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Cover photograph: A farmer and a researcher walking and talking in the veld in the northern Succulent Karoo, South Africa.
Photo: Imke Oncken, Hamburg/Germany.
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Article IV.2
– Author's copy –

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Bush encroached rangeland in the Thornbush Savanna. Photo: N. Dreber.

Part IV

IV.2 Impacts of landuse and climate change on the dynamics and biodiversity in the Thornbush Savanna Biome

Impacts of landuse and climate change on the dynamics and biodiversity in the Thornbush Savanna Biome

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Summary: In this chapter we summarise results of the long-term interdisciplinary BIOTA study investigating the impacts of landuse and climate change on the dynamics and sustainable use of biodiversity in the Thornbush Savanna Biome of southern Africa. This study successfully integrated remote sensing methods, eco-hydrological approaches, and ecological research with socio-economic aspects by combining empirical and modelling approaches. Based on a summary of current knowledge and key challenges, we describe in-depth analyses of the causes and mechanisms of vegetation dynamics in the focal thornbush savannas and the consequences of the resulting vegetation patterns for animal diversity and a single medicinal plant species. These analyses provided risk assessments of vegetation and animal diversity based on future environmental scenarios, and enabled the formulation of recommendations for landusers and conservation planners and the identification of future research needs.

2.1 Introduction: current knowledge and key challenges

[F. Jeltsch, N. Blaum, E. Rossmanith & D. Lohmann]

According to the Millennium Ecosystem Assessment (2005) the greatest pressure on dryland ecosystem services takes place in regions of intermediate aridity and not, as might be expected, in dry subhumid areas where population density is highest, or in hyperarid areas, where population is lowest. The high overexploitation of services is inferred by physical, biological, and social phenomena—soil erosion, reduced biodiversity and biological productivity, and reduced income expressed by reduced human well-being—and it is reflected by the highest rate of infant mortality and hunger among children (Millennium Ecosystem Assessment 2005: chapter 22). This and the fact that (i) drylands cover about 41% of Earth's land surface and are inhabited by more than 2 billion people (about one third of the world's population) and (ii) dryland populations on average lag far behind the rest of the world in terms of human well-being and development indicators (Millennium Ecosystem Assessment 2005), emphasises the need to focus dryland research on regions of intermediate aridity.

Arid and semi-arid Thornbush Savannas of south-western Africa are such hot spots with regard to the relevance of biodiversity, ecosystem processes and services for current and future livelihood of millions of households (Ericson & Watson 2009). They can function as a model-region for a broad range of drylands, especially non-humid savannas, worldwide—in particular as projected climatic changes for most subhumid and semiarid savanna regions indicate major shifts in precipitation towards drier conditions (Solomon et al. 2007). Therefore, research findings especially for the drier end of the savanna continuum are likely to become relevant for areas that are currently more humid.

Previous research in the Thornbush Savanna regions of south-western Africa has shown that management strategies related to current landuse (e.g. extensive livestock production) can be problematic in this fragile dryland ecosystem (e.g. Jeltsch et al. 1997a, b, Weber & Jeltsch 2000, Weber et al. 2000, Popp et al. 2009a, b). In Namibia alone about 50% of the savanna rangeland is purportedly affected (Bester 1998). Projected consequences of the current landuse strategies include desertification with increasing erosion risks in the drier savanna parts and intensified bush encroachment (i.e. the increase of un- or less palatable woody species at the cost of palatable herbaceous vegetation) in the more mesic parts. These problems are aggravated by climate change and rapid population growth. Current climate projections predict increasing temperatures, decreasing precipitation, shorter rainy seasons, and increasing intensity of rain events (Solomon et al. 2007). Desertification and degradation caused by ill-adapted land management or/and climate change are accompanied by biodiversity losses (e.g. Blaum et al. 2007a, b, c, 2009a, b, Wasiolka et al. 2010) adding further risks to ecosystem functioning and stability. Biodiversity plays a crucial role in the provision of dryland savanna ecosystem services. This includes single key species that are directly involved in the provision of a range of ecosystem services (e.g. *Acacia erioloba* in the southern Kalahari providing fuelwood, forage and food, and supporting soil development and conservation (Dean et al. 1999, see also Article III.7.2), or plant communities providing a

package of services through their ground cover and structure (e.g. water regulation and soil conservation as well as forage and fuelwood provision and climate change resilience) (Millennium Ecosystem Assessment 2005: chapter 22). The biodiversity of "vegetation cover" and biological soil crusts is further linked to a diversity of arthropod species that process most of the living plant biomass, constituting the first link of nutrient cycling. Also, the replacement of the herbaceous vegetation by encroaching bush or the reduction of the litter-decomposing termite populations can impair nutrient cycling, primary production, and carbon sequestration (Whitford & Parker 1989, Zeidler et al. 2002).

Clearly, the change in the ecological environment causes economic problems as landusers depend crucially on natural resources. Key ecological and economic services within the Thornbush Savannas of south-western Africa that are at risk include livestock, game and woodfuel production. The latter service is of high relevance since more than 90% of rural households in southern Africa are still dependent on fire wood for cooking and the average household uses up to 3 tons of wood per year of which the majority is harvested from live trees (Kojwang 2000).

In this chapter we will summarise and discuss key results and their implications of almost one decade of interdisciplinary research in the arid to semi-arid Thornbush Savannas of south-western Africa.

2.2 "When is a Change a Change?"

[M. Lück-Vogel & M. Strohbach]

Background

Savannas are described as semiarid transition zones between tropical forests and deserts (Mistry 2000). This definition implies the dynamic character of the "ecotone named Savanna", where the more humid tropical and more arid desert conditions can vary tremendously in space and time. In fact, the last nine years of interdisciplinary BIOTA work have demonstrated a high degree of variability in the dynamics of savanna vegetation at a range of spatial and temporal scales. One

of the major questions several BIOTA experts dealt with was to assess the relevant changes, namely degradation processes, during the period of observation. However, it turned out that the accurate distinction between non-reversible degradation processes (directed change) and the system-inherent interannual variability (non-directed change) is a challenge on its own.

Vegetation surveys

Phytosociological surveys were conducted to better define the Thornbush Savanna vegetation in Namibia. The distributions of individual plant communities within a vegetation type were strongly related to soil characteristics. On the other hand, the individual sub-types of plant communities often showed variations of community structure and composition that could broadly be related to either landuse impacts or rainfall regime at the time of survey. In addition to the surveys, several permanent sites were used to monitor vegetation at regular intervals to gain more insight into the dynamics of the vegetation.

Namibian vegetation scientists have not yet developed a plant indicator system analogous to Ellenberg's indicator value system, which has been used for decades to evaluate natural site conditions and potential alterations thereof due to different landuse practices on Middle Europe's vegetation communities. However, there is already a fair understanding of which perennial (and some annual) grasses and shrubs indicate relatively sustainable or unsustainable rangeland management, although there is still much uncertainty about the ideal species composition that would indicate a highly productive, intact rangeland. This is further complicated by the dynamic nature of the herb layer. In particular the annual species are highly dynamic as rainfall regimes change annually, not only in response to the total amount of rain received, but also in terms of how that rainfall is distributed over the growing season. This observation was confirmed by the analysis of monthly rainfall data derived from the Climatic Research Unit (CRU; http://www.cru.uea.ac.uk; Hulme 1992) and vegetation development on several farms in the study area. In this analysis rainfall data were overlaid with remotely sensed NDVI

data (Normalised Difference Vegetation Index) derived from NOAA-AVHRR and MODIS sensors, which provides an indication of the intensity of photosynthetic activity per area. The close relationship between rainfall and NDVI (Fig. 1) clearly shows that the vegetation responds within only a few weeks after significant rainfall events have occurred.

Furthermore, the interannual variability of plant species composition at fixed monitoring sites was recorded. The results showed that sites, which have been managed sustainably for decades might on occasion also be dominated by so-called degradation species in some years. This phenomenon is most prevalent when rainfall is distributed predominantly during the very early or very late growing season (but not equally throughout). Under these circumstances, these ruderal species are then better-adapted to germinate and grow quickly, whilst growth in the remainder of the vegetation is considerably repressed due to the otherwise unfavourable rainfall. However, the scenario may change entirely during the following growing season, again depending on rainfall distribution.

Remote Sensing

The high variability of species composition as well as total biomass recorded during the botanical surveys also had severe implications for the remote sensing work conducted during the 9 years of BIOTA research.

Vegetation mapping. One of the major goals of BIOTA was to derive a vegetation map for at least a 60 km wide strip along the BIOTA north-south transect. Using Landsat 7 ETM+ data from the rainy season 1999–2000, the vegetation units derived by the plant surveys on the Observatories and beyond served as ground truth information. The resulting vegetation map reveals the small-scale, spatial mosaic of the different vegetation units (Fig. 2). In an attempt to monitor the vegetation development over time, the classification scheme developed for imagery in one year was applied to satellite imagery of the following year. This was made possible through the use of ATCOR 2 and ATCOR 3 software, which used to radiometrically normalise the imagery beforehand (Richter 2009).

monthly NDVI maxima

monthly precipitation sum (mm)

Month

Legend:
- Precipitation (CRU)
- Precipitation (farmers' data)
- NDVI (NOAA-AVHRR)
- NDVI (Terra-MODIS)

Landsat acquisition date: 14.6.84 14.5.90 26.4.95 17.5.00 16.2.02 24.4.03
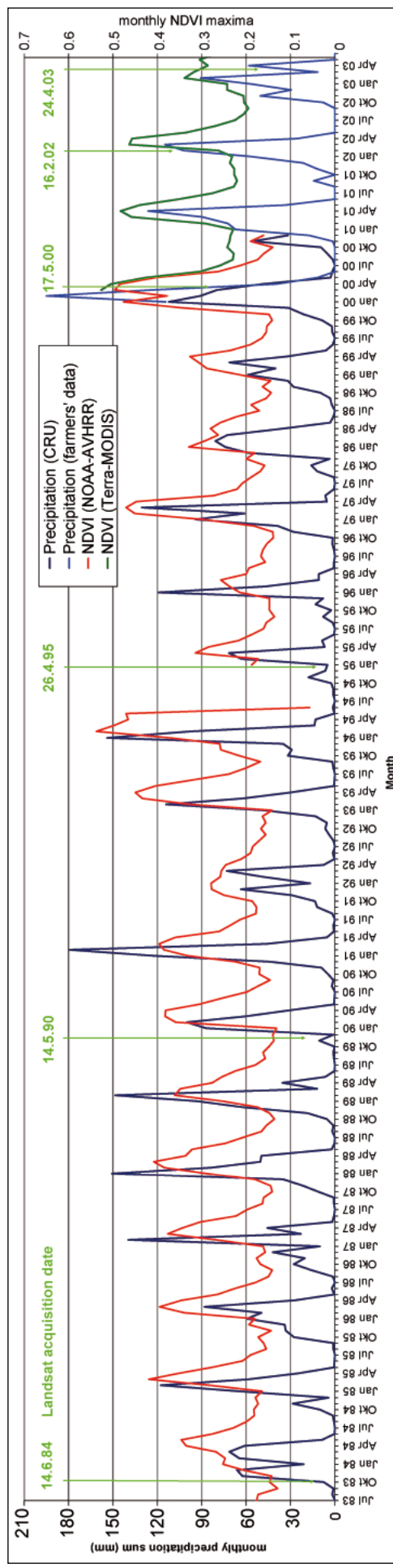
Fig. 1: Plot of monthly rainfall and NDVI development for central Namibia (area between Windhoek and Otjiwarongo) for the years 1983–2003. Green arrows indicate acquisition dates of Landsat TM imagery that was used for regional monitoring of vegetation dynamics.

Camelthorn Savanna
Acacia erioloba - Stipagrostis uniplumis low semi-open bushland

Thornbush Shrubland
Acacia mellifera - Leucosphaera bainesii low-closed shrubland
Acacia mellifera - Stipagrostis uniplumis low semi-open bushland
Acacia mellifera - Monechma genistifolium low semi-open bushland

others
Boscia foetida - Leucosphaera bainesii low semi-open bushland
Catophractes alexandri - Willkommia sarmentosa tall sparse shrubland
Commiphora tenuipetiolata - Grewia villosa short semi-closed woodland
Boscia albitrunca - Eragrostis cylindriflora low open woodland
mountainous areas, unclassified

Otjiwarongo
Windhoek

N   20 km   400 km

Fig. 2: Vegetation map for the area between Okahandja and Otjiwarongo, derived from Landsat 7 ETM+ imagery with a supervised Maximum Likelihood classification. Two white circles in left image centre: masked Omatako Mountains. Source: Strohbach (2002).

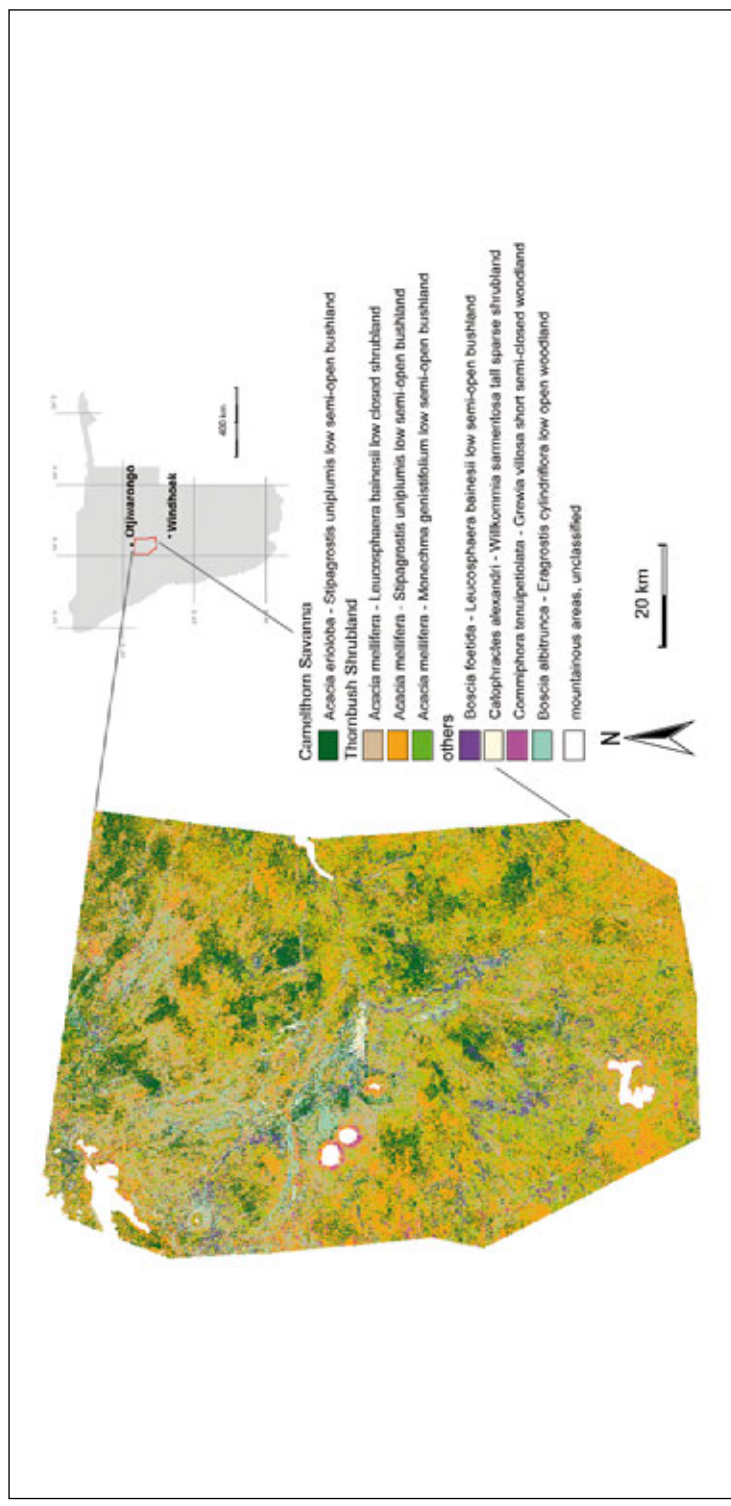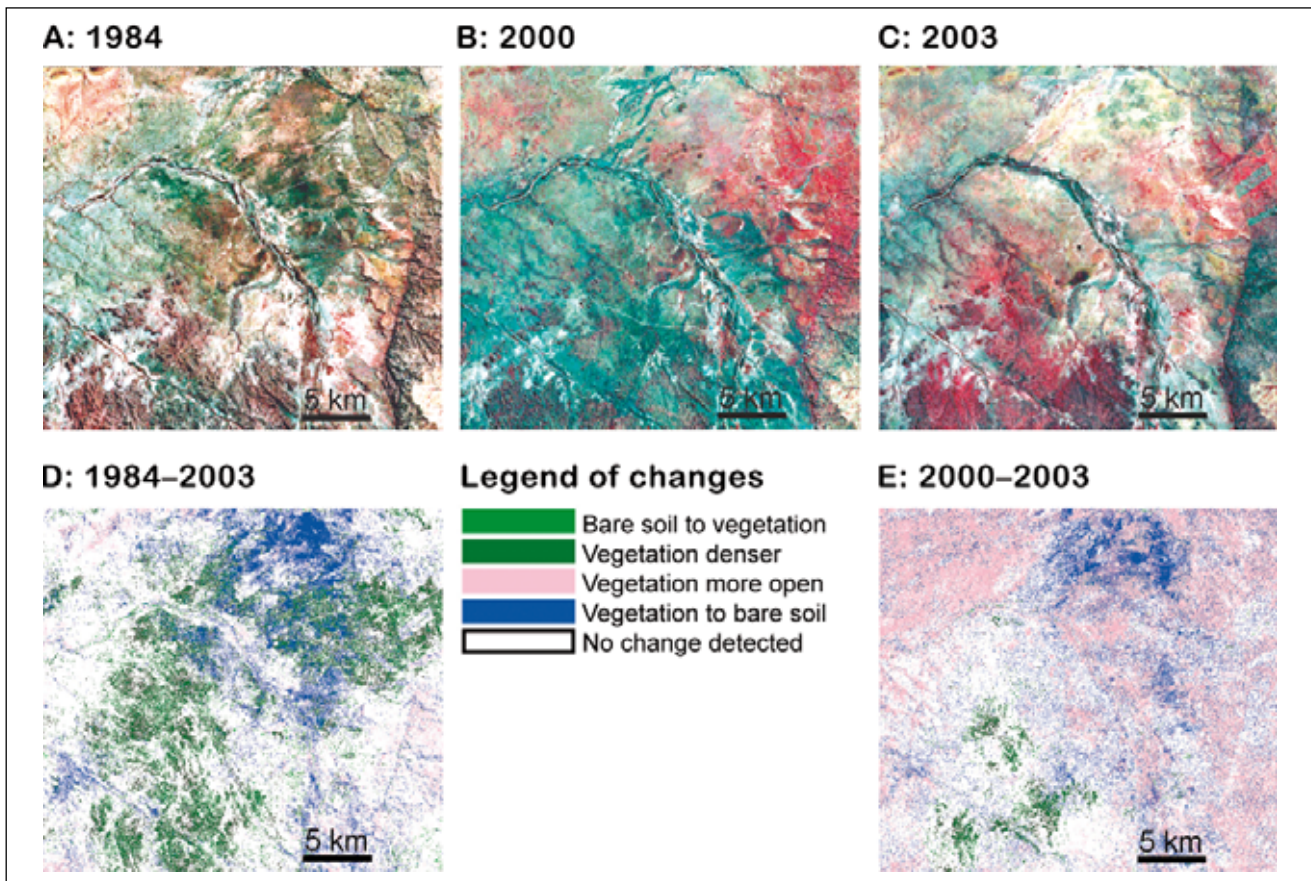Fig. 3: Detected vegetation change for a Landsat TM subset of the area between Okahandja and Hochfeld. The image acquisition dates and rainfall history of the study area can be seen in Fig. 1. The displayed Landsat band combination for A to C is as follows: RGB: b3-b5-b4 (red-MIR-NIR). With this combination dense vegetation appears red, open-dry vegetation appears green and bare soil appears white. The rainfall history of the 1984 and 2003 images was comparable (366 mm vs. 305 mm until respective image acquisition date), but 2000 was significantly wetter (710 mm) than 2003. Thus, the shift from vegetation to bare soil evident in E above should be understood as a non-permanent, phenological response to the lower rainfall in 2003, whereas the detected increase in denser vegetation is probably related to a permanent increase in woody plants (i.e. an increase in bush encroachment). The increase in vegetation in D most likely is related to bush encroachment. Source of Landsat TM imagery: USGS.

However, the vegetation boundaries in the areas, which were compared over two years varied significantly. This led us to ask the question: "Do these changes reflect a permanent change in the vegetation on the ground?" Frequently an analysis of the rainfall data or recent landuse history (from asking the farmers) showed that the changes were not permanent changes but either reflected drought/non-drought related conditions or the impact of recent grazing events, which were superimposed on the vegetation characteristics of the study area.

Significant change processes. In order to identify the relatively permanent, directional changes that have occurred in the vegetation of the central Namibian savanna region, a Landsat TM based change analysis was conducted for the period from 1984–2003 (Vogel 2006, Vogel &

Strohbach 2009). Farmers confirmed that bush encroachment, bush dieback, fires, clear cutting of bushes, and grazing impact are the most important factors leading to changes in the relative composition of woody and herbaceous species. We wanted to know if bush encroachment was indeed a significant phenomenon in this region since this process has been described for other savanna regions in southern Africa and in many other parts of the world (e.g. Sefe et al. 1996). Bush encroachment is a severe problem, as the increase in frequently dense and monospecific woody plant communities often leads to a reduction in plant and animal diversity. Furthermore, bush encroached regions suffer significant loss in economic value, e.g. for livestock farming (de Klerk 2004). Because a vegetation unit-based approach appeared not to be suitable (see

vegetation mapping challenges described above), a more robust change detection approach was developed, using a series of radiometrically-normalised multispectral Landsat TM data. The result was a map of areas where complete vegetation loss (related to "desertification"), recovery of vegetation on bare soil (restoration/rehabilitation), moderate vegetation increase (densification of woody cover: bush encroachment) and moderate decrease of vegetation (bush dieback presumably as a result of a fungal disease) were detected (Vogel 2006) (Fig. 3).

The resulting series for changes between the years 1984–1990, 1990–1995, 1995–2000, 2000–2002, 2002–2003, and 1984–2003 confirmed the high inter-annual and small-scale spatial variability in the density of the vegetation in the savanna vegetation of that region (Fig. 3). Because

of this variability, it was again difficult to detect permanent and significant changes in the vegetation. While there was a close relationship between rainfall history and detected change in vegetation density (see also Fig. 1) in only a few cases, could the change in vegetation cover or density be confidently related directly to bush encroachment or desertification. Most of the detected changes had to be interpreted as phenology-related, non-significant variations in vegetation cover. It was only in areas where the observed changes in vegetation phenology differed from what was expected under a particular rainfall history that a directional change in the vegetation could be inferred with any degree of certainty.

So what are the lessons learnt?

Firstly, we suggest that we need to abandon the idea of the savanna as a mono-static constant system. Secondly, in semi-arid systems such as the observed Thornbush Savannas, we suggest that rainfall is the limiting factor for vegetation development and even single rainfall events can result in a significant response in the vegetation. Therefore we need to accept that vegetation density and also the species composition may vary considerable in space and time (even annually) in response to rainfall.

This strong phenological response to rainfall is detectable with a variety of satellite imagery instruments and processes (see also Wessels et al. 2004, 2007). However, when it comes to the detection of relevant change processes, it is frequently those ('irrelevant') phenological responses, which are superimposed on the 'true' signal of change. Currently the only way to correct for this "noise" in the phenological response of the vegetation is the interpretation of the detected changes with rainfall data. But even this approach has its shortcomings as we are not able to accurately distinguish, for example, between bush encroachment and rainfall-related vegetation increase when comparing a drier year with a wetter year in a multispectral image series.

From the perspective of remote sensing, research needs to explore the use of spatially and spectrally higher resolved data. The analysis of (mainly greyscale) histori-

cal aerial photos can be useful, too, as the high spatial resolution principally allows for the identification of single tree and shrub canopies. However, the technical challenges of aerial photo processing and normalisation currently impede the application of such an approach for large areas.

For the future, monitoring systems based on high resolution multispectral satellite sensors such as Quickbird, RapidEye, and aerial multispectral and hyperspectral data could be developed. The identification of single tree and bush canopies or the distinction of characteristic spectral signatures of vegetation species should allow for the distinction between grasses and woody species. Another option is the further exploration of radar and Lidar sensors, which allow for the assessment of structural landcover characteristics, which overcomes the limitations of spectral approaches.

Several BIOTA teams tried to assess bush encroachment using field observations, field experiments and different remote sensing approaches (see e.g. Subchapter IV.2.7 as well as Articles III.2.3 or III.2.4). However, the data and image time series analysed during the last nine years still do not provide a clear picture of possible long term trends for larger regions. The analysis of long term rainfall data, however, shows a periodicity of about 10–11 years for Namibia of dry and wet spells (Engert & Jury 1997).

In order to determine the long-term trend in vegetation composition a data series of at least several decades would be necessary. Even if we have access to some valuable historical rainfall records from some farms in the region, the currently available satellite archives of about 30 years are still insufficient to establish significant long-term trends. Even worse is the situation with regard to vegetation monitoring data, as most regions were only surveyed for the first time during the BIOTA project. In the absence of any historical survey data, the regular surveys conducted during the BIOTA period have provided an important first step in the development of a long term monitoring data base.

