

Aus dem Institut für Pflanzenernährung und Bodenkunde  
der Christian-Albrechts-Universität zu Kiel

**Nitrogen dynamics as affected by grazing in  
semi-arid grasslands of Inner Mongolia**

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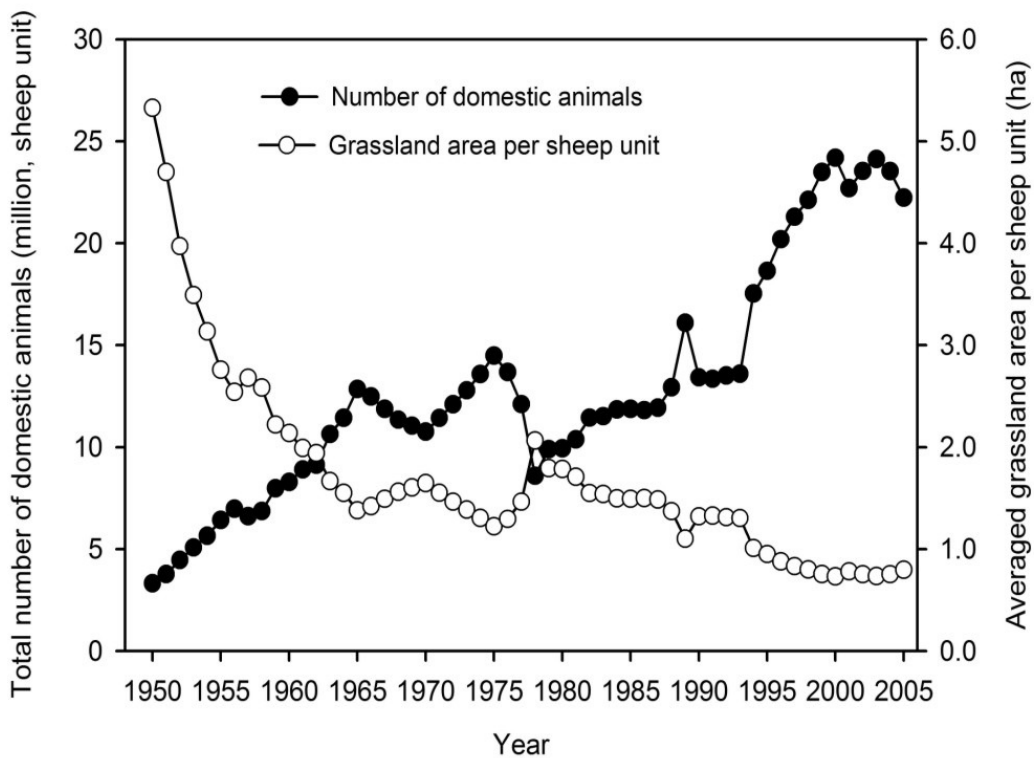
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## General Introduction

The present work was carried out in the frame of the research group 436 MAGIM (**M**atter fluxes of **G**rasslands in **I**nnner **M**ongolia as influenced by grazing) and supported by the German Research Foundation (DFG). As a Sino-German cooperation project, it was created in the context of the dramatic land use changes in the central Asian grassland ecosystems during the last decades and launched in the first operational phase during spring 2004.

In the Chinese semi-arid natural grasslands, the ecosystem carrying capacity for live-stock of was strongly exceeded for a number of reasons. The traditional nomadic live form of the Mongolian minority, which was adapted to the limited availability of natural resources, was more or less completely replaced by intensified stationary livestock farming systems between 1950 and 1970. Further, the socio-economic background of the Chinese society was based on common land, which is in many cases contradictory to sustainable land-use practises. A new established economic policy in the late 1980<sup>th</sup> allowed private profit to individual farmers, which results in drastically increased sheep, goat and cattle stocking rates. End of the last century, the people of the comparable poor Northern Chinese grasslands took the opportunity to benefit from the fast developing Chinese society and their arising giant food markets at the new economic centres at the coastal zone. In the course of this development, the average Inner Mongolian grassland area per sheep unit deceased from 5 hectares in the late 1950<sup>th</sup> to less than 1 hectare at the end of the last century (Fig. 1). This alarming trend corresponds to the total number of domestic animals, which even doubles during the last ten years of the previous century.

Overgrazing is nowadays very common on huge areas of the Chinese natural grasslands with serious consequences for the ecosystem. Studies showed that almost two third of the area is already seriously affected by land degradation (Zhou et al. 2002, Tong et al. 2004). Land degradation involves considerable decrease in primary grassland productivity and fodder quality, loss of soil organic matter and nutrients as well as severe land erosion processes (Fig. 2c). The most prominent off-side effects include severe dust storms of enormous size, not only seriously troubling the local population (Fig. 2e and 2f), but also Chinese coastal



**Fig. 1:** Development of the number of domestic animals and the grassland area per sheep unit from the 1950th until present. The Chinese Statistical Office 2006.

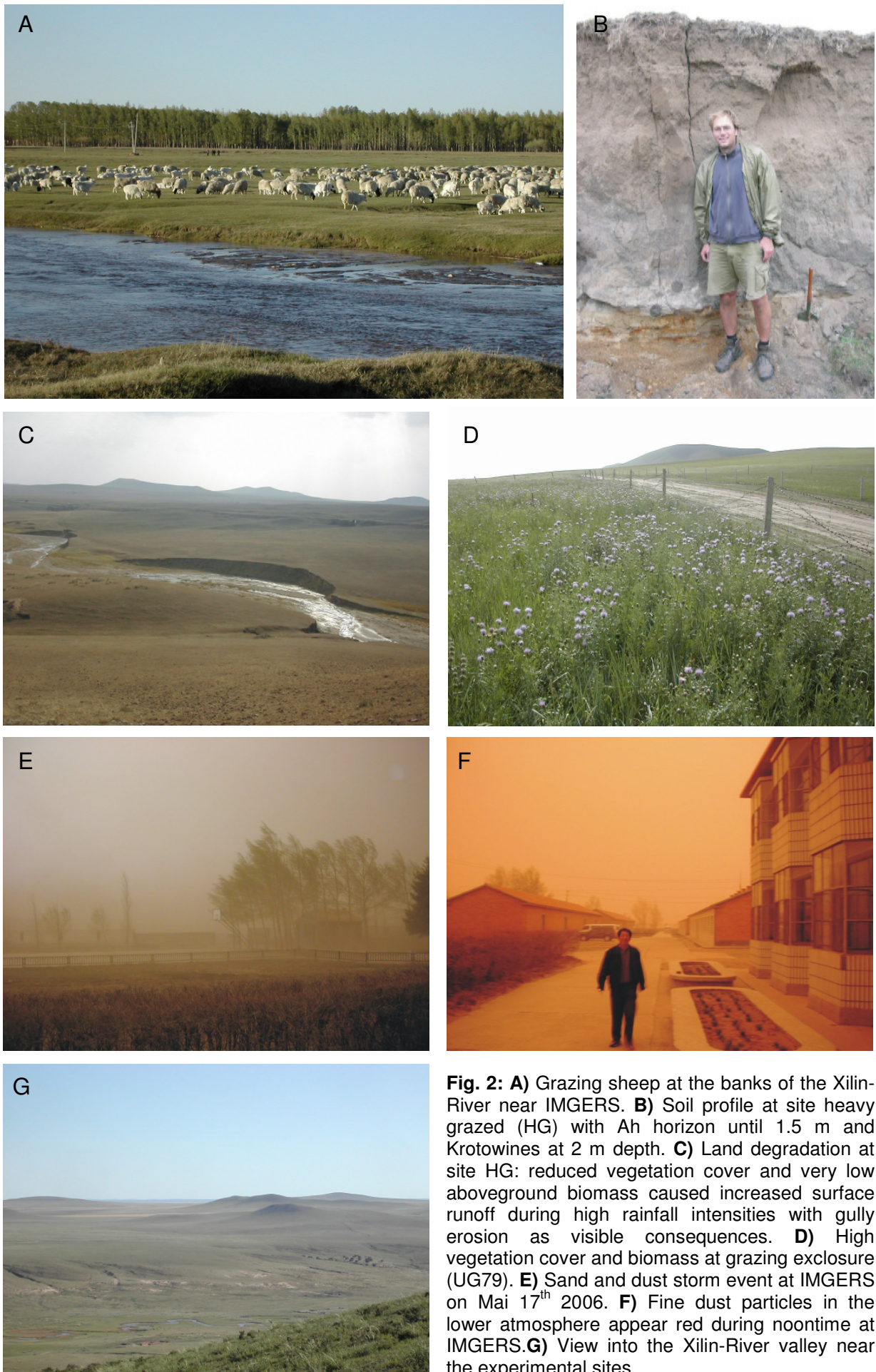
metropolis and furthermore still detectable after intercontinental atmospheric transport.

Tropical (savanna) and temperate (steppe) natural grasslands of the world together represent in total more than 40% of the continental land surface and are of enormous importance for terrestrial carbon and nitrogen stocks and matter fluxes. Within the temperate grasslands, the central Asian steppe ecosystems exceed other grasslands by far with respect to size, and thus for the proportional importance of the relevant carbon and nitrogen fluxes. In contrast to other temperate steppe ecosystems existing in northern and southern America, southern Africa and Australia, the central Asian semi-arid grasslands were much less considered and therefore underrepresented in research literature.

The North American steppe variations (short-, mixed- or long grass prairie and the different sagebrush types) reveal fundamental differences considering their response to grazing (Milchunas et al. 1988). Studies comparing different semi-arid ecosystems regarding the carbon and nitrogen cycles, but also with respect to diversity, illustrate the importance of natural precipitation gradients as well as the

basic ecosystem fertility as important variables how different grasslands respond to grazing (Milchunas et al. 1988, Milchunas and Lauenroth 1993, Olf and Ritchie 1998). But altogether, consequences for the resulting carbon and nitrogen pools, fluxes and budgets are still very variable and more understanding is needed to balance these fundamental processes for regional and global matter flux analysis. Recent studies emphasise the importance of natural climate variability, especially the precipitation pattern on semi-arid ecosystem functions (Knapp et al. 2002, Austin et al. 2004, Heisler and Weltzin 2006). For this reason and in particular that long-term effects of the recent land-use changes in northern China are still unclear, research results from the central Asian steppes are highly requested to continue and enrich the scientific discussion on the grazing-ecosystem interaction of the worlds natural grasslands.

From the 9 subprograms of the MAGIM research group, the present work represents subproject 2, which addresses in particular the carbon and nitrogen dynamics of Inner Mongolia grasslands as influenced by grazing. Project 2 is clearly subdivided into an above- and belowground carbon allocation part on the one hand (Gao 2007) and the part of nitrogen dynamics on the other hand, which is represented by the present work. The preference to the nitrogen dynamics, originates from the fact, that without doubt, this macronutrient currently dominates the discussion of grazing effects on nutrient dynamics (Mazancourt et al. 1998, Proulx and Mazumder 1998, Frank et al. 2000, Le Roux et al. 2003, Augustine and McNaughton 2004, 2006, Bakker et al. 2004, Loiseau et al. 2005, Wang et al 2006). However, within this work, crosscuts to the carbon dynamics will be done whenever essential and beneficial.



**Fig. 2:** **A)** Grazing sheep at the banks of the Xilin-River near IMGERS. **B)** Soil profile at site heavy grazed (HG) with Ah horizon until 1.5 m and Krotowines at 2 m depth. **C)** Land degradation at site HG: reduced vegetation cover and very low aboveground biomass caused increased surface runoff during high rainfall intensities with gully erosion as visible consequences. **D)** High vegetation cover and biomass at grazing enclosure (UG79). **E)** Sand and dust storm event at IMGERS on Mai 17<sup>th</sup> 2006. **F)** Fine dust particles in the lower atmosphere appear red during noontime at IMGERS. **G)** View into the Xilin-River valley near the experimental sites.





*Study sites and grazing history*

The study sites are located about 10 km SW of the IMGERS facility and carefully selected in co-operation with the Chinese Institute of Botany (IB-CAS), according to our focus on long-term grazing effects on natural grassland sites. During the field campaign, we mainly investigated three experimental sites characterized by different grazing intensities, respectively land-use history (Table 1). First, a long-term grazing enclosure (hereafter: UG79, position 43°33'10"N; 116°40'33"E, Fig. 2d), fenced in 1979 and before this lightly grazed. Second, a winter grazed site (WG, position 43°32'57"N; 116°40'04"E), representing moderate grazing intensity, where grazing occurs from October to January with a stocking rate of 1-2 sheep per ha. The winter grazing system was introduced in 1999, before that, the site was moderately grazed through out the year at same stocking rates. Third, a long-term heavily grazed site (HG, position 43°34'54"N; 116°40'37"E, Fig. 2c) located about 3 km NE of the other sites, where heavy grazing occurred for more than 30 years with 2-5 sheep per ha. Two more experimental sites were included during the vegetation analysis: A short-term grazing enclosure site fenced in 1999 and before that moderately grazed (UG99, position 43°33'05"N; 116°40'15"E) and a continuously grazed site, mostly stocked with 1-2 sheep per ha throughout the year (CG position 43°33'09"N; 116°40'02"E). Grazing history of that site prior 1999 is comparable to site WG.

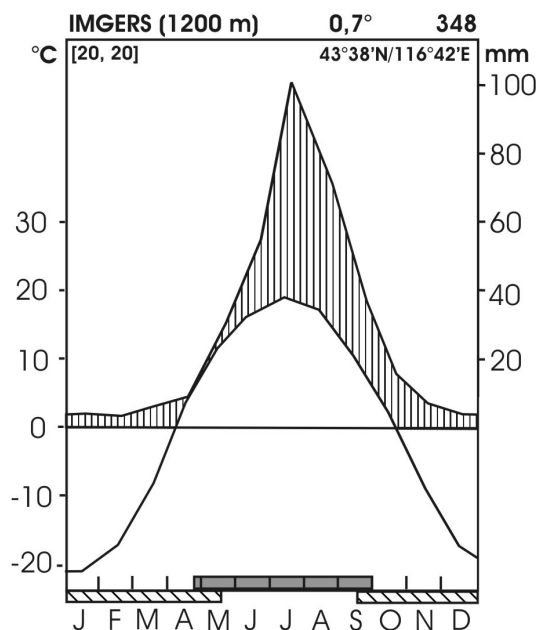
**Table 1:** The different grazing and management history, the recent management practise and stocking rates at the investigated sites of Inner Mongolia grassland ecosystem.

Name of site	Recent management	Average stocking rate	History prior 1979
1979 (UG79)	Fenced in 1979	0	Free grazing at light intensity, no degradation visible. Plant community dominated by <i>Leymus chinensis</i> . Total number of species between 40 and 50.
1999 (UG99)	Fenced in 1999	0	
Winter grazing (WG)	Grazed from October to March	1-2 sheep/ha	
Continuously grazing (CG)	Grazed through out the year	1-2 sheep/ha	
Heavy grazing (HG)	Heavy grazing through out the year	3-5 sheep/ha	Free grazing at heavy intensity due to near settlements and better water supply. Plant community with <i>Artemisia frigida</i> as one dominant species.

The grazing history of the different sites had a strong effect on plant species composition. The long-term enclosure UG79 and site HG showed a comparable large proportion of dicotyledonous plants, while site WG mainly consisted of Graminoids (see chapter 5, and appendix). Northern Chinese grasslands have a long evolutionary history of grazing (Christensen et al. 2004). The evolutionary grazing history was found to be an important factor to affect the ecosystem response to grazing disturbance (Milchunas et al. 1988). Since grazing ever occurred by native large and small mammals, the grassland is adapted to herbivore disturbance and thus might respond with faster regenerating abilities compared to grasslands with a short evolutionary grazing history in comparable climate zones (Milchunas et al. 1988, Milchunas and Lauenroth 1993, Adler and Lauenroth 2005).

#### *Climate and soils*

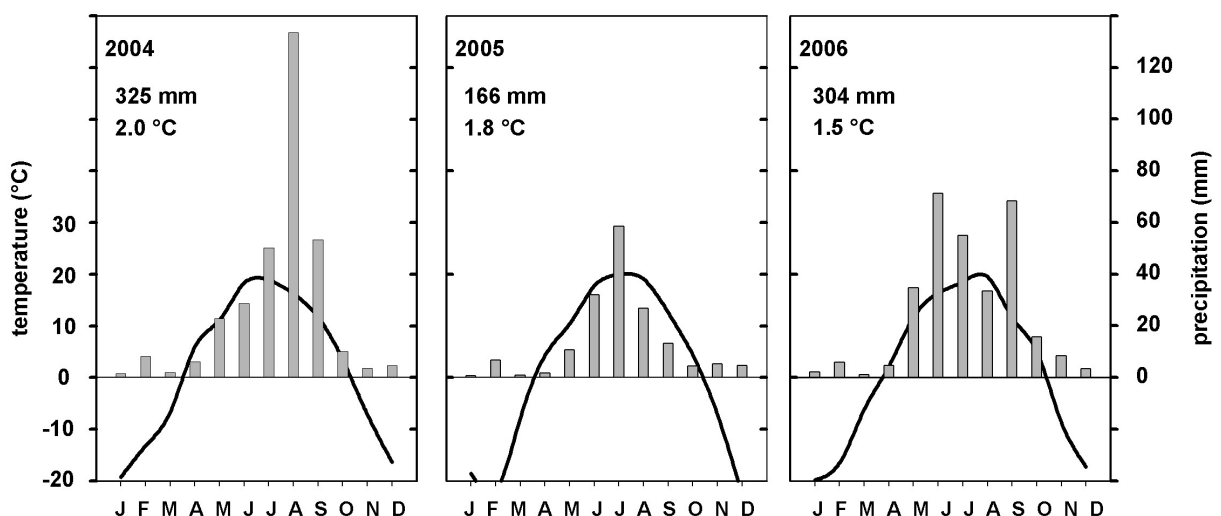
By thermic and hygric criteria, the semi-arid continental climate of temperate grasslands of the central Asian steppe ecosystems defines an annual growing season of about five months between early May and late September (Fig. 4). In this period, average monthly temperatures are above 5°C reaching a long-term



**Fig. 4:** Climate Diagram of Inner Mongolia Grassland Ecosystem Research Station (IMGERS) at 43°38'N, 116°42'E. The striped bar below illustrates the frost period, the grey bar the growing season defined as  $T_{\text{mon}} > 5^{\circ}\text{C}$ ,  $P > 2T$ , (Schultz 2000).

maximum of 19°C in July. Annual mean precipitation (about 350 mm) occurs to more than 90% between May and September with a clear summer peak in July. Due to high variability of rainfall within the growing season, water deficiency is considered as a limiting factor, especially in May and June. The winter is dry and extremely cold and the frost-free period is limited from June to August.

