

Rangeland use in Northwestern Namibia

**An integrated analysis of vegetation dynamics,
decision-making processes and environment perception**

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Jenny Eisold
aus Minden

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Berichtersteller:

Prof. Dr. Helmut Hillebrand

Prof. Dr. Michael Bonkowski

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Hyaku-bun wa ik-ken ni shika-zu

(Einmal sehen ist mehr als hundertmal hören,
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Abstract

Degradation of natural resources has become a serious challenge in rangelands, bearing negative impacts on the pastoral ecosystems, livestock production and livelihoods (Vetter 2005; Kassahun 2008). In Namibia, the driest country of sub-saharian Africa (Hutchinson 1995) 45 per cent of the national land area can only be used as rangeland (Mendelsohn 2003). Many Namibians are pastoralists, whose livelihoods mainly depend on natural rangeland vegetation and water resources. 43 per cent of Namibia's land surface is used by 90 per cent of the population under communal land tenure (Dewdney 1996; Seely 1998). This socio-economic importance makes the sustainable use of the rangeland essential (Kassahun et al. 2008). Furthermore northern Namibia will be heavily affected by climate change (Beyer 2001).

Integrating local and scientific environment perception, decision-making processes of the local land users and the impact of land use practices on vegetation dynamics into ecological research becomes increasingly important in today's complex web of social, ecological and political changes. This thesis develops an integrated approach focusing on ecological aspects in a social-ecological rangeland system in Northwestern Namibia.

Firstly I developed a methodological approach for the synthesis of local ecological knowledge (LEK) and scientific knowledge. I found that to local livestock herders, woody species are more important than herbaceous species, what does not correlate with species' ecological performance in the grazing area. I hypothesise that reliability of forage resources in times of scarcity is important in local perception. This shows that integrating LEK on ecological items into ecological research helps to identify criteria, or indicators for local management decisions.

In the second step I matched local knowledge and local management decisions. In dealing with scarce and variable natural resources it is crucial to acquire a functional understanding of the interactions between management strategy and the mechanisms which buffer the variability of rainfall. I differentiated between abiotic buffers (key resource areas) and biotic buffers (storage tissue and stockpiled forage). The two biotic buffer mechanisms can be managed via herd mobility. I analysed how mobility is connected to the supply of the eco-

system goods 'water' and 'forage'. I related the mobility decisions of local herders to the biotic and abiotic buffers of pastures used to see if they made use of pastures with key resource properties. In the communal areas, the main decision factor for mobility was permanent water availability and herders mainly follow short-term management objectives. Their set of options may be limited by influences that go beyond ecological conditions, such as land pressure, a non-adapted institutional framework and limited options for sustaining their livelihoods. This section shows that to grasp the essential elements of range management it is important to understand user objectives in deciding when and where to move in times of scarce resources.

Reliability, which is perceived as an important criterion for the quality of forage resources, has a lot in common with the concept of key resources, which are defined as forage resources available in times of resource scarcity. While other authors have a descriptive approach to the reliability of key resources, I functionally defined them as biotic and abiotic buffers. In contrast to a common view in rangeland science decision-making of local pastoralists does often not reflect the availability of key forage resources but of the second essential resource for livestock, such as drinking water.

In the third step I evaluate the overall impact of the grazing practices on the vegetation and how to indicate ecological thresholds. Direct measures for system shifts are difficult to obtain, due to multiple factor controls that operate at diverse spatial and temporal scales. Consequently, there is a need for the development of indicators to determine if an ecosystem is approaching a threshold. I aimed to identify potential early-warning indicators and long-term indicators for crossing a degradation threshold in a semi-arid African savanna.

I identified indicators for changes in a rangeland ecosystem on different levels. Long-term indicators were identified as the structural shift from grassland to woodland indicating a major shift in the supply with grazing resources, that indicate major ecological restructuring on a functional level symptomatic of land degradation (Reynolds and Stafford 2002; Scheffer et al. 2009). Furthermore early warning indicators, the ratio between annual and perennial grasses were characterised.

The local range use strategy may not be adaptive, but seems to be the only possible adaptation of local users to the current ecological and socio-economic situation because options for action are restricted. Since the starting point for

the analysis of social-ecological system functioning is the sustainable provision of ecosystem services and the maintenance of livelihoods, this integrated analysis is important to conceptualize the effects of environmental variability, the supply of ecosystem services, and to connect this to management rules.

Progress in rangeland ecology can be achieved by integrating local knowledge, local decision-making processes with regard to their drivers and the impact of land use and environmental variability on the natural resources. If integrated analysis is to meet the challenges facing rangelands, users and scientists, theory and application have to be linked by a common set of objectives in which monitoring could become a vital tool in the quest for sustainable rangeland use.

Zusammenfassung

Die Degradation natürlicher Ressourcen stellt mit ihren negativen Auswirkungen auf das pastorale Ökosystem, den Tierbestand und damit die Existenzgrundlagen der lokalen Bevölkerung (Vetter 2005; Kassahun 2008) eine große Herausforderung für aride und semi-aride Weidegebiete dar. In Namibia, dem trockensten Staat des sub-saharischen Afrika (Hutchinson 1995), können 45 Prozent der Landfläche nur als Weideland und nicht für die Pflanzenproduktion genutzt werden (Mendelsohn 2003). Viele Namibier sind Pastoralisten, deren Lebensunterhalt hauptsächlich von Wasserressourcen und der natürlichen Vegetation des Weidelandes abhängt. 43 Prozent des Kommunallandes in Namibia werden von 90 Prozent der Bevölkerung genutzt. (Dewdney 1996; Seely 1998). Dieser starke Landdruck und die Tatsache, dass das nördliche Namibia stark vom Klimawandel betroffen sein wird (Beyer 2001), machen eine nachhaltige Nutzung der Weideland-Ökosysteme unabdingbar (Kassahun et al. 2008).

Das Integrieren von lokaler und wissenschaftlicher Umweltwahrnehmung, das Einbeziehen von Entscheidungsfindungsprozessen der lokalen Landnutzer und die Analyse der Auswirkungen der Landnutzung auf Vegetationsdynamiken durch ökologische Forschung gewinnen in dem komplexen Netz aus sozialen, ökologischen und politischen Veränderungen zunehmend an Bedeutung. In der vorliegenden Arbeit wurde demgemäß ein integrierter Ansatz mit einem speziellen Fokus auf ökologischen Aspekten in einem sozial-ökologischem Weidesystem im nordwestlichen Namibia entwickelt.

Zunächst wurde ein methodischer Ansatz für die Synthese von lokalem ökologischem Wissen (LÖK) und wissenschaftlichem Wissen entwickelt. Damit konnte herausgefunden werden, dass holzige Pflanzenarten für die lokale Hirten von größerer Bedeutung sind als krautige Pflanzenarten, was nicht mit dem Artenvorkommen im Weidegebiet korreliert. Daraus wurde die Hypothese abgeleitet, dass die Verlässlichkeit der Futterressourcen in Zeiten von knappen Naturressourcen einen essentiellen Faktor in der lokalen Umweltwahrnehmung darstellt. Es hat sich gezeigt, dass das Integrieren von LÖK in die ökologische Forschung dazu beiträgt, Kriterien und Indikatoren für lokale Managemententscheidungen zu identifizieren.

Bei dem Umgang mit knappen und variablen natürlichen Ressourcen ist es unabdingbar, ein funktionelles Verständnis der Interaktionen zwischen Managementstrategie und den Mechanismen, welche die Variabilität von Niederschlägen abpuffern, zu entwickeln. Deshalb wurde in einem zweiten Schritt das lokale Wissen den lokalen Managemententscheidungen gegenüber gestellt und diese miteinander verglichen. Diese Studie unterscheidet zwischen abiotischen Puffern („key resources areas“) und biotischen Puffern („storage tissue“ und „stockpiled forage“). Die zwei biotischen Puffermechanismen können durch die Mobilität der Rinderherden gemanagt werden. Es wurde analysiert, wie die Mobilität mit dem Angebot der „ecosystem goods“ Wasser und Futter verbunden ist, wobei die Mobilitätsentscheidungen der lokalen Hirten mit den biotischen und abiotischen Puffern auf den Weiden verglichen wurden, um herauszufinden, ob die Hirten die Weiden mit den entsprechenden Schlüsselressourcenmerkmalen nutzen. Im Gegensatz zu einem geläufigen Ansatz in der Weidökologie, spiegelt sich nicht die Verfügbarkeit von Futter-Schlüsselressourcen in den Entscheidungen der lokalen Hirten wider, sondern hauptsächlich eine zweite essentielle Ressource für die Tiere, das Trinkwasser.

Zudem verfolgen die Hirten nur kurzfristige Managementziele. Es kann geschlossen werden, dass ihre Handlungsmöglichkeiten von Einflüssen limitiert sind, welche über die ökologischen Bedingungen im Untersuchungsgebiet hinaus gehen, wie beispielsweise Landknappheit, unangepasste institutionelle Rahmenbedingungen und eingeschränkte Möglichkeiten der Einkommenssicherung. Dieser Abschnitt der Arbeit zeigt, dass es wichtig ist, die Ziele der lokalen Ressourcennutzer und Kriterien für ihre Entscheidungen darüber, wann und wo sie mit ihren Tieren hinziehen, zu verstehen. Diese Kenntnisse gewinnen vor allem in Zeiten knapper natürlicher Ressourcen an Bedeutung, um die essentiellen Elemente des lokalen Weidemanagements zu erfassen. „Verlässlichkeit“, was als ein wesentliches lokales Kriterium für die Bedeutung von Futterressourcen wahrgenommen wird, hat erhebliche Gemeinsamkeiten mit dem Konzept der Schlüsselressourcen, welche als diejenigen Futterressourcen definiert sind, die in Zeiten von Ressourcenknappheit zur Verfügung stehen. Viele Autoren beschränken sich auf einen deskriptiven Zugang zur Verlässlichkeit von Schlüsselressourcen. Diese Arbeit dagegen definiert sie funktional als abiotische und biotische Puffer.

Im dritten Schritt evaluiert diese Studie den Einfluss der identifizierten Beweidungsstrategien auf die Vegetation und die Möglichkeiten, wie ökologische

Schwellenwerte (thresholds) identifiziert werden können. Aufgrund multipler Faktoren, die auf verschiedenen räumlichen und zeitlichen Skalen agieren, ist es schwierig, direkte Maße für Systemänderungen zu erarbeiten. Folglich gibt es einen Bedarf für die Entwicklung von Indikatoren, um bestimmen zu können, ob sich ein Ökosystem bereits einem Schwellenwert annähert. Aufgrund der vorliegenden Forschungsergebnisse konnten Frühwarn- und Langzeitindikatoren entwickelt werden, welche zeigen, wann sich ein System einem degradierten Zustand annähert. Diese Indikatoren konnten auf verschiedenen Ebenen identifiziert werden. Der wesentliche Langzeitindikator ist der strukturelle Übergang vom Grasland zu „woodland“, was eine essentielle Veränderung in der Versorgung mit Futterressourcen bedeutet. Diese Veränderung kennzeichnet eine bedeutende Umstrukturierung auf funktionaler Ebene, symptomatisch für Degradationsprozesse (Reynolds and Stafford 2002; Scheffer et al. 2009). Als Frühwarnindikator wurde das Verhältnis von annualen zu perennen Gräsern identifiziert.

Die aktuell praktizierte lokale Strategie der Weidenutzung ist möglicherweise noch nicht als adaptiv einzuordnen, dennoch erscheint sie als die einzige mögliche Adaption der lokalen Nutzer an die derzeitige ökologische und sozioökonomische Situation, denn ihre Handlungsoptionen sind eingeschränkt. Da die Ausgangskriterien für die Analyse eines sozial-ökologischen Systems die nachhaltige Versorgung mit „ecosystem services“ und die Erhaltung der Lebensgrundlagen sind, ist die entwickelte integrierte Analyse notwendig, um die Effekte von Umweltvariabilität, die Versorgung mit „ecosystem services“ und ihre Verbindung zu Managementregeln zu konzeptionalisieren. Ein Fortschritt in der Weideökologie kann erreicht werden, wenn lokales Wissen, lokale Entscheidungsfindungsprozesse im Bezug auf ihre Treiber sowie der Einfluss von Landnutzung und Umweltvariabilität auf die natürlichen Ressourcen in die Forschung integriert werden. Wenn integrierte Analysen den Herausforderungen, vor denen unsere globalen Weidesysteme stehen, gerecht werden sollen, müssen Nutzer und Wissenschaftler sowie Theorie und Anwendung mit einem gemeinsamen Satz von Zielen, bei denen Monitoring ein wichtiges Werkzeug für die Etablierung der nachhaltigen Nutzung von Weidegebieten ist, verbunden werden.

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1 Introduction

Drylands are influenced by specific environmental and anthropogenic conditions, with the climatic features playing a particularly characteristic role. Aridity is high in these areas and annual potential evaporation exceeds the annual precipitation by a factor of at least 1.5. The mean annual precipitation does not exceed 350 mm, while the coefficient of variation can range between 40 and 60 per cent (Middelton and Thomas 1997). This expresses the most defining feature of drylands, namely high variability of environmental conditions in time and space. On a local scale, not only rainfall but soil, geology, topography and herbivory are important drivers of plant productivity and spatial distribution of plant cover (Gillson 2004). A further important determinant is land use. Due to their climatic characteristics drylands are used mostly as rangelands and their users depend mainly on extensive livestock¹ production. Pastoralism is thus one of the main sources of livelihood in these areas (Darkoh 2003; Mendelsohn et al. 2003). Arid and semi-arid rangelands, which cover two-thirds of the African continent, provide a living for more than 370 million people worldwide and about 80 per cent of the nutrition for livestock (Ellis 1994; White et al. 2002).

However, degradation² of natural resources has become a serious challenge in rangelands, bearing negative impacts on the pastoral ecosystems, livestock production and livelihoods (Vetter 2005; Kassahun 2008). Degradation processes in drylands can be caused by maladapted land use. A growing imbalance between the number of livestock, the natural resource supply as well as changing preferences and needs of the human population leads to more intensive land use and changes in traditional rangeland use patterns (FAO 2001). An important factor that has resulted in the transformation of traditional rangeland use patterns is the provision of permanent waterpoints which attract the establishment of permanent settlements. With increased sedentarisation of people and livestock along with improved provision of veterinary and other medical services, migration of entire households has declined considerably,

¹ Livestock: Domesticated animals raised for agricultural production, such as cattle, goats and sheep.

² See Chapter 4 for further reading on the definition of and approach towards the term degradation in this thesis.

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disturbing the balance between the human population, cattle stocks, water and forage³ resources (Niamir-Fuller and Turner 1999; Bollig 2005; Samuels et al. 2007).

This more sedentary form of grazing results in unnaturally constant and heavy grazing pressure on forage resources (Landsberg et al. 2003) as compared to a system where rainwater and high groundwater, accessible via digging, is the only source of water. Under “natural” conditions, livestock movements are driven by the spatio-temporal variability of rainfall typical of semi-arid environments (Tainton 1999; Pickup and Stafford Smith 1993), where grazing is limited by the natural provision of rainfall water, accumulated in clay pans in the rainy season or in the riverbeds in the dry season. Discontinuing such grazing patterns can cause significant changes in vegetation dynamics and declines in soil fertility (Barbier 2008). If land use patterns are not suitably adapted to the spatio-temporal variability of natural resources such as water and forage supply for livestock, these resources may be subject to processes of degradation. Consequences are a shift in vegetation patterns, altered species composition resulting in a loss of species and biomass, and a decrease in pasture productivity (Briske et al. 2006; Kassahun et al. 2008). Estimates show that overgrazing causes 35 per cent of all human-induced soil degradation worldwide and 49 per cent in Africa (Haen 1997; Pinstup-Anderson and Pandya-Lorch 1994). All these forces not only contribute to increased pressure on the environment but also raise the vulnerability of the herders.

The question arises as to which mechanisms determine a sustainable rangeland use in these highly variable and vulnerable ecosystems. There is a huge need to understand the fundamental aspects and elements of sustainable⁴ rangeland use and how assessments can most adequately be made in rangelands.

1.1 Rangelands as social-ecological systems

Since pastoralists’ livelihoods depend directly on the provision of natural resources, and since adaptation is a political process (Eriksen and Lind 2009), we

³ Forage: Plant material eaten by livestock.

⁴ For my evaluation of rangeland use I shall follow Snyman’s (1998) sustainability criteria: “The most important principles for sustainability of the rangeland ecosystem in arid and semi-arid regions are based on: (1) natural resource conservation; (2) decreasing risks; (3) maintaining or enhancing biological productivity; (4) economic viability; and (5) social acceptability.”

assume that herders are compelled by environmental and socio-economic changes to adapt their behaviour and livelihood strategies.

This strong and direct interdependence between people and their actions on the one hand and natural resources and their dynamics on the other is my main motivation for approaching my system of interest as a social-ecological system.⁵ A social-ecological system can be defined as an analytical category enabling formalised analysis of relations between society and ecology (Becker and Jahn 2006). Social-ecological systems are complex in character due to the multiplicity and non-linearity of the mutually dependent factors existing within them (Liehr et al. 2006).

Grazing resources provide pastoral land users with forage for their livestock and this may be seen as an important ecosystem service of rangelands. Ecosystem services can be defined as functions of ecosystems with value for human well-being (Constanza et al. 1997; Tengö and Belfrage 2004). This implies that it is essential to understand the interdependencies between the ecological and socio-economic subsystems. The underlying capacity of the system to generate ecosystem services is herein referred to as resilience (*sensu* Holling 1973). Resilience encompasses the capacity to absorb perturbation⁶ while maintaining function without collapse (Carpenter et al. 2001; Gunderson 2000; Holling et al. 2002). To maintain resilience⁷, it is necessary to understand and manage vital ecosystem functions as well as social mechanisms that can respond and adapt to feedback signals from such systems (Berkes et al. 1998; Kates et al. 2001).

However, the growing acceptance in the scientific community that ecosystems are complex, dynamic, non-linear systems pose new challenges for rangeland research.

Natural science approaches alone offer insufficient understanding of the dynamics underlying social-ecological systems. The interactions between the ecological and socio-economic subsystems are complex and to study these in-

⁵ The underlying theory of Social Ecology is not at all limited to such obvious cases, but is rather a general theory for human-nature interactions (Becker and Jahn 2006).

⁶ Perturbation can be defined as “disturbance or disruption, including management actions, which can force a system into another state” (Suding et al. 2004).

⁷ Ecological resilience can be defined as the “speed at which a system returns to its former state after it has been perturbed and displaced from that state. In the context of restoration, resilience can refer to both a system’s return to a restorative “goal” state following degradative perturbation, and to a system’s return to a degraded state following management perturbation” (Suding et al. 2004).

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teractions in an adequate manner, several aspects belonging to different disciplines have to be integrated. Thus, any analysis of social-ecological systems requires interdisciplinary research approaches (Baumgärtner et al. 2008).

Due to the direct interaction of humans and Nature, disciplines that seem especially suitable for an interdisciplinary analysis of rangeland systems are anthropology (the study of humans and their interactions with each other and the environment [Wolf 1994]), sociology (the study of human societies to develop knowledge on human social activity [Comte 2005]), and geoecology (the study of ecosystems and the human impact on them).

Environment perception, decision-making processes and impact of land use on ecological dynamics are three relevant elements representing the interactions between the ecological and social subsystems which need to be analysed in order to understand resilience mechanisms in a social-ecological system (Schlüter and Pahl-Wostl 2007).

1.2 Perception, decision and impact

Perception indirectly guides human behaviour and helps individuals to make decisions, e.g. on economic matters in various situations of daily life. Perception of their environment is especially important for rangeland users interacting directly with it, e.g. to monitor the status of the resource supply and the condition of their animals (Niamir-Fuller 1998). This direct interaction and monitoring of the environment leads to the build-up of local knowledge under the influence of cultural background, the local environment and personal experiences (see Tönsjost 2007 for review of local knowledge). Knowledge both shapes perception and is shaped by perception.

All these components of environment perception and local knowledge show that there are multiple factors which influence the complex process of *decision-making* (Neubert and Macamo 2004). Local knowledge is embedded in specific conditions and is one framework in the decision-making processes. An important decision in drylands is that of choosing a range management option, e.g. an optimal or sub-optimal grazing area. Natural resources and spatio-temporal vegetation patterns in rangelands are not only influenced and determined by the highly variable environment, but they also mirror land use in dependence of individual or collective decisions by the user groups. Like other grazing systems, pastoral herding and the decisions that affect it occur

across a hierarchy of spatial and temporal scales (see Bailey et al. 1996; Coughenour 1991; and Senft et al. 1987 for general reviews of grazing systems). The spatial aspects of pastoral land use and the importance of mobility and mobility decision processes have long been recognised by anthropologists (reviewed in Dyson-Hudson and Dyson-Hudson 1980). A key element of these debates is the spatial dimension of *impacts* and pastoral herding, e.g. where herding takes place and what factors affect this distribution. Furthermore, the impact of rangeland use on vegetation dynamics is especially interesting from an ecological point of view.

The impact of livestock grazing in drylands is diverse (Westoby et al. 1989; Milton and Hoffman 1994). As southern African ecosystems have experienced a co-evolution with large indigenous herbivores (Owen-Smith and Danckwerts 1997), they should be pre-adaptive and thus relatively more resilient than other ecosystems to the impacts of livestock grazing (Milchunas et al. 1988; Milchunas and Lauenroth 1993). However, even in Africa, with its extended evolutionary history of grazing by large mammals, current grazing regimes frequently differ from historical ones, and there are many examples where livestock grazing has caused major changes in the composition of plant communities (e.g. Illius and O'Connor 1999; Tobler et al. 2003; Landsberg et al. 2003).

Light grazing may result in an increase in plant biodiversity and species richness, most probably as a result of reduced competition (Noy-Meir et al. 1989). Increased grazing intensity can, however, lead to a decrease in species richness (Noy-Meir et al. 1989; Olsvig-Whittaker et al. 1993). A frequently observed change associated with increasing grazing pressure is a shift from perennial to annual vegetation (Milton et al. 1994; Huenneke and Noble 1996; Todd and Hoffmann 1999).

1.3 The tool box of rangeland ecology

In recent years, the debate on essential factors for a sustainable use of natural resources in rangelands has gained new momentum (Walker and Abel 2002; Gillson and Hoffmann 2007; Linstädter 2008). A particularly promising approach for a better understanding seems to be the analysis local of land use strategies (Müller et al. 2007). Only a fundamental understanding of the multiple interactions within the most important set of factors provides the opportunity to develop sustainable land use strategies. To gain an insight into the

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complex dynamics in rangeland ecosystems is a fundamental aim of range ecology in general and of this thesis in particular. The condition of the ecosystem needs to be assessed with appropriate tools, accompanied by an analysis of the consequences of anthropogenic land use.

Besides remote sensing (Matheson and Ringrose 1994), geographic information systems (GIS) (Robbins 2003) and modelling approaches (Jeltsch 1994; Tietjen and Jeltsch 2007), there are three further tools of importance in case study approaches to range ecology, namely are long-term monitoring (Rosenschein et al. 1999), manipulation experiments (O'Reagain and Turner 1992), and the assessment of present range condition (Schulte 2002).

1.3.1 Indicators

For all three approaches, specific ecological indicators are used which can be assigned to different hierarchical levels. Vegetation, as the resource used, lends itself to the provision of specific rangeland indicators, with the different levels in question being community, plant species and individual plants. However, with vegetation as the indicator it is difficult to disentangle the influence of environmental determinants and land use (Dougill et al. 1999). Thus it is still difficult to identify significant indicators for range assessment. This is because they should indicate the current condition of the natural resources along with possible changes from neutral towards directed processes at an early stage, as well as being locally accepted and applicable. In long-term monitoring and manipulation experiments, the specific aim is to assess the precipitation-driven parts of vegetation dynamics on the one hand, and on the other to assess the condition and recovery potential of the pastures. Here the focus lies on the level of plant populations and their dynamics. However, population dynamics as parameters are not easily captured for the assessment of pasture quality and recovery potential (see Baumann 2009 for a review of indicators).

In order to differentiate between the influence of climate and land use, indicators and the results of monitoring need to be compared with those of a benchmark, which, ideally, should show the best rangeland conditions possible in the current macroclimate (de Klerk 2004). In range ecology, a benchmark can be defined as the optimal vegetation of a grazing area with regard to desired land use/management objective (Aucamp et al. 1992). In most studies, two approaches are applied to identify a benchmark: the potential natural

vegetation (Dyksterhuis 1949) or the status under “best practice land use” (Hawley 1944). However, long-standing reserves might be needed for the potential natural vegetation, since “natural” vegetation might not be found in rangelands, and what is considered best practice will differ among land users, depending upon their management objectives (Schulte 2002). The most useful type of benchmark for general rangeland monitoring will be one that exhibits the healthiest ecological processes and is best protected from erosion under the prevailing macroclimatic conditions (de Klerk 2004). Since this benchmark is the expression of a desired condition of the ecosystem, it is important for social values to be integrated (Smyth et al. 2007), to which end local land users can be interviewed.

1.3.2 Grazing gradients

Grazing gradients represent an opportunity for differentiating the long-term effects of livestock activity from other environmental patterns. Gradients of animal impact known as piospheres tend to develop around artificial water-points, particularly in arid ecosystems. Previous studies used the distance from a source of water (e.g. Fernandez-Gimenez and Allen-Diaz 1999; Todd 2006) or from a livestock camp (e.g. Riginos and Hoffman 2003) to assess the impact of grazing on rangeland structure and function, while controlling the background environmental factors (Hoshino et al. 2009). The ultimate cause of this pattern is the fact that, in arid ecosystems, livestock are limited in how far they can move away from water sources, because they have to drink regularly. Furthermore, the grazing area available increases with distance from the waterpoint, resulting in a reduction in the relative grazing intensity with distance (James et al. 1999). Grazing gradients provide a way to observe vegetation changes under different grazing intensities in areas where it is not possible to manipulate livestock densities of free-ranging grazers (Fernandez-Gimenez and Allen-Diaz 2001). The aim is to assess the impact of certain management strategies on the ecosystem.

1.3.3 Local ecological knowledge and interdisciplinarity

Local knowledge⁸ is seen as a key to understanding pastoralist strategies of sustainable resource use (Kaschula et al. 2005). It is particularly interesting to investigate how local knowledge is produced and used in a highly unpredictable system characterised by non-linear dynamics. With the integration of local knowledge into scientific assessments, evaluations and the implementation of management strategies, it is possible to ensure the local applicability and the objectivity of the assessment. The challenge for research lies in the fact that cognitive processes are unobservable and unconscious. To overcome this dilemma, “scientific” ecological data can be matched with local ecological knowledge to achieve a synthesis by comparing the emic and the etic perception⁹, of certain ecological processes, services and items. The aim is to gather new insights into local decision-making processes regarding natural resource management.

One key element in local environmental monitoring in rangelands is the perception and rating of local key forage species (Niamir-Fuller and Turner 1999). However, people in any given community do not use and value all plant species equally, and consequently some researchers have argued that identifying the more relevant groups of plant species for local people may help in defining and implementing priorities for conservation and sustainable management strategies (Camou-Guerrero et al. 2008).

An interdisciplinary team-based research approach combined with participatory methods can integrate local knowledge into scientific knowledge and involve local land users as experts in the research process to enable diverse inputs (Huntington 2000).

⁸ I decided on the term “local knowledge” instead of “traditional” or “indigenous knowledge” because it underlines the spatial and theoretical aspects of such knowledge. Here, in the spatial category, the local knowledge of OvaHerero pastoralists in the local context of the Kunene region in Northwestern Namibia is emphasised.

⁹ The insider’s and the outsider’s view.

1.4 Research objectives and questions

The objectives of this thesis are twofold:

- I. It aims to make a contribution to range ecology by discussing emerging issues of sustainable rangeland use and providing a methodological framework for the integrated analysis of a social-ecological system with special emphasis on environment perception, decision-making processes and impact of range land use on vegetation dynamics.
- II. The application of this approach is presented in a case study.