Finally, the project also contributed significantly to our understanding of savanna dynamics and the factors, which drive them. It pointed us towards many

challenges in savanna research that still need to be resolved and will helped us to formulate the appropriate research questions for projects to come.

## 2.3 Eco-hydrology of Thornbush Savanna

Introduction

As shown in the previous subchapter, vegetation dynamics is largely driven by the variability of precipitation. However, the amount of rainfall that becomes available to the vegetation as soil moisture also depends on the soil structure and type, the topography of the landscape and the vegetation cover itself. Vegetation cover and composition strongly influence rainfall interception, infiltration, run-off, and water losses through evapotranspiration. Understanding the complex interaction of soil, topography, vegetation, and soil moisture is an interdisciplinary key challenge for empiricists and modellers. In the following, we present a field study on the small-scale factors controlling local soil water budget and a set of modelling studies that explore the dynamic interaction of savanna vegetation and soil moisture driven by current and predicted future climate.

Empirical findings: soil water balance at savanna sites

[N. Classen, A. Gröngröft & A. Eschenbach]

On the Farm Otjiamongombe (Erichsfelde), about 110 km north of Windhoek in the central Namibian Thornbush Savanna, four typical sites were selected and detailed studies of in-situ water balances conducted in the years 2007 to 2010. The studies aimed at a quantification of controlling factors for the local soil water budget, especially with regard to the varying soil properties, the patchy vegetation, and the occurrence of physical or biological surface crusts. Here, for one site and season, the results will be summarised and some general conclusions drawn.

Methods. On each site, along a 30 m transect, soil water probes were installed at characteristic surface situations (e.g. canopy of small *Acacia* trees, *Stipagrostis* tussocks, open soil patches). Addi-
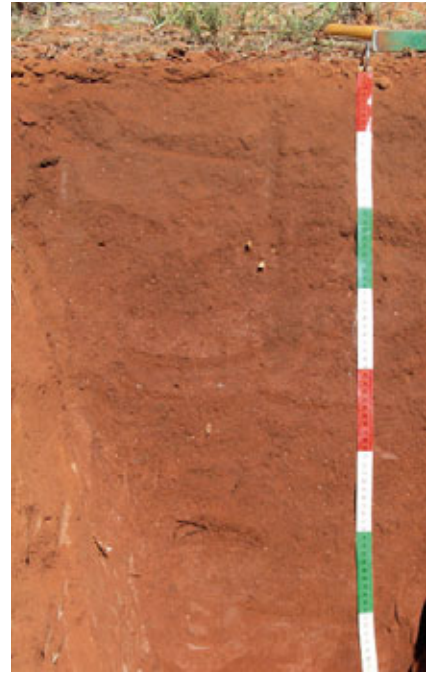
Photos 1 and 2: Study site at the Farm Otjiamongombe (Erichsfelde), general view of the site (left) and soil profile (right).

tionally, at two positions (*Acacia* canopy – open grassy) sensors were installed in the soils at four depths (20, 40, 60, and 80 cm) and connected to loggers for automatic reading. The device consisted of i) sensors for measuring soil water content and temperature (Easytest, Institute of Agrophysics, Lublin, Poland), ii) gypsum probes for the measurement of water tension (Watermark, Irrometer company, Riverside, USA), limited to tensions up to 0.2 Mpa), and iii) climate sensors for the reading of precipitation (Raingauge, Mike Cotton Systems, South Africa), and air temperature (Handylog, Driesen & Kern, Germany). To determine the soil water content in the areas between the sensors the readings were interpolated and the layer above the upper sensor was regarded as of constant moisture. In case of data loss (e.g. by termite destruction of sensor cable), the water retention curves were used to calculate missing values.

Results. In Fig. 4, the precipitation and the course of soil water pressure is depicted for two neighbouring positions on one site and the rainy season 2008/2009. Soil at this site was classified as a Haplic Luvisol (see Table 1). The vegetation is dominated by patches of *Acacias* (*A. mellifera* and *A. tortilis* with a height up to 5 m) and of dwarfshrubs (*Monechma genistifolium*)

(Photos 1 & 2). Protected through the *Acacia* canopies, a dense layer with single grasses is developed, which forms dense stands at the end of the rainy season. The both profiles have a distance of 10 m.

The data indicate:
- The rainy season was split into two distinct phases: a) the first rain events from the end of November until Mid of December (in total 105 mm on 32 days) and b) a phase from mid of January to begin of March (in total 445 mm on 48 days). While the distribution was typical for the region, the total amount of 558 mm was significantly above the annual average of 396 mm.
- At a depth of 20 cm, the first decrease of soil water tension took place ten days directly after the first rain of 14 mm.

This was only true for the grassy site. Below the *Acacia* canopy, even after the sum of 332 mm (227 mm since January) rain the water tension stayed constantly high.
- A comparison of the water tension between grassy and *Acacia* sites shows significant difference: The soil below the *Acacia* canopy got wet substantially later and dried out earlier.
- For the days without rain and with dry subsoil, the change in water storage could be used to estimate the daily evapotranspiration (ET). For the whole period, ET was summed up to about 375 mm.
- Taking rain (P), ET, I and change in water storage (ΔR) into account, the resulting water balance shows a seasonal surplus of ca. 150 mm.

Table 1: Characteristics of the soil on a Thornbush Savanna site on the Farm Otjiamongombe, about 110 km north of Windhoek, central Namibia

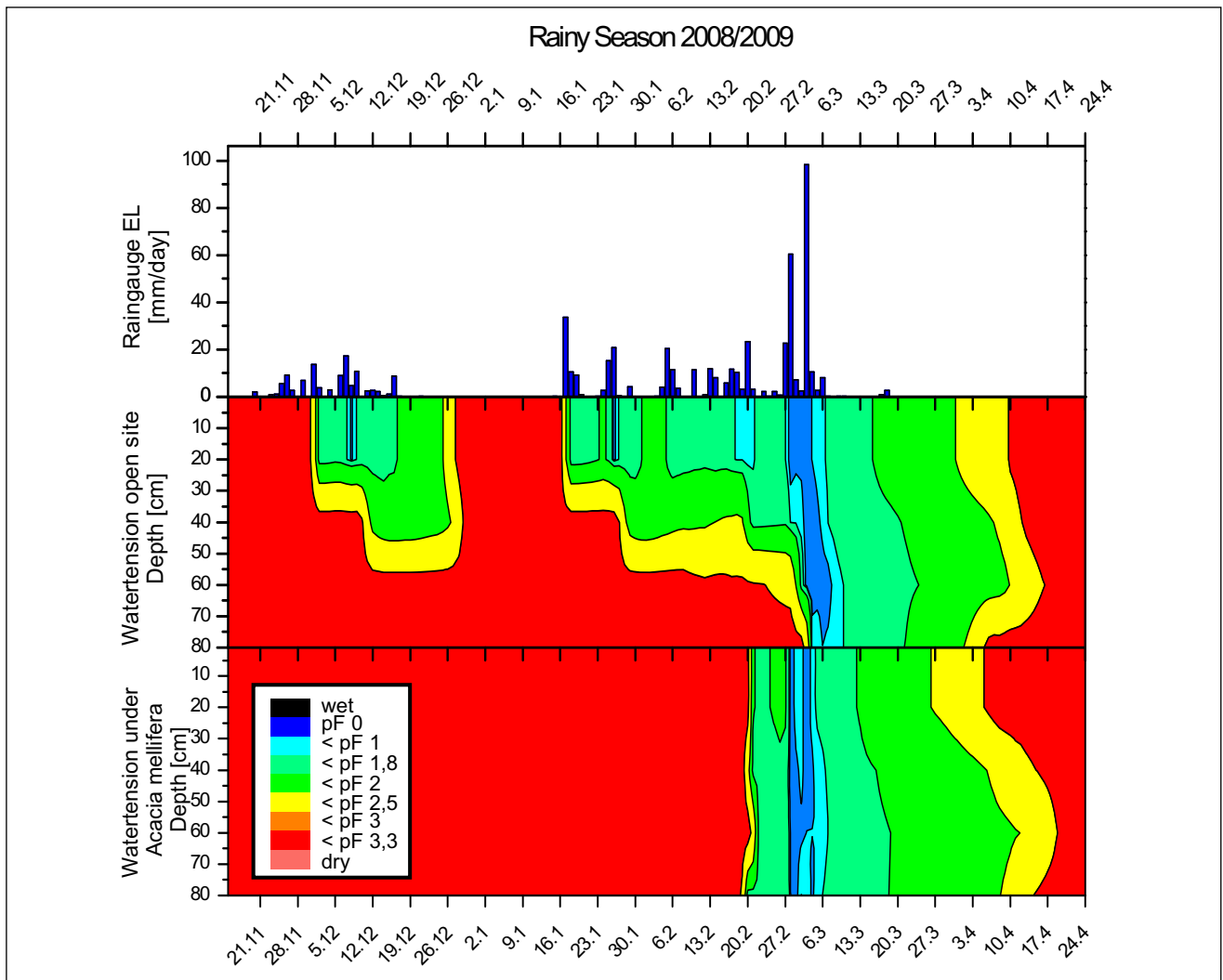| Soil classification (WRB)	Haplic Luvisol, hypereuric
Texture	sandy loam to sandy clay loam
pH-Value [in CaCl$_2$]	5.5–6
EC $_{2,5}$ [µS/cm]	11–22
C$_{org}$ in topsoil [%]	0.350

Fig. 4: Precipitation and water tension at two profiles on loamy soil in Thornbush Savanna. The two profiles are at an open (i.e. grassy site) and an *Acacia* woodland site.

- At the end of the rainy season strong rain events led to subsoil wetting for both sites. At this time, deep drainage was possible. There were no signs of water saturation above the bedrock in this period.
- To maintain the transpirational demand of the vegetation, water storage was needed for 5 weeks after the end of the rainy season.

Conclusions.

- The local water budgets are substantially altered by the vegetation structure and distribution. Here, tree stands lead to canopy interception and thus reduced water infiltration.
- In seasons with sufficient rainfall, it is possible for the deep-draining water to reach the saprolite and the underlying fissures in the bedrock although this

was not recorded in this study. Most likely, part of this water is used by deep rooting trees and thus improves biomass production. It is impossible to quantify this amount of water storage with conventional soil hydrological measurement devices.

- To fulfil the water balance, additional water losses through runoff have to be regarded.

To understand the competition between grasses and trees in mixed savanna sites and thus the drivers of bush encroachment it is necessary to study the hydrological interaction between soils and vegetation in more detail. Especially, the transpirational demand of the grass and tree species under varying soil moisture stress conditions and the redistribution of soil water by some Acacia trees have to be quantified in future. In Namibian soils,

which are characterised by a restricted fine earth cover overlying saprolites and bedrock, the role of the geologic substratum has to be taken into account. Here, the application of sap flow meters, which have been successfully tested on the study site in 2009, promises to be a valuable tool to understand water relations of woody plants in areas with a strongly restricted water supply.

Eco-hydrological modelling and scenario analyses

[F. Jeltsch, M. Wieczorek, S. Meyfarth, N. Blaum, E. Rossmanith, D. Lohmann, A. Popp & B. Tietjen]

Drylands are characterised by a tight coupling of water availability and vegetation dynamics and the risk of deterioration is high due to the extremely variable climatic conditions. Deterioration, often

linked to rangeland overexploitation in combination with unfavourable climatic periods, can either take place in the form of increasing bare patches of soil associated with increasing erosion risks (Kefi et al. 2007) or the fragile equilibrium between woody and herbaceous vegetation can shift towards a shrub-dominated state (Roques et al. 2001). Climate change in drylands is likely to increase these risks of degradation with predictions of decreasing mean annual precipitation (Dore 2005), increasing temperatures and increasing number of extreme events (Easterling et al. 2000, Solomon et al. 2007).

While recent meta-analyses show the principle relationship that exists between woody cover and rainfall in arid and semi-arid savannas with a mean annual precipitation below 650 mm (Sankaran et al. 2005) this relationship cannot easily be extrapolated to the transient dynamics that are to be expected under future climate change conditions. Feedbacks between vegetation cover and soil moisture through infiltration, evapotranspiration, and runoff modification, as well as changes in plant-plant interactions, are likely to lead to complex, non-linear vegetation changes. Computer models, which simulate the key eco-hydrological processes can help to better understand these complex feedbacks and assess risks caused by different climate change scenarios. Fig. 5 shows the principle structure of eco-hydrological models developed and applied in the BIOTA Africa framework.

A key challenge in this eco-hydrological framework is the assessment of the role of different soil types in modifying the effect of climatic changes on vegetation. Using examples of a rainfall gradient in Namibia we compared two Thornbush Savanna sites (Sonop, approx. 495 mm/ year, and Otjiamongombe, approx. 360 mm/ year with an even drier site in the Nama Karoo (Gellap Ost, approx. 150 mm/year) [Note: information on mean annual precipitation (MAP) used for modelling were derived from the Atlas of Namibia, http://www.met.gov.na/programmes/infocom/infocom/atlas.htm]. In a first step, computer simulations were conducted with constant site-

Table 2: Water balance for the 2008/2009 season (15.11.08–22.4.09, Acacia site)

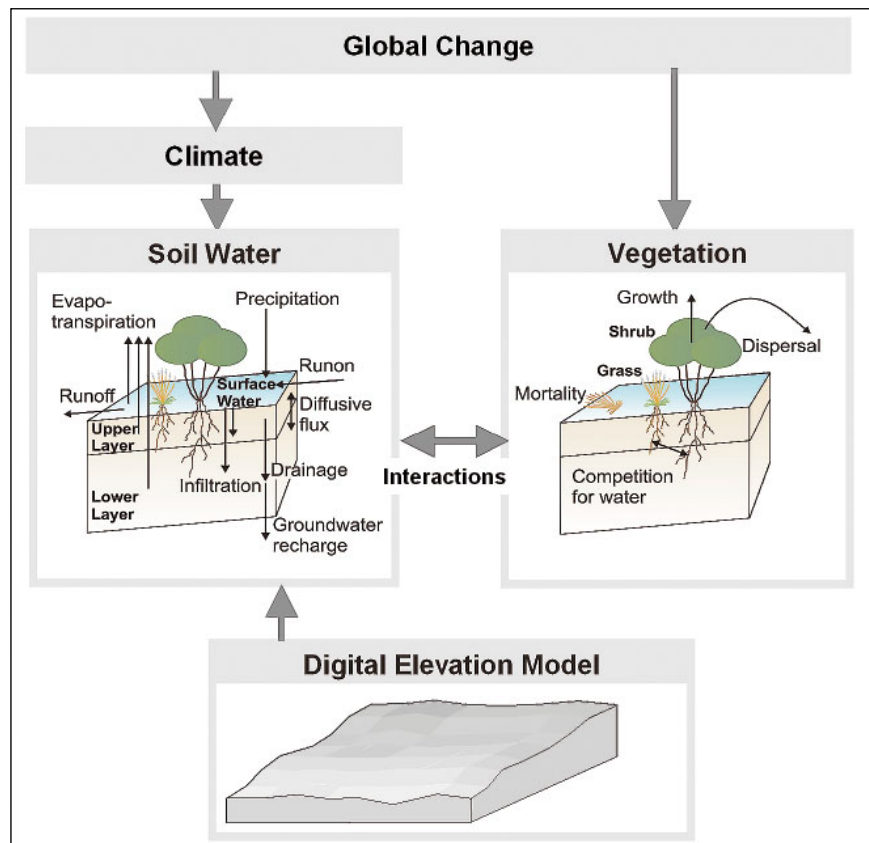| Variable or measurement	Amount (mm)
Precipitation (P)	558,6
Maximum soil water storage (0–80 cm depth)	110,8
Change in water storage (ΔR)	6,8
Evapotranspiration (ET) estimated	374,2
Interception (I) estimated	29,1
Surplus (P-ET-ΔR-I)	148,5

Fig. 5: Schematic description of the eco-hydrological model EcoHyd (Eco-Hydrology in Drylands; Tietjen et al. 2009a, b) dynamically linking vegetation and soil moisture in realistic landscapes under scenarios of climate change. The model accounts for the intensity and duration of single precipitation events and the resulting dynamics of different woody and non-woody plant life forms. The water component of the model simulates the daily dynamics of surface water and water contents in two soil layers. The vegetation elements compete for this soil water and strongly influence hydrological processes.

specific vegetation cover and an identical topography for the three test sites but with variable rainfall conditions to focus on the effects of soil type and precipitation pattern. With regard to the latter, we tested a rainfall time series with an unchanged mean value and a series with a 10% reduction of mean annual precipitation. Further, for both of these variants, we tested the effect of an increase of extreme events, i.e. a reduction of low rainfall events and an increase of the amount of rain falling during high rainfall events without changing the annual mean. As an output measure we used the relative change in number of growth days per rainy season, i.e. the number of days where soil moisture was sufficient for
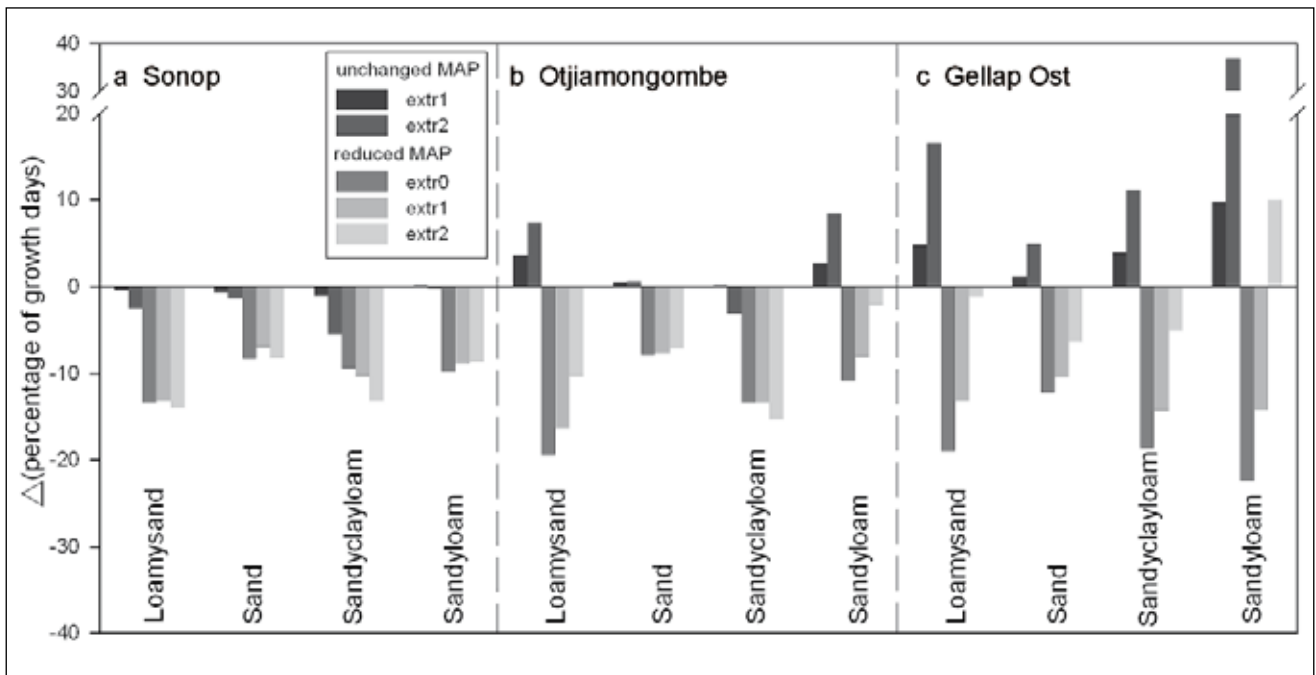
Fig. 6: Effect of different rainfall change scenarios on the number of plant growth days for different soil types at three sites along a rainfall gradient in Namibia (Sonop: approx. 495 mm/year, Otjiamongombe: approx. 360 mm/year, Gellap Ost: approx. 150 mm/year). The columns give the relative difference in number of growth days compared to the unchanged, site-specific rainfall conditions. The left two columns for each soil type are the unchanged MAP scenario but with an increasing shift towards more extreme rainfall events (extr1-2). The three columns on the right for each soil type reflect a 10% reduction in MAP (extr0), plus an increasing shift towards more extreme rainfall events (extr1-2).
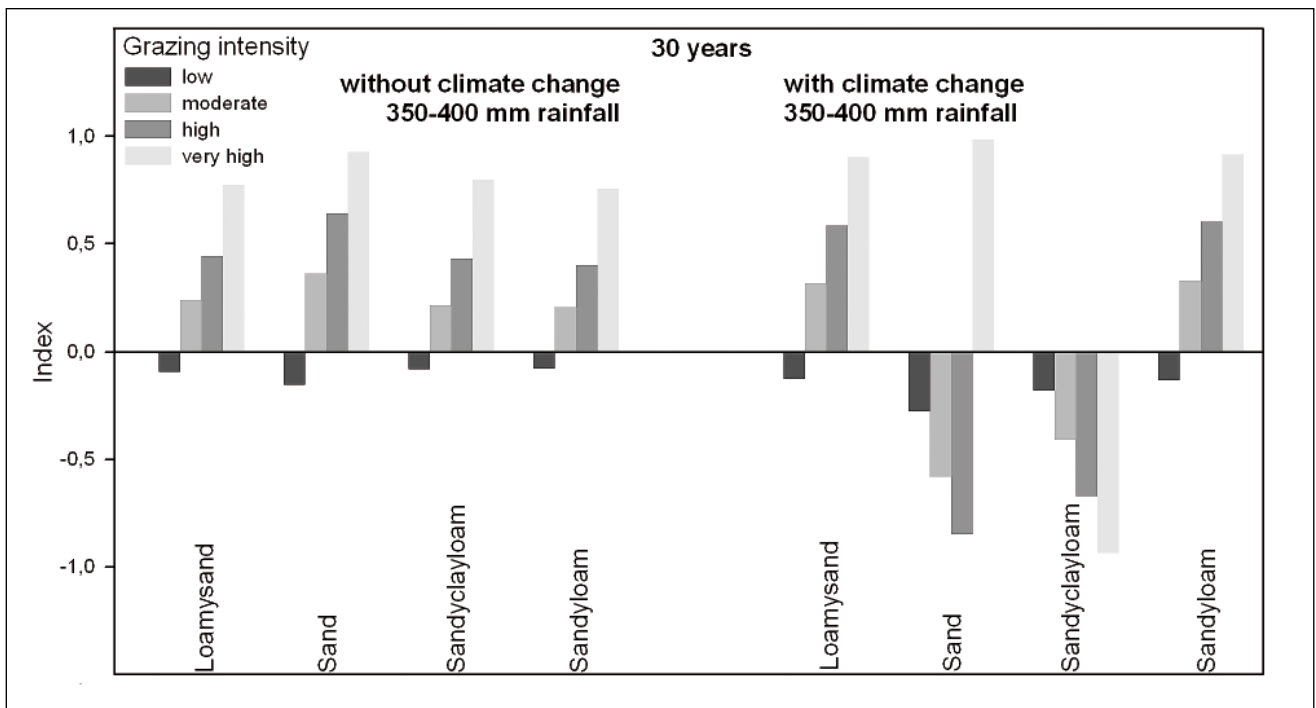

Fig. 7: Simulated vegetation degradation index *deg* (compare Article III.7.2) after 30 years of light, moderate, heavy and very heavy grazing (from left to right) for four different soil types. Left: current climatic conditions for the 'Otjiamongombe' savanna region in Namibia with an average annual rainfall from 350 to 400 mm. Right: climate change scenario without consideration of possible effects caused by CO_2 increases. Note: The absolute value of the index *deg* resembles the relative loss of cover of perennial grasses under different levels of grazing intensity as compared to a situation without grazing. In addition, negative values indicate that the absolute value of shrub cover and the mean increase of shrub cover within the given time frame is less than 5%. This suggests that the risk of soil erosion is more prominent than shrub encroachment, at least if index values are high (otherwise values are positive). Low index values in general indicate a small loss of perennial grass cover and thus a low risk of degradation.

plant growth. Fig. 6 shows that the three test regions show different responses to modified rainfall. As expected, all sites show a strong reduction in growth days with a 10% reduction in annual rainfall. Also, for all sites the specific soil type has a strong influence on the change in growth days. However, while the increase in extreme events has a positive effect for the drier sites it has no or only slightly negative effects for the most mesic site. For the drier sites this finding clearly corresponds to the increased precipitation evident during high rainfall events, which leads, in turn, to soil moisture conditions above a threshold that allows for plant growth. The reduction of low rainfall events has little negative effect since these events are not sufficient to significantly increase the growth period. This is different for the more mesic site, where even the lower rainfall events facilitate plant growth. The reduction of these events at the more mesic site, however, is not fully compensated for by the increase in precipitation of high rainfall events. Again, the specific results for a given site are influenced by the soil type.

