multi-year droughts and can therefore cause severe diebacks of grasses. Juvenile shrubs and seedlings with low competitive strength especially benefit from such situations (Buitenwerf, Swemmer & Peel 2011; Ward & Esler 2011). Additionally, inter-annual variation causes consecutive years with above average precipitation which enable episodic mass recruitment of woody species (Joubert, Rothauge & Smit 2008; Kraaij & Ward 2006).

Inter- and intra-annual precipitation patterns are predicted to change in semi-arid and arid environments in the course of climate change, leading to an increased number of extreme events (IPCC 2007). Furthermore, predictions include increases in mean temperature and atmospheric CO_2 levels as well as changes in mean annual precipitation (MAP) for many dry regions (IPCC 2007). Unfortunately, climate model predictions are subject to high uncertainty, therefore the direction of change can vary between different climate models (IPCC 2007). Increased evaporation and transpiration rates will lead to decreased water availability. Negative and positive changes in MAP can mitigate or exacerbate these effects respectively. The water use efficiency in turn can be positively influenced by increased levels of atmospheric CO_2 (Drake, Gonzàlez-Meler & Long 1997; Scott *et al.* 2006), which potentially favours woody vegetation over grasses due to their different carbon pathways (Bond, Midgley & Woodward 2003; Kgope, Bond & Midgley 2010).

In order to systematically analyse the influence of a realistic range of possible climatic changes on the response of a dryland ecosystem to land use we exemplarily address degradation in an African camelthorn savanna. In such semi-arid savannas degradation in general was found to occur when certain long-term grazing intensities have been exceeded (Joubert, Rothauge & Smit 2008; Weber & Jeltsch 2000). In most cases, the degradation pattern was shrub encroachment, where perennial grass abundance is decreased and woody plant species become dominant (e.g. Skarpe 1990). The consequence is a decrease in productivity and biodiversity of the system (Blaum *et al.* 2009).

We assessed savanna rangeland response for a broad range of grazing intensities and climate change scenarios applying a modified ecohydrological model (Tietjen *et al.* 2010). We simulated changes in intra- and inter-annual patterns of precipitation as well as modifications of long-term means. Further, we assessed the effects of an increase in the atmospheric CO_2 level and mean temperature.

In particular, we address the following questions: (1) How do changes in different climate patterns impact the long-term threshold of livestock density up to which a savanna persists in a non-degraded state and what are their combined effects? (2) If degradation occurs, does the pattern of degradation in terms of relative and absolute abundance of shrubs, perennial- and annual grasses differ between the different climate change scenarios?

1.2 Materials and Methods

1.2.1 Study area

The model was applied to a Namibian Acacia-tree-and-shrub savanna of the Central Kalahari type (Mendelsohn *et al.* 2002) as it is found at the governmental research station at Sandveld (latitude 22°02'S longitude 19°07'E). This area was used for livestock production for about 80 years and has been a research farm since the late 1960s. It is considered a typical shrub-encroached savanna that is invaded by Acacia mellifera BENTH., though the level of degradation is still considered to be moderate.

Precipitation falls during summer months (September to April) and has a high inter- and intra-annual variation. The MAP measured at Sandveld during 23 years (1986–2008) was 408 mm with a standard deviation of 180 mm (Rothauge 2006). The annual mean temperature is about 19 °C, with monthly means ranging from 12 °C (July) to 25 °C (January). The area is characterized by loamy Kalahari sand soils. The topography of the area is very flat with a mean height of 1520 m.

1.2.2 Model description

Our model is based on the grid-based ecohydrological dryland model EcoHyD (Tietjen *et al.* 2010). It is a combination of a process-based savanna vegetation sub-model calculating the biweekly growth of two plant functional types (shrubs and perennial grasses) and a process-based hydrological sub-model, calculating daily moisture dynamics in two soil layers (Tietjen, Zehe & Jeltsch 2009) for 30x30 grid cells each representing 5x5 m² patches resulting in a total simulated area of 2.25 ha. No changes have been made to the hydrological sub-model, but the vegetation model was modified to address land use related questions.

The vegetation sub-model comprises the processes growth, mortality (induced by drought or senescence), competition for water and space, dispersal and establishment for shrubs, perennial grasses and annuals. In addition to the model described by Tietjen *et al.* (2010), we introduced a grazing/browsing algorithm, and the algorithms for dispersal and establishment were changed to account for grazing effects. We included annual grasses as a third functional type in addition to perennial grasses and woody vegetation, since annual vegetation may contribute considerably to the diet of livestock (Rothauge 2006; Tainton 1999). We furthermore changed the rules for establishment and mortality of the woody vegetation, taking into account the ecology of the dominant encroacher species in the study area, i.e. A. mellifera. All simulations were conducted using the parameter set given in Appendices 1.A and 1.B. A comprehensive description of the model rules can be found in Appendix 1.C.

Table 1.1: Climate scenarios and respective combinations of simulated changes in precipitation pattern. Note: scenario c represents simulations with current climatic conditions (control)

Scenario name	Changes in precipitation pattern			Increased Temperature & CO_2
	±10% MAP (±10)	**Increase in size of large rainfall events (s)**	**Increase in inter-annual variation (v)**	
c	–	–	–	–
p	–	–	–	+
s	–	+	–	+
v	–	–	+	+
–10	-10	–	–	+
+10	+10	–	–	+
sv	–	+	+	+
s–10	-10	+	–	+
v–10	-10	–	+	+
sv–10	-10	+	+	+
s+10	+10	+	–	+
v+10	+10	–	+	+
sv+10	+10	+	+	+

1.2.3 Simulations

In this study we applied a full factorial design, to simulate all possible combinations of the implemented levels of land use intensity, precipitation pattern, temperature and atmospheric CO_2 (see table 1.1). Unless otherwise noted, all results are based on 25 repeated simulations of 200 years, each with unique time series of stochastic precipitation and temperature according to the respective climate scenario (see below).

Initial grass cover was randomly drawn from a uniform distribution with values of perennial grass canopy cover between 40% and 80% per cell. Shrub canopy cover was randomly distributed in 20% of all cells with values of 1–80% per cell. In cells with woody vegetation, initial grass cover was limited to 5%.

A model spin-up was performed for each simulation for 50 years with the respective climate scenario and without livestock grazing, to allow for the system to reach a steady state before testing management scenarios.

Land use

Land use scenarios were implemented as different densities of livestock that were kept constant over time. Scenarios range from 3.3 large stock units (LSU) per 100 hectares to 10 LSU 100 ha-1 plus an additional scenario representing no grazing by livestock. We use the definition of a LSU as a 450 kg live weight cattle (Meissner 1982). Natural grazing and browsing of game are not simulated but are implicitly included since the presence of game presumably influenced all data that was used for model parameterization.

Precipitation pattern

We generated precipitation and temperature time-series with hourly resolution as described in Tietjen *et al.* (2010). The derived time-series showed no significant difference in mean value, distribution and variance compared to measured annual precipitation in Sandveld during 1983–2007.

We implemented three types of changes in precipitation patterns following the range of possible changes given by the global climate models (GCMs) in the IPCC report (2007) and available studies of regional climate models (RCMs) for southern Africa (Engelbrecht, McGregor & Engelbrecht 2009, Tadross, Jack & Hewitson 2005): (1) Change of MAP by ±10% (as GCMs and RCMs disagree on the sign of changes). For this, the size of all generated rainfall events was changed by ±10% respectively (2) A change in the intra-annual distribution of precipitation towards more large events and less small events. This was generated by shifting the total precipitation amount of events falling into the lowest quantile (10%) to precipitation events falling into the upper 90% quantile, while keeping the MAP fixed (as given in Tietjen *et al.* 2010). (3) Change of the inter-annual distribution of precipitation towards an increased probability of above (>500mm) and below (<300mm) average precipitation years. For this we reshaped the probability distribution of annual precipitation (clustered in 100mm classes) by reducing probabilities of the two precipitation classes embracing the MAP (300–400 mm and 400–500 mm). The subtracted probability was added to below average and above average precipitation classes respectively revealing a new, stretched distribution with an unchanged median. This increased the coefficient of variation of annual precipitation by 0.1 to 0.45.

In addition, all possible combinations of these three changes were implemented.

Temperature and CO_2

Our temperature and CO_2 scenarios are based on the climate change scenarios for 2080–2099 given by the A1B scenario of the IPCC (2007) report. This includes an increase in annual mean temperature by 3.5 °C and increased atmospheric CO_2–levels up to 700 ppm.

The effect of increased CO_2-levels on the system is regarded in two ways. Following the implementation of Tietjen *et al.* (2010) that is based on free-air CO_2 enrichment experiments (Ainsworth & Long 2005), we decreased transpiration from the lower soil layer by 40% and secondly increased the potential growth rates of plants. The latter was found to be stronger for plants using a C3-pathway for carbon fixation in photosynthesis than for the C4-pathway, since C4-photosynthesis is often saturated or nearly saturated under ambient conditions (Morgan *et al.* 2004). Since most grasses in tropical savannas feature C4-pathways (Polley 1997), we assume that CO_2-enrichment leads to an increase in potential growth rate of grasses by 30% and potential growth rate of shrubs by 90% according to Bond, Midgley & Woodward (2003). Note that these rates are potential growth rates, and growth is additionally strongly limited by water availability and competition.

Sensitivity analysis and validation

In order to improve confidence in simulation results we conducted a sensitivity analyses by varying all parameters by ±10% and ±20%. The model output was furthermore validated against three empirical patterns (relation of annual precipitation to grass biomass production of African savannas, shrub cover and biomass production of the study site). Details of these analyses can be found in Appendix 1.D.

1.3 Results

1.3.1 Present dynamics and effects of livestock grazing

Under present climatic conditions, the simulations without livestock grazing lead to a savanna system that is in a stable state, with especially the annual and perennial grasses strongly responding to fluctuations in the highly variable precipitation (Fig. 1.1a). Perennial grass cover is high (mean value: 65%, SD: 11.3), while shrub cover and annual grass cover are low (mean values: 8%, SD 1.2 and 7%, SD 3.2, respectively), leading to a dense grass matrix with sparse shrubs. Continuous high grazing, however, deteriorates the system and drives it to another dynamic state (Fig. 1.1b) that is dominated by high levels of shrub cover (mean value 29%, SD: 8.5), while perennial grasses are nearly absent (mean value < 1%). In this shrub encroached state, annual grasses vary strongly with precipitation and reach much higher maximum values of cover than in scenarios without livestock grazing (mean value 19%, SD: 7.7).

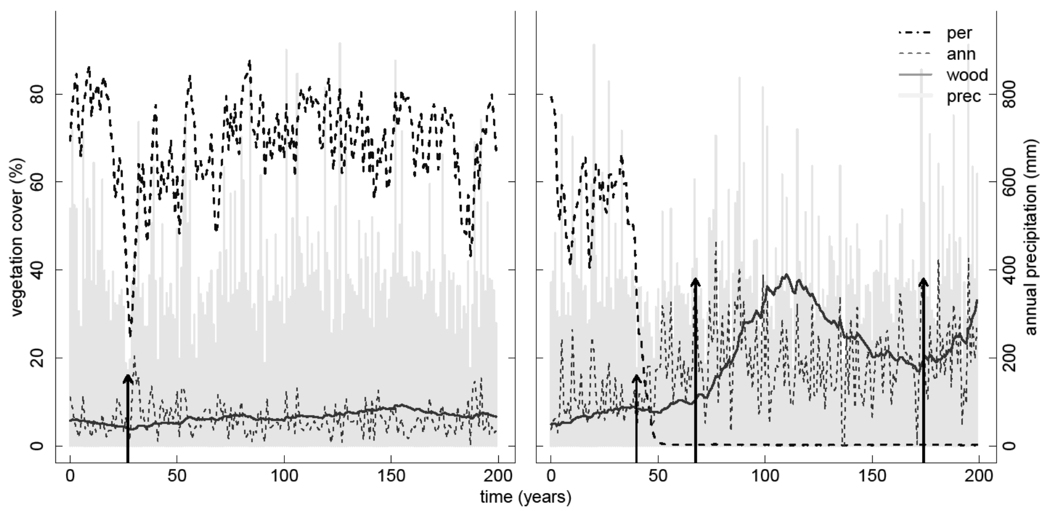

Figure 1.1: Time series of mean vegetation cover [%] and respective annual precipitation [mm] for showcase single runs of 200 years for simulations (**a**) without livestock grazing and (**b**) with intense livestock grazing (8.3 LSU 100 ha^{-1}). Arrows indicate examples of precipitation events causing grass collapse or shrub recruitment.

In general, the perennial grass matrix is severely threatened by rarely occurring severe droughts or series of years with below average precipitation (e.g. Fig. 1.1a years 26–27). If grazing pressure is high in addition to such drought events, this might lead to a complete loss of perennial grasses (see Fig. 1.1b, year 40). However, the loss of perennial grasses does not immediately lead to shrub encroachment, but causes an interim state dominated by annual grasses (see Fig. 1.1b, years 50 et seqq.). If a series of above average precipitation years occurs during this state, events of major shrub establishment success can lead to a shrub encroached system (see Fig. 1.1b, year 67–68 & 173–174). Once the shrubs are established, their cover varies over time as a result of age- and drought-dependent mortality.

1.3.2 Effects of climate change

The simulated scenarios show that climate change will have strong direct and indirect effects on soil moisture. All scenarios lead to decreased mean soil moisture content of the upper soil layer, whereas the change in moisture content of the lower layer shows no consistent trend, but depends on the respective precipitation pattern (Fig. 1.2). Compared to current climatic conditions (scenario c), both the moisture in the upper and lower layer are decreased by simultaneous increases in temperature and CO_2 (scenario p). If, in addition to increased temperature and CO_2, precipitation falls in fewer, but larger events (scenario s) or if MAP is increased (scenario +10), soil moisture in the lower layer increases. In contrast, a reduced MAP (scenario -10) leads to decreased lower layer soil moisture.

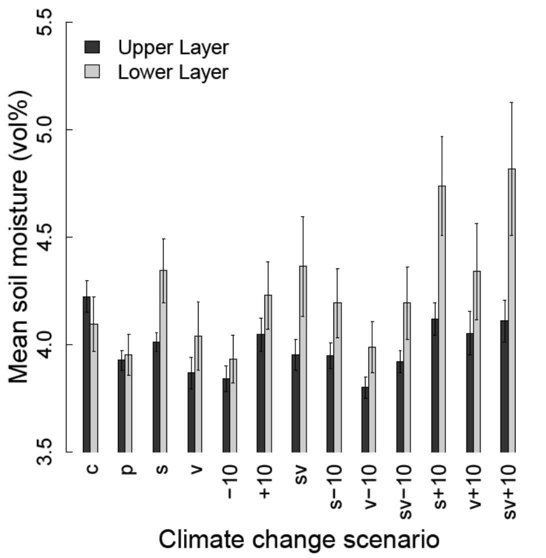

Figure 1.2: Mean and standard error of 25 replicate simulations of initial soil moisture (t=10) during the growing season in upper and lower layer without livestock grazing (see table 1 for scenario description). Scenario c shows results for the scenario with current climatic conditions.

Furthermore, altered climatic conditions impact vegetation cover (Fig. 1.3). Perennial grass cover increases with an increase in size of large precipitation events (s) or in MAP (+10) and is reduced when both a decrease in MAP and increased inter-annual variation of precipitation are assumed (v-10). The pattern is very similar for shrubs, although relative differences between the simulated scenarios are much more pronounced. An increase in size of large events (s) and an increased MAP (+10) have positive implications for the abundance of shrubs, also when combined with increased inter-annual variation. Scenarios of reduced MAP (-10) lead to decreased shrub cover, while an increase in inter-annual variation (v) has no clear effects.

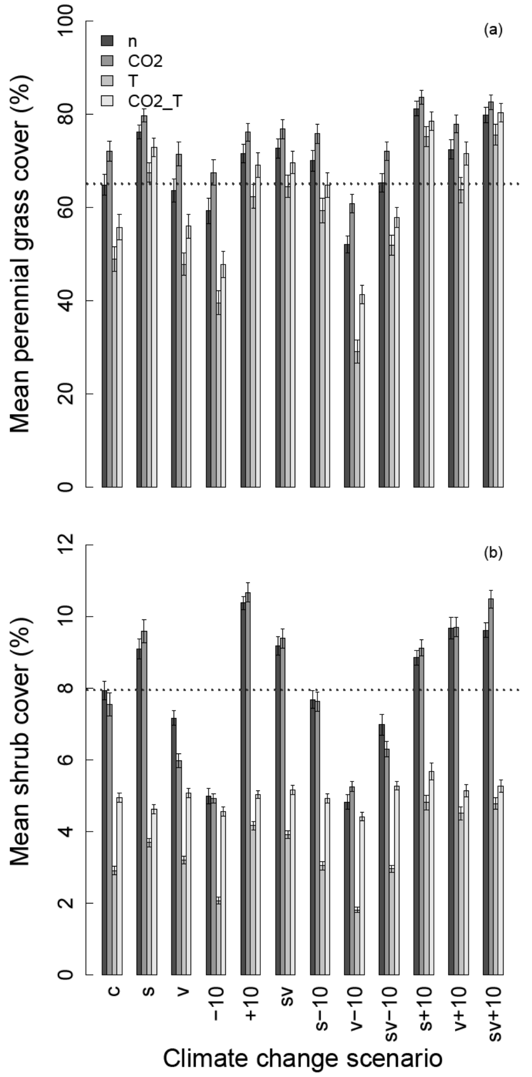

Figure 1.3: Mean and standard error of final cover of (a) perennial grasses and (b) woody vegetation for different precipitation scenarios (see table 1) from 25 repeated simulations of 200 years without grazing; Different bars depict results for simulations with no changes (n) or singular increase in either mean annual temperature (T) or CO_2-levels (CO2) or an increased level of both at a time (CO2_T); the dashed line delineates cover resulting from simulations of current climatic conditions.

Compared to these effects of altered precipitation patterns, the effects of increased CO_2-levels on shrub and perennial grass cover are moderate. Grasses tend to slightly benefit from increased CO_2 for all precipitation scenarios, while shrubs show no pronounced or clear response, especially since they are limited by competitive interactions with dominant perennial grasses. In contrast, increased mean annual temperature has a pronounced negative impact on both, shrub and perennial grass cover in all precipitation scenarios. Consequently, the concurrent effects of CO_2-level and temperature increase are dominated by the unambiguously stronger temperature effects, thus resulting in decreased values of shrub as well as perennial grass cover compared to the respective reference scenario (i.e. the same precipitation pattern).

To further evaluate the importance of water availability in the upper soil layer and related germination and early establishment of shrubs for long-term vegetation effects we compared initial mean soil moisture with final shrub cover of simulations under high livestock grazing intensities (Figure 1.4). Since we are interested in the cause of final vegetation pattern rather than its feedback on soil moisture we use initial soil moisture

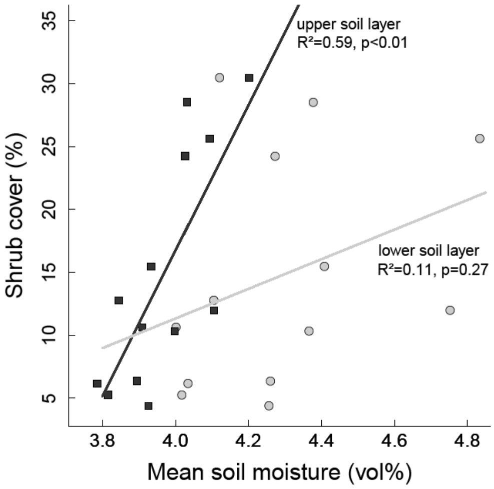

Figure 1.4: Final mean values of shrub cover (t=200) and regression dependent on initial mean soil moisture (t=10) derived from 25 simulations with intense livestock grazing (8.3 LSU 100 ha^{-1}) for all 13 climate scenarios. Squares show data for upper soil layer and circles for lower soil layer.

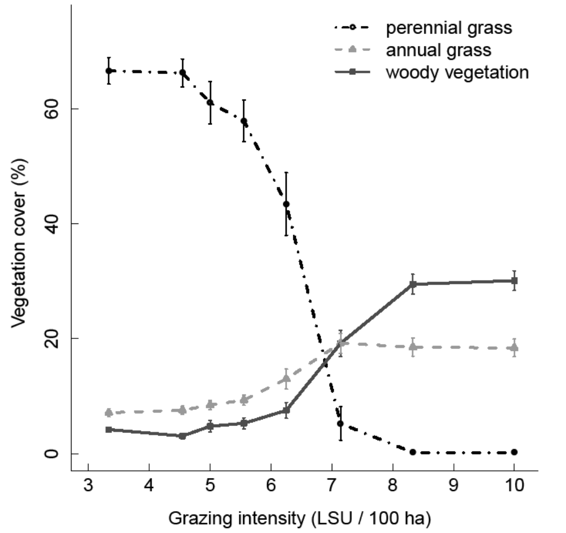

Figure 1.5: Mean and standard error of final cover of perennial and annual grasses and shrubs at different livestock densities. Results were derived from 25 repeated simulations of 200 years under current climatic conditions.

values (mean of 25 replicates at t=10) for this correlation. Interestingly, initial soil moisture in the upper layer and final shrub cover show a clear correlation (R^2=0.59; P<0.01). In contrast, the correlation of final shrub cover to lower layer soil moisture was not significant (R^2=0.11; P=0.27). Grasses were similarly linked to moisture in both layers (R^2=0.55 and R^2=0.78 respectively P<0.001).

1.3.3 Combined effects of climate change and land use

In general, the response of the system to grazing follows a threshold behaviour regarding grazing intensity: under current climatic conditions, the system remains in its original state with stocking rates up to about 6 LSU 100ha-1 (Fig. 1.5). If grazing intensity increases, the probability of degradation towards an increased level of shrubs and annuals and a decreased level of perennial grasses increases rapidly (Fig. 1.5, compare also with single runs in Fig. 1.1).

This response of the system to livestock grazing can be altered substantially by the simulated climate change scenarios (Fig. 1.6a–l) with respect to both, the collapse of the perennial grass matrix and the encroachment of shrubs.

The productivity of perennial grasses and hence the grazing resistance of the system (i.e. the grazing threshold where degradation occurs) are reduced by the simulated increase in temperature (Fig. 1.6e). This situation is worsened compared to current climate conditions if the inter-annual variability of precipitation is increased (Fig. 1.6g) or MAP is decreased (Fig. 1.6a) or both (Fig. 1.6c). However, negative effects of temperature increase can be compensated by the positive effects of an increased MAP (Fig. 1.6i) or increased intra-annual variation of precipitation leading to more large, and thus effective, precipitation events (Fig. 1.6f). A combination of such changes even leads to increased grass cover despite the negative effects of increased temperature (Fig. 1.6j). In general, annual grass cover is negatively related to perennial grass cover and therefore increases with higher livestock grazing intensity.

The response of woody vegetation to land use is also influenced by climate change. As described above, maximum shrub cover values under high grazing pressure are often reduced drastically if the mean water content of the upper soil layer is decreased (Fig. 1.4). While a high grazing intensity leads to shrub encroachment, under present climatic conditions (Fig. 1.5) climate change scenarios that reduce upper soil moisture levels lead to the dominance of annual vegetation or increased bare ground instead of increased shrub cover (Fig. 1.6a–h). The level of shrub encroachment resulting from grazing is highly dependent on MAP: If, in addition to higher temperatures and increased CO_2 levels, MAP decreases by 10%, none of the changes in the intra- or inter-annual rainfall distribution leads to shrub encroachment. If MAP remains the same, we see slight shrub encroachment if grazing intensity increases, but annual vegetation dominates the system. In contrast, high grazing in combination with an increase in MAP by 10% is leading to shrub dominance on levels which are similar to encroachment under current climatic conditions (Fig. 1.6i–l).

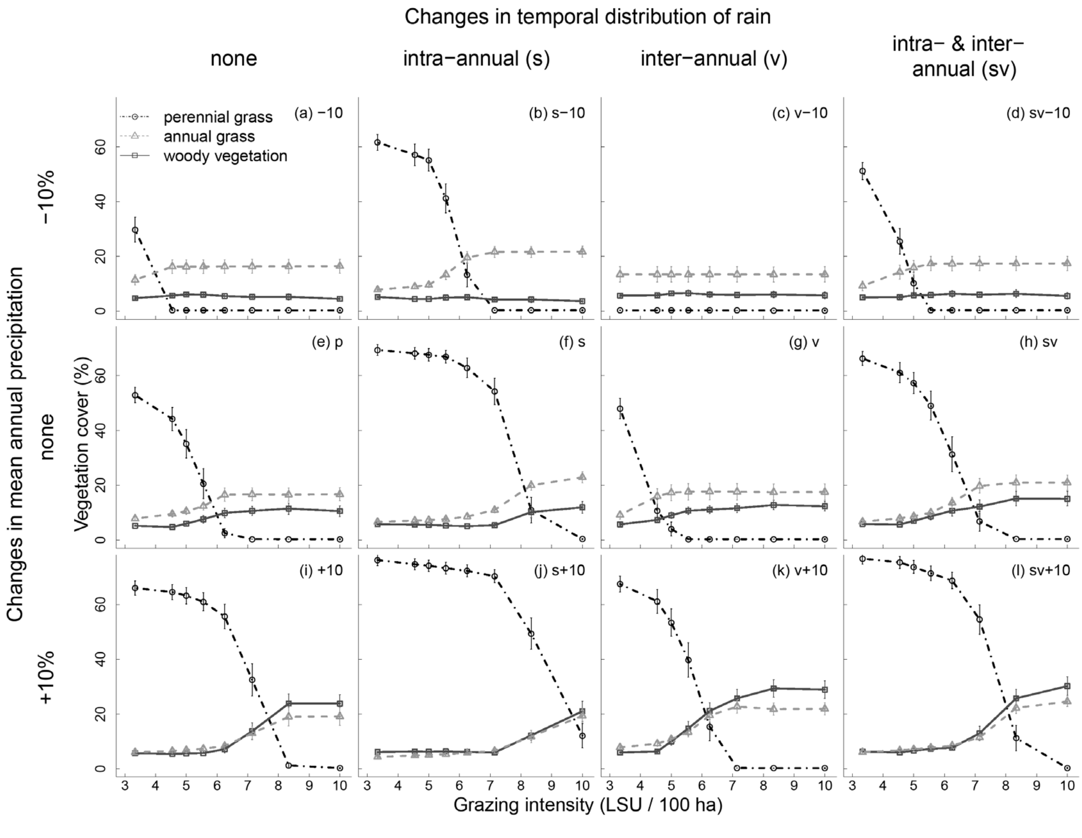

Figure 1.6: Mean and standard error of final cover of perennial and annual grasses and shrubs at different livestock densities. Results were derived from 25 repeated simulations of 200 years under 12 different climate change scenarios comprising all combinations of changes in mean annual precipitation with changes in temporal distribution of rain. All simulations include an increase of mean temperature and atmospheric CO_2 as given by the A1B scenario of the 2007 IPCC report.

1.4 Discussion

Following the state-and-transition concept (Westoby, Walker & Noy-Meir 1989) environmental drivers and land use can cause semi-arid savannas to switch between several possible vegetation states. These can broadly be categorized into a productive and desirable (e.g. in terms of livestock production or biodiversity) grass dominated state and several degraded states dominated by annual vegetation, woody vegetation or bare ground (Reynolds *et al.* 2007; Westoby, Walker & Noy-Meir 1989). In this study, we successfully refined and applied a process-based ecohydrological model (Tietjen *et al.* 2010) to assess the impact of specific patterns of climate change on the response of a semi-arid savanna to land use. We found that the capacity of the system to sustain long-term livestock grazing is strongly influenced by most of the simulated climate change scenarios. Furthermore, our findings suggest that shrub encroachment, currently one of the major problems in management of semi-arid rangelands, will be reduced under predicted future climatic conditions.

In the following, we discuss (i) the threshold of grazing intensity at which a transition between a non-degraded and a degraded state occurs, and (ii) how the pattern of degradation, i.e. the composition of vegetation types of the degraded system, may be influenced by climate change, and (iii) the implications for the management of semi-arid rangelands.

1.4.1 Shifting thresholds

Perennial grass biomass is one of the most important factors for livestock production in drylands. Various studies have shown that perennial grass dynamics are mainly driven by the highly variable water availability, while grazing increases the resulting fluctuations in grass abundance (Buitenwerf, Swemmer & Peel 2011; Fynn & O'Connor 2000; Weber & Jeltsch 2000): If water availability decreases due to changes in climatic conditions, growth decreases and drought induced mortality increases. This can even cause a collapse of the perennial grass matrix (Tietjen *et al.* 2010). If, in contrast, climate change leads to an increase in water availability (i.e. due to changes in intra-annual distribution of precipitation or increased MAP) the perennial grass abundance becomes more stable. In a nutshell, our study shows a shift in the threshold of grazing intensity, at which the system changes from a state dominated by perennial grasses to a state where plant types take over that are less suitable for livestock grazing or where vegetation cover is generally reduced.

Hereby, as also shown by our ecohydrological approach, changes in the intra-annual pulse size of precipitation events play a key role and have the potential to counteract the negative impacts of increased temperature and decreased MAP (Schwinning & Sala 2004). Dependent on soil texture and topography, an increased size of larger precipitation events at the cost of small events can cause a higher rate of water recharge of deeper soil layers (Reynolds *et al.* 2004; Schwinning & Sala 2004; Tietjen, Zehe & Jeltsch 2009) and consequently increased total water availability for plant growth. However, the shift of grazing thresholds that we found is to some extent site specific, since different soil types, soil crusts or slopes can cause limitations in the infiltration speed or changes in top-soil run-off (Tietjen, Zehe & Jeltsch 2009). Furthermore, this shift could potentially be mitigated if stocking rates are dynamically adapted (e.g. to available fodder) instead of keeping them constant over time.

1.4.2 Changing degradation pattern

Degradation of dryland savannas typically shows one of the following general patterns, mainly depending on the precipitation received: Either vegetation composition changes, leading to shrub encroachment (Graz 2008; Skarpe 1990; Wiegand, Ward & Saltz 2005) or vegetation cover in general is drastically reduced and the fraction of bare ground is increased with temporarily dominating annual grasses (Jeltsch *et al.* 1997).

Our results indicate that, if water availability decreases due to increased temperature or decreased MAP, woody vegetation will not be the dominant vegetation type in the degraded system state as under present climate conditions for this semi-arid savanna. The degradation pattern is instead driven towards a pattern typically found for more xeric savannas under current climatic conditions. But even for scenarios with increased MAP, shrub encroachment does not increase above the level that is found under current climatic conditions. This is particularly interesting, since several studies predict the opposite, i.e. an increase in shrub encroachment due to climate change despite temperature changes (Bond, Midgley & Woodward 2003; Kgope, Bond & Midgley 2010; Tietjen *et al.* 2010), which we did not find for any of our scenarios. Such conclusions are based on of the fact that woody plant species, which follow the C3-photosynthetic pathway, generally benefit more from elevated CO_2-levels than perennial grasses following the C4-pathway (Bond, Midgley & Woodward 2003; Morgan *et al.* 2004). However, these studies either neglect the establishment bottleneck of woody savanna species (Tietjen *et al.* 2010), which was found in a wide range of empirical and theoretical studies (e.g. Joubert, Rothauge & Smit 2008; Meyer, Wiegand & Ward 2009; Sankaran, Ratnam & Hanan 2004) or they refer to more mesic conditions (Bond, Midgley & Woodward 2003; Kgope, Bond & Midgley 2010). In such mesic savannas water availability is higher and consequently germination and early stages of establishment of woody vegetation are rather occurring continuously over time, thus being less important for degradation dynamics. In fact, fires play a central role in such systems by controlling woody vegetation (Bond, Midgley & Woodward 2003). Consequently, direct effects of CO_2, like the increased growth rates, could be beneficial by increasing post-fire re-growth (Kgope, Bond & Midgley 2010) as well as growth of saplings to the “fire-escape” zone (Bond, Midgley & Woodward 2003).

To our knowledge, only a few empirical studies have addressed the impacts of CO_2 in combination with temperature and precipitation changes on germination and establishment in general (Classen *et al.* 2010) or for relevant encroaching savanna species in particular (but see Polley *et al.* 2002). Supporting our finding that soil moisture plays a key role, Classen *et al.* (2010) found in their experimental study that tree germination and establishment are mostly determined by soil moisture when temperature and CO_2 were increased concurrently. None of the tree species studied showed an increased germination or establishment rate as a result of elevated CO_2 under dry and warm conditions.

1.4.3 Implications for rangeland management

This study shows that the effects of climate change on semi-arid rangelands are highly dependent on the specific scenario. On the one hand scenarios of increased size of precipitation pulses or increased MAP reveal a stable or even increased carrying capacity of the system. On the other hand, increased inter-annual variation of precipitation or a decrease in MAP reduces carrying capacities considerably. Despite this uncertainty, our study reveals that the risk of shrub encroachment will be reduced irrespective of changes in precipitation pattern due to increased mean temperatures. This has four important implications for rangeland managers and policymakers:

First, a possible loss of woody vegetation in degraded rangelands, especially when this loss is extreme, will pose a new problem to rangeland managers. In particular, at the end of the annual dry season when grass biomass in over-utilized areas is depleted, woody plant species contribute significantly to livestock diet (Tainton 1999, Katjiua & Ward 2007). Increased costs for supplemental feeding or the necessity to reduce livestock numbers would be the consequence.

Second, experiences of decision makers derived under present climatic conditions can be misleading for future management (see also Popp *et al.* 2009). If, under future conditions, rangeland managers considered the absence of encroaching bush as a sign of high carrying capacities or well adapted grazing intensity, they would be at risk of drastically overestimating the carrying capacities of their lands. Therefore, we suggest that sustainable rangeland management in the light of climate change should be determined by available grass biomass and not by woody vegetation cover.