The study aims to answer the following questions:

- i. Perception: which ecological elements of the ecosystem are perceived as important by the local land users in regard to land use decisions?
- ii. Decision: what influences decision-making processes of local land users with regard to mobility patterns?
- iii. Impact: what is the impact of livestock grazing and land use strategy on vegetation dynamics and – structure?
- iv. What are suitable and adequate indicators for range assessment?

The dry season was chosen for most of the investigations because it is the time when limitation of resources in an already resource-scarce area is most profoundly felt. Most of the studies so far were conducted during the rainy season (Becker and Jürgens 2000). Only few also take into account the periods which are more unfavourable for species (Leggett et al. 2003). This sequence of the social-ecological dynamics needs further investigation to detect the possible effects of human or climatic impact on rangeland ecosystems.

1.5 Study area

The study area is located in the Kunene Region in Northwestern Namibia. This administrative region covers about 144000 km² (Mendelsohn et al. 2003). It is framed by the Kunene river, the border river with Angola, in the north, the Etosha National Park (NP) to the south-east, the Atlantic ocean in the west and the Hoanib river in the south. The Kunene Region borders on five Namibian Regions, such as Omusati Region in the north-east and Erongo in the south. The area was delimited in 1998, replacing the boundaries of former Kaokoland and parts of Damaraland, which were homeland areas before Namibia's independence in 1990 (Mendelsohn et al. 2003). The region's topography is charac-

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terised by the transition from the Central Namibian highlands to the low-lying coastal regions through an escarpment, which is mountainous and highly structured by valleys.

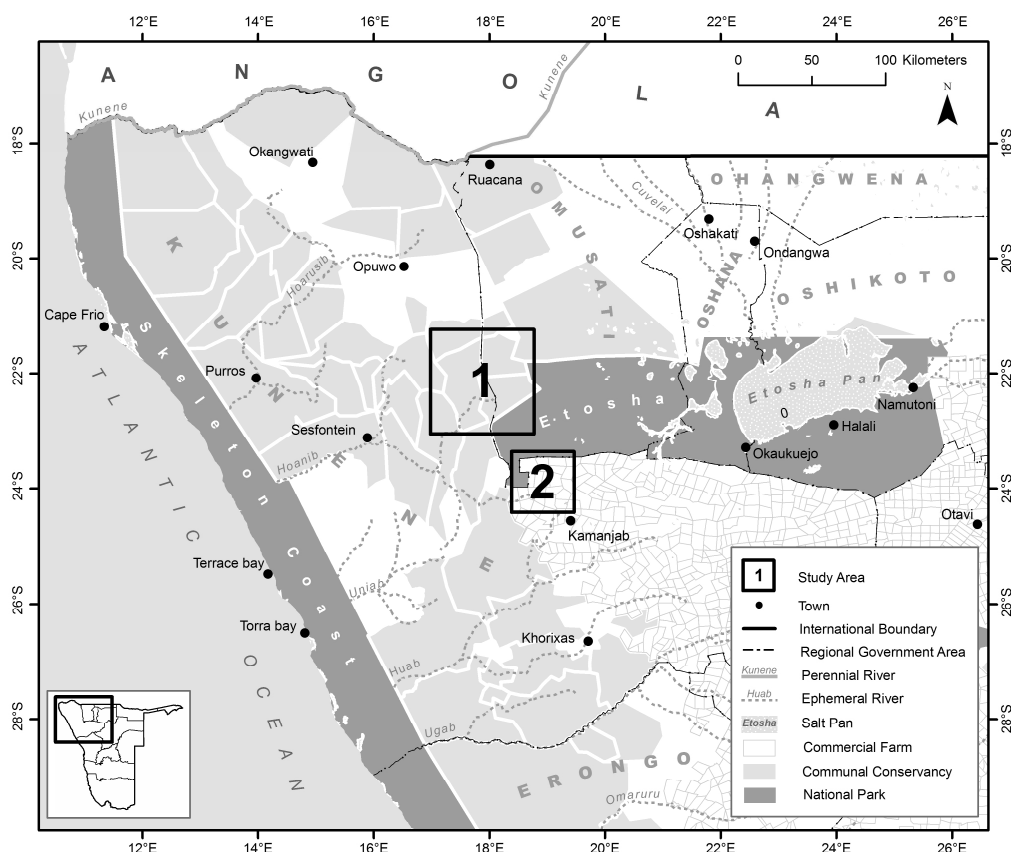


Fig. 1.1: Location of the study areas in Northwestern Namibia, west of the Etosha National Park. Source: ConInfo 2009; Cartography: Holger Vollbrecht.

1.5.1 Land tenure

Land ownership in Namibia is divided into central government (55.8%), private (43.3%) and local authorities (0.9%). Private (or freehold) land may be bought and sold, the owners holding full title to their land. A great part of the government-owned area is allocated communally (Mendelsohn et al. 2003). On private land grazing is controlled entirely by the landowner, comprising only one manager; in communal areas it is controlled on a community basis, comprising multiple managers (Smet and Ward 2005).

Private commercial livestock farming is dominated by cattle, but game farms also occur frequently. On game farms, wild animals, mostly large browsers and

grazers, are kept for wildlife viewing, hunting or for sale (Mendelsohn et al. 2003).

In case of private livestock farming, stocking rates are lower overall than in communal areas, and the animals are kept grazing extensively and rotationally in fenced camps (Mendelsohn et al. 2003). Communal farming is characterised by livestock farming, especially cattle, goats and sheep. Livestock can roam freely because fencing is officially prohibited (Leggett et al. 2003). The grazing regime on the communal rangeland is characterised as being heavily and continuously grazed, while the neighbouring commercial rangeland is moderately and rotationally grazed (Todd and Hoffmann 2000).

In the communal areas, land use strategies have evolved from traditional pastoral nomadism and have been influenced by the restrictive politics of the last century which forbade any commercial trading of livestock until the year 1990 due to quarantine concerns. Since then, stocking rates have nearly doubled (1988–2000) (Bollig 2005). Communal livestock owners in this area do not usually intend to sell their animals at markets. Most products are for their own subsistence. Most herders only have a low or non-existent income (Tönsjost 2007; Faschina 2009).

The communal areas are situated in the emerging Orupupa Conservancy, in Namibia's Kunene region. The Conservancy is an institution of the 'Community-based Natural Resource Management' programme in Namibia. Conservancies take responsibility for the natural resources, mainly wildlife, within their boundaries by monitoring, managing and conserving them. The conservancy in question covers 1650 km² (NACSO 2007).

1.5.2 Climate

Namibia is characterised by frequent droughts (de Klerk 2004) and the absence of permanent rivers. The country is situated at the interface between two climatic systems: the Inter-Tropical Convergence Zone (ITCZ) and the Mid-Latitude High Pressure Zone. Those zones move northward and southward in response to the apparent movement of the sun. The ITCZ is an area of rainfall, whereas the high pressure belt between them is generally an area of very little rainfall. Thus slight inconsistencies in the extent or timing of the movement of these zones will cause considerable difference to the weather from one year to the next (Sander and Becker 2002). That is the cause of the high climatic vari-

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ability in Namibia (Hutchinson 1995; Huenneke and Noble 1996). Rainfall is patchy and highly variable in time and space (Mendelsohn et al. 2002). The study area receives an average annual precipitation of 200 to 300 millimetres, while the coefficient of variation ranges between 40 and 60 per cent (Mendelsohn et al. 2003). Due to the inter- and intra-annual variations of rainfall, this water source is very unpredictable. Annual potential evaporation is estimated to exceed the annual precipitation by a factor of about five (Mendelsohn et al. 2000; Klintonberg and Verlinden 2007). Northern Namibia is characterised by two seasons, the dry season that generally lasts from May to October and the wet season, characterised by occasional rainfall, from November to April. On average, most of the rain falls between January and April.

1.5.3 Geology and soils

Namibia comprises two large-scale geological areas. In the eastern part of the country, most rocks are covered by sands and sediments, whereas the west comprises a great variety of rock formations. Most parts of the study area are situated in the western part; only the most north-easterly communal study site forms part of the Kalahari and Namib sands group. The sites within the emerging communal Orupupa conservancy, east of Etosha NP, are part of the Damara Supergroup and Gariep Complex (850–600 million years), the Otavi group, and consist mainly of limestone and dolomite (Klimm et al. 1994; Grünert 1999).

The commercial farm sites, south-west of Etosha NP are situated on the oldest Rocks group (2600–1650 million years), and also part of the Khoabendus group and Oapuka formation. They consist mainly of rhyolites and sandstones. However, personal observation indicates that to the west of Etosha NP a granite environment can also be found, with corestones, the result of desquamation, occurring as a major feature in the area. Only a few kilometres further north, limestone and dolomite can be found (Mendelsohn et al. 2003). This illustrates the highly variable and fragmented character of the study area's geology.

In general, the study area features low developed soils (van der Merwe 1983). The communal areas are characterised by lithic Leptosols, which are very thin and shallow soils, typically formed in actively eroding landscapes. Leptosols are coarse-textured and have a limited depth caused by the presence of continuous hard rock within 30 cm of the surface (Mendelsohn et al. 2003). They

are azonal soils and common in mountainous regions, where they are found on slopes with continuous erosion, appearing together with Regosols (Bubenzer and Bolten 2007). Their water-holding capacity is low and rates of water run-off and erosion can be high when heavy rains fall. In some plains, eutric Regosols can be found on communal ground. They are rather fertile soils with a high base saturation. Regosols are weakly developed, fine to medium-textured soils of actively eroding landscapes, the thin layers lying directly above the rock surface. They do not usually reach depths of more than 50 cm (Mendelsohn et al. 2003; Bubenzer and Bolten 2007).

The north-eastern part of the communal Conservancy is characterised by ferralic Arenosols which have high contents of combined oxides from iron and aluminium (van der Merwe 1983). They are generally formed from wind-blown sand usually extending to a depth of at least one meter. Sand makes up more than 70 per cent of this soil, while the rest usually consists of particles of clay and silt. This soil is relatively favourable for pastures and other agricultural activities such as gardens, since capillary rise of water and subsequent salinisation of the soil does not occur (English et al. 2005).

The farm areas south-west of Etosha NP are characterised by dystic Leptosols and Cambisols, which are infertile soils with low base saturation. Cambisols were formed recently on the geological time scale, mainly from medium and fine-textured material deposited during sporadic flooding. As the parent material is only slightly weathered, they are characterised by the absence of accumulated clay, organic material, aluminium and iron. They have a relatively high water-holding capacity (Mendelsohn et al. 2003). This again shows the highly variable environmental characteristics in the study area.

1.5.4 Vegetation

On the level of plant biomes, the study area is situated within the tree and shrub savanna (Sarmiento 1984; Mendelsohn et al. 2003). Savannas are defined as tropical communities with a relatively continuous herbaceous layer, usually dominated by C4 grasses, and a discontinuous woody layer of shrubs or trees (Belsky 1995; Solbrig et al. 1996). The impact of Namibia's dry conditions is indirectly expressed in the character of the vegetation, which is thinly distributed over much of the country and adapted to survival under the prevailing climatic conditions (Moorsom 1995).

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The vegetation type is western Kalahari for the eastern part of the study area and western highlands for the rest of the study area (Mendelsohn et al. 2003). Dominant species in the western Kalahari are *Commiphora* species, *Combretum apiculatum*, *Pterocarpus rotundifolius*, *Acacia erioloba* and *Acacia fleckii*. The vegetation in the Kunene region was described in detail by Viljoen (1980) and summarised by Becker (2000), mainly following climatic gradients. The eastern part of the study area is characterised by the *Terminalia sericea-Lonchocarpus nelsii-Sesamothamnus guerichii*-unit of the eastern sand field. The other parts of the area are mainly dominated by *Colophospermum mopane* and *Terminalia prunioides*, typical of the central drainage area. Recordings of this study showed that the most abundant grass species were *Aristida adscensionis*, *Stipagrostis uniplumis var. uniplumis* and *Schmidtia kalahariensis*.

The study is structured as follows: the questions and objectives stated in Chapter 1.4 are examined with data from a communal conservancy and put in the context of a commercially managed system, as a benchmark. In Section Two, important elements of environment perception are analysed within an especially developed methodological framework. Buffer mechanisms are analysed and decision-making processes of the local land users are linked to these dynamics. In Section Three, a monitoring scheme for rangelands with different indicators is developed, especially in the context of ecological thresholds.

The last chapter provides a synthesis and integrates all aspects in the context of socio-ecological conditions, furthermore an outlook for future studies is given.

2 Perception – Towards a synthesis of local and scientific knowledge on ecological items: An interdisciplinary framework

2.1 Introduction

During the past years, there has been a growing recognition that the integration of local ecological knowledge (LEK) into ecological research is important particularly in the fields of biodiversity conservation and sustainable natural resource use (Berkes et al. 2000; Pierotti and Wildcat 2002; Gadgil et al. 1993). Local people can offer alternative knowledge, values and insights on ecological processes and items which go beyond that of researchers or policy makers. Local land users often have an implicit knowledge of interrelated stochastic and deterministic processes. Thus, most progress towards an understanding of complex social-ecological systems can be achieved by a synergy of local and scientific knowledge (Harrison and Burgess 2000; Payton et al. 2003; Kaschula et al. 2005).

However, there is still a lack of formalised, rigorous scientific methods and concepts (Da Cunha and De Albuquerque 2006). Current work with LEK mainly means collecting and documenting it in the form of databases or registries, and a synthesis as the outcome of an integration of LEK into scientific research is hardly ever reached (Twarog and Kapoor 2004). Only few researchers have analysed the practices and strategies involved in integrating the two kinds of knowledge, or have made recommendations for future projects (Calheiros et al. 2000; Klooster 2002; Schultz et al. 2007). This may be due to various conceptual and practical obstacles (Huntington 2000; Davis and Wagner 2003) and to a certain scepticism against LEK displayed by some scientists (Bojorquez-Tapia et al. 2003). We present a scientifically rigorous framework that overcomes these problems with an interdisciplinary methodology which builds on participatory methods to integrate LEK into ecological research.

LEK encompasses many aspects, including local perceptions and explanations of processes, item properties, item classifications and natural resource management. We concentrate on the local perception of ecological items, which we define here as biotic or abiotic entities of a certain ecological category, such as plants, mammals or soil types having a specific economic, cultural, or indicative value (salience) in local perception.

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LEK studies have mostly focused on ecological items such as plants (26%), mammals (17%), soils (17%) and others (Brook and McLachlan 2008). Only a small proportion of LEK studies deal with aspects other than ecological items, such as water quality or the observation of biological phenomena (Fazey et al. 2006). The reasons why ecological items are so highly represented in LEK studies are both practical and conceptual. For instance, items are easier to assess than LEK on interactive processes, especially because of the implicit character of LEK.

When reviewing LEK studies, we have found two different approaches to compare the local value of ecological items (i.e. their salience) to a scientific value or assessment: Firstly, a comparison of two independent rating systems, such as local and scientific grazing values of forage plants (Bollig and Schulte 1999), or the severity of soil erosion (Vigiak et al. 2005), and secondly a comparison of local rating to ecological performance of items (e.g., Castaneda and Stepp 2007; Camou-Guerrero et al. 2008). The studies show that the local value of an ecological item may differ considerably from the scientific assessment, functional explanations for such differences have been proposed (Shipley et al. 2006). By analysing the local value system itself these studies highlight the local criteria for valuation, which play a role in local decision-making processes.

Due to the relevance of provisioning ecosystem services for local livelihoods, societies assess the supply of these ecosystem services, such as local indicators for rangeland condition (Klintenberg et al. 2007). Since LEK is, amongst others, based on the perception of resources such as plant species and their spatio-temporal patterns of production, this approach can also improve our knowledge on criteria and indicators guiding land users in their decisions with respect to spatial and temporal aspects of land management.

The integration of local and scientific knowledge is particularly challenging because of the different characteristics of the two knowledge systems. Scientists are used to the explicit and specific character of scientific knowledge, which is based on effective methods, and is transferable across different spatial and temporal scales (Dewalt 1994). Local knowledge is mostly implicit and indirect. It is the sum of experiences, abilities and world views, of people and institutions, which have emerged in a specific cultural background and environment, on which its meaning depends (Harrison et al. 1998), and which is constantly changing (Schareika 2004). It is the consequence of practical en-

gement, reinforced by experience in the environment and trial-and-error tests. Pastoralists, for example, have to cope with a high variability of natural resources in space and time. Thus they have developed local strategies of risk minimisation (Bollig 2005), which enable them to deal with lack of rain and variable amounts of palatable biomass (Little 2003). On the other hand local perception may also be flawed if causal links between process and effect are not perceived as in the case of degradation in arid rangelands (Bollig and Schulte 1999).

We analyse congruencies and differences between local salience and scientific assessment to get valuable insights on functional aspects of local land management which are not fully understood by scientific approaches alone. Our approach can address the following questions: (i) are there differences in local and scientific perception and valuation of ecological items? (ii) which items are important and what attributes make them important in local perception? (iii) what are local criteria in decision-making processes and can indicators be identified?

We will first present the methodological framework and the methods it is based on, subsequently the case study will be shown where the method was applied. Concluding further fields of application will be discussed.

2.2 Methodology

Our methodological framework parameterises local knowledge and scientific data in such a way as to allow a direct comparison, identifies congruencies and differences between local perception and scientific assessment of ecological items, to then analyse these differences to derive a synthesis between local perception and ecological assessment.

Ecological items which can be used in our methodological framework are a) distinct and roughly equivalent units from a local and a scientific perspective, b) occur in the environment with certain, quantifiable abundances and frequencies, in space or time, and c) can be assigned to functional categories.

Our null hypothesis is that local valuation correlates with ecological importance of ecological items. The expectation of a correlation is based on the assumption that LEK reflects observable ecological attributes, such as species abundance, that also form the basis for scientific assessment (Mackinson

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2001). Observations may be linked to land use activities such as harvesting fruit or fuel wood.

It is important to keep in mind that there are different phases within an interdisciplinary research process. We recommend that the research questions and main hypotheses are formulated together. A side by side phase of disciplines follows, to gather objective data for each discipline. A fully integrated cooperation of disciplines follows (Baumgärtner et al. 2008), to assure a synthesis of the different disciplines.

A summary of our framework is presented in Fig. 2.1.

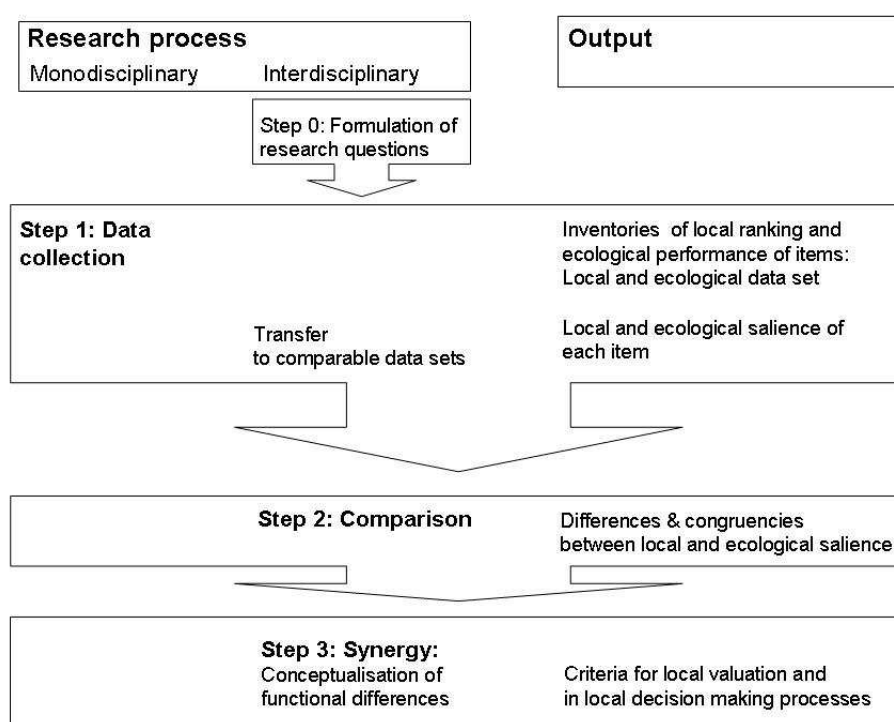


Fig. 2.1: Framework for the integration of LEK into ecological research on ecological items.

2.2.1 Step 1: Collection of ecological knowledge

The first step is the collection of data on the valuation of items in LEK and on ecological performance of the items. When collecting and analysing LEK it is a challenge to account for scientific rigour, because LEK might not lend itself well to mathematical representation (Mackinson 2001). Thus quantitative methods are useful tools. Furthermore LEK should be collected with participatory methods to actively include local people in the research process.

Ranking and scoring methods are good tools to parameterise LEK because, rather than answering questions which might be directed by the values of the researcher, local people are encouraged to explore their own versions of their worlds (Pretty 1995). Methods such as free-listing and pile sorting (Borgatti 1994) from Cultural Domain Analysis (CDA) investigate how items are related to each other in people's minds. The informant is asked to name all the important items of interest that come to mind. A necessary assumption of this method is that the items mentioned first and most frequently by informants tend to be more salient in their domain of knowledge (Bernard 1994). As a result of the free-listing the items are ranked by local informants. Why a high or a low rank is given can yet not be answered, because the parameter ranking the items is unknown. Thus, LEK is still implicit.

Ecological performance of items can be collected via an ecological assessment, quantified by parameters such as frequency or ground cover. Ecological performance can be ranked explicitly by these parameters, e.g. from highest to lowest frequency. Thus two data sets are collected and ranked, the local one where the parameter ranking the items is not known and the ecological one, where the parameter responsible for ranking the items is known. This is important for the next step, the comparison, because the parameters by which ecological performance is ranked must be known to identify the parameter ranking the local data set. In addition, interviews and participatory observation concerning the topic of interest are necessary to gain insights into the application of knowledge, and to interpret the information provided by the local informants (Davis and Wagner 2003; Jones et al. 2008).

2.2.2 Step 2: Comparison of local and ecological salience

a) Salience scores are calculated for the ecological items via salience indices. This results in two data sets, the local and the ecological salience scores. The salience indices include more information than the ranking, and they transfer the "ecological" and "local" information into salience scores with the same dimension. Salience scores can be plotted as x and y coordinates in a scatter diagram, which can later be used to identify differences and congruencies between the data sets. Keep in mind that the parameter ranking ecological performance is still known after the transfer from ranks to salience scores.

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We propose three different salience indices: Cognitive Salience Index (CSI) (Sutrop 2001), Smith's Salience Index (SI) (Borgatti 1999) and the Frequency Index (FI) (Borgatti 1994). The indices differ with respect to the principle parameters they include and in how sensitive they react towards these parameters (Schneegg and Lang 2008) (Table 2.1). CSI and SI originate from anthropological research. They determine the salience of an item considering the rank of the item in the free-list of each informant, its frequency in all collected free-lists, and the individual length of the informant's free-list. This approach can directly be transferred to an ecological data set, if the free-list and the informant have equivalents in the ecological research design. If, for example, the analysed items are plant species, the free-list could be equivalent to a species list and the informant could be equivalent to a sample plot. Note that this has to be considered explicitly in the research design, because only if each index parameter has an equivalent in the ecological data set, can a direct comparison be achieved (see case study step 2). FI originates from ecological research, it is based on item frequency only (Table 2.1).

Table 2.1: Salience indices CSI, SI and FI and their sensitivity to input parameters (-- not considered; - not sensitive; + sensitive; ++ very sensitive).

Salience index	Source	Input parameter		
		item position in list	item frequency in all lists	list length
CSI	(Sutrop 2001)	+	-	+
SI	(Borgatti 1999)	+	-	++
FI	(Borgatti 1994)	--	+	--

b) The local and the ecological salience scores of the items are compared to identify differences and congruencies between local salience and ecological performance to accept or reject the null hypothesis. The salience scores can be compared visually by placing them in a two-dimensional plot, an importance-performance grid, originally developed for marketing purposes (Martilla and James 1977), or by means of correlation methods. The resulting degree of overlap or correlation is an indication of the relationship between the data sets and thus of how important ecological performance is for the salience of ecological items by local people.

The null hypothesis is that a correlation exists. Only if this is rejected the next steps can be applied.

2.2.3 Step 3: Synergy – conceptualisation of functional aspects

To answer the question of why certain items are important in local perception, the data sets must be analysed in further detail. The items need to be characterised by adding information of certain functional attributes or features. If the null hypothesis, i.e. that local salience correlates with ecological performance, is rejected, we can assume that the difference between local salience and ecological performance of ecological items can be explained by other observable (Mackinson 2001) or functional ecological attributes (Walker et al. 2006). If, for example, plant species are these units, they might be classified into plant functional types, according to their habitat preferences, or life forms. For local classification approaches and the assignment of ecological units to scientific classes, further methods from Cultural Domain Analysis, such as pile sorting (Borgatti 1994) can be applied.

2.3 Case study

We apply the methodological framework in a semi-arid rangeland case study, focusing on forage species as ecological items. Since pastoralism is the dominant form of land use in arid and semi-arid rangelands, it is particularly important to investigate how local knowledge is produced and used in a highly unpredictable system, characterised by non-linear dynamics. With this study, we aim to conceptualise the role of key forage species in local environment perception, especially as essential elements for decision-making processes in local range management. We compare the local salience of forage species with their ecological performance, given by the ground cover and frequency of local plant species. With rejection of the null hypothesis we further analyse the functional aspects of the differences.

The case study is situated in Northwestern Namibia (Fig. 2.2), which is characterised by a high variability of natural resources. Annual precipitation averages around 300 mm, its variability exceeds 30% and droughts occur frequently (Mendelsohn et al. 2003). Vegetation is a secondary Mopane savanna.

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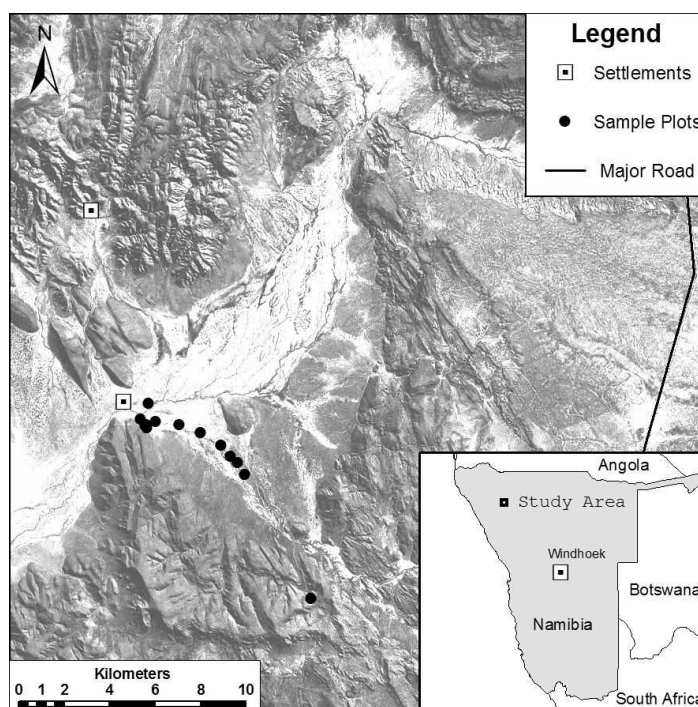


Fig. 2.2: Location of the study area and sample plots, situated in Northwestern Namibia.

Data was collected in the settlement of Okazorongua and its surroundings which have been permanently inhabited by OvaHerero pastoralists since the 1960s. Livestock herds mainly consist of cattle, goats and sheep, additionally donkeys are kept as domestic animals. The study area is under communal land tenure. The settlement was inhabited by a total of 186 people when census data was collected in June 2005. Further data was collected by an ecologist and an anthropologist respectively between April 2006 and January 2007.

2.3.1 Step 1: Collection of ecological knowledge

LEK was collected by free-listing with 19 interview partners, nine women and 10 men from different age groups. Informants were asked to name all the important forage species for local domestic livestock in this area. Ecological performance of plant species was assessed on nine 1000 m² plots within the local grazing area. Species ground cover was estimated visually on each plot, and overall frequency of the plant species was calculated.