Using the example of the 'Otjiamongombe' Thornbush Savanna region with an average rainfall of approximately 360 mm, we next included dynamic vegetation responses (i.e. dynamic vegetation changes as a result of plant-soil moisture interactions) and livestock grazing. Again, model simulations clearly indicate the high relevance of the soil type for climate change effects. Comparing the effect of different grazing intensities on vegetation shows that under current climate conditions different soil types only lead to minor differences in vegetation degradation (Fig. 7). However, under a reasonable scenario of climate change (i.e. 10% reduction of mean annual precipitation and a temperature increase of 2.25°C) simulations show a generally higher risk of losses of perennial herbaceous vegetation cover (i.e. higher absolute values of the index *deg*). Model results further indicate that for two of the four soil categories shrub encroachment accompanying the loss of grass cover gets less important for all but the highest grazing intensity (negative values of the index
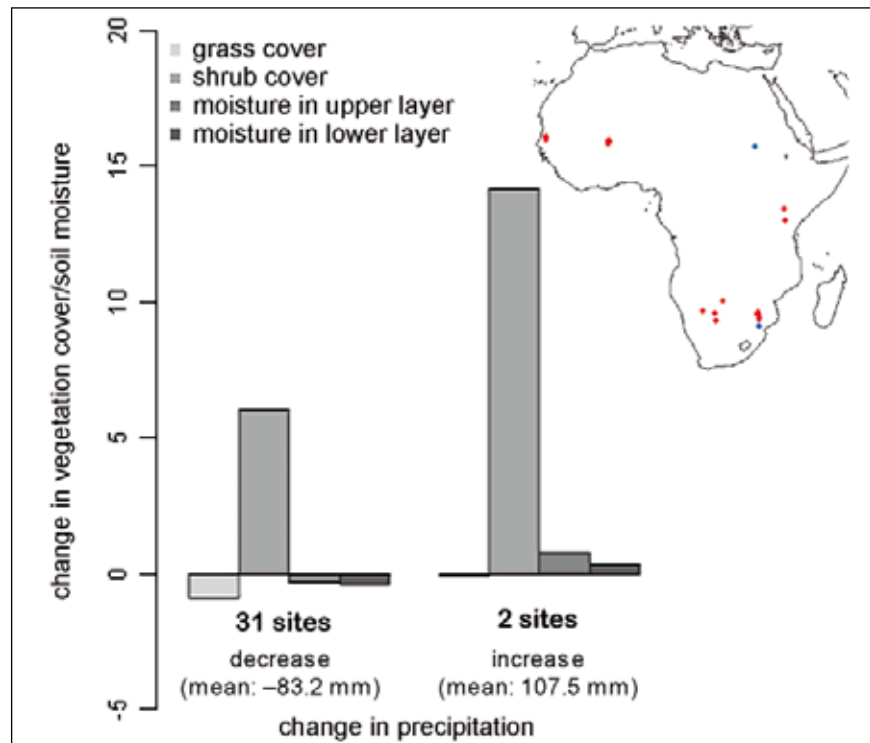

Fig. 8: Simulated combined effects of climate change (based on site specific scenarios of climate change and landuse) and an increase of atmospheric CO_2 on vegetation cover and average soil moisture levels in 33 different arid to semi-arid Thornbush Savanna sites in Africa (see map). Left: mean changes for sites with a predicted decrease in mean annual precipitation (map: red points). Right: sites with a predicted increase in mean annual precipitation (map: blue points). All scenarios are based on IPCC (Solomon et al. 2007) and Sankaran et al. (2005).

deg). As a consequence, the total cover of perennial vegetation is reduced and the risk of soil erosion further increases. Risk assessment maps for the focal Thornbush Savanna region in Namibia under climate change that are based on the described simulation approach and degradation index are given below in Subchapter IV.4.5.

Results of climate change impacts on vegetation presented so far were based on modified rainfall and temperature conditions. A further factor that is assumed to impact vegetation changes (also in savannas) is the increasing level of atmospheric CO_2. Increased atmospheric CO_2 could mitigate the described climate change effects directly through the higher photosynthetic rates of plants, which occur under such conditions, and indirectly by enhancing the water use efficiency (Drake et al. 1997). However, the effects of elevated atmospheric CO_2 can vary strongly between plant types (Ainsworth & Long 2005). The growth rate of juvenile woody spe-

cies with a high carbon demand could be especially sensitive to an increased CO_2 level, which could alter the shrub-grass balance (Bond et al. 2003, Tietjen et al. 2009b).

While effects of altered precipitation levels and temperature on savanna vegetation are relatively well understood the impact of elevated CO_2 generally remains more speculative. Interestingly, including increased growth rates for woody vegetation and enhanced water use efficiency for all vegetation in the described eco-hydrological models leads to a change in the general vegetation response pattern to climate change. Climatic changes alone typically lead to a general loss of perennial vegetation cover and a reduced risk of shrub encroachment. In contrast, the combination of climate change with CO_2 effects leads to an increase in the risk of shrub encroachment for 33 simulated Thornbush Savanna sites across Africa (Fig. 8). This holds both for sites with a predicted decrease and for sites with a predicted increase in mean annual

precipitation. These results clearly show that as long as the effect of increasing CO_2 levels on the complex vegetation-water dynamics are not fully understood any predictions on climate change effects in drylands savannas need to be interpreted with caution.

2.4 Vegetation patterns and biodiversity dynamics

Key vegetation patterns in Thornbush Savanna rangelands

Impacts of rainfall on vegetation cover for two major savanna topsoils in Namibia

[M. Strohbach]

Botanical surveys throughout the Namibian Thornbush Savanna (sensu Giess 1998) revealed a tendency of these landscapes to develop bare patches, ranging from two to four meters to several hundreds of meters in diameter. This problem is most severe on the Etjo Erosion plains. The origins of such bare patches appear to be a combination of natural, erosion related processes, coupled with inappropriate grazing practices. Developing mosaic systems are relatively common in savanna vegetation, but are widely regarded as a form of degradation of a previous continuous vegetation cover (Valentin et al. 1999), which may become more difficult to reverse as rainfall patterns (and other climatic variables) continue to change.

The herbaceous vegetation (i.e. the grass and forb layer) is the most affected by landuse practices as well as the highly variable rainfall experienced in the area. Hence, cover values of this layer show the most dramatic inter-annual fluctuations, making the accurate detection of actual degradation difficult (Wessels et al. 2007, Vogel 2006). To better understand and interpret inter-annual changes in the herbaceous layer, the cover of all these species were documented, together with daily rainfall over a period of four years, at permanent sites on a farm on the western Etjo Erosion plains (21°06' S and 16°36' E, mean annual rainfall: 366 mm).

In this study, the herbaceous vegetation was divided into non-grassy forbs (hl), perennial grasses (gp) and annual grasses (ag). Each of these groups was further divided into grazing response groups, namely Decreaser species (D), Increaser species (IN) and Other, the latter being species whose cover is primarily influenced by rainfall. The change in canopy cover and species number of the herbaceous vegetation growing on different topsoils (on which especially grass species differ due to their ecological preferences) was compared over four years (2005–2008) (Fig. 9).

The sites with loamy sand were grazed by between 30 and 50 medium-frame cattle, rotated on an ad-hoc manner between four camps. During 2006 this was increased to 80 oxen due to the above-average rainfall that season. The sites with sandy loam were grazed by a herd of 100 to 150 Dorper sheep and occasionally a small group of milking cows with their calves (less than ten animals), rotated between four camps, spending a maximum of three weeks in a camp. From June 2006 onwards, the single camp in which the monitoring sites were located was stocked with about 650 Damara sheep, increasing to about 800 when lambs arrived, and rotation was slowed down to about 2.5 month periods per camp. The rainfall recorded during the study ranged from well below to well above average (Fig. 9). The drought during 2006/2007 resulted in the large-scale sale of animals and the flooding of markets country-wide. Because of this, farmers were unable to sell their livestock, and the camps with the monitoring sites could only be de-stocked in December 2007. The camps were left without livestock during 2008.

The results show that on the sites where the topsoil was comprised of loamy sand, perennial grasses were not as severely affected by overgrazing during 2007, as sites with sandy loam and managed to recover to some degree during 2008 (Fig. 9). There was also an almost complete absence of all annual species during the 2007 drought. The sites (and camps) with the sandy loam topsoils, which generally have a lower vegetation cover and often show vast expanses of bare patches, showed a very poor recovery of perennial grasses after the 2006 and 2007 overstocking, despite adequate rains and no

livestock present during 2008. Perennial grasses were instead replaced by a high amount of short-lived annual grasses and an invasive and highly unpalatable forb, *Ondetia linearis*.

Important for management decisions, is the suggestion that palatable, perennial grasses appear to be more abundant and more resilient where topsoils are relatively sandy. It can also be clearly shown that the recovery of perennial grasses, if grazed excessively during a drought period, is not nearly as rapid (within one season), as many farmers would hope, even with sufficient rainfall. This is most evident on the loamier (and hydrologically less favourable) soils. This would imply that erosion of sandy topsoils should be prevented as far as possible, whilst also showing the importance of either reducing stocking rates rapidly during years of drought, or in some way trying to avoid extensive grazing in camps with a high cover of palatable grasses. Most important, however, is the view that long-term stocking rates for an area need to be determined according to average grazing availability during the dormant season, and not during the peak growing season, as is still standard practice with most farmers and agricultural authorities.

The role of mound building termites for the soil structure and nutrient patterns

[C. Grohmann, A. Petersen, T.V. Medinski, A. Mills, A. Gröngröft & K.E. Linsenmair]

Most parts of the Namibian Thornbush Savanna are peppered with termite mounds built by the fungus growing species *Macrotermes michaelseni*. The mounds often occur in high densities. For example, more than four inhabited and uninhabited mounds per hectare were found on the Observatory S04 (Toggekry) (Grohmann et al. 2010). Via the construction of their mounds and the processes within their mounds, termites significantly influence the soil properties and therewith many ecological processes in their habitats. In general, termites modify soil structure and nutrient patterns through at least two processes: i) the allocation of soil material for the construction of mounds and sheetings
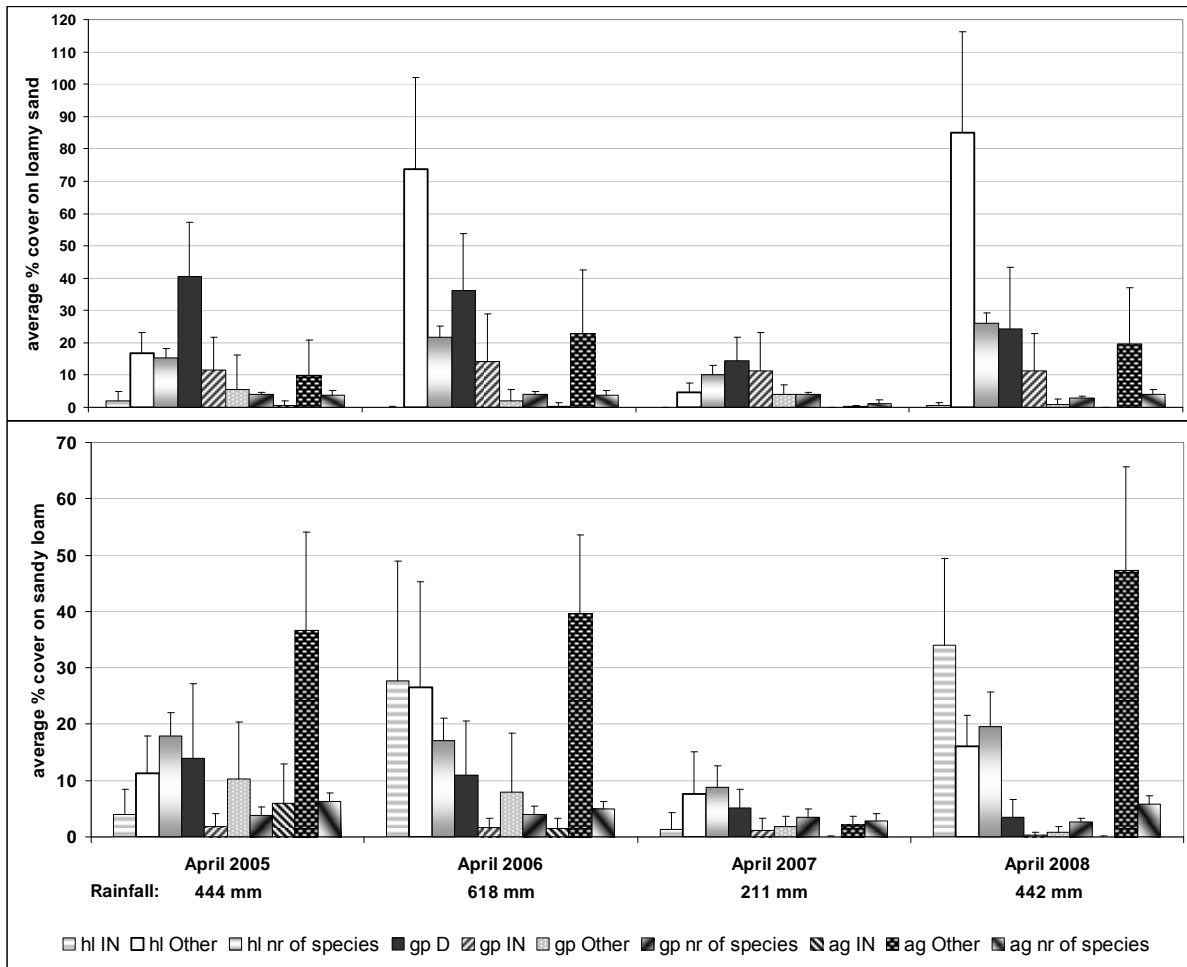
Fig. 9: The change in average canopy cover and species richness (2005–2008) within three groups of herbaceous plants and three grazing response groups growing on loamy sand (top) and sandy loam (bottom) soils on a farm on the western Etjo Erosion plains, Namibia. The three groups of herbaceous plant are (hl) non-grassy forbs, (gp) perennial grasses and (ag) annual grasses while the three grazing response groups are (D) Decreaser species, (IN) Increaser species and (Other) (see text for further details). The rainfall figures reflect annual values for the period May–April of each year. Note the difference in scale of the y-axis in the two graphs.

(soil constructions, which the termites build to protect themselves when they forage) and ii) the transfer of organic material to the fungus gardens and its subsequent mineralisation.

These termite-mediated changes of soil properties have subsequent impacts on plant biomass and vegetation composition. For example, Rogers et al. (1999) found suppressed plant growth on termite mounds, which they attributed to the impenetrability of the mound surface for the roots of the seedling plants. In some regions, mound soils are used as manure (e.g. Nyamapfene 1986). *Macrotermes* mound material is also frequently ingested by wild animals and humans (geophagy), possibly for its nutritional value. Mills et al. (2009) speculated

that mound earths could provide a critical supply of especially scarce nutrients such as selenium, to wildlife and humans alike. They likened the mounds to 'nutrient fountains' within the savannas of Africa and parts of Asia.

The main question of this study was how the structure and nutrient patterns differ between mound-soils of the fungus growing termite, *M. michaelseni*, and the surrounding soils.

For this study, on the BIOTA Observatory S04 (Toggekry) soil samples were taken from the chimneys of 26 inhabited mounds and 26 topsoils (0–10 cm in depth) next to each mound. The total contents of N, K, Ca, Mg, P, and S of the soils were analysed and the logarithms of these values were used for the statistical

analyses. Additional samples were taken from this Observatory as well as from the BIOTA Observatories S05 (Otjiamongombe), S03 (Sonop) and the research farm Omatjenne. These samples were analysed with respect to the exchangeable cations of Ca, Na and Mg, the EC and pH contents, as well as their texture (grain size).

The results show, that the mound soils could clearly be differentiated from their nearby soils with respect to their total N, K, Ca, Mg, P, and S contents (Fig. 10). The difference between the mound soils and the adjacent soils was significant (ANOSIM, $p < 0.001$). Most of the macronutrients (total N, Ca, Mg and S) were enriched in the mounds (Wilcoxon signed rank test, $p < 0.001$). However, P showed
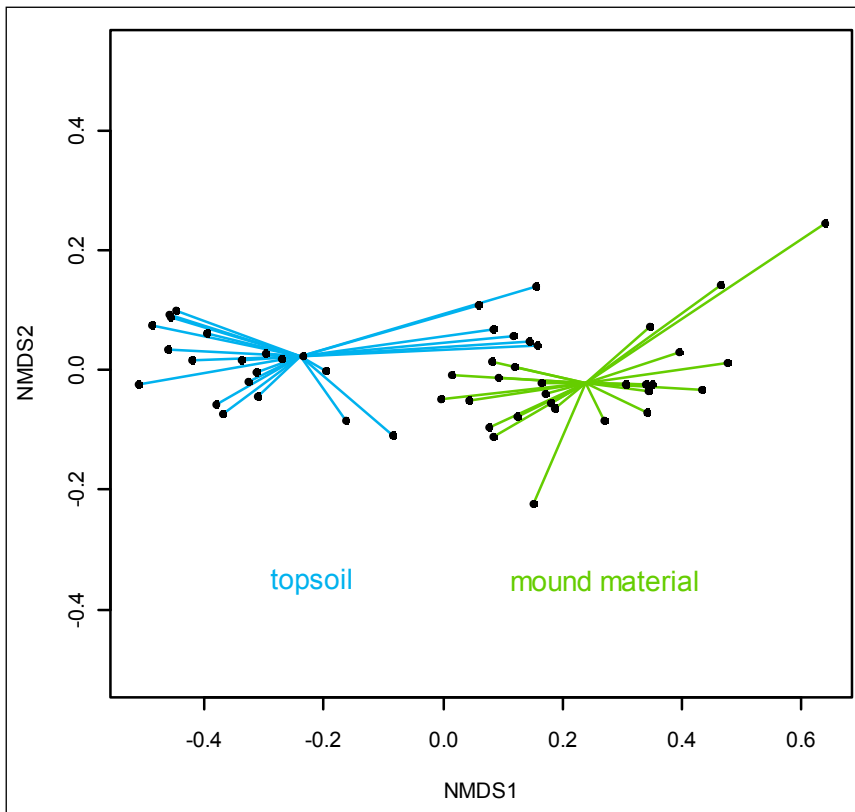
Fig. 10: NMDS-plot of soils from 26 inhabited termite mounds (connected to their group centroid by green lines) and 26 nearby relatively unmodified soils (connected by blue lines), using Euclidean distances. The ordinations arrange the soils (black dots) so that soils, which have similar values of total N, K, Ca, Mg, P, and S contents, are closer to each other compared with dissimilar soils. The plot was rotated so that the variance of sites is maximised on the first dimension. Soil nutrients were added to the plot. Stress: 2.99.

no significant pattern, and K was less concentrated in the mound soils compared to the surrounding soils (Wilcoxon signed rank test, $p < 0.05$). Similar results could be found when analysing the additional samples. Here, higher values of exchangeable Ca, Na and Mg, EC, pH, clay and fine silt contents were found in mounds soils compared to surrounding topsoil.

The enrichment of nutrients within the mound soils is partly caused by the utilisation of subsoil for the building of the mounds and partly by decomposition processes within the fungus gardens of the mounds. The high values of Ca, Mg, and Na can be attributed to the selection of clay-rich subsoil for the mound construction. This clay-rich subsoil exhibits high content of total Ca, Mg and trace elements (Petersen 2008, Mills et al. 2009). The magnitude of clay enrichment of *Macrotermes* mounds from several sites was 4.7 to 6.5 fold compared to adjacent topsoil (Mills et al. 2009).

In contrast, the main source of total N (mainly composed of nitrate) and S in the mounds can be attributed to the highly effective decomposition and mineralisation of organic material by the fungus, which is cultivated by the termites and assists in the digestion of cellulose and other plant material. Through the reconstruction activities of the termites, the N and S were probably distributed from the region of the fungus gardens to the mound chimney. Another path of distribution within the mound might be the indirect transport by water. If nitrate dissolves in water, which then is transported to the chimney, the nitrate can be fixed here by evaporation. Furthermore, at the same time, the leaching of soluble and exchangeable ions is reduced by the lower water permeability of mound walls compared to the adjacent soil (Watson 1976). Another source might be the saliva of the termites with which the soil particles are mixed before they are used in mound building (Holt & Lepage 2000).

It is often assumed that the observed high nutrient contents of the termite mounds have a greater impact on plant biomass than soils from the surrounding areas. However, on the Observatory S04 (Toggekry), the growth of radish and sorghum was reduced on soil from *Macrotermes* mounds compared to soils from the surrounding areas (Graiff 2010). This may have been due to soil water shortages as a result of the high clay contents of the termitaria, or nitrate toxicity or salinity effects. The same effect was found for radish growing on mound material of a different *Macrotermes* species in a study conducted in the framework of BIOTA West in Burkina Faso (Siegle 2008). However, the biomass of herbs and grass growing on termite mounds was much higher compared to the surrounding soil in a year with extraordinary high rainfall, in which the termite mound soils were soaked thoroughly (Graiff 2010). This enhanced growth may have been linked to the greater availability of water in that year as in years with average rainfall, rainwater hardly infiltrates the mound but is mostly lost through run-off processes from the surface of the mound. Other explanations might be reduced nitrate toxicity and salinity effects as a result of soil leaching.

In conclusion these results show that mound-building termites have a strong impact on soil structure and nutrient patterns within the savanna ecosystem. Firstly, through the constant turnover of soil, evident in the massive mounds comprised of nutrient-rich subsoil and by the redistribution of this material via wind and rain, a "rejuvenation" of the soils takes place. In this way, the consequences of the leaching of nutrients from the topsoil to deeper soil layers are reduced by termites through the redistribution of nutrients to the topsoil. Secondly, termites are the main decomposers in arid savannas (Buxton 1981), and play a key role in the humification and mineralisation of plant material.

In addition, termites also influence the water supply of the soils directly, which should be analysed in more detail in the future. The complex underground galleries, which they build to reach their food sources, are important for the vertical and lateral distribution of water in the soils.
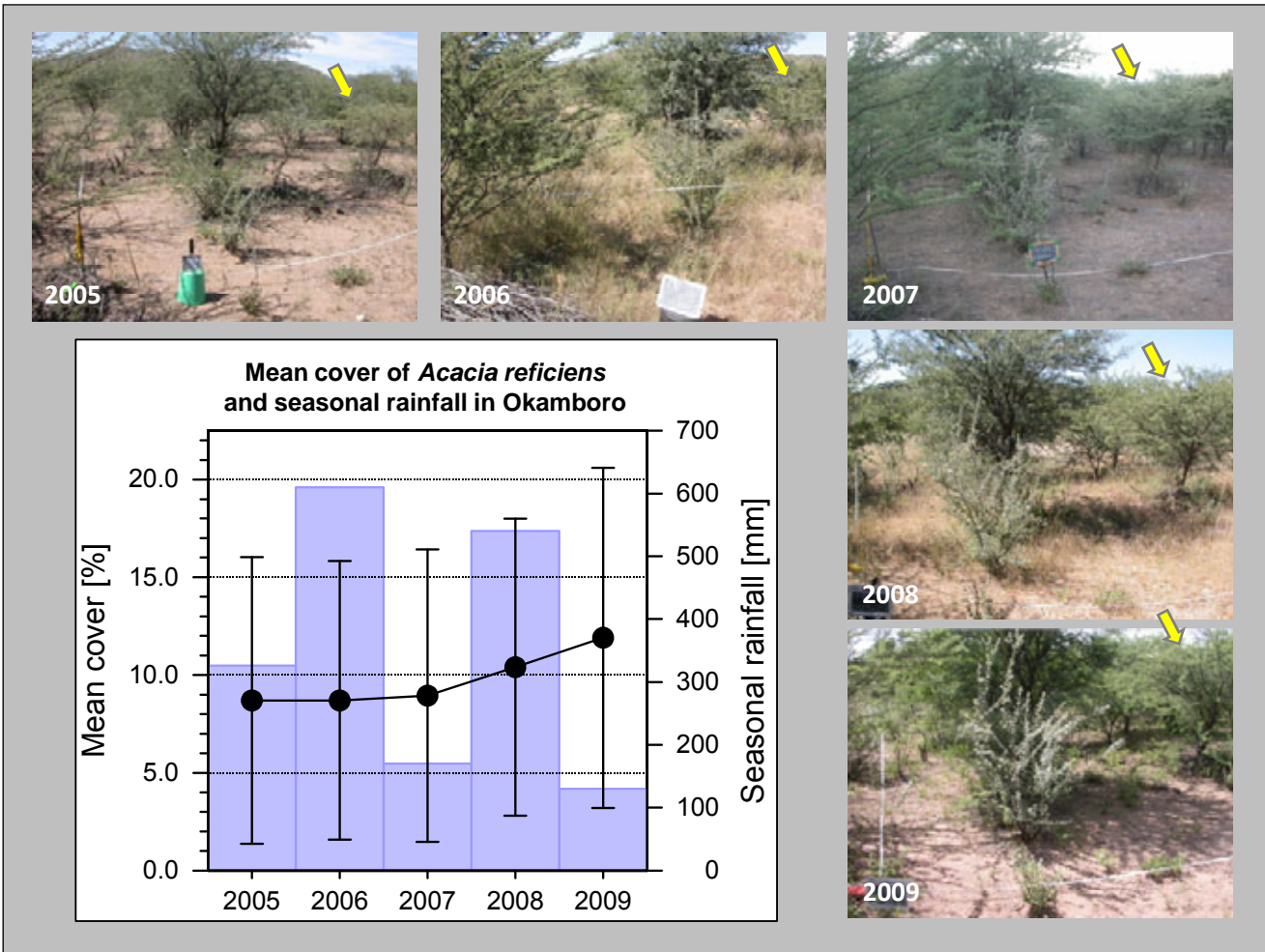
Fig. 11: Mean cover (±std. dev.) of the encroacher species *Acacia reficiens* on 20 permanent monitoring plots (each 1000 m²) of the BIOTA Observatory Okamboro (or Ovitoto, S06) over the period 2005–2009. The fixed point photographs, taken every year in the growing season are used for illustrative purposes to show the changes in vegetation cover and composition over the five-year period within one sampling plot (hectare number 86). Arrows point to the same individual of *A. reficiens* in the respective year.