Third, our findings have major implications for medium and long-term restoration of degraded savanna systems. Declined establishment of woody plants combined with natural senescence of adult plants will lead to a decrease of shrubs in the long run. Thus, elaborate and expensive bush control measures become redundant. The reduced competition from woody species in turn could increase the success of alternative restoration measures such as the (re-)introduction of desired grass species or other investments to improve rangeland quality from both, an ecological as well as an economic perspective.

Finally, savannas are increasingly recognized for their significant contribution to the global carbon cycle (Lehman 2010). An altered risk of degradation as well as changes in degradation pattern, which both change overall standing biomass, could have major implications in terms of carbon sequestration and the long-term feedback between climate and vegetation (Field *et al.* 2007). Altered carbon sequestration should be considered in the national and international planning of e.g. climate change mitigation and carbon storage projects.

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1.6 Appendix 1.A – parameters of vegetation model

Standard parameters of the vegetation model; for further details see Tietjen *et al.* 2010

Name	Description	Value	Unit	Source / parameterization
$W_{WP,pg}$	pg specific wilting point	0.036	vol%	based on Neilson 1995 & Sala *et al.* 1989
$W_{WP,w}$	w specific wilting point	0.036	vol%	based on Neilson 1995 & Sala *et al.* 1989
$W_{WP,ag}$	ag specific wilting point	0.036	vol%	based on Neilson 1995 & Sala *et al.* 1989
θ_{pg}	Potential uptake rate per grass cover	$4.275*10^3$	mm yr^{-1}	according to Tietjen *et al.* (2010) $\theta_{BM,pg} * conv\_c\_bm_{pg} * area_{cell}$
θ_w	Potential uptake rate per woody plant cover	$4.5*10^4$	mm yr^{-1}	according to Tietjen *et al.* (2010) $\theta_{BM,w} * conv\_c\_bm_w * area_{cell}$
$\theta_{BM,pg}$	Relative uptake rate per grass biomass	0.9	mm $(yr*g)^{-1}$	Tietjen *et al.* 2010;van Langevelde *et al.* 2003
$\theta_{BM,w}$	Relative uptake rate per woody biomass	0.5	mm $(yr*g)^{-1}$	Tietjen *et al.* 2010;van Langevelde *et al.* 2003
$conv\_c\_bm$	Perennial grass biomass at 100% cover	$1.9 * 10^6$	g ha^{-1}	Snyman 1998;Snyman & Fouche 1991
$conv\_c\_bm$	Woody biomass at 100% cover	$2.1 * 10^7$	g ha^{-1}	Vanvegten 1984
$conv\_c\_bm$	Annual grass biomass at 100% cover	$1.7 * 10^6$	g ha^{-1}	Snyman 1998
$root_{pg,L1}$	Fraction of pg roots in upper layer	0.63	-	Jackson *et al.* 1996;Tietjen *et al.* 2010
$root_{w,L1}$	Fraction of w roots in upper layer	0.36	-	Jackson *et al.* 1996;Tietjen *et al.* 2010
r_{pg}	Potential growth rate of perennial grasses	0.55	$mm^{-1}yr^{-1}$	Calibrated to gain long-term mean perennial grass cover of 30-70% in undisturbed (low grazing) scenarios according to local expert estimates
r_w	Potential growth rate of shrubs	0.3	$mm^{-1}yr^{-1}$	Calibrated to gain woody cover of max 42% according to Sankaran *et al.* (2005)
r_a	Potential growth rate of annuals	1.5	$mm^{-1}yr^{-1}$	Calibrated to gain long-term mean annual grass cover of 5-40% in degraded savanna according to expert estimates
bm_c_rain	Constant for linear increase of biomass per unit of cover depending on annual precipitation	0.35	-	Calibrated to gain slope of biomass – rain relation in Snyman & Fouche (1993) for savannah in good condition
lap	Extent to which shrubs and grasses can overlap	0.2	-	Tietjen *et al.* (2010)
mrd_{pg}	Mortality rate of perennial grasses due to water stress	0.54	$mm^{-1}yr^{-1}$	Calibrated to gain average mortality of 26% in drought years (<250mm) as given by O'Connor & Everson (1998)
mrd_w	Mortality rate of shrubs due to water stress	0.12	$mm^{-1}yr^{-1}$	Calibrated to gain average mortality of 5% in drought years (<250mm) as given by Meyer *et al.* (2007)

$cmax_{pg}$	Maximum cover for grasses	1.0	-	Expert estimate
$cmax_{w}$	Maximum cover for shrubs	0.8	-	Sankaran *et al.* (2005)
est_{pg}	Rate of successful establishment of grasses	0.05	-	Tietjen *et al.* (2010)
est_{s}	Rate of successful establishment of shrubs	0.005	-	Tietjen *et al.* (2010)
dca	Constant for exponential decline of spatial establishment with distance	0.5	-	Tietjen *et al.* (2010)
dcb	Constant for exponential decline of spatial establishment with distance	0.1	-	Tietjen *et al.* (2010)
$dist_{0}$	Constant for exponential decline of spatial establishment with distance	0.5	-	Tietjen *et al.* (2010)
cd_{min}	Fraction of maximum value of spatial establishment function at which function terminates	0.01	-	See supplementary material SA1 section "dispersal"
gb_{g}	Fraction of grass biomass that cannot be used by cattle	0.15	-	Tainton (1999)
gb_{w}	Fraction of woody biomass that cannot be used	0.5	-	Skarpe (1990)
m_{est}	Factor determining minimum mean soil moisture content needed for establishment relative to Wilting Point	1.21	-	Chosen so that establishment condition occur Joubert, Rothauge & Smit (2008)
$frac_{pg}$	preferred ratio of perennial grasses in cattle diet	0.65	-	Rothauge (2006)
$frac_{ag}$	preferred ratio of annual grasses in cattle diet	0.35	-	Rothauge (2006)
$frac_{w}$	maximum additional ratio of browse in cattle diet relative to grass uptake	0.075	-	Tainton (1999)
inishare	Constant of spatial heterogeneity of grazing	1.5	-	Estimated according to Jeltsch *et al.* (1996) & Weber & Jeltsch (2000)
ga	Constant shaping quadratic function of grazing damage	0.8	-	Expert estimate
gb	Constant shaping quadratic function of grazing damage	0.1	-	Expert estimate
growStart	First day of growing season	150	*d*	Tietjen *et al.* (2010)
growEnd	Last day of growing season	330	*d*	Tietjen *et al.* (2010)

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1.7 Appendix 1.B – hydrological model parameters

Standard parameters of the hydrological model; for further details see Tietjen, Zehe & Jeltsch 2009.

Parameter	Value	Unit	Source / comment
Water content at capillary pressure of 15 bar (-1500kPa)	4.1	Vol %	Rawls *et al.* 1992
Water content at capillary pressure of 0.33 bar (-33kPa)	16.7	Vol %	Rawls *et al.* 1992
Residual water content during dry season	3.5	Vol%	Rawls *et al.* 1992
Effective suction at wetting front	61.3	mm	Rawls *et al.* 1992
Saturated hydraulic conductivity	59.8	mm/h	Rawls *et al.* 1992
Water balancing constant between layers	0.05	-	Tietjen *et al.* 2010
Depth of upper layer	200	mm	Tietjen *et al.* 2010
Depth of lower layer	600	mm	Tietjen *et al.* 2010

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1.8 Appendix 1.C – model rules

Our model is based on the eco-hydrological dryland model *EcoHyD* to which we refer where possible in this description of model rules (Tietjen *et al.* 2010). It is a combination of a savannah vegetation model calculating the biweekly growth of two plant functional types (shrubs and perennial grasses) and an hydrological model, calculating daily moisture dynamics in two soil layers (Tietjen, Zehe & Jeltsch 2009). We do not describe the hydrological model since this was not changed and a full description can be found elsewhere (Tietjen *et al.* 2010;Tietjen, Zehe & Jeltsch 2009).

A list with description, value and source of all parameters is given in Appendices 1.A and 1.B for the vegetation and the hydrological model respectively. Parameter and variable names used are as used in the work of Tietjen *et al.* (2010).

Vegetation model

Changes in vegetation canopy cover (*cveg*) per cell of perennial grasses (*pg*), woody vegetation (*s*) and annual grasses (*ag*) depend on the processes growth (*grveg*), dispersal/establishment (*dsveg*), mortality (*mveg*) and losses due to herbivory ($\mathrm{h_{veg}}$) and (equation 1). The subscription veg refers to the three vegetation types (*veg: pg,ag,s*).

$$\frac{\mathrm{dc_{veg}}}{\mathrm{dt}} = \mathrm{gr_{veg}} + \mathrm{ds_{veg}} - \mathrm{m_{veg}} - \mathrm{h_{veg}}\ [\text{dimensionless}] \qquad \text{eqn 1}$$

In the following description of the model rules we refer to the original model description (*EcoHyD*) of Tietjen *et al.* (2010). Variable names are in-line with those found in Tietjen *et al.* (2010) for good comparability. Detailed descriptions are only given where changes have been made compared to the original model version. Exceptions are the functions for growth and mortality. They are described in detail, despite the fact that no changes have been made because this is crucial for a general understanding of the model structure and functioning.

Plant growth

Plant growth is implemented as increase in vegetation cover calculated in intervals of 14 days during a defined growing season. Growth of perennial grasses and shrubs is hereby based on a logistic function including the cover of the two perennial growth forms ($\mathrm{c_{pg}}, \mathrm{c_s}$) to represent competition for space, assuming a potential overlap of the both growth forms (lap). It furthermore depends on water availability in the two soil layers ($\mathrm{avW_{veg,Lx}}$), competition for water in each layer ($\mathrm{U_{veg,Lx}}$), a site-specific maximum cover for each plant type (cmaxveg) and a potential growth rate ($\mathrm{r_{veg}}$). Hereby Lx refers to the two soil layers (with x=1 for the upper soil layer, x=2 for the lower soil layer).

The growth function for perennial grasses and woody vegetation is (according to Tietjen *et al.* 2010):

$$gr_{veg,Lx} = min\left(U_{veg,Lx} * avW_{veg,Lx}, 1\right) * \, r_{veg} * c_{veg} * \left(1 - \frac{c_{veg}}{cmax_{veg} - \left(c_{-veg} * (1 - \, lap)\right)}\right) [yr^{-1}]$$

eqn 2

Water availability for a vegetation type in a soil compartment ($avW_{veg,Lx}$) is calculated as follows (see Tietjen *et al.* 2010):

$$avW_{veg,Lx} = \begin{cases} 0 & if\ W_{Lx} \leq \ W_{WP,veg} \\ \left(W_{Lx} - \ W_{WP,veg}\right) * depth_{Lx} & if\ W_{Lx} > \ W_{WP,veg} \end{cases} [mm] \qquad \text{eqn 3}$$

with W_{Lx} [vol %] the mean water content of the soil layer during the last 14 days, $W_{WP,veg}$ [vol %] the plant specific wilting point, and the depth of the soil layer $depth_{Lx}$ [mm]. This subroutine results in linearly rising water availability for increasing soil water contents as long as the water content is above $W_{WP,veg}$.

The fraction of available water that can be taken up in each soil compartment by each plant type $U_{veg,Lx}$ is calculated based on a potential water uptake rate per cover θ_{veg} [mm * yr-1], vegetation cover c_{veg} and the fraction of roots $root_{veg,Lx}$ in the respective layer (extending the approaches of Walker *et al.* 1981 and van Langenvelde *et al.* 2003).

$$U_{veg,Lx} \ = \frac{\theta_{veg} * root_{veg,Lx}}{\theta_{pg} * root_{pg,Lx} * c_{pg} + \theta_{w} * root_{s,Lx} * c_{s}} [dimensionless] \qquad \text{eqn 4}$$

Growth of annual plants in contrast is given by equation 5 and does not explicitly include any competition for water but exclusively depends on availability of empty space, growth rate (r_{ag}) and general water availability in the upper soil layer ($avW_{ag,L1}$), since annual plants are not assumed to invest resources in deep and dense root systems. Like in equation 4 we assume a potential overlap (lap) between grasses and shrubs.

$$gr_{ag} = min\left(1 * avW_{ag,L1}, 1\right) * r_{ag} * \left(1 - c_{pg} - c_{s} * (1 - lap) - c_{ag}\right)[yr^{-1}] \qquad \text{eqn 5}$$

Growth consequently decreases with increasing overall vegetation cover and is consequently potentially fastest at the beginning of the growing season.

In equation 4, annuals are not included since they are assumed to be the clearly inferior competitor for water, as perennial grasses and shrubs are already present with extensive root systems when the rainy season begins. However, the impact of annual plants on soil water through transpiration and evaporation is of course taken into account in the hydrological submodel. This is implemented in the same way that evapo-transpiration is considered for the perennial grasses except that we assume annual grasses to only root in the upper soil layer (for details of hydrological model see Tietjen et al 2009).

Plant mortality

Two types of mortality affect vegetation cover. First, drought induced mortality (md_{veg}) is calculated exactly as described in Tietjen *et al.* (2010). It is based on water availability and uptake analogous to growth (see equations 2-4) and depends on a drought mortality rate mr_{veg} , the average available water content in both soil layers during the growing season ($avW_{veg,Lx}$) and the proportional uptake of this water ($U_{veg,Lx}$).

$$md_{veg,Lx} = mr_{veg} * c_{veg} * \left[\left(1 - min\left(U_{veg,Lx} * avW_{veg,Lx}, 1\right)\right) * \frac{root_{veg,Lx}}{\sum_i root_{veg,Li}}\right] \; [yr^{-1}] \quad \text{eqn 6}$$

Secondly, we introduced stochastic age based mortality (ma_s) for woody vegetation, referring to empirical data on Acacia mellifera L. from a semi-arid savannah similar to the one found in the study area (Meyer, Wiegand & Ward 2009). This simulates a mortality that depends on the age of individuals ,since older individuals were e.g. found to be more sensitive to infestations by fungi or other diseases (Joubert, Rothauge & Smit 2008). This senescence is applied to all cells with cohorts of shrubs older than the average age of death of individual trees (ScenAge) (Meyer *et al.* 2007). The age of a cohort is determined by the date of the last establishment event that occurred in the respective cell. Hence, cells where the last establishment event of woody vegetation has been more than ScenAge ago are completely cleared from woody vegetation with an annual probability of mp_{age}.

Biomass production

Ground cover of the different vegetation types is the basic unit used in most equations of EcoHyD (important e.g. for infiltration, evaporation and surface water run-off). Since we are aiming at a quantitative representation of grazing, we need to translate cover values to biomass. This enables us to derive system productivity in terms of livestock carrying capacities from our analyses. In a semi-arid and thus water limited system, biomass production from a given vegetation cover depends on the amount of rain received (Snyman & Fouche 1993). More precisely, biomass production does not only depend on cover, but also on the height of e.g. grass shoots or the thickness of leaves etc. which in term is assumed to mainly depend on precipitation in a water limited system. Biomass (b_{veg}) is deduced from cover (c_{veg}) and the average biomass produced per unit of cover in years with average rainfall quantities ($conv\_c\_bm_{veg}$) depending on the following linear relation:

$$b_{veg} = c_{veg} * conv\_c\_bm_{veg} * cf(rain) \quad \text{eqn 7}$$

According to the abovementioned dependence of biomass production on precipitation, the slope of this relation is varied by the factor cf(rain) according to the actual year's precipitation (rain) and mean annual precipitation (MAP). The value of cf(rain) is 1 in case of an average seasonal rainfall, below 1 in case of lower and above 1 in case of higher rainfall amounts according to the below given linear relation (equation 8) with a constant ($\beta_r \leq 1$) determining the strength of the impact of rain on the cover-biomass relation.

$$cf(rain) = rain * \frac{1-\beta_r}{MAP} + \beta_r \quad \text{eqn 8}$$

Grazing and browsing

We applied an algorithm based on ideas underlying established grazing algorithms (used by e.g. (Weber & Jeltsch 2000). Basic underlying assumptions for grazing are:

1) grazing is heterogeneous in space, i.e. cattle tends to deplete preferred resources where it is if resources are available rather than moving further (Weber *et al.* 1998;Weber & Jeltsch 2000).
2) cattle feeds selectively, i.e. it favours perennial grasses over annual grasses. Shrubs are usually only eaten if no other alternative source of fodder is available (Rothauge 2006;Tainton 1999).
3) savannah vegetation is to some extent adopted to grazing (Tainton 1999).

Total annual biomass demand is calculated dependent on animal numbers per area, average animal's body weight and average daily biomass need of cattle– the latter being 2% of the livestock body mass per day (Tainton 1999). In case that this demand exceeds the available grass (annuals and perennials) biomass, we assume that additional fodder is given to the animals and herd size remains constant.

In the next step, the mean biomass need per cell ($\mathrm{bm_m}$) is derived by dividing total biomass need by the number of cells. Subsequently, individual cells are chosen randomly and from every cell biomass is removed. This biomass removal from random cells is repeated until the total biomass demand is covered. Thus, individual cells may be chosen several times.

Accounting for the spatial heterogeneity of grazing mentioned above, the biomass to be removed from a cell per grazing attempt ($\mathrm{bm_{rem}}$) is calculated as follows:

$$bm_{rem} = bm_m * \gamma_g \quad [kg] \quad with\, \gamma_g > 1 \qquad \text{eqn 9}$$

The heterogeneity factor (γ_g) determines the strength of spatial variation of grazing. We assume $\gamma_g$to be above one, since cattle is known to preferentially feed on sites where enough resource is available before moving to the next site.

Since grazing is considered to be selective, biomass is taken from perennial grasses, annual grasses and shrubs according to a defined ratio ($\mathrm{frac_{veg}}$):

$$bmr_{veg} = bm_{rem} * frac_{veg} \ [kg] \qquad \text{eqn 10}$$

However, if the finally determined demand of biomass of a plant life form in a given cell ($\mathrm{bmr_{veg}}$) is higher than the available biomass, the latter is totally removed. Hereby we assume a limited maximum fraction ($\mathrm{gb_{veg}}$) of the given biomass of a plant type ($\mathrm{BMO_{veg}}$) to be available, because cattle cannot graze the biomass completely down to the ground and not all parts of the different plants are edible (Skarpe 1990;Tainton 1999).

$$bma_{veg} = gb_{veg} * BMO_{veg} \quad [kg] \qquad \text{eqn 11}$$

If the biomass demand (bmr_{veg}) is close or equal to the available grass biomass (bma_{veg}), it is likely, that cells are chosen more than once by random selection. In such situations, when grass biomass is scarce, grazing becomes more homogeneous in space since animals make more efforts to find all available resources. If a cell is "grazed" a second time, the algorithm will still try to remove biomass according to the defined fractions ($frac_{veg}$), but it is likely that the preferred types' biomass is already depleted. Consequently, repeated selections of a cell lead to a shift of effectively realized fractions of biomass removal towards the less preferred types (annuals and shrubs) compared to the ratios given by $frac_{veg}$. In this way we conditionally simulate unselective and spatially homogeneous grazing. In summary this means, that the diet of the animals changes dynamically if grazing pressure increases or resource availability decreases (Rothauge 2006, Tainton 1999).

After determining the total amount of biomass that is removed from a cell (BMG_{veg}), the reduction of cover of the respective vegetation types due to herbivory (h_{veg}) is calculated.

$$h_{veg} = \frac{\propto_{graze} * BMG_{veg}}{conv\_c\_bm_{pgrass} * cf(rain)} \quad [dimensionless] \qquad \text{eqn 12}$$

Reduction of biomass does not directly – i.e. according to the cover-biomass relationship given in equation 7 – translate back into a reduction of cover, accounting for adaptation of grasses to grazing. Therefore we include a relative damage caused to plant cover ($\propto_{graze}$), which increases linearly with an increasing fraction of biomass that was removed by the grazers ($\frac{BMG_{veg}}{BMO_{veg}}$) as given in the following relation:

$$\propto_{graze} = \left(\frac{BMG_{veg}}{BMO_{veg}} * ga + gb\right) \quad [dimensionless] \text{ with } ga + gb \leq 1 \qquad \text{eqn 13}$$

The reduction in cover due to grazing is consequently calculated by incorporating $\propto_{graze}$ in the abovementioned relation between biomass, cover and precipitation (equation 7). The parameters ga and gb define the shape of this quadratic function of cover reduction (equation 11 in equation 12 results in a quadratic function of BM_{rem}).

Dispersal and seedling establishment

Dispersal and establishment are simulated as addition to the cover of a respective growth form (ds_{veg}) to certain cells in the grid. This is rather representing seedling dispersal than seed dispersal. Germination and seedling/juvenile survival are therefore rather implicitly included (Tietjen *et al.* 2010).

Dispersal and establishment of perennial grasses is implemented as described by Tietjen *et al.* (2010). We assume no dispersal limitation on the given spatial scales (Jeltsch *et al.* 1997), i.e. spatially homogeneous distribution of grass cover with the amount of cover depending on the mean perennial grass cover of the whole grid. Annuals are assumed to be always present as seeds and start off at every season without initial cover (i.e. no dispersal and establishment calculation necessary, solely growth function determines occurrence). Woody plants are, in accordance with literature on regional typical shrub and tree species (i.e. Acacia species), assumed to be limited in dispersal, seed production and especially in establishment (Barnes 2001;Joubert, Rothauge & Smit 2008;Meyer *et al.* 2007;Tews, Schurr & Jeltsch 2004).

However, the establishment of shrubs is simulated in more detail compared to the model version of Tietjen *et al.* (2010). Dominant encroacher species in semi-arid African savannahs are known to have relatively high requirements regarding water availability for seed production, seedling germination and successful establishment. Different studies showed, that at least 2 subsequent years of above average rainfall are needed for successful establishment of A. mellifera (Barnes 2001;Joubert, Rothauge & Smit 2008;Meyer *et al.* 2007) and other woody plant species of semi-arid savannahs (Wilson & Witkowski 1998). Hence, successful establishment of woody vegetation is only possible if the mean soil-water content in the upper soil layer during the growing season is well above the wilting point of plants during two subsequent years $(\mathrm{W_{L1,mean}} > \mathrm{m_{est}} * \mathrm{WP_s})$. The factor $\mathrm{m_{est}}$ was calibrated so that establishment conditions at one location occur on average 5-6 times per century (Joubert, Rothauge & Smit 2008). To account for positive impacts of grazing on woody plants' dispersal and establishment (Hiernaux *et al.* 2009;Kraaij & Ward 2006), we add a grazing dependent factor to the function of Tietjen *et al.* (2010), so that amount and spatial extent of shrub establishment increases with increasing grazing pressure (Ward & Esler 2011). This is achieved by linearly altering the parameters that determine the exponential decrease of "seedlings" (i.e. cover) with distance ($\mathrm{distConst(SR)}$) and the maximum dispersal distance ($\mathrm{distmax_s(SR)}$).

The dispersal and establishment of shrub seedlings added as cover to a target cell (ds) is calculated for every source cell in the grid if the target cell had a sufficient water availability during the last and current growing season and its position was within the maximum dispersal distance according to the following term:

$$ds = c_{s\_source} * est_s * dist_0 * e^{-distConst(SR)*dist} * max(1 - c_s - c_{pg}; 0) \ \ [dimensionless]$$

eqn 14

Establishment and dispersal consequently depend on shrub cover in the source cell ($\mathrm{c_{s\_source}}$), mean rate of seedling establishment ($\mathrm{est_s}$), cover of grasses and shrubs in the target cell ($\mathrm{c_s, c_{pg}}$) and the shape of an exponential dispersal decline (dependent on $\mathrm{dist_0}$ and $\mathrm{distConst}$) as well as on the distance of the target from the source cell (dist). Grazing impact on the dispersal kernel is given by the following linear relation being a function of the stocking rate (SR):

$$distConst(SR) = dc_a + (dc_b * SR) \ \ [dimensionless]$$

eqn 15

The maximum dispersal distance $\mathrm{distmax_s(SR)}$ is calculated so that the added cover ds is at least 1% of the maximum possible value of ds at the centre of the source cell (dist = 0).

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1.9 Appendix 1.D – model validation & sensitivity analysis

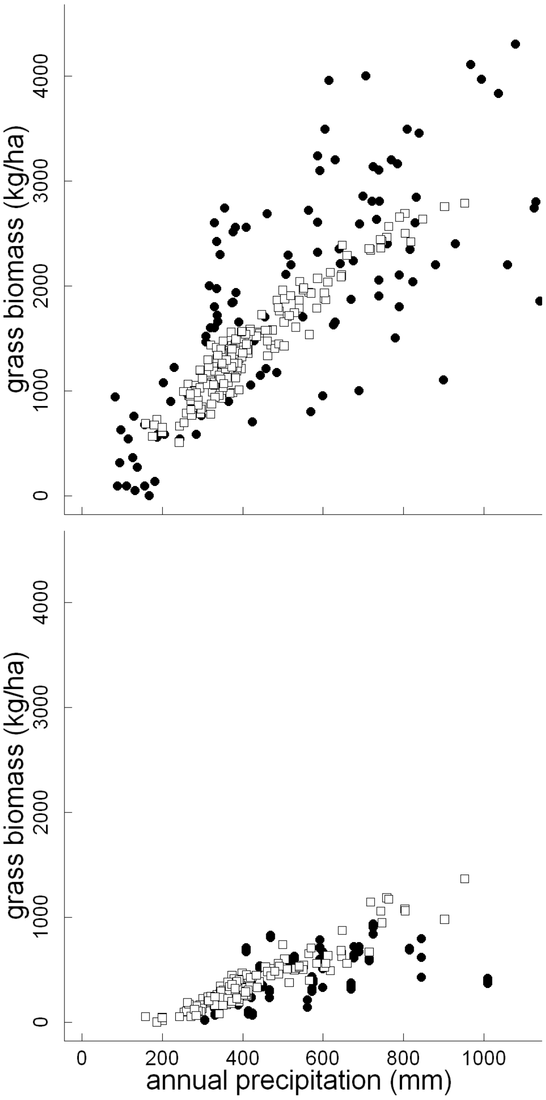

Figure 1.D.1: Biomass production of grasses (perennial and annual grasses) in dependence of annual precipitation from simulated (open squares) and literature (filled circles) data of semi-arid African savannas. Upper figure shows data for savannah in "good" condition, i.e. where no shrub encroachment has taken place and perennial grasses are dominating the vegetation (data from Higgins, Bond & Trollope 2000 and references therein ;Ward & Ngairorue 2000), lower figure shows data of savannas in a "bad" condition, i.e. perennial grasses are drastically reduced or gone and shrubs and annual grasses dominate the system (data from Snyman 1998).

Model validation

For model validation, we compared simulation results with three empirical patterns. First, we tested, whether total grass biomass production, depending on annual precipitation derived from model simulations, is comparable to empirical evidence found in the literature (Higgins, Bond & Trollope 2000 and references therein ;Ward & Ngairorue 2000, Snyman 1998). Hereby we distinguish between data from sources that were labelled in the respective studies as being in "good" or "poor" condition (or as being heavily grazed or only subject to low/moderate grazing intensities respectively). We compared biomass data of "good" condition savannas with simulated biomass gained under low stocking rate scenarios (5 LSU/100ha) i.e. where no degradation occurred. Biomass values for rangelands in "bad" condition were compared to the results of simulations with 8.3 LSU/100ha after a burn-in of 100 years, so that the vegetation was already degraded (i.e. shrub encroached). As shown in figure 1.D.1 our model was very well able to reproduce this important pattern, for both, the non-degraded as well as the degraded system. Simulated biomass production is representing values from empirical data very well, though simulated data shows less variation since several sources of variation like soil heterogeneity are not represented in our simulations.

Second, we used satellite images of the end of the rainy season of 2008 (pan sharpened IKONOS images covering the area of Sandveld, Namibia, between E 19° 3' and E 19° 12'; and S 21° 57' and S 22° 5' in 1 m² resolution from March 26th and 29th, 2008), to identify shrub cover and compared these to simulations with land use and climate scenarios according to the data of the last 23 years given by the Sandveld research station. Therefore we performed simulations with constant stocking rates according to the full range of livestock densities that was applied in Sandveld (2.5 – 8.3 LSU /100ha). The simulations showed that the shrub cover produced by our simulations is in a very realistic range (figure 1.D.2). The reproduction of this pattern is very difficult, since both real and simulated systems are in a transitional, i.e. highly dynamic state. On the one hand juvenile shrubs might not be detected by remote sensing measures and on the other hand climate and land use history beyond the last 23 years are not known exactly although they clearly have an impact on the current state of the system. However, the encroachment pattern found in our results seems to be in a very realistic order of magnitude.

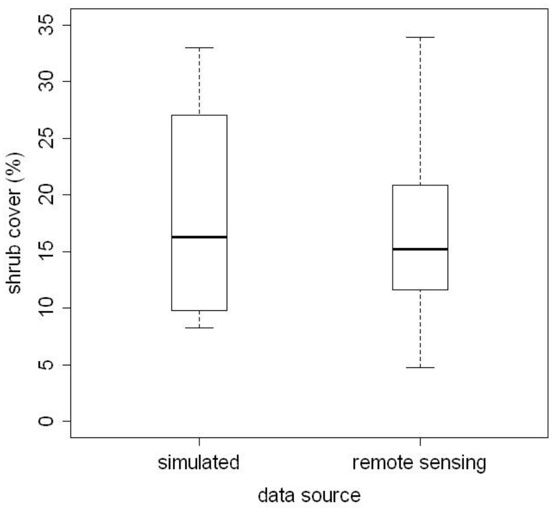

Figure 1.D.2: Shrub cover derived from simulations and satellite data – simulations were performed for a range of stocking rates given by the rates applied in Sandveld during the last 23 years (2.5 – 8.3 LSU /100ha); Climate data was according to annual precipitation data of the last 23 years given by the Sandveld research station

Third, we compared simulated biomass production of grasses with the production measured in Sandveld during four years between 1985 and 1992 (Rothauge 2006). Since the empirical data was assigned to different grazing treatments and derived from years with very different precipitation (and the history of both) we only compared the range given by empirical data and simulated data. We found a grass biomass production ranging from 400 to 1100 kg/ha in our simulations for the given set of precipitation in the sampling years. The empirical data from Sandveld revealed a range from 200 to 1500 kg/ha. Hence, we are also able to cover the biomass production of this specific site quite well.

Sensitivity analyses

To test for sensitivity of model parameters, we varied all vegetation parameters (Appendix A) systematically by ±10% and ±20% according to the sensitivity analyses of Tietjen *et al.* (2010). Exceptions were made for three parameters where such a variation seemed too extreme or results in unrealistic/impossible values (see Tietjen *et al.* 2010). These were (1) first day of growing season) and (2) last day of growing season both varied by ±10 days and ±20 days and (3) the wilting point of all vegetation types: here we varied the distance of the value to the value of the residual water content by ±10% and ±20% instead of varying the parameter value itself. Afterwards we evaluated the effect of these changes on the vegetation cover of perennial grasses and woody vegetation after 200 years for standard simulations with today's climatic conditions and high grazing (8.3 LSU/100 ha) and non-grazed scenarios (1.3 LSU/100 ha), respectively. We do not discuss results for annual grasses here since sensitivity was very low and always directly linked to the response of the other functional types. Annual grasses are inferior competitors and directly linked to the occurrence of the other vegetation types (see model rules in appendix C).

In scenarios of low grazing intensity only shrub cover is sensitive towards variations of the parameter determining the threshold of soil moisture that allows for shrub establishment (m_{est}). This is known to be one of the most critical parameters in the life-history of shrubs of this region and thus, this finding is not surprising (see discussion in main text). However, besides this the dynamics of the undisturbed system are very stable and robust against variations in single parameters.

In scenarios with high grazing pressure the system becomes more sensitive since highly dynamic degradation processes are initiated. Consequently both, grass and shrub cover become sensitive to variations in some parameters.

Perennial grass cover becomes slightly sensitive towards the parameter $W_{WP,ag}$ determining the wilting point of annual grasses. If this is decreased, annuals produce more biomass, which reduces grazing pressure on perennial grasses a little. However, this only causes minor changes to perennial grasses, sustaining about 5% of cover after 200 years instead of being eradicated completely. Both grasses and shrubs are affected slightly by a 20% reduction of the parameter ga that is one of two parameters shaping the quadratic function for grazing damage (see appendix A an C). Such a reduction causes reduced grazing damage to perennials and thus the degradation process is slower compared to standard simulations. However, after 200 years perennial grasses can only sustain 9% instead of 0% of mean cover (average over 25 simulations with individual stochastic precipitation) and shrubs have 25% instead of 30% average canopy cover.

Shrub cover, like in the “no grazing” scenario, responded strongly to variations in the parameter determining the threshold of soil moisture that allows for shrub establishment (m_{est}). Furthermore, variations in the parameters determining shrub dispersal (*dca, dcb,*cd_{min}) caused minor changes in final shrub cover. However, the respective simulations were always still resulting in shrub encroachment with between 25 and 35% of mean canopy cover of shrubs and had no influence on perennial grass cover after 200 years. Finally the growth rate of shrubs when varied by plus or minus 20% was leading to changes in shrub cover after 200 years of 16% and 23% respectively. Considering a change of the growth rate by 20% this response is fairly proportionate and does not question the general outcome of the simulations (i.e. shrub encroachment takes place and perennial grasses decline completely).

These results show that there is no immoderate sensitivity of the model towards any of the parameters. However, establishment, also comprising dispersal of shrubs is clearly a key process of the system dynamics. Empirical knowledge of these parameters is low, so that a certain level of uncertainty regarding simulation results remains. Nevertheless literature suggest, that this bottleneck is essential for system dynamics and we are thus confident (Joubert, Rothauge & Smit 2008;Meyer, Wiegand & Ward 2009;Sankaran, Ratnam & Hanan 2004) that this sensitivity is no drawback of the model but an important trait that needs to be considered.