Local informants named a total of 211 important forage species. The ecological data set comprises 123 plant species occurring in the local grazing area. We included those species in further analyses, for which both the scientific and local name were identified and a frequency higher than three species was observed in at least one of the data sets (Sutrop 2001).

Semi-structured interviews were conducted to collect information on range management decisions and mobility patterns (Tönsjost 2007).

2.3.2 Step 2: Comparison of local ranking and ecological performance

Salience scores for the plant species were calculated by CSI, SI and FI. For the calculation of ecological salience scores the following input parameters were considered: the vegetation plot was taken as equivalent to the informant, and the species list was taken as equivalent to the free-list. For the application of CSI and SI, the plant species of the ecological data set were ranked from highest to lowest ground cover, for calculating FI they were ranked from highest to lowest frequency. This rank was taken as analogous to a species rank in the free-list.

FI is considered for two reasons, first to test the influence of the highly variable ecological parameter ground cover and because in ecology, frequency is a more readily established quantitative measure than the measurement of ground cover. Salience scores were calculated with ANTHROPAC 4.0. (Borgatti 1996). Results were cross-checked with inhabitants of the local community and cattle farmers from an adjacent commercial farm area by expert interviews and further rankings. For SI and CSI we found no correlation between the local and the ecological salience scores with Spearman's rank correlation ($p < 0.05$). For FI a significant correlation was found (Table 2.2).

Table 2.2: Relationship between the local and ecological salience scores for the three salience indices. Correlation coefficients R and significance levels are given (NS = not significant).

	R ($n = 67$)	$p < 0.05$
SI	-0.2	NS
CSI	-0.19	NS
FI	-0.4	

Hence our null hypothesis that ecological performance gives salience in local perception was rejected. Now, by the process of elimination, we conclude that it is not the parameter ground cover which primarily gives salience to local perception, and on the basis of this finding a further analyses of the data can follow.

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We displayed the differences between local and ecological salience scores by placing them as x and y coordinates in an importance-performance plot (Fig. 2.3). The salience scores for each item, derived by applying each of the three salience indices, are plotted in Fig. 2.3.

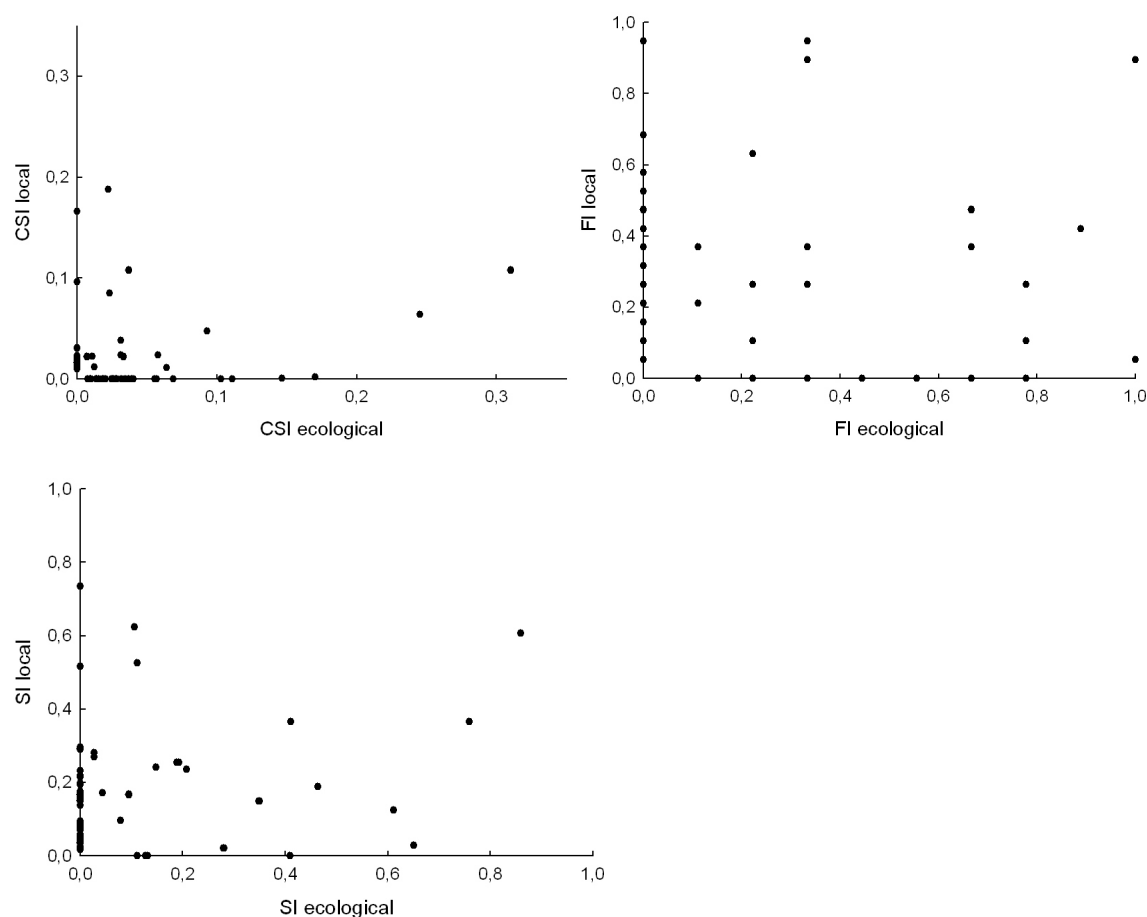


Fig. 2.3: Importance-performance plot. Comparison of local and ecological salience scores (CSI, SI and FI) with Spearman's rank correlation.

It is interesting to note that in the ecological data set we found 16 species (13%) with a zero score, thus not occurring in the local data set. These species grow on local pastures but are not named as important forage species by local informants, hence they are not perceived as important forage species. In contrast, 13 species in the local data set (6%) are zero scores in the ecological data set. These species are named as important forage species but do not occur in the local grazing area. We conclude that the local perception of important forage species is not restricted to the local grazing area. The consecutive question is what makes the items important in local perception.

2.3.3 Step 3: Synergy – conceptualisation of functional aspects

Further information on the plant species was used to come to a functional explanation. Species were assigned to life forms (woody species, perennial and annual grasses and herbs) and abiotic habitats (sandy plains, loamy plains, mountains and riverbeds). The comparison of the different categories of life forms yielded significant results. Combining the visual analysis of the importance-performance plot and the functional classification, here the life forms, reveals that most zero scores in the ecological data set are woody species, while most zero scores in the local data set are herbaceous species. A Wilcoxon signed-rank test found significant differences in the proportion of life forms between the local and the ecological data set for annual grasses and herbs and woody shrubs and trees (Table 2.3).

Table 2.3: Mean (\pm SE) CSI for the local and the ecological data set divided into three life form categories. Results of the paired Wilcoxon signed-rank test are also given (NS = not significant).

Life form	Local	Ecological	Z	P	<i>n</i>
Annual grass and herbs	0.01 \pm 0.02	0.06 \pm 0.06	3.98	0.000069	21
Perennial grass	0.05 \pm 0.07	0.01 \pm 0.01	1.52	NS	7
Woody species	0.03 \pm 0.03	0.01 \pm 0.05	4.34	0.000014	39

The proportion of woody species is significantly ($p < 0,05$) higher in the local than in the ecological data set (Fig. 2.4). Thus we conclude that woody species are more salient in local perception than herbaceous species.

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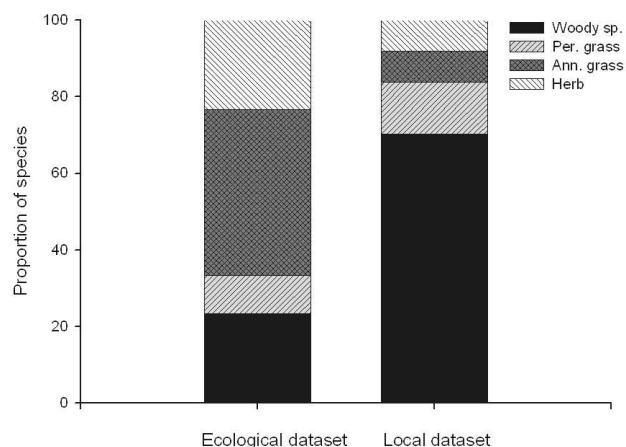


Fig. 2.4: Proportion of the life forms trees and woody shrubs (Woody sp.), perennial grasses (Per. grass.), annual grasses (Ann. grass.) and herbs in the local and the ecological data set, exemplary for the CSI data sets.

A summary of the methods applied in the case study is given in Fig. 2.5.

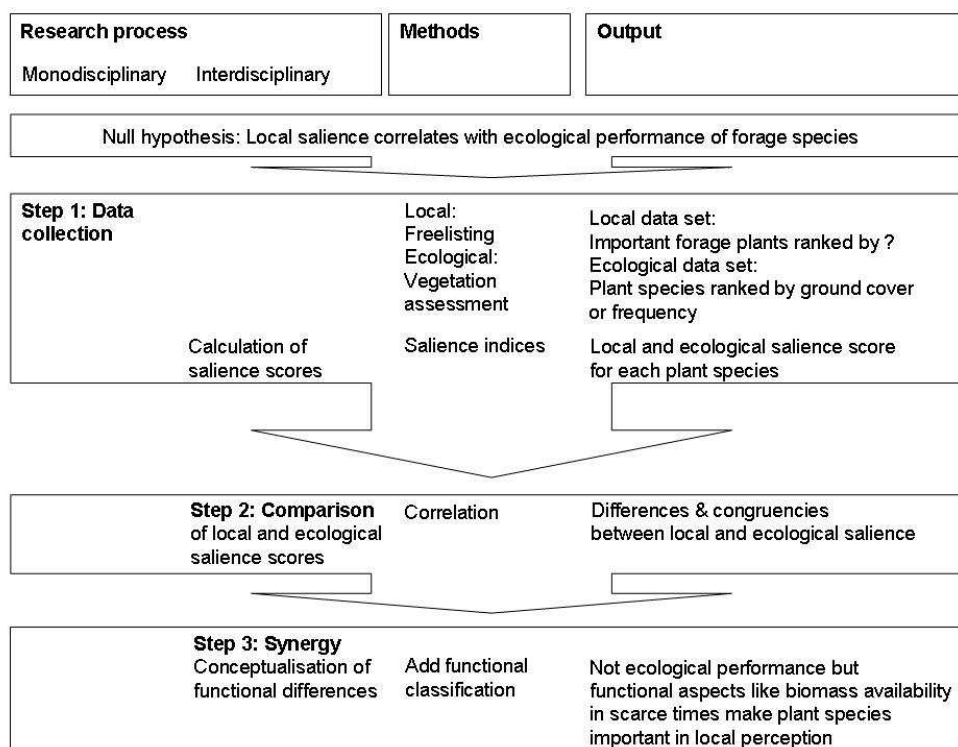


Fig. 2.5: Framework for the comparison of local perception and ecological performance of forage species, as applied in the case study.

Conceptualisation of the functional differences

We analysed the data sets in further functional detail and found that most woody and herbaceous species differ in their annual cycle. The palatable biomass of most woody species in the study area is available within the local dry season (Table 2.4), the time of low resource availability when most of the herbaceous, especially the annual grass cover is already consumed by the livestock (Tönsjost 2007).

Table 2.4: Percentage of species with palatable biomass in rainy and/or dry season occurring in the local and ecological data set

Time of biomass availability [%] (<i>n</i> = 25)	Local data set	Ecological data set
Rainy season	100	100
Dry season	72	32

The results were confirmed by a cross-check with local informants. A ranking revealed that woody species are valued as the main source of palatable biomass in scarce times, such as the dry season, for all kinds of livestock. At least 72% of the palatable biomass during this time is obtained from woody species (Table 2.5) (Tönsjost 2007).

Table 2.5: Local importance of forage plants for different kinds of livestock in the scarce times of the year (late dry season).

[%]	Cattle	Goats	Donkeys
Woody species	72	88	72
Perennial grasses	14	6	14
Annual grasses	14	0	7
Herbs	0	6	7

In interviews with local commercial cattle farmers it was confirmed that leaves and flowers of woody species can make up about 20% of the annual forage. This is obvious for goats and donkeys because they are browsers or have a mixed forage behaviour. But even for cattle, mainly classified as grazers, woody species are named as the most important forage species for the late dry season.

2.4 Discussion

Local land users in our case study perceive woody species to be more important than herbaceous species, because they provide palatable biomass in the dry season. These findings correlate with studies published in 1990 (McKell), where underestimated potentials of woody species, mainly shrubs are discussed. These kinds of findings are still not well integrated into ecological research about semi-arid rangelands. In range ecology herbaceous species' abundance is regarded as a key indicator for rangeland condition influencing management decisions. In particular, the abundance of perennial grasses is named as indicative for rangeland condition. If local perception and salience of forage species is not considered, discrepancies to scientific knowledge, and their functional explanation might be missed. Woody species might be used as local indicators for decision-making processes in range management additional to grass species. This might also be important for the success of local monitoring schemes.

2.4.1 What gives salience in local perception?

One of the reasons why woody species are the most important forage species in local perception is that their palatable biomass is available in times of low resource availability. In contrast, herbaceous species biomass is much more driven by the variability of annual rainfall, particularly in arid and semi-arid rangelands. Woody species are salient in local perception because they provide palatable biomass in the dry season and are thus important for livestock herding in scarce times. For our case study we conclude that it is this reliability that gives salience in local perception. These findings also support the basic idea of pastoralists being a highly reliable institution: pastoralists rather search for reliability than escape from risk (Roe et al. 1998). This is achieved by creating a reliable flow of life-sustaining goods and services from highly variable rangeland ecosystems.

In this way LEK, about processes or items may help in refining research hypotheses and give further insights into locally important and unimportant processes and items for researchers, managers and policy makers. For example, by analysing the traditional range management of the Ova-Himba, Müller et al. (2007), detected intra- and inter-annual heterogeneity of resource use through specific resting periods as crucial for a high productivity of biomass in semi-arid rangelands.

2.4.2 Implications for the methods

The main limitation of our methodological framework is that it is only as accurate as the data that it is based on. Contradictions might arise in the free-listing if plants are more commonly mentioned due to reasons other than their importance as forage species. For example woody species might be more prominent in the informant's minds for their height rather than their value as a forage source. Another reason for the high occurrence of woody species in local free-lists might be the importance of woody species in other areas of daily life, such as religious uses (Brauer 1925) or fire wood (Eichhorn 2004). Another option for the differences of local perception and ecological performance might be the relatively small area in which ecological data was collected since it was also found that local perception of important forage species is not restricted to the local grazing area.

In general, the application of the methodology made an in-depth analysis possible. It also confirmed that integrating LEK into scientific research is a valuable approach because it can complement and strengthen the scientific research with data on local valuation of certain ecological items, e.g. plant species and entities. This provides important aspects of local environmental perception by identifying preferences, and gaining new insights into local decision-making processes regarding natural resource use.

Apart from maintaining biological productivity and natural resource conservation, social acceptability is also regarded as an important principle for the sustainability of natural resource use in arid and semi-arid ecosystems (Snyman 1998). Consequently, the incorporation of local people's ecological knowledge and furthermore, their participation and integration into the research process, are key factors for sustainable development and advances the acceptance of innovative management approaches, such as adaptive management strategies (Berkes et al. 2000) or community-based natural resource management (Stuart-Hill et al. 2005).

The framework proposed here can be applied in various other fields of research. It is particularly interesting for studies on ecological items which supply ecosystem services, thus having a direct or indirect economic value. For example, medicinal plants, native food plants, or fish species have a direct economic value as provisioning ecosystem services, while fertile soil types and target species for conservation have an indirect value. All these ecological items have

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been subject to studies on LEK (Brown et al. 2004; Klintonberg et al. 2007; Angassa and Oba 2008).

The presented framework can identify indicators for changeable conditions, e.g. through the comparison of local perception of changes in ecological performance of an item and the assessed ecological performance of an item, e.g. over-utilization in fishery (Wilson et al. 2006), or for the condition of rangelands. We suggest that there is a role for the application and further development of this approach in ecological and social-ecological research. We recommend the application of the framework to complement ecological research with local ecological knowledge, which can provide important aspects in local natural resource management, biodiversity conservation and ecosystem service provision.

3 Decision – Water, forage or buffer – the management trilemma in semi-arid rangelands

3.1 Introduction

It is predicted that drylands worldwide will suffer particular impact from the consequences of climate change. The forecast for these areas includes not only decreased rainfall, but also an increase in temperature, rainfall variability and extreme events such as droughts and floods (Sivakumar et al. 2005; Scheiler and Higgins 2009). Because of their climatic characteristics drylands are mostly used as rangelands, with users mainly depending on extensive livestock production. These rangelands provide about 80% of the nutrition for livestock, and livelihoods for more than 370 million people, and they cover 40% of the African continent (Ellis 1994; MEA 2005). In addition to a higher probability of extreme events, grazing areas in drylands are also increasingly threatened by degradation caused by changing land use (Milton et al. 1994; Reynolds et al. 2007), a process which is amplified or attenuated by global forces (Lambin et al. 2001; Foley et al. 2005). There is a need to identify adequate management strategies that maintain the productivity of grazing areas in the face of a changing climate and land use (Walker 2002). In this context, it seems promising to analyse local ecological knowledge (Müller et al. 2007; Reynolds et al. 2007; Eisold et al. in review). After all, land users herding animals in a highly stochastic arid environment have always had to face low amounts and a high variability of rainfall, and have had to adapt their management strategies to these climatic patterns which are predicted to occur more frequently in the future. This ecological expertise may help to preserve crucial elements of local range management as well as improving other societies' ability to cope with climatic change and extreme climatic events such as droughts (Meze-Hausken 2000).

Many studies suggest that the adaptation of grazing impact to the spatio-temporal variability of natural resource provision is a crucial component of sustainable land use in drylands (Campbell et al. 2006; Quaas et al. 2007). However, feedback mechanisms between management strategy and natural resource dynamics are still not fully understood (Hein 2006; Tietjen and Jeltsch 2007). It is particularly difficult to predict the impact of climate change

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in arid environments with non-equilibrium herd dynamics (Tietjen and Jeltsch 2007; Gillson and Hoffman 2007).

3.1.1 Natural resource dynamics

As livestock is highly dependent on natural resources such as rangeland vegetation and water, resource dynamics in arid rangelands are driven both by variable rainfall and the impact of livestock itself (Todd 2006; Adler et al. 2005). It is thus crucial to identify mechanisms at the vegetation level which are capable of buffering rainfall variability (Stafford et al. 2009). A fundamental concept here is that of key resources (Illius and O'Connor 1999; Drees et al. 2009). Key resources were originally defined in the context of the rangeland disequilibrium theory as those resources with which livestock populations are in long-term equilibrium (Hamblen et al. 2007). Recently, this concept has been specified as that subset of resources which supports herds through times of scarcity (Ngugi and Conant 2008). In this sense, key resources are resources which buffer negative rainfall anomalies (Drees et al. 2009). Key resources have mostly been associated with certain parts of a grazing area (Illius 1999; Scholte and Brouwer 2008), thus emphasising the spatial heterogeneity of the arid landscape (Linstädter et al. 2007). Key resource areas were often identified in abiotically favourable landscape compartments such as riverbeds with deep soils and a water surplus due to lateral water transport (van de Koppel et al. 2002). Soils in these areas can effectively buffer rainfall variability due to their ability to store water (Reynolds et al. 2004). Here, the vegetation and its ecosystem service of supplying forage resources are partly decoupled from erratic rainfall patterns.

Besides this abiotic buffer mechanism, rainfall variability can also be buffered through vegetation. An important biotic buffer mechanism is storage (Wiegand et al. 2004; Smith et al. 2008; Linstädter et al. in press). This can be important in the case of individual plants in terms of energy accumulated in storage tissue (Owen Smith 2008; Müller et al. 2007; Milchunas et al. 1988). At the level of plant communities, an accumulation of stockpiled forage on pastures may also serve to buffer rainfall variability (Linstädter et al. in press). Storage tissue is comprised of the non-photosynthetic reserve organs which may be located below ground (as in the case of annuals after seed dispersal), above ground (as in the case of perennial grasses and forbs), or both (i.e. woody species where roots and stems may store energy). It is synonymous with reserve biomass

(Noy-Meir 1982; Müller et al. 2007). Stockpiled forage is palatable biomass which may accumulate on a pasture, and is synonymous with ‘green biomass’ (Müller et al. 2007). After the growing season, stockpiled forage, originally defined for perennial grasses, may be alive (in the case of evergreen plants) or dead (e.g. moribund biomass of perennial grasses (Zimmermann et al. 2009).

Range management (i.e. the frequency and intensity of disturbance by grazing animals) has a strong impact on the vegetation’s ability to buffer rainfall variability through storage mechanisms. Thus it is not sufficient to merely identify areas as key resources. We also need to understand functional connections between biotic buffers of plant species and management strategy.

Plants have differing ability to develop biotic buffers. Functional plant types defined for rangelands (e.g. annuals, herbaceous perennials and woody perennials (Gilson and Hofmann 2007) are characterised by a specific buffer capacity, and sensitivity to disturbances affecting this capacity (Table 3.1). These plant types are often applied in rangeland studies to account for functional differences in plants’ use of water resources and their response to disturbances such as fire and grazing (Tietjen and Jeltsch 2007). However, the capacity of functional plant types and pastures to buffer variable rainfall through storage mechanisms has rarely been explicitly addressed in experimental or modelling approaches.

A different management strategy is required to develop and/or maintain the different characteristics of storage tissue and stockpiled forage on a pasture (Table 3.1). In this context, management may either follow a long-term or a short-term strategy, whereby different levels of grazing impact (i.e. frequencies and intensities of grazing) are applied in the years before a drought event. As long as grazing impact is not too high, storage tissue will slowly build up over several years; otherwise, stored energy will either be directly destroyed or depleted. In the case of herbaceous and woody perennials, the production of stock-piled forage is highly dependent on storage tissue. Its accumulation on local pastures requires merely a short-term rest period in the pre-drought year, while in the long term the grazing impact may be medium to high. A pasture with a high biotic buffer capacity based on storage tissue is thus the outcome of a long-term management strategy, while a certain amount of stockpiled forage can also be achieved with a short-term management effort.

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Table 3.1: Functional buffer mechanisms of the two principal biotic buffers on pastures, specific capacity of functional plant types to build up these buffers (- none; + high; ++ very high), and management strategy to protect or maintain these buffers through changes in grazing impact, i.e. the frequency and intensity of grazing.

Biotic buffer	Storage tissue	Stockpiled forage
Description	Non-photosynthetic reserve organs below or above ground. Synonym 'Reserve biomass' (Noy-Meir 1982)	Palatable photosynthetic biomass, alive or dead. Synonym 'green biomass' (Müller et al. 2007)
Buffer mechanisms for negative rainfall anomalies (drought)	(i) Indirectly through positive effects on production of stockpiled forage during and after a drought event; (ii) Directly if storage tissue is itself palatable	Directly through immediate availability of palatable biomass
Management strategy		
Multiple-year grazing impact	Low to medium	High to medium
Grazing impact in pre drought year)	Medium	Very low (resting)
Specific buffer capacity		
Annuals	-	++
Herbaceous perennials	+	+
Woody perennials	++	+

3.1.2 Management strategies

We have argued that rainfall variability may be buffered by herd management mechanisms designed to protect and/or maintain storage tissue and stockpiled forage. We now ask how concrete strategies described for good-practice cases of range management implement these mechanisms. Several inter-dependent strategies have allowed pastoralists to survive for centuries in patchy and unpredictable low-productivity environments while sustaining their resource base. One of these, often regarded as the most important of all, is mobility (Fernandez-Gimenez and LeFebre 2006).

It is well known that herd mobility enables herders to cope with effects of rainfall variability and meteorological drought on the natural resource base (Sandford 1983; Thurow and Taylor Jr. 1999; Adriansen and Nielsen 2002; Bollig 2005). For arid non-equilibrium systems in particular it has been argued that spatial mobility is important in its function of accessing and sustaining forage biomass at a local level (Illius and O'Connor 1999; Illius and O'Connor

2000), and accessing non-local resources in times of drought (Bollig 2005). Herd mobility has thus mostly been described as an opportunistic strategy which enables users to avoid adverse environmental conditions and gain intra- and inter-seasonal access to natural resources in different localities or regions (Bovin 1990; Al-Eisa, 1991; Adriansen 2005; Dwyer and Istomin 2008). This suggests that the main objective of mobility is to take the animals where there is forage and/or water in order to maximise livestock survival. Mobility of this kind ('resource exploitation mobility', Oba and Lusigi 1987) is analogous to the movements of large wild herbivores within their arid environments. Mobility would then be a mere coping or resource exploitation strategy (Oba and Lusigi 1987; Fabricius 2007) on a short-term scale, i.e. a fall-back mechanism aiming at counteracting the detrimental effects of drought after its occurrence. In contrast to this view, mobility may also be a strategy with adaptive character (Roe et al. 2005). Adaptive strategies are adopted as long-term, proactive and planned measures. They aim not only at survival but also at sustainable management, including that of natural resources (Fabricius et al. 2007). In this sense, management of the biotic buffer capacity of forage resources is an adaptive aspect of mobility (Roe et al. 1998), as it acts in a preventive way by increasing plant vigour and growth in storage tissue, and by protecting stock-piled forage on local pastures via rest periods at least in the pre-drought year (Table 3.1).

Mobility decisions in a specific management context are thus to be discussed in the context of the inherent objectives of local land users. These are (i) immediate access to forage resources, (ii) immediate access to water resources (both of which are coping objectives), or (iii) adaptive management of the biotic buffer capacity of forage resources. We call this the management trilemma for mobility decisions. In times of scarce resources such as drought years, we postulate that the motivation trilemma will be reduced to a dilemma because the objectives are merely to maximise livestock survival through immediate access to forage or water resources (i.e. through escape mobility). Following the concept of key resource areas we expect local herders to fall back to grazing areas with high abiotic buffer capacity (or favourability).

To test these assumptions, we used ecological and anthropological data from a case study in semi-arid Namibia. The case study approach (Fotheringham 1997) provided an appropriate means of exploring adaptation, in particular the way that local adjustments to change are shaped by interacting processes. To

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test the first assumption that biotic buffers may be built up through adaptive management of pasture resources, we compared ungrazed areas under communal and commercial management along a gradient of abiotic favourability to take account of the intrinsic buffer potential in the area. This approach allowed us to compare two strategies with respect to pre-drought management, i.e. how much emphasis was placed on the long-term and short-term aspects of adaptive resource management. To test our second assumption that in times of drought herders practise only 'escape mobility', pursuing the objective of feeding or watering their animals and making use of pasture with high abiotic buffer capacity, we analysed the mobility decisions of a pastoral-nomadic user group in a drought year, and related these to the biotic and abiotic buffers of pastures used during this time. Finally, we asked if our results allowed general conclusions to be drawn about sustainable forms of management in semi-arid environments.

3.2 Materials and methods

3.2.1 Study area

Location

We examined two land use systems in a semi-arid savanna ecosystem in north-western Namibia under either private or communal land tenure (Fig. 3.1). The communal areas are situated in the emerging Orupupa Conservancy, in Namibia's Kunene region. The Conservancy is an institution of the 'Community-based Natural Resource Management' programme in Namibia. Conservancies take responsibility for the natural resources, mainly wildlife, within their boundaries by monitoring, managing and conserving them. The conservancy in question covers 1650 km² (NACSO 2007) and two habitats. The north-eastern part is a plain with sandy soils dominated by grasslands. The other parts are plains and hilly habitats dominated by mopane savanna. The commercial farms are situated about 90 km south-east of the Conservancy, on private land. Site conditions and vegetation are similar to the Conservancy. Soils vary from deep sand to loamy clay. Data was recorded at the end of the 2007 rainy season (April to June). For a map of the study area see Fig. 1.1.

Climate

Northern Namibia's climate is semi-arid. The area receives an average annual precipitation of 200 to 300 mm/year. Rainfall is highly variable in time and space with inter-seasonal variability between 40 and 60% (Mendelsohn et al. 2003). Northern Namibia has two main seasons, the dry season that lasts from May to October and the wet season extending from November to April. Annual potential evaporation is estimated to exceed the annual precipitation by a factor of about five (Mendelsohn et al. 2003).