The data presented in this study were collected during a three growing season field campaign from 2004 to 2006. During this time the MAGIM project was faced with growing seasons of highly different rainfall amounts and distribution (Fig. 5). All three years were clearly below the long-term average precipitation of around 350 mm per year according to the long-term meteorological data recorded at IMGERS. Only the year 2004 offered more or less average conditions, with a comparable late start of the mid summer rainy season. The total rainfall amount during the growing season (May – September) was 288 mm (325 mm/ full year). The year 2005 was the driest year ever recorded by IMGERS station with only 142 mm of rainfall between May and September (166 mm/ full year). The growing season 2006 started with more precipitation than average, but left out the summer peak in July and finally reached 263 mm during the growing season (more than 300 mm/ full year) because of high precipitation in September 2006. Due to desiccated soils at the beginning of the growing season resulting from the 2005 drought, and the missing summer rainfall peak, 2006 can be considered as a dry year again with respect to grassland growth and process dynamics, but less extreme compared to 2005. The temperatures showed more or less constant values during the three years.



**Fig. 5:** Monthly mean temperatures and precipitation 2004 – 2006 measured at Inner Mongolia Grassland Ecosystem Research Station (IMGERS).

Soils are classified as Calcic Chernozems (IUSS Working Group WRB, 2006) derived from aeolian sediments above acid volcanic rocks. Chernozems have a characteristic deep Ah horizon, often reaching depth of more than 1 m. The soils started to develop under periglacial conditions, where soil organic matter turnover was strongly inhibited and thus accumulated. Numerous Krotowines, found in soil profiles in more than 2 m depth indicate intensive bioturbation (Fig. 2b). The texture of the topsoil is a sandy loam to loamy sand. Soil parameters as sampled at the experimental sites are shown in Table 2. The nitrogen, carbon and sulphur content as well as the bulk density of the topsoil increased significantly with grazing intensity (Steffens et al. 2007). The topsoil CN ratio was constant between 9.5 at site WG and 9.8 at site UG79. The pH-value of the different experimental sites was measured between 6.6 and 6.7. Significant differences in the topsoil water content were measured as well, showing significant decreasing values with increasing grazing intensity.

**Table 2:** Mean values (ANOVA results, Tukey HSD,  $p = 0.05$ ) of parameters sampled at different grazing intensities in summer 2004 at long-term grazing enclosure since 1979 (UG79), winter grazing (WG) and heavy grazing (HG) of Inner Mongolia grassland ecosystem.

Site	n	OC <sup>a</sup>	N <sub>tot</sub> <sup>a</sup>	S <sub>tot</sub> <sup>a</sup>	CN-ratio	Bulk-Dens. <sup>a</sup>	Water (0-6 cm) <sup>c</sup>	Cover <sup>d</sup>	AGB <sup>b</sup>	pH <sup>a</sup>
		(mg/g)								
HG	75	16.8 <sup>a</sup>	1.7 <sup>a</sup>	0.27 <sup>a</sup>	9.7	1.28 <sup>a</sup>	0.15 <sup>a</sup>	69 <sup>b</sup>	132 <sup>a</sup>	6.6
CG	103	22.8 <sup>b</sup>	2.4 <sup>b</sup>	0.29 <sup>a</sup>		1.17 <sup>b</sup>	0.18 <sup>b</sup>		164 <sup>b</sup>	6.6
WG	119	25.9 <sup>b</sup>	2.7 <sup>b</sup>	0.37 <sup>b</sup>	9.5	1.09 <sup>b</sup>	0.19 <sup>b</sup>	60 <sup>c</sup>	168 <sup>b</sup>	6.7
UG99	77	25.5 <sup>c</sup>	2.6 <sup>c</sup>	0.35 <sup>b</sup>		1.09 <sup>c</sup>	0.21 <sup>c</sup>		399 <sup>d</sup>	6.8
UG79	76	31.2 <sup>c</sup>	3.2 <sup>c</sup>	0.42 <sup>c</sup>	9.8	0.94 <sup>c</sup>	0.21 <sup>c</sup>	77 <sup>a</sup>	313 <sup>c</sup>	6.6

<sup>a</sup> values from topsoil 0-4 cm, OC = organic carbon.

<sup>b</sup> aboveground biomass at 1 cm, incl. dead parts.

<sup>c</sup> topsoil water content 0-6 cm, mean values of 12 measurements.

<sup>d</sup> percent of green aboveground biomass covering soil surface

*Vegetation and productivity*

The vegetation of the Xilin-River catchment is classified as a *Stipa grandis* and *Leymus chinensis* dominated grass steppe. Most of the species are perennials with their persistent organs directly at the soil surface (Hemicryptophytes) or well protected within the topsoil (Cryptophytes). A few exceptions are semi-shrubs (often the legume *Caragana microphylla*) and dwarf-shrubs (Chameaphytes), mainly represented by *Artemisia* species. Annual species are minor within the plant community, but opportunistically some (mainly *Chenopodiaceae*) can appear widespread if growing conditions are favourable. Beside the shrubs, the aboveground parts of the vegetation die at the end of the growing season in August and September. The average vegetation ground cover depends on the soil conditions and topographical position in general, but is highly variable due to growing season's precipitation amount and distribution and the land-use form. In the year 2004, at peak biomass time around mid of august, our experimental sites had a plant cover (green parts) between 77% at the long-term enclosure (UG79) and 60% at site WG. The total aboveground biomass (AGB) was characteristically increasing with decreasing grazing intensity (Table 2).

Compared to the pastures of the moist mid latitudes, the semi-arid natural grasslands are far less productive. During the growing season of 2004, we measured the highest aboveground net primary productivity (ANPP) at the long-term ungrazed site UG79 with approximately 200 g m<sup>-2</sup> dry matter per year. In European pastures 1000 – 2000 g m<sup>-2</sup> dry matter per year is a common annual net productivity. Analysing the average ANPP values for the world's temperate semi-arid grasslands, we find our ecosystem in the middle to low range. The most productive natural temperate semi-arid grasslands (North American long grass steppe) produce up to 700 g m<sup>-2</sup> dry matter per year, while the North American short grass steppe achieves only around 100 g m<sup>-2</sup> dry matter per year.

The belowground net primary productivity (BNPP) is much more difficult to measure and reliable data from semi-arid ecosystems are rare. Nevertheless authors agree, that in semi-arid grasslands belowground productivity weight out the aboveground productivity by far (Milchunas and Lauenroth 2001). Estimates reach up to more than 90% of carbon allocation to belowground. Our projects measurements using the ingrowth core method estimate a belowground allocation for carbon in our study area of about 60% (Gao et al. 2007).

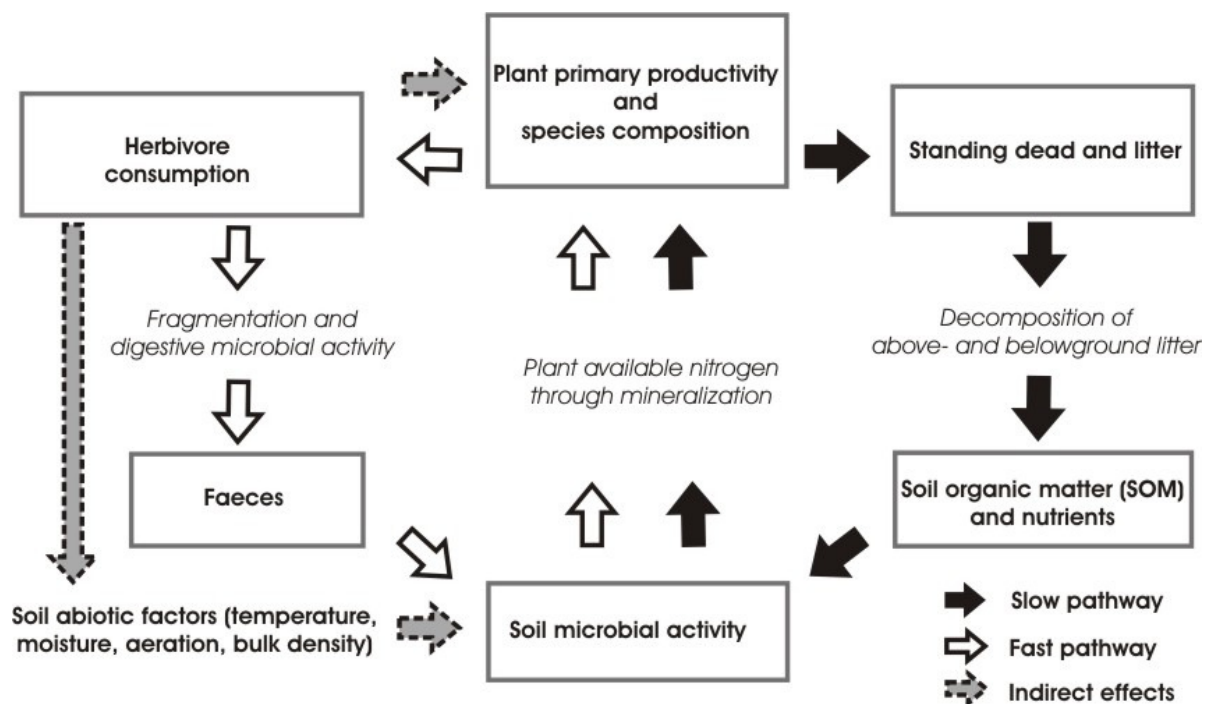
**Nitrogen dynamics as affected by grazing**

The effect of grazing on nitrogen (N) cycling is the focal point of many studies investigating plant-animal interactions (Holland and Detling 1990, McNaughton et al. 1997, de Mazancourt 1998, Bakker et al. 2004). N is of particular importance since most terrestrial ecosystems are limited by this macronutrient (Vitousek 1982, Aerts and Chapin III 2000). Therefore any effects on N dynamics are of profound significance for most ecological processes. A number of studies showed that herbivore consumption could induce improved N cycling (Ruess 1984, McNaughton et al. 1997, de Mazancourt 1998, Frank and Groffmann 1998, 2000), which can lead to compensation or even overcompensation of plant growth (Frank et al. 2002, Garibaldi et al. 2007). On the other hand studies found contradicting results, showing that grazing reduces N cycling (Pastor et al. 1993, Ritchie et al. 1998, Biondini et al. 1998, van Wijnen et al. 1999). These controversial results show that the herbivore impact on N dynamics is continuously debated and additional scientific contributions are highly requested.

In general herbivores directly or indirectly affect the N cycling. Different pathways of N turnover in grassland ecosystems are illustrated in Fig. 6. Without herbivores disturbance, the slow N cycle starts with the litter formation after plants death. Plant material enters the decomposition process, where nutrients partly are directly mineralised and plant available again. But especially under semi-arid conditions most of the material is transferred to the fraction of the soil organic matter (SOM) pool - a large compartment with slow turnover rates (Knops et al. 2002). By consumption and digestion, herbivores directly can shortcut this slow N cycling. The consumed plants are mechanical and biological treated and through the animal's faeces, nutrients are much faster released and therefore plant available again. This process results in a fast N turnover (Ruess and McNaughton 1987, Frank et al. 2000).

In contrary to the direct effects, herbivores alter nutrient cycling indirectly by changing plant species composition. A species shift is often combined with a decrease in litter quality since high palatable species, usually characterized by high N contents, are preferentially consumed (Ritchie et al. 1998, Olofsson and Oksanen 2002). Further, grazers are known to affect soil parameters such as soil bulk density (Tate et al. 2004), soil water content (Naeth et al. 1991) and temperature, which concurrently could also indirectly alter the N cycling.

Regarding the slow or fast N cycling, as well as the direct and indirect effects of herbivores (Fig. 6), it is important to know that at any time the N cycling can be accelerated or decreased. Table 3 gives an overview of mechanisms that enhance or slow down different pathways of the N cycling. The illustrated pathways of N and their possibilities of manipulation become even more complex if combined with effects of natural inner and inter annual climate variability (Biondini et al. 1998, Fernandez-Gimenez et al. 1999, Augustine and McNaughton 2006) or anthropogenic modification such as fertilization, irrigation or different land-use practises. In the following, the thesis chapters are introduced in relation to the central question, how grazing affects N dynamics in semi-arid grasslands.



**Fig. 6:** Schema of nitrogen pathways in grassland under herbivore grazing, modified after Bakker et al. 2004.



*Above- and belowground decomposition (chapter 2)*

To study the impacts of grazing on the N pathway, the above- and belowground decomposition process of plant material was analysed using the litter-bag technique. The decomposition in general is regarded as a key process of ecosystem N and C cycling (Aerts et al. 2003), and an improved understanding of grazing effects on this process is required (Semmartin et al. 2004, Olofsson et al. 2007). We analysed decomposition of uniform and site-specific plant material in order to distinguish the effects of local environmental factors and litter quality as altered by grazing. We ask for differences in the dynamics of root and shoot decay. Growing seasons of average and very low precipitation allowed us to study the effect of rainfall variability on the decomposition process.

**Table 3:** Overview of mechanisms that enhance or slow down nitrogen cycling through different pathways for semi-arid grasslands, modified and extended after Bakker et al. 2004.

	<b>Mechanisms of enhancing N cycling</b>	<b>Mechanisms of slowing down N-cycling</b>
<b>Fast pathway</b>	Microbial digestion in the herbivore gut: high temperature and moisture, fragmentation of plant material, profitable C/N ratios, trampling	Leaching, volatilisation of N from faeces and urine patches
<b>Slow pathway</b>	Increased net primary production, increased litter quality	Selective consumption of high-quality species decreases litter quality, increased aboveground standing dead and litter
<b>Indirect herbivore effects</b>	Increased soil temperature, increased soil moisture, increased soil aeration	Decreased soil temperature, decreased soil moisture, decreased soil aeration
<b>Annual climate variability</b>	Warm and moist periods within growing season	Drought, cold periods
<b>Anthropogenic effects</b>	Fertilization, irrigation	Dung collections, hay making, burning

*Nitrogen availability (chapter 3)*

The plant inorganic N availability was investigated at three different grazing intensities and three growing seasons (2004 – 2006). Ion exchange resins were installed to measure time integrated N fluxes. Additionally, we analysed conventional soil mineral N extractions. The central question was, how grazing affects plant N availability. On the other hand it is known, that climate and especially rainfall variations have considerable effects on soil biogeochemical processes (Schimel and Parton 1986, Varnamkhasti et al. 1995, Austin et al. 2004,

Augustine and McNaughton 2006). Generally, we hypothesise, that the measured N fluxes correlate with the primary plant productivity, respectively the plant N uptake. By analysing the plant N availability, we hope to address a key parameter for plant - animal interactions in this semi-arid environment. Further, the dynamic of plant available inorganic N-form (nitrate and ammonium) raise interesting questions within a cutting edge research field about abiotic effects on plant available N-form and consequences for ecosystem functions.

*Nitrogen cycling and balance (chapter 4)*

Grazing alters ecosystem functions at different levels and with different consequences. However, generally grazing causes long-term N losses (Holland and Detling 1990, Milchunas and Lauenroth 1993, Pineiro et al. 2006, Zhou et al. 2007) and thus endangers sustainable grassland productivity, and finally the existence of the particular farm. The central aspect addressed in this chapter, is the quantification of grazing effects on N fluxes and pools. Available data of the MAGIM research group related to N are brought together in order to balance in- and outputs for different grazing intensities and land-use systems. In a second step, the gains and losses are set in relation to the system internal N cycling illustrated by a summarizing flow chart.

*Grazing and plant community structures (chapter 5)*

In this study, we analysed five different grazing sites representing a gradient of land-use intensity. Canonical correspondence analysis (CCA) was applied to link information about plant species composition, abundance and coverage to variables of nutrient availability (N, P, K), topsoil water content, aboveground biomass, organic carbon, total soil N and sulphur concentration, pH and bulk density. The aim was first to find the most sensitive variable in response to long-term grazing disturbance. In a second step, we illustrate how grazing effects selected vegetation traits and plant species composition by altering different environmental parameters. We hypothesise, that nutrient, in particular N availability has strong effects on the plant species composition. From this analysis we classified the plant species into grazing response groups, from those some species might be able to function as land-use indicators.

*Slope effects on grazed semi-arid grassland ecosystem (chapter 6)*

For a considerable part of the Inner Mongolia grasslands a hilly landscape is characteristic. Since differences of vegetation and soil properties due to exposition were obvious, we investigated slope effects on grazed grassland ecosystem in more detail. Different slope exposures might represent extreme environments of this ecosystem. We hypothesize, that different radiation budgets at north and south facing slopes alter the soil water- and nutrient dynamics. In the focus was the adaptability of the vegetation as influenced by topography. Results might be valuable for extrapolation and evaluation of hilly grassland sites with respect to different land-use options. This study was designed as a master thesis work and carried out by Gong XiaoYing, China Agricultural University.

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## Effects of grazing and rainfall variability on root and shoot decomposition in a semi-arid grassland

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### Abstract

Overgrazing increasingly affects large areas of Inner Mongolia semi-arid grassland. Consequences for ecosystem functions and in particular for the decomposition as a key process of ecosystem carbon (C) and nitrogen (N) cycling are still unclear.

We studied the effects of grazing on shoot and root decomposition with the litter bag method in a long-term grazing enclosure (UG79), a moderate winter grazed site (WG) and a long-term heavily grazed site (HG). We analysed decomposition of uniform and site-specific material in order to distinguish the effects of local environmental factors and litter quality, respectively as altered by grazing. Growing seasons of average and very low precipitation allowed us to study the effect of inter annual rainfall variability on decomposition.

Local environmental factors as altered by different grazing intensities had no effect on decay rates of shoot and root dry mass. Also differences in litter quality between the grazing sites were not reflected by root decomposition dynamics. However, shoot decay was accelerated at site HG, but could not clearly be linked to litter quality parameters. In contrast to the expected grazing effects, we found a pronounced impact of rainfall variability on the decomposition dynamics. In the

year of average precipitation root decomposition was faster than shoot decay, while under dry condition the opposite was observed. Fitting a simple exponential decay function ( $x_0 = x_1 e^{-kT}$ ), we found soil moisture storage was the best parameter explaining the dynamics of root decomposition. Contrarily, shoot decomposition was unaffected by annual rainfall variability. Net nitrogen immobilization was not observed during the decay process of shoot and root at all sites.