In particular, it is the openings of the galleries on the soil surface that appear to enhance several fold the infiltration of rainwater into the soil (Léonard & Rajot 2001, Kaiser et al. 2009).

Finally, climate change will have considerable impacts on ecosystems in the future and will affect termites through several key mechanisms. The predicted increase in temperature for central Namibia will affect, for example, the temperature-sensitive system of the fungus garden cultivated by Macrotermitinae. In this context, changes in vegetation composition (e.g. wood to grass ratio) will also affect termite communities and their role in ecosystems. Experiments on the temperature plasticity of the fungi and the required food composition of different termite species will thus be valuable in predicting at least some of the likely consequences of temperature changes on ecosystem functioning.

Snapshots of grazing induced bush encroachment in the central highland of Namibia

[D. Wesuls]

Although bush encroachment due to overgrazing is widespread in Namibian Thornbush Savannas and is considered as a major form of land degradation (de Klerk 2004), the speed at which encroacher species can grow has been rarely documented. One important bush encroacher species in the central Highland of Namibia is *Acacia reficiens* (Red Thorn). An excellent study system to monitor bush encroachment is the Ovitoto area (22.02°S, 17.06°E), situated in the central Highland of Namibia where landuse has changed rapidly during the last decade. The area of the BIOTA Observatory Okamboro (S06) in the Ovitoto area was a former part of a military base and is now used as a communal rangeland. Since 2001 new settlements have been built close to the Observatory leading to an intensified landuse and, as a consequence thereof, the number of free ranging cattle increased.

In 2005, 20 permanent study plots of 1,000 m² each were established on the BIOTA Observatory to monitor annual changes in vegetation cover and species composition in response to rainfall in this communal rangeland area. Own observations and the inspection of repeated photographs of the monitoring plots (Fig. 11) prove that of the woody species, *Acacia reficiens* in particular has shown a rapid growth response to good rainfall seasons, which were above average in 2006 and 2008.
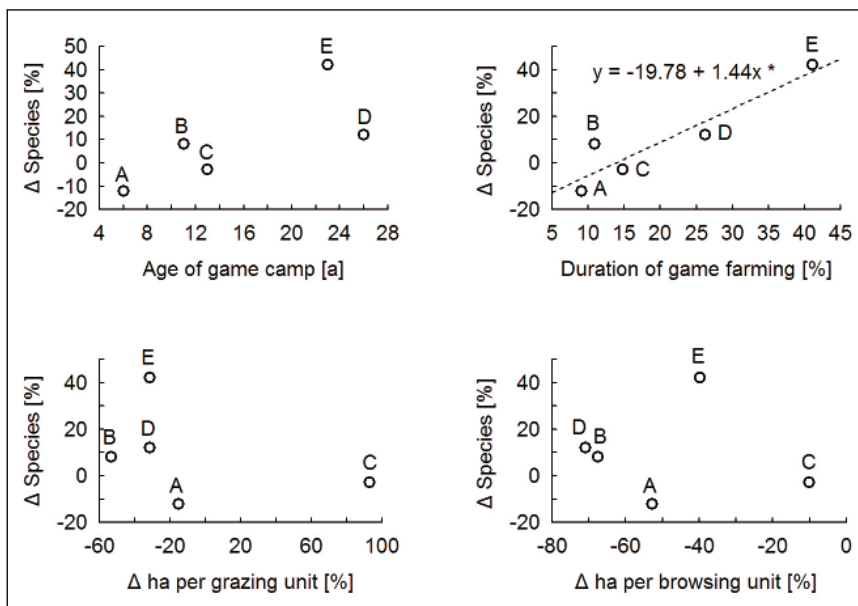
Fig. 12: Positive change of the number of plant species on game farms in comparison to adjacent sheep camps at five different sites (A–E). Grazing and browsing units have been calculated according to Bothma et al. (2004). Spearman Rank Order correlation: * = $p < 0{,}05$.

In general, the monitoring results showed an increase of *A. reficiens* cover by 4% over the period 2005–2009. However, because of the relatively short period of observation the increase in cover was not significant. For a semi-arid rangeland like the Ovitoto area with a high degree of variability in interannual rainfall amounts, longer periods of vegetation monitoring will be necessary to document significant changes in woody cover. Nevertheless, the observed trend indicates that *A. reficiens* has the potential to become dominant within a few years. Once young individuals have established themselves and developed an extensive root system they can grow rapidly and make rangelands inaccessible for livestock within a short period. The monitoring of individual encroacher species such as *A. reficiens* under different treatments will give more detailed information on the potential growth rates of this species independent of landuse intensity and rainfall amounts. These results would be the basis for the development of appropriate management strategies to prevent or combat bush encroachment in the Highland savanna of Namibia.

Changes in plant diversity in relation to duration and intensity of grazing

[A. Horn, C. Reisch & P. Poschlod]
Heavy grazing in conjunction with good rainfall years can lead to a rapid increase of bush encroachment within less than a decade (compare previous example), but many changes in plant species richness and composition occur at a slower pace. For example, degradation zones around watering points on 100 year old farms in the southern Kalahari around the Observatory Alpha (S17) were significantly more extensive than those on 50 year old farms (Horn 2008). This indicates that the composition of the present vegetation depends on the overall intensity of landuse, which has accumulated over many decades. The importance of including long-term processes was further corroborated by the slow recovery of former sheep camps now used for game farming (Horn 2008). The positive change in species richness on game camps in comparison with adjacent sheep camps increased with the age of the game camp (oldest game camp = 26 years) and was significantly correlated with the proportion of game farming to total farm age. In contrast, a reduction in stocking rate did not show a significant correlation with a positive change in the number of species (Fig. 12).

Farming also had effects on the spatial patterns of functional traits. For example the distribution of plants with dispersal syndromes adapted to wind-dispersal was significantly correlated with animal density within the landscape. In other words, there was a lower incidence of wind-dispersed species in intensively-grazed areas such as around 30 year old watering points, while the converse was also true where the incidence of wind-dispersed species was higher in areas with low animal density (Horn 2008). Curiously, wind dispersal potential was generally low, although few other vectors would naturally be available at high frequencies, since moving surface water is hardly ever present and natural animal densities are low. Animal-dispersed species can also be sensitive to changes in available dispersal vectors. For example, antelope act as an important dispersal vector for the tsamma melon, *Citrullus lanatus*. Their loss due to hunting and the erection of fences in the 1960s has had significant effects on seed dispersal as could be shown by changes evident in gene flow and population genetic differentiation (Horn 2008).

We conclude that while in the short-term the vegetation seems to be mostly influenced by rainfall, over longer time frames even mild overgrazing can have severe effects, especially in focal areas such as watering points. Furthermore, severe degradation is almost irreversible due to the harsh climate and associated low frequency of germination events, but also due to the transient nature of the seedbanks and the potential rarity of long-distance dispersal.

Identifying key patterns of animals diversity along landuse and climate gradients

Responses of animal diversity to shrub encroachment

[N. Blaum, E. Rossmanith & F. Jeltsch]
One of the most threatening forms of rangeland degradation in Thornbush Savannas of southern Africa is shrub encroachment resulting from heavy livestock grazing in conjunction with good rainfall years (e.g. Kraaij & Ward 2006). Although the economic effects of
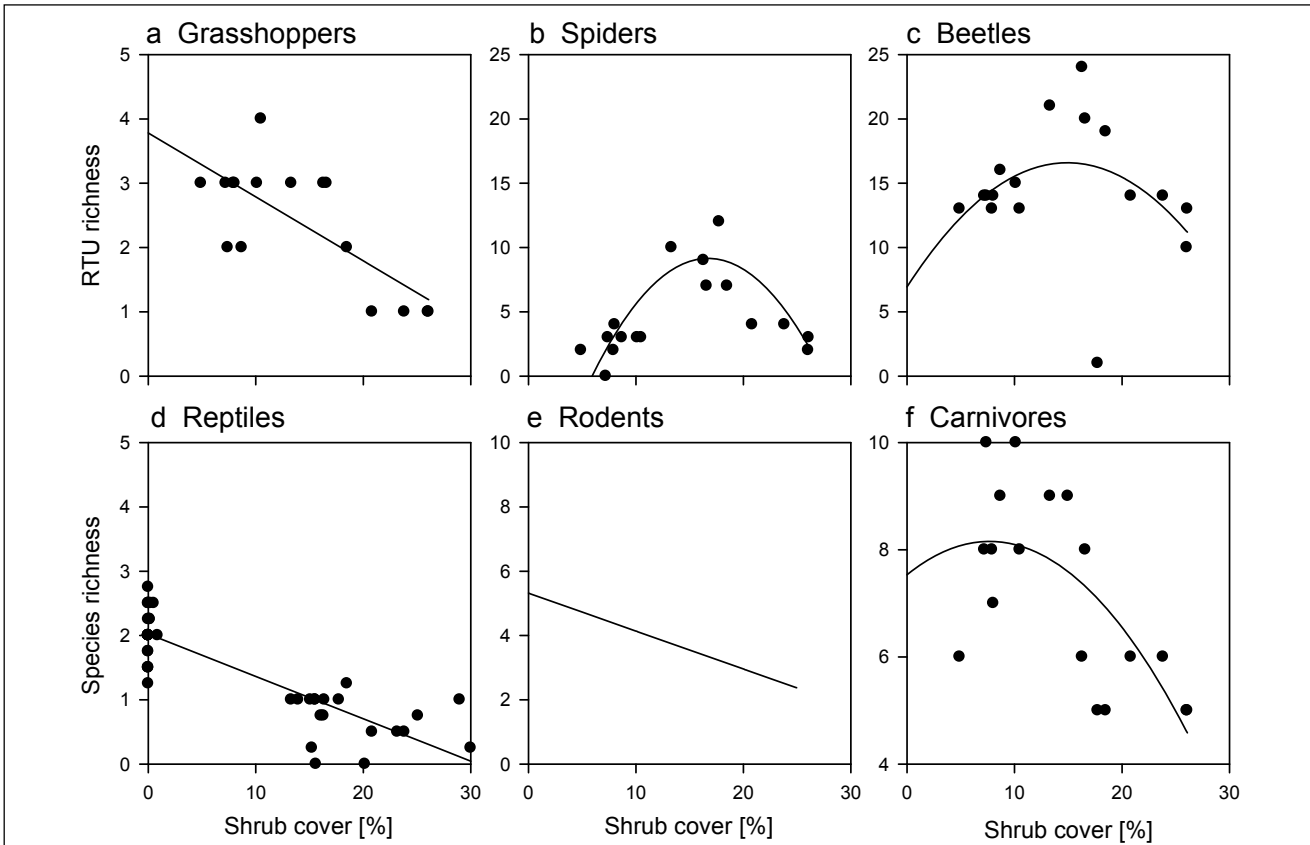
Fig. 13: Changes in animal diversity along a gradient of shrub cover (a: grasshoppers, b: spiders, c: beetles, d: reptiles, e: rodents, f: mammalian carnivores). Species richness of insects (a–c) were determined according to Scholtz & Holm (1989) and then classified as recognisable taxonomic units (RTUs; distinguished by morphological characteristics) using the rapid assessment of biodiversity approach of Oliver & Beattie (1993). Figs. 13a–c was reprinted with the permission of Springer Science and Business Media from Blaum et al. (2009b). Fig. 13e was reprinted with the permission of Blackwell Publishing Ltd from Blaum et al. (2007c). Fig. 13f was reprinted with the permission of Elsevier Ltd from Blaum et al. (2007b).

bush thickening on productive farmland are well known, the effects on savanna biodiversity are only now beginning to emerge. Assessments between habitats (Meik et al. 2002) and across experimentally-cleared plots (Kaphengst & Ward 2008) have shown declines in reptile and avian diversity between open and encroached habitat. However, only few long-term assessments across a wider landscape have been made in southern Africa for birds (Sirami et al. 2009), and within the BIOTA research framework in the Kalahari for arthropods (Blaum et al. 2009a) and mammals (Blaum et al. 2007a, b, c, 2009b).

In the BIOTA Kalahari study (i.e. the rangeland area around Observatory Alpha, S17), the impact of shrub encroachment on animal diversity (and in particular on wingless arthropods, reptiles and mammals) was studied along a gradient of shrub cover ranging from 0–30%. Carnivores were selected as potential

indicators for farmers to detect changes in rangeland quality and to highlight the fact that sustainable farming is a crucial part of nature conservation. Results from the BIOTA long-term studies show that although the responses of animal groups vary between taxa two major patterns are evident. Species richness of grasshoppers, reptiles, and rodents declined with shrub cover while spiders, beetles, and carnivores exhibited bell-shaped responses to shrub cover with species richness maxima at shrub cover values between 12 and 18% (Fig. 13).

The bell-shaped responses of spiders, beetles, and carnivores to shrub cover can be explained by the habitat heterogeneity hypothesis (MacArthur & Wilson 1967), which predicts that structurally complex habitats provide more niches and environmental resources and thus increase species diversity. In southern Kalahari rangelands, the relationship between shrub cover and habitat hetero-

geneity (measured as the horizontal and vertical diversity of the shrubby vegetation) is bell-shaped with a maximum at 10–15% shrub cover (Blaum et al. 2007b). At low shrub cover, savannas are characterised by a structurally-poor grassy matrix with few trees or shrubs scattered in the landscape (Scholes & Walker 1993, Jeltsch et al. 1997a). The increase of shrubs therefore first enriches the structural diversity of savanna habitats (Blaum et al. 2007b), as shrubs such as *Acacia mellifera* provide patches of shade, with concomitant patchiness in soil temperatures, moisture, and nutrient levels (Mazía et al. 2006). In highly overgrazed areas, however, shrubs become the dominant vegetation form, generating homogeneous patches (Skarpe 1990) with low habitat heterogeneity. In relative terms, nutrient levels also become uniformly high in response to nitrogen fixing by woody species and greater amounts of
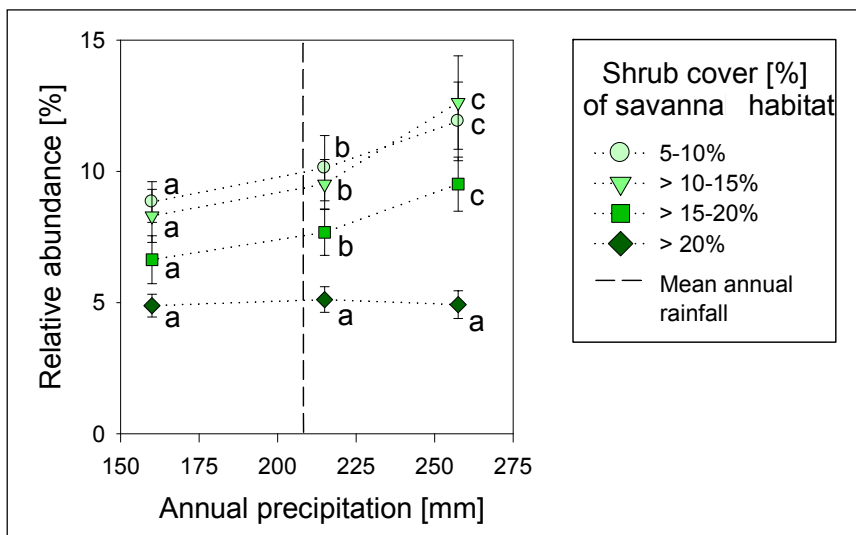
Fig. 14: The influence of annual rainfall on the abundance of small- and medium-sized mammalian carnivores in savanna habitats with different levels of shrub cover (category 1, shrub cover 5–10%, $N = 6$; category 2, shrub cover > 10–15%, $N = 4$; category 3, shrub cover > 15–20%, $N = 6$; category 4, shrub cover > 20%, $N = 4$). Annual rainfall was calculated between two subsequent rainy seasons i.e. from October of one year to September of the following year on the basis of the rainfall data of Twee Rivieren (South African Weather Bureau). Different letters indicate significant differences of carnivore abundance between annual precipitations of sample periods within a shrub cover category (repeated ANOVA; Tukey HSD Post Hoc Test, $p < 0.05$).

dung associated with higher livestock numbers. Despite the different responses, animal diversity across taxonomic groups was lowest for all taxa in areas where shrub cover was highest. Also the regeneration potential of animal populations in shrub-dominated areas during good rainfall years can be limited (Blaum et al. 2007b). For example, the abundance of small- and medium-sized carnivore populations increased with an increase in the amount of annual rainfall in savanna habitats where shrub cover is below 20%, whereas no increase was observed in habitats with shrub cover above 20% (Fig. 14).

As shown here, widespread shrub encroachment, caused by long-term, heavy grazing, threatens animal diversity in Thornbush Savannas. Nevertheless, moderate livestock grazing can even enrich Thornbush Savanna rangelands with positive effects for animal diversity since species richness across taxonomic animal groups was highest in Kalahari rangelands with medium shrub cover (12–18%). This needs to be considered when initiating shrub removal programs that aim at increasing grazing capacity. Understanding the relationships between species diversity and shrub encroachment is a crucial prerequisite to the development of management strategies that sustain both animal diversity and the landuser's profit.

Changes in bird diversity under shrub encroachment along a rainfall gradient

[R. Simmons & C. Seymour]

The effects of bush thickening are not always clear-cut as in the previous examples of mammals, reptiles and wingless arthropods. For birds, Seymour & Dean (2009) found that the presence of large trees within bush-encroached savanna landscapes obscures the effect of bush thickening on bird diversity and abundance. The effect of bush thickening on bird assemblages is, therefore, not simply related to the density of horizontal habitat, but also to the heterogeneity of the vertical habitat. Furthermore, Seymour & Simmons (2008) found that although species density (i.e. the number of species per area) was no different between encroached and non-encroached areas, species richness (species detected per number of birds surveyed) was greater in less encroached areas. Although flight enables birds to include larger shrub-dominated areas in their home ranges, shrub encroachment is nevertheless predicted to increase since under climate change because of the fertilisation effect that increased levels of CO_2 has on woody plants (Polley 1997) where rainfall may be limiting (Bond & Midgley 2000, Bond et al. 2003). Thus, drier areas may mimic what future climate change may bring to areas presently experiencing higher rainfall and bird communities in these areas may tell us how climate change will affect birds in each habitat type. In a recent BIOTA study in Namibia bird species-richness was compared between bush thickets and two alternative habitats (dry riverlines and open grassy areas), across a 260 km rainfall gradient with a drop in mean annual precipitation (MAP) of 257 mm/year. This represents a steep rainfall decline of roughly 1 mm/year for every km travelled mimicking possible climate-induced changes in years to come in Namibia. Within each site, three structurally different vegetation types were identified and sampled for birds. Habitats comprised: "Thicket" (dense shrubland comprising mainly *Acacia mellifera* and *Catophractes alexandri*), ephemeral dry "Riverlines" (with mature *Acacia* spp., *Ziziphus mucronata* and *Boscia albitrunca* trees) and "Open" areas dominated by grassland, with scattered *Acacia* spp. Bird species richness indices in the last dry season of monitoring (Aug. 2009) showed the same trends as in the first dry season (Aug. 2007), indicating across season consistency in resident bird assemblages. The aridity trends for bird richness in the thickets indicate that species richness (and diversity, H' $\log_e$) decline steadily from mesic to arid sites, (i.e. 30–7 species, Fig. 15).

Dry riverlines, however, showed no such decline from east to west (e.g. 28–28 species respectively), while open areas showed only marginal declines (i.e. 24–20 species respectively). This indicates that dry riverlines may play an increasingly important role in bird species diversity as global climate change creates more arid regions, as suggested by Seymour & Simmons (2008).

We next asked if bird assemblages in each area differed according to habitat

and rainfall. Our results showed that for the four more mesic sites, bird assemblages were more similar between habitat types than between sites (e.g. riverine bird assemblages were similar, regardless of aridity), but a threshold was crossed at the most arid site. Bird communities at Rooiklip were more similar to each other regardless of habitat. In line with expected aridity effects on bird communities, the most **dissimilar** sites (at 83%) were those differing most in rainfall (Okasewa and Rooiklip). Combined, these results suggest that habitat is more important than aridity per se on bird community **composition** until very dry conditions are encountered. Across the aridity gradient habitat structure plays a role in species richness with the thickets losing species rapidly, while riverlines retain high species richness.

Understanding biodiversity dynamics—from single species to taxonomic groups

Background

[N. Blaum, E. Rossmanith & F. Jeltsch]
Understanding biodiversity dynamics under global changes in climate and landuse is a major challenge in modern ecology and a prerequisite for conservation planning. In Thornbush Savannas of southern Africa, many lessons on how animal diversity changes along landuse and climate gradients have been learnt from empirical studies within the BIOTA research framework (see under 'Identifying key patterns' in Subchapter IV.2.4; compare also Subchapters III.4.4 and III.4.5, Blaum 2008, Blaum et al. 2007a, b, c, 2008, 2009a, b, Popp et al. 2007, Meyer et al. 2009, Seymour & Simmons 2008, Seymour & Dean 2009, Thiele et al. 2008, Wasiolka et al. 2010). Based on such correlations between species diversity and environmental conditions, predictions on how species diversity may respond under changes in landuse and climate can be calculated. However, this static approach is often criticised because relevant processes such as dispersal, local adaptation, interactions, and transient dynamics are often not included (e.g. Thuiller et al. 2004). Process-based, dynamic simulation models are an alternative and promising approach (Guisan &

Thuiller 2005) and have been applied in several BIOTA studies (e.g. Wichmann et al. 2003, Blaum & Wichmann 2007, Popp et al. 2007, Schwager et al. 2008). In these models, all important processes of a population dynamics are simulated. Based on the known effects of the environment (e.g. climatic conditions) on specific processes of population growth, physiology, or behaviour, dynamic population models project the details of population dynamics into the future under scenarios of environmental change (e.g. induced by climate or landuse). In comparison to static approaches, dynamic population models have the inherent advantage of not accepting a species' response to a certain climate as static but as result of certain mechanisms and processes. Still, most process-based dynamic simulation models require many data for parameterisation and are thus often based on single species approaches (e.g. Wichmann et al. 2003, Tews et al. 2006, Blaum & Wichmann 2007, Popp et al. 2007, Schwager et al. 2008). Although these approaches allow for the development of specific management recommendations

for decision support of a single species, recommendations for a wider range of species with, for example, different home range requirements and dispersal abilities are limited. To predict changes in biodiversity dynamics based on a single species approach is impossible under reasonable time and cost frames. Therefore, a generic modelling approach was developed, where the impacts of landuse and climate changes can be simulated for a set of species.

In the following section, the results of two modelling approaches will be presented, where the impact of landuse on species diversity and abundance was analysed for multiple species.