The 2006/2007 rainy season showed below average rainfall and was characterised by the local land users as a "bad rain year". The Kunene region received an average of 183.67 mm in 2007, while in 2006 it received 460.5 mm/year.¹⁰ Local land users reported low rainfall in the north-eastern part of the Conservancy, in the sandy plains. In the commercial farm areas, 144 mm of precipitation were recorded in the central part of this area between October 18, 2006 and April 4, 2007. This is about 50% below the long-term mean.

Population and land tenure

Pastoralism, including both communal and commercial forms of livestock farming, is one of the main sources of livelihood in Namibia (Mendelsohn et al. 2003). The northern part of Namibia is communal land. Fencing is not allowed and livestock can roam freely (Leggett et al. 2003). Management strategies have evolved from traditional pastoral nomadism and have been influenced by the restrictive politics of the last century which forbade any commercial trading of livestock until the year 1990 due to quarantine concerns. Since then, stocking rates have nearly doubled from 92607 in 1990 to 183512 in 2003 (Bollig 2005). Because of quarantine concerns it is still difficult for livestock owners to sell their animals at markets. Most products are for subsistence and most herders have only a low or non-existent income (Tönsjost 2007; Faschina 2010).

The commercial farms belong to private farmers. In Namibia's Kunene region the average size of a commercial farm in 2007 was 97.3 ha.¹¹ The farms are divided into camps, where rotational grazing has been practised since the 1950s.

¹⁰ http://www.phlosses.net/index.php?form=rainfall&tc_id=14

¹¹ http://www.phlosses.net/index.php?form=average_farm_size&tc_id=14

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3.2.2 Survey design

Vegetation was sampled in communal areas and on commercial farms in areas that were part of the respective grazing system. Plots had comparable site conditions and for each communal grazing area a benchmark was selected that had not recently been grazed.

In each grazing system (commercial and communal), we selected areas with low, medium or high abiotic favourability. This was classified according to top soil characteristics. A high sand content was declared to be indicative for high abiotic favourability, as studies in arid and semi-arid environments show that relatively more water is available to plants on sand-rich soils (English et al. 2005).

To assess the biotic buffer realised by a grazing system, we compared ungrazed communal areas to commercial areas, with the latter serving as a benchmark. For assessing decisions of pastoral mobility, grazed and ungrazed communal areas were compared.

For details on the selected pastures (study sites), see Table 3.3. Data collection on herd mobility in the communal grazing system was carried out by an interdisciplinary team that included a socio-agronomist. We analysed factors influencing mobility by means of participatory observation, area drives and interviews with local herders. Grazed pastures in the communal area were selected according to the sequence of utilisation at the onset of a drought period. This was done by following herds from the home range to further grazing areas. The selection of study sites thus reflects mobility decisions in face of drought and at the same time a range of pastures with different abiotic favourability (low, medium and high; see Table 3.3).

Vegetation was sampled on a total of 57 plots with five to nine 1000 m² plots in each study site (Table 3.3). In selecting the plots, an attempt was made to minimize geographic variability such as slope and aspect.

The species recorded in the study were each classified as one of the following life forms: woody species (trees and woody shrubs), perennial grasses and annual grasses. For each of the nine areas sampled, the mean cover provided by all species and within each life form was calculated.

The biotic buffer was gauged on the basis of stockpiled forage, quantified as ground cover from perennial and annual grasses, and storage tissue of the grass layer, quantified as cover from perennial grasses.

A top soil sample (0–5 cm) was taken from five randomly selected spots on each plot (Ladd et al. 2009). The sand fraction (2–0.063 mm) was sieved. Silt and clay contents (0.063–0 mm) were determined by pipette analyses (Schlichting et al. 1995). Samples were analysed for soil texture at the Soil Laboratory of the Institute for Geography, University of Osnabrück and the Agricultural Soil Laboratory in Windhoek.

3.2.3 Statistical analysis

The paired Wilcoxon signed-rank test was used to test for significant changes in biotic buffer potentials (cover of annual grasses, perennial grasses and woody species) between the ‘ungrazed commercial farm’ and ‘ungrazed communal’ management strategies. In addition to examining abiotic favourability (the sand fraction), the paired Wilcoxon signed-rank test was also used to test the driving factors behind mobility decisions, comparing biotic and abiotic buffers in grazed and ungrazed communal areas.

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Table 3.2: Characteristics of the study sites (pastures) in the communal and commercial grazing system.

Code	Landform	Utilisation	Abiotic favourability ⁺	Mean sand content [%]	<i>n</i>	Permanent water point	Distance to home range [km]	Altitude [m.a.s.l.]
1	Undulating sandy plains interspersed with low hills	Home range, grazing area directly connected to the settlement	Medium	63.16 ± 18.41	7	Yes	-	1016
2	Undulating sand plains	Accessed soon after onset of drought	High	89.97 ± 2.46	7	Yes	35	1234
3	Intramontane stony plain	Accessed later that season	Low	61.83 ± 5.77	8	Yes	5	1059
B*1	Undulating sandy plains interspersed with low hills	20 years of no grazing	Medium	76.30 ± 2.17	5	No	10	1069
B2	Undulating sand plains	Very low grazing intensity	High	89.67 ± 1.38	5	No	40	1203
B3	Intramontane stony plain	Dry season pasture, 3 years of no grazing	Low	60.44 ± 17.53	6	Broken	15	1345
F1	Undulating sandy plains interspersed with low hills	Commercial game farm	Medium	74.79 ± 3.99	6	Yes		1294
F2	Undulating sandy plains interspersed with low hills	Commercial game farm	High	84.56 ± 2.5	5	Yes		1281
F3	Undulating sandy plains interspersed with low hills	Commercial game farm	Low	68.76 ± 8	8	Yes		1282

* B = Benchmark

⁺ Classified according to sand content.

3.3 Results

3.3.1 Buffer potentials in different management strategies –

Comparing ungrazed areas under communal and commercial land tenure

For assessing differences in abiotic buffer capacity, we used the sand content of a plot's top soil. Sand content differs significantly between all favourability classes.

Table 3.3: Difference matrix of abiotic favourability (fav.) calculated on the basis of sand content. The significance of differences was tested using a paired Wilcoxon signed-rank test; n and significance levels are given.

Sand content	Fav._low	Fav._medium	Fav._high
Fav._low	-	0.017 (n = 21)	0.00044 (n = 16)
Fav._medium	-	-	0.00044 (n = 16)

We compared ungrazed communal areas and ungrazed areas on nearby commercial farms to assess how much palatable biomass had been accumulated as biotic buffer by annual and perennial species. Regardless of management strategy or abiotic site condition, just 29% of the 45 cases differ significantly.

In the perennial grass layer, 47% of the cases differ, in the annual grass layer only 20%, and within the woody species 13%. Our results indicate a significant impact of management strategy on perennial grasses (Table 3.4).

All farm sites had higher mean perennial grass cover and often lower annual grass cover than the communal areas (Fig. 3.2). Within a management strategy, perennial grass cover increased in line with abiotic favourability. In contrast, mean annual grass cover was higher in the communal areas than on most farm sites. While relatively constant in communal areas, it increased with abiotic favourability on farms. The influence of the site's abiotic condition on vegetation structure is higher in farm areas than in communal areas.

On average, perennial cover on the communal rangeland was less than one third of the cover on commercial rangelands (Fig 3.2). Annual cover on the commercial rangeland was, however, less than half of the cover of the communal rangeland, indicating that the lower perennial cover has been at least partially compensated for by an increase in annual plant cover. Woody species

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cover was generally higher in the communal area than on the farm sites. It was highest on sites with medium favourability. In the farm areas the amount of woody biomass is hardly affected by abiotic favourability.

Table 3.4: Difference matrix of annual grasses (Ann. grass.), perennial grasses (Per. grass.) and woody species cover calculated between sites with low, medium and high abiotic favourability (Fav._low, fav._med., fav._high) and between and within management strategies, as well as between farm and communal ungrazed areas (Farm_ungr., Com_ungr.). The significance of differences was tested using a paired Wilcoxon signed-rank test; significance levels are given ($n = 5$). Highlighted values denote significant differences between sites, framed are the differences between management strategies.

		Com_ungra			Farm_ungra		
		Fav._low	Fav._med.	Fav._high	Fav._low	Fav._med.	Fav._high
Ann. grass. cover							
Com_ungr.	Fav._low	-	NS	NS	NS	NS	NS
	Fav._med.		-	NS	0.04	NS	NS
	Fav._high			-	0.04	NS	NS
Farm_ungr.	Fav._low				-	no result	0.04
	Fav._med.					-	NS
	Fav._high						-
Per. grass. cover							
Com_ungr.	Fav._low	-	0.04	NS	NS	0.03	0.04
	Fav._med.		-	NS	NS	NS	0.04
	Fav._high			-	NS	NS	0.04
Farm_ungr.	Fav._low				-	NS	NS
	Fav._med.					-	0.04
	Fav._high						-
Woody species							
Com_ungr.	Fav._low	-	NS	NS	NS	0.05	NS
	Fav._med.		-	0.04	NS	0.04	NS
	Fav._high			-	NS	NS	NS
Farm_ungr.	Fav._low				-	NS	NS
	Fav._med.					-	NS
	Fav._high						-

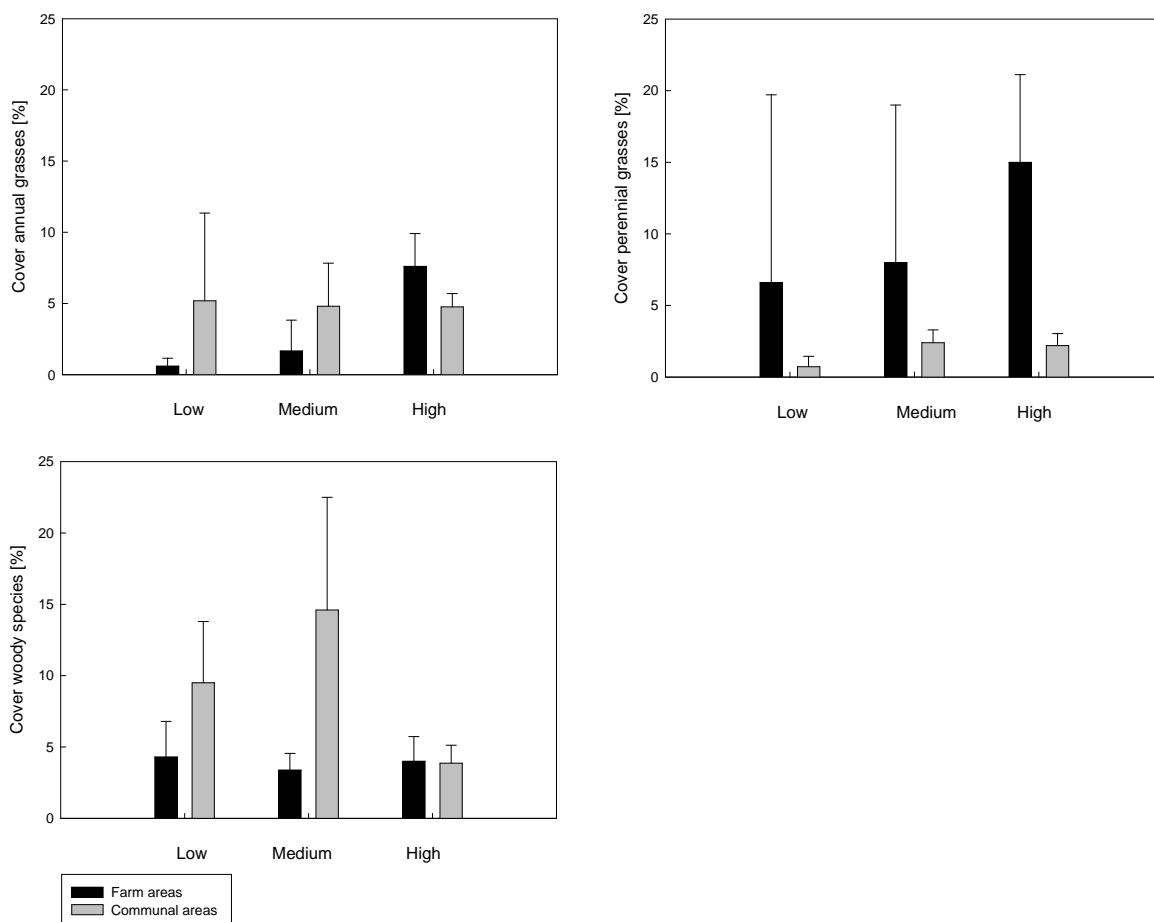


Fig. 3.2: Means and standard errors of annual grasses, perennial grasses and woody species cover for farm and communal areas with low to high abiotic favourability.

3.3.2 Driving factors behind mobility decisions – comparing grazed and ungrazed communal areas

In the communal areas we compared the differences in abiotic site condition, biotic buffer, and availability of water for livestock in the context of mobility decisions on the part of local herdsman. We analysed what made the herders utilise a certain area at the onset of a drought period in the context of the postulated motivation dilemma, i.e. whether mobility decisions were more dependent on the availability of forage or of water.

After examining the pastures surrounding the permanent settlement ('home range'), two grazing areas within the Conservancy were accessed consecutively between May and June 2007 (Fig. 3.3). From the home range, parts of the herds went into an area with high abiotic favourability and a correspondingly high amount of biotic buffer (Pasture 2). However, further mobility decisions were taken in favour of areas with lower amounts of biotic buffers. From the

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home range which had no perennial grass cover, a few cattle were moved to an area 35 km away, with higher abiotic favourability and a nine times higher amount of herbaceous stockpiled forage. But most cattle stayed in the home range, while in an adjacent area, only 10 km distant from the waterpoint of the home range, the level of stockpiled forage of annual grasses was almost ten times higher, and perennial grass cover rose to 2.4%. Some cattle were even moved to another adjacent area which was also without perennial grass cover and where annual grass cover was half that in the home range (Pasture 3).

Of these three pastures, the home range had lower annual and perennial grass cover and abiotic favourability than Pasture 2, but higher vegetation cover and abiotic favourability than Pasture 3. Woody species cover was high in all areas and water was permanently available. The ungrazed areas close to these three areas had higher mean perennial grass cover and generally higher annual grass cover. Perennial cover on the grazed areas was only a small fraction of that existing in ungrazed areas (Fig 3.4). The level of annual cover on the grazed areas was on average one quarter lower than that on the ungrazed areas, indicating that the lower perennial cover had not been compensated for by an increase in annual plant cover. Woody species cover did not differ significantly between grazed and ungrazed areas. Permanent water was not available on any of the ungrazed sites.

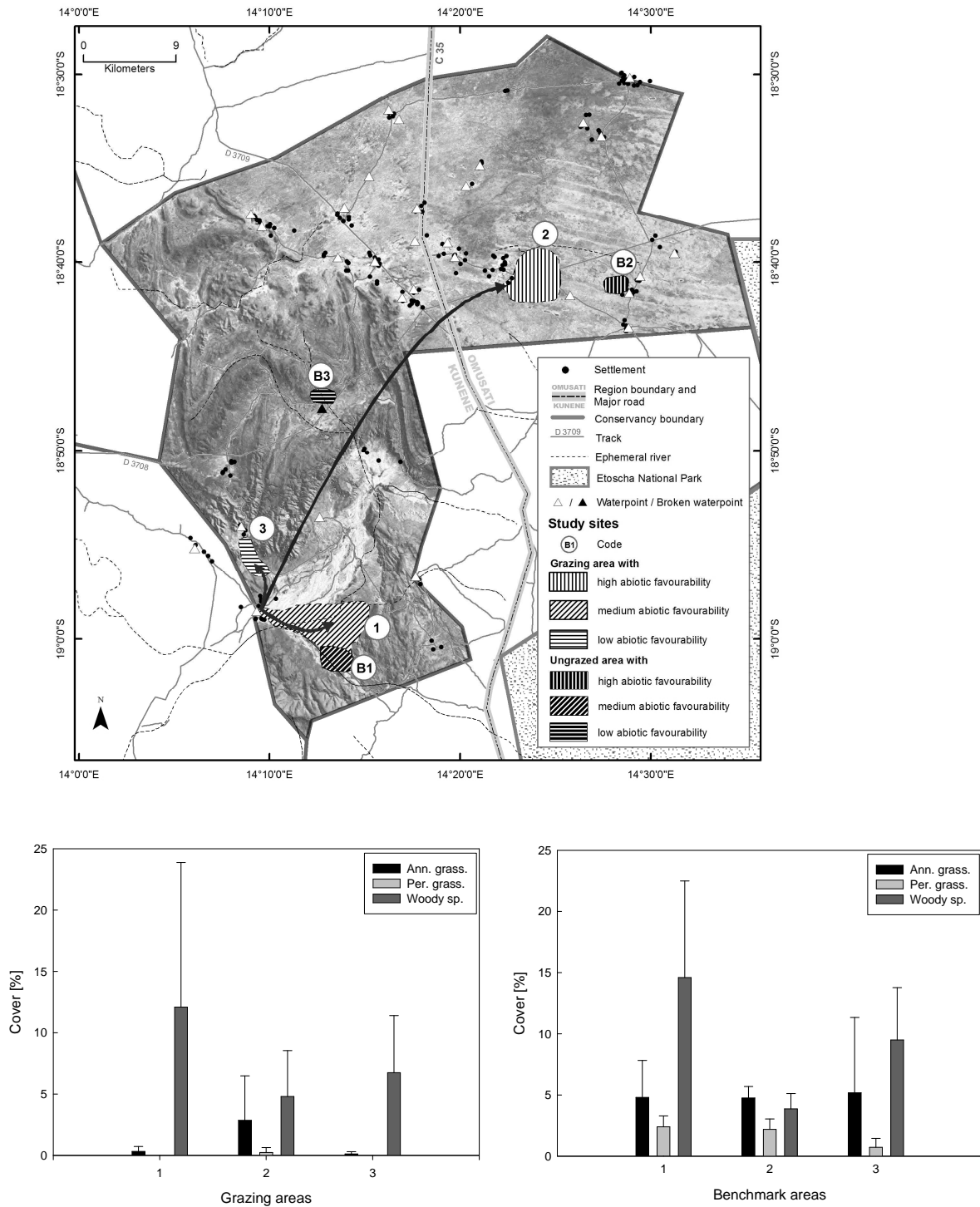


Fig. 3.3: Map of the study areas with means and standard errors of life forms for communal pastures compared with corresponding ungrazed reference sites, following the sequence of access from 1 (home range) to 3, compared with the corresponding ungrazed reference sites (B1, B2, B3). Black arrows indicate the grazing routes. See Table 3.1 for further details. Source: ConInfo 2009; Cartography: Holger Vollbrecht.

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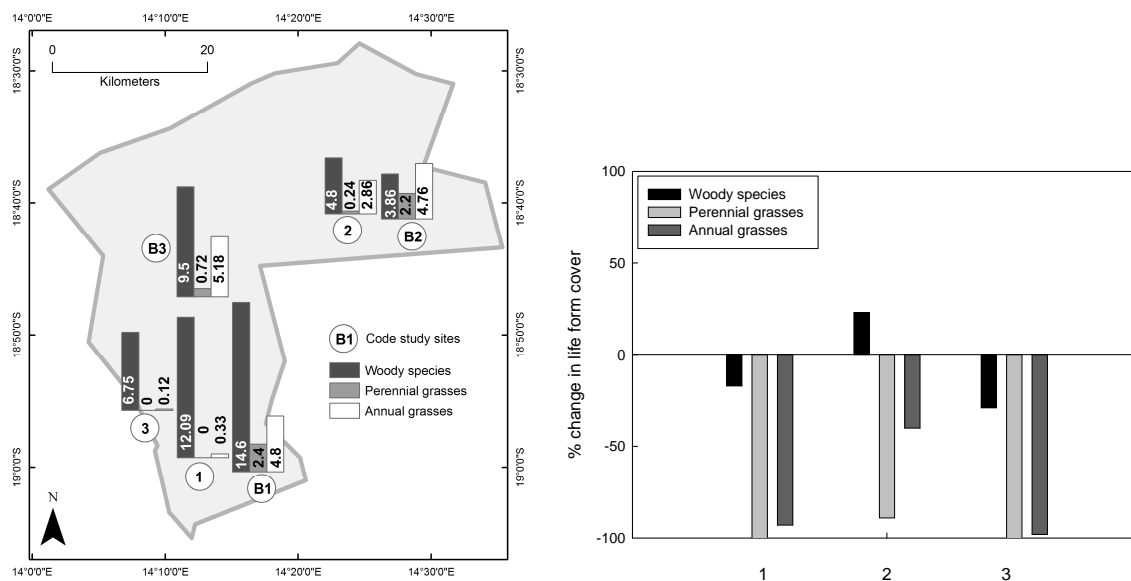


Fig. 3.4: The biotic buffer realised in the communal areas, shown as the accumulation of stockpiled forage of annuals (as a direct buffer) and of storage tissue of perennials (as a mixture of direct and indirect buffering effects). The cover in the ungrazed communal areas is set as a benchmark, set as 0%. (Note that in the two visualisations bars indicating life form cover have different fill colours.)

3.4 Discussion

3.4.1 Aspects and objectives of management strategies

Farm management results in a higher yield of storage tissue, accumulated by perennial species, than of stockpiled forage, accumulated by annual grasses. In communal areas there is more annual than perennial grass, resulting in a higher yield of stockpiled forage than storage tissue. Findings indicated that buffer potential could not be adequately utilised on any of the communal pastures. As regards the grass layer, a long-term buffer of perennial grass storage tissue was not established, and the existing short-term buffer potential of stockpiled forage from annual grasses was not fully exploited.

This indicates that in the communal areas, users emphasize short-term rather than long-term aspects of grazing management strategies. Compared to the farm areas, communal users do not practice long-term management of buffer mechanisms. They allow the accumulation of stockpiled forage, particularly of annual species, via short-term rest periods for these areas. They do not, however, protect the storage tissue of perennial grasses. Furthermore we showed that in the communal areas, the vegetation on pastures with abiotic buffers is in a better condition than in other communal areas. The abiotically favourable

areas manifest higher productivity and a higher amount of storage tissue, implying a higher recovery potential after a drought (see Table 1), but are still in a poorer condition overall than the neighbouring farm areas. Here the better relation of grazeable area and animal numbers (i.e. a lower overall stocking density) may allow for a more long-term management strategy. We conclude that more features of an adaptive strategy are to be found in farm than in communal areas.

In a year with below average rainfall and scarce natural resources we analysed why certain communal grazing areas were accessed and others not. The hypothesis we put forward was that areas with a high buffer capacity will be used preferentially. This was only partly confirmed. The results indicate that it was not the amount of stockpiled forage in the grass layer that drove mobility decisions of local herdsman (Fig. 3.3), but water availability, with only partial pursuit of biomass. An important point is that all the areas accessed contained woody palatable biomass and permanent waterpoints.

During a drought period where both forage and water were scarce natural resources, a common feature of those areas accessed was not a high abiotic or biotic buffering capacity, but permanent waterpoints. We conclude that it is the availability of water that mainly drives mobility decisions. Neither the presence of accumulated palatable biomass nor other factors which might possibly influence mobility decisions (e.g. a pasture's position within a landscape, or spatial extent) were important for a specific decision. In our case study, one of the three ungrazed areas not initially accessed, and used as a benchmark, was accessed very late in that season when water became available there. We explain this in two different ways. Firstly, in the motivation dilemma water is favoured; thus the availability of forage plays a minor role for mobility decisions in times of scarcity. Secondly, the biotic buffer capacity of the grass layer does not play an important role in local range management. This corresponds to the observation that in local perception woody species are more important than herbaceous species (Kemmerling et al. resubmitted; Eisold et al. in review; Baumann 2009). In our case, local land users perceived woody species to be more important than herbaceous species because these species provide palatable biomass in times of low resource availability, and are thus an important objective of management decisions for livestock herding in resource-scarce times. In contrast, the availability of herbaceous biomass is far more subject to the high variability of annual rainfall, particularly in arid and semi-

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arid rangelands. It is thus the reliability of woody species that is uppermost in local perception (Eisold et al. in review; Kemmerling et al. resubmitted). As a last resort, livestock can forage on the palatable biomass of the woody species, which “does not make them fat but it makes them go by” (pers. comm.).

In this study we neglected feedbacks between abiotic and biotic buffers. Perennial species, in particular perennial grasses, can protect the soil from erosion processes more effectively than annuals (Cerdà 1997), thus maintaining the abiotic buffers of a site. Range management which keeps perennial plant cover above a critical threshold of 10% assumed to be significant for erosion (Thornes and Francis 1990) therefore amounts to indirect management of abiotic buffers (Linstädter et al. in press). It may also increase an area’s ability to sustain its productivity in the face of increasingly frequent and severe extreme climatic events (Coppus and Imeson 2002). A short-term strategy may still be appropriate in areas where potential erosion rates are low, and/or alternative pastures are accessible. Here, users enjoy good conditions for working with annual vegetation.

3.4.2 Implication for management

In the communal areas, the vegetation’s buffer potential is not exploited with the current grazing system, which does not per se mean unsustainable land use. This is only the case if alternative options or fall-back mechanisms such as alternative pastures or alternative livelihood strategies are missing. Still, in our case study we assumed that the criteria for sustainability are not fulfilled. This cannot be judged from the pasture itself, but needs to be assessed on the basis of what other possible livelihood strategies are available to people (Snyman 1998). In our case, the set of possible actions is limited. For several reasons, herders have no alternative pastures for times of low-resource availability. This is partly the result of an increasing settlement and livestock density (Bollig 2005; Klintonberg and Verlinden et al. 2007; own data). Limited grazing areas and additional pressure on the grass resources (Pieri 1992; Maire and Delpuech 2004; Guenguant 2005) result from an increase in livestock numbers (Jacobs and Coppock 1999). According to local herders, most of the strategies have been adopted because of growing land scarcity rather than because of climate variability, which is also confirmed in a study by Barbier (2008) in a semi-arid rangeland. We assume livestock numbers have reached a critical threshold, while land scarcity, declining soil fertility and reduced animal mo-

bility have pushed farmers to neglect their sustainable strategies (Barbier 2008). Traditional mechanisms of risk minimisation such as the resting of specially accounted reserve areas become increasingly ineffective, and buffer capacity degrades (Samuels et al. 2007). Land pressure is most acute in marginal pastoral areas, where livestock husbandry can have unfavourable effects on natural resource supply.

Our results show that there is not automatically a best management practice for a certain area, but there are different options available, with different means of managing the buffer mechanisms. We conclude that there are different coping or adaptive strategies open to pastoralists in arid environments. There can be a wide spectrum ranging from users investing strongly in biotic buffers and those more inclined to pursue short-term strategies. The latter case is common in areas where only few alternatives such as forage resources outside the local grazing area are available (Fernandez-Gimenez and Le Febre 2006; Breuer et al. 2007). As far as local and regional mobility are concerned, there is a choice between preventive (Linstädter et al. in press) and coping mobility; as well as a series of possible combinations. Various alternatives are available, but in general it is advisable to conserve biotic buffers because otherwise the area is threatened by vegetation and soil degradation. There could be a compromise between short- and long-term objectives in rangeland management in order to optimise the quantity and quality of plant production. The short-term productivity of the rangeland ecosystem is a function of climatic variations, soil characteristics, stocking rate, management systems and the type of animal farmed. On the other hand, long-term optimisation of plant and animal production involves the prevention of natural resource degradation (Snyman 1998). A holistic approach to the investigation of rangeland ecosystems is essential for sustainable future production and stability.

4 Grazing impact in space – indicators for rangeland assessment

4.1 Introduction

Rangelands cover more than 40% of the African continent, provide about 80% of the nutrition for livestock and the livelihoods for more than 370 million people (Ellis 1994; White et al. 2002). Degradation of natural resources in rangelands has become a serious challenge, bearing negative impacts on the pastoral ecosystems, livestock production and livelihood thereof (Vetter 2005; Kassahun 2008). This socio-economic importance makes it crucial to assess the impact of land use on this type of ecosystem. However, it is difficult to disentangle the effects of land use, such as potential long-term degradation, from environmental fluctuations in arid and semi-arid rangelands because they are characterised by a high environmental variability and non-linear dynamics (Campbell et al. 2006; Gillson and Hoffmann 2007). Thus it is crucial to find suitable indicators that take into account environmental variability as well as the effects of land use on the ecosystem including non-linear dynamics (Pyke et al. 2002; Ludwig et al. 2004). However, this is still a very difficult task (Reed et al. 2008).