Consequently, a separate analysis of shoot and root decay dynamics is required in order to describe carbon and nitrogen cycling in this semi-arid grassland. At site WG, the low initial litter N content decreased the nitrogen release from root and shoot material, while carbon dynamics were unaffected. When root decomposition is strongly reduced in dry periods, shoot decomposition becomes relatively more important for plant nutrient supply. Further, a stronger N limitation for plant productivity at grazed sites results from general low aboveground biomass and decreased shoot nitrogen release. Under average rainfall conditions differences between grazing sites with respect to nutrient supply through decomposition are less pronounced because of more active belowground processes. The grazing impact on C and N fluxes through decomposition of plant material likely exhibits a strong interaction with seasonal rainfall pattern.

**Key words:** steppe, nitrogen and carbon cycling, litter bag, Inner Mongolia



## Introduction

One of the world's largest C stocks is located in soils of natural temperate semi-arid steppe ecosystems of Central Asia. During the recent decades excessive overgrazing has affected large areas of these grasslands and consequences for ecosystem functions are intensively discussed (Fernandez-Gimenez and Allen-Diaz 1999, Barger et al. 2004, Christensen et al. 2004, Tong et al. 2004). The decomposition of organic matter is regarded as a key process of N and C cycling (Aerts et al. 2003) and an improved understanding of grazing effects on this process is required (Semmartin et al. 2004, Olofsson et al. 2007).

Grazing is expected to alter the decomposition process mainly via effects on local environmental parameters and litter quality (Semmartin et al. 2004). These effects include increased light penetration (Ritchie et al. 1998) and albedo as well as decreased surface roughness (Li et al. 2000) and soil water content (Naeth et al. 1991). These altered parameters can result in a strong *environment-effect* on above- and belowground decomposition rates.

Furthermore, selective grazing changes plant species composition (Milchunas and Lauenroth 1993), which concurrently may affect decomposition rates through altered litter quality (Ritchie et al. 1998, Olofsson and Oksanen 2002). Litter quality is usually the best predictor of decomposition within a particular climatic region. However, the semi-arid grasslands were found to form an exception (Aerts 1997, Parton et al. 2007). The contradicting results from several grassland studies (Murphy et al. 1998, Moretto et al. 2001, Semmartin et al. 2004, Liu et al. 2006) prompted us, to consider the litter *quality-effect* in more detail.

Since the decay of above and belowground organic matter is affected by different environmental parameters, contrasting decomposition rates between root and shoot litter are likely, but evidence for semi-arid grasslands is controversial, with roots decomposing being faster (Seastedt et al. 1992, Moretto 2001) or slower (Biondini et al. 1998) than shoot litter. Differences in decomposition dynamics between shoot and root organic matter pools would have considerable consequences for nutrient cycling and balancing.

Besides grazing effects, climatic fluctuations have significant impacts on decomposition, particularly in ecosystems of high inter- and intra-annual variability of rainfall. Results considering the interactive impact of both, grazing and climate,

on nutrient cycling are rare (Semmartin et al. 2004). Actual evapotranspiration shows usually a high correlation to decomposition intensity, when this process is investigated in regions of different climate (Meentemeyr 1978, Aerts 1997). Under the assumption that this mechanism is not only valid in a spatial dimension but also with time, we hypothesise, that the intensity of the decomposition process in semi-arid grasslands correlates stronger to variables of inter-annual climate variability than to different land-use practices.

We studied the decomposition dynamics of Inner Mongolia semi-arid grassland with root and shoot litter bags at sites of different grazing intensity in order to test the following hypothesis:

1. Grazing alters local environmental parameters, which have a significant effect on decomposition rates and related C and N matter fluxes (*environment-effect*).
2. By changing plant species composition, grazing has pronounced effects on chemical quality of the plant material and therefore decomposition rates and related C and N matter fluxes are significantly different (*quality-effect*).
3. Above- and belowground decomposition rates and associated C and N matter fluxes should be different.
4. Decomposition rates and related C and N matter fluxes are more affected by inter annual climate variability than by grazing.

## Material and Methods

### *Study area and experimental sites*

Detailed information about the study area and experimental sites are given in the general introduction part (Chapter 1). During this study, we investigated three sites characterized by different grazing intensities. A long-term grazing enclosure (hereafter: UG79, position 43°33'10"N; 116°40'33"E), a moderate winter grazing site (WG, position 43°32'57"N; 116°40'04"E), and a long-term heavily grazed site (HG, position 43°34'54"N; 116°40'37"E).

### *Litter bag incubation*

Above- and belowground decomposition was studied by the litter bag method. On 10 May 2004, litter and standing dead material from the last growing season was collected at a site protected from grazing since 5 years. This material was used as uniform material to test the *environment-effect*. Roots were taken from 0-20 cm depth and washed in order to remove soil particles. In order to study the *quality-effect*, site-specific shoot litter and roots were collected at the tree sites of different grazing intensities (HG, WG and UG79) on 15 May 2005. According to long-term data of IMGERS station, we met a very dry incubation period from Mai 2005 to Mai 2006 after average conditions in 2004 (Table 1).

**Table 1:** Selected climate parameters for the first year after incubation of shoot- and root litter bags from 2004 to 2006 and long-term annual averages as recorded by IMGERS (Inner Mongolia Ecosystem Research Station).

Parameter	30. May 2004 – 24. May 2005	25. May 2005 – 23. May 2006	Annual averages IMGERS (1983 –2006)
Mean temperature (°C)	0.9	1.9	0.7
Mean temperature Mai – Sept. (°C)	15.1	16.7	15.0
Precipitation (mm)	306	150	336
Precipitation October - April (mm)	28.5	26.4	48.2
Accumulated temperature Mai – Sept. (sum of daily means > 10°C)	2172	2524	n.a. <sup>1</sup>
No. of days with mean temp > 10°C	131	145	n.a. <sup>1</sup>

<sup>1</sup> not available

Grazing history of the different sites had a strong effect on plant species composition (Table 2). The long-term enclosure UG79 and the HG site show a large proportion of dicotyledonous plants, while the WG site mainly consists of Graminoids. Chemical characteristics of the incubated material are shown in Table 3. After drying litter and roots at 40 °C to constant weight, 8 g of litter and 6 g of root material were weighted to the nearest 0.02 g and filled into 15 x 20 cm nylon-bags with a mesh size of 1.25 mm. On May 30, 2004, the first series of 264 root and shoot litter bags was started. Root bags were buried in the main rooting zone at 15 cm depth and shoot bags were fixed at soil surface. In the first year, litter and root bags were harvested every 2-3 weeks (8 sampling dates) to obtain information about seasonal dynamics of decomposition and thereafter less frequent with 3 and 2 samplings in 2005 and 2006, respectively. On 22 May 2005 the second series with site-specific material was started with 216 root and shoot litter bags. This series was sampled 5 times in 2005 and 4 times in 2006. Bags were collected with four replicates in a minimum distance of 20 m to each other at each harvest date. Before weighting, bags were dried for 48 h at 85°C and cleaned to remove extraneous material. A sub-sample of each bag (shoot and roots) was taken to determinate the loss-on-ignition (muffle furnace 600°C, 4h). The value was used for the correction of dry weight caused by soil contamination.

**Table 2:** Contribution of major plant species to bulk litter at different grazing intensities (HG = heavy grazing, WG = winter grazing, UG79 = enclosure since 1979) and a control site (uniform) used in litter bag experiments in Inner Mongolia grassland ecosystem. Data based on visual estimates of ground coverage in % ( $\pm$  S.E.) of green biomass sampled at 1 m<sup>2</sup> in August 2004.

Species	Uniform (n = 77)	HG (n = 75)	WG (n = 119)	UG79 (n = 76)
<i>Stipa grandis</i>	25.3 ( $\pm$ 1.5)	2.2 ( $\pm$ 0.6)	35.0 ( $\pm$ 1.7)	16.0 ( $\pm$ 1.1)
<i>Leymus chinensis</i>	17.6 ( $\pm$ 1.4)	2.4 ( $\pm$ 0.4)	17.4 ( $\pm$ 1.1)	7.2 ( $\pm$ 1.0)
<i>Achnatherum sibiricum</i>	10.4 ( $\pm$ 0.8)	0	8.5 ( $\pm$ 1.0)	16.3 ( $\pm$ 1.1)
<i>Carex spec.</i>	17.9 ( $\pm$ 1.0)	18.2 ( $\pm$ 1.1)	14.7 ( $\pm$ 0.7)	10.1 ( $\pm$ 1.0)
<i>Cleistogenes squarrosa</i>	8.0 ( $\pm$ 0.8)	18.0 ( $\pm$ 1.1)	13.3 ( $\pm$ 1.5)	7.0 ( $\pm$ 0.9)
<i>Potentilla acaulis</i>	0.2 ( $\pm$ 0.1)	25.9 ( $\pm$ 3.5)	0.3 ( $\pm$ 0.2)	3.5 ( $\pm$ 0.9)
<i>Artemisia frigida</i>	0.1 ( $\pm$ 0.1)	15.9 ( $\pm$ 1.3)	0	2.5 ( $\pm$ 0.6)

**Table 3:** C, N (n = 6) and lignin (n = 3) content (mean  $\pm$  s.e.) of initial shoot and root material from different grazing sites (HG = heavy grazing, WG = winter grazing, UG79 = enclosure since 1979 and Uniform) used for litter bag experiments started in 2004 and 2005 in Inner Mongolia grassland ecosystem. Significant differences are indicated by different letters (Tukey's HSD,  $p = 0.05$ ).

Site	C (% DM)	N (% DM)	C/N	Lignin (% DM)
<b>Shoot</b>				
<b>HG</b>	43.8 ( $\pm 0.1$ ) <sup>a</sup>	0.85 ( $\pm 0.02$ ) <sup>b,c</sup>	52	6.3 ( $\pm 0.5$ ) <sup>ab</sup>
<b>WG</b>	44.5 ( $\pm 0.1$ ) <sup>a</sup>	0.50 ( $\pm 0.01$ ) <sup>a</sup>	90	7.1 ( $\pm 0.01$ ) <sup>b</sup>
<b>UG79</b>	43.8 ( $\pm 0.2$ ) <sup>a</sup>	0.77 ( $\pm 0.03$ ) <sup>b</sup>	58	5.0 ( $\pm 0.1$ ) <sup>a</sup>
<b>Uniform</b>	44.1 ( $\pm 0.2$ ) <sup>a</sup>	0.87 ( $\pm 0.01$ ) <sup>c</sup>	49	5.5 ( $\pm 0.2$ ) <sup>a</sup>
<b>Root</b>				
<b>HG</b>	45.8 ( $\pm 0.2$ ) <sup>b</sup>	1.04 ( $\pm 0.03$ ) <sup>b</sup>	44	23.0 ( $\pm 1.4$ ) <sup>a</sup>
<b>WG</b>	35.7 ( $\pm 1.3$ ) <sup>a</sup>	0.83 ( $\pm 0.02$ ) <sup>a</sup>	43	20.0 ( $\pm 1.2$ ) <sup>a</sup>
<b>UG79</b>	37.8 ( $\pm 1.1$ ) <sup>a</sup>	1.01 ( $\pm 0.06$ ) <sup>b</sup>	38	20.4 ( $\pm 0.3$ ) <sup>a</sup>
<b>Uniform</b>	46.0 ( $\pm 0.4$ ) <sup>b</sup>	1.13 ( $\pm 0.04$ ) <sup>b</sup>	41	19.5 ( $\pm 0.5$ ) <sup>a</sup>

### *Chemical analysis*

C and N content of shoot litter and root were analysed by dry combustion with CN elemental analyzer (EA 1108, Italy). Acid detergent lignin (ADL) was analysed in the initial bulk shoot and root material by sequential extractions (Ankom 200 Fiber Analyzer) after van Soest (1963).

### *Statistical analysis*

Data were analysed with the statistical software SAS Version 8.2. Litter decay was analysed with a two-factorial model with grazing sites and root and shoot material as fixed effects. Comparison of means was based on Tukey's HSD test. Since processes affecting shoot and root decomposition were different, we additionally analysed differences between sites using one way ANOVA. We are aware that, due to our focus on long-term grazing effects on natural field sites, treatments were pseudo-replicated. We did our best to choose representative areas within this grassland in co-operation with the Chinese Institute of Botany.

Decomposition rate ( $k$ ) was estimated by the negative exponential decay function  $X/X_0 = e^{-kt}$  (Olson 1963), where  $X$  is the amount remaining of an initial quantity  $X_0$ . We calculated  $k$ -values for both, the first year decay and the entire period of three growing seasons (803 days) for the *environment-effect* experiment and two growing seasons (447 days) for the *quality-effect* experiment. As we started our series in May, first year  $k$ -values were calculated until May of the following year.  $K$ -values of the entire period were adjusted to one year for comparability.

In a second step we fit exponential decay functions with Sigma-Plot (version 9.0), where decomposition rates were subjected to regression analysis against climate data, which were monitored by IMGERS (Inner Mongolia Grassland Ecosystem Research Station).

## Results

### *Environment-effects on decomposition*

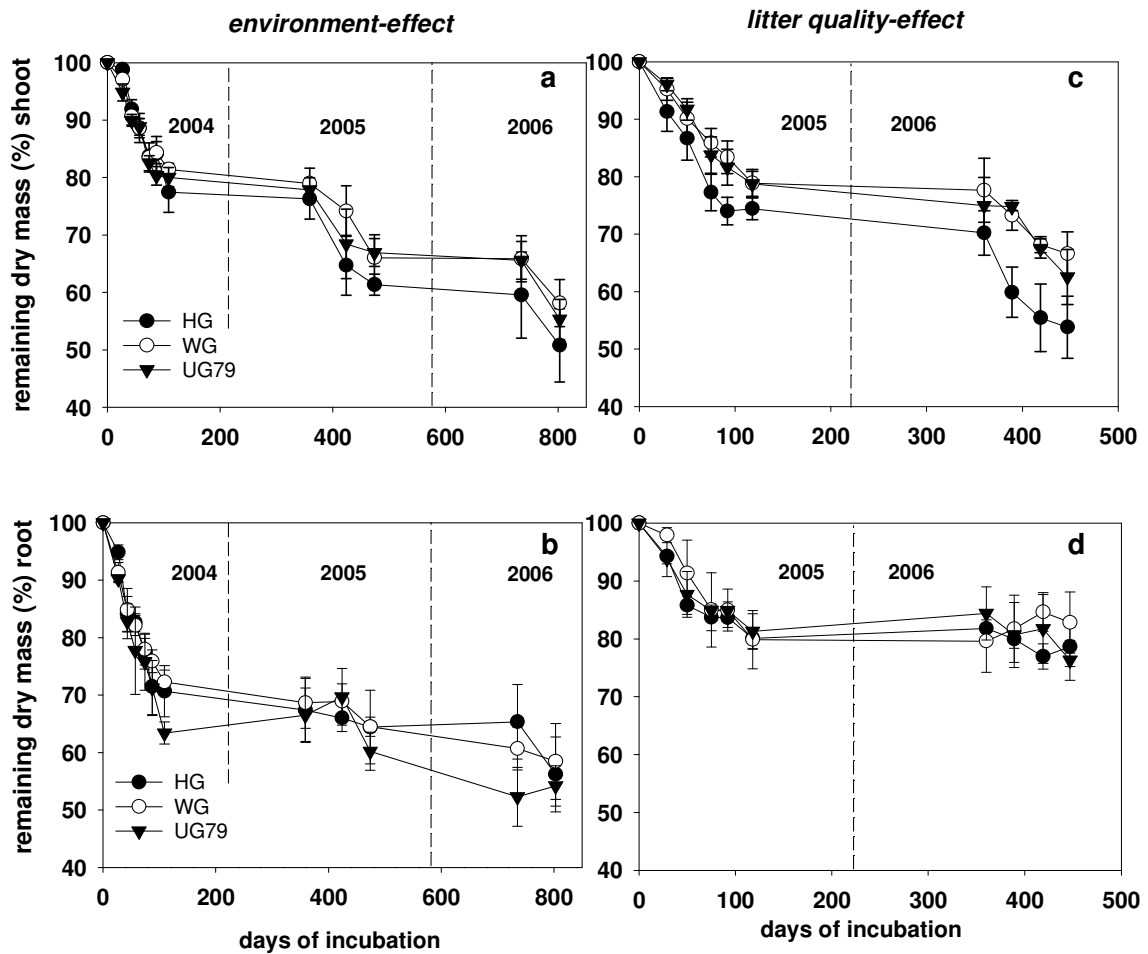
The grazing sites exhibited no significant difference in decomposition of uniform shoot litter after the first year of incubation (Table 4, Fig. 1a). We measured a remaining shoot dry mass between 76.3% and 78.9% and corresponding *k*-values between 0.24 and 0.28. At the last sampling date, 803 days after incubation, we still found no significant differences between the sites. At the four last harvest dates an increase of litter decay rate was observed at site HG compared to sites UG79 and WG (Fig. 1a). However, this effect was only significant ( $p < 0.05$ ) at two dates. No *environment-effect* was found for decomposition of uniform root material after the first year (Table 4, Fig. 1b). Remaining dry mass varied from 66.5% (UG79) to 68.7% (WG), resulting in *k*-values between 0.38 (WG) and 0.41 (UG79). Considering the long-term effects, no significant differences in root decomposition rates were found at the last harvest date (after 803 days).

**Table 4:** Shoot and root decomposition and fractions of initial N and C of *uniform* material incubated at three sites of different grazing intensities in Inner Mongolia semi-arid grassland ecosystem. Data reported as % remaining dry mass, *k*-values calculated as  $-\ln(X/X_0) = kt$  (Olson 1963).