Sensitivities of mammalian carnivores to landuse at scales relevant for decision makers

[N. Blaum, E. Rossmanith & F. Jeltsch]
Analysing the effects of land management on biodiversity dynamics requires the explicit consideration of scale-crossing mechanisms and process-interactions. For example, the germination and establishment of plants act at very small and local
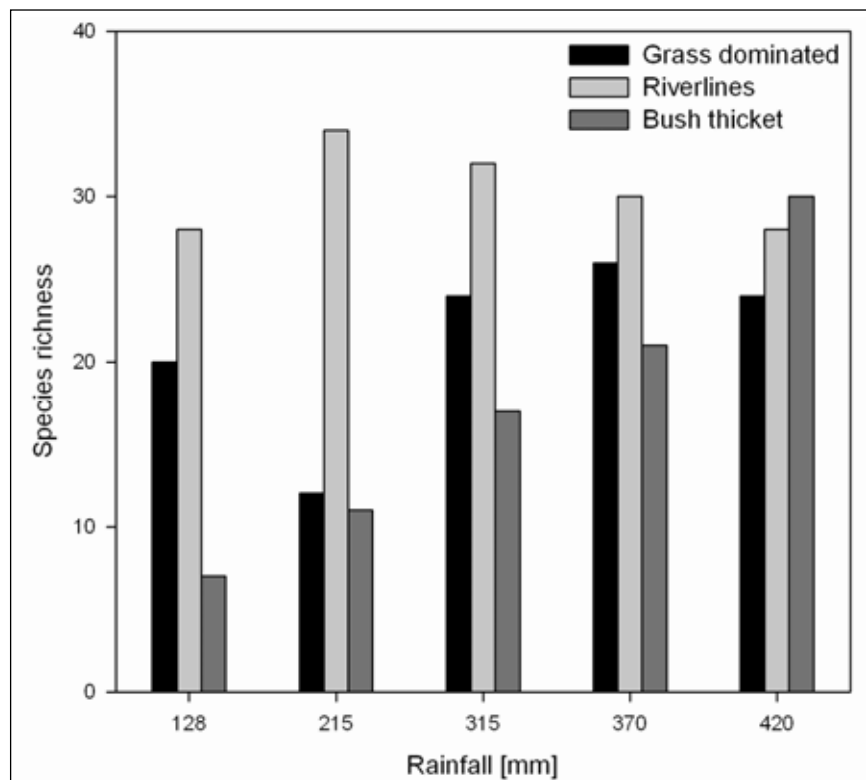

Fig. 15: Responses of bird species richness along an East-West rainfall gradient in Namibia for three major savanna habitat types. Black bars show grass-dominated savanna habitats, dark grey bars show bush thickets and light grey bars riverlines.
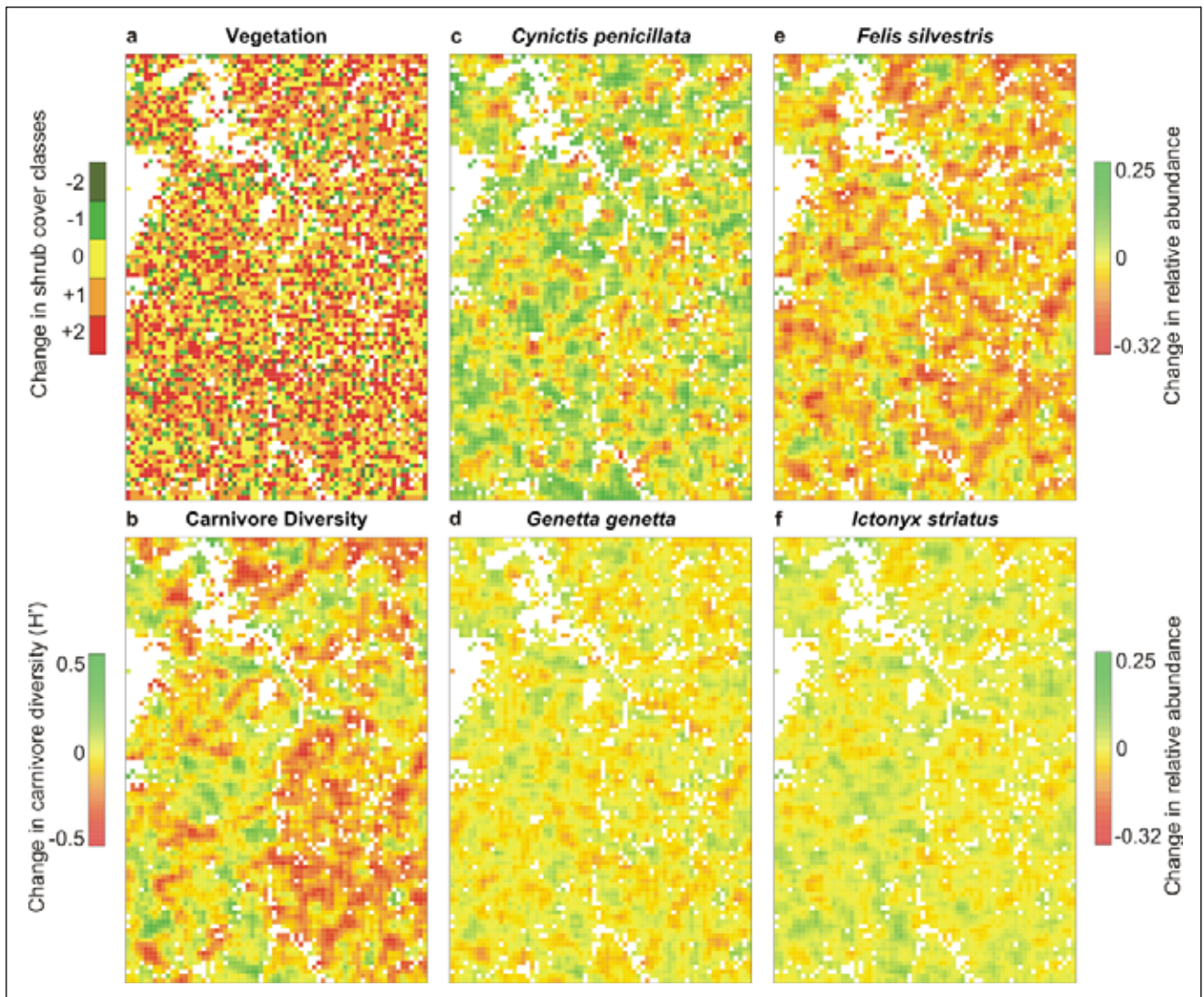
Fig. 16: Simulation results of the upscaled vegetation and carnivore model. Tested scenario: 20% increase in grazing intensity on all farms. Shown are differences between current status quo and changes after 20 years. Panel a (top left): The upscaled model assumes three shrub cover classes: low shrub cover < 10%, medium shrub cover 10–20% and high shrub cover > 20%. Panel a shows the changes in shrub cover classes (–1 indicates a decrease of shrub cover of one class, i.e. from high to medium shrub cover or from medium to low shrub cover). Panel b (bottom left): Changes in carnivore diversity (Shannon Index). Panels c–f: Changes in relative abundance of c: Yellow Mongoose (*Cynictis penicillata*), d: Small Spotted Genet (*Genetta genetta*), e: African Wildcat (*Felis silvestris lybica*), and f: Striped Polecat (*Ictonyx striatus*).

scales while the dispersal of animals and the decisions made by land managers can act at large and often regional scales. The integration of such contrasting scales was successfully done combining the analysis of a small scale vegetation model and a state-and-transition approach (Jeltsch et al., in press). This upscaled vegetation model was combined with a multiple species approach to analyse the impacts of land management on mammalian carnivores in a southern African Kalahari rangeland system. The simulated Kalahari rangeland system (75 km x 100 km) is characterised by a mosaic of livestock farms, with different stocking rates and

farm sizes (3,000 ha to 20,000 ha). In a first step, a grid-based landscape model (grid-cell size 1 km²) was developed, which includes specific grazing intensities for all farms. The savanna vegetation was simulated dynamically using a state-and-transition approach (Westoby et al. 1989). Transition probabilities of typical Thornbush Savanna vegetation states (defined by a specific proportion of shrub and perennial grass cover) under different grazing intensities and annual rainfall were derived from simulation experiments with a small-scale process-based vegetation model (5 m by 5 m grid-cell size, Jeltsch et al. 1997a) on 1 km². In

particular, the frequency of occurrence of transitions between two vegetation states served as transition probabilities that were used to simulate the change in vegetation for each of the 7,500 one square kilometre patches of the landscape model (i.e. the whole rangeland area; see Fig. 16 for an example).

In a second step, the large scale vegetation model was linked to statistical models of carnivore abundance and species richness field data (Blaum 2008, Blaum et al. 2007a, b, 2008, 2009a) including an algorithm describing neighbourhood effects. The current vegetation for the study area was derived from remote sens-

ing data, which were ground truthed by several hundred vegetation surveys in the field. Current grazing impact was derived from own farm surveys and stocking data from the extension office in Upington. In the sample scenario shown in Fig. 16, stocking rates were increased by 20% for a period of 20 years. The results of this combined large-scale savanna landscape and carnivore diversity model shows a large scale degradation of the savanna vegetation with differing sensitivities of carnivores within one guild (Fig. 16). While Yellow Mongooses (*Cynictis penicillata*) showed positive as well as negative responses to the predicted changes, the abundance of Wild Cats (*Felis sylvestris lybica*) declined in most habitats. Similarly, the abundance of Small-spotted Genets (*Genetta genetta*) decreased slightly while minor effects were predicted for Striped Polecats (*Ictonyx striatus*). The ambivalent responses of Yellow Mongooses to predicted habitat changes can be explained by their dependency on single shrubs for sheltering and reproduction (for details see Article III.4.6). This large-scale vegetation model can also be used for other applications such as landuse optimisation models (for an example see Article III.7.4), linkage to dynamics population models of animals with large home ranges or dispersal models (Chapter III.2).

Effects of landuse on bird populations—a generic functional type modelling approach

[E. Rossmanith & N. Blaum]
As stated above (see 'Uderstanding biodiversity dynamics' in Subchapter IV.2.4), to better understand the dynamics of biodiversity we need to transfer knowledge gained from single species to a more generalised multiple species approach. This has led to the development of categorisation systems, where species' similarities in life strategies and traits within and between communities are classified into functional groups or types (e.g. Lavorel & Garnier 2002, Blondel 2003). Functional types were initially defined as "a non-phylogenetic classification leading to a grouping of organisms that respond in a similar way to a syndrome of environmental factors"

(Gitay & Noble 1997). The classification is based on a set of functional traits that are considered important for a species' response to the environment. While much emphasis has been set on the identification of key traits and functional types in plant ecology (e.g. Lavorel & Garnier 2002, Cornelissen et al. 2003, Poschlod et al. 2003) examples where animal species were identified as functional groups are scarce. For savannas, functional approaches for animals are mainly restricted to grazing ungulates and their effects on vegetation (e.g. McNaughton 1993, du Toit & Cuming 1999).

One reason that functional classification schemes have hardly been developed further in animal ecology might be the difficulty in defining clear core traits due to the high variability in behaviour, foraging strategies, resource needs and morphology of animals compared to plants. For savanna systems, the combined effects of landuse and climate on key habitat structures and on resource availability demand the identification of (i) core animal traits describing species' dependency on their habitat as well as (ii) life history traits that are related to the processes of birth, survival, and movement. Orienting process-based, bottom-up models towards the response of functional types instead of single species and identifying traits that react most sensitively will allow for the derivation of more general results with regard to understanding and predicting the effects of environmental change on savanna biodiversity.

As indicated earlier, one major environmental change in the savannas of southern Africa is shrub encroachment and the loss of perennial grass cover due to overgrazing. This change in habitat can reduce the availability of food resources, such as grass seeds, insects, and vertebrate prey species, for birds (e.g. Blaum et al. 2007b, c, 2009a). Habitat change also decreases the availability of grass needed for nest construction. On the other hand, shrub encroachment will increase shrub-associated prey species as well as nest sites for shrub-nesting bird species. Besides grazing, wood cutting for fire wood production is an important landuse option, which decreases tree density and changes spatial patterns of trees (Anderson & Anderson 2001). For many bird species, trees are

important structures for nesting or as a raised stand for hunting. Thus, depending on the traits and trait combinations of birds, predicted landuse induced changes will have different effects.

To investigate the effect of shrub encroachment and tree removal on bird species in Thornbush Savannas, a generic functional type model was developed for birds, which simulated the population dynamics of a large variety of functional types, characterised by the combination of functional traits. Parameterisation of life history traits (reproductive rate, survival rate, age of maturity, home range size and dispersal distance) were derived from the allometric relationships with body mass. Since reproductive rate is strongly influenced by latitude (Lack 1947, Skutch 1985), this trait was derived from a meta-analysis of 135 bird species in the study area (southern Kalahari, including Observatory Alpha, S17). All other allometric relationships were derived from meta-analyses in the literature (Calder 1984, Saether 1987, Prinzinger 1993, Bowman 2003, Ottaviani et al. 2006). Traits that are particularly important for birds in Thornbush Savannas under changing habitat due to landuse were food preference [(i) herbivorous/carnivorous, (ii) insectivorous/omnivorous, and (iii) carnivorous], and nest site needs (trees, shrubs, ground). Together with body mass, these three traits were systematically varied and combined to test their effect on a large variety of bird functional types.

The model is spatially-explicit and is based on a landscape grid, which is characterised by the proportion of shrub and grass cover as well as the number of trees and the spatial patterns of these vegetation structures (see Fig. 19 as an example of different tree clumping scenarios). A home range grid is projected on top of the landscape grid, with 15 x 15 cells of the size of a home range where the cell size for a particular bird functional type is determined by the respective body mass. For each home range cell, vegetation composition was calculated, and this in turn influenced (together with simulated stochastic rainfall) the habitat quality. Changes in the population dynamics proceeded in annual time steps and the habitat quality of the home range then influenced
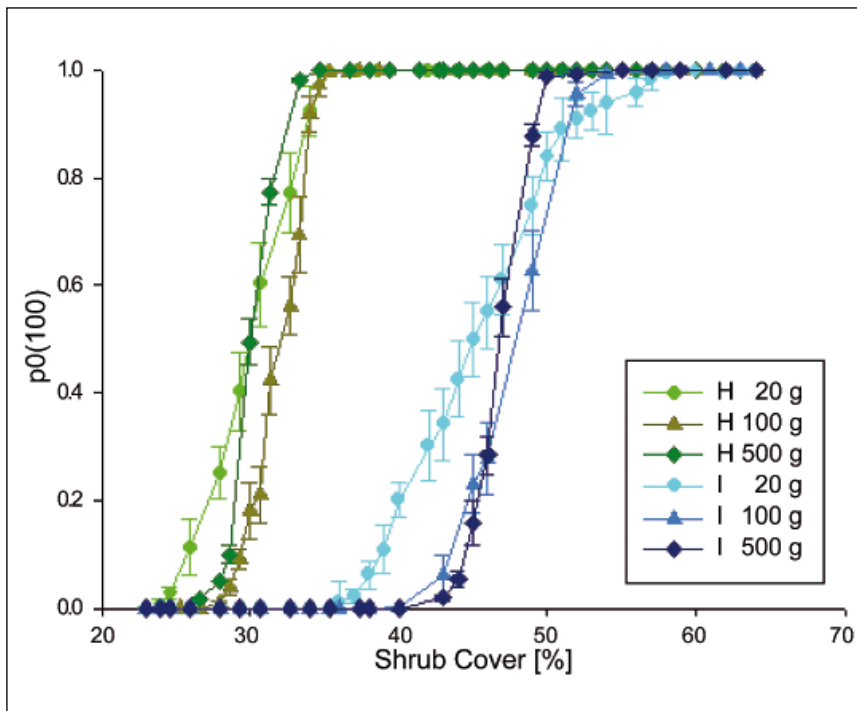
Fig. 17: Effects of shrub encroachment on extinction probability within the next 100 years (p_0(100) for herbivores/granivores (H) and insectivorous/omnivorous (I) birds of three size classes. The class of small-sized birds (0–50 g) was represented by simulating a 20 g bird, the class of medium sized birds (51–300 g) by a 100 g bird and the class of large sized birds (> 300 g) by a 500 g bird. The mean and standard error of 30 runs are shown. Each run simulates 100 populations with identical landscape and parameterisation.
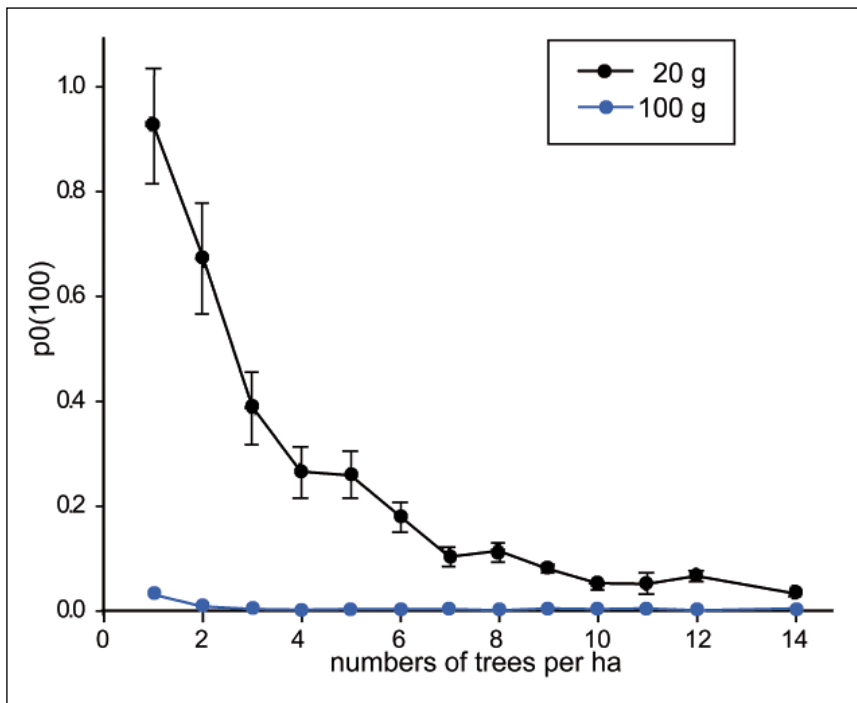

Fig. 18: Effect of tree density on extinction probability within the next 100 years of tree nesting birds in three size classes. The mean and standard error of 30 runs are shown. Each run simulates 100 populations with identical landscape and parameterisation. Spatial pattern was scenario "low clumping", which is comparable to the existing pattern of trees in the study area.

the reproductive success and the survival rate of its inhabitant. Using this model, the effect of different shrub encroachment as well as tree removal scenarios on the persistence of functional bird types could be tested.

The results of the simulation model show that shrub encroachment will increase the risk of extinction for all tested functional types. However, herbivorous birds respond more sensitively to shrub encroachment then insectivorous birds (Fig. 17). This is supported by Sirami et al. (2009), who found for bird assemblages that carnivorous and other birds associated with grasses decreased as the percentage of shrubs increased whereas insectivorous species increased. Even though there are slight differences in extinction risk between birds of different size classes, the results show that the trait reacting most sensitively to shrub encroachment is food preference.

For tree nesting species, the model predicts a strong increase in extinction risk with decrease of tree density for small birds but almost no effect on medium-sized and large birds when trees are distributed randomly (Fig. 18). The reason for this pattern is that the smaller home range sizes evident within small birds (Ottaviani et al. 2006), will result in a lower likelihood of finding a tree at given densities then for birds with large home ranges. However, the spatial pattern has a large influence on the persistence of populations, since stronger clumping of trees will also affect negatively the extinction risk of medium-sized and large tree-nesting birds (see Fig. 19 for medium-sized bird). Model results show that not only the remaining tree density but also their spatial pattern should be considered when removing trees for firewood and charcoal production.

2.5 Risk assessments

Savanna vegetation

[M. Wieczorek, N. Blaum, H. Rubilar, E. Rossmanith, D. Lohmann & F. Jeltsch] Namibian savannas are threatened by climate change and unsustainable land-use. Both, taken separately or in combination, can lead to degradation. This can

occur either as a general loss of vegetation cover and an increase of bare patches or an increase in woody and less palatable vegetation (see 'Remote sensing' in Subchapter IV.2.2). While the loss of palatable, perennial herbaceous vegetation is a general risk accompanying all types of savanna degradation the specific outcome of climate change and overgrazing depends not only on the rainfall amount and regime but also on the soil type and the actual vegetation type and cover (see Subchapter IV.4.3 and Article III.2.4).

To assess the risk of degradation in Namibian savannas, we applied the ecohydrological model EcoHyD (Tietjen et al. 2009a, b; see also Subchapter IV.2.3) to six different savanna sites that represent rainfall regimes with MAP (mean annual precipitation) between 200 mm and 500 mm and to four different soil types (loamy sand, sand, sandy clay loam, and sandy loam). These 'site – rainfall regime – soil' combinations represent a broad range of environmental conditions specific to a large part of Namibian savannas (Fig. 21). In addition to the variable combination outlined above, grazing was also varied in these areas according to four different categories, from low to very high grazing intensity, quantified as 2%, 5%, 10%, and 20% grass cover removal per year by livestock grazing that needed to be compensated for by regrowth. To allow for better comparison, the same topography was applied to all areas in the model. For all rainfall-soil combinations within this focal savanna region we systematically tested the risk of vegetation degradation under the different levels of livestock production for scenarios with and without climatic change. For climate change scenarios the MAP of each region was assumed to decrease by 10% and the mean annual temperature was assumed to rise by 2.25°C (compare Subchapter IV.2.3 and Article III.7.2).

To calculate the risk of degradation for every site, we assessed the relative decrease of perennial grass cover as the most important aspect of degradation risk. We also looked at the risk of shrub encroachment, which we defined as a relative increase of shrub cover of more than 5% in the given time frame and an
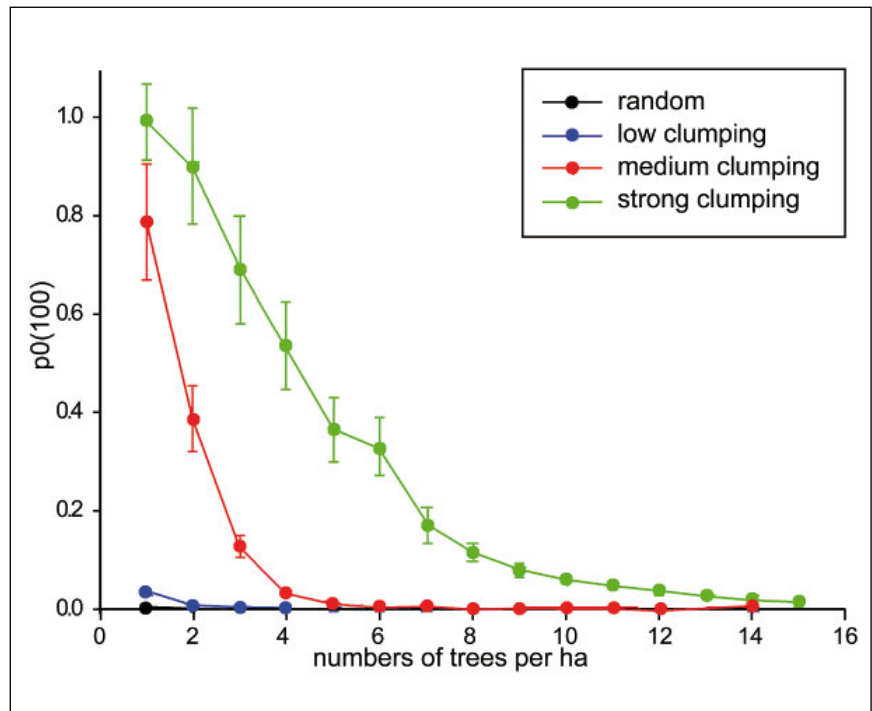
Fig. 19: Effect of tree density on extinction probability of medium-sized birds under four different scenarios of spatial patterns of trees. The mean and standard error of 30 runs are shown. Each run simulates 100 populations with identical landscape and parameterisation.

absolute value of shrub cover higher than 5% (striped areas in Figs. 21 & 22).

The risk assessment maps (Figs. 21 & 22) clearly show the increasing degradation risk with higher grazing intensities, which is further enhanced by expected climate change. Different soil conditions also have a strong influence on the risk of vegetation degradation and on the impact of climate change. Soil type also influences the risk of shrub encroachment, which is further modified by climate change. In general, the tested climate change scenarios, which did not include increased CO_2 levels (compare Subchapter IV.2.3 and Article III.7.2), reduced the tendency towards shrub encroachment but also led to an decrease in overall vegetation cover and thus an increased risk of erosion (see Figs. 21 & 22).

Risk assessment of a medicinal plant—the Devil's Claw

[S. Schütze, N. Blaum, H. Rubilar, E. Rossmanith, D. Lohmann & F. Jeltsch]
To assess the risk of population decline of the Devil's Claw (*Harpagophytum procumbens*) under landuse and climate change in Namibian savannas, we used the above-predicted changes in vegetation

cover (Figs. 21 & 22) as landscape input for the Devil's Claw model described in Article III.7.6 (Schütze 2009). For each of three landuse scenarios (low grazing, moderate grazing and high grazing impact), and one climate change scenario (details described above in 'Savanna vegetation', Subchapter IV.2.5), the impacts of two different harvesting strategies on the change in Devil's Claw population size after 100 years were simulated. Harvesting scenario 1 was a sustainable harvesting strategy with a moderate removal rate of secondary tubers and a three year harvesting cycle that was identified by the model of Schütze (2009) and supported by expert knowledge (Strohbach, pers. comm.). Harvesting scenario 2 reflects a common strategy where high amounts of secondary tubers are harvested annually (Strohbach & Cole 2007, see also Article III.7.6).