Many different definitions of degradation exist (see Reynolds and Stafford 2002 for a review on degradation). Some authors define degradation as processes that irreversibly deprive a system of ecosystem service supply (Gorse and Steeds 1987; Mainguet 1991). Others focus on the reduction of ecosystem service supply, without a reversibility criterion (Hellden 1991; UNEP 1992). What both approaches have in common is the recognition that relationships, e.g. between plant community structure and grazing intensity are not linear (Thrash 1998; Cingolani et al. 2005). Recently there has also been much debate on degradation not being a linear process and that thresholds in space and time can be recognized (Lindenmayer and Luck 2005; Briske et al. 2006; Groffman et al. 2006).

Non-linear response means that abrupt changes in ecological conditions occur at ecological thresholds, rather than a continuum of variation (Gillson and Hoffmann 2007). Thresholds are defined as points at which relatively rapid changes occur from one ecological condition to another, along a gradient in a prevailing disturbance regime (Briske et al. 2003; Radford et al. 2005; Sasaki et

al. 2008); they are “points where even small changes in environmental conditions will lead to large changes in system state variables” (Suding et al. 2004). Ecological thresholds can be categorized into two groups, structural or compositional thresholds and functional thresholds (Briske et al. 2005). Structural thresholds are based on changes in community composition, plant growth form, and the occurrence of invasive species (Scheffer et al. 2001; Stringham et al. 2003). Functional threshold components are defined by positive or negative changes in various ecological processes, such as soil properties, nutrient cycling and productivity. Due to the time required for such processes, functional processes are expected to lag behind structural thresholds.

The threshold concept in rangeland science was originally associated with state-and-transition models (Westoby et al. 1989; Milton and Hoffmann 1994). These models can accommodate discontinuous and irreversible vegetation change (Briske et al. 2003). However, arid and semi-arid rangelands may show both, continuous (reversible), and discontinuous (nonreversible) vegetation dynamics (McAllister et al. 2006). The first implicates shifts within stable states, thus from one plant community to another, the later represents a threshold from one stable state to another (Briske et al. 2005) (Fig. 4.1). Once a threshold has been crossed, the vegetation changes are not easily reversible, depending on the interactions of numerous climatic, edaphic, and biological factors in combination with the economic feasibility of rehabilitation (Reynolds and Stafford 2002).

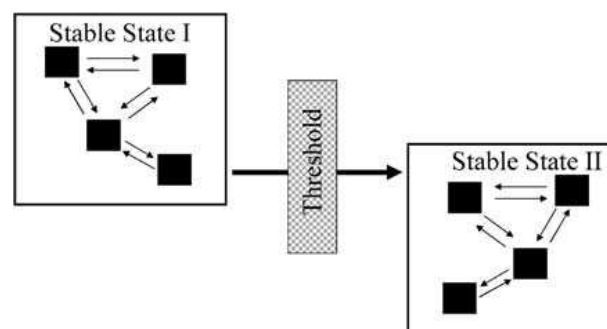


Fig. 4.1: Potential pathways of vegetation dynamics. The big squares illustrate the plant communities between stable states, where transitions are nonreversible. The filled smaller squares illustrate plant communities within stable states, here transitions are reversible. Source: Briske et al. 2005.

Today special emphasis is placed on management frameworks for thinking about the dynamics involved in rangeland degradation and restoration (King and Hobbs 2006). This is often done in terms of preventive or restoration

4. Grazing impact in space

thresholds, aiming at the re-establishment of former states following active restoration of autogenic repair mechanisms and feedbacks (Briske et al. 2006; Hobbs and Suding 2009).

Within this concept important elements especially for management frameworks are a reference state, an alternative state, and an at-risk community phase within a state. Thresholds exist from one state to another, as the limits of resilience for an ecosystem. The alternative state is reached once a threshold is crossed, while the reference state can be defined as the optimal vegetation of a grazing area with regard to desired land use (Aucamp et al. 1992). What is considered best practice will differ among land users, depending upon their management objectives (Linstädter 2008). The type of reference or benchmark which will be most useful for general rangeland monitoring will be one that exhibits the healthiest ecological processes and is best protected from erosion under the prevailing macroclimatic conditions (de Klerk 2004). An at-risk community phase is a plant community phase that is most vulnerable to exceeding state resilience. Within these states indicators, feedbacks and triggers must be identified, as well as thresholds between the states. Triggers are variables or events that initiate thresholds by contributing to the immediate loss of ecosystem resilience, such as a severe drought. Within a state feedback mechanisms exist, ecological processes that enhance (negative) or decrease (positive) ecosystem resilience (Briske et al. 2008).

However, ecosystem state changes are still often 'ecological surprises' (Peters et al. 2004). Our difficulties to anticipate (or reverse) such catastrophic transitions result from our inability to understand the full suite of mechanisms driving and maintaining them (Bestelmeyer et al. 2006; Groffman et al. 2006, Peters et al. 2009). From a rangeland management perspective, degradation thresholds need to be identified before they are crossed, by providing early warning indicators (Sasaki et al. 2008; Scheffer et al. 2009). A promising approach is a threshold evaluation on the basis of disturbance regimes, as it can more explicitly identify the drivers of vegetation change and provide additional insight into ecosystem dynamics (Briske et al. 2003; Jentsch 2007). In addition, the effects of multiple processes, scale, spatio-temporal pattern, and soils have to be considered (Bestelmeyer et al. 2006).

4.1.1 Methodological framework – spatial gradients of land use

Studies of grazing gradients are an important tool in rangeland ecology to address such complex system dynamics, particularly to analyse the functional response of an ecosystem along spatial gradients of land use (e.g. Tilman et al. 2001; Adler and Hall 2005). Two types of functional response have to be distinguished: first the response along a disturbance gradient, caused, e.g. by land use impact (Smet and Ward 2006; Abule et al. 2007), and second the response along a resource gradient. Feedbacks between these two types may occur, such as an impact of land use on resource availability via nutrient redistribution and discharge (Smet and Ward 2006; Baumann 2009), or through the change of physical soil characteristics (Greenwood and McKenzie 2001). The most important feedback exists between the impact of disturbances (such as grazing) and the availability of water, the latter being the main limiting resource for plant growth. It is thus crucial to analyse potential early-warning indicators and threshold dynamics along a broad range of combinations between water resource supply and disturbance impact.

In arid and semi-arid grazing systems, spatial gradients of animal impact are also known as piospheres (Todd 2006), and are defined as “the zone of ecological impact surrounding a water point” (Andrew 1988). They are used as model systems for studying vegetation patterns in response to grazing. This space-for-time approach is seen as an alternative to long-term studies (Pickett 1989) and has been frequently applied because vegetation is thought to show comparable responses along temporal and spatial gradients of grazing impact (Jeltsch et al. 1997; Thrash 1998, Landsberg et al. 2003; Todd 2006). Recently threshold dynamics in rangelands have also been studied along piospheres (Sasaki et al. 2008), focusing on the response of floristic composition.

Many empirical studies have reported different responses along land use gradients, from positive to negative, depending on spatial and temporal scales (Olff and Ritchie 1998), intensity and duration (Waser and Price 1981), evolutionary history of grazing (Milchunas et al. 1995; Laurenroth 1998), rainfall and soil nutrients (Meserve et al. 2003), and livestock density (Kelt and Valone 1995) among the most important factors. Most studies agree that the impact of concentrated grazing on natural resource dynamics leads to a marked reduction in forage resources mainly due to shifts in vegetation composition, e.g. from dominance by perennial grasses towards dominance of annuals (Schulte 2002; Sasaki et al. 2008). This process could be categorised as the first ecological

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shift before grazing sensitive species are eliminated from the system (Briske et al. 2006).

Indicators applied in rangeland studies to detect potential threshold dynamics should be capable of capturing and integrating processes of change on different temporal scales, and respond with different sensitivity and memory to grazing impact (Albon 2007; Beever et al. 2003; see overview on indicator traits in Table 4.1).

For example, plant functional types are biotic indicators on the level of species with a long to short-term memory for grazing impact and various responses to grazing impact and variable rainfall (Abule et al. 2007; Díaz et al. 2007). In contrast, soil parameters are good indicators for a long-term degradation of abiotic site conditions. They are less affected by short-term rainfall fluctuations and drought than plant functional types – except some woody species – are (Dougill and Cox 1995; Smet and Ward 2006).

Table 4.1: Rangeland indicators and their response (sensitivity and memory) to grazing and environmental variability. Direct indicators for grazing, and (indirect) abiotic and biotic indicators are distinguished. PFT = Plant Functional Type. Sensitivity of response –: none, o: low, +: high, ++: very high. The indicators analysed in this study are marked in bold.

Indicator group	Example	Memory for grazing impact	Sensitivity to grazing impact	Sensitivity to environmental variability
Direct indicators				
Recent grazing	Trampling, dung, bare ground	Short-term	++	-
Abiotic indicators				
Soil physical indicators	Compaction, infiltration	Long-term	+/o	-
Soil chemical indicators	C _{org} , N, P	Long-term	+/o	-
Biotic indicators: Species level				
PFT: Annual herbaceous plants	Annual grasses	Short-term	++	++
PFT: Perennial herbaceous plants	Perennial grasses	Medium	++	+
PFT: Woody plants	Trees, shrubs	Long-term	+	o
Biotic indicators: Populations and individuals				
Palatable perennial plants	Forage trees and grasses (age structure, damage of individuals)	Medium to long-term	Variable	o

The parameters “memory for grazing impact” and “sensitivity to environmental variability” show a wide range. In this study we systematically analyse a set of indicators which cover a broad range of response, particularly to grazing impact. We further test whether a piosphere, as a spatial gradient approach, is suited to identify early-warning indicators and thresholds under different range management, i.e. whether the space-for-time-substitution is appropriate in this case.

Specifically, the present study addressed the following questions.

- i. Are transects (spatial gradients of land use) appropriate tools for disentangling the effects of environmental variability and land use impact in a semi-arid savanna?
- ii. Can we identify indicators for threshold dynamics (early-warning-indicators) along spatial gradients?

For analysing these questions, gradients of land use have to be sufficiently long to cover potential system shifts, environmental variability should be high and specific response of indicators to disturbance (grazing) and environmental variability should cover a broad range.

Since resilience encompasses the capacity to absorb perturbation while maintaining function without collapse (Carpenter et al. 2001; Gunderson and Holling 2002) we collected data at the onset of a drought period.

4.2 Materials and methods

4.2.1 Study area

Location

We examine two types of management regime in a semi-arid savanna ecosystem in Northwestern Namibia under either private or communal land tenure (Fig. 4.2). The communal areas are situated in the emerging Orupupa Conservancy, in Namibia’s Kunene-Region. The conservancy is an institution of the “Community based natural resource management” programme in Namibia. Conservancies take responsibility for the natural resources, mainly wildlife, within their boundaries by monitoring, managing and conserving them (NACSO 2009). The emerging conservancy Orupupa covers 1650 km² (Faschina 2010) and two habitats. The north-eastern part is a plain with sandy soils dominated by grasslands. The other parts, as well as the commercial farms are

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plains interspersed with hills dominated by mopane savanna. The commercial farm area is situated about 90 km southeast of the conservancy on private land. Site conditions and vegetation are similar to the conservancy. Soils vary from deep sand to loamy clay (Eisold, unpublished data). Data were recorded at the end of the rainy season 2006/2007 from April to June.

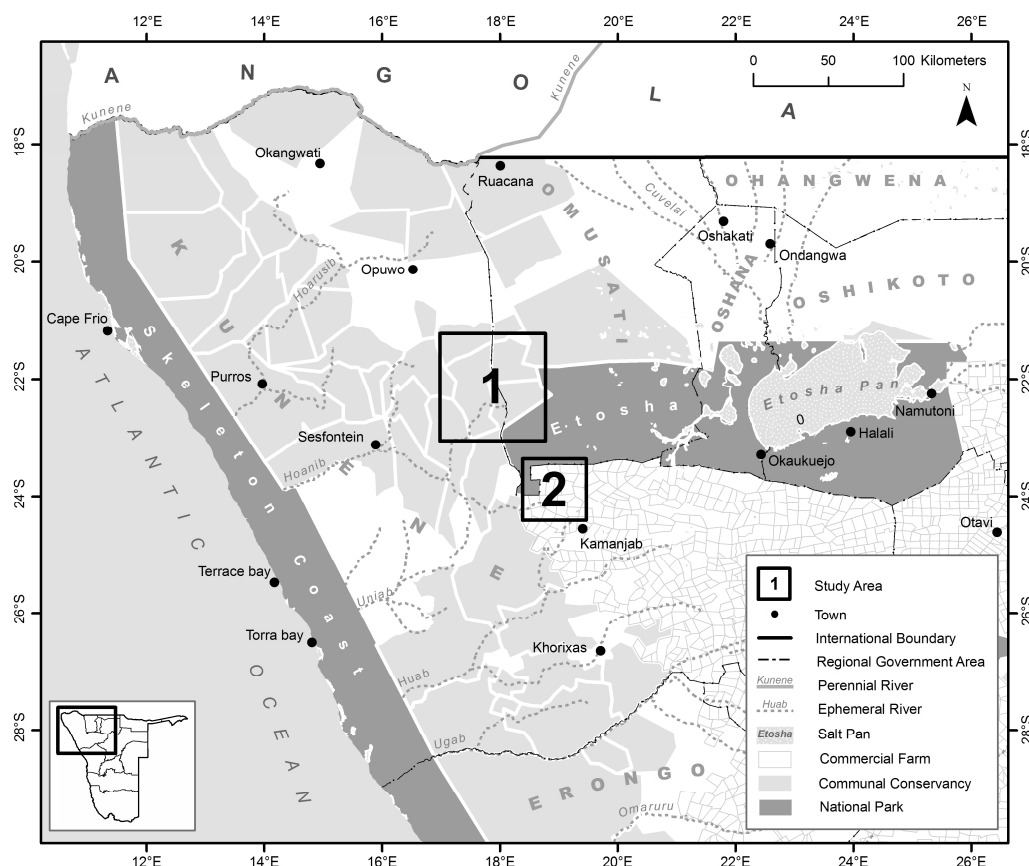


Fig. 4.2: Location of the study areas on communal land (1) and on commercial farms (2), situated in Northwestern Namibia. Source: ConInfo 2009; Cartography: Holger Vollbrecht.

Climate

Northern Namibia's climate is semi-arid, the area receives an average annual precipitation of 200 to 300 mm. Rainfall is highly variable in time and space with an interseasonal variability between 40 and 60% (Mendelsohn et al. 2003). Northern Namibia has two main seasons, the dry season that lasts from May to October and the wet season, from November to April. Annual potential evaporation is estimated to exceed the annual precipitation by a factor of about five (Mendelsohn et al. 2003).

The rainy season 2006/2007 received below average rainfall and was characterised by the local land users as a “bad rain year”. The Kunene region received an average of 183.67 mm in 2007, while in 2006 it received 460.5 mm/year.¹² Local land users reported low rainfall in the north-eastern part of the conservancy, in the sandy plains. In the commercial farm area, 144 mm of precipitation were recorded in the central part of this area between October 18, 2006 and April 4, 2007. This is about 50% below the long-term mean.

Population and land tenure

Pastoralism, both communal and commercial forms of livestock farming, is one of the main sources for livelihood in Namibia (Mendelsohn et al. 2003). The northern part of Namibia is under communal land tenure, fencing is not allowed and livestock can roam freely (Leggett et al. 2003). The commercial farms belong to private farmers. In Namibia’s Kunene Region commercial farms had an average size of 97.3 ha in 2007,¹³ and are divided into camps, where rotational grazing is practised since the 1950ies.

4.2.2 Survey design

Grazing gradient

We selected two study sites which covered a broad range of grazing impact, as well as the two most important types of range management found in semi-arid rangelands (i.e. communal and commercial livestock farming). To evaluate the impact of grazing on the rangeland system, three types of indicators (direct, abiotic and biotic, see Table 4.1) were sampled along piospheres. Because potential correlations between the grazing gradient and other environmental factors often create confounding effects due to sampling across significantly different vegetation types and soil textures (Landsberg et al. 2003; Todd 2006), we selected piospheres where these effects appeared to be smallest. Specifically we used the distance from a permanent water source (waterpoint) to represent grazing intensity, as artificially created waterpoints are frequently not associated with or confounded by other environmental factors (Landsberg et al. 2003).

¹² http://www.phlosses.net/index.php?form=rainfall&tc_id=14

¹³ http://www.phlosses.net/index.php?form=average_farm_size&tc_id=14

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The ecological sites were not selected to be statistical replicates of grazing impact within each management area; rather, each site was intended to provide an independent test for the existence of ecological thresholds along a grazing gradient (Sasaki et al. 2008).

In each grazing area the direction of grazing was identified via expert interviews with the herders and by participant observation. From each waterpoint cattle moved in one general direction for most of the year. Due to this fact, the relatively small size of grazing areas and because of the heterogeneity of the habitat parameters, (only) one transect was laid out from each waterpoint and sampling distance varied along individual transects. Each grazing gradient stretching away from a waterpoint was sampled with one 500 to 7000 m long transect. We aimed at sampling the transects at distances of 100 m, 500 m, 1000 m, 2000 m, 3000 m etc. up to 5500 m. A 20 x 50 m (1000 m²) plot was selected at each distance, respectively. Because of the spatially heterogeneous vegetation in these ecosystems, one relatively large plot seemed more appropriate than several smaller plots. As our study – like the recent study by Sasaki et al. (2008) – uses biotic indicators on the level of species to identify potential thresholds, we were not interested in vegetation patches such as islands of shrubs or tree islands, which would have required smaller plots.

Altogether 10 transects were sampled with a total of 63 plots. Four transects were situated in the communal area, four on cattle farms and two on game farms. Plots were selected based on minimizing geographic variability such as slope and aspect.

The proportion of bare ground is often applied in empirical rangeland studies as a direct indicator for herbivore impact along grazing gradients (Smet and Ward 2006). As (indirect) abiotic indicators, we analysed chemical soil properties known to respond to herbivore density (Derner et al. 1997), i.e. organic carbon and total nitrogen content of the top soil. Representing biotic parameters we analysed vegetation cover, subdivided into plant functional types (PFT's). The species recorded in the study were classified into one of the following PFT's: woody species (wood. sp.) (trees and woody shrubs), perennial grasses (per. gra.) and annual grasses (ann. gra.). In pre-studies these groups were found to be the most suitable as an approach to a classification into PFT's (Eisold, unpublished data). For each of the 10 areas sampled, mean cover of all species and within each life form was calculated.

Table 4.2: Characteristics of the transects in the communal and commercial grazing system.

Landform	Land use and range of grazing intensity	<i>n</i>	Range of altitude [m.a.s.l.]
Undulating sandy plains interspersed with low hills	Communal, heavily grazed to dry season pasture	28 plots (four transects)	1016 to 1345
Undulating sandy plains interspersed with low hills	Commercial game farm	13 plots (two transects)	1282 to 1294
Undulating sandy plains interspersed with low hills	Commercial cattle farm	22 plots (four transects)	1270 to 1300

Soil sampling

From each plot a bulk topsoil sample (0–5 cm) was taken from five randomly selected spots. Samples were analysed at the soil Laboratory of the Institute for Geography, University of Osnabrück for texture, and the University of Dresden for organic Carbon (C_{org}) and total Nitrogen (N_{tot}) content. One communal transect and two transects on cattle farms were analysed by the agricultural soil laboratory in Windhoek in an identical manner.

To determine the soil texture, the sand fraction (2–0.063 mm) was sieved, silt and clay contents (0.063–0 mm) were determined by pipette analyses (Schlichting et al. 1995). Total soil nitrogen content of the top soil was measured using Dumas' total combustion method (Duma 1831; Bremner 1996) with a Leco nitrogen analyser. Organic carbon content of the top soil was determined using the Walkley-Black method (Walkley 1947; Nelson and Sommers 1996).

4.2.3 Statistical analysis

We applied a stepwise approach for data exploration (Zuur et al. 2009). We used scatterplots and multivariate analyses to test for outliers and for collinearity among variables. Scatterplots were also applied to evaluate the strength of correlations between distance to waterpoint and abiotic and biotic response variables, and to visually estimate discontinuities across gradients (Bestelmayer et al. 2009) to detect threshold response. We applied Spearman's Rank correlation to evaluate the strength of the relationship between cover of PFT's and abiotic variables in species groups with distance to waterpoints. Regres-

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sion analysis was then used to investigate the relationship of these grazing response variables to distance from water.

Differences between response groups were examined with the Wilcoxon signed-rank test to avoid the assumption of normality (Conover 1980). This test assessed differences between grazing regimes by ranking the relative composition of abiotic and biotic parameters among treatments.

4.3 Results

Soil texture, slope and aspect were held to a minimum along the grazing gradients to provide comparable ecological sites. The only significant correlation of soil texture along the gradient was found in the communal areas, where clay content correlated with distance ($R = -0.38$; $p < 0.05$; $n = 9$).

4.3.1 Impact of grazing intensity and management strategy

Overall the indicators showed different responses along the grazing gradients and between management strategies. Bare ground cover and woody species cover showed no response in relation to grazing. Only organic carbon content and perennial grass cover showed responses in two out of three management strategies (Table 4.3).

Table 4.3: Correlation coefficients (R) for distance to waterpoint and potential indicators. (n.s. = not significant)

	R ($p < 0.05$)		
	Communal area	Farm cattle	Farm game
Bare ground	n.s.	n.s.	n.s.
C_{org}	- 0.6	n.s.	n.s.
N_{tot}	- 0.4	- 0.52	n.s.
Annual grasses	0.5	n.s.	0.62
Perennial grasses	n.s.	0.62	0.66
Woody species	n.s.	n.s.	n.s.

Direct indicators – recent grazing

Between the two management strategies, bare ground cover differed significantly (Wilcoxon signed-rank test; $p < 0.001$; $n = 24$). Overall, bare ground cover was higher in the communal areas than in the farm areas (Fig. 4.5).

Abiotic indicators – Soil chemical indicators

Organic carbon content of the topsoil decreased along the grazing gradient in the communal areas ($r^2 = 0.04$; $p < 0.05$). Total nitrogen content decreased with distance to waterpoint in the communal areas and in the cattle farms ($r^2 = 0.27$; $p < 0.05$). In all cases a certain value, 0.2% of organic carbon content and 0.02% of total nitrogen content was not exceeded (Fig. 4.3). The variation within the data set is high, standard deviation being nearly half of the mean value in all cases (Fig. 4.5). The depletion of organic carbon content does not go beyond 0.18% of total carbon content (Fig. 4.3).

Between the communal areas and the game farms organic carbon (Wilcoxon signed-rank test; $p < 0.03$; $n = 11$) and total nitrogen content of the topsoil (Wilcoxon signed-rank test; $p < 0.005$; $n = 13$) differed significantly. Overall, no clear trend could be detected. Between the communal areas and the game farms organic carbon content of the topsoil differed significantly (Wilcoxon signed-rank test; $p < 0.03$; $n = 11$).

4. Grazing impact in space

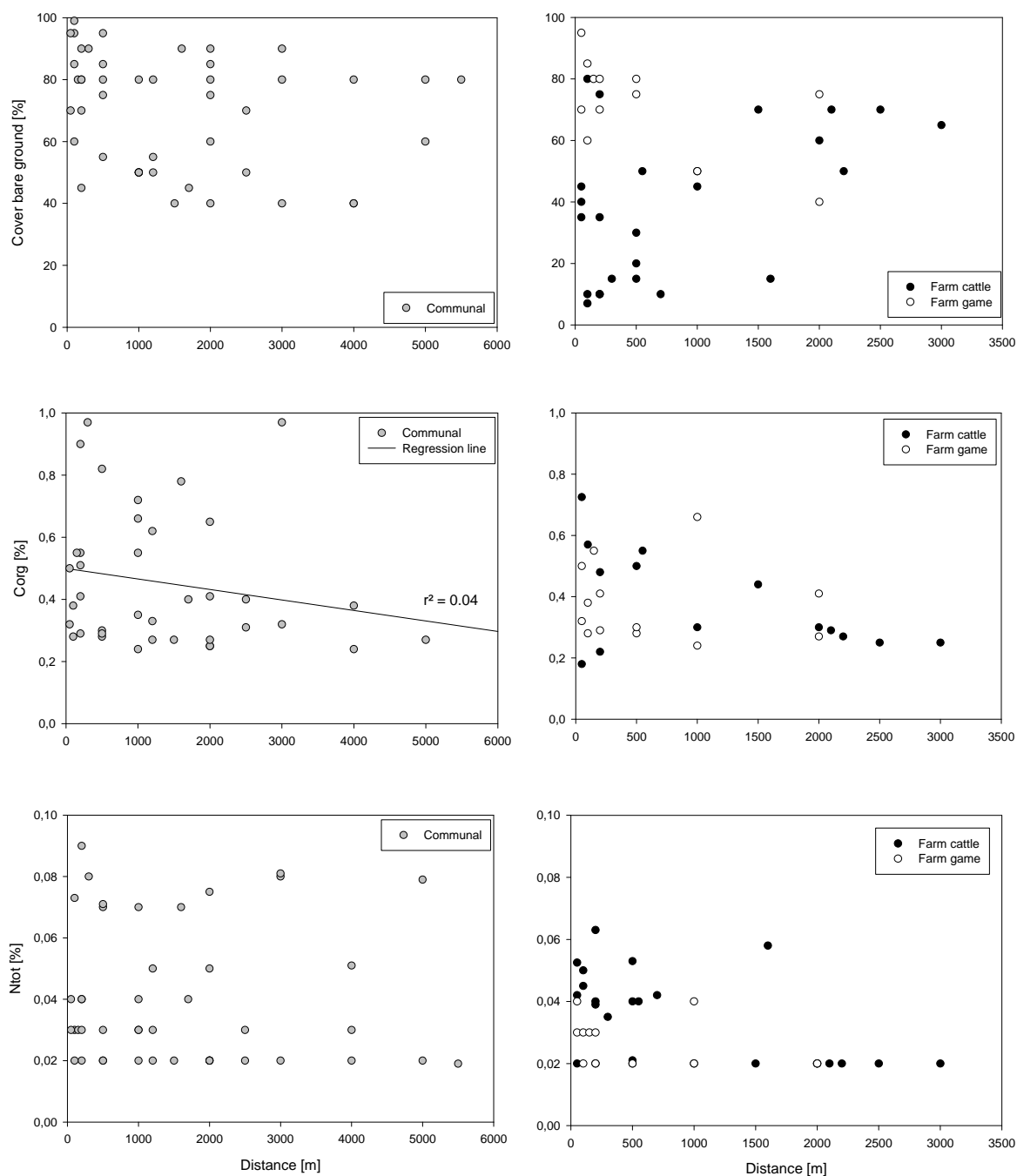


Fig. 4.3: Cover of bare ground, total organic carbon and nitrogen content of the topsoil within three grazing regimes imposed by different management strategies. Simple linear correlations between distance to waterpoint and abiotic indicator (solid lines) represent the spatial trend in each grazing regime.

Biotic indicators

Perennial grass cover decreases with grazing intensity in the cattle farm ($r^2 = 0.14$; $p < 0.05$) and game farm ($r^2 = 0.34$; $p < 0.05$) areas. Annual grass cover responded to grazing in the communal areas ($r^2 = 0.04$; $p < 0.05$) and game

farms ($r^2 = 0.08$; $p < 0.05$) (Fig 4.4). Overall, perennial grass cover (Wilcoxon signed-rank test; $p < 0.00007$) and annual grass cover (Wilcoxon signed-rank test; $p < 0.0001$) were higher in the farm areas than in the communal areas.