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Link to the following chapter

In chapter 1 I described an extended version of the eco-hydrological dryland model *EcoHyD* which I parameterized and validated for a semi-arid savanna in Namibia. My simulations showed that carrying capacities of the system might be significantly reduced under climate change, though remarkably, not for all of the tested scenarios. Further, I found that the dynamics of shrub encroachment might differ significantly from the current pattern under future climatic conditions.

At this, my model predictions differ in important aspects from those of other studies and earlier versions of *EcoHyD*. This difference is mainly based on the specific inclusion of the severe bottleneck at the seedling stage (that especially encroaching woody species in semi-arid savannas feature) into the present version of the model.

Hence, the results of chapter 1 underline two key aspects of semi-arid savanna rangelands: First, the results highlight the necessity of precautious sustainable management of these rangelands, especially in face of the uncertainty and potential threats exhibited by future climatic conditions. Second, the study presented in chapter 1 points out the high relevance of the recruitment bottleneck of woody plants for the overall dynamics of the semi-arid savanna vegetation.

Consequently, a process that would further narrow the bottleneck of encroaching tree and shrub species should have high potential to provide measures for sustainable management of semi-arid savannas. In this context, recent empirical findings that indicate high post-fire mortalities of woody plant seedlings provide an interesting opportunity for rangeland management.

Therefore, in chapter 2, I use the abovementioned simulation model to assess the effects of different fire management strategies with regard to their effect on degradation and long-term carrying capacity of the system.

Chapter 2

Prescribed fires as a tool for sustainable management of semi-arid savanna rangelands.[1]

Summary

1. Savanna rangelands are threatened by degradation in the form of woody plant encroachment, causing a strong decline in the productivity of domestic livestock production. Although recent studies indicate that fire might be of great importance for semi-arid and arid savanna dynamics, its potential use for the management of semi-arid rangelands has been largely neglected.

2. We used the eco-hydrological savanna model *EcoHyD* to simulate the long-term effects of different fire management strategies for semi-arid savannas. All strategies aim at the early seedling recruitment of encroaching woody plant species, since adult trees or saplings are not sufficiently damaged by the low intensity fires. We simulated different modes of fire management for a broad range of grazing intensities to analyse the long-term suitability of these strategies for semi-arid rangeland management.

3. Prescribed fires, timed to kill young tree seedlings were found to be very efficient in preventing shrub encroachment for a broad range of livestock densities. However, when grazing intensity was too high, fire management failed in preventing woody plant increase.

4. The application of this fire management strategy increased the maximum long-term carrying capacity of the semi-arid rangeland by more than 30%. In addition, we found that that the risk of degradation by shrub encroachment into the savanna vegetation was lower with the application of fires.

5. *Synthesis and Applications* Although opportunistic management, i.e. the tracking of available forage with livestock numbers in combination with fire suppression seems appealing in the short run, long-term strategies with an active use of fires have great advantages for semi-arid rangeland management. Hereby, not frequency but timing of fires relative to episodic events of shrub seedling recruitment should be the basis of fire management strategies. The application of fire on fractions of a farm at rare intervals alone, or in combination with other measures of shrub control, offers a practicable solution to the encroachment problem in semi-arid rangelands.

[1] Submitted as Lohmann, D., Tietjen, B., Blaum, N., Joubert, D.F. & Jeltsch, F. "Prescribed fires as a tool for sustainable management of semi-arid savanna rangelands." to *Journal of Applied Ecology*.

2.1 Introduction

For decades the degradation of rangelands has been observed in savannas worldwide. Vegetation composition has changed drastically from a state dominated by perennial grasses to a state in which a few woody plant species encroach and dominate the plant community (Buitenwerf *et al.* 2012;Fensham, Fairfax & Archer 2005;Wigley, Bond & Hoffman 2010). This so-called shrub or bush encroachment is often irreversible for several decades, and reduces the provision of fodder biomass for livestock production as well as other ecosystem services like water retention and protection from soil erosion (Gillson & Hoffman 2007;Graz 2008). Careful management is required to avoid degradation of savannas. However, the prevailing conditions of high climatic variability, non-linear dynamics of degradation, and increasing pressure due to climatic changes and population growth pose a difficult task to local farmers and land managers (Gillson & Hoffman 2007;Lohmann *et al.* 2012;UNCCD 1994) and a universal explanation for the degradation of savannas has not been identified yet.

The interplay of several factors is generally assumed to cause savanna degradation, including livestock (over-) grazing, the suppression of the natural fire regime and an increasing level of atmospheric CO_2 (Tietjen & Jeltsch 2007;Wigley, Bond & Hoffman 2010). These factors can cause a shift in the competitive balance between the two major plant functional types, i.e. perennial grasses and woody vegetation (Graz 2008). However, literature disagrees about the question to what extent the different factors influence this competitive balance.

Clearly, grazing by domestic livestock is the process that is most often stressed to be responsible for shrub encroachment (Watkinson & Ormerod 2001). The selective removal of grass biomass by cattle has the potential to release trees and shrubs from competition with grasses. Several studies found that especially seedlings of woody plants suffer from strong competition with perennial grasses and that grasses dominate competition for water in the topsoil (Kulmatiski *et al.* 2010;Ward & Esler 2011). Further, Kambatuku, Cramer & Ward (2012) showed that water absorption by grasses in the topsoil decreased deep infiltration: if grasses were clipped, more water was available in deeper layers of the soil and thus for woody species with a deep rooting system. Consequently, the establishment success and growth of encroaching shrubs is greater when grasses are prone to intense livestock grazing (Kambatuku, Cramer & Ward 2011;Ward & Esler 2011). However, grazing alone does not seem to be sufficient to explain shrub encroachment (e.g. Browning & Archer 2011;Wigley, Bond & Hoffman 2010).

Several studies suggest that direct and indirect human interventions in natural fire regimes are essential drivers of the degradation of mesic savanna rangelands (Browning & Archer 2011;Rohde & Hoffman 2012). If the grass fuel biomass allows for frequent and hot fires, these are capable of controlling bush densities by destroying the aboveground biomass (topkill) of trees (Bond & Keeley 2005;Sankaran, Ratnam & Hanan 2008). Further, fires play a key role in shrub demography of mesic savannas, since they potentially prevent woody plant saplings from growing to maturity and consequently from producing seeds (see Gulliver syndrome in Higgins *et al.* 2007a;Midgley, Lawes & Chamaille-Jammes 2010). In contrast, the generally low fuel load in semi-arid and arid savannas does not allow for hot fires, therefore adult shrubs or trees and saplings are normally not damaged effectively by fire events (Higgins, Bond & Trollope 2000;Meyer *et al.* 2005). Therefore, fires are often assumed to be of little importance for arid and semi-arid savannas, although occurring frequently under natural conditions (e.g. Higgins, Bond & Trollope 2000;Sankaran, Ratnam & Hanan 2008). However, evidence has been found that also in these drier regions fires can severely impact the early recruitment of tree species (Casillo, Kunst & Semmartin 2012;Harrington 1991;Joubert, Smit & Hoffman 2012;Taylor *et al.* 2012). In a recent review on the effects of fire and herbivory on savanna trees Midgley, Lawes & Chamaille-Jammes (2010) suggest that young seedlings, which have a thin bark and only few carbon storage, are likely to face high post-fire mortalities. This was confirmed by Joubert, Smit & Hoffman (2012) and Casillo, Kunst & Semmartin (2012), who both tested the in-situ effect of fires on woody seedling recruitment in semi-arid savannas and found that tree recruitment is strongly inhibited by fires. Particularly seedlings of *Acacia mellifera* BENTH., which is one of the most important encroaching shrub species in semi-arid southern Africa, experienced post-fire mortalities of 97 to 99% after moderately hot fires (Joubert, Smit & Hoffman 2012).

Encroaching woody species have been found to be strongly limited in recruitment by the highly variable water availability in semi-arid and arid savannas (Joubert, Rothauge & Smit 2008;Kraaij & Ward 2006), leading to mass recruitment events after series of years with considerably above-average precipitation. Therefore, a fire event immediately after a year of mass germination might further narrow this recruitment bottleneck. Following this idea, fires should be of fundamental importance for the overall establishment success of woody vegetation in semi-arid savannas and could thus provide a valuable tool for rangeland management, which has been largely unused so far (Harrington 1991;Joubert, Rothauge & Smit 2008;Midgley, Lawes & Chamaille-Jammes 2010).

Yet, fire is seen critically by many land users, since it is potentially fraught with risks for infrastructure and livestock and, not least because scientific evidence regarding the role of fire on semi-arid savannas is controversial. Further, fire removes the desired biomass of grasses, which is the fodder for livestock production. Especially in semi-arid and arid regions, fires are thus often extinguished by land users or are prevented due to high levels of grazing and the respective lack of grass fuel biomass (Joubert, Smit & Hoffman 2012;Scholes 2009). For many regions fire suppression was linked to colonial expansion and the advent of European settler farmers, who mostly perceived fire as a problem (Rohde & Hoffman 2012).

In this study, we assess the long-term effect of a fire-based management strategy for semi-arid and arid savanna rangelands that explicitly targets the demography of woody species encroachment, i.e. infrequent years of high seedling emergence. Under these conditions, which are characterized by prolonged (multi-season) periods of above average soil-water availability, semi-arid and arid savannas will also carry relatively high amounts of grass biomass and thus high fuel loads. This in turn favours intense fire event, which is capable of suppressing a mass recruitment of woody species (Harrington 1991).

Based on the experimental evidence found by Joubert, Smit & Hoffman (2012), we assess the long-term influence of systematic prescribed fires on the carrying capacity and degradation of a semi-arid savanna. For this, we use the savanna vegetation model *EcoHyD* (Lohmann *et al.* 2012;Tietjen *et al.* 2010), which explicitly addresses specific demographic features of the typical encroacher plant *A. mellifera* and its competition with grasses. The eco-hydrological approach of the model assures that the important dynamics of soil-water availability are represented in sufficient detail and thus enable a realistic simulation of the water-triggered demographic bottleneck of the woody plant type. On this basis, we simulate the application of prescribed fires after woody species germination in combination with a broad range of grazing intensities. In particular we assess how such a controlled fire regime interacts with different grazing intensities especially with regard to (1) its efficiency in controlling woody plant encroachment and (2) its capability of increasing semi-arid savanna capacity for livestock production.

2.2 Materials and Methods

2.2.1 Study area

The model was parameterized to conditions of a typical semi-arid Namibian Acacia-tree-and-shrub savanna of the Central Kalahari type (Mendelsohn *et al.* 2002) as found at the governmental research station at Sandveld (latitude 22°02'S longitude 19°07'E). The study area, which has been a research farm since the late 1960s, was used for livestock production for about 80 years. The vegetation is a typical shrub-encroached savanna that is invaded by *Acacia mellifera* BENTH., though the level of degradation is still considered to be moderate.

Precipitation mainly occurs during the summer months (September to April) with a high inter- and intra-annual variation. Mean annual precipitation (MAP) is 408 mm (1986-2008) with a standard deviation of 180 mm (Rothauge 2006). The mean annual temperature is about 19 °C, with monthly means ranging from 12 °C (July) to 25 °C (January). The area is characterized by loamy Kalahari sand soils. The topography of the area is very flat with a mean altitude of 1520 m.

2.2.2 Model description

We used the grid-based, eco-hydrological savanna vegetation model *EcoHyD* (Tietjen *et al.* 2010), which has been successfully applied to questions on the dynamics of semi-arid savannas (Lohmann *et al.* 2012;Tietjen *et al.* 2010). It comprises two sub-models, a process-based savanna vegetation model calculating the growth of three plant functional types (shrubs, perennial and annual grasses) in biweekly steps (extended by Lohmann *et* *al.* 2012) and a process-based hydrological model that calculates the daily moisture dynamics of two soil layers (Tietjen, Zehe & Jeltsch 2009) for 30x30 grid cells each representing 5x5 m² patches resulting in a total simulated area of 2.25 ha. The model was successfully tested against pattern of biomass production in relation to precipitation under different levels of degradation (Lohmann *et al.* 2012). No changes have been made to the hydrological sub-model, but the vegetation model was modified to be able to explicitly simulate fire management.

The vegetation sub-model simulates the processes growth, mortality (induced by drought or senescence), competition for water and space, grazing, browsing and the dispersal and establishment of shrubs, perennial grasses and annuals. Besides a new fire algorithm we included moribund grass biomass into the model since it plays a role for fire intensity calculation. The woody plant functional type is parameterised to resemble *A. mellifera*, which is one of the most dominant encroaching species in southern African savannas (Joubert, Rothauge & Smit 2008) and for which sufficient data was available from other studies.

Since the model has been published elsewhere (see above), a comprehensive description of the model rules and equations can be found in Appendix 2.B. The newly included fire simulation is described below. All simulations were conducted using the parameter set given in Appendix 2.A. Parameters of the hydrological model are identical to the ones given in chapter 1 of this thesis (Appendix 1.B).

2.2.3 Fire

The basic underlying assumption for the fire implementation is that if fire is applied, it is ignited at the end of the dry season and disperses to all cells of the simulated 2.25 ha grid. For each grid cell, fire damage to each vegetation type is calculated separately. We simulated the effect of fire on woody seedlings of the age of one and two years based on the findings of fire experiments by Joubert, Smit & Hoffman (2012). If local fire intensity exceeds a threshold of 300 kJ $s^{-1} \cdot m^{-1}$, shrub seedlings in the specific grid cell will die with a probability of 97% (in the standard scenarios) To estimate fire intensity we use a regression of Trollope, Trollope & Hartnett (2002) for fires in African savannas:

$$I = 2729 + 0.8684 \cdot FL - (530 \cdot \sqrt{FM}) - (0.1907 \cdot RH^2) - (596/WS) \qquad \text{eqn. 1}$$

where FL is fuel load ($kg \cdot ha^{-1}$), FM is fuel moisture (%), RH is relative humidity (%), and WS wind speed ($m \cdot s^{-1}$).

If a fire is identified to be hot enough to spread (I> 300 kJ $s^{-1} \cdot m^{-1}$), it is assumed to additionally cause topkill leading to a loss of shrub cover (LS) (following van Langevelde *et al.* 2003):

$$LS = \min((sLS \cdot FL \cdot cF \cdot sC); mLS) \qquad \text{eqn. 2}$$

Where sLS is the specific loss of shrub cover per unit of energy (W^{-1}), FL is the fuel load, cF is a coefficient determining the increase in fire intensity with fuel load ($W \cdot m^{-2} \cdot g^{-1}$), sC is the cover of shrubs and mLS is the maximum loss of shrub cover due to fire.

Perennial grasses suffer a mortality of 10% of the current cover irrespective of the intensity of fire (van Langevelde *et al.* 2003). All grass biomass, moribund and alive is removed from the grid. Grasses can re-grow during the growing season which follows the burning. Fire does not have any impact on annual grasses as burning takes place before germination.

2.2.4 Scenarios

All results presented here are based on 25 repeated simulations of 200 years, each with a unique time series of stochastic precipitation and temperature with a resolution of one hour, generated as described in Tietjen *et al.* (2010). Mean value, distribution and variance of the derived time series do not differ significantly from measured annual precipitation in Sandveld during 1986–2008.

Simulations are initialized with shrub canopy cover that was randomly distributed in 20% of all cells with values drawn from a uniform distribution between 1 and 80% per cell. In cells with woody vegetation, initial grass cover was set to 5%. Then, perennial grass cover of the remaining cells is randomly drawn from a uniform distribution with values between 40% and 80%. Annual grass seeds are assumed to be omnipresent.

We simulate three scenarios of fire management that are combined with several scenarios of grazing intensity.

Land use

The scenarios of livestock density range from 2.5 to 10 large stock units (LSU) per 100 ha^{-1}, with constant density during one simulation. We use the definition of a LSU as a 450 kg live weight cattle. Natural grazing and browsing of game are not simulated as an exclusive use of the rangeland by domestic cattle is assumed.

Fire management

Fire is considered as a management measure in this study and the extinction of all naturally occurring fires is assumed. We simulate three kinds of fire application: In the baseline fire scenario fire is applied when tree seedlings emerge in more than one third of the grid cells, which only occurs after at least two consecutive years with above average rainfall (Fig. 2.1a). No change in applied stocking rates is assumed. Since this is not realistic on a farm scale, because grasses needed resting for at least some weeks after a fire event to produce new biomass, we simulate a second scenario of fire management. This strategy is based on the finding of Joubert, Smit & Hoffman (2012) who found, that burning in the first and second year after germination caused equal seedling mortalities. For reasons of simplicity we only present simulations of one of the two halves of a farm here. We assume that the livestock density on the burnt area is reduced by 33% in the

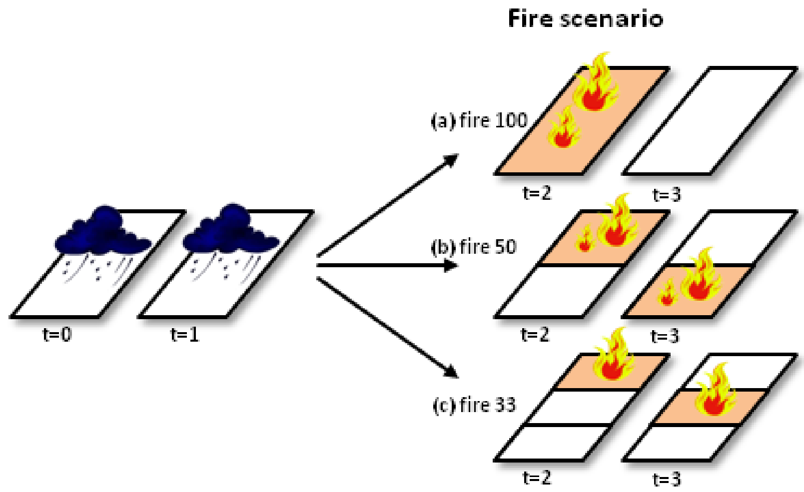

Figure 2.1: Schematic representation of fire application. After at least two consecutive years of good rainfall (t=0 & t=1) seedlings emerge. Fire can be effectively applied at the beginning of the rainy season of the two following years (t=2 & t=3). Scenario (a) fire 100: fire is applied in the year after seedling emergence on the whole farm. Scenario (b) fire 50: fire is applied on half of the farm in year 2 and 3 respectively. Scenario (c) fire 33: fire is only applied on two thirds of the farm. At the next occasion of shrub seedling emergence the now unburned plot will be burned and one of the other two will remain unburned.

year of fire application, and increased by 33% in the subsequent year (Fig. 2.1b). On a farm scale, this would mean, that while half of the area is burnt and rested for a certain time after burning, the other half has to carry more animals. In the following year, the other half of the farm is then burnt and rested, while the cattle density is increased on the formerly burnt site. Third, we further reduced the fraction of the farm that needs to be burnt during one season from 50% to 33%. Whenever mass seedling emergence occurs, two of the three parts of the farm are burnt (Fig. 2.1c), the first part in the first year and the second part in the second year, following the idea of the second fire scenario. The remaining third of the farm is not burnt but will be burnt at the next occasion of seedling emergence.

In order to also assess how sensitive these results are to the fire-induced mortality of seedlings, we additionally simulate the baseline fire scenario with fire-induced seedling mortality rates ranging from 50% to 97%, i.e. mortality rates that are lower or equal to the one found by Joubert, Smit & Hoffman (2012)

.

2.3 Results

In the following we show how different stocking rates and fire applications influence the cover of perennial grasses and woody vegetation and the long term carrying capacity of the savanna. We use the term "target stocking rate" to describe constant density of animals that the farmer tries to keep on the land. In cases of fodder scarcity, i.e. when grass biomass is not sufficient to support the desired livestock density, the realized density of animals might be lower than the desired target stocking rate (see Appendix 2.B). We use the term "realized stocking rate" for this temporarily lowered stocking rate and “realized mean long-term stocking rates” (RLSR) for the average realized stocking rate that the simulated savanna could support during 200 years of simulation.

The simulated long-term livestock carrying capacity (maximum RLSR from the range of target stocking rate simulations) and the risk of savanna degradation highly depends on the specific stochastic time series of precipitation. Consequently, vegetation cover and RLSR vary greatly among the 25 repeated simulations for a given management scenario (Fig. 2.2).

Simulations without fire application show that livestock farming with high target stocking rates (6-10 LSU 100 ha^{-1}) lead to a high variability in the RLSR over 200 years (Fig. 2.2a). The optimal “no fire” scenario in terms of the highest realised long-term stocking rate allows for a RLSR of 5.3 ± 0.57 LSU 100 ha^{-1} at a target stocking rate of 6.3 LSU 100 ha^{-1}. Higher target stocking rates do not increase the RLSR when rangelands are managed without fire. For target stocking rates above 5 LSU 100 ha^{-1}, the perennial grass matrix is lost after 200 years in more than 50% of the simulations (Fig. 2.2e). At the same time, shrub encroachment (shrub cover >20%) occurs in most simulations without fire after 200 years (Fig. 2.2i). However, depending on the actual rainfall conditions, the system can resist grazing pressure for up to 200 years before the grass layer collapses and shrubs dominate the system (see sample run in Fig. 2.3).

If fire is applied on the whole farm in the year after shrub seedling emergence (“fire 100”, see Fig. 2.1 for explanation), variability in RLSR is much lower compared to the “no fire” scenario for all target stocking rates below 7.7 LSU 100 ha^{-1}. The maximum RLSR in the “fire 100” scenario is 6.9 ± 0.71 LSU 100 ha^{-1}, i.e. 30 % higher than the maximum RLSR without fire. In addition to this increase in rangeland productivity, this management strategy also leads to a more sustainable resource use compared to the strategy without fire: in more than 75% of the simulations with the optimum target stocking rate of 7.1 LSU 100 ha^{-1}, the system remains in a grass dominated state (Fig. 2.2 f&k). For all scenarios with a target stocking rate below the optimal rate (2.5-6.7 LSU 100 ha^{-1}), shrubs do not encroach and perennial grasses remain in the system (Fig. 2.2f) with at most one exception per 25 repeats (Fig. 2.2k). In contrast, the application of fires has no clear effect on long-term vegetation cover and RLSR for target stocking rates above 7.1 LSU 100 ha^{-1}. In other words, if fire is applied every year after seedling emergence (approximately every 14.9 years in our simulations) and target stocking rates are below 7.1 LSU 100 ha^{-1}, this management enables both, the sustainable use of the natural resources under cattle grazing and an increased livestock productivity.

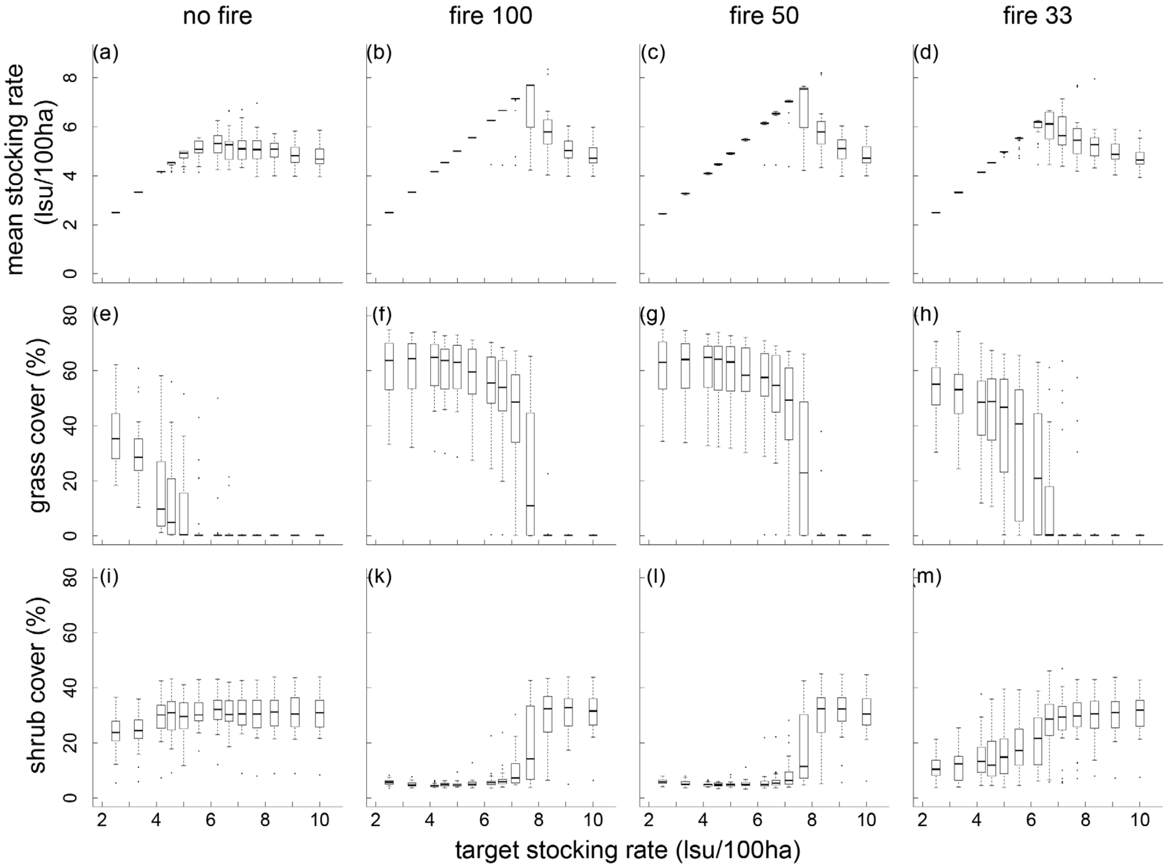

Figure 2.2: Effects of target stocking rates under four fire management scenarios on mean RLSR (over 200 years) and on grass and shrub cover after 200 years. Fire scenarios: without fire ("no fire"), with fire applied in years after seedling emergence on a whole farm ("fire 100"), with fire applied and additional adaptation of stocking rates, i.e. a burning of 50% of the farm in one year ("fire 50") and with fire applied with exception of every third event of seedling recruitment resulting in a burning of 33% of a farm in one year ("fire 33"). Boxplots show results of 25 repeated simulations.

If only 50% of the farm are burnt after years of seedling emergence (with adapted livestock densities as described above: "fire 50" scenario), results are similar to those of the scenario "fire 100" (Fig. 2.2). Finally, if fire is applied only after two out of three events of shrub seedling emergence, simulating a farm where 33% of the area are burnt at a time (scenario "fire 33"), the effects of fire application are weakened compared to the other strategies with fire. However, the maximum RLSR is still 13 % higher than in simulations without fire (Fig. 2.2d, mean 6.0 ± 0.64 LSU 100 ha^{-1} . Further, the depletion of the grass layer is much lower for target stocking rates below 6.7 LSU 100 ha^{-1} (Fig. 2.2h) and shrub cover is lower below a target stocking rate of 7.1 LSU 100 ha^{-1}. These findings indicate that the pace of savanna degradation is lowered, but degradation is not completely prevented by this strategy.

To understand the reasons leading to the reduced shrub encroachment under fire application, we show exemplary 200-year dynamics of the four fire management strategies for the same underlying precipitation time series. A series of drought years can significantly reduce the proportion of grass cover. If such a series is followed by a series of years with above average rainfall (e.g. years 22-25 in Fig. 2.3a-d), events of mass recruitment of shrub/tree seedlings occur, since grasses, which are the stronger competitors, are substantially reduced (e.g. years 26-28 in Fig. 2.3a-d). While these shrub seedlings grow and can lead to encroachment in the no-fire scenarios (Fig. 3a), fires applied at these mass recruitment events largely kill the seedlings (Fig. 2.3b-d). However, also in simulations with fire application shrub recruitment is sometimes successfully. A strong decline of the perennial grass cover induced by unfavourable precipitation events (see Fig. 2.3b years 160 & 162) can lead to strong recruitment events of shrubs (see Fig. 2.3b year 167).

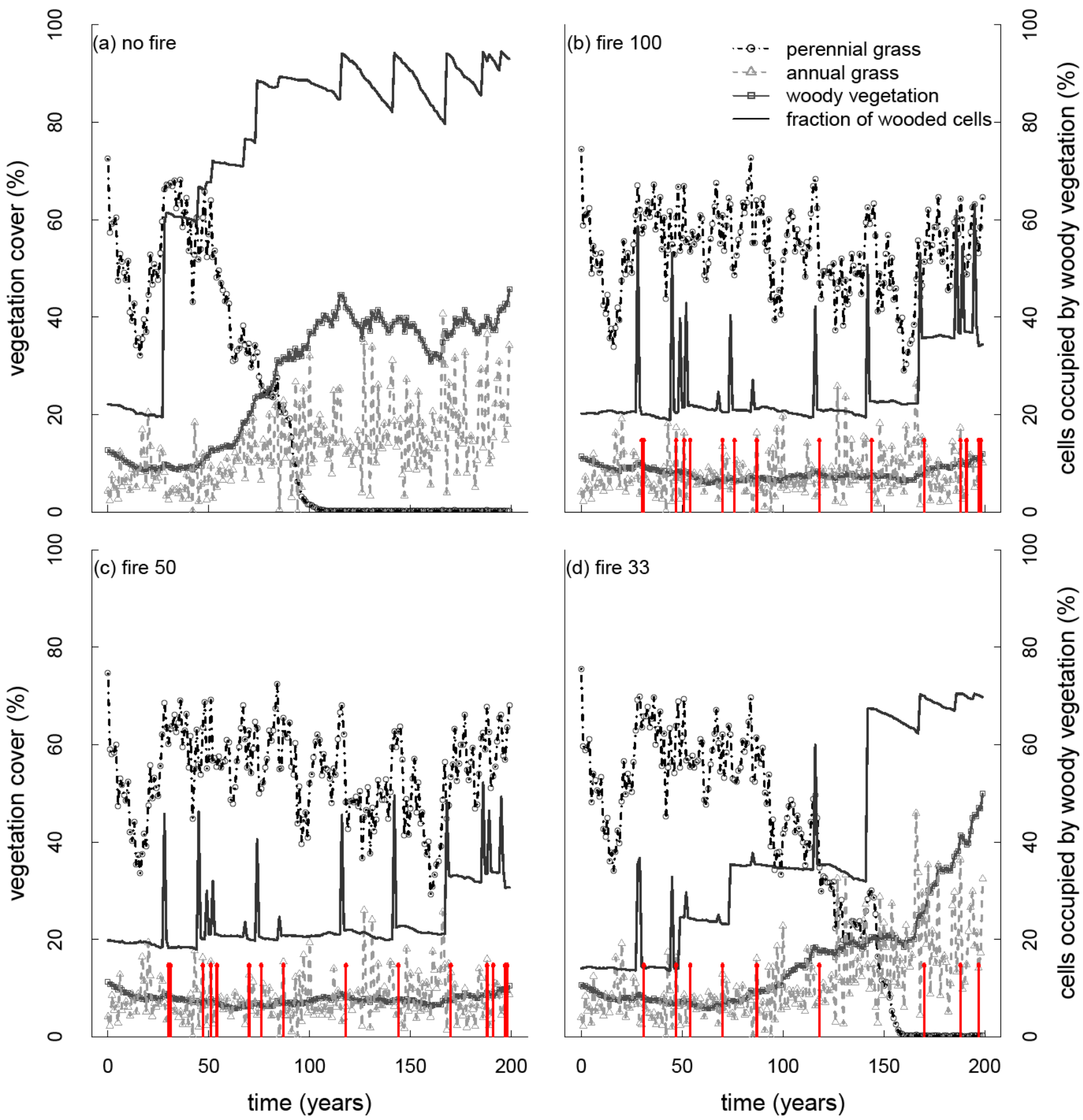

Figure 2.3: Exemplary 200 year time-series of cover of perennial and annual grasses and woody vegetation and the number of cells that are occupied by woody vegetation for simulations with four different fire scenarios. Simulations have been conducted with a moderately high target stocking rate of 7.1 LSU 100 ha^{-1}. Vertical arrows indicate the application of a fire.

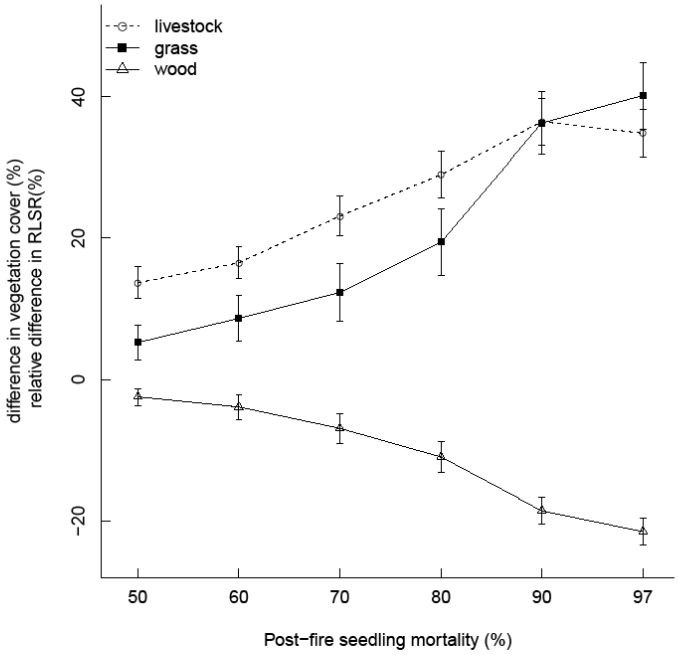

Figure 2.4: Impact of different post-fire seedling mortalities on vegetation cover and realized long-term stocking rate. Data shown are mean and standard error of the absolute difference in perennial grass ("grass") and woody vegetation ("wood") cover and the relative difference in RLSR ("livestock") between simulations with (scenario "fire 100") and without fire (scenario "no fire") from 25 repeated simulations of 200 years with a target stocking rate of 7.1 ha 100 LSU^{-1}.