Model results show that the impacts of the harvesting strategy on the relative change in size of Devil's Claw populations is much stronger than the predicted impacts of the livestock management and climate change scenarios (Figs. 23 & 24). However, predicted climate change increases the negative effects of unsustainable harvesting on Devil's
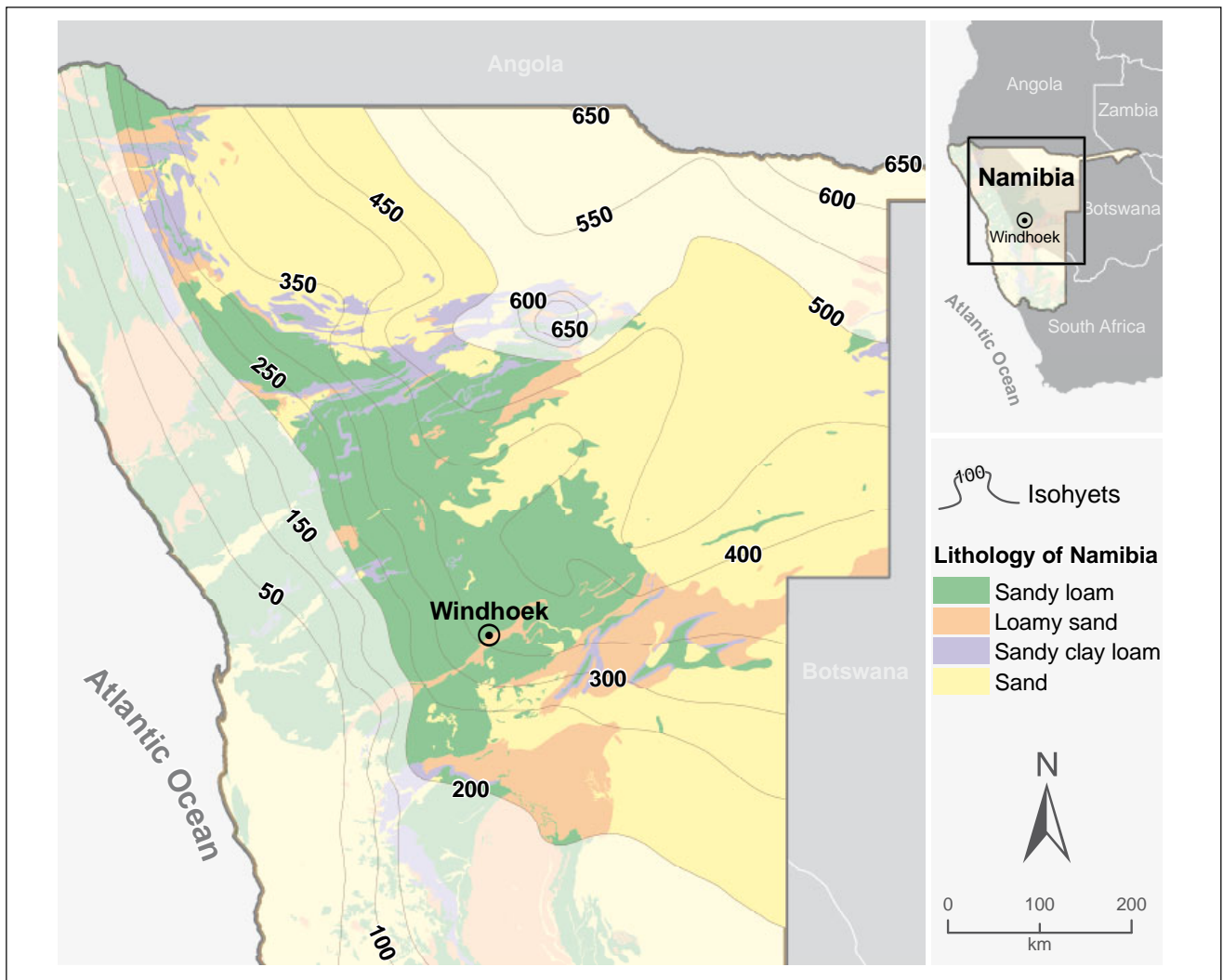
Fig. 20: Lithology of Namibia with rainfall isohyets for the risk assessment maps in the focal Namibian savanna region. Figure modified from Christelis & Stuckmeier (2001), with permission from the Ministry of Agriculture, Water and Rural Development, Department of Water Affairs, Division Geohydrology, Namibia.

Claw population size across the study region (Fig. 24). The model predicts that the most sensitive region in Namibian Thornbush Savannas is the most southern region with the lowest amount of rainfall.

Risk assessment of bird functional types

[E. Rossmanith, N. Blaum, H. Rubilar, D. Lohmann & F. Jeltsch]

To assess the extinction risk of birds under landuse and climate change in Namibian savannas, we used the above predicted changes in vegetation cover after 30 years (Fig. 22) as landscape input for the animal functional type model described under 'Understanding biodiversity dynamics' in Subchapter IV.2.4. The Figures shows the extinction risk of insectivorous (Fig. 25) and herbivo-

rous (Fig. 26) birds. Since there were only slight differences in the results for different size classes, only results for small birds (body mass < 50 g, which comprise 54% of the species in the area) are shown. The extinction risk is given as the probability of extinction within 100 years ($p_0[100]$).

The model predicts no threat for small insectivorous birds as a result of habitat change, except for a slight increase in extinction risk under very high grazing impact (Fig. 25). For herbivorous birds, however, the model predicts increased extinction probabilities of up to 50% under high and very high grazing impacts (Fig. 26). Especially in areas with low rainfall, the extinction risk increases even under moderate grazing impacts when the climate change scenario is

added to the model. This spatial pattern is determined by the relative loss of perennial grass, which is the primary food resource (compare Fig. 22). The results emphasise the high sensitivity that herbivorous birds have to changes in habitat caused by overgrazing or climate change. However, we need to be cautious in applying these results to specific bird species, since species-specific life-history traits, potential buffer mechanisms and behavioural adaptations to environmental changes were not considered in this analysis and might have an important influence on extinction risk.

Fig. 21: Risk assessment maps of predicted vegetation changes for the focal Namibian savanna region over 10 years under low to very high grazing intensities without and with climate change. Colour shades indicate different levels of relative loss of perennial grass cover. Striped areas indicate a principle, additional risk of shrub encroachment (for details, see text).
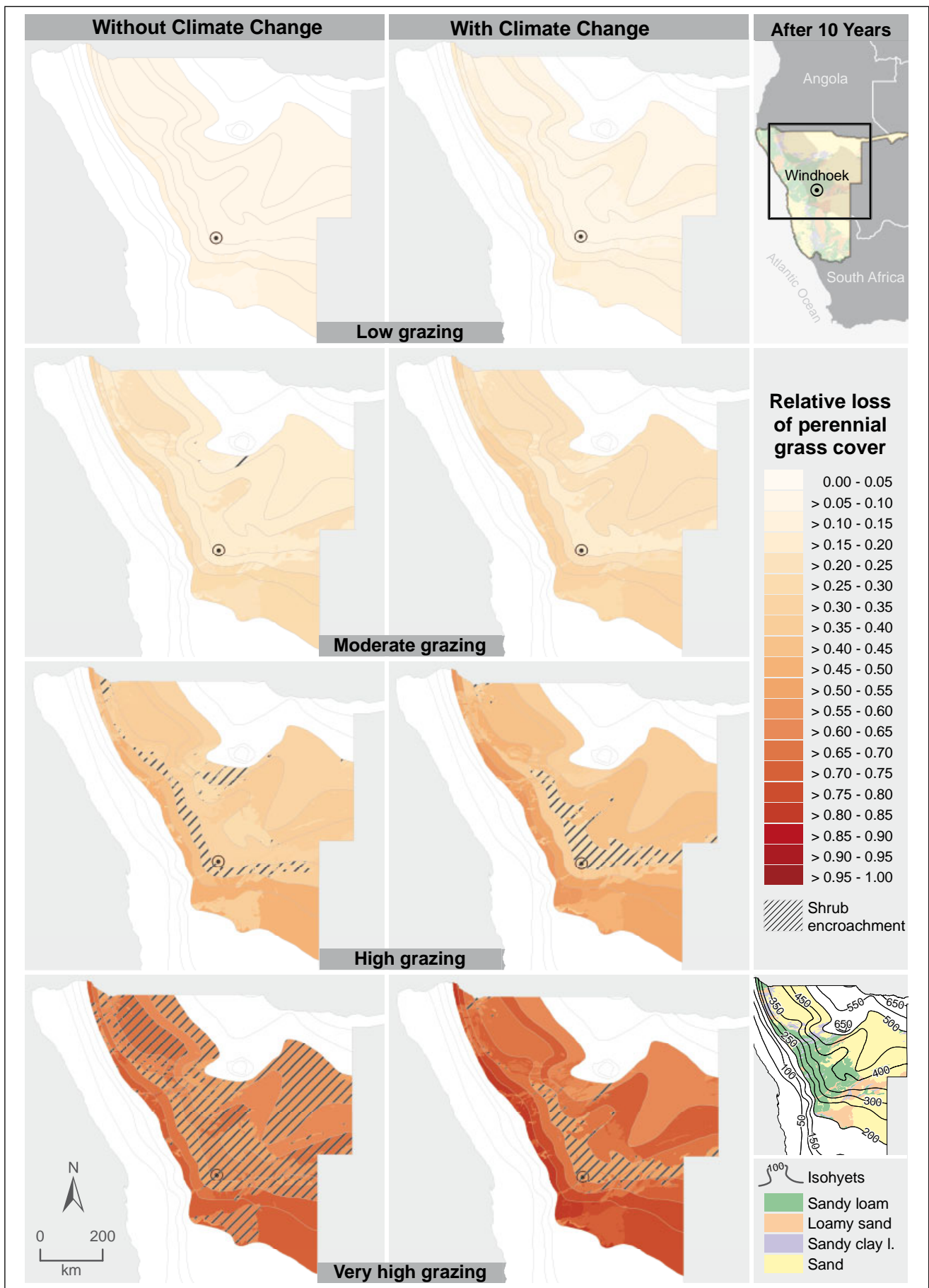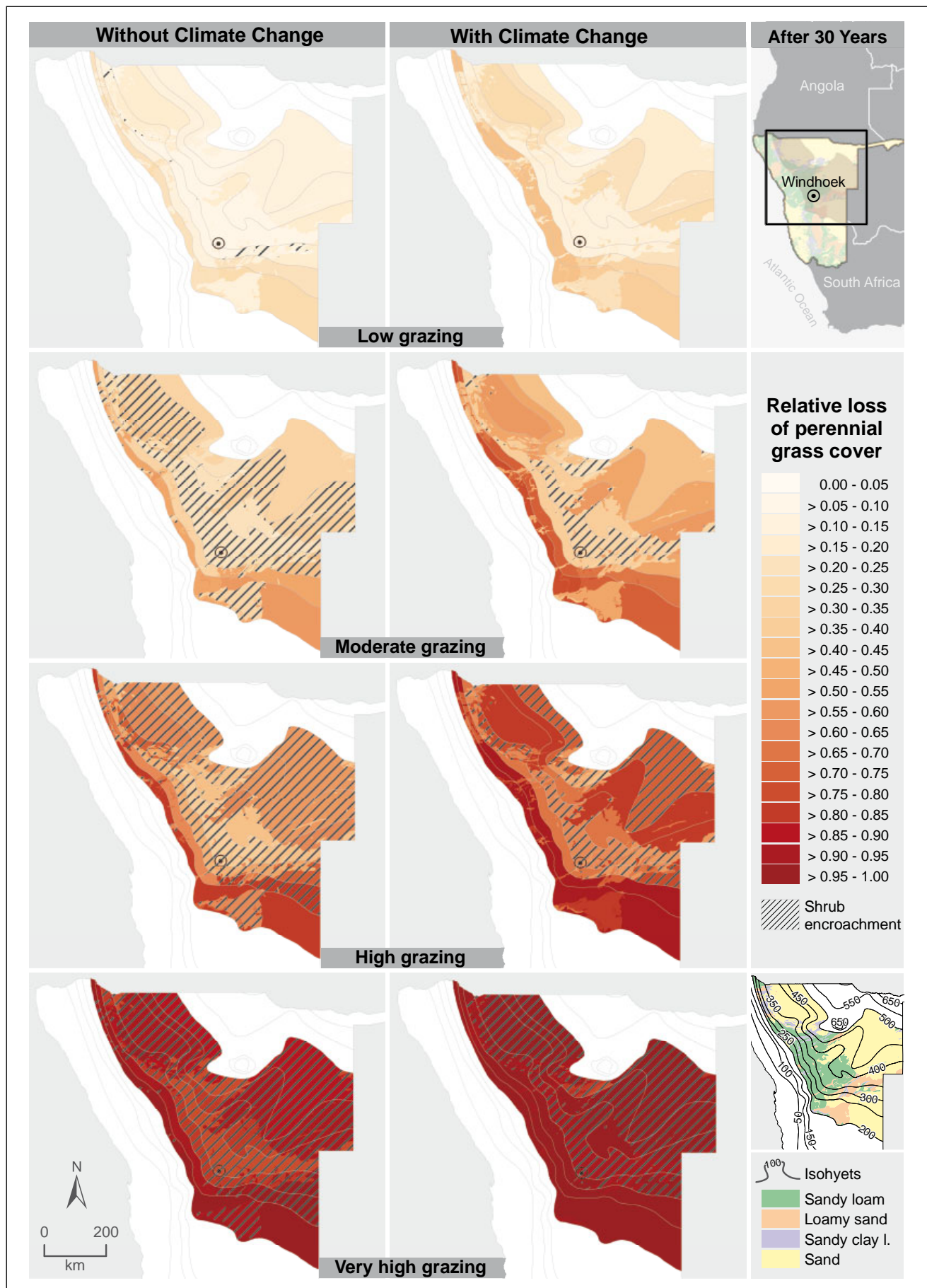
Fig. 22: Risk assessment maps of predicted vegetation changes for the focal Namibian savanna region over 30 years under low to very high grazing intensities without and with climate change. Colour shades indicate different levels of relative loss of perennial grass cover. Striped areas indicate a principle, additional risk of shrub encroachment (for details, see text).

Without Climate Change

| Sustainable harvesting | High harvesting |

Low grazing

Moderate grazing

High grazing

Relative change in population size of the devil's claw

Increase   Decrease

| 0.00 - 0.05 |
| > 0.05 - 0.10 |
| > 0.10 - 0.15 |
| > 0.15 - 0.20 |
| > 0.20 - 0.25 |
| > 0.25 - 0.30 |
| > 0.30 - 0.35 |
| > 0.35 - 0.40 |
| > 0.40 - 0.45 |
| > 0.45 - 0.50 |
| > 0.50 - 0.55 |
| > 0.55 - 0.60 |
| > 0.60 - 0.65 |
| > 0.65 - 0.70 |
| > 0.70 - 0.75 |
| > 0.75 - 0.80 |
| > 0.80 - 0.85 |
| > 0.85 - 0.90 |
| > 0.90 - 0.95 |
| > 0.95 - 1.00 |

Isohyets
Sandy loam
Loamy sand
Sandy clay l.
Sand

Fig. 23: Risk assessment maps for Devil's Claw (*Harpagophytum procumbens*) in the focal Namibian savanna region under three grazing scenarios and two harvesting strategies without climate change. Colour shades indicate the relative change in population size of Devil's Claw after 100 simulation years.
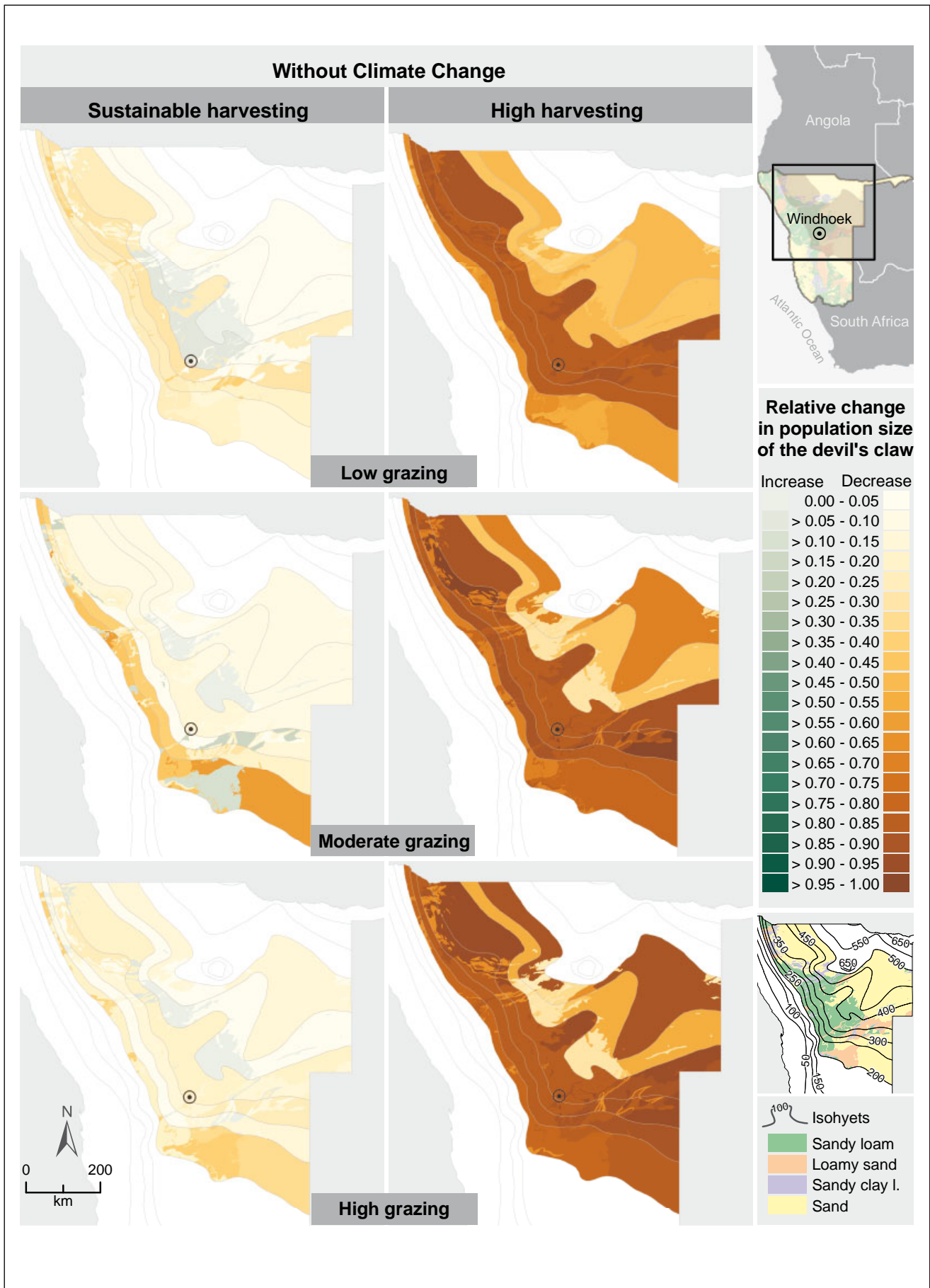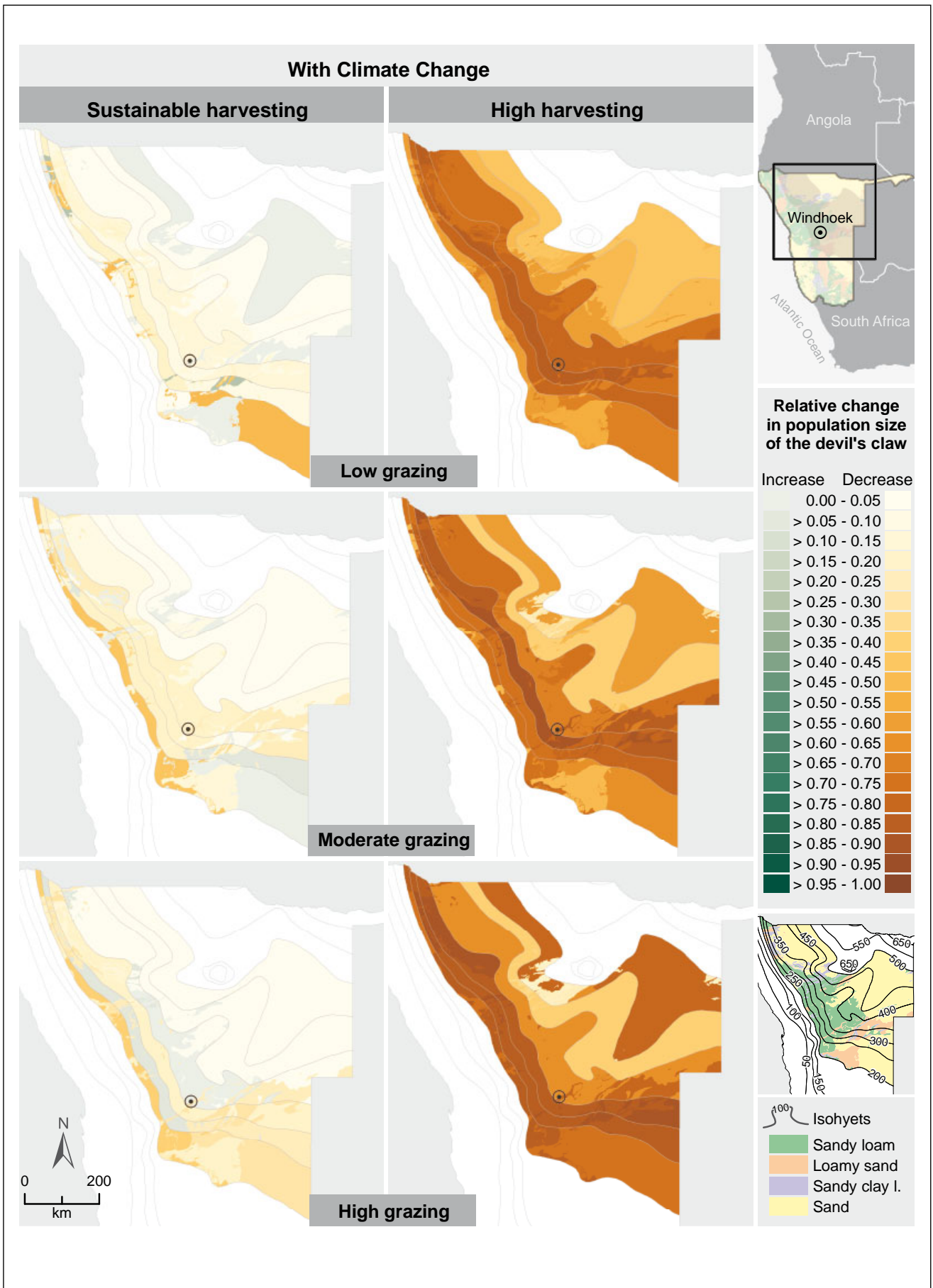
Fig. 24: Risk assessment maps for Devil's Claw (*Harpagophytum procumbens*) in the focal Namibian savanna region under three grazing scenarios and two harvesting strategies with climate change. Colour shades indicate the relative change in population size of Devil's Claw after 100 simulation years.
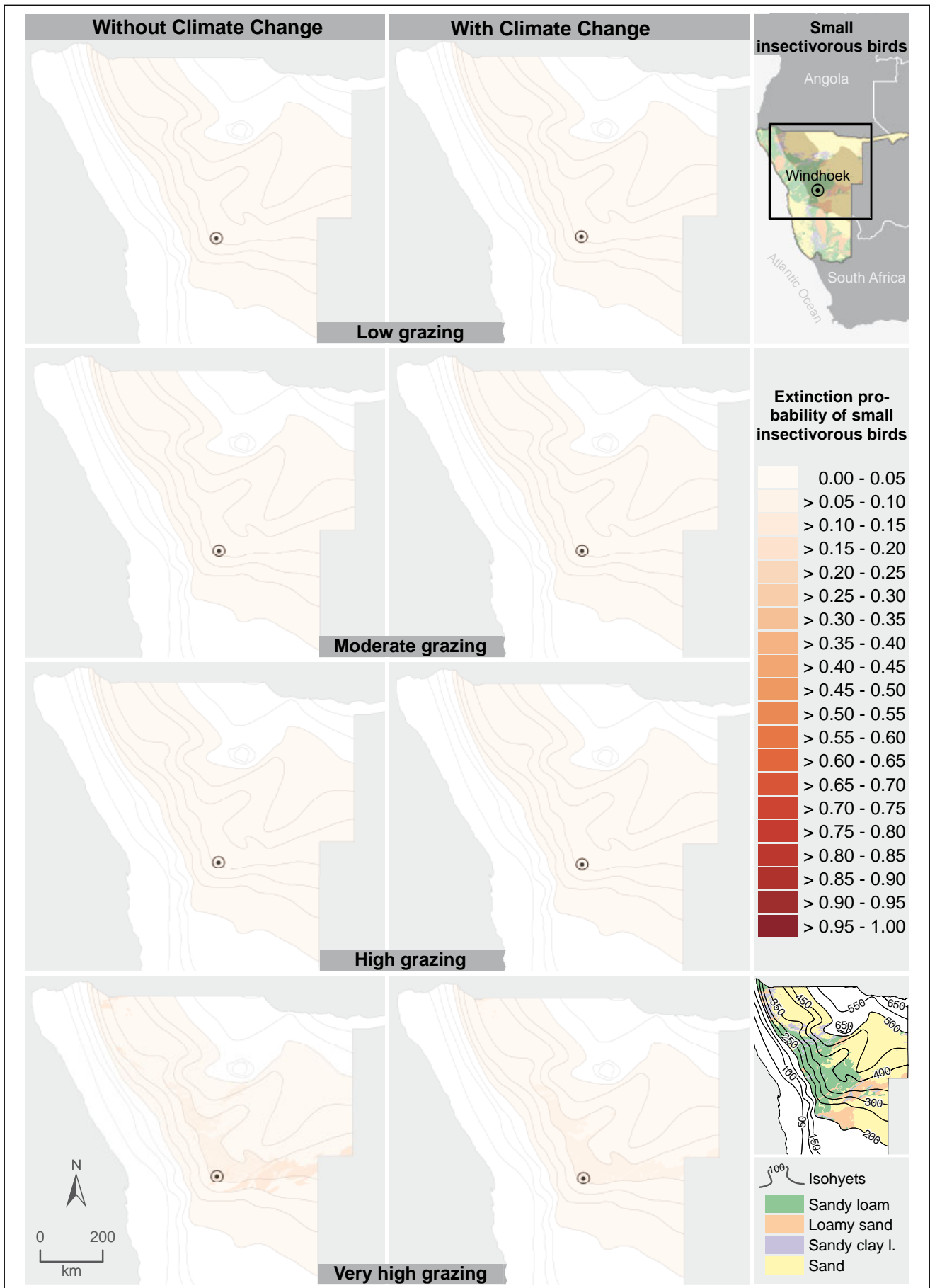
Fig. 25: Risk assessment maps for small (< 50 g) insectivorous birds in the focal Namibian savanna region based on predicted vegetation changes after 30 years for different grazing intensities with and without climate change. Colour shades indicate the extinction probability within the next 100 years, $p_0(100)$.