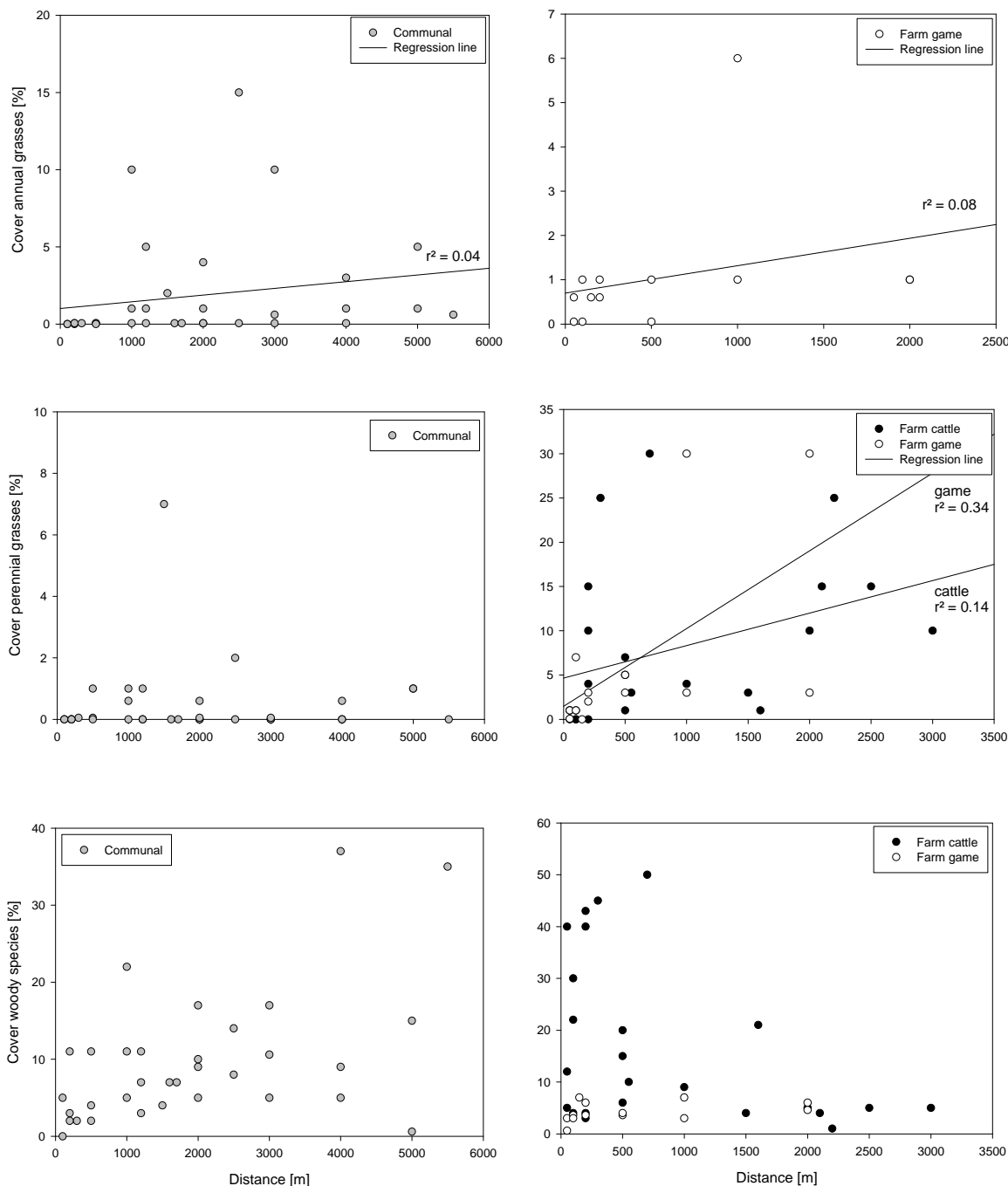


Fig. 4.4: Cover of perennial and annual grasses and woody species within three grazing regimes imposed by different management strategies. Simple linear correlations between distance to waterpoint and life form cover (solid lines) represent the spatial trend in each grazing regime.

4. Grazing impact in space

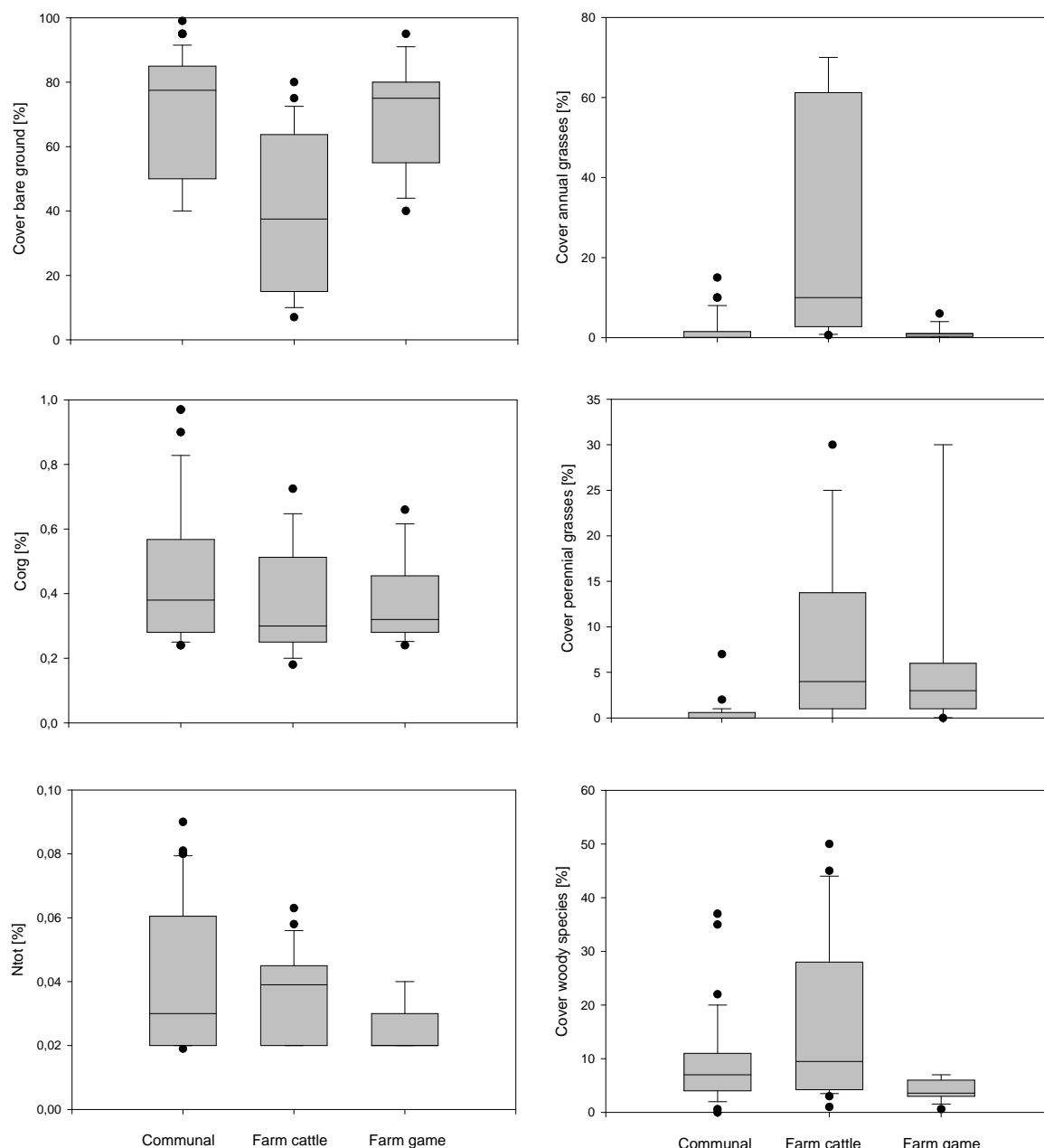


Fig. 4.5: Cover of perennial and annual grasses and woody species, bare ground and total organic carbon and nitrogen content of the topsoil within three grazing regimes imposed by different management strategies. Boxes show medians and 25th to 75th percentiles, whiskers stand for the non-outlier ranges of the data.

4.4 Discussion

In this study we aimed at understanding the response of various rangeland indicators to different levels of grazing impact, and at disentangling these effects from environmental variability. We further asked whether the frequently applied space-for-time approach is suited to detect threshold dynamics and potential early-warning signs of degradation in a semi-arid rangeland (Sasaki

et al. 2008). These questions are crucial for the management of arid and semi-arid ecosystems for both biological conservation and sustainable land use (Hoshino et al. 2009). Our methodological approach was on examining a number of grazing gradients spread across different grazing areas under three different management regimes, rather than providing descriptions of localised response (Landsberg et al. 2003).

4.4.1 The indicators

Direct indicators

We analysed bare ground cover to assess physical presence of livestock as a direct measure of grazing impact. In our study bare ground showed no clear response along grazing gradients, neither a linear response (correlation analyses, linear regression) nor a threshold behaviour (visual estimation of discontinuities) and thus seems not to be a suitable indicator.

Between the management strategies bare ground differed significantly, indicating that trampling and grazing might have a stronger negative effect in the communal areas than on cattle farms. This could be due to a overall lower stocking density and to the rotational grazing system on the farms, where the fenced camps around the waterpoints are rested in regular intervals (Smet and Ward 2006) of several weeks.

The occurrence of bare ground can be interpreted as cattle's grazing behaviour following the supply of forage. This means that bare ground indicates a reaction of grazing patterns to changing pasture quality or condition. These indicators show a short-term response of livestock to resource supply, while functional interpretations concerning, e.g. degradation are difficult to assess. We may thus conclude that direct indicators do not necessarily indicate changes in the systems, as they can merely follow changes in the system.

Although an at-risk community phase, which is most vulnerable to exceeding state resilience can be indicated by large interconnected areas of bare ground in response to grazing and drought, due to higher vulnerability to soil erosion, leading, e.g. to nutrient losses, (Bestelmayer et al. 2009), and Sasaki et al. (2008) state that "in future studies, the preventive threshold should be quantified using more direct indicators (...)", this seems not appropriate in our case study in Northwestern Namibia.

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Abiotic indicators

In our study a significant response of abiotic indicators along the grazing gradients can only be found with organic carbon in the communal areas ($r^2 = 0.04$; $p < 0.05$). While trends can be visually estimated from the scatterplots (Fig. 4.3) in both management regimes. Organic carbon and total nitrogen content of the topsoil does not show a response in its minimum level, thus no potential threshold is indicated for low ranges of organic carbon and total nitrogen. The upper boundary of the data clouds indicates a decrease with distance to the waterpoint. This can be due to an accumulation of nutrients through dung close to waterpoints. Empirical rangeland studies show a redistribution of nutrient and an accumulation due to dung accumulation in close vicinity to waterpoints, because especially cattle tend to rest there after drinking. Vice versa a depletion of organic carbon can be detected further away. In most studies organic carbon increases with decreasing distance to the waterpoint (Smet and Ward 2006; Derner et al. 1997).

Biotic Indicators

In a drought year, cover values of plant functional types were generally low. Annual grass cover increases with distance from waterpoint in communal areas ($r^2 = 0.04$; $p < 0.05$) and game farms ($r^2 = 0.08$; $p < 0.05$), while on cattle farms no trend can be identified. Perennial grass cover responded along grazing gradients on cattle farms ($r^2 = 0.14$; $p < 0.05$) and game farms ($r^2 = 0.34$; $p < 0.05$), while in the communal areas perennial grasses did not occur in most plots.

We conclude that perennial grasses tend to become eliminated from the system, e.g. in cases of overutilization via high grazing pressure and drought, and may thus have no indicative value along the spatial gradient in cases where both grazing pressure and resource stress are high. Annuals may be a good indicator if herbaceous perennials fail in cases of high grazing pressure and high resource stress. However, along spatial gradients they may be less valuable because of their direct response to rainfall. Particularly in years with very low or no rainfall they might totally lose their indicative value.

Visual estimation shows that woody species tend to increase with distance to waterpoint in the communal area (Fig. 4.4). This can be explained by the collection of fire wood in the vicinity of settlements in the communal areas, which usually include the waterpoint. This process cannot be found in the

farm areas. Furthermore, on cattle farms a low density of browsers is present, while in communal areas usually livestock also comprises goats which are browsing herbivores (Skarpe et al. 2007).

4.4.2 Indicators and their indicative value – in response to grazing

The exact value of the indicator depends on the objectives of the manager. On game farms woody species are especially needed for browsers. While on commercial cattle farms the threat of bush encroachment exists. Communal farmers need woody species for daily life and value them as reserve biomass for very scarce times (Eisold et al. in review; Kemmerling et al. resubmitted). Bestelmayer et al. (2009) give 10% as a critical level of perennial grass cover. In our study in the communal areas perennial grasses have surpassed the critical level but few annual grasses and woody species palatable biomass are still available. If the cattle are kept for commercial purposes they need a relatively higher amount of palatable biomass in comparison to subsistence – farming. Thus a state dominated by woody species would not be sufficient for meat production but on the game farms a dominance of woody species would be desired if mainly browsers are kept.

Since vegetation attributes respond to both grazing and environmental factors (Fuhlendorf et al. 2001) a spectrum of vegetation attributes provides a thorough interpretation of vegetation dynamics and thus for monitoring approaches (Fernandez-Gimenez and Allen Diaz 1999). Since the cover of annual grasses can be highly variable in rangelands the ratio between annual and perennial grasses seems to be a good indicator for grazing impact and the suitability for grazing of an area. In times of scarce natural resource availability, such as drought, forage supply would then have to fall back on the palatable biomass of woody species, which is possible but an emergency strategy. Abiotic indicators should not undergo a certain threshold, but seem more or less unaffected by grazing.

4.4.3 Methodological implications

Transects do not offer an appropriate tool for disentangling environmental variability and non-linear effects of grazing impact at the applied spatial scale and resolution in a highly variable environment. We conclude that the space-for-time approach (Pickett 1989) is not sufficient for threshold detection in

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ecosystems that hold both a high spatial and temporal variability. It might be generally inappropriate, as spatial vegetation patterns are not necessarily analogous to temporal patterns. For example, a detection of threshold response along climatic gradients requires a detailed observation of patchiness, as spatial autocorrelation might be an early-warning indicator for systems approaching a threshold (Scheffer et al. 2009).

Patterns of ecological dynamics emerge at various spatial and temporal scales due to site conditions, ecological processes and units. To assess the impact of grazing on ecological dynamics, to disentangle it from environmental variability and to identify the emerging patterns it is important to choose the appropriate scale and resolution for analysis. Spatial patterns emerge at a certain scale, below this they can be perceived as a diffuse noise and above it can merely be the environmental background (Jentsch et al. 2002). In highly variable systems the scale of analyses becomes increasingly important to identify spatial patterns, due to the big range in the data set and many feedback processes. These effects of landscape heterogeneity are usually overcome by focusing on individual sites (Briske et al. 2005).

In at-risk community phases, approaching a threshold to another state, perennial grass cover is low, annuals dominate (Briske et al. 2008) and woody species' palatable biomass is available for scarce times to circumvent a total breakdown of animal numbers. Vegetation patterns are supposed to be patchy, with large interconnected areas of bare ground in response to intensive grazing and drought, indicating that rather patchiness is a potential indicator for a state approaching a threshold. Thus critical transitions may be indicated by particular spatial patterns (Scheffer et al. 2009).

This pattern was observed during data collection but would need a more specified collection method and analyses in future studies. The spatial distribution of vegetation represents an important structural threshold component because it influences the potential for erosion and resource retention on ecological sites (Ludwig et al. 2000; Briske et al. 2005). Vegetation patterns can be analysed via smaller but a higher number of plots (relatively to the approach of this study), or mapped so that the vegetation patterns can be evaluated e.g. via spatial statistics (e.g. Ripley's K-Function), similarity indices or GIS analyses (Juraskinski et al. submitted; Hassler et al. resubmitted).

It seems not appropriate to evaluate threshold dynamics with traditional linear analyses along spatial gradients of land use due to their emphasis on mean response (Bestelmayer et al. 2009). Different processes are active on different levels, reflected by different responses along the gradient of the lower range of the data points and the maximum range of the data points. For example in the soil, depletion and accumulation processes are possible. In this highly variable system a differentiation between depletion and accumulation processes is difficult to show with median and linear analyses. Alternatively, linear piecewise quantile regression (Cade and Noon 2003) offer the ability to examine abiotic indicators (Bestelmayer et al. 2009). Through the analysis of percentiles, indicating the highest and lowest level of values, depletion and accumulation processes can be characterised at other parts of the response distribution, the extreme quantiles. At the same time the range, i.e. the spatial dimension of these processes can also be assessed. However, even with such statistical tools, transects are still limited in their value for detecting thresholds due to the importance of spatial patterns (patchiness; see above).

4.4.4 Implications for monitoring – Key elements of state-and-transition models

According to the resilience based concepts, the reference state of a savanna rangeland is characterized by a high perennial grass cover, a low cover of woody species, and minimum of bare soil (Briske et al. 2008). In our study such a state could only be found in a cattle farm area, identified by the commercial farmer as his desired state of land use (pers. comm.). This state developed due to a temporal resting from cattle grazing over a period of several years, due to the areas relative distance to the waterpoint. In the reference state, as a feedback mechanism, perennial grass cover minimizes soil and nutrient loss and water and soil movement to decrease soil erosion. At-risk community phases, where herbaceous plant cover is low could be found in the grazing areas close to waterpoints where grazing pressure is high.

Triggers would be intensive grazing and/or drought that predispose the sites to soil erosion processes. A threshold is reached if through soil erosion processes, water, soil and nutrients are channelled away from grasses which initiates greater soil erosion and leads to additional grass loss. If this threshold is passed an alternative state is reached. For this state indicators would be major soil and water movement and woody species dominance, low annual grass cover. In this state feedback are few or no perennial grasses and continued

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water, soil and nutrient losses with subsequent rain storms that lead to additional (annual) grass and further vegetation loss.

No clear thresholds were found along spatial gradients of grazing, but shifts between different plant communities can be detected (Briske et al. 2009). Due to their different memory levels and sensitivity to grazing impact, indicators could be found for different processes. In our study especially biotic indicators respond to the different grazing impact between management strategies, and can be assigned to the different memory levels (Table 4.5). Since degradation processes of the vegetation are generally characterized by changes in plant functional types, such as an increase in the abundance of annual species with a concomitant decrease in perennial species (McIntyre and Lavorel 2001; Reynolds and Stafford 2002; Pakeman 2004; Diaz et al. 2007; Hoshino et al. 2009), this shift in floristic composition/species dominance could indicate the continuous and reversible shift from one plant community to another, occurring within a stable state. Perennial grass cover is supposed to show a medium response to climatic variability and short term response to grazing impact. It can be classified as a relatively robust indicator for a desirable state of the grazing area (Aucamp et al. 1992) and is widely applied as an indicator for rangeland condition (Schulte 2002). A significant decrease of perennial grasses indicates a transition among plant communities within a state (Bestelmayer et al. 2009), but not a threshold yet. With a shift from perennials to annuals, forage supply is still given, but the supply is not as reliable as before, with the dominance of perennial grasses because the response to climatic variability of annuals is higher. A decrease of perennial grasses could thus function as an early warning indication, approaching a threshold to leave the current state.

While the further shift from annual grassland to woodland could be a long-term indicator, a threshold component marking the discontinuous and irreversible replacement of a stable state (Briske et al. 2005). Woody species are relatively robust in their response and show a long term memory for grazing. Thus they can indicate long term changes in rangelands (Sasaki et al. 2008). A response could be indicative for a degradation threshold because of the threat of a depletion of reserves and also bush encroachment. Woody species can act as forage supply reserves (Eisold et al. in review; Kemmerling et al. resubmitted) in small or spatially constrained grazing areas. If this last reserve is depleted, a threshold is overcome because no natural forage supply is available.

We identified indicators for changes in a rangeland ecosystem on different levels. Long-term indicators were identified as the structural shift from grassland to woodland indicating a major shift in the supply with grazing resources, that indicate major ecological restructuring on a functional level symptomatic of land degradation (Reynolds and Stafford 2002; Scheffer et al. 2009). Furthermore early warning indicators, the ratio between annual and perennial grasses were characterised.

We also conclude that the space-for-time approach, widely applied in range ecology in gradient analyses along transects, is not suitable for an application to the threshold concept.

With this basis restoration pathways, for the re-establishment of pre-threshold states following active restoration of autogenic repair mechanisms (Briske et al. 2009) can be identified. This is important because long term changes alter the resource base of the entire ecosystem such that it may move beyond a threshold whereby degradation accelerates and may become irreversible.

5 Synthesis

It seems that the degradation of natural resources is a serious challenge to Northwestern Namibia. Maladapted land use being one of the drivers. However, local land users can still adapt to their ever changing environment. With an integrated analysis of the interactions in a social-ecological system from an ecological view point, I have analysed important aspects of resilience-building mechanisms, especially in the case of drought as perturbation.

In summary the study answered the following questions:

i) Which ecological elements of the ecosystem are *perceived* as important by the local land users in regard to land use decisions? I could demonstrate that land users perceive resources as important that bring reliability in the context of rangeland use decisions.

ii) What influences *decision-making processes* of local land users in regard to mobility patterns? Water and ecological buffer mechanisms were identified as crucial elements, expressed in a trilemma of choices. This may be important in understanding the household and landscape factors underlying observed distributions of grazing intensity.

iii) What is the *impact* of livestock grazing and land use strategy on vegetation dynamics and – structure, particularly on the presence/quality of ecological buffers?

The results indicate a change in vegetation structure that are likely to be related to the accumulated long-term impacts of grazing. They are also likely to be the long-term detriment of the local vegetation (Landsberg et al. 2003, Klintonberg et al. 2007).

iv) What are suitable and adequate *indicators* for range assessment? Plant functional types provide suitable indicators for shifts and thresholds in vegetation dynamics. The ratio between annual and perennial grasses are useful early warning indicators for shifts within a stable state. Woody species provide an early warning indication for an ecological threshold crossing into an alternative state.

One impulse of my study is the importance to identify and further work on those elements of the system that are also locally perceived as salient. This could be ecological items, such as plant species in their function as forage resources. For example, woody species are mostly neglected in ecological studies. However, I could demonstrate that they are perceived as reliable and serve as a buffer in the system, though their abundance is rather low. I developed a method to integrate the local and the scientific view on ecological items to identify what are local criteria for the valuation of forage resources.

Reliability, which is perceived as an important criterion for the quality of forage resources, has a lot in common with the concept of key resources, which are defined as forage resources available in times of resource scarcity. While other authors have a descriptive approach to the reliability of key resources, I functionally defined them as biotic and abiotic buffers.

Furthermore, I found a functional understanding of how rainfall variability is buffered through vegetation traits, by biotic buffers; and through site characteristics and the spatial re-distribution of rainfall (abiotic buffers).

In contrast to a common view in rangeland science decision-making of local pastoralists does often not reflect the availability of key forage resources but of the second essential resource for livestock, such as drinking water. The local range use strategy may not be adaptive, but seems to be the only possible adaptation of local users to the current ecological and socio-economic situation because options for action are restricted.

5.1 Outlook on the socio-economic environment

The consequence of increased land pressure is not only increased persons/land ratios, reduced resting periods of grazing areas and land degradation, but also pressure on the laws and customs, which have in the past assured the sustainability of the rangeland use (Ostrom et al. 1999). Rangelands have experienced many ecological, social and institutional changes and effects (Bollig 2005; Faschina 2010). For instance, Faschina (2010) notes that the economic and social conditions of the Ovaheroo living in the communal areas of Northwestern Namibia have changed throughout their history in response to many factors operating over a variety of spatial and temporal scales, such as the coexistence of traditional and modern local institutions.

5. Synthesis

Different forms of land tenure have important implications for the management of grazing resources and their preservation in the longer term, especially in communal areas. A common property regime, on the one hand consists of a well-defined group of authorized users, a well-defined resource that the group manages, and a set of institutional arrangements that define both of these. There are also rules of use for the resource in question. Conversely, in open-access situations users have privilege with respect to the use of the resource as nobody has the legal right to exclude them (Bromley 1989). However, they have no actual rights to the resource (Bromley 1989). With open access, grazing management decisions are essentially taken on an individual or 'clique' basis with the sole intention of maximizing benefit to the individual and there is little or no incentive to manage the resource productively and sustainably in the long term (Berkes 1989). In a common property regime the resource is managed on a consensus basis to the mutual benefit of the community and there is, therefore, an incentive to ensure its productivity in the long term.

According to Hardin's "tragedy of the commons" (1968) degradation occurs because in communal land use there are multiple managers and it is more profitable for the individual to overstock because he derives the entire benefit but the costs are shared by all. But the reality can be quite different and the inevitability of resource destruction in open access has been heavily contested since 1968. As Ostrom et al. noted in their review of 1999, "although tragedies have undoubtedly occurred, it is also obvious that for thousands of years people have self-organised to manage common pool resources, and users often do devise long-term sustainable institutions for governing these resources". Hardin (1998), commenting later on his 1968 paper and the criticisms he had received, admitted that "repeatedly I found fault with my own conclusions. The weightiest mistake was the omission of the modifying adjective 'unmanaged', adding that 'with an unmanaged commons ruin is inevitable'".

Livestock owners can be bound to unwritten rules that govern the use of the communal rangeland (Tapson 1993; Everson and Hatch 1999; Smet and Ward 2005). These shared norms have resulted in a pastoral system that through ownership of livestock, sharing of livestock products and the recognition of livestock keeping as a way of life, is a unifying feature of the southern African communal communities. Despite of a long history of restriction to small areas of land and the imposition of colonial and apartheid policies, this pastoral system has persisted, suggesting that informal institutions have been effective in

ensuring adherence to norms, and that tacit knowledge supporting herders' practices are effective in this environment. In future studies the following aspects should be analysed and discussed: The local coping mechanisms may be failing or becoming less viable due to rangeland degradation, droughts and lack of supported and locally adapted politically motivated changes (Faschina 2010; Kassahun et al. 2008). These processes of change also require the adoption of the institutional frame work and socio-economic options. Rational decisions are restricted by various factors, such as norms and affects (Esser 1996; Moran 2006). The institutional framework of the area might not be sufficient to guide today's grazing patterns because with the growing land scarcity and degradation of the vegetation cover they are not able to account for an accumulation of reserve biomass and a reduction of herd mortality. Thus, it must be asked if the local range use system is still able to cope with changing conditions.

5.2 Implication for management and monitoring

As well as extending the previous literature by a new context, and providing useful information to guide planning decisions for sustainable natural resource use, this approach is a methodological advance which provides further insights into local decision-making processes. With the integrated approach of this study local knowledge and local action could be matched providing an important base for monitoring schemes. Altogether, the spatial distribution of pastoral grazing as it is important to provide a quantitative foundation for ecological monitoring, is a critical element of natural resource management and conservation projects, aimed at engaging local people (Kremen et al. 1994).

I conclude that waterpoints are a primary determinant of the vegetation structure and composition of communal Northwestern Namibia. Thus areas with a high grazing impact tend to be overutilized and associated with a reduction in herbaceous plant cover. The relative low herbaceous vegetation cover in heavily used rangelands is of concern, because it is not buffered by reserve biomass, accessible in scarce times, especially since most rangeland areas are now grazed. From a rangeland management perspective, my study confirms that those areas remote from waterpoints are valuable buffers for rainfall variability (Landsberg et al. 2003), thus acting as key resources. The most grazing-sensitive species are at most risk. Areas distant from waterpoints that provide a refuge for grazing-sensitive species that might otherwise be lost from the

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rangeland are very scarce. My results indicate that those may be perennial grass species. For example, in the grazed areas cover of perennial grasses found was 0.11 ± 0.33 per cent, while in the ungrazed areas it covered up to 2.4 ± 0.89 per cent.

Since water availability was identified as a key driver in mobility decisions which is even more important than biomass availability as a driver, the provision of further permanent waterpoints might seem a way in the future but is also very challenging. This study provides evidence that to expand the availability of permanent waterpoints is likely to result in a further degradation of the natural resources due to the high grazing impact around these waterpoints (Kabubo-Mariara 2005; Barbier et al. 2008).