Simulations with reduced post-fire seedling mortality (50%-97%, Fig. 2.4) lead to an increase in shrub cover ($F=17.45$, $p<0.001$), and a decrease in perennial grass cover ($F=13.33$, $p<0.001$) and a decreased mean RLSR ($F=10.5$; $p<0.001$). With regard to the RLSR, as a measure for the productivity of the system, even simulations with low mortality rates lead to an increased rangeland productivity compared to the "no fire" scenarios. However, rather high mortality rates (≥90%) seem necessary to sustain significant positive effects of fire on sustaining high proportions of perennial grass cover and preventing shrub encroachment in the long run.

2.4 Discussion

We applied the eco-hydrological savanna model EcoHyD to assess the suitability of prescribed fires for semi-arid savanna management. In contrast to many other studies (see below), our study revealed that fires can be of great importance for the dynamics and the management of semi-arid savanna vegetation. In particular, we could show that the application of fire in rare years of shrub seed germination and establishment has the potential to efficiently prevent shrub encroachment and to increase the long-term carrying capacity for extensive livestock production.

In agreement with many other studies, our eco-hydrological simulations revealed that semi-arid savanna vegetation dynamics are triggered to a large extent by erratic and highly variable environmental conditions (Buitenwerf, Swemmer & Peel 2011;Hodgkinson & Muller 2005). Unfavourable rainfall conditions can lead to a shift of the savanna rangeland from a vegetation state dominated by perennial grass with scattered shrubs and trees to a state dominated by annual grasses and woody vegetation (the so-called shrub encroachment). The risk for such a shift strongly depends on rangeland management, i.e. intensity of livestock grazing, but also the application of fires. Our simulation results show that prescribed fires which aim at preventing establishment of shrubs, which is their key demographic bottleneck, have great potential to prevent semi-arid savanna degradation by shrub encroachment (Joubert, Rothauge & Smit 2008;Joubert, Smit & Hoffman 2012). This is in line with other studies, which show that seedlings are especially susceptible to fire damage (Midgley, Lawes & Chamaille-Jammes 2010), that their growth is reduced after burning (Hean & Ward 2012) and that shrub seedling recruitment success is strongly limited by fires (Casillo, Kunst & Semmartin 2012;Harrington 1991;Joubert, Rothauge & Smit 2008). The potential benefit of fire management depends on the long-term stocking rates: At low to moderate stocking rates fires can substantially improve the state of the rangeland, while under continuous very intense livestock grazing the standing biomass of perennial grasses is insufficient to sustain hot fires which prevent shrub seedling establishment. This relation was also found for mesic savannas (D'Odorico, Laio & Ridolfi 2006;O'Connor, Mulqueeny & Goodman 2011;van Langevelde *et al.* 2003).

These findings contradict those of other studies, which claim that fires do not play an important role in determining the vegetation dynamics of semi-arid to arid savannas (e.g. Higgins *et al.* 2007a;Sankaran, Ratnam & Hanan 2008) and that rangeland degradation and in particular shrub encroachment are rather triggered by global drivers (Buitenwerf *et al.* 2012;Higgins *et al.* 2007a) or are part of natural cyclical dynamics (Meyer *et al.* 2007). However, these findings are based on a perspective derived from research in rather mesic savannas, where the germination and establishment of tree species does not depend on rare years but is more continuous. Thus, these studies do not explicitly consider or even completely neglect the effects of fires on tree seedling survival (Higgins *et al.* 2007a;Midgley, Lawes & Chamaille-Jammes 2010). Instead, they focus on the frequency of fires and their impact on population size or canopy cover of woody vegetation (Higgins *et al.* 2007a;Meyer *et al.* 2005). Also the study period of the investigations just did not necessarily coincide with the rarely occurring germination events (Kraaij & Ward 2006;Meyer *et al.* 2005) or grazing pressure was uncontrolled and potentially very high (Biggs *et al.* 2003;Higgins *et al.* 2007a), which according to our findings is leading to very low fuel loads and consequently to insufficiently hot fires.

Based on our results we propose to reconsider the role of fires for semi-arid and arid systems and to include the full life cycle of savanna vegetation and in particular the seedling stage in future studies. Especially since fires are also under unmanaged conditions likely to occur in years of seedling recruitment (Mulqueeny, Goodman & O'Connor 2011;Nano *et al.* 2012), since both events are coupled to above average rainfall conditions, the application of fires after mass recruitment might be a highly effective measurement to prevent shrub encroachment. Therefore, further experimental work to identify the effects of fires under different environmental and ecological conditions is emphasized by our simulation results, which show that post-fire mortalities need to be high for fires to develop the desired effects.

In addition to reduced shrub encroachment we found that fire management leads to high increases in long-term average livestock stocking rates in the semi-arid savanna rangeland. The reasons for this are twofold: If recruitment of shrubs is reduced effectively, also the risk of a collapse of the perennial grass matrix is largely reduced, since under grazing pressure grasses suffer strongly from competition by juvenile trees (Kambatuku, Cramer & Ward 2011;Kambatuku, Cramer & Ward 2012). In addition, grass growth is furthermore promoted by the fire-induced removal of accumulated moribund grass biomass, which otherwise constrains grass growth (Zimmermann *et al.* 2010).

Implications for land use

Based on our findings we strongly recommend applying fires for the management of semi-arid rangelands. While in mesic savannas the frequency of fires is critical for the success of fire management (D'Odorico, Laio & Ridolfi 2006;Smit *et al.* 2010;Van Auken 2000), fire management in more arid rangelands depends on the timing. Semi-arid rangeland managers need to flexibly respond to environmental (i.e. rainfall) and ecological (i.e. seedling emergence) conditions instead of applying a fixed burning schedule.

Fires are viewed unfavourably by many land users (Joubert, Smit & Hoffman 2012;Scholes 2009), since they are difficult to manage, have inherent risks and immediate costs, while the benefits are realised only in the longer term. When fires are applied a rangeland manager faces opportunity costs since grass biomass is not used as fodder for cattle, but to fuel fires instead. Consequently, an opportunistic use of grass biomass resources in years of above average rainfall is often more appealing (Behnke & Scoones 1993). However, also more general and economic considerations of extensive livestock production in semi-arid rangelands suggest a non-opportunistic and conservative stocking in combination with the use of fire, since the resulting increased long-term economic revenue will be more sustainable and less variable (Higgins *et al.* 2007b;Quaas *et al.* 2007).

In practice, a mode of fire management that requires the burning of a whole farm at in the same time is hardly feasible. After burning at the end of the dry season, the vegetation needs to be rested for several weeks to ensure that grasses can re-grow and recover (Tainton 1999). As a consequence, rangeland managers would have to face a temporal lack of fodder for livestock. A possible solution could be implied by the study of Joubert, Smit & Hoffman (2012), who found that a fire event in the second season after germination is as effective to reduce tree seedlings as a fire event in the first season. This allows a farmer to only burn 50% of the land after a shrub recruitment event while burning the other half in the subsequent year. In addition, cattle can feed on the other half of the farm, while the burnt area is rested for some weeks.

As shown by our simulations also a strategy with an even lower fraction of a farm being burnt during one season seems possible. If less than half of a farm (e.g. one third) is burnt at a time the fraction of the farm remaining un-burnt following shrub recruitment will be colonised by numerous woody plant saplings. However, as this area will then be burnt the two subsequent recruitment events, it will not experience any significant further recruitment for several decades. If in this situation additional measures are undertaken to control growth of the once established saplings by e.g. the application of browsers (Nano *et al.* 2012;Prins & Vanderjeugd 1993;Staver *et al.* 2012) or interim burning the intensity and speed of encroachment will be reduced and natural mortalities could keep shrub covers on acceptable levels in the long run (Joubert, Rothauge & Smit 2008;Meyer *et al.* 2007).

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2.6 Appendix 2.A – parameters of vegetation model

Name	Description	Value	Unit	Source / parameterization
$W_{WP,pg}$	pg specific wilting point	0.036	vol%	based on Neilson 1995 & Sala et al. 1989
$W_{WP,w}$	w specific wilting point	0.036	vol%	based on Neilson 1995 & Sala et al. 1989
$W_{WP,ag}$	ag specific wilting point	0.036	vol%	based on Neilson 1995 & Sala et al. 1989
θ_{pg}	Potential uptake rate per grass cover	$4.275*10^3$	mm yr^{-1}	according to Tietjen et al. (2010) $\theta_{BM,pg} * conv\_c\_bm_{pg} * area_{cell}$
θ_{w}	Potential uptake rate per woody plant cover	$4.5*10^4$	mm yr^{-1}	according to Tietjen et al. (2010) $\theta_{BM,w} * conv\_c\_bm_{w} * area_{cell}$
$\theta_{BM,pg}$	Relative uptake rate per grass biomass	0.9	mm (yr*g)$^{-1}$	Tietjen et al. 2010;van Langevelde et al. 2003
$\theta_{BM,w}$	Relative uptake rate per woody biomass	0.5	mm (yr*g)$^{-1}$	Tietjen et al. 2010;van Langevelde et al. 2003
$conv\_c\_bm_p$	Perennial grass biomass at 100% cover	$1.9 * 10^6$	g ha^{-1}	Snyman 1998;Snyman & Fouche 1991
$conv\_c\_bm_w$	Woody biomass at 100% cover	$2.1 * 10^7$	g ha^{-1}	Vanvegten 1984
$conv\_c\_bm_a$	Annual grass biomass at 100% cover	$1.7 * 10^6$	g ha^{-1}	Snyman 1998
$root_{pg,L1}$	Fraction of pg roots in upper layer	0.63	-	Jackson et al. 1996;Tietjen et al. 2010
$root_{w,L1}$	Fraction of w roots in upper layer	0.36	-	Jackson et al. 1996;Tietjen et al. 2010
r_{pg}	Potential growth rate of perennial grasses	0.55	$mm^{-1}yr^{-1}$	Calibrated to gain long-term mean perennial grass cover of 30-70% in undisturbed (low grazing) scenarios according to local expert estimates
r_w	Potential growth rate of shrubs	0.3	$mm^{-1}yr^{-1}$	Calibrated to gain woody cover of max 42% according to Sankaran et al. (2005)
r_a	Potential growth rate of annuals	1.5	$mm^{-1}yr^{-1}$	Calibrated to gain long-term mean annual grass cover of 5-40% in degraded savanna according to local expert estimates
bm_c_rain	Constant for linear increase of biomass per unit of cover depending on annual precipitation	0.35	-	Calibrated to gain slope of biomass – rain relation in Snyman & Fouche (1993) for savannah in good condition
lap	Extent to which shrubs and grasses can overlap	0.2	-	Tietjen et al. (2010)

mrd_{pg}	Mortality rate of perennial grasses due to water stress	0.54	$mm^{-1}yr^{-1}$	Calibrated to gain average mortality of 26% in drought years (<250mm) as given by O'Connor & Everson (1998)
mrd_w	Mortality rate of shrubs due to water stress	0.12	$mm^{-1}yr^{-1}$	Calibrated to gain average mortality of 5% in drought years (<250mm) as given by Meyer et al. (2007)
$cmax_{pg}$	Maximum cover for grasses	1.0		Expert estimate
$cmax_w$	Maximum cover for shrubs	0.8		Sankaran et al. (2005)
est_{pg}	Rate of successful establishment of grasses	0.05		Tietjen et al. (2010)
est_s	Rate of successful establishment of shrubs	0.005		Tietjen et al. (2010)
dca	Constant for exponential decline of spatial establishment with distance	0.26		Tietjen et al. (2010)
$dist_0$	Constant for exponential decline of spatial establishment with distance	0.5		Tietjen et al. (2010)
cd_{min}	Fraction of maximum value of spatial establishment function at which function terminates	0.01		See supplementary material SA1 section “dispersal”
gb_g	Fraction of grass biomass that cannot be used by cattle	0.15		Tainton (1999)
gb_w	Fraction of woody biomass that cannot be used	0.5		Skarpe (1990)
m_{est}	Factor determining minimum mean soil moisture content needed for establishment relative to Wilting Point	1.21		Chosen so that establishment condition occur Joubert, Rothauge & Smit (2008)
$frac_{pg}$	preferred ratio of perennial grasses in cattle diet	0.65		Rothauge (2006)
$frac_{ag}$	preferred ratio of annual grasses in cattle diet	0.35		Rothauge (2006)
$frac_w$	maximum additional ratio of browse in cattle diet relative to grass uptake	0.075		Tainton (1999)
$inishare$	Constant of spatial heterogeneity of grazing	1.5		Estimated according to Jeltsch et al. (1996) & Weber & Jeltsch (2000)
ga	Constant shaping quadratic function of grazing damage	0.8		Expert estimate
gb	Constant shaping quadratic function of grazing damage	0.1		Expert estimate
dec_{ag}	Decay rate of annual grass biomass per year	0.9	yr^{-1}	Expert estimate
dec_{pg}	Decay rate of annual grass biomass per year	0.75	yr^{-1}	Expert estimate
$growStart$	First day of growing season	150		Tietjen et al. (2010)
$growEnd$	Last day of growing season	330		Tietjen et al. (2010)
FM	Fuel Moisture	19.6	%	Joubert et al. (2012)
RH	Relative humidity	35.6	%	BIOTA database www.biota-africa.de
WS	Wind speed	1.43	$m\ s^{-1}$	BIOTA database

sLS	Specific fire induced loss of shrub cover per unit of energy	0.01	W^{-1}	vanLangenfelde et al. (2003)
sLG	Specific loss of grass cover if fire occurs	10	%	vanLangenfelde et al. (2003)
mLS	Maximum loss of shrub cover due to fire	15	%	Joubert et al. (2012)
cF	coefficient determining the increase in fire intensity with fuel load	0.5	$W \cdot m^{-2} \cdot g^{-1}$	vanLangenfelde et al. (2003)

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2.7 Appendix 2.B – model rules

Our model is based on the eco-hydrological dryland model *ECOHYD* to which we refer where possible in this description of model rules (Tietjen *et al.* 2010). It is a combination of a savannah vegetation model calculating the biweekly growth of two plant functional types (shrubs and perennial grasses) and an hydrological model, calculating daily moisture dynamics in two soil layers (Tietjen, Zehe & Jeltsch 2009). We do not describe the hydrological model since this was not changed and a full description can be found elsewhere (Tietjen *et al.* 2010;Tietjen, Zehe & Jeltsch 2009). The vegetation model was changed in order to simulate grazing in more detail. In this version we added a fire module, which is described in the main text of this paper. We only made very minor changes to the rest of the vegetation model (e.g. including moribund grass biomass).

A list with description, value and source of all parameters is given in tables ST1 and ST2 in supporting information for the vegetation and the hydrological model respectively. Parameter and variable names used are as used in the work of Tietjen *et al.* (2010).

Vegetation model

Changes in vegetation canopy cover (c_{veg}) per cell of perennial grasses (pg), woody vegetation (w) and annual grasses (ag) depend on the processes growth (gr_{veg}), dispersal/establishment (ds_{veg}), mortality (m_{veg}) and losses due to herbivory (h_{veg}) and (equation 1). The subscription veg refers to the three vegetation types (veg: pg,ag,w).

$$\frac{dc_{veg}}{dt} = gr_{veg} + ds_{veg} - m_{veg} - h_{veg} \; [dimensionless] \qquad \text{eqn 1}$$

In the following description of the model rules we refer to the original model description (*EcoHyD*) of Tietjen *et al.* (2010). Variable names are in-line with those found in Tietjen *et al.* (2010) for good comparability. Detailed descriptions are only given where changes have been made compared to the original model version. Exceptions are the functions for growth and mortality. They are described in detail, despite the fact that no changes have been made because this is crucial for a general understanding of the model structure and functioning.

Plant growth

Plant growth is implemented as increase in vegetation cover calculated in intervals of 14 days during a defined growing season. Growth of perennial grasses and shrubs is hereby based on a logistic function including the cover of the two perennial growth forms (c_{pg}, c_w) to represent competition for space, assuming a potential overlap of the both growth forms (*lap*). It furthermore depends on water availability in the two soil layers ($avW_{veg,Lx}$), competition for water in each layer ($U_{veg,Lx}$), a site-specific maximum cover for each plant type ($cmax_{veg}$) and a potential growth rate (r_{veg}).

The growth function for perennial grasses and woody vegetation is (according to Tietjen *et al.* 2010):

$$gr_{veg,Lx} = min\left(U_{veg,Lx} * avW_{veg,Lx}, 1\right) * r_{veg} * c_{veg} * \left(1 - \frac{c_{veg}}{cmax_{veg} - \left(c_{-veg} * (1 - lap)\right)}\right) [yr^{-1}]$$

eqn 2

Note, that since we included moribund grass biomass to this model version, the growth of perennial grasses in the upper soil layer ($gr_{pg,L1}$) is additionally limited by the presence of moribund grass material (c_{mor}) so that the l last term in eqn 2 is changed to

$$\left(1 - \frac{c_{veg}}{cmax_{veg} - \left(c_{-veg} * (1 - lap)\right) - c_{mor}}\right).$$

eqn 2a

Water availability for a vegetation type in a soil compartment ($avW_{veg,Lx}$) is calculated as follows (see Tietjen *et al.* 2010):

$$avW_{veg,Lx} = \begin{cases} 0 & if\ W_{Lx} \leq W_{WP,veg} \\ \left(W_{Lx} - W_{WP,veg}\right) * depth_{Lx} & if\ W_{Lx} > W_{WP,veg} \end{cases} [mm]$$

eqn 3

with W_{Lx} [vol %] the mean water content of the soil layer during the last 14 days, $W_{WP,veg}$ [vol %] the plant specific wilting point, and the depth of the soil layer $depth_{Lx}$ [mm]. This subroutine results in linearly rising water availability for increasing soil water contents as long as the water content is above $W_{WP,veg}$.

The fraction of available water that can be taken up in each soil compartment by each plant type $U_{veg,Lx}$ is calculated based on a potential water uptake rate per cover θ_{veg} [mm * yr^{-1}], vegetation cover c_{veg} and the fraction of roots $root_{veg,Lx}$ in the respective layer (extending the approaches of Walker *et al.* 1981 and van Langenvelde *et al.* 2003).

$$U_{veg,Lx} = \frac{\theta_{veg} * root_{veg,Lx}}{\theta_{pg} * root_{pg,Lx} * c_{pg} + \theta_w * root_{w,Lx} * c_w} [dimensionless]$$

eqn 4

Growth of annual plants in contrast is given by equation 2.5 and does not explicitly include any competition for water but exclusively depends on availability of empty space, growth rate (r_{ag}) and general water availability in the upper soil layer ($avW_{ag,L1}$), since annual plants are not assumed to invest resources in deep and dense root systems. Like in equation 4 we assume a potential overlap (*lap*) between grasses and shrubs.

$$gr_{ag} = min\left(1 * avW_{ag,L1}, 1\right) * r_{ag} * \left(1 - c_{pg} - c_s * (1 - lap) - c_{ag} - c_{mor}\right)[yr^{-1}]$$

eqn 5

Growth consequently decreases with increasing overall vegetation cover and is consequently potentially fastest at the beginning of the growing season if growth is not limited by a big amount of moribund grass material.

In equation 4, annuals are not included since they are assumed to be the clearly inferior competitor for water, as perennial grasses and shrubs are already present with extensive root systems when the rainy season begins. However, annual plant's impact on soil water contents by transpiration and evaporation is of course taken into account in the hydrological submodel in the same way that it is considered for the perennial grasses but assuming the annuals to root in the upper soil layer only (for details of hydrological model see Tietjen et al 2009).

Plant mortality

Two types of mortality affect vegetation cover. First, drought induced mortality (md_{veg}) is calculated exactly as described in Tietjen *et al.* (2010). It is based on water availability and uptake analogous to growth (see equations 2-4) and depends on a drought mortality rate mr_{veg} , the average available water content in both soil layers during the growing season ($avW_{veg,Lx}$) and the proportional uptake of this water ($U_{veg,Lx}$).

$$md_{veg,Lx} = mr_{veg} * c_{veg} * \left[\left(1 - min\left(U_{veg,Lx} * avW_{veg,Lx}, 1\right)\right) * \frac{root_{veg,Lx}}{\sum_i root_{veg,Li}}\right] \; [yr^{-1}] \qquad \text{eqn 6}$$

Secondly, we introduced stochastic age based mortality (ma_s) for woody vegetation, referring to empirical data on *Acacia mellifera* L. from a semi-arid savannah similar to the one found in the study area (Meyer, Wiegand & Ward 2009). This simulates a mortality that rather depends on the age of individuals than on water stress like for example infestation by fungi (Joubert, Rothauge & Smit 2008). This senescence is applied to all cells with cohorts of shrubs older than the average age of death of individual trees ($ScenAge$) (Meyer *et al.* 2007). The age of a cohort is determined by the date of the last establishment event that occurred in the respective cell. Hence, cells where the last establishment event of woody vegetation has been more than $ScenAge$ ago are completely cleared from woody vegetation with an annual probability of mp_{age}.

Biomass production

Ground cover of the different vegetation types is the basic unit used in most equations of EcoHyD (important e.g. for infiltration, evaporation and surface water run-off). Since we are aiming at a quantitative representation of grazing, we need to translate cover values to biomass. This enables us to derive system productivity in terms of livestock carrying capacities from our analyses. In a semi-arid and thus water limited system, biomass production from a given vegetation cover depends on the amount of rain received (Snyman & Fouche 1993). More precisely, biomass production does not only depend on cover, but also on the height of e.g. grass shoots or the thickness of leaves etc. which in term is assumed to mainly depend on precipitation in a water limited system. Biomass $\left(b_{veg}\right)$ is deduced from cover $\left(c_{veg}\right)$ and the average biomass produced per unit of cover in years with average rainfall quantities ($conv\_c\_bm_{veg}$) depending on the following linear relation:

$$b_{veg} = c_{veg} * conv\_c\_bm_{veg} * cf(rain) \qquad \text{eqn 7}$$

According to the abovementioned dependence of biomass production on precipitation, the slope of this relation is varied by the factor $cf(rain)$ according to the actual year's precipitation ($rain$) and mean annual precipitation (MAP). The value of $cf(rain)$ is 1 in case of an average seasonal rainfall, below 1 in case of lower and above 1 in case of higher rainfall amounts according to the below given linear relation (equation 8) with a constant ($\beta_r \leq 1$) determining the strength of the impact of rain on the cover-biomass relation.

$$cf(rain) = rain * \frac{1-\beta_r}{MAP} + \beta_r \qquad \text{eqn 8}$$

Grazing and browsing

We applied an algorithm that based on ideas underlying established grazing algorithms (used by e.g. (Weber & Jeltsch 2000a). Basic underlying assumptions for grazing are:

1) grazing is heterogeneous in space, i.e. cattle tends to deplete preferred resources where it is if resources are available rather than moving further (Weber *et al.* 1998;Weber & Jeltsch 2000b).
2) cattle feeds selectively, i.e. it prefers to feed on perennial grasses before annual grasses before shrubs (Rothauge 2006;Tainton 1999).
3) savannah vegetation is to some extent adopted to grazing (Tainton 1999).

Total annual biomass demand is calculated dependent on animal numbers per area, average animal's body weight and average daily biomass need of cattle– the latter being 2% of the livestock body mass per day (Tainton 1999). In case that this demand exceeds the available grass (annuals and perennials and moribund) biomass, we assume that additional fodder is given to the animals and herd size remains constant.

In the next step, the mean biomass need per cell (bm_m) is derived by dividing total biomass need by the number of cells. Subsequently, individual cells are chosen randomly and from every cell biomass is removed. This biomass removal from random cells is repeated until the total biomass demand is covered. Thus, individual cells may be chosen several times.

Accounting for the spatial heterogeneity of grazing mentioned above, the biomass to be removed from a cell per grazing attempt (bm_{rem}) is calculated as follows:

$$bm_{rem} = bm_m * \gamma_g \quad [kg] \quad with\, \gamma_g > 1 \qquad \text{eqn 9}$$

The heterogeneity factor (γ_g) determines the strength of spatial variation of grazing. We assume γ_g to be above one, since cattle is known to preferentially feed on sites where enough resource is available before moving to the next site.

Since grazing is considered to be selective, biomass is taken from perennial grasses, annual grasses and shrubs according to a defined ratio ($frac_{veg}$):

$$bmr_{veg} = bm_{rem} * frac_{veg} \quad [kg] \qquad \text{eqn 10}$$

However, if the finally determined demand of biomass of a plant life form in a given cell (bmr_{veg}) is higher than the available biomass, the latter is totally removed. Hereby we assume a limited maximum fraction (gb_{veg}) of the given biomass of a plant type (BMO_{veg}) to be available, because cattle cannot graze the biomass completely down to the ground and not all parts of the different plants are edible (Skarpe 1990;Tainton 1999). Note, that whenever biomass of perennial or annual grasses is removed, the model first removes biomass from the moribund biomass of the respective vegetation type according to its fraction of total biomass (e.g. if moribund biomass is 10% of all biomass, cattle will cover 10% of its' biomass need with moribund biomass).

$$bma_{veg} = gb_{veg} * BMO_{veg} \quad [kg] \qquad \text{eqn 11}$$

If the biomass demand (bmr_{veg}) is close or equal to the available grass biomass (bma_{veg}), it is likely, that cells are chosen more than once by random selection. In such situations, when grass biomass is scarce, grazing becomes more homogeneous in space since animals make more efforts to find all available resources. If a cell is "grazed" a second time, the algorithm will still try to remove biomass according to the defined fractions ($frac_{veg}$), but it is likely that the preferred types' biomass is already depleted. Consequently, repeated selections of a cell lead to a shift of effectively realized fractions of biomass removal towards the less preferred types (annuals and shrubs) compared to the ratios given by $frac_{veg}$. In this way we conditionally simulate unselective and spatially homogeneous grazing. In summary this means, that the diet of the animals changes dynamically if grazing pressure increases or resource availability decreases (Rothauge 2006, Tainton 1999).

After determining the total amount of biomass that is removed from a cell (BMG_{veg}), the reduction of cover of the respective vegetation types due to herbivory (h_{veg}) is calculated.

$$h_{veg} = \frac{\propto_{graze} * BMG_{veg}}{conv\_c\_bm_{pgrass} * cf(rain)} \quad [dimensionless] \qquad \text{eqn 12}$$

Reduction of biomass does not directly – i.e. according to the cover-biomass relationship given in equation 7 – translate back into a reduction of cover, accounting for adaptation of grasses to grazing. Therefore we include a relative damage caused to plant cover ($\propto_{graze}$), which increases linearly with an increasing fraction of biomass that was removed by the grazers ($\frac{BMG_{veg}}{BMO_{veg}}$) as given in the following relation:

$$\propto_{graze} = \left(\frac{BMG_{veg}}{BMO_{veg}} * ga + gb\right) \quad [dimensionless] \text{ with } ga + gb \leq 1 \qquad \text{eqn 13}$$

The reduction in cover due to grazing is consequently calculated by incorporating $\propto_{graze}$ in the abovementioned relation between biomass, cover and precipitation (equation 7). The parameters ga and gb define the shape of this quadratic function of cover reduction (equation 11 in equation 12 results in a quadratic function of BM_{rem}).

Dispersal and seedling establishment

Dispersal and establishment are simulated as addition to the cover of a respective growth form (ds_{veg}) to certain cells in the grid. This is rather representing seedling dispersal than seed dispersal. Germination and seedling/juvenile survival are therefore rather implicitly included (Tietjen *et al.* 2010).

Dispersal and establishment of perennial grasses is implemented as described by Tietjen *et al.* (2010). We assume no dispersal limitation on the given spatial scales (Jeltsch *et al.* 1997), i.e. spatially homogeneous distribution of grass cover with the amount of cover depending on the mean perennial grass cover of the whole grid. Annuals are assumed to be always present as seeds and start off at every season without initial cover (i.e. no dispersal and establishment calculation necessary, solely growth function determines occurrence). Woody plants are, in accordance with literature on regional typical shrub and tree species (i.e. *Acacia* species), assumed to be limited in dispersal, seed production and especially in establishment (Barnes 2001;Joubert, Rothauge & Smit 2008;Meyer *et al.* 2007;Tews, Schurr & Jeltsch 2004).

However, the establishment of shrubs is simulated in more detail compared to the model version of Tietjen *et al.* (2010). Dominant encroacher species in semi-arid African savannahs are known to have relatively high requirements regarding water availability for seed production, seedling germination and successful establishment. Different studies showed, that at least 2 subsequent years of above average rainfall are needed for successful establishment of *A. mellifera* (Barnes 2001;Joubert, Rothauge & Smit 2008;Meyer *et al.* 2007) and other woody plant species of semi-arid savannahs (Wilson & Witkowski 1998). Hence, successful establishment of woody vegetation is only possible if the mean soil-water content in the upper soil layer during the growing season is well above the wilting point of plants during two subsequent years $\left(W_{L1,mean} > m_{est} * WP_s\right)$. The factor m_{est} was calibrated so that establishment conditions at one location occur on average 5-6 times per century (Joubert, Rothauge & Smit 2008).

The dispersal and establishment of shrub seedlings added as cover to a target cell (ds) is calculated for every source cell in the grid if the target cell had a sufficient water availability during the last and current growing season and its position was within the maximum dispersal distance according to the following term:

$$ds = c_{s\_source} * est_s * dist_0 * e^{-dc*dist} * max\left(1 - c_s - c_{pg}; 0\right) \ [dimensionless] \qquad \text{eqn 14}$$

Establishment and dispersal consequently depend on shrub cover in the source cell (c_{s_source}), mean rate of seedling establishment (est_s), cover of grasses and shrubs in the target cell (c_s, c_{pg}) and the shape of an exponential dispersal decline (dependent on $dist_0$ and dc) as well as on the distance of the target from the source cell ($dist$).

The maximum dispersal distance $distmax_s$is calculated so that the added cover ds is at least 1% (cd_{min}) of the maximum possible value of ds at the centre of the source cell ($dist = 0$).

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Link to the following chapter

In chapters 2 and 3 I identified the impact of environmental and ecological conditions on semi-arid savanna dynamics and the implications for sustainable land use. My simulations showed that climatic variability is of central importance for semi-arid rangeland dynamics. Series with above average precipitation trigger pulses of shrub recruitment, while protracted droughts can cause severe declines of perennial grasses, especially when combined with high grazing pressure.

The resulting variation in perennial grass biomass which is available as fodder for livestock is of major importance for land users in semi-arid savannas and will have strong impacts on the productivity of rangeland businesses. In order to facilitate the sustainable use of the resources provided by semi-arid savannas, land users should thus be aware of vegetation dynamics and manage the ecosystem accordingly.

However, rangeland managers are not only influenced by ecological and environmental conditions but also by economic, social and political frame conditions of land use. In southern Africa, the commercial livestock production sector is subject to transformations according to the political changes during the second half of the 20th century. Land reform projects, mainly aiming at poverty alleviation and the compensation of imbalances that root back to the past discriminatory political systems, redistribute land in order to enable formerly disadvantaged citizens to get involved in (commercial) livestock production.

In the following chapter, I want to analyse how beneficiaries of the Namibian ´commercial` land reform decide upon resource use. To achieve this, I develop an ecological-economic model simulating commercial livestock farms which is then applied to conduct interactive role plays with Namibian land reform beneficiaries. In particular, I want to identify whether their decisions concerning herd size adjustments are determined by environmental variation (i.e. precipitation and vegetation state) or economic aspects (i.e. available assets or running costs). I further want to assess whether the identified management strategies are successful with regard to their ecological and economic performance.

Chapter 3

Determinants of semi-arid rangeland management in a land reform setting in Namibia[1]

Summary

To assess the ecological and economic implications of the redistributive land reform in semi-arid Namibia, we investigated to what extent land reform beneficiaries adjust herd size and herd composition according to environmental (rainfall, vegetation state) and economic variables (herd size, financial assets, running costs). We performed model-based role plays with Namibian land reform beneficiaries, simulating 10 years of rangeland management.

Our study revealed that the surveyed farmers mainly manage their herds according to their economic situation (herd size and account balance) but do not take environmental variability (rainfall and vegetation state) into account. Further, our results indicate that, due to financial pressure, farmers are not able to apply their desired management strategies and that owners of small farms have a higher risk for economic failure. However, farmers applied rather conservative and constant stocking rates and would thus, given the current economic limitations, likely not contribute to semi-arid savanna degradation.

We conclude that land reform beneficiaries need support to be able to apply straightforward and efficient management strategies. This could be achieved by facilitating cooperation between small farming businesses and by supporting an initial build-up of a productive cattle herd at the time of redistribution of the land.

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3.1 Introduction

Since decades, semi-arid African rangelands are prone to degradation, mostly in form of bush encroachment at the cost of palatable perennial grasses (Skarpe 1990; Jeltsch, Weber & Grimm 2000; Graz 2008). This form of rangeland degradation leads to severe declines in ecosystem services such as the production of grass and livestock biomass, groundwater recharge, carbon sequestration or prevention from soil erosion (UNCCD 1994; Graz 2008; Lehmann 2010) but also to significant losses in biodiversity across taxonomic groups (Blaum *et al.* 2009; Blaum, Tietjen & Rossmanith 2009). This degradation is mainly attributed to unsustainable rangeland management (e.g. high livestock densities or fire suppression) which is aggravated by increasing pressure on land due to human population growth (causing even higher livestock densities), but also to global changes such as climate change and rising atmospheric CO_2 levels (UNCCD 1994; Reynolds *et al.* 2007; Lohmann *et al.* 2012).