	Site <sup>1</sup>	Remaining dry mass		K-values		Fraction of initial	
		first year	last harvest <sup>2</sup>	first year	last harvest <sup>2</sup>	N	C
shoot	HG	76.3	50.8	0.28	0.31	0.77	0.75
	WG	78.9	58.1	0.24	0.25	0.82	0.78
	UG79	77.8	55.3	0.26	0.27	0.86	0.78
root	HG	67.4	56.2	0.40	0.26	0.68	0.72
	WG	68.7	58.5	0.38	0.24	0.69	0.73
	UG79	66.5	57.7	0.41	0.25	0.75	0.71
	<i>Site</i>	n.s.	n.s.			n.s.	n.s.
	<i>Root/Shoot</i>	.0001	n.s.			.001	.05
	<i>S*R x S</i>	n.s.	n.s.			n.s.	n.s.

<sup>1</sup> HG: heavy grazing, WG: winter grazing; UG79: ungrazed since 1979

<sup>2</sup> harvest of litter bags after 803 days, values are adjusted to 1 year ( $k/803*365$ ) to enable comparison.



**Fig. 1 a-d:** Above- and belowground decomposition as influenced by different grazing sites: heavy grazing (HG), winter grazing (WG) and ungrazed since 1979 (UG79) in Inner Mongolia semi-arid grassland ecosystem. Data from litter bag experiments, where decomposition is expressed as % remaining dry mass in relation to days of incubation. Litter bags with uniform shoot and root material (*environment-effect* study) were incubated at 30.05.2004. Litter bags with site specific shoot and root material (*litter quality-effect* study) were incubated at 22.05.2005. Last harvest for both litter bag experiments was 12.08.2006. Mean values of four replicates with standard error bars.

### *Quality-effects on decomposition*

Considering the chemical composition of the initial aboveground site-specific litter, the N content was between 0.77% and 0.85% at sites HG and UG79, while site WG exhibited a clearly reduced N content of 0.50% (Table 3). The carbon content at all sites was more or less constant, which resulted in a strongly increased C/N



ratio of 90 at site WG. Root N content was also reduced at site WG (0.83%) compared to that at sites UG79 (1.01%) and HG (1.04%), while the C/N ratio was less affected compared to aboveground litter. Lignin content of the shoot material was highest at site WG, but not significantly different to that at site HG (Table 3). Acid detergent lignin content of roots was about three to four times higher than of shoot material.

Decomposition of site-specific shoot material at site HG was significantly ( $p = 0.05$ ) faster compared to the other investigated sites after the first year (Table 5) and at eight of ten sampling dates over the entire incubation period (Fig. 1c). Accordingly, after one year,  $k_{shoot}$  was higher at site HG (0.36) compared to that of sites WG (0.26) and UG79 (0.29). Contrarily to shoot material, site-specific roots showed no significant site difference in decomposition rates after the first year and the last harvest date (Table 5, Fig. 1d). After the first year, between 79.6% and 84.4% of dry matter remained in the litter bags resulting in values of  $k_{root}$  between 0.17 and 0.23. The remaining dry masses at the last harvest date were between 76.3% ( $k = 0.22$ ) at site UG79 and 82.8% ( $k = 0.15$ ) at site WG.

#### *Differences between above- and belowground decomposition and inter-annual variability*

The decomposition rate of roots ( $k_{root} = 0.38 - 0.41$ ) incubated in May 2004, a year of average rainfall amount and distribution was significantly ( $p = 0.001$ ) faster compared to shoot decomposition ( $k_{shoot} = 0.24 - 0.28$ , Table 4). Contrarily, first year decomposition rate of roots incubated one year later in May 2005, characterized by very low rainfall, was significantly slower ( $p = 0.001$ ,  $k_{root} = 0.17 - 0.23$ ) compared to shoot decomposition ( $k_{shoot} = 0.26 - 0.36$ ). After an additional year of low rainfall (2006), this trend was even stronger ( $p = 0.001$ ) with values of  $k_{root}$  ranging from 0.15 - 0.22 compared to  $k_{shoot}$  values of 0.33 - 0.51). While root decomposition rates were strongly reduced or generally low during periods of low rainfall, shoot decomposition rates were more or less constant between the years (Table 4, 5).

**Table 5:** Shoot and root decomposition and fractions of initial N and C of *site-specific* material incubated at three sites of different grazing intensities in Inner Mongolia semi-arid grassland ecosystem. Data reported as % remaining dry mass, *k*-values calculated as  $-\ln(X/X_0) = kt$  (Olson 1963). Latin letters indicate significant differences between sites pooled over shoot and root material. Greek letters indicate significant differences between sites based on one-way ANOVA separated for root and shoot.

	Site <sup>1</sup>	Remaining dry mass		K-values		Fraction of initial	
		first year	last harvest <sup>2</sup>	first year	last harvest <sup>2</sup>	N	C
shoot	HG	70.2 <sup>α</sup>	53.8 <sup>αα</sup>	0.36	0.51	0.80 <sup>αα</sup>	0.71 <sup>α</sup>
	WG	77.5 <sup>β</sup>	66.6 <sup>ββ</sup>	0.26	0.33	1.06 <sup>βα</sup>	0.77 <sup>α</sup>
	UG79	74.9 <sup>β</sup>	62.5 <sup>αββ</sup>	0.29	0.38	0.78 <sup>αα</sup>	0.79 <sup>α</sup>
root	HG	82.2 <sup>α</sup>	78.4 <sup>αα</sup>	0.20	0.20	0.72 <sup>αα</sup>	0.76 <sup>α</sup>
	WG	79.6 <sup>α</sup>	82.8 <sup>βα</sup>	0.23	0.15	0.83 <sup>βα</sup>	0.90 <sup>α</sup>
	UG79	84.4 <sup>α</sup>	76.3 <sup>αβ</sup>	0.17	0.22	0.69 <sup>αα</sup>	0.86 <sup>α</sup>
	<i>Site</i>	n.s.	0.004			0.004	n.s.
	<i>Root/Shoot</i>	0.001	0.001			0.012	0.047
	<i>S*R x S</i>	n.s.	n.s.			n.s.	n.s.

<sup>1</sup> HG: heavy grazing, WG: winter grazing; UG79: ungrazed since 1979

<sup>2</sup> harvest of litter bags after 447 days, values are adjusted for one year

### *Predicting shoot and root decomposition*

We fitted a simple exponential decay function ( $x_1 = x_0e^{-kt}$ ) to selected environmental parameters to find the best prediction of above- and belowground decomposition for the *environment-effect* litter bag experiment. Shoot decomposition as a function of accumulated number of days of incubation gave  $r^2$  values of 0.64 - 0.80 (Table 6, Fig. 2a). In order to consider the long winter period, when nearly no decomposition occurred (Fig. 1a), a fit of decay by using accumulated daily mean temperature  $> 10^\circ\text{C}$  increased  $r^2$  values to 0.91 - 0.93 (Table 6, Fig. 2b). This correlation was higher compared to temperature threshold values  $< 10^\circ\text{C}$ , while higher values of temperature did not increase the coefficient of determination (data not shown). The alternatively used accumulated precipitation and accumulated soil water content resulted in comparably good

fittings (Table 6, Fig. 2c-d). Root decay was poorly predicted by accumulated days of incubation ( $r^2 = 0.10 - 0.24$ ). Prediction was improved by using the accumulated temperature ( $r^2 = 0.56 - 0.65$ ) or accumulated rainfall ( $r^2 = 0.63 - 0.74$ ) approach and was best predicted ( $r^2 = 0.73 - 0.83$ ) by accumulated soil water content during the frost-free growing season with a threshold value  $> 10\%$  soil water content (Table 6, Fig. 2e-h).

**Table 6:**  $r^2$  values of exponential decay function ( $x_0 = x_1 e^{-kT}$ ) in relation to time of incubation and selected environmental variables (see also Fig. 2 a – h). Higher  $r^2$  values indicate better decay process prediction of the model. Data from above- and belowground litter bag experiment of three growing seasons (May 2004 to August 2006) in semi-arid grassland of Inner Mongolia.

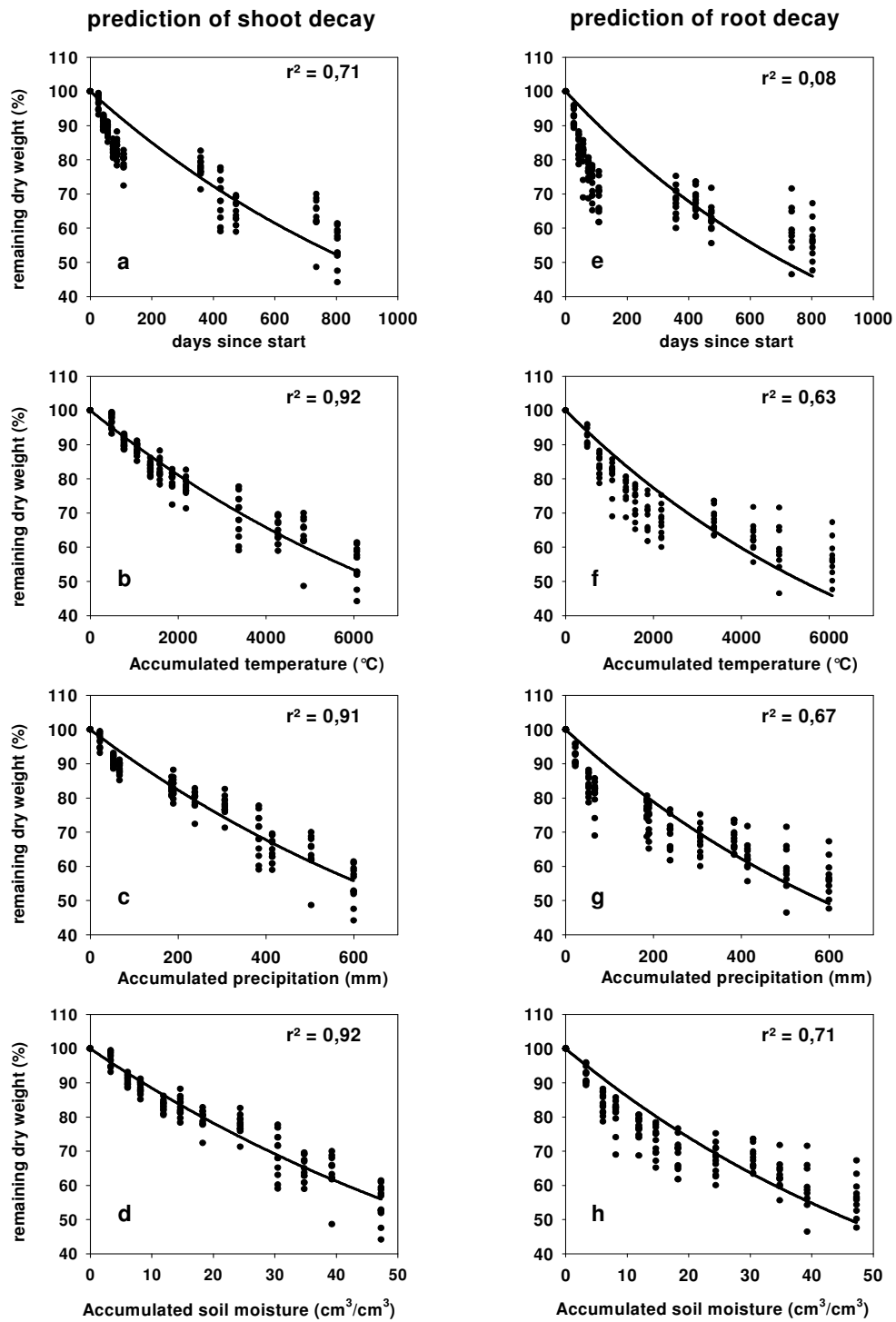
Variable of X-axis	$r^2$ aboveground <sup>1</sup>	$r^2$ belowground <sup>1</sup>
Time (days)	0.64 – 0.80	0.10 – 0.24
Accumulated temperature ( $>10\text{ }^\circ\text{C}$ )	0.91 – 0.93	0.56 – 0.65
Accumulated precipitation	0.91 – 0.92	0.63 – 0.74
Soil water content ( $> 10\text{ vol}\%$ )	0.89 – 0.90	0.73 – 0.83

<sup>1</sup> range for three different grazing sites (HG, WG and UG79)

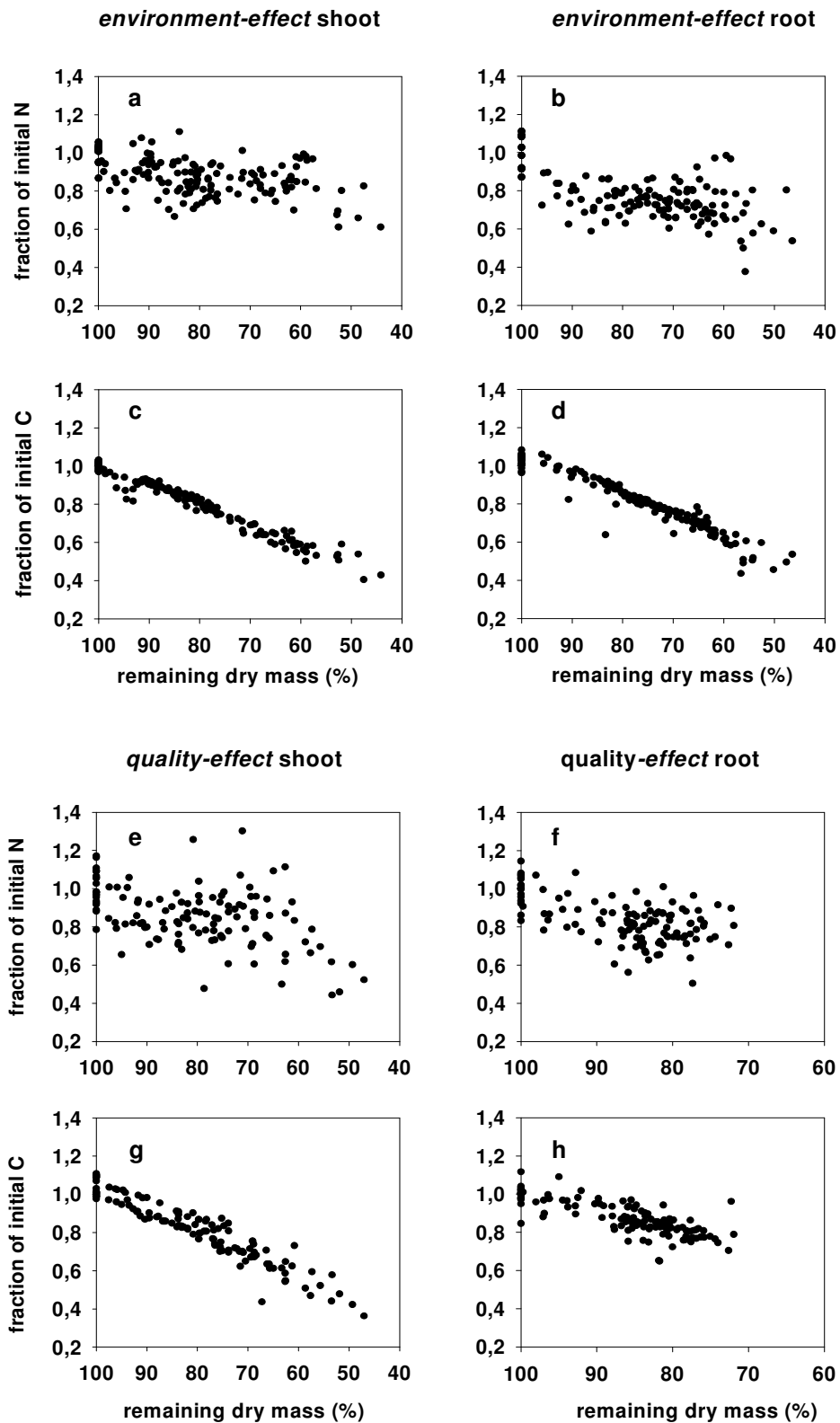
### *Carbon and nitrogen release*

The C content of remaining litter mass showed constant proportions to the initial C content and carbon release showed a high correlation with dry matter loss ( $r^2_{\text{root}} = 0.95$ ,  $r^2_{\text{shoot}} = 0.93$ , Fig. 3c, d, g, h). Consequently, site differences in C release rates were equivalent to the results presented for dry mass loss (see above, Table 4, 5).

The fraction of initial N of above- and belowground material of the *environment-effect* experiment was not different between the three sites after the first year (Table 4). The *quality-effect* experiment, however, revealed significantly increased remaining fractions of initial N at site WG compared to the other treatments if statistical analysis is pooled over root and shoot values (Table 5). The comparison of remaining N fractions further revealed some differences in above- and belowground decomposition dynamics. In general, no net N immobilization was observed (Fig. 3a, b, e, f). But, while roots released N almost proportional in relation to the remaining dry weight, we found that N was relatively enriched in aboveground biomass during the decay process.



**Fig. 2a-h:** Fitted exponential decay function ( $x_0 = x_1 e^{-kx}$ ) by adjusting selected environmental parameters (x-axis) to find the best prediction of decomposition of above- and belowground decomposition. Based on combined data from three different grazing intensities.



**Fig. 3 a-h:** Fractions of initial N and C in relation to remaining dry mass measured in litter bags incubated at semi-arid grassland ecosystem of Inner Mongolia from May 2004 until August 2006. Results shown for *environment* and *litter quality* effect experiments. The remaining fraction of initial N and C is shown in relation to remaining dry mass of above and belowground decomposition.

## Discussion

### *Environment-effects on decomposition*

Due to biomass consumption and trampling effects, grazing increases soil bulk density (Tate et al. 2004), albedo and surface roughness (Li et al. 2000) and reduces topsoil water content (Naeth et al. 1991). Site UG79 had higher aboveground biomass (data not shown), higher soil water content (Zhao et al. 2007) and lower soil bulk density (Steffens et al. 2007) compared to sites WG and HG. We therefore presumed an *environment-effect* on decomposition dynamics, but root and litter decomposition revealed no pronounced difference between sites. This is in agreement with results of Knops et al. (2001), who found no location effect on decomposition rate in grassland ecosystems. At site HG (Table 4, Fig. 1a), we found a trend of increased litter decomposition at two of ten harvest dates, but existing differences in environmental parameter between grazing sites were not strong enough to have pronounced effects on root and litter decomposition. For this reasons *hypothesis 1* is rejected, *environment-effects* as a result of grazing have no significant influence on decomposition dynamics in this semi-arid grassland ecosystem.