If future management options do not take the integrated management of natural resources into account, there would be very few areas of productive rangeland remaining sufficiently far from water to provide potential reserves for biomass and for perennial herbaceous vegetation. Thus indicating a potential degradation threshold. The very real worry remains that if the trend of borehole intensification continues at its recent pace (Tsimako 1991; White 1993) the relative importance of degradation processes, e.g. mirrored in sacrifice zones¹⁴, will also increase. In the grazed areas the sacrifice zone, where hardly any herbaceous vegetation cover was found extended out to 2000 meters. And herbaceous reserve biomass was only found in a distance as far as out to 5500 meters. While results are variable for every case study, this study nonetheless gives some indication of the scale of the potential threat, if the development of further waterpoints proceeds without specific environmental impact assessments and integrated management approaches, that include specific measures to safeguard ecosystem service provision. A promising approach is integrated water resource management, an holistic approach integrating the management of land and water resources, as well as participation of all stakeholders in regard to the management, such as the local land user and administrative (Kluge and Moser 2008).

For further management implications it can be concluded that the depredations of drought may be limited by access to more land (Samuels et al. 2007). My analysis of the impact of rangeland use on vegetation dynamics suggests

¹⁴ A belt of land devoid of vegetation in the dry season develops around waterpoints and is called the sacrifice zone (e.g. White 1993).

that resting for biomass accumulation and conservation of grazing sensitive species is crucial for the preservation of ecological buffers, as key resources. Constraints include the requirement for more land, the need to facilitate movements of herds over any great distances (transport facilities at lower costs, etc.). But still more specific measures should be carried out to assess its impact in different situations and to improve the efficiency of seasonal movements to guarantee the long-term sustainable use of natural resources (Simons and Allsopp 2007). However, selective resting of waterpoints offers prospects as a strategic tool for improving rangeland condition.

Understanding the nature and drivers of decision-making by land users whose livelihoods are dependent to a great extent on natural resources is one key requirement in the implementation of appropriate adaptation options for vulnerable communities and for the management of ecosystem services (Schlüter and Pahl-Wostl 2007; Kemmerling et al. resubmitted; Thornton et al. 2007).

According to Coughenour (1991) regarding spatial components of plant-herbivore interactions in pastoral ecosystems, especially traditional pastoral movement patterns, different dispersal-convergence movement patterns arise in response to the spatial dispersions of the most limiting resources. It is important to consider the spatial distribution of non-forage resources (such as water) within a landscape and their impact on spatial and temporal herbivory patterns; and develop options for redistributing livestock over larger spatial scales in response to changing patterns of rainfall and forage availability. In my case study, decision-making processes of a communal user group do not fit that because options for land use actions are limited. This means that adaptive aspects of the management are rather rare. Congruently the impact on grazing resources is very high, and even those elements that are regarded as reliable in the system are in an overall lower condition in comparison to the benchmark system. This heavy grazing causes significant changes in vegetation. An increasing population might use all land for grazing until there is no more land for reserves, causing declines in soil fertility and a change of vegetation patterns (Barbier et al. 2008).

Since the starting point for the analysis of social-ecological system functioning is the sustainable provision of ecosystem services and the maintenance of livelihoods, I am sure that this integrated analysis is important to conceptualize the effects of environmental variability, the supply of ecosystem services, and to connect this to management rules. This study focused on interactions

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between the ecological and cultural subsystem of a social-ecological system from an ecological point of view. Reliability of ecosystem service supply, key resources and ecological buffers are three important aspects of resilience mechanisms that I identified to functionally mean the same. Interaction effects are a defining feature of resilience and research connected to it. This fact makes an integrative and holistic approach in rangeland science even more necessary.

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Appendix

Table A2.1: Species list for correlation of LEK and scientific perception of plant forage species. Life form and salience indices are given. Ann. grass = Annual grass; Per. grass = Perennial grass.

Species	Family	Vernacular name	Life form	CSI		SI		FI	
				ecolog.	local	ecolog.	local	ecolog.	local
<i>Acacia erioloba</i> E. Mey.	Mimosaceae	Omumbonde	Tree	0,00	0,02	0,10	0,17	0,00	0,32
<i>Acacia hebeclada</i> DC.	Mimosaceae	Otjimbuku	Tree	0,00	0,01	0,00	0,08	0,00	0,26
<i>Acacia mellifera</i> (Vahl) Benth.	Mimosaceae	Omusaona	Tree	0,00	0,02	0,00	0,21	0,00	0,53
<i>Acacia nilotica</i> ssp. <i>kraussiana</i> (Benth.) Brenan	Mimosaceae	Orusu	Shrub	0,01	0,02	0,00	0,23	0,00	0,37
<i>Acacia reficiens</i> ssp. <i>reficiens</i> Wawra	Mimosaceae	Oungondo	Shrub	0,00	0,02	0,00	0,19	0,11	0,37
<i>Acacia senegal</i> (L.) Willd.	Mimosaceae	Omu-ryangava	Shrub	0,00	0,01	0,00	0,13	0,00	0,21
<i>Aristida adscensionis</i> L.	Poaceae	Ohoke	Ann. grass	0,06	0,01	0,35	0,15	0,78	0,26
<i>Berchemia discolor</i> (Klotzsch) Hemsl.	Rhamnaceae	Omuve	Tree	0,00	0,01	0	0,13	0,00	0,42
<i>Boscia albitrunca</i> (Burch.) Gilg & Benedict	Capparaceae	Omuntendeeti	Tree	0,03	0,04	0,03	0,28	0,33	0,95
<i>Boscia foetida</i> Schinz	Capparaceae	Otjinautoni	Shrub	0,09	0,05	0,41	0,37	0,67	0,47
<i>Catophractes alexandri</i> D.Don	Bignoniaceae	Omukaravize	Shrub	0,04	0,11	0,11	0,62	0,33	0,89
<i>Cleome foliosa</i> Hook.f.	Capparaceae	Ombowa	Herb	0,02	0,00	0,08	0,10	0,22	0,21
<i>Colophospermum mopane</i> (J.Kirk ex Benth.) J.Kirk ex J.Léonard	Caesalpinaceae	Omutati	Shrub	0,31	0,11	0,86	0,61	0,11	0,89
<i>Combretum apiculatum</i> ssp. <i>apiculatum</i> Sond.	Combretaceae	Omumbuti	Tree	0,00	0,10	0,00	0,52	0,00	0,68
<i>Combretum imberbe</i> Wawra	Combretaceae	Omumborombonga	Shrub	0,00	0,02	0,00	0,17	0,00	0,42
<i>Combretum wattii</i> Exell	Combretaceae	Omutapati	Tree	0,00	0,01	0,00	0,07	0,00	0,21
<i>Commiphora glandulosa</i>	Burseraceae	Omboo	Shrub	0,00	0,01	0,00	0,09	0,00	0,16
<i>Commiphora glaucescens</i> Engl.	Burseraceae	Omutungi	Shrub	0,00	0,02	0,00	0,22	0,00	0,47
<i>Commiphora mollis</i> (Oliv.) Engl.	Burseraceae	Omurenda	Shrub	0,00	0,02	0,00	0,19	0,00	0,32
<i>Commiphora multijuga</i> (Hiern) K.Schum.	Burseraceae	Omuzumba	Shrub	0,00	0,03	0,00	0,30	0,00	0,53
<i>Crotalaria</i> sp.	Fabaceae	Onduyaturawa	Herb	0,06	0,01	0,34	0,05	0,44	0,11
<i>Dichrostachys cinera</i> (L.) Wight & Arn.	Mimosaceae	Omutjete	Shrub	0,00	0,01	0,00	0,08	0,00	0,16
<i>Dicoma tomentosa</i> Cass.	Asteraceae	O-nyainya	Herb	0,02	0,00	0,48	0,00	0,89	0,00
<i>Ehretia rigida</i> (Thunb.) Druce	Boraginaceae	Omusepa	Per. grass	0	0,004	0,00	0,04	0,00	0,21
<i>Eragrostis annulata</i> Rendle ex Scott-Elliot	Poaceae	Ongwengwe	Ann. grass	0,06	0,00	0,41	0,00	0,67	0,00

Species	Family	Vernacular name	Life form	CSI		SI		FI	
				ecolog.	local	ecolog.	local	ecolog.	local
<i>Eragrostis dinteri</i> Stapf	Poaceae	Onyase	Per. grass	0,03	0,00	0,19	0	0,33	0,00
<i>Eragrostis nindensis</i> Ficalho & Hiern	Poaceae	Onyase	Per. grass	0,00	0,02	0,00	0,07	0,00	0,58
<i>Eragrostis porosa</i> Nees	Poaceae	Orueyo	Ann. grass	0,17	0,00	0,65	0,03	0,78	0,11
<i>Euphorbia</i> sp.	Euphorbiaceae		Herb	0,03	0,00	0,15	0,00	0,33	0,00
<i>Faidherbia albida</i> (Delile) A.Chev.	Fabaceae	Omue	Shrub	0,00	0,01	0,00	0,09	0,00	0,37
<i>Ficus sycomorus</i> L.	Moraceae	Omikuyu	Shrub	0,00	0,01	0,00	0,09	0,00	0,32
<i>Geigeria acaulis</i> Benth. & Hook.f. ex Oliv. & Hiern	Asteraceae	Okamuti kovipindo	Herb	0,03	0,00	0,13	0,00	0,44	0,00
<i>Geigeria alata</i> (DC.) Benth. & Hook.f. ex Oliv. & Hiern	Asteraceae		Herb	0,04	0,00	0,19	0	0,67	0,00
<i>Gisekia africana</i> (Lour.) Kuntze	Gisekiaceae		Herb	0,07	0,00	0,37	0	0,67	0,00
<i>Grewia bicolor</i> Juss.	Tiliaceae	Omuvapu	Shrub	0,00	0,03	0,00	0,29	0,00	0,47
<i>Grewia flava</i> DC.	Tiliaceae	Omundjembere	Shrub	0,00	0,01	0,00	0,17	0,00	0,32
<i>Grewia flavescens</i> Juss.	Tiliaceae	Omu-he	Shrub	0,00	0,01	0,00	0,11	0,00	0,21
<i>Grewia tenax</i> (Forssk) Fiori	Tiliaceae	Omundjendjere	Shrub	0,00	0,00	0,00	0,07	0,00	0,16
<i>Grewia villosa</i> Willd.	Tiliaceae	Omu-hamati	Shrub	0,00	0,01	0,00	0,12	0,11	0,32
<i>Hermannia tigrensensis</i> Hochst. ex A.Rich.	Sterculiaceae		Herb	0,02	0,00	0,12	0,00	0,33	0,00
<i>Indigofera</i> sp.	Fabaceae	Onyiva	Herb	0,15	0,00	0,28	0,02	1,00	0,05
<i>Kohautia</i> sp.	Rubiaceae		Herb	0,03	0,00	0,1	0,00	0,44	0,00
<i>Leucosphaera bainesii</i> (Hook. F.) Gilg	Amaranthaceae	Otjipembati	Shrub	0,03	0,02	0,15	0,24	0,33	0,37
<i>Limeum argute-carinatum</i> ssp. <i>argute-c.</i> Wawra & Peyr.	Molluginaceae		Herb	0,03	0,00	0,2	0,00	0,56	0,00
<i>Lonchocarpus nelsii</i> (Schinz) Heering & Grimme subsp. <i>nelsii</i>	Fabaceae	Omupanda	Tree	0,00	0,01	0,00	0,15	0,00	0,37
<i>Monelytrum luederitzianum</i> Hack.	Poaceae		Ann. grass	0,10	0,00	0,61	0,13	0,78	0,00
<i>Myrothamnus flabellifolius</i> Welw.	Myrothamnaceae	O-handukaze	Herb	0,00	0,01	0,00	0,13	0,00	0,32
<i>Nelsia quadrangula</i> (Engl.) Schinz	Amaranthaceae		Herb	0,07	0,00	0,42	0,00	0,78	0,00
<i>Pechuel-Loeschea leubnitziae</i> (Kuntze) O.Hoffm.	Asteraceae	Otjindumba	Shrub	0,00	0,01	0,00	0,09	0,00	0,16
<i>Phragmites mauritianus</i> Kunth	Poaceae		Per. grass	0,00	0,01	0,00	0,10	0,00	0,26
<i>Pogonarthria fleckii</i> (Hack.) Hack.	Poaceae		Ann. grass	0,04	0,00	0,32	0,00	0,67	0,00
<i>Ptaeroxylon obliquum</i> (Thunb.) Radlk.	Ptaeroxylaceae	Omumbungururu	Tree	0,00	0,01	0,00	0,07	0,00	0,21
<i>Rhigozum virgatum</i> Merxm. & A.Schreib.	Bignoniaceae	Omunditi	Shrub	0,01	0,02	0,04	0,17	0,22	0,26
<i>Salvadora persica</i> L.	Salvadoraceae	Omungambu	Shrub	0,00	0,02	0,00	0,16	0,00	0,26
<i>Schmidtia kalahariensis</i> Stent	Poaceae	Ongorondji	Ann. grass	0,25	0,06	0,76	0,37	0,89	0,42
<i>Sesamothamnus guerichii</i> (Engl.) E. A. Bruce	Pedaliaceae	Ongumbati	Shrub	0,00	0,02	0,00	0,20	0,00	0,32

Species	Family	Vernacular name	Life form	CSI		SI		FI	
				ecolog.	local	ecolog.	local	ecolog.	local
<i>Sterculia africana</i> (Lour.) Fiori	Sterculiaceae	Omu-hako	Tree	0,00	0,01	0,00	0,14	0,11	0,32
<i>Stipagrostis hirtigluma</i> (Henrard) De Winter	Poaceae	Okatjirakonduno	Ann. grass	0,03	0,02	0,21	0,24	0,33	0,37
<i>Stipagrostis uniplumis</i> var. <i>Uniplumis</i> (Licht.) De Winter	Poaceae	Ongumba	Per. grass	0,02	0,09	0,11	0,53	0,22	0,63
<i>Terminalia prunioides</i> M.A.Lawson	Combretaceae	Omuhamama	Shrub	0,00	0,17	0,00	0,74	0,00	0,95
<i>Tribulus</i> sp.	Zygophyllaceae	Ohongo	Herb	0,06	0,02	0,46	0,19	0,67	0,37
<i>Triraphis ramosissima</i> Hack.	Poaceae	Oru-renda	Per. grass	0,00	0,01	0,00	0,16	0,00	0,21
<i>Urochloa brachyura</i> (Hack.) Stapf	Poaceae	Ehozu	Per. grass	0,02	0,19	0,19	0,25	0,33	0,26
<i>Vangueria infausta</i> Burch. subsp. <i>Infausta</i>	Rubiaceae	Omudjenya	Tree	0,00	0,00	0,00	0,06	0,00	0,16
<i>Ximenia americana</i> L.	Olacaceae	Omu-ninga	Tree	0,00	0,01	0,00	0,14	0,00	0,32
<i>Zanthoxylum ovatifoliolatum</i> (Engl.) Finkelstein	Rutaceae	Omuhandua	Tree	0,00	0,00	0,00	0,04	0,00	0,21
<i>Ziziphus mucronata</i> Willd. subsp. <i>mucronata</i>	Rhamnaceae	Omukaru	Shrub	0,00	0,02	0,00	0,18	0,00	0,42

Table A3.1: The mean (\pm SD) percentage cover per site contributed by each life form subscribed to classes of abiotic favourability (Fav._low to Fav._high) and the different management strategies communal grazed (Com._gr.), communal ungrazed (Com._ungr.) and farm ungrazed (Farm_ungr.).

	Fav._low			Fav._med.			Fav._high		
	Ann. grass	Peren. grass	Woody species	Ann. grass	Peren. grass	Woody species	Ann. grass	Peren. grass	Woody species
Com._gr.	0.12 \pm 0.19	0.00	6.75 \pm 4.65	0.33 \pm 0.4	0	12.09 \pm 11.79	2.86 \pm 3.62	0.24 \pm 0.4	4,8 \pm 3.74
Com._ungr.	5.18 \pm 6.16	0.72 \pm 0.73	9.5 \pm 4.28	4.8 \pm 3.03	2.4 \pm 0.89	14.6 \pm 7.89	4.76 \pm 0.94	2.2 \pm 0.84	3.86 \pm 1.26
Farm_ungr.	0.6 \pm 0.55	6.6 \pm 13.11	4.3 \pm 2.49	1.67 \pm 2.16	8 \pm 10.99	3.83 \pm 1.17	7.6 \pm 2.3	15 \pm 6.12	4 \pm 1.73
Sand fraction									
Com._gr.	61.83 \pm 5.77 ($n = 8$)			63.16 \pm 18.41 ($n = 7$)			89.97 \pm 2.46 ($n = 7$)		
Com._ungr.	60.44 \pm 17.53 ($n = 6$)			76.30 \pm 2.17 ($n = 5$)			89,67 \pm 1.38 ($n = 5$)		
Farm_ungr.	68.76 \pm 8 ($n = 8$)			74.79 \pm 3.99 ($n = 6$)			84.56 \pm 2.5 ($n = 5$)		

Table A3.2: Number of plots per management strategy and abiotic favourability (Fav.) unit, for buffer analysis.

	Farms	Communal areas	
	Ungrazed (Farm_ungr.)	Ungrazed (Com._ungr.)	Grazed (Com._gr.)
Abiotic favourability high (Fav._high)	5 plots (F2)	5 plots (B2)	7 plots (2)
Abiotic favourability medium (Fav._med.)	6 plots (F1)	5 plots (B1)	7 plots (1)
Abiotic favourability low (Fav._low)	8 plots (F3)	6 plots (B3)	8 plots (3)

Table A3.3: Species list for biotic buffer analysis: Land use and abiotic favourability are given. H = herb; AG = annual grass; PG = perennial grass; S = shrub; P = phanerophyt (further abbreviations see chapter 3).

Species	Family	Life form	Land use	Land use and abiotic favourability
<i>Abutilon fruticosum</i> Guill. & Perr.	Malvaceae	H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav.
<i>Acacia arenaria</i> Schinz	Mimosaceae	S	communal ungrazed	com_ungr. high_fav.
<i>Acacia erioloba</i> E. Mey.	Mimosaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_gra. high_fav., com_ungr. high_fav.
<i>Acacia erubescens</i> Welw. ex Oliv.	Mimosaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav., com_gra. low_fav.
<i>Acacia hebeclada</i> DC.	Mimosaceae	S	communal ungrazed	com_ungr. low_fav.
<i>Acacia hereroensis</i> Engl.	Mimosaceae	S	farm ungrazed, communal ungrazed	farm_ungr. low_fav., com_ungr. high_fav.
<i>Acacia karroo</i> Hayne	Mimosaceae	S	farm ungrazed	farm_ungr. low_fav.
<i>Acacia luederitzii</i> Engl. var. <i>luederitzii</i>	Mimosaceae	P	communal ungrazed	com_ungr. high_fav.
<i>Acacia mellifera</i> (Vahl) Benth.	Mimosaceae	S, P	communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav., com_ungr. high_fav.
<i>Acacia nilotica</i> ssp. <i>kraussiana</i> (Benth.) Brenan	Mimosaceae	S	communal grazed	com_gra. high_fav.
<i>Acacia reficiens</i> Wawra subsp. <i>reficiens</i>	Mimosaceae	S	farm ungrazed	farm_ungr. high_fav.
<i>Acacia</i> sp.	Mimosaceae	S, P, H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. high_fav.
<i>Acalypha fruticosa</i> Forssk.	Euphorbiaceae	S	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_gra. high_fav.
<i>Acrotome inflata</i> Benth.	Lamiaceae	H	farm ungrazed, communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. high_fav., farm_ungr. high_fav.
<i>Aizoon virgatum</i> Welw. ex Oliv.	Aizoaceae	H	communal ungrazed	com_ungr. low_fav.
<i>Albizia anthelmintica</i> (A. Rich.) Brongn.	Mimosaceae	S, P	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. high_fav.
<i>Alternanthera pungens</i> Kunth	Amaranthaceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_gra. high_fav.
<i>Amaranthus dinteri</i> Schinz subsp. <i>dinteri</i>	Amaranthaceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_ungr. med_fav.
<i>Antheplora pubescens</i> Nees	Poaceae	PG	communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
<i>Antheophora schinzii</i> Hack.	Poaceae	AG	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., farm_ungr. high_fav., com_ungr. med_fav.
<i>Aptosimum lineare</i> Marloth & Engl.	Scrophulariaceae	S, H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. med_fav.
<i>Aristida adscensionis</i> L.	Poaceae	AG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav.
<i>Aristida hordeacea</i> Kunth	Poaceae	AG	farm ungrazed	com_ungr. low_fav., farm_ungr. high_fav.
<i>Aristida vestita</i> Thunb.	Poaceae	PG	farm ungrazed	farm_ungr. high_fav.
<i>Asparagus nelsii</i> Schinz	Liliaceae	H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. high_fav.
<i>Barleria prionitoides</i> Engl.	Acanthaceae	H	communal ungrazed	com_ungr. high_fav.
<i>Barleria</i> sp.	Acanthaceae	H	farm ungrazed	farm_ungr. high_fav.
<i>Boscia albitrunca</i> (Burch.) Gilg & Benedict	Capparaceae	S, P, H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_ungr. med_fav., com_ungr. high_fav.
<i>Boscia foetida</i> Schinz	Capparaceae	S, P	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_ungr. high_fav.
<i>Brachiaria nigropedata</i> (Ficalho & Hiern) Stapf	Poaceae	PG	farm ungrazed	farm_ungr. high_fav.
<i>Bulbostylis hispidula</i> (Vahl.) R.W.Haines	Cyperaceae	H	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. high_fav.
<i>Cadaba aphylla</i> (Thunb.) Wild	Capparaceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. med_fav.
<i>Catophractes alexandri</i> D.Don	Bignoniaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav., com_ungr. high_fav.
<i>Cenchrus ciliaris</i> L.	Poaceae	PG	farm ungrazed, communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. med_fav., farm_ungr. high_fav.
<i>Chloris virgata</i> Sw.	Poaceae	AG	farm ungrazed, communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav.
<i>Colophospermum mopane</i> (J.Kirk ex Benth.) J.Kirk ex J.Léonard	Caesalpiaceae	S, P, H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav.
<i>Combretum apiculatum</i> Sond. subsp. <i>apiculatum</i>	Combretaceae	S, P	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
Commicarpus fallacissimus (Heimerl) Heimerl ex Oberm.	Nyctaginaceae	S	communal ungrazed	com_ungr. low_fav.
Commiphora glandulosa Schinz	Burseraceae	S, P	communal ungrazed	com_ungr. low_fav., com_ungr. low_fav.
Commiphora pyracanthoides Engl.	Burseraceae	S, P	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_gra. low_fav., farm_ungr. high_fav., com_ungr. high_fav.
Commiphora sp.	Burseraceae	S	farm ungrazed	farm_ungr. high_fav.
Corchorus asplenifolius Burch.	Tiliaceae	H	communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav.
creeper 1		H	communal ungrazed	com_ungr. low_fav.
creeper 2		H	communal ungrazed	com_ungr. low_fav.
creeper 3 OKP2		H	communal ungrazed	com_ungr. low_fav.
creeper like Grewia		H	communal grazed	com_gra. high_fav.
creeper round holz		S, H	communal ungrazed	com_ungr. high_fav.
Crotalaria podocarpa DC.	Fabaceae: Papilionoideae	H	communal ungrazed	com_ungr. med_fav.
Crotalaria sp.	Fabaceae	H	farm ungrazed, communal ungrazed	farm_ungr. high_fav., com_ungr. med_fav.
Cullen obtusifolia (DC.)C.H. Stirt.	Fabaceae	H	communal ungrazed	com_ungr. low_fav.
Dichrostachys cinera (L.) Wight & Arn.	Mimosaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav., com_gra. low_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. high_fav.
Dicoma tomentosa Cass.	Asteraceae	H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav.
einfach shrub		S	farm ungrazed	farm_ungr. low_fav.
Elephantorrhiza suffruticosa Schinz	Mimosaceae	S	communal ungrazed	com_ungr. high_fav.
Enneapogon cenchroides (Roem. & Schult.) C.E.Hubb.	Poaceae	AG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav., com_gra. low_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav., com_ungr. high_fav.
Enneapogon desvauxii P.Beauv.	Poaceae	AG	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav.
Eragrostis annulata Rendle ex Scott-Elliot	Poaceae	AG	farm ungrazed, communal ungrazed	farm_ungr. high_fav., com_ungr. med_fav.
Eragrostis echinochloidea Stapf	Poaceae	PG	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
Eragrostis nindensis Ficalho & Hiern	Poaceae	PG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. med_fav., com_gra. low_fav., farm_ungr. high_fav., com_ungr. med_fav.
Eragrostis porosa Nees	Poaceae	AG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., com_ungr. med_fav.
Eragrostis superba Peyr.	Poaceae	PG	communal ungrazed	com_ungr. med_fav.
Eriocephalus luederitzianus O Hoffm.	Asteraceae	S	farm ungrazed, communal ungrazed, communal grazed	com_ungr. low_fav., farm_ungr. high_fav.
Euphorbia sp.	Euphorbiaceae	H	communal grazed	com_gra. high_fav.
Evolvulus alsinoides (L.) L.	Convolvulaceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. high_fav.
Flaveria bidentis (L.) Kuntze	Asteraceae	S	communal ungrazed	com_ungr. low_fav.
Geigeria acaulis Benth. & Hook.f. ex Oliv. & Hiern	Asteraceae	H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., farm_ungr. high_fav.
Geigeria alata (DC.) Benth. & Hook.f. ex Oliv. & Hiern	Asteraceae	H	farm ungrazed, communal grazed	com_gra. low_fav., farm_ungr. high_fav.
Geigeria ornativa O.Hoffm.	Asteraceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. high_fav.
Geophyt OZ3000		H	communal grazed	com_gra. high_fav.
Geophyt sukk poison		H	communal grazed	com_gra. high_fav.
Gisekia africana (Lour.) Kuntze	Gisekiaceae	H	communal ungrazed, communal grazed	com_gra. low_fav., com_gra. med_fav., com_ungr. med_fav.
Gossypium triphyllum (Harv.)Hochr.	Malvaceae	S	farm ungrazed, communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav.
gras OKP3		AG	communal ungrazed	com_ungr. low_fav.
Grewia bicolor Juss.	Tiliaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. high_fav.
Grewia flava DC.	Tiliaceae	S	farm ungrazed, communal ungrazed	farm_ungr. high_fav., com_ungr. high_fav.
Grewia flavescens Juss.	Tiliaceae	S	communal ungrazed	com_ungr. high_fav.
Grewia sp.	Tiliaceae	S	farm ungrazed	farm_ungr. high_fav.
Grewia tenax (Forssk) Fiori	Tiliaceae	S	farm ungrazed, communal ungrazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. low_fav., farm_ungr. high_fav.
Grewia villosa Willd.	Tiliaceae	S	communal ungrazed, communal	com_ungr. low_fav., com_gra. med_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
			grazed	
Helichrysum tomentosulum (Klatt) Merxm. subsp. tomentosulum	Asteraceae	S	farm ungrazed, communal ungrazed	farm_ungr. low_fav., com_ungr. low_fav., farm_ungr. high_fav.
Heliotropium lineare A. DC., herb	Asteraceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. high_fav., com_ungr. med_fav., com_ungr. high_fav.
herb creep OZ3000		H	communal ungrazed	com_ungr. low_fav., com_ungr. high_fav.
herb ERRef1		H	communal grazed	com_gra. high_fav.
herb KG200J		H	farm ungrazed	farm_ungr. high_fav.
herb OKP2		H	farm ungrazed	farm_ungr. low_fav.
herb OKP4		H	communal ungrazed	farm_ungr. med_fav.
herb OR 2000		H	communal ungrazed	com_ungr. low_fav.
herb OZ 4000		H	communal grazed	com_ungr. low_fav.
herb OZ1200		H	communal grazed	com_gra. med_fav.
herb OZ200		H	communal grazed	com_gra. high_fav.
herb OZ2000		H	communal grazed	com_gra. high_fav.
herb OZ3000		H	communal grazed	com_gra. high_fav.
herb OZ4000		H	communal grazed	com_gra. high_fav.
herb2 OZ1200		H	communal grazed	com_gra. high_fav.
herb2 OZ2000		H	communal grazed	com_gra. high_fav.
herb2 OZ3000		H	communal grazed	com_gra. high_fav.
herb3 OZ3000		H	communal grazed	com_gra. high_fav.
Hermannia glandulosissima Engl.	Sterculiaceae	H	communal grazed	com_gra. high_fav.
Hermannia tigrensis Hochst. ex A.Rich.	Sterculiaceae	H	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. med_fav.
Hermbstaedtia linearis Schinz	Amaranthaceae	H	farm ungrazed	farm_ungr. high_fav.
Hermbstaedtia odorata (Burch.) T.Cooke var. odorata	Amaranthaceae	H	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. high_fav.
Hibiscus calyphyllus Cav.	Malvaceae	S	communal ungrazed	com_ungr. low_fav.
Hiernia angolensis S.Moore	Scrophulariaceae	H	communal ungrazed	com_ungr. high_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
<i>Indigofera alternans</i> DC.	Fabaceae: Papilionoideae	H	communal ungrazed	com_ungr. high_fav.
<i>Indigofera auricomia</i> E.Mey.	Fabaceae: Papilionoideae	H	communal ungrazed	com_ungr. low_fav.
<i>Indigofera charlieriana</i> Schinz	Fabaceae: Papilionoideae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., com_ungr. high_fav.
<i>Indigofera heterotricha</i> DC.	Fabaceae: Papilionoideae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav.
<i>Indigofera</i> sp.	Fabaceae	H	communal ungrazed	com_ungr. med_fav.
Kraut dickblatt		H	communal ungrazed	com_ungr. low_fav.
Kugeldistel		H	communal ungrazed	com_ungr. low_fav.
<i>Leucas pechuelii</i> (Kuntze) Gürke	Lamiaceae	S	communal ungrazed	com_ungr. med_fav.
<i>Leucosphaera bainesii</i> (Hook. F.) Gilg	Amaranthaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. med_fav., farm_ungr. high_fav.
like Dicho cin fruit		S	farm ungrazed	farm_ungr. med_fav.
<i>Limeum pterocarpum</i> (J.Gay) Heimerl	Molluginaceae	H	communal ungrazed	com_ungr. med_fav.
<i>Limeum</i> sp.	Molluginaceae	H	communal ungrazed	com_ungr. med_fav.
<i>Melhania damarana</i> Harv.	Malvaceae	H	communal grazed	com_gra. high_fav.
<i>Melinis repens</i> ssp. <i>Grandiflora</i> (Hochst.) Zizka	Poaceae	PG	farm ungrazed, communal ungrazed	farm_ungr. high_fav., farm_ungr. med_fav., com_ungr. low_fav., com_ungr. med_fav., com_ungr. high_fav.
<i>Monechma genistifolium</i>	Acanthaceae	S	farm ungrazed	farm_ungr. low_fav., farm_ungr. high_fav.
<i>Monelytrum luederitzianum</i> Hack.	Poaceae	PG	communal grazed	com_gra. low_fav., com_gra. med_fav.
<i>Montinia caryophyllacea</i> Thunb.	Montiniaceae	S	communal ungrazed	com_ungr. low_fav., com_ungr. high_fav.
<i>Mundulea sericea</i> (Willd.) A.Chev.	Fabaceae: Papilionoideae	S	communal grazed	com_gra. high_fav.
<i>Nelsia quadrangula</i> (Engl.) Schinz	Amaranthaceae	H	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. med_fav., com_ungr. low_fav., com_gra. med_fav., farm_ungr. high_fav., com_ungr. med_fav.
nessel shrub		S	communal grazed	com_gra. low_fav.
no leaves, dry		H	communal ungrazed	com_ungr. high_fav.
<i>Ocimum canum</i> Sims	Lamiaceae	H	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_ungr. high_fav.
OZ crawler		H	farm ungrazed	farm_ungr. low_fav.
<i>Pavonia burchellii</i> (DC.)R. A. Dyer	Malvaceae	H	farm ungrazed, communal ungrazed	farm_ungr. med_fav., com_ungr. low_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
Pechuel-Loeschea leubnitziae (Kuntze) O.Hoffm.	Asteraceae	S	communal grazed	com_gra. low_fav.
Petalidium sp.	Acanthaceae	S	farm ungrazed	farm_ungr. low_fav., farm_ungr. med._fav., farm_ungr. high_fav.
Phaeoptilum spinosum Radlk.	Nyctaginaceae	S	farm ungrazed, communal ungrazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav.
Phyllanthus pentandrus Schumach. & Thonn.	Euphorbiaceae	H	communal ungrazed	com_ungr. low_fav.
Pogonarthria fleckii (Hack.) Hack.	Poaceae	AG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav., com_gra. low_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav., com_ungr. high_fav.
Pollichia campestris Aiton	Caryophyllaceae	S	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. high_fav.
powdery stem		H	communal ungrazed	com_ungr. med_fav.
Pupalia lappacea (L.) A.Juss.	Amaranthaceae	H	farm ungrazed, communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav.
Requienia sphaerosperma DC.	Fabaceae	H	communal ungrazed	com_ungr. high_fav.
Rhigozum brevispinosum Kuntze	Bignoniaceae	S	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_ungr. med_fav., com_ungr. high_fav.
Rhus tenuinervis Engl.	Anacardiaceae	S	communal ungrazed	com_ungr. high_fav.
Rhynchosia candida (Welw. ex Hiern)Torre	Leguminosae	H	farm ungrazed, communal ungrazed	com_ungr. low_fav., farm_ungr. high_fav.
Rhynchosia minima (L.)DC.	Leguminosae	H	farm ungrazed, communal grazed	farm_ungr. low_fav., com_gra. high_fav.
Ruelliopsis damarensis S. Moore	Acanthaceae	H	communal ungrazed	com_ungr. low_fav.
s woll hair tooth/gros stern		S	farm ungrazed	farm_ungr. med._fav.
schmalblashru		S	communal ungrazed	com_ungr. low_fav.
Schmidtia kalahariensis Stent	Poaceae	AG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med._fav., com_ungr. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav., com_ungr. high_fav.
Schmidtia sp.	Poaceae	AG	communal grazed	com_gra. low_fav.
Senna italica Mill.	Fabaceae	H	communal ungrazed	com_ungr. low_fav.
Sericorema sericea (Schinz) Lopr.	Amaranthaceae	S	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. high_fav.
Sesamum sp.	Pedaliaceae	H	communal grazed	com_gra. high_fav.
shr wo lea		S	communal ungrazed	com_ungr. high_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
shrub OZ2000		S	communal grazed	com_gra. high_fav.
shrub 1 OKP06		S	communal ungrazed	com_ungr. low_fav.
shrub 2 OKP3		S	communal ungrazed	com_ungr. low_fav.
shrub big leaves		S	communal ungrazed	com_ungr. low_fav.
shrub dreiblatt		S	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. high_fav.
shrub haariges dreiblatt OZ		S	farm ungrazed	farm_ungr. low_fav.
shrub OKP3		S	communal ungrazed	com_ungr. low_fav.
shrub OPP3		S	communal grazed	com_ungr. low_fav.
shrub OR 3000		S	communal grazed	com_gra. low_fav.
shrub OZ 3000		S	communal ungrazed	com_gra. med_fav.
shrub OZ1200		S	communal grazed	com_ungr. high_fav.
shrub OZ5000		S	communal grazed	com_gra. high_fav.
shrub OZRef5		S	communal ungrazed	com_gra. high_fav.
shrub Petal		S	communal grazed	com_ungr. high_fav.
shrub rund holz		S	communal ungrazed	com_gra. high_fav.
shrub without leaves		S	communal ungrazed	com_ungr. high_fav.
shrub without leaves		S	communal ungrazed	com_ungr. high_fav.
shrub1 OKP4		S	communal ungrazed	com_ungr. low_fav.
shrub2 OKP4		S	communal ungrazed	com_ungr. low_fav.
shrub2 OZ3000		S	communal grazed	com_gra. high_fav.
shrubRO2100		S	farm ungrazed	farm_ungr. high_fav.
Solanum catombelense Peyr.	Solanaceae	S	communal ungrazed	com_ungr. high_fav.
spikey citrus		S	communal grazed	com_gra. high_fav.
ss ER150J		S	farm ungrazed	farm_ungr. low_fav.
ss OZ5000		S	communal grazed	com_gra. high_fav.
sshub		S	communal ungrazed	com_ungr. high_fav.
sshub OZ 4000		S	communal grazed	com_gra. high_fav.