Against this background of complex degradation dynamics land reform policies of southern African countries additionally change the frame conditions for land use (Clover & Eriksen 2009; Marchant 2010). Amongst others, land reforms shall redress past discriminative imbalances in the distribution of economic resources. In particular, land shall improve food self sufficiency, bring small-holder farmers into the mainstream economy, create employment, and alleviate human and livestock pressure in communal areas (Adams 1993; Republic of Namibia 1995). Hereby, the success of these projects obviously depends on the application of ecologically and economically sustainable and viable land use strategies.

However, already irrespective of the changes going along with the abovementioned redistribution of land, the fundamental question of how to manage semi-arid rangelands in a sustainable manner, i.e. how to avoid degradation while maintaining economic profitability is still not finally solved. There is an ongoing debate on whether management should be opportunistic, i.e. resource availability should be tracked with animal numbers, or whether conservative strategies should be favoured, where more or less fixed and low stocking densities are applied (Illius & O'Connor 2000; Cowling 2000; Sandford & Scoones 2006). Different ecological theories have described semi-arid savannas as either, equilibrium or non-equilibrium systems (Jeltsch, Weber & Grimm 2000; Gillson & Hoffman 2007). However, recent studies show that savanna vegetation dynamics are most likely determined by a combination of top-down, ‘non-equilibrium’ processes (i.e. fires and large herbivores) as well as bottom-up, ‘equilibrium’ processes (i.e. strong competition for the scarce and variable key resource water) (Kulmatiski *et al.* 2010; Nano & Clarke 2010; Kambatuku, Cramer & Ward 2012; Taylor *et al.* 2012; Nano *et al.* 2012). Consequently, one would expect different rangeland management strategies to be optimal, depending on which of the abovementioned ecological concepts is applied.

However, as rangeland management is not only depending on ecological but also on economic conditions, further complexity arises. Already on a farm scale, additional factors such as opportunity and transaction costs, risk aversion, and discounting have to be considered (Buss 2006; Quaas *et al.* 2007). In addition we also need to understand the impact of political measures, like e.g. redistributive land reforms, but also of social and cultural aspects of land use on land users decisions (Ostrom 2009). For the identification of viable sustainable solutions, all these aspects have to be considered in trans- and interdisciplinary research approaches that do explicitly account for the different stakeholders' perspectives (Neef & Heidhues 2005; Eriksen & Watson 2009; Marchant 2010).

In southern Africa, redistributive land reform programs change conditions under which rangeland management takes place (Clover & Eriksen 2009). After redistribution land is often managed in smaller units (Werner & Odendaal 2010). Previous land users are replaced by new ones who have a different cultural and economic background and who often start farming with little or no experience in commercial rangeland management (Falk *et al.* 2010). Since the farming expertise of these emerging farmers is mostly based on experiences in communal livestock farming, their farming goals might differ from those of existing commercial farmers and hence goals other than financial revenue like e.g. insurance or social status could lead to a strategy of herd size maximisation with the respective ecological consequences (Cousins *et al.* 2007; Allsopp *et al.* 2007; Falk 2008; Prediger, Vollan & Frolich 2011). However, the success of southern African land reform programmes is often seen critically (Adams 1993; Clover & Eriksen 2009) and there is still insufficient knowledge about how land reform beneficiaries cope with challenge of managing a highly variable natural resource that is threatened by non-linear and partly irreversible degradation.

Having in mind the negative ecological implications of savanna degradation (e.g. Blaum, Tietjen & Rossmanith 2009) it is not only of social and economic importance that land reform beneficiaries succeed in managing their rangelands. Besides the identification of theoretically optimal management strategies research should also assess the status quo of how land reform beneficiaries manage semi-arid rangelands to be able to provide decision makers (e.g. farmers, policy makers, extension services, NGOs) with useful information (Marchant 2010). In particular this should comprise the assessment of economic and ecological preconditions of such farmers, but also the question of how and to what extent they adapt to the environmental and economic conditions they experience.

In this study, we assessed the management decisions of Namibian land reform beneficiaries in the Omaheke region. In cooperation with the Emerging Commercial Farmers Support Program (ECFSP), the GIZ Namibia, the Polytechnic of Namibia, and the Namibian Ministry of Agriculture, Water, and Forestry (MAWF) we developed an agent-based ecological-economic simulation model of extensive cattle farms in a semi-arid African savanna. We used this model to conduct simulation-based role plays with land reform beneficiaries to assess what factors influenced their decisions regarding alterations in the size of their herd. By using the model-based approach, we created realistic scenarios, replicating the real life situation of the participants (i.e. regarding herd and farm size and variability of resources). We were particularly interested in the factors determining the farmers' decisions to buy or sell animals. This decision mainly characterizes the interaction between ecological and economic system, as herd size is the fundamental determinant of rangeland management. In particular, we assess to what extent their decisions are based on basic environmental (rainfall, vegetation state) and economic variables (herd size, financial assets, running costs). Further, we discuss the implications of our findings for the future of redistributive land reform programs in semi-arid Africa.

3.2 Materials and Methods

3.2.1 Study system

The research units of this experimental study are farms that have been allotted to beneficiaries of the Namibian land reform (Republic of Namibia 1995). At this, the acquisition of land is based on the preferential right of the Namibian state to purchase agricultural land whenever any owner of such land intends to dispose of it (willing-seller willing-buyer principle). We included farms that have been redistributed according to the two most important land reform instruments (Republic of Namibia 1995; Republic of Namibia 2002) namely the Affirmative Action Loan Scheme (AALS), where beneficiaries buy the land with subsidized loans and the National Resettlement Programme (NRP), where relatively small parts of land are allotted to mostly very poor people.

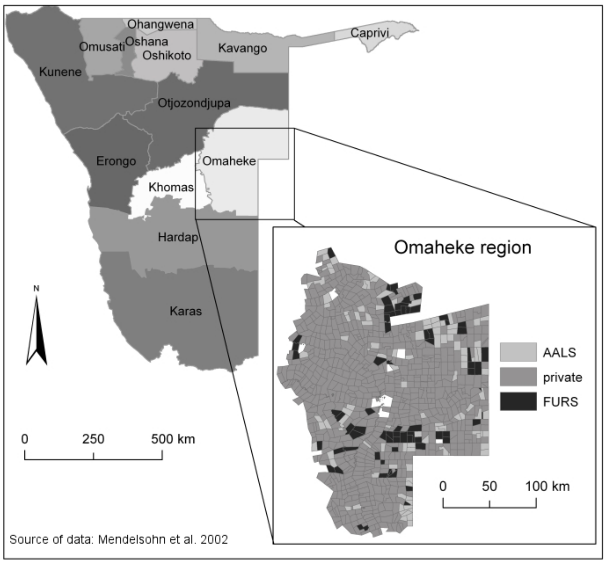

Figure 3.1: Location of the study region in Namibia and farm borders with land tenure information.

The study was conducted in the Omaheke region of Namibia (see Fig. 3.1). Our research concentrated on the eastern part of the region where the vegetation is dominated by Acacia-tree-and-shrub savanna of the Central Kalahari type (Mendelsohn *et al.* 2002). The mean annual precipitation (MAP) is between 300 mm – 400 mm and the region is considered to be a high potential livestock farming area. The vegetation module of the model applied here (Lohmann *et al.* 2012) was parameterized for the conditions found at the Sandveld research station (latitude 22°02´S, longitude 19°07´E) that is run by the Namibian Ministry of Agriculture, Water and Forestry (MAWF).

3.2.2 Study Design

We assessed 65 land reform beneficiaries' responses to different environmental (rainfall, vegetation state, animal condition) and capital variables (costs, herd size and available money). The performed survey was a simulation-based interactive role-play in which participants were asked to play 10 years of farming. At this, the respondents were requested to make decisions regarding the amount of animals and the type of animals (i.e. cow, oxen, male/female weaned calf or heifer) to buy or sell in every year depending on the information that was given to them by the simulation model. Hereby the interaction between the computer model and the role-players was facilitated by the research team throughout the duration of the game. A comprehensive protocol of the simulation-based role plays including a list of in and output variables/parameters is given in Appendix 3.A). The results of a parallel survey of the same farmers based on semi-structured interviews can be found in Falk *et al.* (2010).

3.2.3 Ecological-Economic Model

We use an agent-based ecological-economic model to simulate rangeland management during the role-plays. A farm sub-model simulates animal herd dynamics (including different classes of cattle, i.e. cows, oxen, male and female weaners and heifers), different fixed and variable costs and allows for interactive trading of livestock at two times during one simulated season. A grid-based vegetation sub-model (100x100 m^2 resolution) simulates the dynamics of a semi-arid savanna based on a state-and-transition approach (Westoby, Walker & Noy-Meir 1989). At this, vegetation dynamics (i.e. perennial grass dynamics are derived from simulations with an established eco-hydrological simulation model which simulates the dynamics of trees, perennial and annual grasses (Tietjen *et al.* 2010; Lohmann *et al.* 2012). Structure and process flow of the ecological-economic farm model are depicted in Fig. 3.2. All model rules have been discussed and advanced with members of the Namibian Emerging Commercial Farmers Support Programme (ECFSP) and rangeland experts from the MAWF research station at Sandveld. Further the model was improved according to the results of several test runs of the model-based role play with Namibian land reform beneficiaries. Rules are based on the recommendations of the joint presidency committee (JPC) of the Namibia National Farmers Union (NNFU) and the Namibia Agricultural Union (NAU) who developed management guidelines for the ECFSP (Stehn 2008a; Stehn 2008b).

A full description of how we generated the transition probabilities for the vegetation- sub-model by means of simulations with the established high-resolution eco-hydrological model EcoHyd (Tietjen *et al.* 2010; Lohmann *et al.* 2012) is given in Appendix 3.B. Further, we describe in detail the structure, rules and parameterization of the interactive farm-scale ecological-economic model that we used for the interactive and simulation-based experiments in the field in Appendix 3.C.

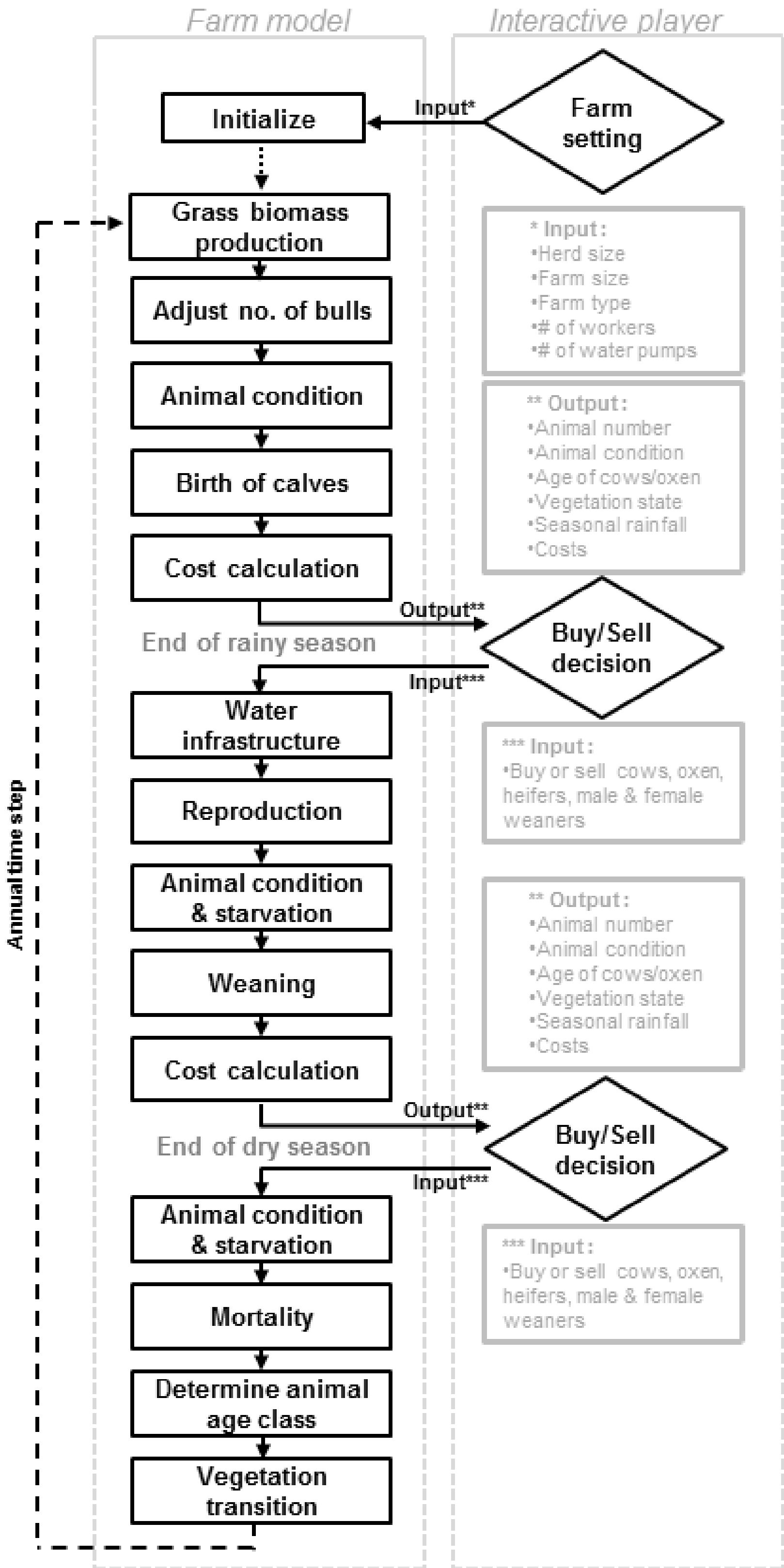

Figure 3.2: Flow chart representing the order of processes in the model and model in-/output. At the end of the rainy season (Apr/May) and of the dry season (Sep/Oct) the players are requested to enter a decision regarding how many animals to buy or sell. The grey boxes on the right describe the variables relevant for data in- and output at the interface between model and role-players.

3.2.4 Data analyses

During the experiment all input and output data of the simulation model were recorded. We calculated the number of animals on the farm in large stock units (LSU) by summing up all animals weighing them according to livestock classes (female and male weaner, heifer, cattle, oxen). The size of the farms varies in real life as well as in the experiment and consequently the economic potential differs. For this reason we set the numbers of bought and sold animals, the account balance, the costs and the livestock numbers in relation to the farm size.

In order to identify the effects of different ecological and economic variables on the farmers decision how many animals to buy or sell we applied a linear mixed effects model in R, version 2.10.1 (R Development Core Team 2009) using the packages "nlme" (Pinheiro *et al.* 2009) and "MASS" (Venables & Ripley 2002). We wanted to know whether farmers were influenced by the variables annual rainfall (rain), average vegetation state (vegstate), animal condition (cond), incurring annual costs per hectare (cost_ha), account balance per hectare (account_ha) and herd size per hectare (lsu_ha) regarding their decision of buying and selling animals (lsu_bs_ha). Since we were interested in the adaptation of livestock densities we use the difference between bought and sold large stock units as dependent variable. First we visually analysed the distribution of the dependent and the different independent variables. Inspection of the dependent variable, i.e. the number of animals bought minus sold per year and hectare revealed that three of the 65 surveyed farmers showed very unrealistic behaviour and were thus excluded from the analyses. One farmer did not show any response (neither sell nor buy animals) throughout the role-play. Two other farmers made several trading decisions that were very extreme regarding the amount of animals that were bought or sold, i.e. they repeatedly traded 1 to 2.5 times the regional carrying capacity during one single year and thus produced several outliers. With regard to the independent variables we found that the variable "animal condition" was extremely stable throughout the role-plays and only hardly ever varied from its initial value. Thus we did not assume any effect of this variable on the behaviour of the farmers and we decided to not include it into the model. Further, we had to exclude the variable incurring costs, since it was strongly correlated with the variable herd size ($r=0.58$, $p<0.001$). This can be explained by the fact that costs are mainly calculated on the basis of animal numbers in the underlying simulation model.

We performed an analysis identifying the effect of the abovementioned independent variables on the farmers' decision to trade animals. Since we performed 10 simulated years of farming with each of the participants, and since we were especially interested in the variation within the single decisions of the individuals we added the identity of the farmer (id) as a random effect, while we added all other variables (i.e. rain, vegstate, lsu_ha, account_ha) as fixed effects to a linear mixed effects model (Pinheiro *et al.* 2009). Based on this we formulated the following starting model:

$$lsu\_bs\_ha_{ij} = \alpha + \beta_1 \times rain_{ij} + \beta_2 \times vegstate_{ij} + \beta_3 \times lsu\_ha_{ij} + \beta_3 \times account\_ha_{ij} + a_i + \varepsilon_{ij} \quad \text{eqn 1}$$

with indices i referring to the farmer id (i=1,...,62) and j referring to the simulated years (j=1,...,10). Backwards model selection through stepwise deletion of variables by comparison of model AIC by maximum likelihood estimation was performed (Venables & Ripley 2002). This resulted in the optimal model given by:

$$lsu\_bs\_ha_{ij} = \alpha + \beta_1 \times lsu\_ha_{ij} + \beta_2 \times account\_ha_{ij} + a_i + \varepsilon_{ij} \quad \text{eqn 2}$$

We tested model adequacy with graphical methods (Normality of residuals was examined by plotting theoretical quantiles versus standardized residuals and homogeneity of variance was assessed by plotting residual versus fitted values). Further, we assessed the extent of autocorrelation in the data using an autocorrelation plot of residuals (function ACF in R), which revealed no problematic results. We finally tested for significance of the independent variables by using the likelihood ratio test.

Further, we performed a simple linear regression to test whether the standard deviation of traded animals (bought – sold) in the 10 years depends on the size of the farm, as we expect higher variation between years for smaller farms due to stochastic effects.

3.3 Results

3.3.1 Livestock trading decisions

The results of the applied minimal regression model (see methods) for the trading of livestock showed that only the two capital variables, herd size per hectare and account balance per hectare significantly influenced the farmers' decision. The net trade of animal numbers decreased, that is the role players sold more animals if they had higher livestock density (β=-0.2728, L.Ratio=173.1, df_1, $p<0.001$) and if their account balance was lower (β= 0.000017, L.Ratio=19.0, df_1, $p<0.001$). They did in contrast, not respond to variations in annual precipitation or the state of the vegetation. There was a significant tendency for a relation between the variation in year to year trading numbers and farm size (see Fig. 3.3). This is indicated by the fact that the standard deviation in bought minus sold animal numbers was higher for smaller farms (Fig. 3.3, F=5.27, $df_{1,60}$, p=0.025).

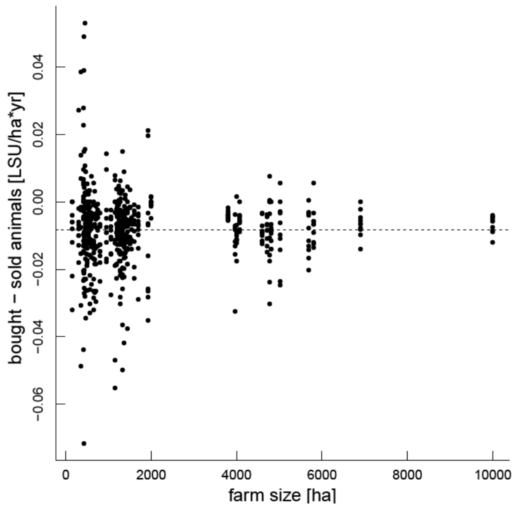

Figure 3.3: Net trade of animal numbers per hectare (bought-sold) in dependence of farm size. Dashed line shows mean trade of all farms.

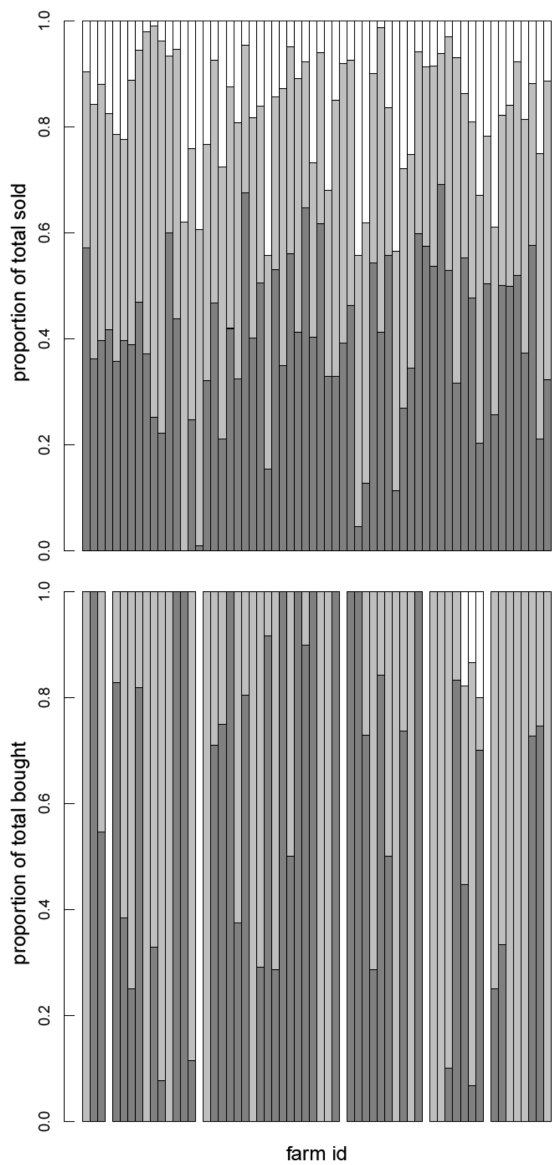

Figure 3.4: Proportions of weaned calves (dark grey), cows (light grey) and oxen (white) of total number of sold (upper diagram) and bought (lower diagram) animals.

3.3.2 Production system

In order to identify whether the players applied a straightforward production system we plotted fractions of different animal categories from sold and bought animals respectively (Fig. 3.4). There is no clear trend in the dataset, some farmers sold high numbers of oxen and almost no weaners and some others rather sold weaners and only very few oxen. A certain amount of cows was regularly sold (24-74%, mean 40%, SD: 0.11) and on average 2.9 times more male weaners have been sold than female ones. This indicates that female weaners were kept for replacement of old cows which then in turn have been sold.

3.3.3 Economic and ecological performance during role plays

In general, more than two thirds of the farmers did reduce their herd size during the course of the experiment, although initial herd sizes were already mostly below the official carrying capacities for the study region (Fig. 3.5). Note, that initial herd sizes of the experiment equal real herd sizes of the surveyed farms. Nevertheless, also two thirds of the farmers had positive total financial revenue after 10 years, though this was very small in most cases (see Fig. 3.6). Generally, farmers were quite successful in keeping their account balanced. In accordance with the low livestock densities, average vegetation condition did improve slightly over time (see Fig. 3.7).

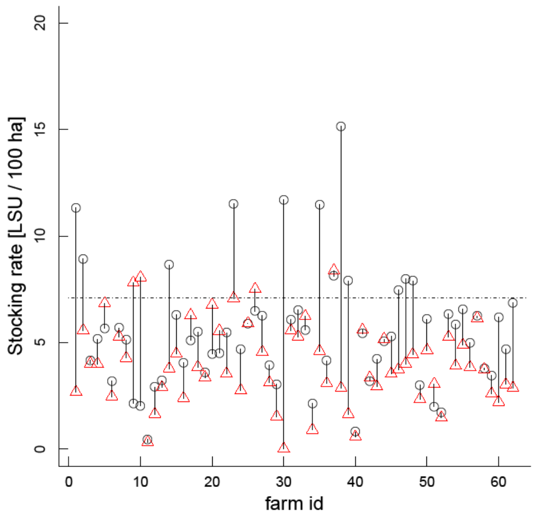

Figure 3.5: circles represent animal densities at the beginning of the experiment (real numbers of participants` farms) and triangles show livestock densities at the end of the experiment. The dashed line depicts the stocking rate recommended by the Namibian ministry of Water Agriculture and Forestry (MAWF).

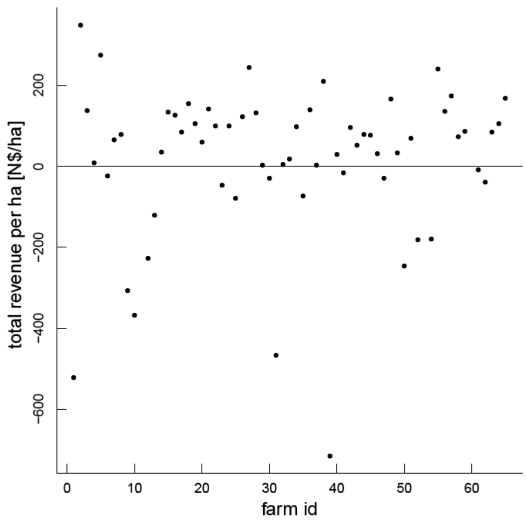

Figure 3.6: Total revenue of the 62 farms per hectare after ten years. Revenue is the difference in the sum of the value of the herd and the account balance between beginning and end of the experiment. Black line delineates zero revenue after ten years.

3.4 Discussion

We successfully parameterized and implemented an ecological-economic model in cooperation with ecologists, economists, experts from a governmental research farm located in the study region and a Namibian NGO (Emerging Commercial Farmers Support Programme). We used this model to assess land reform beneficiaries' (emerging farmers) responses in cattle herd management to different environmental and economic variables in interactive role-plays. The application of a simulation model for this survey enabled an assessment of decision making under realistic conditions: Farmers had to make decisions under, stochastic environmental conditions and resource dynamics and feedbacks between their decisions and these resources. Further they had to deal with realistic herd dynamics as well as costs and financial revenues resulting from their farming decisions. The modelling approach also allowed for an inclusion of local knowledge during model development and parameterization.

According to rangeland theory, we expected farmers to either stock conservatively, i.e. with fixed and rather low livestock densities, or to apply an opportunistic strategy, which aims at making maximum use of peak biomass availability when much grass is available or in good rainfall years. In the first case, one would expect no pronounced response of the land users to environmental conditions, while in the latter case farmers would buy and sell livestock according to biomass availability (in this case represented by rainfall, or vegetation state or both). We found, that the majority of the 62 surveyed emerging farmers from the semi-arid Omaheke region in Namibia applied a conservative management strategy. This is indicated by the fact that we could not identify clear effects of the environmental variations in rainfall or vegetation state on the farmers trading decisions. In contrast, even though vegetation state on average improved during the role-plays, the average livestock density decreased at the same time. Interestingly, our data clearly show that instead of responding to the key environmental variations (rainfall, vegetation state) farmers stabilized their herd size, as indicated by an increasing number of sales with rising animal numbers. This is supporting the conclusion drawn above, that farmers applied a rather conservative strategy, which is determined by the assumption of a fixed capacity of the savanna to support livestock.

In addition, our analyses also revealed that farmers based their trading decision on their financial assets as they sold more animals if they had less money on their virtual account. Accordingly, most of the farmers kept their account balance positive throughout the experiment. This behaviour indicates that the role players were giving high priority to the goal to avoid the risk of (further) debt and high variations in income. Nevertheless, about two thirds of the farmers had positive total revenues after 10 simulated years of range management (including value of money and animals) supporting the assumption that preventing indebtedness is one of their major goals.

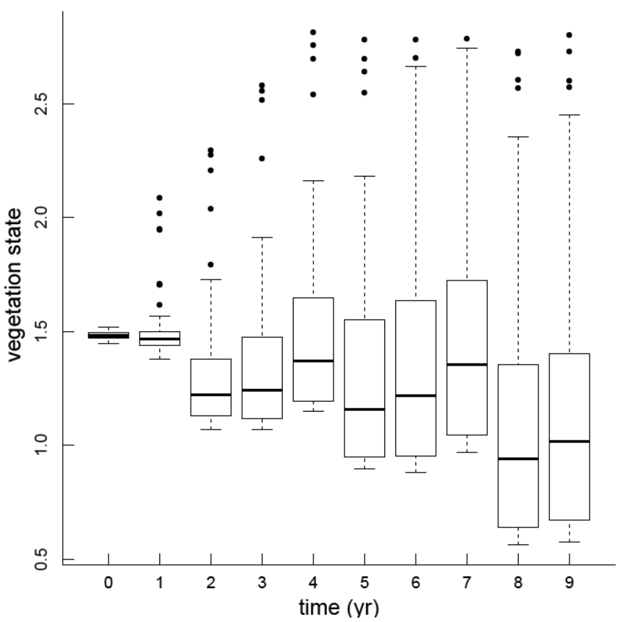

Figure 3.7: Vegetation state over time. At t=0 the 62 farms were initialized with half of the cells in state 1 and state 2 respectively. State 0 exhibits highest and state 3 lowest cover of perennial grass. Due to the stochastic nature of the vegetation sub-model, vegetation composition became naturally more variable over time.

This finding is explicitly important in the context of many players real life situation, where they mostly struggle to make a significant or even any positive profit on their farms and often also have high debts (see data on income of our sample farms in Falk *et al.* 2010, see data on other farms of the region in Werner and Odendaal 2010). The indebtedness is mainly a problem for farmers who received subsidized loans according to the Affirmative Action Loan Scheme (Adams 1993; Falk *et al.* 2010). One further important reason for economic problems of the farmers could be found in the small size of the farms allotted under the Farm Unit Resettlement Scheme instrument (average size of FURS farms 864 ha; min 50 ha, max 2000 ha). Such farms allow for only small or none net revenues as costs are high in relation to gross income and farming businesses that are this small will always be at risk to fail (Tomlinson, Hearne & Alexander 2002; Werner & Odendaal 2010). On farms with sizes below the official minimum for the region of 1000 ha with a recommended stocking rate of about 8 LSU per

100 ha a farmer can hardly afford a stable reproductive herd (Werner & Odendaal 2010). Natural stochastic variations in the fertility of individual cows will have high impacts on farm revenues but also on ecological impacts of the farming. This is also visible in our sample, where farmers with very small farms produced very variable outcomes regarding total revenue or herd size per ha.

Knowing that farmers tended to apply a conservative management strategy, we tried to identify the production system in more depth. As expected for a non-opportunistic management, none of the farmers applied a speculation farming strategy, indicated by low numbers of bought weaners. Instead, most of the farmers, though not all, followed no clear but rather a mixed strategy of weaner and oxen production (see Fig 3.4). Both production strategies are common in the study region (Buss 2006; Olbrich 2011). In a so called weaner production system, farmers sell all male calves (and those female calves that are not needed to sustain the herd) after weaning. In the oxen production system, these calves are "finished" and sold as oxen for slaughter. Producing oxen adds much value to the product, but strongly increases the number of non-reproductive animals that need to be sustained by the farms fodder resources. However, though mixing the two strategies is not economically efficient, it to some extent indicates opportunistic behaviour: farmers eventually keep weaners to produce oxen if financially or ecologically possible (or opportune), but they sell animals already as weaners if necessary to cover incurring costs. Especially the latter, i.e. the selling of weaners due to financial constraints was also reported in Werner and Odendaal (2010) who also surveyed AALS and FURS farms in the Omaheke region. Both, oxen and weaner production are suitable for rather non-opportunistic management strategies as the relative number of reproductive cows in a herd needs to be high and stable, to sustain a constant production of weaners or oxen for sale (Buss 2006; Olbrich 2011). However, oxen production is less flexible, since male calves need about three years until they are ready for slaughter in an extensive rangeland system and losses (opportunity costs) are high if they are sold before time.

The finding that land reform beneficiaries stock conservatively is to some extent surprising as it is often assumed, that land reform contributes to land degradation as a consequence of overgrazing (Hunter 2004). Due to their background of mainly communal or subsistence farming and due to the fact that only about 20% of the farmers in our sample stated to have had some kind of farming training (Falk *et al.* 2010) one would eventually expect strategies of either herd maximization or short term opportunistic tracking of resources (Hunter 2004; Hahn *et al.* 2005; Allsopp *et al.* 2007). This was clearly not the case. In fact, their management behaviour of stocking conservatively with relatively constant livestock densities is in accordance with a large body of literature recommending such management for commercial livestock production in semi-arid to arid savanna rangelands (Illius & O'Connor 2000; Cowling 2000; Higgins *et al.* 2007; Quaas *et al.* 2007; Muller, Frank & Wissel 2007). The main arguments for such conservative strategies are of both, ecological and economic nature. On the one hand, from an ecological perspective, conservative and stable stocking rates allow for recovery of grasses during years with above average rainfall (Muller, Frank & Wissel 2007; e.g.

Buitenwerf, Swemmer & Peel 2011). Further, the excess standing grass biomass can serve as buffer in years of drought (Higgins *et al.* 2007; Quaas *et al.* 2007; Muller, Frank & Wissel 2007). Finally, grass biomass is to some extent necessary to fuel fires, which can be very important for rangeland management (Higgins *et al.* 2007). On the other hand, also more economic arguments indicate a non-opportunistic management of semi-arid rangelands. Amongst the most important arguments are, risk aversion of farmers, the preference of a stable income and the high transaction costs due to high livestock turnover rates occurring in unfavourable sequences (i.e. highly variable) of rainfall conditions (Illius & O'Connor 2000; Higgins *et al.* 2007; Quaas *et al.* 2007; Muller, Frank & Wissel 2007).

However, the reason for the application of a conservative management strategy can likely be found in the real life economic situation of the surveyed Namibian land reform beneficiaries and the corresponding willingness and capability to take the risks associated with variable income and indebtedness. It needs to be mentioned though, that from our study we cannot derive whether farmers would manage the land differently (i.e. opportunistically or with higher stocking rates) if they had more assets.

Conclusion

Our study provides evidence that under the given circumstances, emerging farmers will likely not contribute to the vast phenomenon of rangeland degradation as their de facto management behaviour is ecologically sustainable and does not follow the undesirable pattern that can be found in some communal, but also in commercial rangelands of the study region. However, this effect is likely the result of extreme financial pressure experienced by land reform beneficiaries (Falk et al 2010, Falk et al submitted). This pressure is exhibited by small farm sizes (FURS) or high indebtedness (AALS) depending on the applied land reform measure. This in turn puts the success of these land reform programmes, especially with regard to their economic goals in general at risk (compare with Adams 1993).