Fig. 26: Risk assessment maps for small (< 50 g) herbivorous birds in the focal Namibian savanna region based on predicted vegetation changes after 30 years for different grazing intensities with and without climate change. Colour shades indicate the extinction probability within the next 100 years, $p_0(100)$.

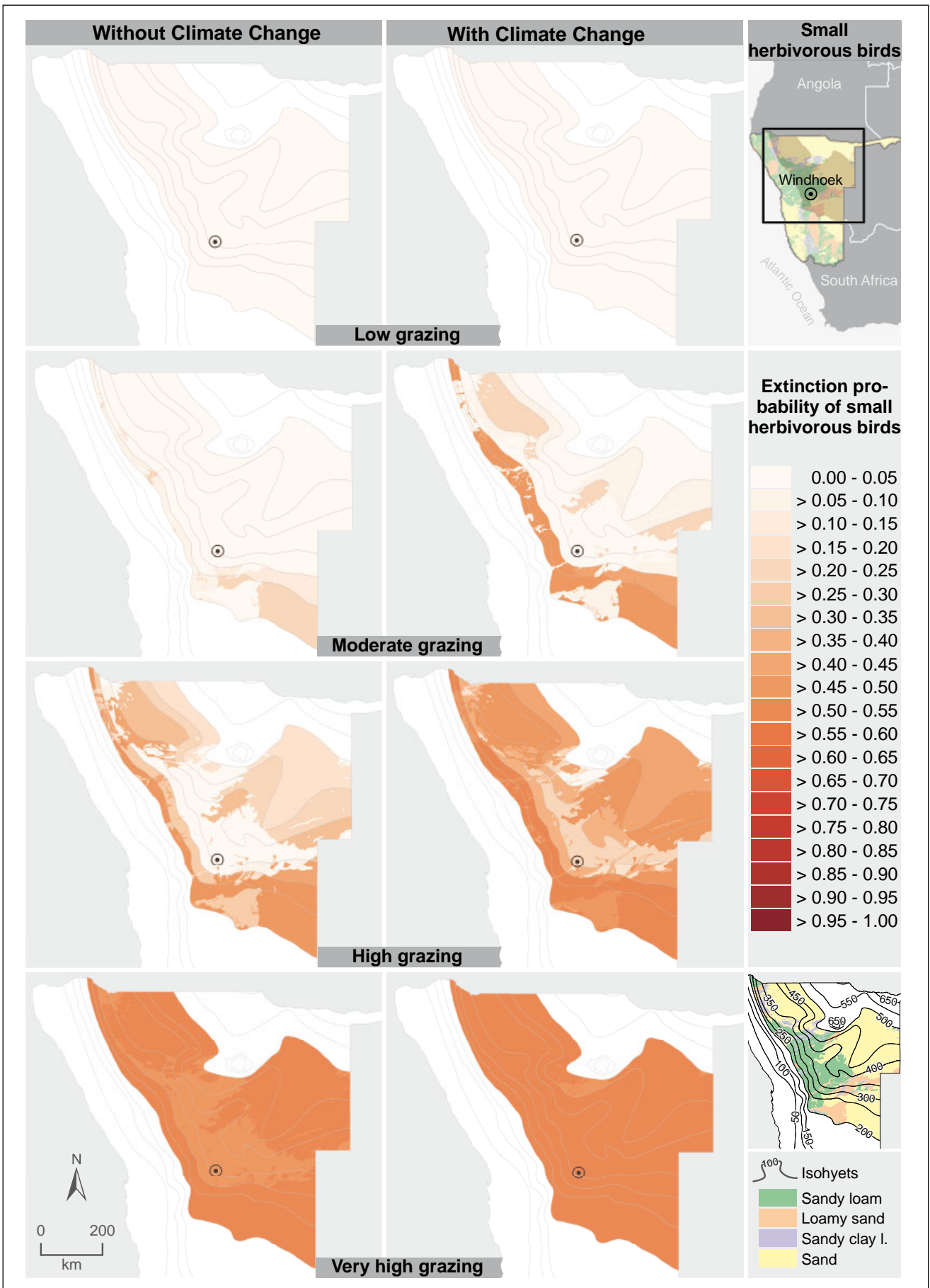## 2.6 Integrated socio-economic and ecological research

[D. Lohmann, T. Falk, E.-A. Nuppenau, E. Rossmanith, N. Blaum & F. Jeltsch]

Background

The semi-arid Thornbush Savannas of southern Africa are not only unique, diverse and fascinating ecosystems, they also contribute to the regional and national economy since they are widely used, for example, for livestock production and tourism. At the same time, these ecosystems are sensitive to over-utilisation resulting in ecological degradation and decreased capacity to deliver ecosystem services. The degradation of the vegetation (i.e. the loss of perennial grasses often accompanied by an increase of woody vegetation) causes a loss of palatable plant biomass with good nutritional value for livestock. It can further result in lower soil moisture levels caused, for example, by a decrease in water retention due to lower infiltration rates.

There are many reasons why some natural resource utilisation systems result in the degradation of savannas. For example, the limited availability of information on complex biotic and abiotic processes and feedbacks and short- or long-term consequences of specific management actions can promote degradation processes. The semi-arid Thornbush Savannas of southern Africa are highly complex eco-hydrological systems that are driven by highly erratic rainfall conditions (see e.g. Article III.7.2). Understanding observed dynamics and assessing possible future responses is a challenge for resource managers especially since human impacts are often only visible on large temporal scales. One objective of integrated research is therefore to present the key aspects of complex systems in a simplified way (e.g. by modelling or developing generalising theories) in order to make it easier for decision-makers to deliberate about the consequences of their decisions. But even if causal links of natural processes are clear, human decisions depend on the dynamics of complex social systems, which need to be understood as well. A typical challenge for sustainable natural resource management is the occurrence

of externalities. If a person does not have to cover the full costs or does not receive the full benefits of resource use then she/he might be more inclined to overexploit the resource. Cultural, customary, and statutory institutions have the objective to internalise externalities but often fail due to high transaction costs.

Therefore, developing well-adapted resource use strategies requires a good understanding of both the ecological as well as the socio-economic systems. This includes the assessment of the feasibility and acceptance of such strategies amongst relevant stakeholders. Fulfilling this task is only possible by strong interdisciplinary cooperation as has been practiced within the BIOTA project.

In the special case of the Thornbush Savanna that is addressed in this chapter, the main focus of ecological-economic research lies in the identification of sustainable use practices of the savanna ecosystem for livestock production. Socio-economic studies using the Institutional Analysis and Development (IAD) framework (Ostrom et al. 1994) as well as the Capital-Need-Institution framework (Falk 2008) assessed the farmers' capital availability, their landuse objectives, and the institutional set up as key determinants of resource use decisions. Empirical data were collected using in-depth case studies as well as semi-structured interviews. In addition, group discussions and participatory observations were utilised in order to learn about common practices and perceived challenges (see e.g. Article III.5.3 and III.7.5). Ecological studies are also available, which describe the drivers and dynamics of the shrub encroachment problems and vegetation dynamics in general (e.g. Article III.7.2 and Subchapter IV.2.5).

Integrated projects including both ecological and economic research have been conducted in the Thornbush Savannas of Namibia during the BIOTA project. Two relevant studies are introduced briefly below in order to depict the structure and potential of the conducted integrated projects. More detailed information about the results of these studies can be found elsewhere in this book (Articles III.7.5 and III.5.3) or in Buß (2006).

Integrated study 1: bio-economic modelling in central Namibia—using optimisation to identify suitable landuse strategies

This chapter introduces a study on optimal farm management in the Okahandja District of central Namibia that is based on an empirical study and the dynamic, non-linear programming of a bio-economic model (Article III.7.5, Buß 2006). The study in general deals with several questions related to sustainable rangeland management, one of which will be described here in order depict its methodology, structure and potential. The example that we present here contributes to an understanding of the potential influence of climate change on an optimal rangeland management strategy.

The empirical part of the study (consisting mainly of interviews) was conducted on regional "best-practice farms". These data, in combination with data from research stations in the area and information from local experts have been used to identify possible landuse options and their costs. In order to simulate the impact of management options on natural resource dynamics a state-and-transition conceptual model (Westoby 1989, Popp et al. 2009c, Joubert et al. 2008a, Article III.7.2) was applied. In cooperation with rangeland ecology experts (Rothauge 2002) different stable vegetation states and probabilities for transitions between these states have been defined. The various sets of management options and corresponding ecological impacts that were developed from this approach then served as the input for a dynamic bio-economic model using the General Algebraic Modelling System (McCarl 2002) in order to identify optimal management strategies. Simulations of a 10,000 ha farm for 30 years were conducted, which accounted for the average farm size of surveyed farms and a time horizon that is relevant from a landusers' but also from an ecological perspective. Stochastic rainfall time-series were also applied, since unpredictable rainfall is one of the key problems for landusers in semi-arid savannas.

In general, the results of such a data-based simulation-optimisation approach provide optimal landuse strategies for
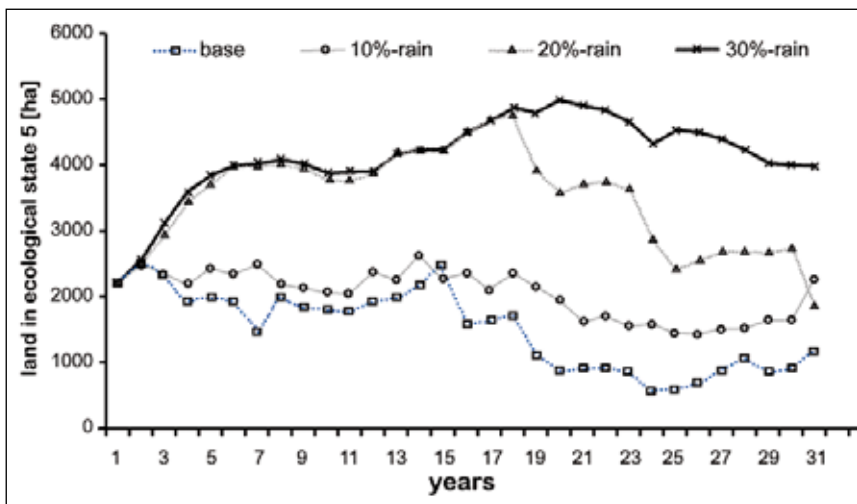
Fig. 27. Area of a farm under optimal management that is degraded (shrub encroached, state 5) for a baseline scenario and three scenarios with 10, 20, and 30% reduction of annual precipitation. Source: Buß (2006).

given objectives. Such objectives include, for example, the development of maximum profit, maximum ecological sustainability, or hybrids of these objectives. In more detail this means that the model determines the best option from the above described set of options for every decision in each of the 30 years.

The programmed model is capable of simulating different scenarios. As mentioned above we wanted to show only one snapshot result as an example here. Fig. 27 shows differences among optimal rangeland management strategies for four climate change scenarios regarding the fraction of the farm that is heavily degraded. Climate change is implemented by applying different rates of rainfall reductions. The graph shows the area, which is considered to be in a degraded state over time for the optimal management approach under a baseline and three climate-change scenarios. Results show that optimal management strategies comprise a certain level of ecological sustainability even though the amount of rainfall is reduced due to climate change. Only in the extreme scenario with an unlikely reduction of precipitation by 30% is it economically favourable to end up with a high fraction of the farm being in a degraded state.

In summary these results show, that ecological sustainability and economic success of a business are not necessarily contradictory. More detailed results for analyses of other management options and scenarios can be found in Article III.7.5 and in Buß (2006).

Integrated study 2: sustainable landuse on resettlement farms in the Omaheke Region in Namibia

Our second example is a study that was conducted in 2009 in the Omaheke Region at the eastern border of Namibia. This study aimed at identifying key problems of and revealing possible strategies for farmers that are farming on land that was redistributed as part of the Namibian Land Reform process. In this chapter we want to show how the economic output of a farm can be improved by applying more adaptive and moderate stocking strategies and that ecological and economic preferences need not necessarily be contradictory.

The study included three parts: 1) an economic survey including semi-structured interviews that identified the key socio-economic parameters of the resettlement farms (e.g. size of the farm, costs, income, indebtedness, ethnic background, number of livestock, etc.); 2) an agro-ecological estimation of range condition according to a standardised method of Zimmermann (2009) in order to quantify the ecological situation on the farms; and 3) an agent-based ecological-economic model of a livestock production farm. The details concerning the design and results of parts one and two can be found in Article III.5.3.

The ecological-economic model is based on an economic model for a single livestock producing farm and a well-established, eco-hydrological model (EcoHyd, Tietjen et al. 2009a) that was parameterised for typical soils and precipitation in the study region (see Article III.7.2) as well as on an agent- and rule based model simulating cattle herd dynamics, costs, profits and animal condition. The data that were needed to parameterise the model was taken from the survey mentioned above, so that the model simulations are representative for an average resettlement farm in the study region.

The model was designed so that it could also serve as a training tool (see Article III.7.6). A first version of the tool (and thus also of the model) was presented in pre-tests to resettlement farmers in advance, so that their feedback could be included in the model version that is presented here. However, in this chapter we present results of recently-repeated 50-year simulations with different, stochastic rainfall time-series. Results presented here are derived from simulations of a 3,000 ha farm producing cattle. We compared different strategies regarding absolute stocking rate, reaction to environmental variation and biennial resting of 50% of the farm in order to identify the economic (mean annual profit) and ecological (fraction of farm that is in a degraded state at the end of the simulation) consequences for a 50 year application of the respective management schemes.

Results show that generally moderate stocking rates result in highest mean annual profits for the farming business (Fig. 28a). The highest profit, however, was generated by the application of a biennial resting strategy, with a herd size reduction to 70% of the average herd size in drought years (< 300 mm). In contrast, a strategy with a fixed herd size and no application of resting resulted in significantly lower average profits.

Interestingly, the response of the vegetation to this management shows, that a more sustainable management, where the degraded fraction of the farm is significantly lower than in the non-resting scenario, leads to a higher average profit (see Fig. 28b). This means, that the ecologi-

cally more desirable strategy (biennial resting) also reveals the economically most beneficial output. However, all scenarios that have been feasible from an economic perspective (i.e. positive mean annual profit over 50 years) lead to a significant degradation of the savanna ecosystem. The calculated maximum annual profits show that the income level of the simulated resettlement farm is very low even in the long run. This result is consistent with the survey data (see Article III.5.3), which showed even lower profits with many respondents making losses. This might be due to the influence of poorly-adapted farming practices, with higher costs than necessary, having either too many or too few livestock, or grazing on a degraded ecosystem. The ecological, on-site analyses indicated that the ecosystem is already in a degraded and below-average ecological condition. This consequently only allows for even lower stocking rates compared to the ones that were given by simulation results, since our simulations are based on an initial moderate to good condition of the veld. Hence, the present interdisciplinary study clearly shows the necessity to further deepen the understanding of the situation of resettlement farmers and to develop and implement adequate schemes for the ongoing land reform process, including more detailed management and governance options like, for example, multi-farm cooperation in combination with spatio-temporal grazing schedules.

2.7 Recommendation for landusers and conservation planners

Relevance of fire management for bush control

[D.F. Joubert]

In historic times, rangeland utilisation by game, livestock, and pastoralists in Namibia was constrained by available perennial surface water. This allowed sufficient rest for the grasses. As a consequence the grass sward was dominated by vigorously-growing perennial grasses. Fires were more frequent, usually coinciding with the establishment of encroaching shrubs particularly in years
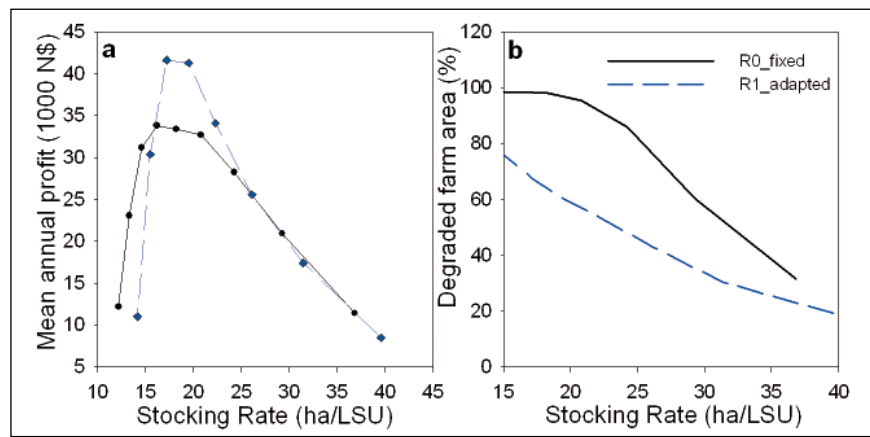

Fig. 28: Effects of stocking rates. a) Mean annual farm profit (N$1,000) and b) mean fraction of rangeland that is in a degraded state (%) of repeated 50 year simulations of a 3,000 ha resettlement farm in the Omaheke region in dependent of the average long term stocking rate (ha/LSU) for a fixed herd size scenario with no resting (R0_fixed) and a strategy that adapts herd size to droughts and applies a biennial resting (R1_adapted).

of high rainfall, thus maintaining an open savanna structure. In areas where natural perennial water occurred, such as around natural fountains, continuous grazing prevented extensive fires from occurring, and in some cases is likely to have resulted in encroachment, as witnessed by explorers as far back as the 1850s (Andersson 1856). Within the last century, boreholes and waterpoints have effectively made surface water available throughout the year and throughout the savannas, thus allowing sedentary game and livestock populations to graze more or less continuously. Grasses did not have a sufficiently long rest period to recover, which has resulted in earlier successional stage species to dominate, as well as a lower fuel load to build up. This in turn has resulted in a reduction in the number of fires. This reduction of fuel load, along with the deliberate exclusion of fire, has resulted in the periodic successful recruitment of woody shrubs on a landscape scale.

The exclusion of fire has been largely responsible for the transition of open savanna to shrub dominated savanna at critical times (Fig. 30). *Acacia mellifera* subsp. *detinens* has probably only undergone mass recruitment around six times in the Highland Savanna (adjacent to the Thornbush Savanna) since rainfall records in 1892 (Joubert et al. 2008a). This would also reflect the situation in the Thornbush Savanna. Despite the few

chances for recruitment, *A. mellifera* is dominant in the landscape in most of the Thornbush Savanna. *A. mellifera* only produces seeds en masse during times of particularly high rainfall. This needs to be followed directly by at least another season of high and evenly spread rainfall to ensure establishment (Joubert et al. 2008a). This is because seeds germinate easily and thus seed banks are ephemeral (Joubert et al. 2008a). Field observations suggest that a month of almost daily effective rain, just prior to the seeds having fallen, as was witnessed in January 2010, allows seedling establishment, even in average rainfall years. Early evidence from BIOTA research shows that competition with climax grasses has little effect on seedling vigour, but that fire may completely prevent an establishment event.

Rangeland managers need to consider applying well planned fires during times of *A. mellifera* establishment. Fortunately, conditions for establishment coincide with ideal conditions for burning since high rainfall events produce good fuel loads provided the veld is well rested. Based on the frequency of recruitment events, a farmer is likely to only need to burn each portion of land every 20 years or so (Joubert et al. 2008a). Slight decreases in productivity in the season following a burn are more than offset by the long term maintenance of high grass production due to the prevention of shrub encroachment. Decision support
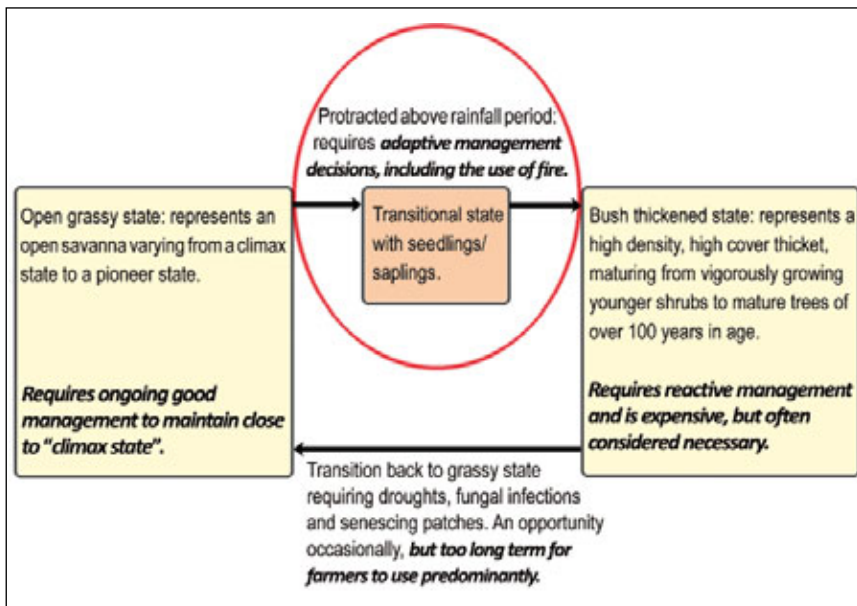
Fig. 29: A simplified version of the conceptual state-and-transition model for Thornbush Savanna.

systems such as wikiChameleon (Joubert et al. 2008b, 2009) are useful in assisting farmers in taking decisions regarding shrub encroachment. This is described in Article III.8.5.

Identification of key/indicator species for vegetation states—input for land managers to evaluate the ecological condition of their rangelands

[M. Strohbach]

The absolute definition of vegetation states according to species composition and the percentage cover of such species will remain problematic in the Namibian Thornbush Savanna, as both variables are, apart from rangeland management practices, strongly influenced by the timing of rainfall and soil type, especially at the soil surface.

Co-occurrence analysis with the software packages EcoSim and COOC, together with the analysis of Plant Functional Types (PFT's) showed that species interactions in the savanna are much more complex than those that exist between shrubs and grasses only. Different types of grasses, for example, are also strong competitors, and this is strongly influenced by grazing—not just overgrazing, but also selective grazing that occurs during the under-utilisation of the veld. From preliminary results it can be assumed that

the dynamics of the herb layer strongly influences the stability of the topsoil. The loss of topsoil further influences the persistence of herbaceous species, and their loss may enable more shrubs to become established. Shrubs, in turn, have a major influence on the persistence and composition of herbaceous species. However, this is not just dependent on shrub density alone, but also on the height and actual shape of the shrubs. This can be seen in Figs. 30 and 31, which show a principal component analysis of grass, and sedge PFT's, together with a second habitat matrix.

To evaluate the ecological condition of rangelands, a practice of annual fixed-point monitoring over several years must be followed as a single year evaluation will be misleading. The three variables that will need to be monitored are: species composition (simplified with the use of PFT's), the percentage cover of these PFT's as well as soil surface condition. The most suitable method for such monitoring is the Landscape Function Analysis (LFA) approach along transects, but incorporating PFT's to describe the vegetation more accurately. Very importantly, the LFA needs to be conducted during the dormant season, which in the Namibian Thornbush Savanna is from July to August. The variables that will give the best indication as to the deterio-

ration or regeneration of rangelands are as follows:

• Shrub cover: Whether thorny or non-thorny, a sudden increase in young shrubs less than 60–70 cm in height, indicates a definite ongoing degeneration of the veld. It is at this time that active intervention against the shrub cover must be undertaken, as this is the stage where shrub/grass dynamics undergo their first relatively significant and usually irreversible changes. A high cover of tall shrubs will result from a long period of continued, unchecked degradation, and is usually associated with a considerable loss of original topsoil as well as the seed banks that were present in these topsoils.

• Types and cover of perennial grasses: The most productive veld has a relatively high and diverse (i.e. more than two species) cover of perennial grasses with very dense basal tufts and either very fine leafy culms and non-spiny seeds (e.g. *Stipagrostis uniplumis*), or short culms with long, relatively broad leaves and only the flower-bearing culms being relatively thick (e.g. *Brachiaria nigropedata*). A first sign of degradation may be the gradual increase of perennial grasses that either have a strong tuft with relatively thick, leafy culms or very low, small tufts with many long flowering culms bearing spiny florets (e.g. *Aristida congesta*). A low cover of perennial grasses overall will indicate that the veld is most likely prone to future shrub encroachment or it is at risk of becoming completely degraded.

• During the growing season, the type and amount of annual grasses and herbs may also give an indication of veld condition. Should the veld be dominated by one or two specific species of annual grasses or herbs, this may be an indication of how the rain has been distributed in that particular year. If, however, a similar dominance of annual species is present every year, especially weedy species such as *Tribulus* spp., *Tagetes* spp., or bulbous species, or when annual grass and herb species are far more common than desirable perennial grasses, then this indicates that the veld is relatively degraded and has a low productivity.