Species	Family	Life form	Land use	Land use and abiotic favourability
sshub OZRef5		S	communal ungrazed	com_ungr. high_fav.
sticky herb yellow flower		H	farm ungrazed	farm_ungr. high_fav.
Stipagrostis hirtigluma (Henrard) De Winter	Poaceae	AG	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav.
Stipagrostis sp.	Poaceae	PG	communal ungrazed	com_ungr. med_fav.
Stipagrostis uniplumis var. Uniplumis (Licht.) De Winter	Poaceae	PG	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., com_gra. high_fav., farm_ungr. high_fav., com_ungr. med_fav., com_ungr. high_fav.
subshrub 1 OK P02		S	communal ungrazed	com_ungr. low_fav.
subshrub 2 OK P03		S	communal ungrazed	com_ungr. low_fav.
Tephrosia lupinifolia DC.	Fabaceae: Papilionoideae	H	communal ungrazed	com_ungr. high_fav.
Tephrosia monophylla Schinz	Fabaceae: Papilionoideae	S	communal ungrazed	com_ungr. high_fav.
Terminalia prunioides M.A.Lawson	Combretaceae	S, P	farm ungrazed, communal ungrazed, communal grazed	farm_ungr. low_fav., farm_ungr. med_fav., com_ungr. low_fav., com_gra. low_fav., farm_ungr. high_fav., com_ungr. high_fav.
Terminalia sericea Burch. ex DC.	Combretaceae	S	communal ungrazed	com_ungr. high_fav.
Tragus sp.	Poaceae	AG	communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav.
Tree OR 1000		P	communal grazed	com_gra. med_fav.
Triaspis hypericoides (DC.)Burch. subsp. nelsonii (Oliv.) Immelmann	Malpighiaceae	S	farm ungrazed, communal ungrazed	farm_ungr. med_fav., com_ungr. low_fav.
Tribulus sp.	Zygophyllaceae	H	farm ungrazed, communal ungrazed, communal grazed	com_ungr. low_fav., com_gra. low_fav., com_gra. med_fav., farm_ungr. high_fav.
Triraphis purpurea Hack.	Poaceae	AG	communal grazed	com_gra. high_fav.
Xenostegia tridentata (L.)D.F. Austin & Staples subsp. angustifolia (Jacq.) A. Meeuse		H	communal ungrazed, communal grazed	com_gra. high_fav., com_ungr. high_fav.
Ximenia americana L. var. microphylla Welw. ex Oliv.	Olacaceae	S	communal grazed	com_gra. low_fav.
yellow red stem		H	communal ungrazed	com_ungr. high_fav.
Ziziphus mucronata Willd. subsp. mucronata	Rhamnaceae	S	communal grazed	com_ungr. low_fav., com_gra. high_fav., farm_ungr. high_fav.

Table A4.1: Summary of the results and application of memory level and predicted grazing impact. Ann. gra. = Annual grasses; Per. gra. = Perennial grasses.

Indicator	Memory	Predicted sensitivity to spatial variability	sensitivity to grazing impact	Communal			Farm cattle			Farm game		
				Mean	SD	r ²	Mean	SD	r ²	Mean	SD	r ²
Recent grazing												
Bare ground	Short-term	o	++	70.18	18.93	ns	38.83	24.57	ns	70	15.81	ns
Abiotic												
C _{org}	Long-term	++	+	0.52	0.25	0.04	0.38	0.16	ns	0.38	0.13	ns
N _{tot}		++	+	0.05	0.03		0.03	0.02	0.27	0.03	0.008	ns
Biotic												
Ann. gra.	Short-term	+	++	2.07	3.74	0.04	25.43	27.89	ns	1.07	1.53	0.08
Per. gra.	Medium	+	++	0.25	0.49	ns	7.75	8.88	0.14	6.77	10.49	0.34
Woody sp.	Long-term	+	+	10.76	8.7	ns	16.79	15.89	ns	4.23	1.84	ns

Table A4.2: Species list for transect analysis. Sampling distances of species are given. H = herb; AG = annual grass; PG = perennial grass; S = shrub; P = phanerophyt

Species	Family	PFT	Sampling distance																			
<i>Abutilon fruticosum</i> Guill. & Perr.	Malvaceae	H	50	100	150	200	300	500	700	1000	1500	1600	1700	2000	2500	3000	4000					
<i>Acacia erioloba</i> E. Mey.	Mimosaceae	S	100	200	1200	2000	3000	4000	5000													
<i>Acacia erubescens</i> Welw. ex Oliv.	Mimosaceae	S	100	200	2000	500	1000	1600	2000	2500	3000	4000										
<i>Acacia hebeclada</i> DC.	Mimosaceae	S	300	500																		
<i>Acacia hereroensis</i> Engl.	Mimosaceae	S	500	2000																		
<i>Acacia karroo</i> Hayne	Mimosaceae	S	2000																			
<i>Acacia luederitzii</i> Engl. var. <i>luederitzii</i>	Mimosaceae	P	50	700	1500																	
<i>Acacia mellifera</i> (Vahl) Benth.	Mimosaceae	S, P	50	100	200	500	1000	1200	1500	2000	2500											
<i>Acacia nilotica</i> ssp. <i>kraussiana</i> (Benth.) Brenan	Mimosaceae	S	2000	3000																		
<i>Acacia reficiens</i> Wawra subsp. <i>reficiens</i>	Mimosaceae	S	3000																			
<i>Acacia</i> sp.	Mimosaceae	S, P, H	50	100	200	500	1000	1200	1500	1700	2000	3000										
<i>Acalypha fruticosa</i> Forssk.	Euphorbiaceae	S	200	500	2000	1200	1600															
<i>Acrotome inflata</i> Benth.	Lamiaceae	H	300	500	1200	4000	3000															
<i>Aizoon virgatum</i> Welw. ex Oliv.	Aizoaceae	H	2000																			
<i>Albizia anthelmintica</i> (A. Rich.) Brongn.	Mimosaceae	S, P	50	100	550	2000	2100	3000	4000													
<i>Alternanthera pungens</i> Kunth	Amaranthaceae	H	50	300	1200																	
<i>Amaranthus dinteri</i> Schinz subsp. <i>dinteri</i>	Amaranthaceae	H	50	100	500	1700	2000															
<i>Anthepphora pubescens</i> Nees	Poaceae	PG	200	1200	2000																	
<i>Anthepphora schinzii</i> Hack.	Poaceae	AG	50	100	200	300	500	1000	1200	1500	1600	1700	2000	2500	3000	4000						
<i>Aptosimum lineare</i> Marloth & Engl.	Scrophulariaceae	S, H	50	200	300	500	1000	1200	2500	3000												
<i>Aristida adscensionis</i> L.	Poaceae	AG	50	100	150	200	300	500	550	700	1000	1200	1500	1600	1700	2000	2500	2500	3000	4000	5000	5500
<i>Aristida hordeacea</i> Kunth	Poaceae	AG	100	200	500	1500	3000															
<i>Aristida</i> sp.	Poaceae	PG	200	1500																		
<i>Aristida vestita</i> Thunb.	Poaceae	PG	100	200	500	550	1000	1500														
<i>Asparagus nelsii</i> Schinz	Liliaceae	H	50	100	200	300	2000	3000	4000													

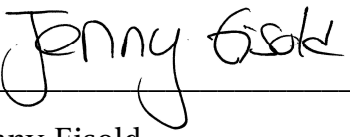
Species	Family	PFT	Sampling distance																					
Barleria prionitoides Engl.	Acanthaceae	H	1000																					
Barleria sp.	Acanthaceae	H	100	550																				
Blepharis pruinosa Engl.	Acanthaceae	H	300																					
Blepharis spinosa	Acanthaceae	H	1500																					
Boscia albitrunca (Burch.) Gilg & Benedict	Capparaceae	S, P, H	50	100	150	200	500	1000	1700	2000	3000	5000	5500											
Boscia foetida Schinz	Capparaceae	S, P	50	100	200	300	500	700	1000	1500	2000	2500	3000	4000	5000									
Brachiaria nigropedata (Ficalho & Hiern) Stapf	Poaceae	PG	3000																					
Bulbostylis hispidula (Vahl.) R.W.Haines	Cyperaceae	H	2000	3000	4000	5000																		
Cadaba aphylla (Thunb.) Wild	Capparaceae	H	300	500	1000	1200	2000	3000																
Catophractes alexandri D.Don	Bignoniaceae	S	50	100	150	200	300	500	550	700	1000	1200	1500	1700	2000	2100	2500	3000	4000					
Cenchrus ciliaris L.	Poaceae	PG	100	200	300	500	550	700	1500	2000	2500	4000												
Cephaloctrion mollis		S	100	200																				
Chloris virgata Sw.	Poaceae	AG	50	100	500	550	1500	2000																
Cleome foliosa Hook.f. var. foliosa	Capparaceae	H	200																					
Colophospermum mopane (J.Kirk ex Benth.) J.Kirk ex J.Léonard	Caesalpinaceae	S, P, H	50	100	150	200	300	500	550	700	1000	1200	1500	1600	1700	2000	2100	2200	2500	3000	4000	5000	5500	
Combretum apiculatum Sond. subsp. apiculatum	Combretaceae	S, P	50	100	100	100	200	200	500	500	1000	1000	1000	1000	1200	1600	2000	2000	2000	2500	3000	4000		
Combretum imberbe Wawra	Combretaceae	S	200	300	500																			
Commicarpus fallacissimus (Heimerl) Heimerl ex Oberm.	Nyctaginaceae	S	500																					
Commiphora angolensis Engl.	Burseraceae	S	200																					
Commiphora glandulosa Schinz	Burseraceae	S, P	50	100	200	300	500	700	1500	2500														
Commiphora pyracanthoides Engl.	Burseraceae	S, P	50	50	100	150	200	500	1000	1500	2000	2500	3000											
Commiphora sp.	Burseraceae	S	2200																					
Corchorus asplenifolius Burch.	Tiliaceae	H	300	200																				
Crinum minimum Milne.-Redh.	Amaryllidaceae	H	100	200	500																			
Crotolaria sp.	Fabaceae	H	550																					
Cucumis anguria L.	Cucurbitaceae	H	50																					
Cullen obtusifolia (DC.)C.H. Stirt.	Fabaceae	H	300																					
Dactyliandra welwitschii Hook. F.	Cucurbitaceae	H	100	500																				

Species	Family	PFT	Sampling distance																	
<i>Grewia villosa</i> Willd.	Tiliaceae	S	1000	2000																
<i>Helichrysum tomentosulum</i> (Klatt)	Asteraceae	S	100	200	550	1200	1500													
<i>Merxm. subsp. tomentosulum</i>																				
<i>Heliotropium lineare</i> A. DC.,	Asteraceae	H	1000	2000	3000	4000	5000													
<i>Hermannia glandulosissima</i> Engl.	Sterculiaceae	H	1200																	
<i>Hermannia tigrensis</i> Hochst. ex A.Rich.	Sterculiaceae	H	100	200	4000															
<i>Hermbsstaedtia linearis</i> Schinz	Amaranthaceae	H	100	550	1500	2000	2100	2200												
<i>Hermbsstaedtia odorata</i> (Burch.) T.Cooke var. <i>odorata</i>	Amaranthaceae	H	200	700	4000	5000														
<i>Hibiscus caesius</i> Garcke	Malvaceae	H	500																	
<i>Hibiscus calyphyllus</i> Cav.	Malvaceae	S	500	2000																
<i>Hibiscus micranthus</i> Linn. f.	Malvaceae	H	300	1500																
<i>Indigofera auricomata</i> E.Mey.	Fabaceae: Papilionoideae	H	50	100	1200															
<i>Indigofera charlieriana</i> Schinz	Fabaceae: Papilionoideae	H	50	100	200	300	500	1000	1200	1600	1700	2000	3000	4000	5000					
<i>Indigofera heterotricha</i> DC.	Fabaceae: Papilionoideae	H	100	200	500	700	1000	1200	1500	1600	2000	2500								
<i>Ipomoea sinensis</i> (Desr.) Choisy	Convolvulaceae	S	100																	
<i>Kohautia</i> sp.	Rubiaceae	H	200																	
<i>Lantana dinteri</i> Moldenke	Verbenaceae	S	700																	
<i>Leucas ebracteata</i> Peyr. var. <i>kaokoveldensis</i>	Lamiaceae	H	50	100	200															
<i>Leucas pechuelii</i> (Kuntze) Gürke	Lamiaceae	S	500	1500																
<i>Leucosphaera bainesii</i> (Hook. F.) Gilg	Amaranthaceae	S	50	100	150	200	300	500	550	700	1000	1500	2000	3000						
<i>Lycium bosciifolium</i> Schinz	Solanaceae	S	1500																	
<i>Megalochlamys marlothii</i> (Engl.) Lindau	Acanthaceae	S	200	500																
<i>Melhania damarana</i> Harv.	Malvaceae	H	2000																	
<i>Melinis repens</i> ssp. <i>Grandiflora</i> (Hochst.) Zizka	Poaceae	PG	100	550	1500	2100	2200	2500	3000											
<i>Melinis repens</i> (Willd.) Zizka ssp. <i>Repens</i>	Poaceae	AG	100	200	300	500	700	1000	1200	2500										
<i>Momordica humilis</i> Wall.	Cucurbitaceae	H	200	500																
<i>Monechma genistifolium</i>	Acanthaceae	S	100	200	500	700	1500	2000	3000											

Species	Family	PFT	Sampling distance																
Verdc.																			
Setaria verticillata (L.) P. Beauv.	Poaceae	AG	50	1500															
Stipagrostis hirtigluma (Henrard) De Winter	Poaceae	AG	200	1000	1200	1500	1700	2000	2500	3000	4000	5500							
Stipagrostis hochstetteriana (Beck ex Hack.) De Winter var. Hochstetteriana	Poaceae	AG	1500																
Stipagrostis uniplumis var. Uniplumis (Licht.) De Winter	Poaceae	PG	50	100	200	300	500	550	700	1000	1200	1500	2000	2100	2200	2500	3000	4000	5000
Talinum sp.	Portulacaceae	H	100	200															
Terminalia prunioides M.A.Lawson	Combretaceae	S, P	50	100	150	200	300	500	550	1000	1200	1500	1600	1700	2000	2500	3000	4000	
Tragus sp.	Poaceae	AG	50	50	100	300	500	700	1200	1500	1700	2500	3000						
Triaspis hypericoides (DC.)Burch. subsp. nelsonii (Oliv.) Immelmann	Malpighiaceae	S	500	1000	2000														
Tribulus sp.	Zygophyllaceae	H	50		100	200	500	700	1200	1500	1600	1700	2000	2100	2200	2500			
Tricholaena monachme (Trin.) Stapf ex C.E.Hubb	Poaceae	AG	200																
Triraphis purpurea Hack.	Poaceae	AG	4000	5000															
Urochloa brachyura (Hack.) Stapf	Poaceae	AG	50	100	500														
Xenostegia tridentata (L.)D.F. Austin & Staples subsp. angustifolia (Jacq.) A. Meeuse		H	4000	5000															
Ximenia americana L. var. microphylla Welw. ex Oliv.	Olacaceae	S	1600																
Ziziphus mucronata Willd. subsp. mucronata	Rhamnaceae	S	100	200	4000														

Erklärung

„Ich versichere, dass ich die von mir vorgelegte Dissertation selbständig angefertigt, die benutzten Quellen und Hilfsmittel vollständig angegeben und die Stellen der Arbeit – einschließlich Tabellen, Karten und Abbildungen –, die anderen Werken im Wortlaut oder dem Sinn nach entnommen sind, in jedem Einzelfall als Entlehnung kenntlich gemacht habe; dass diese Dissertation noch keiner anderen Fakultät oder Universität zur Prüfung vorgelegen hat; dass sie – abgesehen von unten angegebenen Teilpublikationen – noch nicht veröffentlicht worden ist sowie, dass ich eine solche Veröffentlichung vor Abschluss des Promotionsverfahrens nicht vornehmen werde. Die Bestimmungen der Promotionsordnung sind mir bekannt. Die von mir vorgelegte Dissertation ist von Prof. Dr. Helmut Hillebrand betreut worden.“



Jenny Eisold

Curriculum Vitae

Persönliche Daten

Jenny Eisold

* 13.04.1977 in Minden

Ledig

Deutsche Staatsangehörigkeit

Schulbildung

1987–1996

Nord-Pfalz Gymnasium in Kirchheimbolanden
Abschluss: Allgemeine Hochschulreife

Berufstätigkeit

seit 02.2009

wissenschaftliche Mitarbeiterin am Institut für sozial-
ökologische Forschung (ISOE), GmbH, Frankfurt, M.
Projektkoordinatorin des Verbundprojekts
„CuveWaters-IWRM in north central Namibia“
www.isoe.de

09.2008–02.2009

wissenschaftliche Mitarbeiterin am Helmholtz Zentrum
für Umweltforschung (Ufz), Leipzig, im Department
Ökologische Modelbildung.
http://www.nomadsed.de/projects_08-12/en_e10.html

11.2005–12.2007

wissenschaftliche Mitarbeiterin in der VW-Nachwuchs-
gruppe „Ökologische und ökonomische Nachhaltigkeit
verschiedener Nutzungsformen einer Savannenland-
schaft Nordnamibias“ (Dr. A. Linstädter), Univ. zu Köln;
http://www.savanna.uni-koeln.de/index_e.htm

Hochschule

10.1999–10.2005

Studium der Geoökologie an der Univ. Bayreuth

Diplomarbeit: „Spatial patterns of plant biodiversity in
savanna ecosystems in Northwestern Namibia“

seit 10.2006

Promotion im Fach Biologie/Botanik an der Universität
zu Köln

Auszeichnungen

- 10.2006 Poster-Award für
Eisold, J, Tönsjost, S, Bollig M & Linstädter, A:
„Local and Ecological Knowledge on Natural Resource
Management – A Case Study from north-western Na-
mibia“. – Tropentag – Conference on International Ag-
ricultural Research for Development, Bonn (October 11-
13, 2006).
<http://tropentag.de>

Stipendien

- 10.2007–1.2008 Stipendium der Hans Merensky Stipendienstiftung
„Nachhaltiges Management natürlicher Ressourcen –
lokale und wissenschaftliche Umweltwahrnehmung“

Praktika

- 05.2004–07.2004 Forschungspraktikum am Helmholtz Zentrum für
Umweltforschung (Ufz), Leipzig, im Department
Naturschutz „Vegetationsökologie und nachhaltige
Landnutzung in semiariden Landschaften“
(Kontakt: Jun.-Prof. Dr. Anke Jentsch)
- 08.2003–10.2003 Praktikum der Permakultur in den Niederlanden
„Nachhaltiger Gartenbau & Landschaftsdesign“
(Kontakt: Dip.Perm.Des. Harald Wedig)

ehrenamtliche Tätigkeiten

- seit 07.2009 Beirat der namibischen NRO EduVentures (Umweltbil-
dung)
- 3.2008 Wissenschaftliche Beratung und Koordination bei den
Dreharbeiten der 360° GEO Reportage „Vom Klassen-
zimmer in die Kalahari“ (ausgestrahlt am 31.07.2008
auf ARTE)
Innerhalb der Dokumentation Teilprojektleitung „The
Daily Life of the Jul’hoansi today-with special focus
on hunting & gathering “ zur Teilnahme von namibia-
nischen Schulkindern am nationalen „Sciencefair“.
Beitrag wurde mit einer Medaille ausgezeichnet.

weiterbildende Seminare

2008 Einführung ins Projektmanagement
 Grundlagen erfolgreicher Kommunikation und
 Gesprächsführung
 Speed reading
 Frauen zeigen Präsenz – Strategien professionellen
 Auftretens (female career center der Univ. zu Köln)

Frankfurt, den 28.11.2009

Jenny Eisold