Adaptations of the Namibian land reform program are needed in order to enable land reform beneficiaries to farm ecologically and economically viable and not mainly according to the fear of financial failure (Orenstein, Jiang & Hamburg 2011). Additional capacity development efforts need to be backed up by measures of giving beneficiaries more financial flexibility, allowing farmers to decide for straightforward (instead of mixed) production systems and management strategies that enable maximum revenues (Werner & Odendaal 2010). In a first step, this could be achieved by prolonging the repayment plan of AALS loans. Further more attempts could be made to help emerging farmers (especially those being part of FURS) in establishing cooperative infrastructure and rangeland management practices, which could enable them to accomplish economies of scale similar to the ones accomplished on large commercial farms (Falk, Lohmann & Azebaze ; Tomlinson, Hearne & Alexander 2002; Werner & Odendaal 2010).

In the context of financial pressure, we question more fundamentally, whether the redistributive land reform in Namibia as it is currently implemented is an adequate instrument for poverty alleviation (see also Adams 1993; Werner & Odendaal 2010; Orenstein, Jiang & Hamburg 2011). It needs to be understood, that giving land to the poor without strong additional and long term material and training support, can turn out to be economically rather a burden than a gift for them. Further, perspectives other than the commercial livestock production, like e.g. residential purposes, subsistence food production, small-scale irrigation or large-scale game farming should eventually be taken into account in future planning and implementation of possible alternative land reform instruments (Cousins *et al.* 2007; Werner & Odendaal 2010).

3.5 References

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3.6 Appendix 3.A – protocol of role plays

This document describes the overall process of conducting role plays with land reform beneficiaries on the basis of the ecological-economic simulation model. The role plays were conducted between January 2009 and April 2009. Instructions given to the players are formatted in normal fond. General information and instruction given to the experimenter and not meant to be communicated to the players are formatted in italics.

Selection of farms and recruitment of participants

The research team cooperated with the Emerging Commercial Farmers Support Program (ECFSP). The program supports land reform beneficiaries by providing trainings and mentoring as well as extension services. In order to reduce the costs of logistics for our study it was jointly decided to focus on the Omaheke region in central east Namibia (see Fig. 3.1 in main text of chapter 3).

We focus our analyses on Namibia's two main land reform instruments, namely the Farm Unit Resettlement Scheme (FURS) and the Affirmative Action Loan Scheme (AALS). The FURS is administered by the Ministry of Lands and Resettlement. It targets poor and landless Namibians by redistributing land on state-acquired commercial farms. This acquisition is based on the preferential right of the Namibian state to purchase agricultural land whenever any owner of such land intends to dispose of it (Republic of Namibia, 1995). Any Namibian citizen who has been socially, economically or educationally disadvantaged by past discriminatory laws can apply for an allotment of land acquired for resettlement (Republic of Namibia, 2002). Successful applicants are supposed to sign a 99-year lease agreement with the government. The Affirmative Action Loan Scheme (AALS) is implemented by the parastatal Agricultural Bank of Namibia on behalf of the Ministry of Agriculture, Water and Forestry in collaboration with other Ministries. It assists so-called emerging commercial farmers to purchase commercial farms through subsidised interest rates and loan guarantees by the state.

ECFSP provided the research team with lists of land reform projects in the Omaheke region. On the list were 196 FURS farm units and 108 AALS farms. Out of these 45 FURS farmers (23 percent) were included in the study and 15 AALS farms (14 percent). We more strongly focused in FURS farms in order to be able to better study the cooperation challenges associated with this land reform instrument.

The project collected contact details of farmers through key stakeholders such as farmers associations, individual farmers and extension officers. In addition, many farms have been visited personally in order to make appointments with the land reform beneficiary. The selection of participants was not random but predetermined by the accessibility of beneficiaries and their willingness to participate. One challenge has been that many farmers are visiting their farms only very irregularly while employed farm managers keep the farm business running. Since we only included the actual land reform beneficiaries in our study it was difficult to meet the land holders. The team arranged, however, also *game sessions with part time farmers in urban centers in order to avoid a too strong structural bias.*

Location of experiment to be conducted

The role plays as well as the interviews where mainly conducted on the farms of beneficiaries. The exceptions are sessions with some of the part time farmers who were interviewed at their private homes in urban centers.

Compensation for participation

The participants of the game have not been compensated for their participation. The game was largely perceived to be a free training.

Introduction

This is a research exercise conducted as part of the BIOTA project. BIOTA is a large research study financed by the German Ministry for Research and Education. It has the objective to learn about how people benefit from natural resources and how their resource use affects the state of the resources. In this particular study the project is interested in the way how farmers use their land and how this effects the vegetation. For this purpose we will ask you some questions and will play a game with you. The interview will take approximately 90 minutes and the game probably 120 minutes. The game is supposed to be an entertaining way to play with farming decisions and natural resource management. It will help us to understand how farmers manage their pastures and water infrastructure. Any data collected during the course of this game would only be used confidentially for research purposes; no part of this data would be available to any external person. Please listen to the instructions very carefully and ask questions at any point.

The game design

As part of our research, many scientists in the BIOTA project studied the ecology of the veld here in the Omaheke region and how it is affected by grazing. With all this information we developed a computer program which simulates how grasses and bushes grow over the years depending for instance on the previous state of the veld, the rain, and the number of animals grazing on the land. The computer program can further tell us how many animals the land can support and how the state of the livestock is likely to change if there is too little fodder on the veld.

The game we are going to play is based on this computer program. In the game you will be given a farm of exactly the size of your farm in real life. You told us earlier how many animals you posses and you will start the game with exactly this number of animals. This means that your farm in the game is at the beginning of the game stocked at the rate your farm is stocked at in real life.

Game rounds

For simplification, the game is played in 20 rounds. You can make decisions in April and September meaning that you make two decisions per year and we play over a period of 10 years.

Veld conditions

During the farming game the condition of your veld can change. At the beginning your veld condition will be moderately good, equivalent to the pictures we show you. Depending on the rainfall and your management the veld condition can change. We assume that the veld is resilient up to a certain point but changes abruptly once a certain threshold is reached. We assume that the veld of your whole area does not change simultaneously. We will tell you at the beginning of each round which percentage of your veld is in which condition. We will do this by showing you picture.

Please show the players the pictures of the vegetation states.

In addition to the veld condition, we will also report you in every experiment period how much rain in mm has been falling in the previous season/round.

Herd dynamics and state of livestock

In the game your livestock give birth, grow old, and can die. The computer program distinguishes between age classes (calves, weaner, heifer, cows/oxen) and sex of your animals. Your livestock can be in good and bad condition. We distinguish between five classes of condition. Depending on the state of your veld the condition of your livestock can change. Take into account the condition faster worsens than it improves. At the beginning of each game round we will show you the state of your livestock using the pictures.

Healthier livestock has better reproduction rates and the computer program considers this. The lower the state of your livestock the less calves you will get.

In case there is insufficient fodder on your veld due to poor rain or your management, livestock can die. Livestock will automatically be culled by the program if it grows older than 10 years. If old animals are culled you get a minimal price of N$ 700 for it.

At the beginning of each game round we will give you a list showing your herd composition. This includes: Calves, female weaners, male weaners, heifers, cows, cows with calf, pregnant cows, oxen and bulls as well as the total number of animals.

In addition, we will let you know in each game round whether you faced livestock losses for the follow reasons: Starved animals (pasture could not feed them), culled due to old age, died due to other reasons (random).

Selling and buying of livestock

You can sell or buy livestock in the game. Livestock prices are the approximate prices of what you currently get at Meatco. For animals in very good condition in the April round you get the following prices in the game:

Weaner:	N$ 2500
Heifer	N$ 3500
Oxen	N$ 4000
Cow	N$ 4500

Animals in worse conditions get lower prices. In order to get the prices for animals in lower states you have to multiply the prices by the following factors:

State 4: 1.0

State 3: 0.8

State 2: 0.65

State 1: 0.5

Sate 0: 0.3

In addition, observing the market prices over the last years we can see that the prices vary between April and September. Usually more people like to sell in April at the end of the rainy season than in September short before the rain starts again. Therefore you get a seven percent higher price in September.

Throughout the experiment you can also buy livestock. After talking to farmers and experts we assume that the prices for buying livestock are higher than for selling them. The computer program assumes that buying prices are ten percent above selling prices.

But you do not have to calculate the prices all the time. We will provide you with price tables and tell you which one applies to you in each round.

Your individual account

In the game you will be given an individual bank account. Any money you earn by selling livestock will be saved on this account. From this account you also can buy livestock. You can run the account into minus. On any positive balance of your account you will get for each game period an interest of 2 percent. On any negative balance you will have to pay an interest of 15 percent. At the beginning of each game period we will report you the balance of your bank account in that game period.

Farm expenses

As on your real farm you will have expenses on your farm. We do not want to make the game too complicated. Therefore you will not have to decide on each of the small expenses you usually do.

For players who are resettlement farmers (FURS):

Based on research of the ECFSP and various consultants we assume that you have costs of 15 N$ per ha which are therefore constant for your farm over the game and 75 N$ per head of livestock. The costs per head of livestock will vary depending on how much livestock you own in a game period.

For players who are Affirmative Action Loan Scheme farmers (AALS):

Based on research of the ECFSP and various consultants we assume that you have costs of 15 N$ per ha which are therefore constant for your farm over the game and 75 N$ per head of livestock. The costs per head of livestock will vary depending on how much livestock you own in a game period.

In addition, we consider your approximate real life expenses on water and labour in the experiment. You will tell us at the beginning how much wind and diesel pumps you have on your farm. Based on this information you will be charged in the game every year the following amounts which are typical average maintenance costs for this type of pump:

Per wind pump: N$ 750

Per diesel pump: N$ 2,350

We further ask you how many labourers you employ and how much you pay them. In every game period this amount will be deducted from you account.

All expenses will be automatically covered from your individual game account. At the beginning of each game period we will report you the expenses you had in the previous game period.

Only for players who are resettlement farmers: Maintenance of water infrastructure

In addition to the general annual expenses you have to pay, we are interested in one particular issue which we have commonly found on land reform farms. Very often the government buys big farms and splits them into farm units which are allotted to the applicants. The big farms usually had a centralised water infrastructure with few pumps and pipes. As a result not all land reform beneficiaries have access to an own pump or dam and the group of beneficiaries sharing the former big farm have to maintain the infrastructure jointly. We are interested in how you are doing this and simulate the situation in our game.

We play the game with a group of farmers who share the infrastructure also in real life. Imagine that you as a group have a wind and a diesel pump on your farm. As you know, the pumps need regular maintenance and there is always the risk that something breaks and needs to be repaired. You never know, however, how much you have to pay in a particular year as the maintenance cost may vary. Your group will have to pay a random amount which is based on typical maintenance costs of pumps. This amount will be automatically deducted from a group account. Each of you has to decide individually each September how much she wants to pay into the group account from her individual game account. Be aware that you do not know in advance how high the costs will be in the particular game round.

In the case that the money available in the fund is insufficient to cover the maintenance costs, the infrastructure breaks down. In this case you have to take your cattle to the neighbouring farm where you have to pay, however, for getting access to water. In the game you have to pay a fee of N$ 50 per head of cattle which will be automatically deducted from you individual game account. If there is more money in the group account than needed to cover the maintenance costs, the amount will remain there and can be used to cover water costs for upcoming game periods.

At the beginning of each April game period we will report you the amount which had to be paid in the previous September game period. At the beginning of each game period we will report you the amount available in your group's water account.

Trial rounds

In the case of playing with a group of resettlement farmers the players are seated in a way that they can see each other and have the opportunity to communicate with each other in case they feel like doing this. The players are supposed to have everything they need (paper, pen, space) to make personal notes in front of them.

Before we start with the actual game, we would like you to get used to the procedures. Therefore we would like to play some trial rounds with you. Can you please give us the following information:

Resettlement farmers: Size of your farm and number of livestock you own:

Affirmative Action Loan Scheme farmers: Size of your farm, number of livestock you own, number of wind pumps on your farm, number of diesel pumps on your farm, number of workers employed on your farm, average annual salary of your workers.

In which language would you like us to give you the game information?

The experimenter gives the player an ID and enters this information into the initialization interface of the model and click on the "initialize button" to initialize the mode variables. The model is now ready for the role-play to begin. The experimenter prints a list with the information given to the players at the beginning of the experiment. The list can be printed in English, Afrikaans or Otjiherero depending on the preference of the player.

We are now giving you a list with information about your individual account, your costs in the first game period, the rainfall during the first game period and your herd composition. The game starts in the month April.

The experimenter explains the sheet step by step to the players.

In addition to this information on the list we will show you pictures of the vegetation of your farm as well as of the body condition of your livestock.

All state variables like the state of the vegetation of the farm as well as the body score of the player's livestock, animal numbers and age distribution of the herd are visible for the experimenter in the model interface. She shows the farmer the respective pictures of the vegetation state and the livestock condition score.

Now we are giving you a form. In this form you can write in the first column how much livestock you want to sell or buy. In addition, we give you the list of the livestock prices.

The experimenter gives the player only the particular price list for the livestock condition of the player (at the beginning of the game the condition is always 3/good).

The experimenter gives the players the form for entering their decisions.

Please write into the form how much livestock you want to sell or buy!

The experimenter assists the players and repeats individually whatever is unclear.

The experimenter collects the forms after all players have made their decisions. She than enters the decisions of the players into the model interface. By clicking "next time step" the model proceeds and calculates the new states of all variables. The new states are printed again and given to the players. The new states of the vegetation as well as the livestock condition score are shown to players with the respective pictures.

Now that you have made your first decision we will see how the state of your farm has changed in the game. Half a year has passed in the game and you are now in the month September. You can see how your account has changed, which costs you have in the second game period, how your herd is composed as well as whether you faced any livestock losses.

Based on this information, please fill into your form how much of which kind of livestock you want to sell or buy.

The experimenter gives the player only the particular price list for the livestock condition of the player in this round. The experimenter assists the players and repeats individually whenever something is unclear.

For resettlement farmers only:

Remember that your group has to jointly maintain the common farm water infrastructure. Your group will have to pay a random amount for the maintenance after each September round. This amount will be automatically deducted from a group account. Each of you has to decide now individually how much she wants to pay into the group account from her individual game account. Be aware that you do not know in advance how high the costs are going to be.

In the case that the money available in the fund is insufficient to cover the maintenance costs, the infrastructure breaks down. In this case you have to take your cattle to the neighbouring farm where you have to pay, however, for getting access to water. In the game you have to pay a fee of N$ 50 per head of cattle which will be automatically deducted from you individual game account. If there is more money in the group account than needed to cover the maintenance costs, the amount will remain there and can be used to cover water costs for upcoming game periods.

As you see on your information sheet, the balance in your group account is still zero. From now on the list will always show you whether there is excess money in your account which you can use to cover future repairs.

Now please enter the amount you want to pay to the group's water fund into your form.

The experimenter assists the players and repeats individually whatever is unclear.

The experimenter collects the forms after all players have made their decisions. She then enters the selling and buying decisions of the players into the model interface. She enters the water maintenance decision into the model interface. She then clicks the button "next time step" and the model calculates the new states of all variables. The new states are printed again. The new print outs are given to the players again. The new states of the vegetation as well as the livestock condition score are shown to players with the respective pictures.

The experimenter continues the abovementioned steps until she has the impression that all players feel confident with the game procedures. Then the actual game starts.

The actual game

The players are seated in the way as for the trial rounds.

Now we will play the same game for 10 game years.

For the actual game the experimenter initializes a new session again with the initialization information given before.

The experimenter gives the player an ID and enters this information into the initialization interface of the model and click on the "initialize button" to initialize the mode variables. The model is now ready for the role-play to begin. The experimenter prints a list with the information given to the players at the beginning of the experiment. The list can be printed in English, Afrikaans or Otjiherero depending on the preference of the player.

Based on what is reported in the model interface the experimenter shows the players the pictures illustrating the state of the vegetation on their virtual farm as well as the livestock condition score.

Here are now the lists for the first April game period of the real game. In addition, these pictures show the state of the veld on your farm and the condition of your livestock.

New empty forms for buy/sell decisions are given to the players.

Please write into the form how much livestock you want to sell or buy!

The experimenter collects the forms after all players have made their decisions. She then enters the decisions of the players into the model interface and clicks the button "next time step" and the model calculates the new states of all variables. The new states are printed again and print outs are given to the players. Based on what is reported in the model interface the experimenter shows the players the pictures illustrating the state of the vegetation on their virtual farm as well as the livestock condition score.

Now that you have made your first decision we will see how the state of your farm has changed in the game. Half a year has passed in the game and you are now in the month September. Here is the list with the information for the September game period of year 1. In addition, you can see on these pictures in which state the veld of your farm is now as well as the condition of your livestock.

Please fill into your form how much of which kind of livestock you want to sell or buy.

The experimenter gives the player the particular price list for the livestock condition of the player in this round.

For resettlement farmers only:

Please enter into your form as well how much you pay into the group's water fund!

The experimenter collects the forms after all players have made their decisions. She than enters the selling and buying decisions of the players into the model interface. She enters the water maintenance decision into the model interface. She than presses the button "next time step" and the model calculates the new states of all variables. The new states are printed again. The new print outs are given to the players again. Based on what is reported in the model interface the experimenter shows the players the pictures illustrating the state of the vegetation on their virtual farm as well as the livestock condition score.

Now the experimenter repeats the role play for 20 game periods simulating 10 years.

After the game

After the game is finished time series figures for key variables of the game (account, vegetation state, livestock came and condition score, livestock numbers) are printed and *handed out to each player. They can discuss the game and the figures with each other as well as the experimenter.*

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3.7 Appendix 3.B – Up scaling and simulation of vegetation dynamics

Landscape scale vegetation dynamics were simulated using the state-and-transition approach (e.g. Westoby *et al.*, 1989; Popp *et al.*, 2009). The underlying transition probabilities were derived from vegetation dynamics simulated with the eco-hydrological model *EcoHyd* (Tietjen *et al.*, 2010). Following the idea of the state-and-transition concept, we first defined ecological states between which the semi-arid savannah can switch and then defined probabilities for the transition between these states.

The definition of the vegetation states was exclusively based on perennial grass and not on woody vegetation cover or annual grasses for several reasons. First, perennial grasses are the vegetation type providing the desired fodder biomass for livestock and are thus most important from a land user perspective. Second, perennial grasses respond quickly to changes in seasonal precipitation and livestock densities, while woody vegetation rather responds on the scale of decades (see Lohmann *et al.*, 2012) and annuals rather respond directly to the presence or absence of the two other types (see e.g. Lohmann *et al.*, 2012). Consequently, as we want to simulate only 10 years of land use in the role-plays an inclusion of woody vegetation is not necessary. Further, a simple description of the ecological states is important for a clear communication of the model state to the participants during the role plays in the field. We defined four vegetation states and five classes of precipitation which are given in Tables 1 and 2.

Table 3.B.1: Definition of vegetation states based

Perennial grass cover (%)	Vegetation state
>40	0 (very good)
20-40	1 (good)
10-20	2 (moderate)
<10	3 (bad)

Table 3.B.2: definition of rainfall classes

Rainfall (mm)	Class
>500	4
400-500	3
300-400	2
200-300	1
<200	0

We conducted 100 repeated simulations of 100 years for each of 13 different grazing intensities (2-26 ha LSU^{-1}). Every single simulation had a unique stochastic time series of precipitation (Tietjen *et al.*, 2010; Lohmann *et al.*, 2012). In every time step, the change of the average cover of perennial grasses (on the simulated 2.25 ha grid), the respective seasonal amount of precipitation and the biomass of annual and perennial grasses was recorded. This resulted in 65 transition probability matrices (for 13 different grazing intensities and 5 classes of seasonal rainfall intensity) and 4 regressions (one per vegetation state) for both, annual and perennial grass biomass production as a function of rainfall.

Finally, vegetation dynamics and biomass production are stochastically simulated with the state-and-transition approach for a grid of variable size (depending on the interactive initialization at the beginning of the role-play) with a cell size of 1ha in annual time steps.

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3.8 Appendix 3.C – model rules

In the following we describe the processes and model rules according to the flow chart given in Fig. 3.2 of the main text of chapter 3. All model rules have been discussed with members of the Namibian Emerging Commercial Farmers Support Programme (ECFSP) and rangeland experts from the MAWF research station at Sandveld in several feedback sessions and model test runs. Rules are based on the recommendations of the joint presidency committee (JPC) of the Namibia National Farmers Union (NNFU) and the Namibia Agricultural Union (NAU) who developed management guidelines for the ECFSP (Stehn, 2008a; Stehn, 2008b).

The model implementation allows for interactive decisions regarding cooperation in water infrastructure maintenance between several farmers. During the role-plays participants were able to also contribute to cooperative funds. This however, is not in the scope of the study described here, but explained and analysed in detail elsewhere (Falk *et al.*, Appendix A).

Initialisation

At the beginning of every role-play (simulation) few parameters and the state variables of the model need to be initialized. Therefore, every player needs to input the type of farm with regard to the land reform measure, the size of the farm, the number of animals kept, the number and type of water pumps run on the farm, the number of farm workers employed and the salary paid for them. After the input of this basic information the composition of the cattle herd and the state of the vegetation (see table 3.B.2) are initialized. We assumed a moderate to good condition of the on-farm vegetation and randomly set the vegetation state in each cell to state 1 or 2 with an equal probability. The animal herd was composed assuming a good management resulting in a herd of mainly young animals in good condition (see initial age class distribution in table 3.C.1). Further we assumed the herd to consist of mainly fertile adults (56% cows and 4% bulls), 10% oxen, 10% heifers and 20% weaned calves (weaners). This composition represents a fertile breeding herd and allows for a quick adaptation of the management to either a cow/oxen or weaner production strategy.

Table 3.C.1: Age class definition and age dependent parameter values

Age class	Age [years]	Cows [%] in initial herd	Oxen[%] in initial herd	Price coefficient
1	3-5	25	20	1.0
2	6	20	20	0.95
3	7	15	20	0.8
4	8	15	20	0.7
5	9	10	20	0.6
6	10	8	0	0.4
7	11	5	0	0.4
8	12	2	0	0.4

Biomass Production

Based on the current vegetation state (see Appendix 3.B) and the rainfall of the current season the model calculates the biomass production of perennial and annual grasses for all cells according to the linear relation derived in the up scaling process (see Appendix 3.B).

Number of bulls

In order to reduce complexity of the role-plays the number of bulls is automatically adapted on an annual basis, assuming that always at least 1 bull is available for 25 cows (as recommended in Stehn, 2008a). Bulls are taken from male weaned calves (weaners).

Animal condition score and starvation

We categorized the animals' condition from "very lean" to "fat" according to a scoring system that was developed by the Emerging Commercial Farmers Support Program of Namibia (ECFSP). The score of an animal relates to its nutritional condition and more specifically to its body mass (Table 3.C.2).

Table 3.C.2: Condition score of animals

Condition score	Body mass [kg]
0	<270
1	270-320
2	320-380
3	380-440
4	>440

Condition of animals was calculated for an "average individual" based on the number of large stock units (LSU) (Meissner, 1982), and the available fodder on a farm irrespective of the current composition of the herd. Hereby we calculate an average daily weight gain for the three periods of the year that are given by the model structure (see Fig. 3.2 in main text) since animal numbers and thus fodder availability might change during the course of the year. Animals feed with a daily intake rate (r_{in}) of up to 5% ($r_{in,max}$) of their body mass depending on fodder availability (Tainton, 1999). Hereby we assume the intake rate r_{in} during the respective part of the year (with d_y days) to be as high as possible according to the available grass biomass gb_{avail} (biomass produced minus biomass that was already eaten). Accordingly, the maximum value of $r_{in} < r_{in,max}$ is chosen so that:

$$gb_{avail} \geq d_y \times r_{in} \times w_{LSU} \times N_{LSU} + d_{left} \times 0.01 \times w_{LSU} \times N_{LSU} \qquad \text{eqn C.1}$$

with w_{LSU}, the mean weight per LSU (450kg) and N_{LSU} the current number of animals in LSU. For example, during the first 120 days of a year (late rainy season from Jan-April) the animals will feed with a rate of r_{in} if biomass availability enables at least a rate of 1% of their body mass for the rest of the year (245 days from May to December). By this, we achieve the typical seasonal changes in body mass that the animals undergo (Tainton, 1999) and simulate body score dynamics that are part of the output that is given to the farmers during the role play.

After the intake rate was determined according to eqn. 1 the daily weight gain of the animals and thus the current average condition score of an individual can be calculated. If the daily intake rate (r_{in}) lies above the minimum daily intake that enables weight gain($r_{in,lim}$) the daily weight gain is:

$$Dailygain = ((r_{in} - r_{in,lim})/(r_{in,max} - r_{in,lim})) * gr * bm * (bm_{max}-bm)/ bm_{max} \qquad \text{eqn C.2}$$

with, bm the biomass of an average individual, gr the potential growth rate of an animal and bm_{max} , the maximum body mass of an individual.

For rates below $r_{in,lim}$ animals start losing weight according to a linear relation leading to a daily weight loss of max 0.2 kg per day if the available fodder biomass only allows for an intake rate of 1% or less.

Animals are assumed to starve whenever the average body mass is less than 220 kg. In times of very scarce fodder resources the model will reduce animal numbers so that surviving animals will at least have an average body mass of 220 kg. Note, that this is an extreme case which will only occur under very extreme conditions.

Reproduction, Birth of calves and weaning

We assume one single breeding season per year (early winter breeding season). Consequently calves are born in the second half of the rainy season. Weaning takes places after 9 months at the end of the dry season/beginning of the rainy season. Conception and weaning are stochastic and depend on probabilities which in turn depend on the animals' condition score and are given in table 3.C.3.

Mortality

Every individual animal experiences stochastic mortality on an annual basis. The probability to die depends on the condition score of the herd and is given in table 3.C.3.

Vegetation dynamics

At the end of a year the average size of the herd that was feeding on the farm during the year is used to calculate the changes in vegetation state for the next time step. Together with the total seasonal precipitation, the stocking rate can be used to identify the respective transition probability matrix from the matrices produced during the up-scaling procedure (see above). Then, the transition of the vegetation in every cell of the vegetation grid is drawn by chance.

Table 3.C.3: Parameters depending on animal condition score

Parameter	Condition score				
	0	1	2	3	4
Mortality rate	0.07	0.05	0.04	0.04	0.035
Pregnancy rate	0.1	0.25	0.4	0.6	0.65
Weaning rate	0.5	0.6	0.7	0.8	0.9
Price coefficient	0.3	0.5	0.65	0.8	1.0

Calculation of costs

According to interviews with experts and farmers from the ECFSP and the Sandveld research station as well as data from both sources we derived all-inclusive costs of the farming business. The costs are split in two categories: Fixed costs, which are calculated on the basis of farm size and variable costs that are related to the size of the herd. Hereby, a minimum amount of costs is fixed and will always incur, irrespective of the herd size (i.e. costs corresponding to a herd size of 25 LSU). In addition to these costs, we calculate the expenses for water infrastructure maintenance and labour according to the respective on-farm setting. Half of the costs have to be covered at the end of rainy season and the other half after the second time-step, at the end of the dry season (see Fig. 3.2 in the main text). In this way, the participants of the role-play will know the costs when they are requested to make their livestock trading decisions. Costs are automatically deducted from the individual's bank account.

Bank account

Each player has an individual bank account in the game. We calculate credit and debit interest at annual rates of 15% and 2% respectively. At the beginning of the game the account balance is zero.

Table 3.C.4: Model parameter values

Parameter description [name]	Value	Unit
intake threshold growth [$r_{in,lim}$]	0.038	-
max. ind. bodymass [bm_{max}]	470	kg
max intake rate [$r_{in,max}$]	0.05	-
growth rate of animal [gr]	0.004	-
factor weaner/heifer LSU	0.6	-
factor cow/ox LSU	0.9	-
lateBSfac	0.5	-
price for culled cow/oxen	700	N$
price for selling heifer	3000	N$
price for selling weaner	2000	N$
price for selling heifer cow	4000	N$
price for selling ox	3500	N$
factor for dry season price	1.07	-
factor for purchase price	1.10	-
fixed farming costs	15	$N\$\cdot ha^{-1}$
variable farming costs	75	$N\$\cdot LSU^{-1}$

Trading of livestock

At two stages during the simulation of one year the role-players have the opportunity to interact with the model. The participant is requested to make a decision what amount of what kind of animals to buy or sell. The categories of animals are oxen, cows, heifers and male and female weaners (weaned calves). Animal prices are depending on the age and condition of an animal (see tables 3.C.1 & 3.C.3). Selling prices are below purchase prices and prices in September differ from prices in April (see table 3.C.4). Whenever animals are sold, older animals and cows that are not pregnant and have no calf with them are preferentially sold. Prices have been defined according to the 2008 prices[2] at the livestock auction in Gobabis, the capital of the Omaheke region. We assumed constant prices for the duration of the simulation experiment. Revenues from livestock sales are automatically saved on the individual's bank account and expenses for livestock purchases deducted from it.

[2] 2008 livestock prices have been derived from the internet database oft AGRA Namibia co-operative Ltd. at http://www.agra.com.na

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Chapter 4

General Discussion

In my PhD thesis I successfully applied two modelling approaches to assess options and conditions for the sustainable management of semi-arid savannas. First, I amended and applied an eco-hydrological model to improve the understanding of ecosystem dynamics under land use. I was particularly interested in how different land use practices affected vegetation cover and degradation dynamics under climate change and whether controlled fires could be applied as a management option to prevent bush encroachment and to increase rangeland productivity. Second, I developed an ecological-economic model which simulates farm-scale processes. This model was applied in the field to conduct role plays with Namibian land reform beneficiaries, to investigate the influence of environmental and economic variables on their rangeland management decisions.

In the initial section of this general discussion (4.1), I will discuss the results of my thesis with regard to the general understanding of semi-arid savanna vegetation dynamics under livestock grazing. First, I will recapture the implications of my simulations for the general response of a semi-arid savanna to livestock grazing (4.1.1). Second, I will discuss the differences between my key findings and opposed views in literature regarding the effects of climate change and CO_2 increase (4.1.2) and fire (4.1.3) on the response of semi-arid savannas to land use. Third, concluding from the previous discussion, I will emphasize the importance of the pronounced recruitment bottleneck of woody plants for semi-arid savanna vegetation dynamics (4.1.4).

The second section (4.2) deals with the Namibian land reform. First, I will discuss the results of the role plays for rangeland management and especially for land reform policies (4.2.1). Second, I will discuss the applied methodology of simulation-based role plays (4.2.2). Note, that in chapter 4.2.1 I will introduce additional key findings resulting from the role plays which have so far not been presented in this thesis but are presented within another study (Appendix A). These findings are based on the ecological-economic model and its application as presented here and provide important aspects for the final discussion of the findings of this study (see structure of this thesis in chapter 0.3).

Finally, in the third section (4.3) I draw general conclusions from the overall findings of my thesis for sustainable management of semi-arid savannas.

4.1 Semi-arid savanna vegetation dynamics

4.1.1 Basic response of the semi-arid savanna vegetation to climate and grazing

The results presented in chapters 1 and 2 showed that semi-arid savanna dynamics are strongly driven by the inter-annual variability of precipitation and water availability. Certain sequences of below and above average precipitation primarily trigger vegetation dynamics (Schwinning & Sala 2004; Buitenwerf, Swemmer & Peel 2011). While droughts can have strong negative impacts on perennial grasses, especially when occurring in several subsequent years (Schwinning & Sala 2004; Zimmermann *et al.* 2010; Buitenwerf, Swemmer & Peel 2011), series of above average precipitation years can cause mass recruitments of woody plants (Joubert, Rothauge & Smit 2008; Nano & Clarke 2010; Joubert, Smit & Hoffman 2012a). These patterns are also evident in my simulation results: Mass recruitments of woody plants are especially pronounced when series of drought years are followed by series of years with above average precipitation. This is reasonable since young seedlings and saplings of woody plants experience less competition for water with perennial grasses as grass cover is greatly reduced after a prolonged drought (Kambatuku, Cramer & Ward 2011; Kambatuku, Cramer & Ward 2012). Grazing by e.g. domestic livestock is strengthening this effect, as grass mortalities are clearly increased by the combination of grazing and drought (Zimmermann *et al.* 2010; Buitenwerf, Swemmer & Peel 2011). Notably, heavy grazing is not a pre-requisite for woody plant encroachment but can facilitate it. Degradation of the vegetation will occur earlier and with a higher probability under intense livestock grazing. However, according to my results, encroachment continues irrespective of the grazing intensity applied once large quantities of woody plants successfully recruited.

The resulting dynamics of semi-arid savanna degradation are not gradual but strongly non-linear. Once encroached, a simulated patch will furthermore remain shrub dominated for several decades. However, my simulations show diebacks of woody plants on the patch scale after some decades as I assume age dependent mortality, which in reality could also be triggered and synchronized by processes like frost or diseases (Joubert, Rothauge & Smit 2008). If we consider spatial scales larger than the simulated patch scale, spatial variability in precipitation, grazing pattern and fire should result in a heterogeneous landscape with woody and grassy patches. Consequently, spatially homogeneous and continuous high grazing pressures will lead to a spatio-temporal equalisation of the otherwise patchy and asynchronous encroachment process on the landscape scale. This results in the widespread degradation of the savanna vegetation which can be found in many regions of the world (citations). This explanation for landscape scale degradation based on patch scale processes is in line with findings of other studies, suggesting that savannas cycle naturally between a tree dominated and a grass dominated state on the patch scale (Meyer *et al.* 2007; Moustakas *et al.* 2009). These studies did however not assess the effect of different rangeland management strategies on degradation.