### *Quality-effects on decomposition*

Selective grazing affects plant species composition (Milchunas and Lauenroth 1993) and thereby chemical composition of the decomposable plant material. Clear differences in species composition between the sites were found in this study (Table 2), and chemical analysis showed a strongly reduced N content of shoot litter at site WG compared to sites UG79 and HG (Table 3). Differences in decomposition rates resulting from different N content of selected species were reported by Moretto et al. (2001) and Liu et al. (2006) for semi-arid grassland ecosystems. However, the quality differences of shoot litter in our study were not reflected in decomposition dynamics. Shoot decay at site HG was faster compared to other sites, but the chemical composition of shoot litter at site HG was similar to that of the slower decomposing material of site UG79 (Table 3). Aboveground litter at site UG79 exhibited similar decay rates compared to site WG, but had a different chemical composition. These findings agree with studies of Aerts (1997), Parton et al. (2007) and Knops et al. (2001), who reported a poor correlation

between decomposition rates and initial chemical litter composition in semi-arid grasslands. However, accelerated shoot decomposition at site HG likely indicates interactive effects between site microclimate (e.g., radiation) and site specific plant species (e.g. *Potentilla acaulis*, *Artemisia frigida*, *Cleistogenes squarrosa*), which could not be explained by chemical parameters recorded in our study.

Root chemistry is considered as the primary controller of root decomposition (Silver and Miya 2001). Root material in this study differed for N content, but not for C/N ratio and lignin content (Table 3) and we did not find pronounced differences in root decay rates. Considering the small differences in quality of root material between the sites, we could not confirm root quality as a factor affecting the decay process, which might be different when quality characteristics are more distinct. We conclude that *hypothesis 2* is rejected.

#### *Differences between root and shoot litter decomposition*

The number of studies which combined information about shoot and root litter decomposition is limited. For semi-arid grassland ecosystems, belowground productivity usually outweigh aboveground production by far (Milchunas and Lauenroth 2001). Therefore the investigation and comparison of both, shoot and root decomposition is essential to understand and quantify matter fluxes of grassland ecosystems. Studies that investigated root decomposition of semi-arid grassland ecosystems reported first year  $k_{root}$  values of 0.2–0.3 (Moretto et al. 2003), 0.26 (Gill and Burke 2002), and 0.20–0.42 (Semmartin et al. 2004), which is similar to the range of  $k_{root}$  values we observed (0.17–0.41). First year shoot decay values ranged between 0.21–0.32 (Pastor et al. 1987), and 0.21–0.42 (Moretto et al. 2003), which also matches the  $k_{shoot}$ -values we found (0.24–0.36). While in some studies faster root decay compared to litter decay was reported (Seastedt et al. 1992, Moretto 2001), others found lower root decay rates (Biondini et al. 1998, Vivanco and Austin 2006). Remarkably, our two litter bag experiments delivered inconsistent results for the first year  $k$ -values for root and shoot decomposition. Even though lignin content was 3-4 times higher in roots (Table 3), we measured faster root than shoot decomposition in the period of 2004–2005, but higher shoot litter decomposition in 2005 and 2006. As a conclusion, we can confirm *hypothesis 3* for differences in decomposition dynamics of root and shoot material. These

dynamics are triggered by inter-annual differences most obviously originating from climatic variability (see below).

#### *Differences of decomposition between the years*

Differences in root decomposition between years correlated with precipitation pattern showing a high  $k_{root}$  in a year of average precipitation and low  $k_{root}$  values during a drought period. In contrast to the conclusions of Silver and Miya (2001), our results reveal that climate factors (precipitation) control belowground decomposition dynamics to a major extent in the semi-arid grasslands of Inner Mongolia. The effect of rainfall on shoot decomposition was obviously limited and we could not detect a significant reduction of litter decay during the dry conditions in 2005 and 2006. The dynamics of aboveground decomposition were well explained by cumulative temperature with a simple temperature threshold value of 10° C (Fig. 2b). Belowground decomposition, however, was best explained by the more complex pattern of soil water content (Fig. 2h). The best fitting with the simple exponential decay function ( $x_1 = x_0e^{-kt}$ ) still resulted in an underestimated decay dynamic in the first year and an overestimated decomposition in the following dry periods, indicating the limits of simple model under highly variable environmental conditions. Accumulated precipitation, which equals more or less the actual accumulated evapotranspiration (AET) values for this semi-arid ecosystem, surprisingly failed to deliver higher accuracy to predict aboveground decomposition compared to temperature (Fig. 2c). This indicates the possibility of photodegradation to control aboveground decomposition in semi-arid ecosystems as suggested by Austin and Vivanco (2006). Soil moisture generally follows the precipitation pattern during the growing season, but rainfall gave lower correlation to root decay (Table 6, Fig. 2g). The advantage of soil water content values is to smooth out extreme rainfall events, which obviously contributed less to the decomposition rate as their intensity would predict. Reasons can be found in possible water losses by drainage and surface runoff.

Decomposition rates were more affected by inter-annual climate (rainfall) variability than by grazing, which confirms *hypothesis 4* for the belowground part. The aboveground decomposition was more or less constant under the environmental conditions we met during our studying period ranging from average



precipitation (2004) to very dry conditions (2005 and 2006). Thus, for further investigations of the decomposition dynamics of semi-arid ecosystems, it is necessary to split above and belowground processes. Precipitation or better soil water storage may be used as (simple) indicators to predict or model the variable decomposition rates of roots in semi-arid grasslands.

### *Carbon and nitrogen fluxes*

The C content (%) of shoot and root bags remained almost constant with increasing incubation period and thus carbon release is highly correlated with dry matter loss (Fig. 3 c, d, g, h). It further indicates the accuracy of ash correction procedure (see methods). No net N immobilisation was found during the decomposition experiment (Fig. 3 a, b, e, f), which was different to observations in many other ecosystems but similar to semi-arid grasslands (e.g. Seastedt et al. 1992 and Parton et al. 2007). The high shoot C/N ratio at site WG (mainly graminoids) resulted in significant higher remaining N fractions during the decay process compared to the other sites, which prolonged N turn-over times compared to heavily grazed sites and long-term enclosures.

The reduced N release in relation to dry matter loss of shoot litter indicates a relatively slow N mineralisation rate, whereas belowground N release was fast at the beginning of the decomposition process but decelerates under dry conditions and matched the proportion of dry matter loss after three growing seasons (Fig. 3a, b). In years of high aboveground biomass productivity, comparably high proportions of N will be locked in shoot litter during the following growing seasons, especially at sites with high proportions of graminoids. Since the majority of N supply for net primary productivity is provided by root decay, we conclude minor grazing impacts on N cycling during years with balanced soil moisture conditions. Dry seasons, however, strongly decreased belowground mineralisation and aboveground mineralisation became more important for N supply. Under heavy grazed conditions with reduced aboveground litter, N limitation can be expected under such conditions. Thus, the grazing impact on C and N fluxes through decomposition of plant material likely exhibits a strong interaction with seasonal rainfall pattern.

### *Conclusions*

Grazing effects on the decomposition process of above and belowground plant litter are limited. Neither *environment-effects* (local environmental parameters as altered by grazers) nor *quality-effects* (chemical litter quality changed by selective grazing) were observed to affect the decay rates of shoot litter and roots. The accelerated shoot decomposition at the heavily grazed site could not clearly be linked to *environment-* or *quality-effects*. In place of grazing, differences in decay rates over time and between above- and belowground matter could be explained by inter annual rainfall variability. Root decomposition was strongly affected by soil moisture, whereas aboveground decomposition was more or less independent of water availability. Consequently a separate analysis of shoot and root decay dynamics is required for semi arid grasslands.

With respect to C and N cycling, we found a decreased N release for root and shoot material at a medium land-use intensity as represented by a winter-grazed site. N release rate of shoot litter was generally reduced compared to that of roots, but a net N immobilization was not observed during the litter bag study. Aboveground decomposition becomes relatively more important for plant N supply, when root mineralisation is strongly reduced during dry periods. Consequently enhanced N shortage is expected at grazed sites, where aboveground litter is characteristically low and N release reduced. Under average rainfall conditions differences between grazing sites with respect to nutrient supply through decomposition are less pronounced because of more active belowground processes. Thus, the grazing impact on C and N fluxes through decomposition of plant material likely exhibits a strong interaction with seasonal rainfall pattern.

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## Nitrogen availability as affected by grazing and variable rainfall in a semi-arid grassland

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### Abstract

In this case study of a semi-arid grassland ecosystem, we ask for the impact of grazing and rainfall variability on plant available inorganic Nitrogen (N). Ecosystem net primary productivity (NPP) and plant N uptake was linked to inorganic N availability. The availability of inorganic N-form ( $\text{NO}_3^-$ ,  $\text{NH}_4^+$ ) was analysed in the context of seasonal and inter-annual rainfall variability.

We combined time-integrated analysis of N fluxes measured with ion-exchange capsules and conventional soil analysis of mineral N to analyse inorganic N availability for plants. Three grazing sites (a long-term grazing enclosure (UG79), a moderate winter grazed site (WG) and a long-term heavily grazed site (HG) were included to cover different land-use variations.

Grazing had comparable minor impact on the N availability dynamics compared to seasonal and annual rainfall variability. In one of three analysed growing seasons the long-term grazing enclosure showed significant higher N availability compared to site WG and site HG. Annual differences of N availability were much more pronounced as a result of rainfall variability.

In general, grassland NPP correlated positive to inorganic N availability as measured by ion exchange resin and soil extractions. Since productivity in semi-arid grasslands occurs to mayor part below ground, the root production needs to

be taken into account if inorganic N availability is correlated to NPP. No temporal asynchrony was observed between N availability and ecosystem productivity pattern. Ion exchange resins could indicate the plant N uptake at the different grazing sites. However, during growing seasons of comparable low N availability as a result of low rainfall, grazing site-specific N dynamics were not reflected.

Rainfall controlled the  $\text{NO}_3^-$ - $\text{NH}_4^+$  ratio in the soil solution by altering soil water content (SWC), which in dependence of soil texture affects the resulting soil water potential. Below 8 vol.% SWC ammonia dominated the available inorganic N-form, until 13 vol.% SWC the ratio is variable while under more moist conditions >13 vol.% the nitrate proportion of inorganic N was at least higher than 40%.

Climate seasonality and variability finally triggers N-availability processes in semi-arid ecosystems by altering the soil water potential. Positive correlations between N availability and PPN respectively plant N-uptake indicates ecosystem N limitation if water is sufficiently supplied. We conclude that in times of relatively dry conditions, plants that preferentially take up ammonium have a competitive advantage in the semi-arid grasslands of Inner Mongolia. These findings suggest intensifying the research on interactions of precipitation pattern and semi-arid ecosystems functions, which is also of current interest in the frame of the prognoses towards climate change. The knowledge about the role of herbivores in this frame is still fragmentary, which demands for further research. Since climate obviously dominates basic processes of available inorganic N, grazing effects are most likely limited as long intensity is controlled. Decreased soil moisture with increasing grazing intensity however, might possible alter the dynamic of plant available N (and the relation of N-forms) and by this important processes of central Asian semi-arid grassland carbon and N cycling, with consequences for regional and global balances of matter fluxes.

**Key words:** N-cycling, N-uptake, ammonium, nitrate, precipitation, soil water content, Inner Mongolia, steppe

**Introduction**

Nitrogen (N) cycling is often the focal point of studies analysing ecosystem and grazing interactions (Mazancourt et al. 1998, Proulx and Mazumder 1998, Le Roux et al. 2003, Bakker et al. 2004, Loiseau et al. 2005). Since N is considered as an important limiting factor of productivity in most terrestrial ecosystems (Vitousek 1982, Aerts and Chapin III 2000, Burke et al. 1997) the understanding of N dynamics is crucial for a sustainable productivity of grazed natural grasslands. Through intensive interactions, the N cycle is intrinsically coupled with the carbon balance of this large terrestrial ecosystem.

Grazing usually reduces the aboveground biomass and thus the organic matter available for the mineralization process. This might have negative effects on N availability and therefore the productivity of grassland ecosystems. However, accelerated N cycling by increased urine deposition from herbivores could compensate or even overcompensate the N availability (Augustine and McNaughton 2006).

The influence of grazing on plant N availability differs largely between ecosystems and environmental conditions. On a local scale, several studies indicated that the abundance of vertebrates could stimulate ecosystem N cycling, plant N availability and also the productivity (Holland and Detling 1990; Pandey and Singh 1992, McNaughton, et al. 1997; Frank et al. 2000, Garibaldi et al 2007). However, usually long-term grazing is associated with shifts to unpalatable slow-growing plant species and nutrient losses (Milchunas and Lauenroth 1993), which concurrently reduce plant N availability (Lavado et al. 1996, Ritchie et al. 1998, van Wijnen et al. 1999, Wang et al 2006).

Furthermore, in semi-arid grassland ecosystems, soil biogeochemical processes often interact with seasonal or inter-annual rainfall variation and rainfall pulse events (Austin et al. 2004, Augustine and McNaughton 2006, Varnamkhasti et al. 1995, Schimel and Parton 1986). This natural ecosystem variability might finally be of more importance for N dynamics compared to the grazing effects (Biondini et al. 1998). In particular, the impact of soil moisture on the mineralization process and inorganic N availability is poorly understood in semi-arid ecosystems (Yahdjian et al. 2006). An increased number of studies reported a pronounced asymmetry of resource availability for N and water resulting from soil wet-dry cycles (Austin et al. 2004, Augustine and McNaughton 2006), which contrasts the common view of



synchronized mineralization rates, N availability and plant productivity dynamics (Pastor et al. 1984, Vitousek and Matson 1985, Frank and Groffman 1998).

Another promising aspect of ecosystem N dynamics, the research on the effects of different N-form availability (i.e. nitrate and ammonium) on ecosystems functions has just begun and first results suggest considerable impacts on species competition (Weigelt et al. 2005) and carbon turnover (Austin et al. 2006). How these natural driven N dynamics interact with grazing is largely unknown, but climate consumer interactions are most likely affecting important ecosystem processes such as above and belowground productivity (Frank 2007).

In central Asian steppe ecosystem, one of the most distinct disturbances of temperate semiarid grasslands is realized by excessive overgrazing within the last decades (Tong et al. 2004). How these large-scale land-use changes alter ecosystem processes especially in combination with pronounced seasonal climate variability (Fernandez-Gimenez and Allen-Diaz 1999, Snyder and Tartowski 2006) is still unclear. Increased landscape degradation including long distance off-site effects such as increasing dust storm events, could clearly be linked to overgrazing. Thus, a broad interest addresses these processes and demands for management options to reduce negative consequences.

In this case study of a semi-arid grassland ecosystem, we ask for the impact of grazing and rainfall variability on plant available inorganic N. We link ecosystem primary net productivity to inorganic N availability and discuss the availability and appearance of nitrate and ammonium in the context of seasonal and inter-annual rainfall variability. The time-integrated analysis of plant inorganic N availability by ion exchange resins addresses not only the mineralization process of organic N by soil microbes, but includes also possible direct herbivore effects through urine deposition. Since most studies analysing grazing effects on N cycling focus only on the mineralization process (McNaughton et al. 1997, Frank and Groffmann 1998, Barger et al. 2004), while only a few studies include inorganic N pool dynamics (Augustine and McNaughton 2006), a combined analysis of ion-exchange and conventional soil analysis may provide promising insights into inorganic N availability dynamics in the framework of plant-soil-animal interactions and rainfall variability.

## **Material and Methods**

### *Study area and experimental sites*

Detailed information about the study area and experimental sites are given in the general introduction part (chapter 1). During this study, we investigated three sites characterized by different grazing intensities. A long-term grazing enclosure (hereafter: UG79, position 43°33'10"N; 116°40'33"E), a moderate winter grazing site (WG, position 43°32'57"N; 116°40'04"E), and a long-term heavily grazed site (HG, position 43°34'54"N; 116°40'37"E). We are aware that the transferability of results from this study might be limited due to the problem of pseudoreplicates. In cooperation with Institute of Botany of CAS we carefully chose sites being representative for the ecosystem, approved by several published data (e.g. Bai et al. 2004).

### *N dynamics by ion exchange resins*

Several studies illustrated that the ion exchange resin technique can be successfully used for the time integrated monitoring of plant available nutrients in semi-arid ecosystems (Augustine and McNaughton 2006, Hook and Burke 2000, Robson et al. 2007, Dodd et al. 2000).

We used PST-1 ion exchange resin capsules (UNIBEST, Montana) filled with mixed anion-cation resins to measure the plant available ammonium and nitrate in the soil solution. Resin capsules function as a sink, accumulating ions that contact the surface of the capsule. Some ions are present in the soil solution when the capsule is introduced, and other ions will diffuse to the capsule surface. Diffusion occurs because the continual removal of ions from the solution at the resin surface creates a concentration gradient. Multi-element diffusion is simultaneous and independent Yang and Skogley (1992), and the longer the accumulation period, the greater the distance from which the ions may diffuse. Each soil has its own diffusion characteristics, hence, the volume of soil contributing ions to the capsule during a specific interval is soil-dependent. Unless additional measurements are made, this volume (or soil mass) will not be known. What is known after capsule analysis is the quantity of each target ion that was bioavailable at a point in the sample during the adsorption period. This is like analysing plants to determine quantities accumulated; the quantity of soil contributing to plant uptake of elements also generally is not known (UNIBEST 2007).

Capsules were installed with five replications per site into the main rooting zone at 15 cm depth and were exchanged bi weekly (2004) to monthly (2006). At the following dates (days of year: DOY) capsules were exchanged: 2004: 150-177, 177-194, 194-208, 208-225, 225-238, 238-260 and 260-109 (winter). 2005: 109-131, 131-152, 152-172, 172-193, 193-217, 217-235, 235-261 and 261-135 (winter). 2006: 135-163, 163-197, 197-224 and 224-258. For the incubation, an approximately 15 x 15 cm grass sod was trumped every time and a small cavity was formed to assure good contact of the capsule to the soil after replacing the sod. We followed instructions of UNIBEST to analyse the amount of ammonium and nitrate, exchanged by capsules. Soil particles adhering to the surface of the capsule were rinsed off by directing a stream of deionized water onto the capsule while rotating it. Capsules were stored near 0°C until extraction. Ions were recovered by sequential shaking in three batches of 20 ml of 2 M HCl. For nitrate and ammonium analysis, the extract was analysed with a Bran+Luebbe rapid flow auto analyser after neutralization with 2 M NaOH.