Fig. 30: The association of different grass and sedge Plant Functional Type (PFT) groups with the percentage cover of tall shrubs (mostly Acacia mellifera subsp. detinens), as shown by PCA. Within each PFT group, only a few species actually prefer a high cover of tall shrubs as indicated by a reduction in the size of the symbol. Thus, as shrub cover increases, most grass species disappear.
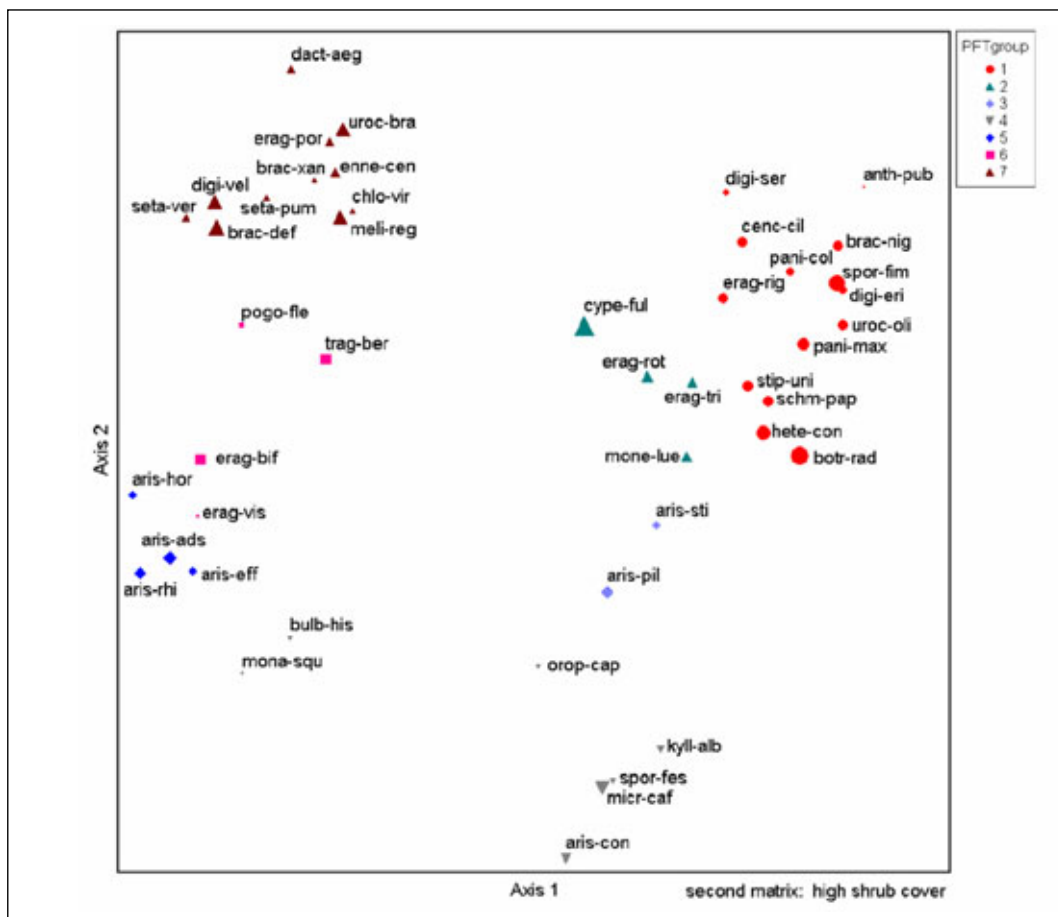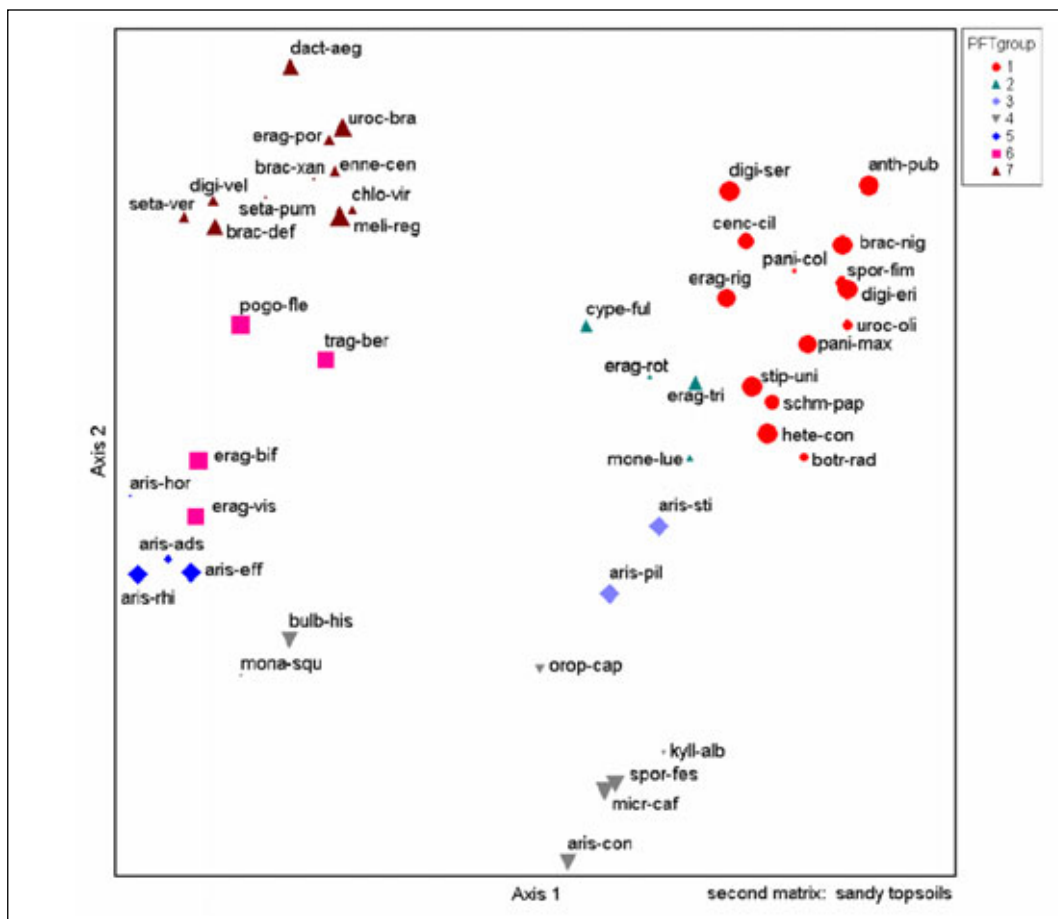

Fig. 31: The association of different grass and sedge PFT groups (the same as in Fig. 30) with soil surface texture and in particular, the dominance of sandy topsoils. Only a very few species actually prefer loamy topsoils as indicated by a reduction in the size of the symbols.

Grazing management during droughts

[M. Strohbach]

Palatable perennial grasses appear to be more abundant and more resilient where topsoils are relatively sandy (see 'Impacts of rainfall on vegetation cover' in Subchapter IV.2.4). This has important implications for management decisions in an area. In addition, the recovery of perennial grasses, if grazed excessively during a drought period, is not nearly as rapid, within one season, as many farmers would hope, even with sufficient rainfall. This is most evident on the loamier and hydrologically less favourable soils. Therefore, erosion of sandy topsoils should be prevented as far as possible, either through the rapid reduction of stocking rates during years of drought or by avoiding extensive grazing in those camps, which possess a high cover of palatable grasses. Most important, however, is that long-term stocking rates for an area should be determined according to the average availability of grazing biomass during the dormant season and not during the peak growing season, as is still standard practice with most framers and agricultural authorities.

Guidelines on best-practice restoration techniques in Namibia

[M. Strohbach]

Restoration experiments carried out on the Etjo Erosion plains highlighted the importance of the following best practices restoration techniques:

- It is imperative that the landuser understands what is happening in the field and is able to provide answers to questions such as: Is there sheet erosion? Does rainwater infiltrate easily or does the majority run off and if so, in which direction? What type and how much perennial grasses are there? What other vegetation, if any, is there in the camps?

- For any effort for veld regeneration, it is much more viable to create many smaller, 'fertile islands' comprised of strips of about 50 m by 3–5 m, that are distributed throughout a camp or farm, rather than to try and restore an entire continuous area. During adverse years, a mosaic of runoff and run-on-patches will remain much more resilient than uniformly homogenous areas with only a low cover of plants.

- Soils that are bare and unvegetated usually have a relatively thick, impenetrable upper layer that often forms an ecological barrier to the establishment of plants. This layer needs to be ripped open to at least 25 cm depth, preferably more. It will be more effective to do few, deep rips with a single ripping implement than many shallow rips.

- No matter how flat the area appears to be, it is advisable to always rip along a slight contour, with the lowest point in the direction of the water runoff. In this manner resources such as soil, seed and organic matter, which are washed off bare areas become trapped and accumulate to help enrich the restored patches, which can then grow bigger by themselves over time.

- Often, it may be necessary to sow in seeds of perennial grass species. It may be possible to purchase some seeds, but then it is advisable to use species always that occur naturally in the area. Road verges are usually a good source for collecting seeds. However, since all grass seeds have an inherent dormancy of 7–10 months, they should be collected during the season prior to when the ripping takes place.

- Timing: rip early in the growing season, preferably just after and within days of the first rains (> 5 mm). If you sow in grass, do so very sparingly to avoid creating patches of preferential grazing. After sowing loosely, cover the seeds with cut bushes to protect the seedlings from being grazed during the first year.

Summary of recommendations for farmers

[I. Zimmermann & G.N. Smit]

Estimation of grazing capacity

Due to the fluctuating nature of Namibia's climate, it is essential to include provision of a drought reserve when estimating the grazing capacity of the rangeland. If the stocking rate needs to be reduced, the best time to do so for the benefit of rangeland condition and animal performance is at the start of the dry season, although seasonal changes in price may sometimes warrant holding on to animals for longer. If the stocking rate can be raised after a drought, it may be helpful to wait until perennial grasses have had an opportunity to regain vigour before increasing the stocking rate. It is recommended that farmers who have not yet gained experience in estimating the grazing capacity by visual judgement should initially apply a quantitative method such as that of van Wyk (1988) before progressing to the use of reference photographs (Klintenberg et al. 2007).

Provision of rest

Periods of absence should ideally be flexible so that they may be adjusted by adaptive management. The condition of grass in the destination paddock provides a more valuable indicator of when to move, for the health of the grass, than the condition of grass where livestock are currently grazing, although the latter may have a greater impact on animal performance. If periods of absence during the growing season need to be fixed in advance, then longer periods are better than shorter periods, during which time there might not be enough rain for grazed grasses to regain vigour. Annual resting, such as proposed by Dames (2009), may hold promise as a simple grazing strategy that cares for both rangeland and animal condition. Where farmers have insufficient fenced paddocks for provision of adequate rest, they could combine their herds with neighbours and rotate through all their paddocks (Kruger 2008) or apply herding on unfenced land (Kangombe & Kapi 2008).

Trampling

Farmers should pay close attention to the effects of trampling on their farms, in different seasons and on different soils, to determine whether it could be applied to help certain parts of their rangeland. They need to bear in mind that long rest should be provided after trampling and that animal performance will be sacrificed to some extent.

Burning

Farmers should also determine whether the occasional use of controlled burning could help a portion of their rangeland,

whether for the control of bush saplings, the rejuvenation of perennial grass, or the creation of nutrient hotspots. If fire is considered helpful, then the conditions required for successful burning will either need to be in place, waited for or encouraged to develop, such as sufficient fuel and sufficient residual water and organic matter in the soil.

Bush control

It is recommended that farmers consult the decision support system (Joubert et al. 2009) accessible at: http://chameleon. polytechnic.edu.na/wiki/. This should preferably be to prevent bush encroachment from taking place, by applying management closer to the root causes in a diagnostic problem tree (Zimmermann et al. 2008). However, in cases where the symptom of bush encroachment needs to be treated, the decision support system mentioned above provides links to a data base of experiences by Namibian farmers (Barac et al. 2004). If arboricides are to be used, the risk of killing non-target bushes and trees should be minimised.

Ground water

Management of ground water, from where most farmers obtain drinking water for their animals, is crucial for the sustainability of farming. It is recommended that farmers monitor their ground water levels and, where they are found to be lowering, farmers should consider taking action to recharge the aquifer such as reducing bush cover (Bockmühl 2009) or aligning roads towards recharge sites.

Parasite control

To avoid harming non-target organisms, such as dung beetles, that provide essential ecological services for rangeland and animal health, farmers should control parasites without, or with minimal use of, toxic chemicals. Those who are unable to apply rotational grazing to disrupt parasite life cycles, and those who are still forced to kraal their animals at night, could at least try non-toxic treatments or chemicals that result in minimal harm to dung beetles, while at the same time making an effort to eliminate animals with high parasite loads from the herd.

Vision of achievable rangeland condition

Due to the fluctuating nature of Namibia's climate, the condition of even the best rangeland changes annually. It is useful for farmers to know what condition of rangeland is achievable under the prevailing climatic conditions. Such a vision could be provided by a benchmark site, where the same overall stocking rate as the surrounding area is applied twice a year, followed by long periods of absence. It is therefore recommended that farmers establish small benchmark sites for their rangeland so that they may differentiate between the effects of climate and management. Apart from providing the farmer with information upon which to base management decisions, the benchmark may serve as a source of seed of good grass species, for spreading out into the surrounding rangeland.

Financial management

Farmers could benefit by examining their financial records, to determine whether their expenses are justified, if they are not already doing so. If farmers focus on reducing non-essential expenditure they are likely to increase profitability (von Alten 2008). Before making any new investment it is recommended that farmers perform cost-benefit analyses on the intended investment and on alternatives, to determine which brings the best returns per dollar spent, especially when those dollars need to be repaid at interest. Reliance on overstocking to increase profitability provides only short-term relief, as the resulting reduction in carrying capacity limits this option for the future. Farmers who obtain poor prices for their products should seek alternative markets, including those that pay premium prices for "green" products raised on natural rangeland.

Monitoring

It is also recommended that farmers undertake more formalised monitoring together with record keeping of their rangeland. This could be as simple as taking regular fixed point photographs at labelled sites and perhaps estimating abundances of bushes and grasses at regular intervals. Those who are more adventurous could back up the visual impressions with measurements. Farmers who co-operate with neighbours to form study groups can benefit by the sharing of information and joint monitoring of their test sites.

Benefiting from spatial heterogeneity

Farmers should exploit the spatial heterogeneity that occurs within their farms, and consider movements of livestock off the farm at critical periods. Fynn (2009) proposed the re-introduction of transhumance pastoral migrations between natural wet and dry season grazing areas where unfenced land still exists. In situations where land subdivision is irreversible, he proposed cooperation between neighbouring farmers to jointly manage their herds over larger areas, such as the practice of agistment applied by some Australian farmers (McAllister et al. 2006).

Conservation of biodiversity

[N. Blaum, A. Horn, F. Jeltsch, E. Rossmanith, R. Simmons & C. Seymour]
BIOTA results in Thornbush Savanna rangelands show that grazing induced shrub encroachment threatens species diversity of plants and animals across taxonomic groups (details are described in Subchapter IV.2.4). Nevertheless, grazing is not negative for species diversity per se. On the contrary, in Kalahari Thornbush Savanna rangelands for example, moderate livestock grazing has been shown to have a positive effect on species richness of different arthropod taxa, rodents, and small and medium sized carnivores (e.g. Blaum et al. 2007a, b, c, 2009a, b). This is an important finding for both rangeland managers, and conservation planners. In particular, when planning shrub removal programs that aim to increase grazing capacity, some shrubs should be kept to sustain species diversity in Thornbush Savanna rangelands. Although single shrubs do not comprise hot spots of species diversity they serve key functions for many plant and animal species as they provide shade, shelter and may also serve as nest sites for some bird species (for details see Subchapter IV.2.4).

A key finding for sustaining bird diversity in arid Namibia was that land managers and conservation managers should reduce the extent of bush thicket areas and

conserve all dry riverlines. Bird diversity decreased in bush thickets while open (grassy) areas showed marginal declines and dry riverlines showed no declines in avian richness (see 'Identifying key patterns' in Subchapter IV.2.4 for details). This indicates that sensitive riparian environments are key habitats sustaining bird diversity. Furthermore, given that birds appear to be attracted to tall trees across all savanna types (Kaphengst & Ward 2008, Seymour & Dean 2009), land managers are encouraged to retain the taller trees when thinning bush-encroached areas. This would not only benefit biodiversity but would also provide shade for livestock and contribute to nutrient cycling processes within rangelands soils.

2.8 Future research needs

Background

[F. Jeltsch, N. Blaum, E. Rossmanith, D. Lohmann, D.F. Joubert, I. Zimmermann, R. Simmons, C. Seymour, C. Grohmann, A. Horn, C. Reisch, P. Poschlod, M. Strohbach & D. Wesuls]

BIOTA research has significantly improved our understanding of how populations, communities, and biodiversity may respond under different landuse options and climate change scenarios in African Thornbush Savannas. The key scientific results and simulation models were further developed into several educational and management tools for rangeland management and biodiversity research (compare Article III.7.6). On this basis, we identified three major topics for future research needs, which require inter-, but particularly transdisciplinary approaches: (i) linking dynamics landscapes with biodiversity dynamics, (ii) understanding the impacts of landuse options and climate change on ecosystem functions and services (EF&S), and (iii) restoration of degraded savanna rangelands.

Linking dynamic landscapes with biodiversity dynamics

A next crucial step in biodiversity research in Thornbush Savannas is the development of a mechanistic understanding of biodiversity dynamics under the coupled effects of landuse and climate change within a dynamic savanna landscape. For example, dry riverlines support higher bird diversity and riparian vegetation habitats are at least 3°C cooler at midday than surrounding bush-encroached and grass-dominated savanna habitats (Seymour & Simmons 2008) (see 'Identifying key patterns' in Subchapter IV.2.4). Although this clearly indicates the importance of conserving riverine habitats for sustaining bird diversity, the dynamics of habitat mosaics and bird diversity remains unclear and makes solid management recommendations difficult.

Although biodiversity can play a crucial role for ecosystem functioning, the effects of spatio-temporal changes in biodiversity pattern on ecosystem functions in changing landscapes are largely unknown (Lundberg & Moberg 2003, Srivastava & Vellend 2005, Duffy et al. 2007, Cardinale et al. 2009). In particular, migration and spatial avoidance are key features of adaptation to highly frequent disturbances in anthropogenic landscapes.

Land management that largely differs in space and time will affect biodiversity at different trophic levels with implications for trophic interactions. For example, climate or landuse impacts can affect predator densities and foraging strategies that are characterised by predator-specific animal movements. Such changes in predator densities or movement behaviour for foraging are likely to modify the abundance of small mammals, dung beetles etc., which provide tunnel systems and perturb the soil and may affect ecosystem functioning by, for example, changing infiltration rates. Another example is the spatial-temporal change in the diversity of pollinators, which provide important ecosystem services.

A mechanistic understanding of biodiversity dynamics, trophic interactions, and their consequences for ecosystem functioning (see below) requires the explicit consideration of dynamic landscapes, vegetation and landuse patterns at different scales and most importantly the simultaneous investigation of multiple interacting species or functional types. This approach will allow for the identification and testing of land management options, which support the functioning of healthy ecosystems and which are adapted to the predicted changes in climate.

Understanding the impacts of landuse options and climate change on ecosystem functions and services (EF&S)

Rangeland degradation in south-western Africa is significant and in Namibia alone about 50% of the savanna areas are purportedly affected (Bester 1998). A variety of savanna studies have analysed the available options to stabilise or restore specific ecosystem services by sustainable land-management systems and appropriate governance arrangements. In most studies the main focus is either on the impact of land degradation on forage and livestock productivity (e.g. Hudak 1999, Briske et al. 2008) or on biodiversity (e.g. Richardson 1998, Blaum et al. 2007a, b, c, 2009a, b). However, there is a lack of studies, which focus simultaneously on multiple EF&S, and only a very limited understanding exists of the trade-offs between different socio-economic and ecological management options and aims. For example, the build-up of woody biomass supports an increase in carbon stocks and thus a reduction in global greenhouse gas emissions (Grace et al. 2006). On the other hand area-wide bush encroachment causing a significant reduction of economic profitability (e.g. livestock production) (Grace et al. 2006), and biodiversity (Richardson 1998, Blaum & Wichmann 2007, Blaum et al. 2007a, 2009a, b). Furthermore, the increase of woody vegetation can alter the regional water balance by increasing evapotranspiration and reducing groundwater recharge (Simioni et al. 2003). This clearly shows the strong trade-off between provisioning services such as livestock production and regulating ecosystem services such as carbon sequestration and groundwater recharge and biodiversity. It also shows the scale-dependence of competing services and related stakeholder interests with local and the shorter-term interest of farmers involved in livestock production and global and longer-term benefits evident in carbon sequestration.

The global character of externalities requires global governance instruments of benefit-sharing and compensation. However, it remains unclear as to how land-

use optimisations towards either direction may affect other ecosystem services and their complex interdependencies. A similar trade-off exists for farmers who rely on income from wildlife species. The management of endemic wildlife for hunting, meat production and ecotourism (Barnes & de Jager 1996, Rao & McGowan 2002) becomes an increasingly important landuse option and is also assumed to contribute to the control of undesirable growth of woody species (McGranahan 2008). However, when wildlife populations increase to unsustainable levels as observed, for example, on many private lands in Namibia where there has been an increase of approximately 70% between 1972 and 1992, the risk of overutilisation of grazing or browsing resources increases significantly (McGranahan 2008).

Future landuse options also include the production of biofuels in the form of fuel wood and charcoal production, which is suitable for marginal lands (Koiwang 2000). The removal and use of undesired woody vegetation for energy production can contribute to the restoration of degraded areas for future livestock production, but it is unclear what effects it has on other services such as soil quality, carbon sequestration as well as on biodiversity. It is also not known whether such practices are sustainable in the long term, especially given the fact that growing urban demands might cause an overutilisation of all types of woody vegetation including ecologically-important trees (Arnold et al. 2006).

Setting the right incentives for ecosystem management, which maximises the equal distribution of welfare and benefits at local, national and global levels, requires a good understanding of the costs and benefits experienced by different stakeholders. Current behaviour is adapted to market mechanisms such as prices and capital access, as well as formal and informal social rules. Too often prices do not cover all the costs associated with landuse impacts such as the costs that are put on current neighbours and on future generations through the impact of different landuse practices on EF&S. If prices for resource use are too low, overutilisation is to be expected (compare Worldbank 2008).

The extent of this overexploitation depends on additional factors. Tenure systems have an influence by assigning property rights and, thus, allotting personal interests in current and future returns to different landuse options and intensities. In addition, tenure systems determine who is making actual decisions and this results in system-dependent learning curves, which in turn may influence future landuse options (Popp et al. 2009b). The social organisation of families and local units also determines the extent to which future generations are included in decision-making processes.

Restoration of degraded savanna rangelands

Efforts to restore shrub encroached land in Namibia are escalating, yet the impacts of these measures on biodiversity and ecosystem functioning have largely not been tested (Joubert & Zimmermann 2002). There is thus an urgent need to focus research on the impacts of different restoration measures and strategies on biodiversity and ecosystem functioning.

From a rangeland manager's perspective, bush encroachment reduces livestock production. For example, an estimated N$700 million per year of beef production has been lost in Namibia due to a decline in rangeland grazing potential (de Klerk 2004) and by 2009, this figure has likely doubled. Bush encroachment also negatively affects other ecosystem functions and services, such as the replenishment of groundwater, forage production and biodiversity (Barnard 1998; see also previous chapters and compare Article III.7.3). On the other hand, however, bush thicket patches in a matrix of open savanna can also provide important functions for biodiversity and soil quality (Mills & Fey 2004) and landscape scale bush thickets were already a feature of the landscape in the 1800s (Andersson 1856). Concerns over the economic costs of bush encroachment have led to wide-scale clearing in the focal area (van den Berg & Kellner 2005), along with the utilisation of the cleared wood, particularly for charcoal. The demand for wood for other products, including electricity is increasing rapidly.

However, the impacts of bush control on ecosystem functions and services have not been adequately analysed, nor has the sustainable success of debushing strategies been monitored and achieved as yet. In order that future restoration is both economically and ecologically sustainable and truly restorative, and not damaging to ecosystem functions and services, this lack of documented information needs to be addressed. In addition, innovative restoration techniques, such as the manipulation of soil chemical and physical properties including soil nutrient levels, need to be evaluated experimentally. The edaphic drivers of bush encroachment are largely unknown, thus there is a need to research the differences in soil properties between bush-encroached landscapes and those that are not bush-encroached. The implementation and adoption of Bush Expert (Barac et al. 2004) and other Decision Support Systems (DSS), such as wikiChameleon (Joubert et al. 2008b) is limited due to a lack of awareness and training opportunities. A consolidation and coordination of existing DSS approaches is likely to facilitate this sustainable development. Such approaches could be supported by best practice demonstration sites, where successful farmers show how they achieve good rangeland condition and buffer drought. In this way, other farmers could be encouraged to apply such strategies.

Bush encroachment is largely related to poor grazing management, which may include both under or overgrazing (e.g. Roques et al. 2001). Sustainable restoration of bush-encroached areas, however, cannot be undertaken if the root cause of bush encroachment is not addressed. Over the last 100 years Africa has switched from dominantly large-scale migratory grazing to small-scale sedentary or ranch-style grazing (Fryxell & Sinclair 1988). The scale at which grazing occurs has large effects on grassland productivity with small-scale ranches generally exhibiting a decrease grassland productivity and large-scale migratory grazing exhibiting an increase grassland productivity over time (Milchunas & Lauenroth 1993, Frank et al. 1998). Moreover, it has been demonstrated that grazing cattle at the same stocking rate but changing the scale at which they graze a particular area through the manipulation of herd size, density, and movement, for example, can

result in dramatically different effects on grassland composition and woody encroachment (Fuhlendorf & Engle 2004, Zimmerman & Smit 2008). This is consistent with the prediction that scale may provide a unified understanding of the response of various ecosystem properties to some environmental influence (Sandel & Smith 2009). A deeper understanding of the influence of the scale of grazing management on the tree/grass balance is important.

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