To summarize, the simulations point out the outstanding importance of climatic variations and soil-water availability for the dynamics of semi-arid savanna vegetation. Consequently, I consider it essential to apply eco-hydrological approaches for the simulation of semi-arid ecosystems.

4.1.2 Environmental change and shrub encroachment

It is certainly one of the most interesting findings of my thesis that the predicted response of the semi-arid savanna vegetation to climate change clearly differed from the prediction of other studies (see chapter 1). While earlier studies expect increasing levels of shrub cover under scenarios of climate change (Bond & Midgley 2000; Bond, Midgley & Woodward 2003; Tietjen *et al.* 2010; Higgins & Scheiter 2012) or suggest increased levels of CO_2 as drivers of current encroachment (Kgope, Bond & Midgley 2010; Buitenwerf *et al.* 2012; Rohde & Hoffman 2012), I found that the risk for encroachment as well as its intensity will decrease or remain at the status quo under future climatic conditions and irrespective of the effects of CO_2.

In order to understand these discrepancies in the predictions of vegetation composition under future climatic conditions I want to briefly explore the specific explanations for these patterns that the different studies provide. For example Bond & Midgley (2000) suggested that savanna tree species will benefit from increased levels of atmospheric CO_2 because of their C_3 photosynthetic pathway. Accordingly, the resulting increased growth rates of woody plants will allow for faster regeneration and growth after fires and consequently an escape from the so-called ‘flame-zone’ (Higgins, Bond & Trollope 2000; Kgope, Bond & Midgley 2010; Higgins & Scheiter 2012). Other studies argue with a shift in the competitive balance between trees (C_3) and perennial gasses (C_4) to explain the predicted increase in tree cover (Tietjen *et al.* 2010; Ward 2010). My prediction of constant or declining woody cover under future climatic conditions is, however, based on the decreasing soil-water availability in the topsoil layer (see chapter 1 fig. 1.4). This in turn, is mainly caused by elevated mean temperatures and evaporation rates and has negative consequences for the severely drought sensitive seedling germination and establishment of woody plant species (compare with Kraaij & Ward 2006; Joubert, Rothauge & Smit 2008; Nano & Clarke 2010; Joubert, Smit & Hoffman 2012a).

4.1.3 The role of fire in semi-arid rangelands

My simulations in chapter 2 underlined that fire can substantially influence the vegetation dynamics of semi-arid savannas. Again, my result contradict the findings of a number of studies which suggest that fires only play an important role for mesic savanna dynamics (Sankaran *et al.* 2005; Higgins *et al.* 2007a; Sankaran, Ratnam & Hanan 2008).

The explanation for this discrepancy is analogue to the explanation given in the last paragraph (4.1.2). The different predictions can be attributed to varying underlying conceptual models of savanna functioning. Both views, the one represented by my work, and the view of studies that come to alternative conclusions explain the effects of fire via the link of a key demographic bottleneck of woody plant species. However, the particular life stage at which fire is assumed to have its main impact on woody plants differs between the two perspectives. Authors neglecting the role of fires for semi-arid and arid systems likely derived their system understanding from mesic savannas. This is also reflected in the design of the respective studies, which mostly focus on the effects of fire frequency and fire intensity on topkill of woody plants (van Langevelde *et al.* 2003; Sankaran *et al.* 2005; Higgins *et al.* 2007a; Sankaran, Ratnam & Hanan 2008). As a consequence, they assume a bottleneck of woody plant species at the transition of individuals from the sapling stage to larger size classes, which is narrowed with increasing fire frequency and intensity as plants are repeatedly top-killed and hindered in growth and reproduction (Higgins, Bond & Trollope 2000; Bond & Midgley 2000). Accordingly, these studies are not considering effects on seedling survival and the timing of fires relative to recruitment events as suggested in my work (chapter 2) and in other recent studies that also consider fire to be very important in shaping semi-arid savanna vegetation composition (Harrington 1991; Joubert, Rothauge & Smit 2008; Midgley, Lawes & Chamaille-Jammes 2010; Nano & Clarke 2010; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012b).

4.1.4 The role of the woody plant recruitment bottleneck

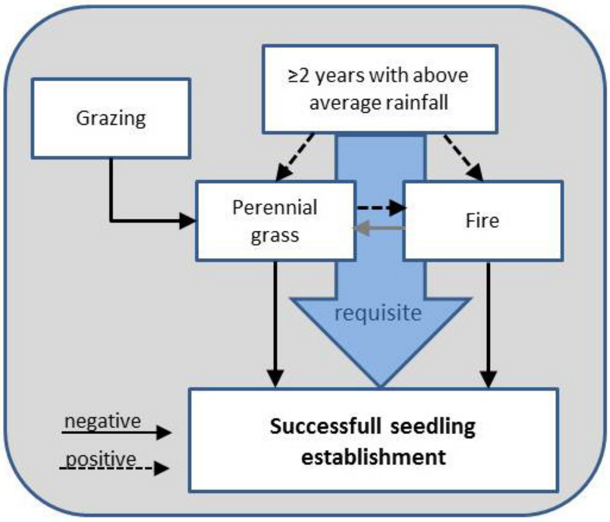

Figure 4.1: Causal relations of factors acting on the woody plant recruitment bottleneck in a semi-arid savanna. A series of at least two subsequent years with above average, well distributed precipitation, leading to high soil water availability, is the prerequisite for successful establishment of the drought sensitive seedlings. Fire and competition with grasses can further hinder this success. Both, perennial grass density and fire occurrence are also favoured by the above average precipitation as above rainfall promotes grass growth and accompanying thunderstorms increase fire probability. The latter is also increased by high grass biomass serving as fuel. Thus, increased soil water availability due to series of years with high quantities of rainfall both trigger the recruitment of tree seedlings, but also promote mechanisms controlling seedling establishment. In contrast, grazing can strongly reduce perennial grass abundance and biomass and thus indirectly promote woody plants seedling success.

Highlighted by the findings presented in the previous two paragraphs (4.1.2 & 4.1.3) and supported by strong evidence from literature, my thesis indicates that the demography of encroaching woody plant species is a key factor for the overall dynamics of the semi-arid savanna vegetation composition (Harrington 1991; Higgins, Bond & Trollope 2000; Midgley & Bond 2001; Kraaij & Ward 2006; Joubert, Rothauge & Smit 2008; Midgley, Lawes & Chamaille-Jammes 2010; Nano & Clarke 2010; Nano *et al.* 2012; Joubert, Smit & Hoffman 2012b). The central bottleneck at the seedling stage of woody plants is in the first place determined by water availability but also by grazing, fire and tha abundance of perennial grasses. These factors in turn are partly interdependent which leads to a strong control of shrub seedling recruitment opportunities (See figure 4.1).

It is important to emphasize the central role and specific nature of the recruitment bottleneck of woody plant species, as this distinguishes the vegetation dynamics of semi-arid savannas from the one of mesic savannas. However, the fact that literature refers to a ´recruitment bottleneck` for all types (i.e. mesic and semi-arid to arid) of savannas (see above), might lead to some confusion. It needs to be kept in mind that in mesic savannas tree recruitment is thought to be mainly limited by frequent hot fires inhibiting the growth of saplings to the size of trees (Higgins, Bond & Trollope 2000; Bond & Midgley 2000) and not, as depicted in figure 1 for semi-arid savannas, by water availability and seedling survival to the sapling stage.

However, for many woody savanna plant species it is largely unknown how their germination and growth depends on water availability. This is not only unknown for the process germination itself , but particularly for the growth during the subsequent years when seedlings have to establish a root system and carbon reserves that allow them to survive fire, drought and browsing (Midgley, Lawes & Chamaille-Jammes 2010; Hean & Ward 2012; Joubert, Smit & Hoffman 2012a; Joubert, Smit & Hoffman 2012b). In this study, I assume encroaching woody plant species to typically be sensitive to drought and fire. However, especially on the time-scales of environmental change other species could benefit in case they are more robust against drought or fire, and especially if they invaded from regions with other climatic conditions. Hence, we need to improve our understanding of the eco-physiological basics of seed germination, seedling establishment and seedling and sapling growth of savanna plants to be able to make reliable predictions of vegetation dynamics under alternative land use and climate scenarios (Midgley & Bond 2001; Midgley, Lawes & Chamaille-Jammes 2010). Thus, some urgent key challenges for future eco-physiological research arise: (i) the drought sensitivity of woody plant species germination and seedling establishment should be quantified for a range of savanna species (from mesic to arid savannas), (ii) it should be analysed whether CO_2 concentrations and temperature increase affect germination and establishment success of these plants, and (iii) it should be investigated whether encroaching and non-encroaching woody plant species differ in this regard. Further, in matters of fire resistance of woody plants, (iv) specific research is needed to understand how seedlings and saplings of different woody plants respond to fire of different intensities and at different times after germination.

Finally, in view of the importance of the bottleneck of woody plant species and its dependence on water availability I strongly emphasize the application of eco-hydrological approaches as represented by the present modelling approach for the assessment of land use and climate change impacts on semi-arid savanna dynamics.

4.2 Role plays with Namibian land reform beneficiaries

4.2.1. Rangeland management of land reform beneficiaries

The model-based role plays were used to assess farmers' decisions with regard to two different problems at a time. First, the basic herd management strategy (chapter 3) and second the cooperation in water infrastructure maintenance (Appendix A). During these role plays farmers could decide upon their annual herd management. Those farmers who actually share water infrastructures in their real life (14 groups) were additionally asked to make payments to a cooperative fund for water infrastructure maintenance (see Appendix A). Ten years of farming were simulated with Namibian land reform beneficiaries who either bought land with a subsidised loan according to the Affirmative Action Loan Scheme (AALS) or who have been allotted land to according to the Farm Unit Resettlement Scheme (FURS). Thus, I considered the two most important instruments of the Namibian redistributive land reform (Adams 1993; Republic of Namibia 1995; Falk *et al.* 2010).

Herd management decisions

As described in chapter 3, I wanted to identify basic management decisions of land reform beneficiaries. During the role plays, farmers were asked to decide how many animals of what type to buy and/or sell at every time step (i.e. cows, oxen, heifers, male or female weaners). Interestingly, my results showed that farmers based their management decisions on capital variables (account balance, herd size) rather than on environmental variables (rainfall, vegetation state). I found that farmers applied a conservative management strategy with stable and relatively low, in many cases even decreasing, herd sizes. This implies that under the given circumstances the management of these farmers will not per se cause (or further worsen) the problem of savanna degradation and shrub encroachment due to overgrazing. However, as this management is rather based on high financial pressure, forcing the farmers to stock conservatively and to discard their original management plans (i.e. to apply an efficient production system and to build up a certain herd size) it is not an indicator for successful rangeland management. This is underlined by the details of their production system that we gained from the role plays, which indicate that the land reform beneficiaries often sold animals before they were finished for marketing (chapter 3). Also the findings of the accompanying interview with the farmers (Falk *et al.* 2010) and of a further study performed in the study region suggest the major importance of financial pressure for management decisions (Werner & Odendaal 2010).

The financial pressure experienced by farmers is furthermore strongly related to the size of their farms and the instrument of redistribution. Owners of larger farms that usually acquired their rangeland with subsidised loans according to the AALS suffer from high payments for these loans (Falk *et al.* 2010, Werner & Odendaal 2010). In contrast, as also visible in our role plays, beneficiaries on small farms (FURS) suffer from limitations in herd size and high costs and thus hardly manage to make any revenue from their business (Tomlinson, Hearne & Alexander 2002; Werner & Odendaal 2010; Falk *et al.* 2010).

Water infrastructure cooperation

I was further interested in the cooperation behaviour of farmers and included farmer's decisions regarding their contribution to maintain shared water infrastructures in the role plays (see Appendix A for a detailed description). Only farmers that shared water infrastructure in real life (45 farmers on 14 farms who all received land according to FURS) were included in this part of the role play. In particular, groups of farmers sharing a water pump were asked to contribute (from their virtual account) to a collective fund from which running costs for maintenance can be covered. Note, that farmers were not informed about the amount of costs incurring in the upcoming period which were randomly drawn from a normal distribution in the simulation model. Hence, farmers had to make their decision about a contribution to the fund in uncertainty. They were furthermore enabled to conjointly discuss eventual rules for contributions to the fund face to face at any time during the role play. If in reality water infrastructure breaks down and there is no money to instantly solve the problem, farmers have to take their cattle to water points of neighbouring farms on an interim basis, where they are charged money for this service. In the experiment we charged the farmers N$ 50 per head of cattle in case the incurring water maintenance costs exceeded the money available in the water fund.

The main questions of particular interest in this part of the study were: Do land reform farmers cooperate successfully, and what is the mode of cooperation that they explicitly agree on or implicitly apply, and what are the determinants or underlying fairness norms (especially with regard to the heterogeneity within the groups) for the respective decisions (see Appendix A for details).

One main result of these role plays is that farmers had problems in agreeing on rules that determine the contribution of individual group members. This is important since groups without an agreement are assumed to likely not apply an optimal cooperation strategy (Ostrom 1998; Nowak 2006; Ostrom 2009). Particularly, we found, none of the groups with ethnic heterogeneity came to an agreement (0 of 6 groups), while 75% (6 of 8 groups) of ethnically homogeneous groups agreed upon a rule. This however did not mean that members of groups without an agreement did not contribute to the fund. Rather, they decided conditionally on an annual basis and did not apply one clear rule that they agreed upon and considered appropriate indicating a sub-optimal mode of cooperation.

Most of the groups which came to an agreement decided to pay an equal amount per farm, while only one group agreed to pay per head of cattle. The latter however, is the payment scheme recommended by the Namibian governments' extension office and literature, since this rule is balanced with regard to the congruence between provision and appropriation (Ostrom 2009). Livestock is consuming by far the largest amount of water provided by water pumps in rural semi-arid rangelands (Falk, Bock & Kirk 2009).

A working water infrastructure is essential for the success of a rangeland business in this semi-arid environment. Hence, our finding indicates that cooperation, although in the case of water infrastructure obligatory, is still sub-optimal and should be improved. In particular as farms that need to share infrastructure are always small, they cannot afford any unnecessary additional costs or losses (compare with chapter 3, Werner & Odendaal 2010, Falk *et al.* 2010).

Implications for the Namibian land reform

We gain three key insights from our interactive model-based role plays. (i) The participating farmers are facing a difficult economic situation, which determines their management practices (ii) they however, do likely not contribute to shrub encroachment and rangeland degradation through overstocking, and (iii) they are often not tapping the full bio-economic potential of their on-farm situation, i.e. they manage with suboptimal production systems, herd sizes below the ecological capacity of the system and due to non-optimal cooperation they also risk avoidable costs.

Based on these results I draw the following four major conclusions with regard to the implementation of the redistributive land reform in Namibia: (i) resettlement farms allotted according to the FURS instrument in the Omhake region should not be smaller than the official recommendation for a lower limit of 1.000 ha (compare with Tomlinson, Hearne & Alexander 2002; Werner & Odendaal 2010, citation farmsize), (ii) initially, farmers should be enabled to start farming with a productive herd. In particular, at the time of redistribution a certain minimum number of healthy reproductive animals should be assured and respective trainings should be conducted to enable the implementation of an efficient production system, (iii) cooperation between farmers should be facilitated (especially between small FURS farms). For example, specific training measures such as the interactive role plays could be an approach to help farmers exploring the advantages of cooperation and could facilitate the building of trust (Gurung, Bousquet & Trebuil 2006; Guyot & Honiden 2006; Becu *et al.* 2008). According to our findings, groups of farmers with relatively equal assets and a homogeneous ethnic origin might perform better in this regard. (iv), in order to prevent overstocking in case of a release of the farmers from the present financial burdens, which then indeed could cause rangeland degradation, farmers should offered rangeland ecology trainings in order to provide them with the knowledge for estimating the capacity of their range to sustain livestock.

4.2.2. Benefits and limitations of the interactive model-based approach

The approach presented in my thesis is often referred to as companion or participatory modelling (Becu *et al.* 2008). Hereby, modelling is not just used to represent a certain aspect of reality, which is then used to e.g. communicate certain correlations or ideas between researchers and stakeholders, but the model development itself is already involving stakeholders. In our case, stakeholders have participated in generating model rules during several ′test-run` sessions with local farmers and members of the emerging commercial farmers support programme (a Namibian NGO[1]). Therefore, the approach is not only interdisciplinary, as ecologists and economists contributed to the study, but also trans-disciplinary, as stakeholders have directly been involved.

Our interactive role plays were generally perceived positively by the participating farmers. They repeatedly stated that they experienced the role plays rather as training than as participation in a survey or research study. The conjointly conducted surveys have likely benefitted from the very good working atmosphere (see e.g. Falk et al 2010) and both sides have consequently profited from the study approach (compare with Gurung, Bousquet & Trebuil 2006; Guyot & Honiden 2006; Becu *et al.* 2008). Especially with regard to water management, such role plays have the potential to facilitate future cooperation. The participants had the opportunity to gain a deeper understanding of the relevance of a working cooperation and the experiment caused at least an occasion to

[1] See website of the ECFSP on http://www.agrinamibia.com.na (projects→emerging farmer)

communicate and discuss the issue of collaboration with all involved members of a group. Generally, such role plays offer the chance to learn by experimental trial and error and thus for “learning by simulating” instead of learning by doing (Barreteau, Bousquet & Attonaty 2001).

A further advantage of this simulation-based approach is that farmers have to make decisions under very realistic conditions compared to more abstract and controlled experiments. Management decisions of participants are made under uncertain conditions (e.g. regarding cooperation, rainfall, vegetation state) and they directly experience the consequences of their own decisions. Further, role players make decisions that directly relate to their real life experiences (e.g. same farm size, number of animals, equal costs, same neighbours), which increases the probability of a good understanding of the role play and therefore generates realistic results with a high validity.

However, the approach also has its limitations. The dynamic simulation underlying the role play induces high variability in the respective variables. Consequently, during the experiment, the independent variables of interest are not fully manageable. This causes problems in the reproducibility of the experiment and also in the ability to detect clear pattern in statistical analyses. The latter would be easier to handle in a controlled experiment, where e.g. only the variable rain is deliberately varied and all other variables (e.g. vegetation, herd size, account) are fixed if one wanted to test for the respondents' reaction to rain. However, in such a situation the role player might only react according to his theoretical understanding of the system dynamics. We could thus learn about his perception of the system in general, but not however, how important this variable would be for his decision in the overall context of his real life situation. In other words both approaches, the (more) controlled experiment and the simulation-based role play have advantages and disadvantages and could be described as complementary. It would thus be interesting to conduct more controlled experiments subsequent to our approach to deepen the understanding of the decision making of the role players.

4.3 Conclusions for semi-arid rangeland management

My thesis reveals that rangeland management in semi-arid savannas is a complex and challenging task. A precautious management of semi-arid rangelands' natural resources seems indispensable. The ecosystem dynamics exhibit a strong threshold with regard to the capacity of the system to support livestock production (see chapter 1) and the resulting dynamics of vegetation degradation are non-linear and not immediately reversible. To prevent degradation and to achieve maximum long-term productivity I strongly recommend a conservative management of semi-arid rangelands as also advocated by others (Quaas *et al.* 2007; Muller, Frank & Wissel 2007; Higgins *et al.* 2007b). This I do for several reasons that arise from or are emphasized by the results of my thesis:

1. Climate change predictions comprise a range of future scenarios and therefore future capacities of the system cannot reliably be predicted now but might be strongly reduced under future climatic conditions (see chapter 1). Therefore, to account for the uncertainty, it is necessary to be precautious and to not manage at the edge of the system's capacity. This is especially relevant since once shrub encroachment or the loss of vegetation has been induced, the productivity of the system might be strongly impaired for the duration of several decades.
2. I could show that fire is a powerful tool for sustainable management of semi-arid savannas which is efficient in preventing land degradation in form of bush encroachment (chapter 2). A conservative management with rather low stocking rates allows for the accumulation of perennial grass biomass needed to enable efficient fires in years of woody plant seedling establishment and is thus a prerequisite for the successful application of fires in semi-arid rangeland management. Overstocking will disable this important option (see chapter 2)
3. The costs of herd adjustments and the risks involved in highly dynamic opportunistic management strategies might further increase under climate change, as climatic variability will likely increase (chapter 1 & chapter 3).
4. In the face of a changing land tenure in the (commercial) livestock production sector due to redistributive land reforms in Namibia and elsewhere, aiming at an alleviation of poverty of as many people as possible, it seems fundamentally important to apply cautious and conservative management strategies to avoid high temporary investments for herd adjustments as well as highly variable revenues (chapter 3).

In addition, my results clearly indicate the importance of the recruitment bottleneck of woody plants for semi-arid savanna vegetation. A successful management of these ecosystems should consequently be aware of this fact and specifically aim at narrowing this bottleneck by the use of fire or at least at preventing to loosen it by applying sustainable stocking rates (see chapter 2).

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Appendix A

The impact of heterogeneity in endowments and norms on public good provision [1]

The manuscript presented in this appendix is based on the methods and role plays described in chapter 3 of this thesis and its key results are also discussed in the general discussion (chapter 4).

Summary

Achieving cooperation in natural resource management is always a challenge when incentives exist for an individual to maximise her short term benefits at the cost of a group. We assess the situation of a social dilemma in water supply cooperation within land reform projects in Namibia. In the context of the Namibian land reform, beneficiaries share the operation and maintenance of water infrastructure in order to gain economies of scale. Our paper assesses how alternative fairness norms affect the probability of cooperation.

In first step we conducted an exploratory overview of the social-ecological system of central Namibian land reform projects. The Social Ecological System (SES) Framework of Ostrom (2009) served as a guideline for this assessment. Taking the complexity of the cooperation situation into account we designed a role-play which is based on a social-ecological simulation model. The role-play simulates the real life decision situations of land reform beneficiaries where equilibriums are permanently changing. This approach helped us not only to better understand the cooperation challenges of Namibian land reform beneficiaries but provided support to stakeholders in their decision making and institution building.

Our study provides evidence that different fairness norms overlap: Land reform beneficiaries increase their contributions (i) as other group members increase their payments (conditional cooperation), (ii) as they are more productive (inequality aversion) and (iii) as they own more livestock (congruence of appropriation and provision & inequality aversion). Decisions are made considering the overall context. We further see evidence that norm homogeneity is in particular critical for the success of collective action if there is a high degree of endowment heterogeneity.

[1] Submitted as Falk, T., **Lohmann, D.**, & Azebaze, N. "The impact of heterogeneity in endowments and norms on public good provision" to: *Ecological Economics*.

A.1 Introduction

Achieving cooperation in natural resource management is always a challenge when incentives exist for an individual to maximise her short term benefits at the cost of a group. Since everybody in the group has the same rationality, the group would be trapped in a situation where it misses out on potential gains. The Prisoners Dilemma or Hardin's Tragedy of the Commons (Hardin 1994) are illustrations of this problem (Bardhan 1993). From a more natural scientist point of view, Nowak (2006) argues that defectors in a group of co-operators have a higher average fitness than co-operators who vanish from the system as a result of selection. Populations consisting only of co-operators, however, have a higher average fitness.

Observations both of natural and social systems demonstrate that it is possible to achieve cooperation (Nowak 2006; Ostrom 1998, 2010). Therefore, the focus of attention shifted towards the assessment of factors increasing or decreasing the probability of cooperation.

One such factor receiving attention in recent years is the heterogeneity of cooperating groups. There is a popular logic that social, cultural, and economic homogeneity of cooperating groups increases the predictability of interactions and therefore supports collective action (Poteete & Ostrom 2004; Ruttan 2008). Baland & Platteau (1996) distinguish three sources of heterogeneity:

1) ethnic, race or other cultural divisions;
2) variations in endowments;
3) differences in interests.

Cultural differences can result in different interpretations of rules (Baland & Platteau 1996). It might impede the development of trust (Poteete & Ostrom 2004) and potentially increases the transaction costs of monitoring and enforcement as social and moral consequences for complying with institutions may be less effective (Rustagi *et al.* 2010, Falk *et al.* 2012).

Inequality of wealth interacts with the relative costs and benefits of cooperation (Poteete & Ostrom 2004). Due to differences in endowments diverse user groups may have different opportunity costs related to the resource use. Both cases can hamper cooperation but can be solved by adapted appropriation and provision institutions. Oppositional, Olson (1995) argues that in cases when the rich receive disproportional high benefits from the resource they are willing to make disproportional provisions (Baland & Platteau 1996). This requires, however, low emotional costs of subsidising free-riding of the poor.

Heterogeneity of interests may affect incentives to cooperate if for instance the dependency on the resource varies and consequently discount rates are not the same. Less dependent users may try to exploit short term benefits which can lead to a collapse of collective action.

The empirical evidence on the role of group heterogeneity is, however, ambivalent (see e.g. Baland & Platteau 1996; Varghuese & Ostrom 2001; Ruttan 2008) and a more thorough assessment of its impact on the success of cooperation is needed. Varghuese & Ostrom (2001) argue that groups can overcome challenges related to heterogeneity by crafting appropriate institutions. They claim that if benefits are substantial, users are likely to invest in effective rules which are perceived to be fair (see also Buckley & Croson 2006). What happens, however, if the perceptions on fairness differ within the group (Ostrom 1998)? In this paper we will look at the impact of heterogeneity on the provision of a collective good in general and the interactions between types of heterogeneity in particular.

We study the case of water supply cooperation within land reform projects in Namibia. In the context of the Namibian land reform, beneficiaries share the operation and maintenance of water infrastructure in order to gain economies of scale. Even though there are small groups with direct communication possibilities, it is not uncommon that groups cannot come to an agreement in time such that water points are not maintained causing additional costs. Taking a broad summary of the socio-ecological system as a starting point, we will assess whether we can observe signs of alternative fairness norms in role-play exercises. Specifically we look for a) norms of the congruence of appropriation and provision (Ostrom 1998, 2010, design principle 2B) and b) norms of inequality aversion (Fehr & Schmidt 1999) in the sense of conditional cooperation (Fischbacher & Gächter 2010). This distinction is relevant as individual payments under the two norms differ depending on the endowments of group members.

In summary, we want to answer the following questions:

Which fairness norms dominate in our sample? Is fairness understood as congruence of appropriation and provision or as inequity aversion/conditional cooperation?

Does the interaction between norm heterogeneity and heterogeneity of endowments have an impact on the success or failure of collective action?

In following section we will give an explorative overview of our case study based on the Social-Ecological-System Framework of Ostrom (2009). We will then describe in Section 3 how we translated this situation into simulation-model-based role-plays. Section 4 summarizes our theoretical deliberations before we present the empirical results of the role-plays in Section 5. We draw our conclusions in Section 6.

A.2 Explorative description of the social-ecological systems of land reform projects in central Namibia

Cooperation patterns of land reform beneficiaries are the outcome of complex features of social-ecological systems (SES). We structure our explorative assessment according to the SES framework of Ostrom (2009) into the main sub-systems (i) the governance system (GS), (ii) the resource system (RS) and resource units (RU) as well as (iii) the users as actors (A).

Governance System (GS)

Land Reform is an important project of independent Namibia as it not only cures an unfair land distribution but also maintains political stability in the country. For more than 18 years, land has been redistributed to previously disadvantaged groups of the Namibian society using a broad range of instruments, such as group resettlement, subsidized loans, redistribution of government land and in a few cases also expropriation. In this paper we focus our attention on the *Farm Unit Resettlement Scheme* (FURS) which is based on the willing-seller willing-buyer principle. The acquisition is based on the preferential right of the Namibian state to purchase agricultural land whenever any owner of such land intends to dispose of it (RoN 1995a). The government divides the farms into smaller portions and any Namibian citizen who has been socially, economically or educationally disadvantaged by past discriminatory laws can apply for an allotment of land acquired for resettlement (e.g. RoN 2002). Successful applicants are supposed to receive a 99-year lease agreement with the government. In contrast to this we observe that by 2008 exactly half of the FURS farmers in our sample did not receive a leasing contract from the Ministry of Lands and Resettlement which administers FURS. These beneficiaries therefore hold no written proof of their rights on the allotted land (Falk *et al.* 2010, Werner & Odendaal 2010). One reason for the delay in issuing the contracts is the obligation of the government to fully maintain and repair the water infrastructure on the farm before redistribution. The responsible units lack, however, the capacity to cover all farms in time.

FURS farmers can benefit from cooperation with neighbouring farmers especially because they tend to use relatively small farm units and can accomplish economies of scale. The beneficiaries, however, face the challenge that they have to establish totally new collective choice and operational rules and need to agree on a monitoring and enforcement system. The farmers did not know each other before resettlement and often come from different ethnic groups with different value sets.

Resource System (RS) and Resource Units (RU)

The research was conducted in the Omaheke region in east central Namibia (Fig. 3.1, chapter 3). Our research concentrated on the eastern part of the region where the vegetation is dominated by an *Acacia-Terminalia* tree-and-shrub savannah of the Central Kalahari (Mendelsohn 2002). With mean annual precipitation (MAP) between 300 mm – 400 mm, the region is considered to be a high potential livestock farming area. The carrying capacity is estimated to be between 12 and 18 ha/LSU (Mendelsohn, 2006).

All land reform beneficiaries in our sample are active in the agricultural sector of livestock farming mainly with cattle. FURS farmers are allotted individual clearly marked sections of land. The size of the farms, however, varies widely between 50 ha and 2,000 ha within our sample. Concerns have been expressed repeatedly that government allotments are too small to accomplish economies of scale and thus economically unviable land units (Werner & Odendaal 2010). This problem can be partly solved by establishing cooperation and collective action between users of small farms as is common with regard to the management of water infrastructure. Typically, prior to resettlement water infrastructure was centrally managed by one decision unit. After splitting up the farms into smaller units, some of those do not often have exclusive access to a water pump. As a consequence, the occupants of such farm units are forced to cooperate in the provision of infrastructure maintenance and appropriation of water. A lack or breakdown of water infrastructure restricts the opportunities to make full use of the land, causes costs of livestock losses, or requires making use of more costly water supply options. In addition, a high concentration of livestock around few water points potentially causes localised degradation. On the farms in our sample, one working pump had to serve between 50 and 2,000 ha (mean = 1,369 ha).

Land users as actors (A)

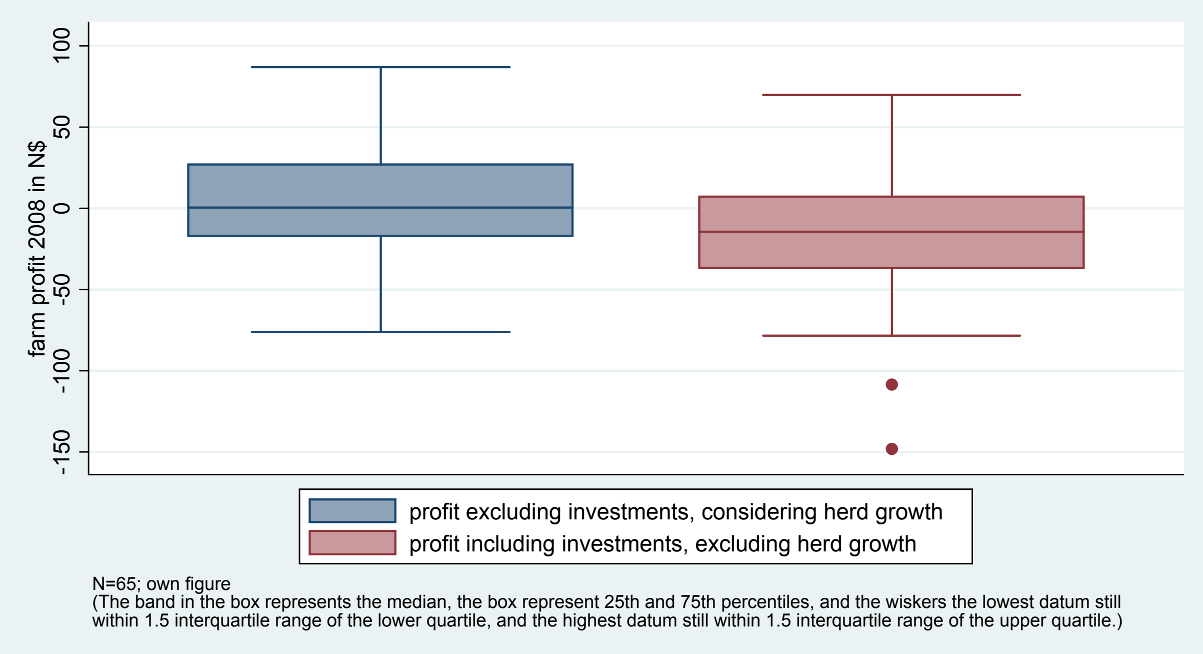
Our sample consists of individual farmers using one farm unit exclusively. The government's selection criteria for FURS beneficiaries have repeatedly changed but the program generally focuses on the poor. 60 percent of the household heads of the sample are full-time farmers.[2] The remaining part-time farmers spend on average 75 days per year on the farm. 80 percent of the respondents claimed to have non-farm income. 44 percent of the household heads did not finish secondary school, 47 percent finished secondary school as the highest degree, and 9 percent hold a technical or university degree. 72 percent of our sample had previous farming experience but only a minority of them in a commercial setting. Only 23 percent of the respondents received some kind of farming training.

The median of the size of the groups sharing water infrastructure was 2.5 (min=2, max=6). In 43 percent of the groups more than one ethnic group was represented (maximum = 3).