The general formula to calculate RAQ – (Resin Adsorption Quantity in  $\text{mmol cm}^{-2}$ ) is given by:

$$\text{RAQ} = c V/M A$$

where  $c$  is the concentration of nutrient in the HCl extract obtained from the capsule ( $\text{mg l}^{-1}$ ),  $V$  is the volume of the extract (ml),  $M$  is the molar mass of the nutrient, and  $A$  is the capsule surface ( $11.4 \text{ cm}^2$ ). Data were calculated as average daily N flux through resin capsule surface ( $\text{mg N m}^{-2} \text{ d}^{-1}$ ).

#### *N dynamics by soil coring*

At each sample date from 2005 on, we extracted soil cores of 4 cm diameter from 0–20 cm depth in 4 replications; each replicate was pooled from 3 sub samples. Replicates within the grazing sites were at least in a distance of 20 m to each other. Sample dates (2005 and 2006) were kept identically to the resin capsules (see above). The soil samples were cooled in ice-boxes and deep frozen until analysis. Water content of fresh samples was determined gravimetrically.  $\text{NH}_4^+$  and  $\text{NO}_3^-$  from 12 g soil (fresh weight) were extracted with 100 ml 0.01 M  $\text{CaCl}_2$ . Concentrations of ammonium and nitrate were analysed with a Bran+Luebbe rapid

flow auto analyser. Data were corrected for soil water content and bulk density of the different grazing sites (1.0 g/cm<sup>3</sup> for site UG79, 1.1 g/cm<sup>3</sup> for site WG and 1.2 g/cm<sup>3</sup> for site HG).

#### *Data analysis*

We used mixed model analysis of variance with years as random, block and grazing sites as fixed factors. To separate the different years effects, we applied repeated measurement ANOVA analysis to address the seasonal dynamic sampling design. Statistical analyses were performed using SAS - software version 8.0. Curve fitting for nitrate-ammonium ratio in relation to soil water content (Fig. 4) were performed with Sigma-Plot (version 9.0) by selecting non-linear (sigmoid, 3 parameter) regression.

## Results

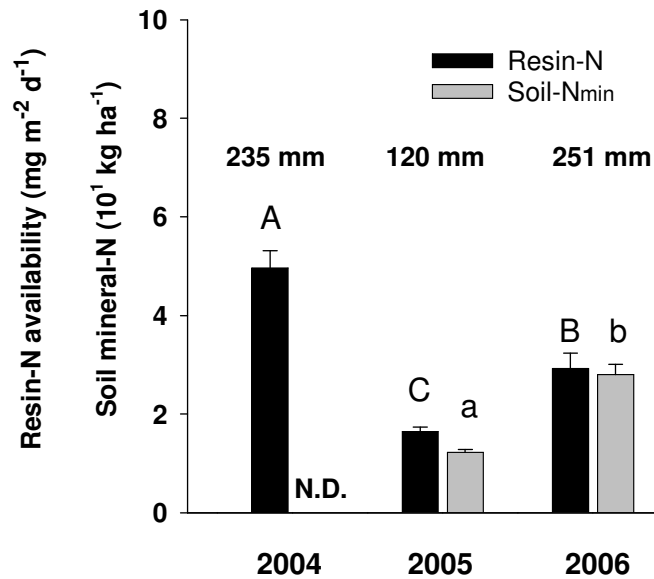
### *Effects of grazing and rainfall variation on N availability*

The results of the mixed model analysis of variance are summarized in Table 1. We tested for block effects with a negative result. Since only a weak ( $P < 0.058$ ) interaction was found, the main effects were analysed. Resin-N availability significantly varied over the three growing seasons 2004–2006 ( $P < 0.013$ ), but there were no significant differences between the three grazing sites. Soil mineral-N extractions were carried out in 2005 and 2006. Statistics indicated no interaction between sites and years. Like for the resin-N results, no block and grazing effects on N-availability were found, but the effect of years was significant ( $P < 0.018$ ).

Fig. 1 shows annual effects on the mean daily resin-N and soil mineral-N availability for the growing seasons 2004–2006. Highest resin-N availability was measured in 2004, followed by 2006, both years with about the same precipitation amount during resin incubation time (235 mm, respectively 251 mm). The lowest resin-N availability for all treatments was observed in the very dry growing season 2005 (120 mm). For the soil mineral–N results, we observed a higher soil mineral-N status during the growing season 2006 compared to the dry season 2005, which supports the resin N availability data.

**Table 1:** Results of mixed model ANOVA for resin N-availability (2004–2006) and soil mineral N (2005–2006) at three grazing treatments, HG, WG, UG79 (site).

Effect	Soil resin N-availability		Soil mineral N by extraction	
	<i>F</i>	<i>P</i>	<i>F</i>	<i>P</i>
<b>Block (fixed)</b>	1.18	0.3378	2.26	0.1239
<b>Site (fixed)</b>	2.04	0.2446	0.78	0.5631
<b>Year (random)</b>	15.61	0.0129	53.41	0.0182
<b>Site x year</b>	2.55	0.0580	2.40	0.1250

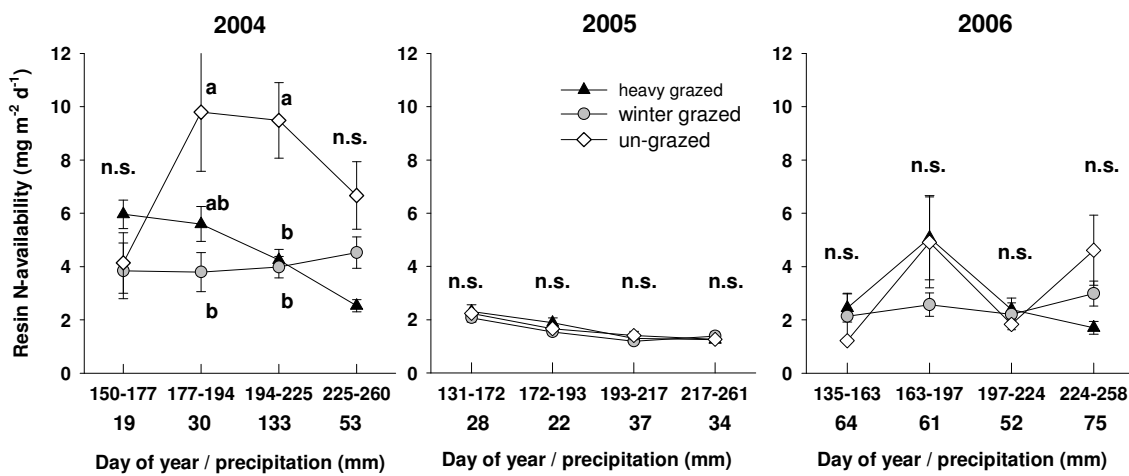


**Fig. 1:** Annual effects of resin-N and soil mineral-N (0-20 cm) availability, pooled over three different grazing intensities (HG, WG and UG79). Tukey grouping based on mixed model ANOVA. The precipitation during sampling period (May-September) is indicated above the bars. No soil mineral-N data available for 2004. Resin data were calculated as mean daily uptake rates of incubation periods, soil N<sub>min</sub> was calculated from mean values of all extractions during the growing season.

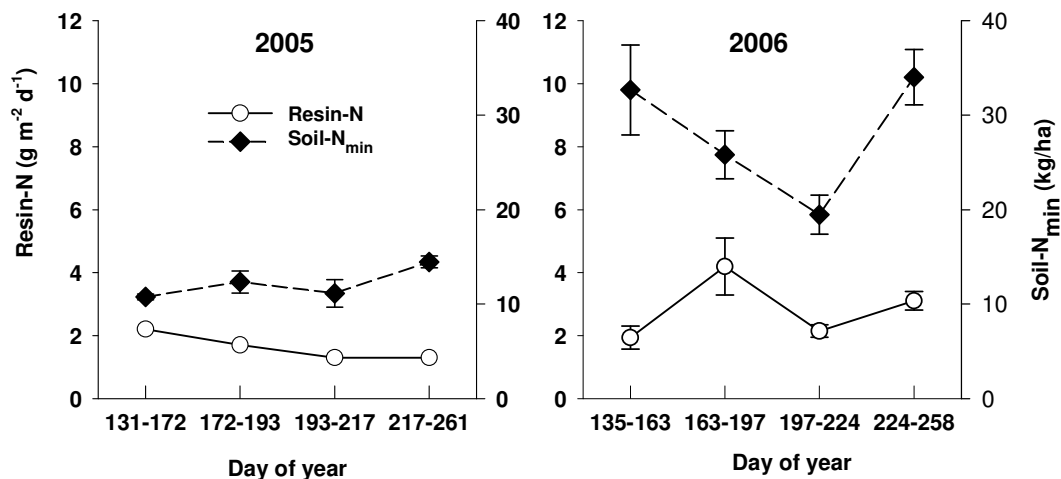
However, the weak site x year interaction in the resin-N data ( $P < 0.058$ , Table 1) should be addressed. Furthermore, due to major rainfall and N-availability differences among years, we separated each growing season for detailed repeated ANOVA analysis (Fig. 2). Separated by years, in 2004 a significant site effect was identified ( $P < 0.001$ ), where UG79 revealed higher resin-N availability compared to site WG ( $P < 0.007$ ) and site HG ( $P < 0.046$ ). Fig. 2 indicates the significant differences for the year 2004 during two sampling periods, with the most pronounced effect during the third sampling period (DOY 194–225). This period concurrently revealed the highest precipitation recorded during the three observed growing seasons. In the years 2005 and 2006 we observed no significant difference between the grazing sites.

Resin-N and soil mineral-N showed basically the same annual and seasonal trends (Fig. 3). In 2005, both methods reflect the relatively small dynamics of N availability, while in 2006 the seasonal N-dynamics were much more pronounced, which coincide with higher rainfall amounts during the sampling periods (see Fig. 2). Except of the early 2006 measuring period (DOY 135–163), when relatively

high soil mineral-N availability contrasts the relatively low resin-N availability, both methods indicated comparable low N availability during the third sample time (DOY 197–224) as well as an increased N availability during the last sampling period of 2006 (DOY 224–258). The last sample period was characterized by relatively high precipitation amounts (75 mm) and soil water content (12 vol.%), while the sample period before gained less rainfall (51 mm) and concurrently had low average soil water content (9 vol.%).



**Fig. 2:** Seasonal dynamics of the mean daily resin N-availability ( $\text{mg N m}^{-2} \text{d}^{-1}$ ) separated by site heavy grazed, winter grazed and un-grazed as well as precipitation amounts during sample periods for the growing seasons 2004–2006 in Inner Mongolia semi-arid grassland. Error bars represent the standard error, letters indicate significant differences by repeated ANOVA analysis separated by years with  $P < 0.05$ .



**Fig. 3:** Seasonal dynamics of resin-N and soil mineral-N (0-20 cm) availability during growing season 2005 and 2006 in Inner Mongolia semi-arid grassland. The graph shows mean values from three different grazing sites with error bars representing the standard error.

*N availability and grassland productivity*

We correlated inorganic N availability as measured by resin-N and soil mineral-N extractions with net primary productivity (NPP) and N uptake data of the plant canopy from the three investigated growing seasons 2004–2006 at the grazing sites.

The highest resin-N availability and aboveground net primary productivity (ANPP) were both measured in the growing season 2004 (Table 2). No significant difference in ANPP was found between 2005 and 2006. Thus, the significantly higher N availability as indicated by resin-N and soil mineral-N extractions for 2006 compared to 2005 was not reflected by increased ANPP. Nevertheless, we found an increased belowground net primary productivity (BNPP) in 2006 compared to 2005. Including the BNPP dynamics, results from resin-N and soil mineral-N extractions measures confirm ecosystem net primary productivity (NPP) pattern.

Analysing the grazing effects, only in the growing season 2004 the highest resin-N availability could clearly be linked to the site of highest NPP (UG79). Site WG and HG showed no significant difference for N-availability and productivity data. In the dry year 2005, low N-availability was associated with a general low productivity at all investigated sites. However the significantly lower NPP of site HG compared to sites WG and UG79 in 2005 and 2006 was not reflected by N-availability values of resin capsules or soil mineral-N extractions.

Since we found high fluctuations in the N content of the plant canopy within one growing season and also between the years, we correlated plant N availability with N content of the plants. Fig. 4 illustrates the relation between the aboveground N uptake by plant canopy and the N availability as indicated by resin capsules. We calculated the daily average N uptake ( $\text{mg m}^{-2} \text{d}^{-1}$ ) by resins for the time period until highest aboveground N content ( $\text{g m}^{-2}$ ) was measured for the plant canopy of the respective grazing site. Different colours indicate the growing seasons from 2004–2006 and abbreviations are attached to identify the grazing sites. The general trend was a linear relationship, showing an increased resin N availability, which coincident with increased aboveground N allocation. The growing season 2004 shows highest aboveground plant N content correlating with highest resin N availability. Differences between the sites are indicated as well. Site UG79 exhibits highest values of resin-N availability and plant N uptake, while site WG was lowest



in both parameters. The growing season 2006 delivered medium N availability values ranking between the other two seasons, which was also reflected by the aboveground N uptake by plants. We also found disagreements between the two parameters. Lowest resin-N availability was observed for the year 2005 with no significant differences between the sites. However, site UG 79 showed a relative high aboveground plant N uptake during that growing season, which was not indicated by the resins N availability.

**Table 2:** N availability measured as resin-N ( $\text{mg N m}^{-2} \text{d}^{-1}$ ) and soil mineral-N extractions ( $\text{kg ha}^{-1}$ ) and above- and belowground net primary productivity (ANPP, BNPP, in  $\text{g/m}^2$ ) at different grazing intensities (HG = heavy grazing, WG = winter grazing and UG79 = ungrazed since 1979) in Inner Mongolia semi-arid grasslands. Data are given as mean values  $\pm$  SE. Significant differences between grazing and years are indicated by different small and capital letters, respectively.

	Site	Resin-N <sup>1</sup>	Mineral-N <sup>2</sup>	ANPP <sup>3</sup>	BNPP <sup>4</sup>	NPP <sup>5</sup>
2004	HG	4.4 $\pm$ 0.1bA	N.D.	121.6 $\pm$ 7.9bA	182	304
	WG	3.9 $\pm$ 0.4bA	N.D	126.3 $\pm$ 1.9bA	189	316
	UG79	6.6 $\pm$ 1.0aA	N.D	175.5 $\pm$ 6.2aA	263	439
	<i>mean</i>	5.0 $\pm$ 0.5	<i>N.D</i>	141.1 $\pm$ 7.3	212	353
2005	HG	1.7 $\pm$ 0.2aC	13.3 $\pm$ 0.8aB	76.6 $\pm$ 3.8 bB	107 $\pm$ 7	184
	WG	1.6 $\pm$ 0.3aC	12.9 $\pm$ 1.2aB	99.1 $\pm$ 3.0 aB	142 $\pm$ 5	241
	UG79	1.7 $\pm$ 0.1aC	10.6 $\pm$ 0.6aB	111.7 $\pm$ 11.3 aB	180 $\pm$ 10	292
	<i>mean</i>	1.6 $\pm$ 0.2	12.3 $\pm$ 0.5	95.8 $\pm$ 5.1	143 $\pm$ 8 B	239
2006	HG	3.0 $\pm$ 0.6aB	34.8 $\pm$ 4.1aA	75.3 $\pm$ 2.6 bB	139 $\pm$ 8	214
	WG	2.5 $\pm$ 0.4aB	25.2 $\pm$ 2.2aA	95.7 $\pm$ 2.9 aB	171 $\pm$ 3	277
	UG79	3.3 $\pm$ 0.5aB	24.5 $\pm$ 3.7aA	106.5 $\pm$ 4.8 aB	203 $\pm$ 14	310
	<i>mean</i>	2.9 $\pm$ 0.3	28.0 $\pm$ 2.0	92.5 $\pm$ 3.3	171 $\pm$ 8 A	267

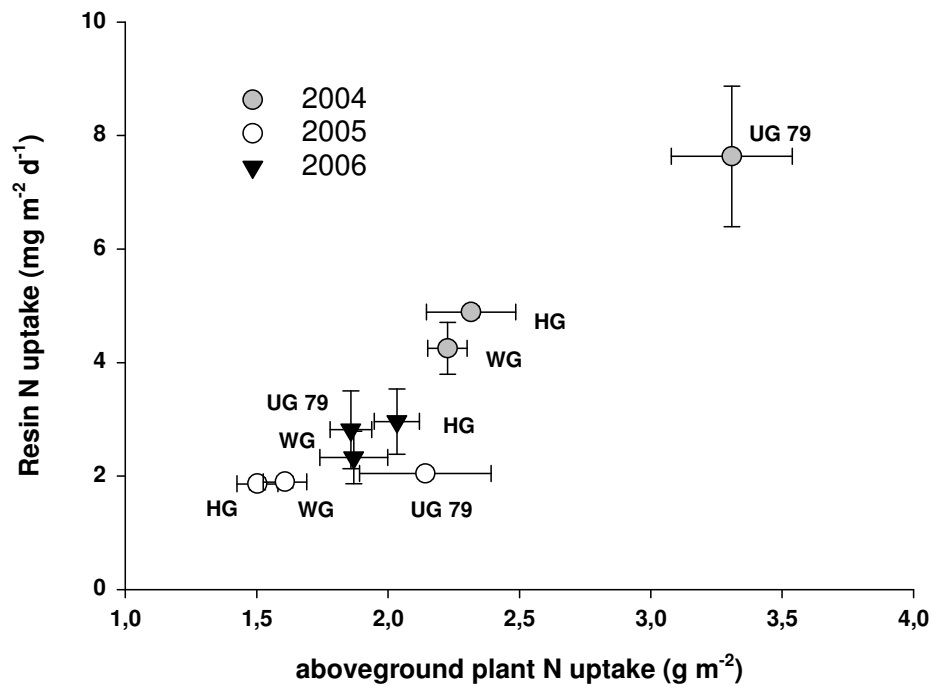
<sup>1</sup> Resin-N availability as mean daily fluxes during growing season.

<sup>2</sup> Soil mineral-N: mean of four sampling dates. no data available for 2004

<sup>3</sup> ANPP calculated as peak green biomass (middle of August) for ungrazed sites WG and UG79, ANPP at site HG was measured with moving cages.

<sup>4</sup> Data for 2004 estimated from BNPP/NPP ratio = 0.6 (Gao 2007, p. 60). 2005 and 2006 measured data by ingrowth core method (Gao 2007, p. 59)

<sup>5</sup> ANPP + BNPP



**Fig. 4:** Resin N uptake in relation to aboveground plant N uptake measured during the growing seasons 2004–2006 at different grazing intensities (UG79 = grazing exclusion, WG= winter grazing, HG = heavy grazing) in semi-arid grassland of Inner Mongolia. Resin average daily N uptake ( $\text{mg m}^{-2} \text{d}^{-1}$ ) was calculated for the period from beginning of growing season until aboveground plant N content ( $\text{g m}^{-2}$ ) reaches annual maximum (error bars represent standard error).

*Nitrate and ammonium availability*

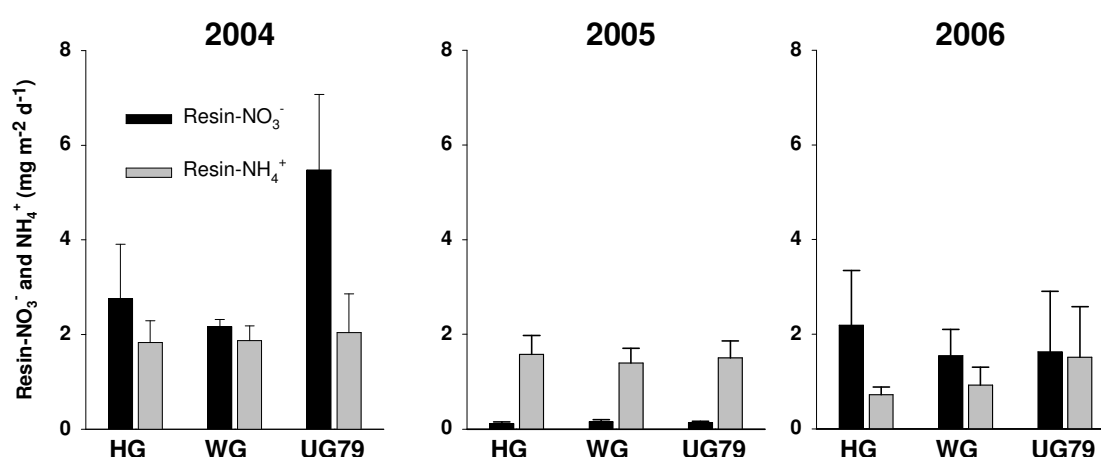
The resin nitrate- and ammonium-N availability of the different grazing sites (HG, WG and UG79) during the growing seasons 2004–2006 is illustrated in Fig. 5.

The strong site x year interaction ( $P < 0.007$ , Table 3) indicated by mixed model ANOVA originates from significantly higher resin-nitrate availability at site UG79 compared to site HG ( $P < 0.026$ ) and site WG ( $P < 0.002$ ) in the growing season 2004. To analyse the annual effects, we separated by grazing sites and found a significantly lower resin nitrate-N availability in 2005 compared to 2004 at site HG ( $P < 0.004$ ) and site UG79 ( $P < 0.001$ ), while at site WG only a weak significance ( $P < 0.068$ ) was observed. Except of site HG ( $P < 0.022$ ), no significant difference was found between the years 2005 and 2006 for resin nitrate-N availability.

Analysing the resin ammonium-N availability, we found no interaction between sites and years according to mixed model ANOVA (Table 3). No significant differences for resin ammonium-N availability between the grazing sites and only weak ( $P < 0.054$ ) annual effects. During the growing seasons of 2004 and 2005, the resin ammonium-N availability remained at a constant level. In contrast to the resin nitrate-N availability, we did not record an effect of the pronounced rainfall variability between these two years. 2006 however, exhibited significantly lower resin ammonium-N availability at site HG ( $P < 0.001$ ) and site WG ( $P < 0.05$ ) compared to the other years, while at UG79 no effect between the years was observed at all.

**Table 3:** Results of mixed model ANOVA for resin nitrate-N and ammonium-N availability (2004–2006) at three grazing treatments HG, WG, UG79 (site).

Effect	Resin nitrate-N availability		Resin ammonium-N availability	
	<i>F</i>	<i>P</i>	<i>F</i>	<i>P</i>
<b>Block (fixed)</b>	1.04	0.4003	1.12	0.3638
<b>Site (fixed)</b>	0.94	0.4620	1.37	0.3531
<b>Year (random)</b>	9.25	0.0316	6.65	0.0535
<b>Site x year</b>	4.26	0.0071	1.94	0.1278

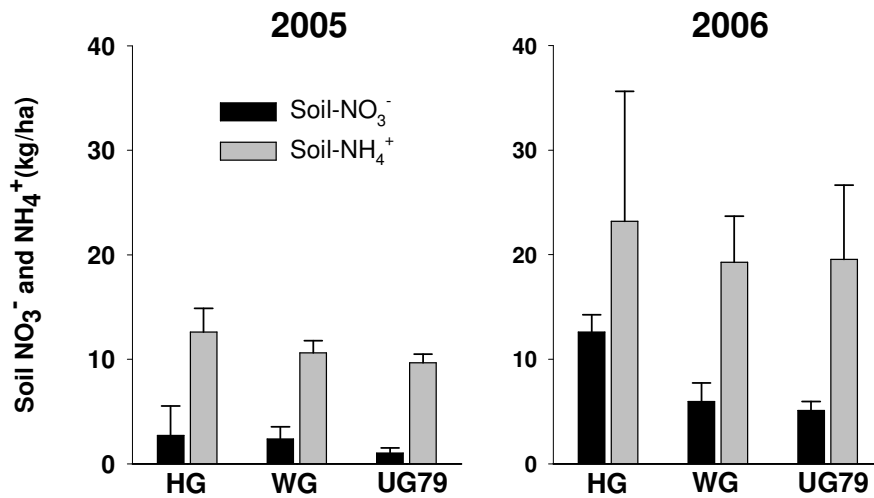


**Fig. 5:** Resin nitrate- and ammonium-N availability from growing seasons 2004 – 2006 at different grazing treatments (error bars represent standard deviation).

The nitrate- and ammonium-N availability at the different grazing sites as indicated by soil mineral-N extractions in 2005 and 2006 is shown in Fig. 6. Mixed model ANOVA indicated no interaction, no block and no site effect for soil ammonium-N availability (Table 4). The annual effect was significant ( $P < 0.009$ ) with higher ammonium-N availability in 2006 compared to 2005. For the soil nitrate-N availability an interaction ( $P < 0.002$ ) was identified (Table 4). Nitrate availability was significantly higher at site HG compared to the other sites in 2006. Separated by sites, we analysed the differences of nitrate availability between the years: all sites showed significantly lower nitrate availability in 2005 compared to 2006.

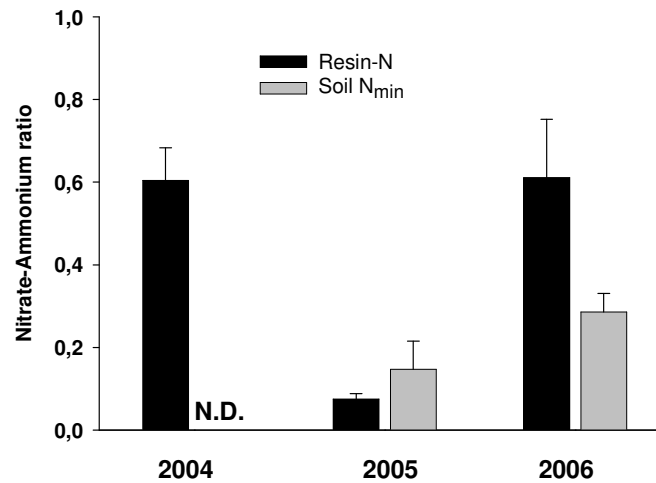
**Table 4:** Results of mixed model ANOVA for soil nitrate-N and ammonium-N availability (2005–2006) at three grazing treatments HG, WG, UG79 (site).

Effect	Soil nitrate-N availability		Soil ammonium-N availability	
	<i>F</i>	<i>P</i>	<i>F</i>	<i>P</i>
<b>Block (fixed)</b>	1.42	0.2763	1.48	0.2607
<b>Site (fixed)</b>	1.04	0.4902	1.09	0.4787
<b>Year (random)</b>	14.57	0.0623	109.2	0.0090
<b>Site x year</b>	10.77	0.0013	0.50	0.6162



**Fig. 6:** Soil nitrate- and ammonium-N availability from growing seasons 2005–2006 at different grazing treatments (error bars represent standard deviation).

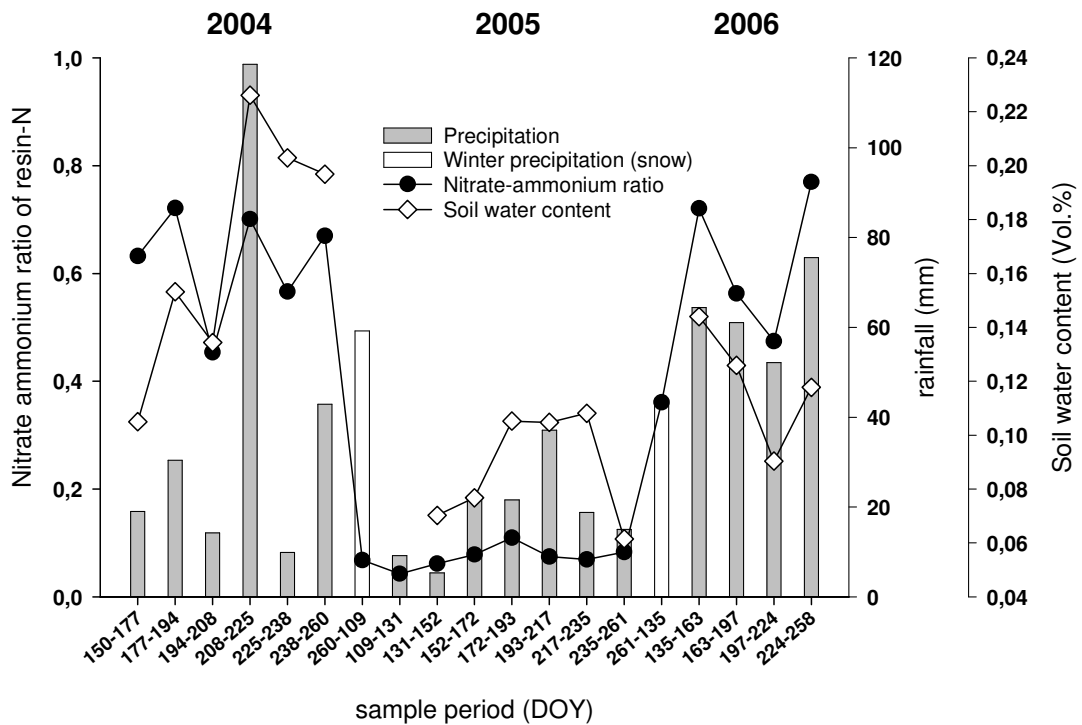
The nitrate-ammonium ratio changed clearly between the investigated years (Fig. 7). In the growing season 2004 (ratio 0.60) and 2006 (0.61) the plant available N-form measured by ion exchange resins was slightly shifted to nitrate, while 2005 revealed a completely different situation. The resin nitrate-ammonium ratio decreased to 0.07, indicating that ammonium was the dominant available inorganic N-form. Soil mineral-N analysis showed the same trend for the year 2005, where ammonium was clearly the primary available N-form (average nitrate-ammonium ratio 0.14). However, despite in 2006 the nitrate-ammonium ratio increased (ratio 0.28), ammonium was still the dominant available N-form recorded, which contrasts the results from ion exchange resins.



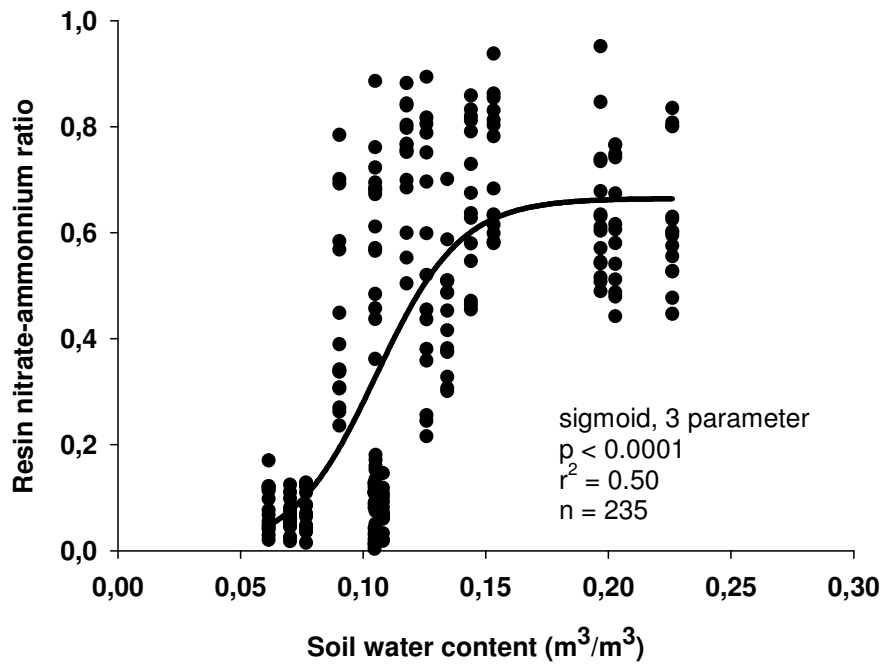
**Fig. 7:** Nitrate-ammonium ratio of resin-N and soil mineral-N availability (0–20 cm) during growing seasons 2004 to 2006. Mean values and standard deviation for resin-N from 17 sample periods and soil mineral-N from 11 sample periods separated by year (6 in 2004, 7 in 2005, 4 in 2006). Each sample period represented by mean values of five (resin-N) and four (soil mineral-N) replicates separated for 3 grazing sites (HG, WG and UG79).

Analysing the seasonal dynamics of resin nitrate-N respectively ammonium-N availability, we found basically the same trend as indicated by the inter-annual pattern. In Fig. 8, for the sampling periods during the growing seasons 2004–2006 the resin nitrate-ammonium ratio, average soil water content (SWC in vol.%) and the precipitation (mm) are plotted. SWC increased with increasing rainfall amount during sampling period, but fluctuations of the soil water content are much less pronounced compared to rainfall variability. The seasonal changes of soil water content and precipitation pattern are in general reflected by the dynamics of the resin nitrate-ammonium ratio. In the year 2004 and 2006 the ratio increases with increasing soil water content and at the other side usually decreases when soil became drier. During the sampling periods 2005, where the average water content of the loamy-sand to sandy-loam textured soil (sand 60–80%, silt 15–22% and clay 10–15%) never exceeded 12 vol.% the resin nitrate-ammonium ratio remained at a low level through out the season. According to soil texture analysis the permanent wilting point (-1.5 MPa water potential) at the investigated sites is calculated to a SWC of 6.4 vol.% for site HG, 7.5 vol.% for site UG79 and around 10 vol.% for site WG (Krueemmelbein 2007). In Fig. 9, the resin nitrate-ammonium ratio correlates positive to soil water content. Non-linear (sigmoid, 3 parameter)

regression was fitted to 236 observations and was significant at  $p < 0.0001$  level with  $r^2$  of 0.50. If the SWC decreases below 8 vol.%, ammonium clearly dominates the resin available N-form. Between 10-13 vol.% SWC, the nitrate-ammonium ratio is variable. Above 13 vol.% of SWC the resin available N-form is balanced or dominated by nitrate.



**Fig. 8:** Soil water content, precipitation and nitrate ammonium ratio of resin available N during 19 incubation periods from 2004–2006. No soil water content data available from sampling periods with frost.



**Fig. 9:** Correlation between nitrate-ammonium ratio of available resin N and soil water content. Data from 16 sampling periods during the growing seasons 2004 (6), 2005 (6) and 2006 (4) at three different grazing intensities (HG, WG and UG79) measured with five replications.



## Discussion

### *Grazing and rainfall impact on N availability*

In our study of semi-arid temperate grasslands in Inner Mongolia, large herbivores had comparable minor impact on the N availability dynamics. Only in one of three observed years (2004), we found a significant grazing effect. Within that growing season, the long-term grazing enclosure exhibited higher N availability compared to the winter- and heavy grazed site. This confirms studies reporting negative effects of herbivores on N cycling and availability (Lavado et al. 1996, Ritchie et al. 1998, van Wijnen et al. 1999, Wang et al. 2006). During the other two growing seasons (2005 and 2006) no grazing effect on the N availability was recorded. At no time we observed indications of increased N availability under grazing as reported by several other studies (Holland and Detling 1990; Pandey and Singh 1992, McNaughton, et al. 1997; Frank et al. 2000, Garibaldi et al. 2007).

In fact, we found rainfall amount and distribution and the corresponding soil water content and soil water potentials having a higher impact on inorganic N availability compared to grazing effects. The drought period during the growing season in 2005 exhibited the lowest inorganic N availability, while the two other years with double precipitation amounts compared to 2005 revealed significantly higher inorganic N availability during growing seasons. This trend was confirmed by the two different methods (resin-N and soil mineral-N) we used to measure N availability (Fig. 1). Soil biochemical processes affecting mineralization are highly depending on temperature and rainfall pattern. Water limited ecosystems however, are highly sensitive to water pulses controlling the belowground processes through soil wet-dry cycles (Austin et al. 2004). In semi-arid grasslands, Ford et al. (2007) found an N availability flush after re-wetting accounting for more than 90% of total N mineralized during the incubation time. N mineralization, microbial biomass and microbial activity were negligible once soils had dried to -1 MPa. According to the pF-curve calculated from soil texture measurements (Kruemmelbein 2007), this water potential is realized at our sites at SWC around 10 vol.%.

During our field campaign, we experienced very low to average precipitation amounts during the growing seasons (May–September) according to IMGERS long-term meteorological data. 2005, as a year of an extreme drought, might have prolonged effects into the next growing season, where N availability was still very low despite slightly above average early season rains. Results, we observed for

2004–2006 need to be confirmed for wet growing seasons as well. N dynamics in general, and in particular with respect to grazing effects, might differ from average and dry years.