The average annual gross profit of our respondents adding up all on-farm income, deducting only running costs and adding the average annual herd growth was N$ 5,463 with a high standard deviation of σ = 32,466. Almost half of our sample was making losses under this most optimistic income calculation. The figures for the net farm profit, which include fixed costs, 2008 investment costs, and liability payments but exclude herd growth, are even worse. The net farm loss was on average N$ 14,395 (σ = 37,708). According to this indicator almost two third of the sample was making losses (Fig. A1, Falk *et al.* 2010).

[2] There has been no random allotment of farm units.

Figure A1: Farm profit calculations considering herd growth and investments



A.3 Role plays based on ecological-economic modelling

The application of the SES framework makes us aware of the complexity of the challenge to collectively maintain water infrastructure on Namibian land reform farms. Namibian land reform beneficiaries operate in systems of dynamic complexity. Amongst others these are marked by permanent and often delayed changes, multiple feedbacks at different speeds, nonlinear relationships of variables, and often irreversible developments (Sterman 2001, 2006, Barreteau *et al.* 2001). These systems are reflexive, acting on decision makers, who through their actions act on various components of the system (Bousquet *et al.* 2002). We represent a situation where the absolute and relative endowments of land reform beneficiaries are permanently changing, resulting in erratically changing equilibriums. As a consequence the farmers are confronted with the permanent need to review institutions taking internalised fairness norms into account. Since the social-ecological system we are dealing with has a considerable complexity we decided to use a computer simulation to represent it (see also Sterman 2001).

Following the general approach of Bousquet *et al.* (2002) we applied role-plays in order to acquire knowledge, build a model, validate the model, and support decision making processes. Our model and its conversion into a role-play simulate the complexity and dynamics of important parts of the social-ecological system and provide support for negotiation (see also Barreteau *et al.* 2001). Our model created a virtual world in which farmers could experiment, rehearse decision making, and play in a compressed time and space (see also Barreteau *et al.* 2001, Sterman 2001, 2006). The model provided them with immediate feedback and allowed them to adjust decisions. Experimenting with the simulation model induces much lower costs and risks for the players than a real life trial and error process of institutional change (Barreteau *et al.* 2001, Sterman 2006).

Compared to standard experiments this approach has a number of obvious disadvantages. The internal validity is low as it is difficult control parameters. As a consequence the results are difficult to compare (Bousquet *et al.* 2002). The role-plays are not suitable to test general theoretical hypothesis. Generating accepted scientific evidence requires controlled experiments which discriminate hypotheses and produce replicable results (Sterman 2006). Nevertheless, the more complex the phenomenon, the more difficult it is to draw conclusions from standard experiments on real life decision situations.

The advantage of the simulation model based role-plays is a higher external validity and a more realistic reproduction of real life decision situations (Barreteau *et al.* 2001). The objective of the role games is to assess a representation of reality rather than studying a theoretical pre-given one (Bousquet *et al.* 2002).

As a starting point we used an existing vegetation model (Tietjen *et al.*, 2010) and parameterised it for the Omaheke region/Namibia based on empirical field work, expert knowledge and a literature review. The model simulates the dynamics of natural resources depending on environmental conditions (precipitation/climate, hydrology, ecological interactions) and land use impacts (see also Chapter 1). Resource dynamics were derived from this ecological model and then dynamically linked to a social model that allows for the inclusion of livestock related decisions of farmers as well as cooperation in water management. For a more detailed model description please see chapter 3 of this thesis.

The ecological-economic model was then converted into a computer based role-play representing the provision situation of a public good. The plays simulate basic farming decisions and the voluntary contribution to water provision. We designed a user interface that allows communication between a facilitator and the model. The interface presents an output of all important state variables and allows for a subsequent input of the farmers' decisions. Based on illustratively communicated ecological and economic information calculated by the model, farmers make decisions regarding their stocking rates as well as their individual contribution to the maintenance of water infrastructure (the amount of money to be paid to a water fund). The decisions on stocking rates are relevant for our assessment of cooperation patterns because the livestock numbers change opportunity costs of infrastructure maintenance.

Outputs of the vegetation state are given using exemplary photographs taken from different vegetation states in the research region. The same approach of showing pictures is used for showing the body score of the livestock. Printouts of outputs are generated for every time step to present all other relevant numbers. Farmers receive a list in their mother tongue with the following information:

- rainfall in the previous role-play period,

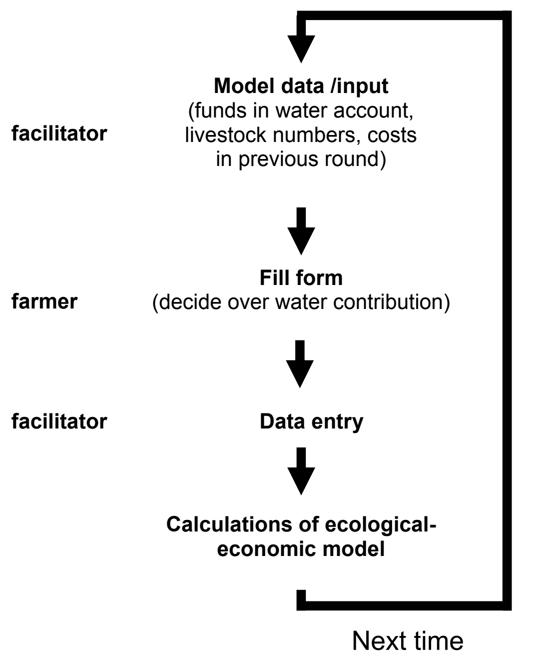
- number of livestock at the beginning of role-play period,
- age structure of livestock at the beginning of role-play period,
- number of livestock losses in the previous role-play period,
- individual farmer's account balance at the beginning of role-play period,
- total farms expenses to be covered in role-play period,
- account balance of the group's water fund at the beginning of role-play period.

In every time step the farmers can make two kinds of decisions: 1) they can buy or sell livestock, and 2) they have to decide how much to pay into their group's water fund. The role-play is set up in a way that all players have the opportunity to continuously communicate face to face as this is the most efficient form of communication for developing institutions (Balliet 2011) and a realistic representation of the real-life situation.[3] Depending on the players' decisions, new ecological and economic states (e.g. condition of livestock, account balance) are calculated, again as the basis for the next steps' decisions. Fig. A2 illustrates the role-play process.

The modeled water infrastructure costs vary from year to year, reflecting randomly appearing maintenance costs. In the role-play, each group shares water infrastructure which consists of a diesel driven and a wind driven pump. The costs are modeled on the basis of expert interviews and on average set at N$ 2,350 (σ = 785) for the diesel driven pump and N$ 750 (σ = 250) for the wind driven pump. Farmers are not informed about the periods' water costs before making their contribution and therefore have to make decisions under uncertainty.

In the case that the money available in the fund is insufficient to cover the maintenance costs, the infrastructure breaks down. In reality the farmers usually take their cattle to the neighboring farm where they have to pay, however, for getting access to water. We assume a hypothetical fee of N$ 50 per head of cattle which is based on interviews with farmers even though the amount strongly varies in reality.

Figure A2: The cyclic process of the role plays



[3] There are several explanations why communication supports cooperation. Balliet (2011) mentions in particular receiving signals about other's willingness to cooperate, group identity and the development of shared norms.

Between January and April 2009 we conducted role-play sessions with 45 land reform beneficiaries on 14 resettlement farms. In an attempt to simulate the real life cooperation situation, the role-plays where played in groups of farmers who in fact share a water point. Players were given a virtual farm of the same size as their real life one and started with virtual livestock numbers being equal to their real livestock numbers in 2008.

For the data analyses, we identified groups agreeing to a rule by the standard deviation of their payments. If the group's standard deviation of the water payments was zero in a role-play period, we concluded that the group was following the rule to pay equally per person representing the norm of conditional cooperation. If the group's standard deviation of the water payments divided by livestock numbers was zero in a role-play period, we concluded that the group was following the rule to pay per head of livestock representing the norm of congruence of appropriation and provision. Descriptive statistics and correlation analysis were used for basic analyses.

In addition, the role-play data were analyzed calculating Random Effects Regression Models. These methods are applied to panel data sets where each role-play period of each player is analyzed considering dependencies between the behaviors of one player in different role-play periods. The empirical model explains for each year the individual's contribution to the maintenance of water infrastructure depending on the input variables in the role-play and group characteristics.

A.4 The theory of water infrastructure provision

In the following section expected outcomes of the role-plays are described theoretically. The model aims at drawing basic conclusions regarding the individual contribution and free ride incentives of farmers and analyzes the decision on the amount the farmers pay into the water fund. The decision to sell or buy livestock is not part of this analysis. Other costs like production costs, transactions costs, etc. are neglected.

Consider a group of two farmers. Each farmer has to contribute an amount C_i, $(i \in \{1,2\})$, into the water fund to cover the maintenance costs. In the role-play the farmers play 10 rounds. The relation between the role-play rounds t and t+1 is the amount remaining in the fund after subtracting the maintenance costs. The model neglects time preferences and interest rates. After the farmers have paid their contribution into the water fund, its value is given with:

$WF_t=C_1+C_2+WF_{t-1}$

The basic condition for the use of the water infrastructure by a farmer or a group is the fulfillment of the individual or the group participation incentive.

$OC_i > C_i$ or $OC_N > C_N$

At the group level, the total opportunity costs (OC_N) of both farmers must be higher than their total contribution (C_N). At the individual level the opportunity costs of a farmer (OC_i) must be higher than her contribution into the water fund (C_i).

The maintenance costs (K) are uncertain. The costs are continuously equally distributed in the interval [0, V], with V being the value of building a totally new water infrastructure. If the amount in the water fund (WF) does not cover the maintenance costs, the infrastructure breaks down and the group faces the opportunity costs of N$ 50 per head of livestock. The survival of the water infrastructure depends on the probability that the amount in the water fund is higher that the maintenance costs: $P(K \leq WF) = F(WF)$. Each farmer acts to minimize her expected costs:

$$EC_1 = C_1 \cdot F(WF) + \left(1 - F(WF)\right) \cdot OC_1 \qquad \text{eqn 1}$$

Once a water point has been allocated to a group of farmers, they cannot exclude a group member from its use. They can, however, exclude external intruders. Public good experiments representing the provision challenge have shown that people cooperate much more than predicted by some standard economic models considering the individuals as being selfish. A possible explanation of this behavior is that some individuals are conditional cooperators as a result of inequity-averse preferences (Fischbacher *et al.* 2001). In this case a player experiences some disutility, if her contribution is different to the contribution of the other farmer. This intrinsic utility element is deducted from her expected costs. In contrast to the model of Fehr & Schmidt (1999) our model does not assume different disutility levels for inequity. After transformation of (1), replacing $F(WF)$ with its value and including the fairness term, the new optimization problem of farmer 1 is given with:

$$\mathrm{Min}_{C_1} EC_1(C_1, C_2) = OC_1 - \frac{WF}{V} \cdot (OC_1 - C_1) + \alpha_1 \cdot (C_1 - C_2)^2 \qquad \text{eqn 2}$$

$$\text{Subject to } WF = WF_{t-1} + C_1 + C_2$$

α_1 is hereby the inequity aversion parameter of farmer 1. OC_1, the opportunity costs of farmer 1, which depends on the number of livestock she possesses. The second term of equation 2 represents the benefit that farmer 1 gets when she pays her contribution into the water fund, namely the difference between her contribution and her opportunity costs weighted with the probability of the infrastructure not to break down. As long as the individual participation condition holds, the expected costs will be smaller than her opportunity costs, if she contributes $C_1 \geq 0$. This does not mean that the expected costs function is monotonically decreasing for any contribution smaller that the opportunity costs. The optimal contribution of farmer 1 resulting from this decision situation is:

$$C_1 = \frac{OC_1 - WF_{t-1} + (\alpha_1 \cdot 2 \cdot V - 1) \cdot C_2}{(2 + \alpha_1 \cdot 2 \cdot V)}$$

Proposition 1: If farmer 1 is acting opportunistically, she will decrease her contribution, when the contribution of farmer 2 increases.

$$C_1 = \frac{OC_1 - WF_{t-1} - C_2}{2}$$

Without the fairness norms, $\alpha_1 = 0$, the contribution of farmer 1 would be negatively correlated with the contribution of farmer 2. When farmer 1 computes her optimal decision, she takes the contribution of farmer 2 as given. Her expected costs reach the minimum at a certain amount in the water fund. If WF remains constant and the other farmers increase their contribution, farmer 1 must decrease her contribution to reach the minimum of her expected costs. This behavior conflicts with the concept of inequity aversion of Fehr & Schmidt (1999) and conditional cooperation of Fischbacher & Gächter (2010).

Proposition 2: The higher the livestock number of farmer 1, the higher her contribution into the water fund.
The farmer's opportunity costs are determined by her number of livestock. Independent from fairness norms she is willing to increase payments if her opportunity costs increase. She strives to avoid higher opportunity costs if the infrastructure breaks down, thus she has an incentive to contribute. The congruence of appropriation (livestock number) and provision (individual contribution) can be observed in this result (Ostrom 2010). A farmer with a high number of livestock will pay more than a farmer with a smaller herd.
Proposition 3: For $\alpha_1 > \frac{1}{2 \cdot V}$ the contribution of farmer 1 is positively correlated with the contribution of farmer 2.
Due to the uncertainty of the maintenance costs, farmer 1 will increase her contribution, if $\alpha_1 > \frac{1}{2 \cdot V}$. The uncertainty about the maintenance costs gives farmer 2 the possibility to have to pay either C_2 or OC_2. Since farmer 1 is not sure about the final costs for farmer 2, the influence of her fairness factor α_1 will be reduced by the uncertainty of the maintenance costs. If the costs were certain, farmer 1 would increase her contribution for any $\alpha_1 > 0$. In the presence of uncertainty she will, however, decrease or maintain her contribution for $\alpha_1 < \frac{1}{2 \cdot V}$, although she has some disutility resulting from violating her norm of inequity aversion (Fehr & Schmidt 1999, Fischbacher & Gächter 2010).
Our theoretical deliberations allow us to draw first conclusions which we summarize in Table 1. In the case of homogeneous endowments and consequently homogeneous opportunity costs the payments under the norm of inequity aversion are equal to the payments under the norm of conditional cooperation. In this case it is likely that the group will come to an agreement even if they do not share the same norms. In the case of homogeneous norms group members will most likely contribute to the public good even if their contributions exceed their opportunity costs. This can be explained by the fact that they try to avoid the intrinsic disutility resulting from norm violations. This case is consistent with Ostrom's (2005) statement that groups are likely to develop an adequate set of institutions when they share a common set of values.

The risk of failing collective action must be taken into account, however, when there is both a high heterogeneity of endowments/opportunity costs and norms. This is the case when smaller farmers demand a payment according to their appropriation and the richer ones propose equal payments per person. Under this condition the disutility of violating one's norm hinders the possibility to come to an agreement and unless process of harmonizing values takes place cooperation will be improbable.

Table A1: Interactions between the heterogeneity of endowments and norms

<table>
<tr><td colspan="2" rowspan="2"></td><td colspan="2">Heterogeneity of endowments/opportunity costs</td></tr>
<tr><td>Low</td><td>high</td></tr>
<tr><td rowspan="2">Heterogeneity of norms / α</td><td>high</td><td rowspan="2">payments under norm of inequality aversion
=
payments under norm of congruence of provision and appropriation
➔ high likelihood of collective action to succeed</td><td>payments under alternative norms differ and internally perceived costs of breaking norms vary too ➔ risk of failing collective action</td></tr>
<tr><td>low</td><td>payments under alternative norms differ but all participants agree to one norm and breaking the norm would lead to internally perceived costs which are weight against opportunity costs ➔ internal costs reduce risk of collective action to fail</td></tr>
</table>

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A.5 Results of the role plays

In the role-plays, 8 out of 14 groups did not come to a reliable agreement and did not follow a clear payment system. One group agreed at the beginning of the role-play to pay per head of livestock but in round two one player defected and the cooperation could not be established again. Six groups reliably agreed on a system. Five groups cooperated from the first role-play round on and one group started to cooperate after round four. Five groups agreed to the payment scheme per person while one group switched in the course of the role-play from payment per person to payment per head of livestock. There are negative correlations between the gini-coefficient of the livestock possession in a group and the fact that a group came to an agreement.[4]

31 percent of our players are conditional cooperators increasing their payments if the other group members increase their contributions.[5] 13 percent of the sample adjusted their payments to their share of the group's livestock herd. The payments of another 13 percent of the players correlated both with the payments of the other players and the livestock numbers. This is possible if there is a relatively stable relation of livestock numbers amongst the group members. Half of them are unconditional cooperators who made the same contribution in all role-play rounds. The payments of 42 percent of the players neither correlated with the other group members' payments nor their share of the livestock herd. We do not observe consequent free riding or straight opportunistic behavior, which means that all players contributed to the maintenance of the water infrastructure.

All reliably or non-reliably cooperating groups were ethnically homogeneous (Table 2).

[4] Spearman rank correlation for variables "group came to agreement" and "gini-coefficient of livestock possession of group": coefficient: -0.3808, p=0.000, N=14;

[5] Where there is a correlation between the own payment and the payments of the rest of the group in a particular experiment period (Pearson correlation calculated and considered to be correlated if $p<0.05$).

Analyzing the role-plays' contributions using regression models (Table 3) reveals that players with lower livestock numbers tend to make lower contributions. It can be further observed that the higher the contributions of the other group members the higher the own contribution will be. When standardizing the variables to $\mu = 0$ and $\sigma = 1$ one can see that the coefficients are approximately in the same range (0.447 vs. 0.490).

Significantly higher payments were made in role-play rounds when group members came to an agreement. Contributions did not decline over the role-play periods.[6] The higher the player's account balance in a role-play period the higher her contribution. The amount in the water fund was taken into consideration by the players. In our multivariate analysis the size of the group did not have a significant impact in individual contributions even though the descriptive statistics show that all reliably cooperating groups had only two group members.

We asked our respondents how many water pumps they have available on their farm and how many pumps are indeed working. There is a correlation between the total water payments of a group and the share of the operating water infrastructure.[7]

Table A2: Cross tabulation for ethnic heterogeneity and group coming to agreement (absolute frequencies of groups with relative frequencies in parentheses**)**

	number of ethnic groups in role-play session group			Total
	1	2	3	
no agreement	1 (7.14%)	3 (21.43%)	3 (21.43%)	8 (57.14%)
came to agreement	6 (42.86%)	0 (0.00%)	0 (0.00%)	6 (42.86%)
Total	8 (57.14%)	3 (21.43%)	3 (21.43%)	14 (100.00%)

[6] We excluded the last role-play round in order to avoid any possible end round effects.

[7] Spearman rank correlation for variables "total individual payment over all role-play periods" and "operating share of total infrastructure": coefficient: 0.3179, p= 0.0378, N=43;

A.6 Discussion and conclusion

The explorative assessment of the social-ecological systems of land reform projects shows that arbitrarily mixed groups of farmers are facing the challenge to solve the provision problem of a public good. They cannot build on a long-enduring history of joint resource management. The situation is aggravated if newly formed groups are ethnically heterogeneous. In our small sample no mixed group came to an agreement on water payments. Forcing ethnically mixed groups into cooperation situations might support nation building and potentially help to overcome tribalism, but in the short run the heterogeneity of norms makes it more difficult to establish a stable foundation of social capital. At the same time we observe that groups which came to an agreement made significantly higher payments.

Table 3: Random effects regression models explaining the natural logarithm of the individual players' payment for covering the water infrastructure maintenance costs: coefficients and cluster robust standard errors in parenthesis (minimum of dependent variable = 0, maximum = 9.6) *(*** p<0.01, ** p<0.05, * p<0.1)*

Variable	Minimum and maximum of variable	Random effects model	model with standardised variables
1) role-play round t	2 - 9	0.0787** (0.0371)	0.226** (0.107)
2) average payment of other players in all previous rounds	500 - 20880	-0.00005 (0.00008)	-0.130 (0.201)
3) all other group member's payment in t	0 – 8300	0.00020*** (0.00007)	0.447*** (0.159)
4) account balance in group water fund at beginning of t	0 – 28787	-0.00005** (0.00003)	-0.424** (0.208)
5) individual account balance at beginning of t	-494,954 – 414,650	0.000003** (0.000001)	0.368** (0.143)
6) livestock number at beginning of t	0 - 151	0.0152*** (0.0038)	0.490*** (0.122)
7) did the group come to an agreement	0/1	0.761** (0.380)	0.342** (0.171)
8) number of farmers in group	2 – 5	-0.193 (0.143)	-0.288 (0.214)
Constant term		5.344*** (0.847)	6.140*** (0.154)
Number of observations		360	360
Number of individuals		45	45
Number of observations per individual		8	8
Prob > F/chi²		0.0000	0.0000
R^2 within		0.1172	0.1172
R^2 between		0.4273	0.4273
R^2 overall		0.2644	0.2644
Wooldridge test for autocorrelation Prob > F		0.3281	0.3281
Robust Hausman test Prob > F		0.2892	0.2892

The most commonly used payment systems of rural water supply in Namibia are payment per head of livestock and payment per person (Bock *et al.* 2006, Falk *et al.* 2009). The two systems differ with regard to the congruence between provision and appropriation, which, according to Ostrom (2010), affects the probability of successful joint resource management. The provision and appropriation is most congruent under the rule to pay per head of cattle, because livestock is consuming the greatest share of water provided by rural water pumps in Namibia (Bock *et al.* 2006, Falk *et al.* 2009). This argument is the main reason why this payment system is actively advocated by the Namibian government extension services. A payment system where each group member pays an equal amount is more in line with fairness norms of conditional cooperation, which, according to Fischbacher & Gächter (2010), supports cooperation.

The rule of payment per animal as recommended by the government was chosen by one group in our role-games. This rule best achieves congruence between provision and appropriation. Five groups shared the costs equally. One explanation for this tendency is possibly that the transaction costs for determining the individual payments are higher for the rule to pay per head of livestock. In this case the payment has to be adopted every time to changing livestock numbers while the amount under the rule of equal payment per farmer is fixed.

Nonetheless, the regression model reveals that even if the players did not agree on a clear payment system per head of livestock the ones owning more livestock tend to make higher contributions. In theory, a system of paying equal amounts achieves a relatively high congruence of provision and appropriation if the gini-coefficient of livestock possession is low in a group. Since the payments under the norm of congruence of provision and appropriation and the one of conditional cooperation match in groups with a relatively equal distribution of endowments, these groups more probably cooperate. Our role-play revealed a correlation between choosing the equal payment rule and the gini-coefficient of livestock possession in the group.

We observe that more homogeneous groups in terms of endowments and ethnicity came more probably to an agreement (see also Bardhan 1993, Meinzen-Dick *et al.* 1997). At the same time, groups which came to an agreement made higher payments. We tried to test the interaction between endowment and norm heterogeneity using interaction terms but the results were unsatisfactory due to high multi-co-linearity. As a result we cannot draw final conclusions about this interaction. Further research is needed.

Our role-plays using a social-ecological model simulated a real life cooperation situation. The virtual environment was sufficiently similar to reality but simple enough to be played (Gurung *et al.*, 2006). In this way we increase the potential to learn from the role-plays about the real life behavior of the players. Using the terminology of Roe & Just (2009) we increase our ecological validity as the extent to which the context of the research is similar to the context of interest. As a consequence, the possibility to replicate our results is limited. There is a high probability of the presence of uncontrolled variation in unobserved variables. We also have only restricted control over subject characteristics but played with the subjects who are in the center of the context of interest in order to make statements specifically about their behavior. We see it as an indicator for the success of our approach that individuals which made higher payments in the role-plays manage in real life to keep the infrastructure given to them in better conditions. The role-plays allow us to draw conclusions on a complex real life decision situation. We compromised on internal validity but gained external one.

The simulation model based role-plays produced not only knowledge but provided support to stakeholders in their decision making (Barreteau *et al.*, 2001; Barreteau, 2003; Gurung *et al.*, 2006; Guyot & Honiden, 2006; Becu *et al.*, 2008). There was uniform response from the participants that they perceived the exercise as training rather than a research activity. Gurung *et al.* (2006) emphasizes that one key objective of participatory modeling is to facilitate dialogue, shared learning, and collective decision making through interdisciplinary research to strengthen the adaptive management capacity of local communities. Modeling in combination with role-plays is a tool to play with rules and strategies and in this way explore probable ecological and economic consequences. It limits the costs of trial and error methods and shifts the approach from costly learning by doing towards learning by simulating (Barreteau *et al.* 2001). Our approach simultaneously deepens the understanding of cooperation processes and serves as a tool to encourage discussion and institution building. In this sense, we supported Namibian land reform beneficiaries in a current and relevant challenge.

Which policy implications can be drawn from our research? First of all, the research confirms the ongoing challenge of institution building faced by land reform beneficiaries. This is not a short term issue anymore as some of the beneficiaries have been resettled more than 20 years ago (mean = 9 years). Considering the importance of water supply for farming in Namibia, pre- and post-resettlement support should not only pay attention to technical aspects of water infrastructure but should facilitate as well the process of institution building. Larger groups of farmers and less homogeneous ones in terms of endowments and ethnic origin need special attention. We further learned that there are no uniform fairness norms. Achieving congruence between provision and appropriation provides theoretically most reliable material incentives to cooperate. Nonetheless, any distribution of the expected costs works as long as the group reliably agrees to it. The intention should therefore not be to impose specific rules on groups but help them to harmonize their norms.

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Zusammenfassung

Nachhaltige Nutzung semiarider Savannen in Afrika unter dem Einfluss von klimatischem und politischem Wandel

Trockengebiete bedecken etwa 40% der globalen Landfläche und stellen die Existenzgrundlage für ca. 38% der Weltbevölkerung dar. Diese Ökosysteme sind weltweit von schwerwiegender Degradierung bedroht in deren Folge ein starker Rückgang von Ökosystemdienstleistungen wie zum Beispiel Grundwasser-Neubildung, Erosionsschutz und die Bereitstellung pflanzlicher Biomasse zu verzeichnen ist. Zudem sind diese Ökosysteme durch die zu erwartenden Veränderungen im Rahmen des vorhergesagten Klimawandels zusätzlich gefährdet. Entsprechend bedarf es globaler Anstrengungen um die spezifischen Gründe für Degradierung von Trockengebieten sowie deren Auswirkung zu analysieren und nachhaltige Landnutzungsstrategien gerade im Hinblick auf eine sich im Wandel befindenden Welt zu entwickeln.

In diesem Sinne befasse ich mich in meiner Dissertation mit der Degradierung semiarider Savannen, welche häufig in Form von Verbuschung, also einer Zunahme holziger Vegetation auftritt. Hierbei konzentriere ich mich insbesondere auf die Identifizierung nachhaltiger Landnutzungsstrategien und die Verbesserung des generellen Verständnisses der Vegetationsdynamik unter den spezifischen Gegebenheiten extensiver Viehhaltung. Darüber hinaus will ich den Einfluss externer Faktoren, insbesondere von Umweltwandel und Landreform, auf die Nutzung der Savanne sowie auf die daraus resultierende Vegetationsdynamik des Ökosystems bestimmen. Schlussendlich möchte ich, aufbauend auf dem so erlangten Verständnis der Ökosystemdynamik, Bedingungen und Strategien identifizieren, welche die nachhaltige Beweidung semiarider Savannen unterstützen und ermöglichen.

Simulationsmodelle bieten die Möglichkeit, Systemdynamiken unter verschiedenen zukünftigen Bedingungen vorherzusagen, alternative Landnutzungsstrategien zu testen und das Wissen aus verschiedenen Disziplinen in einem Ansatz zu integrieren. Dementsprechend verwende ich in meiner Arbeit ein öko-hydrologisches Simulationsmodell, welches ich um wichtige Funktionen erweitert habe, um den Einfluss einer Vielzahl von Klimawandel- und Landnutzungsszenarien auf die Vegetationsdynamik einer typischen semiariden namibischen Savanne zu simulieren. Im Rahmen dieser Simulationen untersuche ich insbesondere den Einfluss von verschiedenen Niederschlags- Temperatur-, sowie CO_2-Szenarien sowie den Effekt verschiedener Beweidungsintensitäten und Feuer.

Nicht nur die Umweltbedingungen, sondern auch politische und ökonomische Rahmenbedingungen wie zum Beispiel Landreform Programme haben einen Einfluss auf Landnutzungsstrategien. Dementsprechend war es ein weiteres Ziel meiner Arbeit, auch den Einfluss der laufenden Landreform im südlichen Afrika auf die Landnutzung und die semiaride Savanne selbst zu verstehen. Um dies zu erreichen, entwickelte ich ein

Agenten-basiertes ökologisch-ökonomisches Simulationsmodell, welches ich im Rahmen einer interdisziplinären, empirischen Studie für interaktive Rollenspiele mit Landnutzern in Namibia eingesetzt habe. Ziel dieser Studie war es die grundlegenden Entscheidungsmuster der Farmer sowie deren Kooperation untereinander zu identifizieren.

Die öko-hydrologischen Simulationen zeigten auf, dass die zukünftige Dynamik der simulierten Savannenvegetation stark vom jeweiligen Klimawandelszenario abhängt. Es zeigte sich, dass insbesondere die Kapazität des semi-ariden Ökosystems in Bezug auf die extensive Viehwirtschaft in Zukunft stark von der Menge und zeitlichen Verteilung von Niederschlägen abhängen wird. Außerdem weisen meine Ergebnisse darauf hin, dass Degradation in Form von Verbuschung unter künftigen klimatischen Bedingungen sowohl unwahrscheinlicher als auch weniger ausgeprägt sein wird. Dieses Ergebnis ist hierbei unabhängig von eventuellen positiven Einflüssen erhöhter CO_2 Konzentrationen auf Wachstum und Transpiration der holzigen Vegetation, obwohl frühere Studien aufgrund dieser Effekte zunehmende Verbuschung vorhersagen. Die Erklärung für die Abnahme der Verbuschung in meinen Ergebnissen findet sich in den negativen Effekten der Temperaturerhöhung auf die Keimung und Etablierung von Keimlingen holziger Savannenpflanzen, welche sehr empfindlich auf Trockenheit reagieren.

In weiteren Simulationsexperimenten konnte ich zeigen, dass der gezielte Einsatz von Feuer ein effizientes Werkzeug für nachhaltiges Weidelandmanagement sein kann, da Feuer die Etablierung von Keimlingen holziger Vegetation erfolgreich unterdrücken kann. Die entsprechenden getesteten Landnutzungsstrategien führten neben einer Verminderung des Risikos für Verbuschung auch zu einer Erhöhung der langfristigen Produktivität der semi-ariden Weidelandschaften. Dieses Ergebnis steht im Gegensatz zu den Aussagen vieler früherer Studien, welche Feuer lediglich eine Bedeutung in feuchteren Savannen zuordnen. Auch in diesem Fall begründen sich die Unterschiede in den Vorhersagen zwischen meiner und früheren Studien durch die unterschiedliche Berücksichtigung des Engpasses im Lebenszyklus der holzigen Pflanzen bei deren Keimung und Keimlingsetablierung.

Die ökologisch-ökonomischen Rollenspiele mit Begünstigten der namibischen Land Reform haben in erster Linie ergeben, dass diese ihre Entscheidungen im Bezug auf jährliche Anpassungen von Viehzahlen weniger an Umweltvariablen, als vielmehr an Kapitalvariablen ausgerichtet haben. Folglich verfolgen diese Farmer keine opportunistischen Beweidungsstrategien, bei welchen die Viehdichten an die jeweils verfügbare Grasbiomasse angepasst werden, sondern wenden vielmehr konservative, niedrige Viehdichten an. Dies impliziert, dass die Farmer, in Anbetracht Ihrer derzeitigen ökonomischen Situation, nicht *per se* durch Überbeweidung zur (weiteren) Verbuschung und Degradierung der Savannenvegetation beitragen. Nichts desto trotz lässt sich dieser Umstand nicht als Indikator für den Erfolg der Landreform in Namibia werten, da die Landnutzungsstrategie hauptsächlich auf der prekären ökonomischen Situation der Landbesitzer zu beruhen scheint. Im Gegenteil, meine Ergebnisse deuten an, dass die Farmer sich schwer tun, überhaupt Gewinne mit der Viehwirtschaft zu erwirtschaften. Im

Rahmen dieser Rollenspiele konnte außerdem gezeigt werden, dass die obligatorische Zusammenarbeit zwischen Farmern im Bezug auf den gemeinschaftlichen Betrieb von Wasserinfrastrukturen nicht optimal verläuft, wodurch ihnen zusätzliche Kosten entstehen.

Die öko-hydrologischen Simulationen lassen den Schluss zu, dass die Berücksichtigung der Demographie von Bäumen und Sträuchern und hier insbesondere der Engpass bei der Keimung und Keimlingsetablierung, eine grundlegende Voraussetzung für ein umfassendes Verständnis der Vegetationsdynamik in semiariden Savannen und deren nachhaltige Nutzung ist. In diesem Zusammenhang werbe ich auch insbesondere für einen gezielten Einsatz von Feuer zur Vermeidung von Verbuschung und Degradation. Schließlich – auf Grundlage der Gesamtheit der Ergebnisse meiner Dissertation – leitet sich die Empfehlung ab, bei der Nutzung semi-arider Savannen für die extensive Viehhaltung auf konservative Beweidungsstrategien zu setzen. Der Hauptgrund für diese Empfehlung besteht in der Tatsache, dass solche vorausschauenden und nicht-opportunistischen Strategien finanzielle aber auch ökologische Risiken puffern können. Desweiteren fördert eine solche Strategie aufgrund erhöhter Grasbiomasseverfügbarkeit die erfolgreiche Anwendung von Feuern. Außerdem entsprechen konstante Herdengrößen den Bedürfnissen von Landnutzern, welche in den seltensten Fällen in der Lage sind, die durch opportunistisches Herdenmanagement entstehenden Schwankungen von Kosten und Einkünften auf sich zu nehmen.

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