Higher impacts of climate related factors on N dynamics compared to grazing from North-American semi-arid grasslands were reported by Biondini et al. 1998. Frank (2007) observed higher impacts of annual rainfall on belowground ecosystem dynamics compared to grazing, but concurrently suggested that climate-consumer interactions might potentially affect important ecosystem processes. On the other hand, other studies provide clear evidence that despite pronounced annual variability in rainfall, large herbivores strongly influence productivity and N-cycling in semi-arid tropical grasslands (Augustine and McNaughton 2006). The number of studies however, analysing the combined effects of climate variation and grazing on semi-arid ecosystem processes in general and N availability in particular is still very limited and results are controversial.

Until now, the high variability of factors, which could affect plant-soil-animal interactions, interferes with the development of general conclusions of grazing impacts on ecosystem processes (Bardgett and Wardle 2003). These limited knowledge strengthen in particular the need to study impacts of climate variation on semi-arid ecosystem processes. This evidence arises not only in the context of possible climate change scenarios, which predict an increased intensity and frequency of extreme climate events including droughts (Easterling et al. 2000). It is well known that semi-arid ecosystems are characterized by a high natural climatic variability (Varnamkhasti et al. 1995, Fernandez-Gimenez and Allen-Diaz 1999, Snyder and Tartowski 2006), which might be one reason that evidence about grazing-ecosystem interactive processes in these ecosystems is controversial and limited so far.

#### *N dynamics and plant productivity*

Does the inorganic N availability measured by ion exchange resin and soil mineral-N extractions reflect the ecosystem net primary productivity (NPP) and N uptake? In temperate semi-arid ecosystems the low winter temperatures and drought periods during summer usually defines plant and microbial activities. In the limited time of preferential growth conditions, temperature and moisture pattern usually

synchronize plant productivity and mineralization rates (Pastor et al 1984, Vitousek and Matson 1985, Frank and Groffman 1998).

In all, we found the annual ecosystem net primary productivity dynamics well indicated by the soil inorganic N-availability measurements. Years of relative high productivity corresponded with relatively high inorganic N availability (Table 2). Since productivity in semi-arid grassland ecosystems occurs to mayor part below ground, the root production needs to be taken into account if inorganic N availability is correlated to net primary productivity. Furthermore, we found very high N concentrations in the leaves during the early growing season 2006, which indicated higher net N uptake per unit dry matter compared to the other growing seasons (Gao 2007). This corresponds to the increased N availability measured by resin- and soil mineral-N for the year 2006, which was not evident by dry weight analysis only based on ANPP.

However, the different grazing intensities were less well correlated to resin-N and soil mineral-N status. Only in the year 2004 we were able to confirm grazing effects on net primary productivity with inorganic N availability data. A possible explanation could be, that ungrazed grassland built up more reserves in belowground organs to support growth in dry periods compared to grazed systems, where nutrients are continuously required to compensate herbivore consumption. Such plant internal nutrient cycling (e.g. N retranslocation) is consequential not reflected by soil N availability dynamics.

The high fluctuations in the N content of the plant canopy within one growing season and also between the years, indicated that net primary productivity and N uptake from soil have different dynamics. From this consideration we hypothesized that resin N availability should correlate better to plant N uptake compared to net primary productivity. In Fig. 4, the general good correlation of plant N availability with N uptake of the plant canopy is demonstrated. Ion exchange resins thus could indicate different seasonal dynamics of plant N uptake at the grazing sites. However, during seasons of comparable low N availability as a result of low rainfall, site-specific N dynamics were not reflected, although this method is still able to indicate annual differences under these water-limited conditions. We are still facing the problems that plant internal N translocations most likely interfere the correlations with resin N availability, which might explain the high aboveground N allocation at site UG79 in 2005.

The seasonal dynamics of semi-arid ecosystem N mineralization rates are recently debated, since some studies found considerable amounts of N mineralization occurring out of growing season. This was found in California grassland ecosystems, where soil inorganic N pools and microbial biomass N increases during dry season when plants are senescent (Jackson et al. 1988). Yahdjian et al. (2006) found an accumulation of inorganic N during dry episodes in Patagonian steppe suggesting periods of maximum water availability and nutrient availability occur at different times. In a 3-years study including dry and wet growing seasons Augustine and McNaughton (2004) showed consistent results of temporary asynchrony of N availability dynamics and plant productivity in semi-arid grassland. The results obtained from Inner Mongolia grassland did not confirm a nutrient accumulation during periods of drought. Soil mineral-N extractions did not show higher inorganic N concentrations after dry periods during the growing season 2005. In the first sampling period of 2006 (DOY 135–163) however, a comparably high N availability was measured by soil mineral N extractions, while resin-N indicated relatively low N availability. In the same sampling period high leaf N concentrations (3 to 4.5%) were measured at the grazing sites (Gao 2007). The results together indicate the possibility of an N availability flush in the early growing season of 2006. The high leaf N concentrations could indicate limited water availability, resulting from the last growing seasons drought. Soil water content oscillating around the permanent wilting point, might be one reason for the low resin-N absorption at the same time. The amount of ions delivered to the resin capsule surface strongly depends on soil hydroconductivity and the diffusion pattern of the ion itself. A plant root is still able to take up water and nutrients until the permanent wilting point is reached, which equals water potential of -0.7 to -3 MPa for drought adapted plants (Larcher 1994, Schulze et al. 2004). Diffusion processes and thus ion uptake by resins are strongly inhibited at this water potential, which might explain the observed discrepancies between the methods.

*Plant available N-form in the annual and seasonal ecosystem climate variability*

Plant available N-forms and consequences for ecosystem processes were only very recently noticed and exploration is about to be intensified (Austin et al. 2006, Weigelt et al. 2005, McKane et al. 2002). Almost no information is available on the soil inorganic N-form as influenced by rainfall variability in semi-arid ecosystems. Yahdjian et al. (2006) found a general trend of higher nitrate-N concentrations in the soil extracts with decreasing rainfall input, while ammonium-N concentrations were unaffected by rainfall interception treatments. High inorganic N accumulations during drought could be explained by reduced ion mobility in thin water films of dry soils. Furthermore plant and microbe assimilation as inorganic N sinks are inhibited. Stark and Firestone (1995) found soil water potential below -0.6 MPa inhibiting the nitrification process. According to the pF-curve of the soils at our sites (Kruemmelbein 2007) this threshold value is realized at soil water content of approximately 10–12 vol.%, which is exactly the transition zone where soil nitrate-ammonium ratio was indifferent at our sites (Fig. 4). Thus, low soil water potentials during several sample periods characterized by low rainfall input might explain the low nitrate-ammonium ratio we observed. However, an increased nitrate availability associated with reduced rainfall as reported by Yahdjian et al. 2006 in semi-arid grasslands is contrasting our findings.

Seasonal and inter-annual differences in plant available N-form might explain competitive dominance of grass species in semi-arid grasslands. Studies in Patagonian steppe indicate that the majority of grassland species preferentially take up nitrate (Austin et al 2006). Evidence comes from experimental studies showing the preferential uptake of inorganic and organic N-forms from temperate grassland species (Weigelt et al. 2005).

We conclude that in times of relatively dry conditions, plants that preferentially take up ammonium have a competitive advantage in the semi-arid grasslands of Inner Mongolia. Changes of N form availability thus might have considerable consequences for ecosystem processes including the carbon turnover, which concurrently underline the major role of nitrifying bacteria (Austin et al. 2006). On the other hand, climate seasonality and variability finally triggers N-availability processes in semi-arid ecosystems by altering the soil water potential. These findings suggest intensifying the efforts to explore interactions of precipitation dynamics and ecosystems functions. The role of herbivores in this frame is unclear

at all. Since climate obviously dominates basic processes of available inorganic N-form, grazing effects are most likely limited as long the intensity is controlled. Decreased soil moisture with increasing grazing intensity however, might possible alter the dynamic of plant available N form and by this important processes of semi-arid grassland carbon and N cycling.

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## Nitrogen cycling and balance in grazed steppe ecosystems of Inner Mongolia

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### Abstract

The understanding of the complex pathways of nitrogen (N) semi-arid grassland ecosystems under grazing is fragmentary. This demands for a balance of the different N pools and fluxes as well as N gains and losses as influenced by grazing. The identification of key processes within the N dynamics of Inner Mongolia grasslands might indicate possible options for the application of adapted and sustainable land-use management practices.

We defined a functional unit of a typical grassland farm as our system boundary for N balancing. Two land-use practices were analyzed, a long-term heavy grazed site (annual stocking rate between 2 and 4 sheep/ha), a light to medium grazed site (average stocking rate 1 sheep/ha) with additional haymaking. Furthermore, a long-term grazing enclosure was included.

Ungrazed sites represent N sinks ( $0.7$  and  $1.7 \text{ g N m}^{-2} \text{ yr}^{-1}$ ), while heavy grazed areas are N sources with losses up to  $1.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Areas of moderate grazing only are N sources if combined with haymaking, since this land-use type causes highest ( $\pm 1.0 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) N losses per area.

Dry deposition was identified as the main pathway contributing to N accumulation with more than  $1 \text{ g N m}^{-2} \text{ yr}^{-1}$  at ungrazed areas. Concurrently wind erosion counts for high N losses up to  $0.6 \text{ g N m}^{-2} \text{ yr}^{-1}$  at heavy grazed sites. The wet deposition

counts for N input with up to  $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Heavy grazing causes N losses ( $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) due to the common management practice to keep sheep in paddock overnight (dung not returned). N losses due to meat and wool harvest are comparable small ( $0.2 \text{ g N m}^{-2} \text{ year}^{-1}$ ).

High potential ecosystem fertility (N stored in soil organic matter  $800 - 1000 \text{ g N m}^{-2}$ ) indicated considerable regeneration potential even of long-term heavy grazed areas. However, the complex interactions between climate variability and ecosystem productivity demand for flexible grazing systems. Grazing would be compensated in the N-balance as soon as stocking rate is reduced to a level where the remaining aboveground biomass prevents wind erosion and the involved N losses. A rotational system between hay and grazing sites, as well as growing season rainfall amounts as indicator for next season stocking rates could be a management option.

**Key-words:** N-cycling, N-balance, semi-arid, grassland, grazing

### **Introduction**

Understanding of the complex pathways of nitrogen (N) in grazed and ungrazed semi-arid grassland ecosystems requires a detailed analysis of the different N pools and fluxes as well as ecosystem gains and losses. Since most studies focused on sub-processes of the N cycle (but. Coupland 1979, 1993, Brey Meyer and Van Dyne 1981), there is still a substantial lack of comprehensive reports of N balances from grazed semi-arid grassland ecosystems. Since Inner Mongolia natural grasslands are largely affected by excessive overgrazing (Zhou et al. 2002, Yu et al. 2004, Tate et al. 2004), the understanding the long-term effects of different land-use management on the ecosystem N balance is of particular value. Most terrestrial ecosystems are N limited (Vitousek 1982, Aerts and Chapin III 2000, Burke et al. 1997) and therefore the understanding of N dynamics is crucial to maintain a sustainable grassland productivity under grazing and prevent land degradation. Continuous grazing alters ecosystem functions in a multiple way, at different levels, and with different consequences, but grazing generally causes long-term N losses (Milchunas and Lauenroth 1993, Holland and Detling 1990 Pineiro et al. 2006, Zhou et al. 2007).

The MAGIM project, a DFG-financed research program which allowed for the simultaneous work of different research disciplines on the same sites offered the rare opportunity to combine information about the multiple N pathways and compounds of this semi-arid grassland ecosystem. The central aspect addressed in this chapter, thus, is the compilation and aggregation of data from several subprojects in order to quantify grazing effects on N cycling, internal fluxes and pools as well as N gains and losses. The first step is to define the system boundaries for the balance. We concentrate on the grazing area within the functional unit of a common sheep farm in Inner Mongolia. Three land-use practices were selected, a long-term heavily grazed site (annual stocking rate between 2 and 4 sheep/ha), a light to medium grazed site (average stocking rate 1 sheep/ha) with additional haymaking and a long-term grazing enclosure.

By analyzing the complete N cycle, we are able to compare different processes of the N cycle and their relevance in relation to the overall N dynamics. The identification of key processes within the N dynamics might indicate possible options to influence ecosystem functions and enable the application of adapted and sustainable land-use management practices.

## **Material and Methods**

### *Study area, grazing sites and stocking rates*

Detailed information about the study area and experimental sites are given in the general introduction (see chapter 1). Data included into this study, originated mainly from three investigated sites characterized by different grazing intensities. A long-term grazing enclosure (hereafter: UG79, position 43°33'10"N; 116°40'33"E), a moderate winter grazing site (WG, position 43°32'57"N; 116°40'04"E), and a long-term heavily grazed site (HG, position 43°34'54"N; 116°40'37"E). Information about the N pathway through sheep were sampled at a grazing experiment located close to the previously mentioned sites (position 43°34'00"N; 116°40'00"E).

We defined the grazing area of a typical grassland sheep farm as the unit for N balancing (Figure 1). In total we considered the N balance of three grazing intensities and land-use practices, respectively. A long-term heavily grazed site (annual stocking rate between 2 and 4 sheep/ha), a light to medium grazed site

(average annual stocking rate 1 sheep/ha) with additional haymaking and a long-term grazing enclosure. All N gains and losses, fluxes and pools of these different areas were calculated in  $\text{g N m}^{-2} \text{ year}^{-1}$ .

Stocking rates are not constant throughout the year since one third to half of the heard is sold in late summer. The bulk of a heard consists of mother sheep, each usually raising a lamb from early spring on. The year is roughly divided into three grazing periods. Summer grazing is carried out from June to September (120 days), during which the stocking rate is at maximum and calculated with 4 sheep/ha at site HG and 1.5 sheep/ha at site MG. Winter grazing occurs from October to January (120 days) with a reduced stocking rate (2 sheep/ha) at site HG and 0.75 sheep at site MG. Due to lambing and government regulations (early growing season grassland protection), the sheep are kept in paddocks from February to Mai (120 days with stocking rate 0 sheep/ha).

#### *Data collection*

Data of N fluxes and pools presented in this study were recruited from different MAGIM subprojects. A detailed listing of all the materials and method applied during the field campaigns would go beyond the scope of this chapter. For detailed information concerning the N pathway through sheep and excrements refer to Glindemann (2007). N gains and losses through dry deposition respectively wind erosion were measured by Hoffmann et al. (2007). Information about N stocks in soil profiles with special interest to the topsoil N dynamics as affected by grazing can be found in the study of Steffens et al. (2007). N mineralization rates and microbial dynamics are published by Holst et al. (2007). All data and methods concerning shoot and root biomass dynamics are explained in detail in the study of Gao (2007). N concentrations in groundwater were sampled by Schneider (2006) (unpublished data). Wet deposition was sampled during growing seasons 2005 (5 times) and 2006 (11 times). Collected rain samples were immediately deep frozen and later analyzed for  $\text{NH}_4^+\text{-N}$  and  $\text{NO}_3^-\text{-N}$  with a Bran+Luebbe rapid flow Autoanalyzer (B&L, Hamburg, Germany). The mean value of all samples of one growing season was extrapolated to annual rainfall amount. All N pools and fluxes not investigated by the MAGIM project were taken from published data sampled at the same location or in comparable ecosystems.

**Figure 1:** Functional unit of an Inner Mongolian sheep farm. Main building and paddock, where the sheep are enclosed overnight. The surrounding grassland of about 100 ha is grazed during growing season and partly during the winter. Picture from August 2006, at peak biomass time.



**Figure 2:** The sheep are locked in the paddock overnight. About 40% of the excrements are released to the paddock area. Mother sheep (dark white) with lamb (bright white) beginning of the growing season in Mai 2006.



**Figure 3:** The trampled dung in the paddock is accumulated on a dung heap. The dry dung is used as fuel for heating and cooking.



## Results

### *Nitrogen gains and losses*

The processes, which contribute to N input and output at the different grazing sites, are calculated to the unit of  $\text{g N m}^{-2} \text{ yr}^{-1}$ . The balance of these processes gives the net N gain or loss of sites characterized by different land-use (Table 1). The heavy and light to moderate grazing intensities were balanced with variable stocking rates over the year (see material and methods).

For all different land-use types, we quantified on the N input side the pathways of atmospheric deposition including rainfall and solid particles, biological  $\text{N}_2$ -fixation, overland flow, ascending groundwater and interflow. Possible pathways of N losses are leaching, interflow and surface runoff, wind erosion and  $\text{N}_2\text{O}$  emissions via denitrification and  $\text{N}_x\text{O}_x$  emissions. On grazed areas, N losses through volatilization ( $\text{NH}_3$ ) from urine patches, dung deposition to paddocks, and the sheep's wool and meat production were added. Furthermore, the N pathway through hay making as a very common land-use form is included (Table 1).

During the years 2005 and 2006, the annual N gains through wet deposition of  $\text{NH}_4^+\text{-N}$  and  $\text{NO}_3^-\text{-N}$  was calculated with approximately  $0.3 - 0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ . The average  $\text{NH}_4^+\text{-N}$  concentrations of rainwater were between  $1.2 - 1.4 \text{ mg NH}_4^+\text{-N l}^{-1}$  in 2005 and  $0.9 - 1.3 \text{ mg NH}_4^+\text{-N l}^{-1}$  in 2006. There was no pronounced seasonality of N concentrations in the collected rainwater. 90% of the precipitation occurred during the vegetation period from Mai to September.

N losses by leaching/drainage were not experimentally measured but are expected to be very small due to the low annual total rainfall in this semi-arid climate (long-term mean: 342 mm). The soil texture (sandy loam) and depth of the Ah horizon often reaching more than 1 m suggest a comparably high water holding capacity. Even after high intensity rainfall events the usually high evapotranspiration forces a quick depletion of soil water content and induces capillary rise rather than drainage. The insignificance of N losses by drainage is supported by the accumulation of secondary  $\text{CaCO}_3$  in the soil profiles and by data indicating very low groundwater N concentration (Schneider 2006, pers. communication). During the wintertime from October to April, soil water fluxes are almost entirely inhibited by frost.



**Table 1:** Nitrogen gains and losses ( $\text{g N m}^{-2} \text{ yr}^{-1}$ ) under different grazing intensity and land-use (HG = long-term heavy grazing, MG = moderate grazing and hay making, UG79 = long-term grazing exclosure since 1979) in semi-arid grassland ecosystem of Inner Mongolia based on measurements from 2004–2006 and literature data. Detailed explanations of the N fluxes are given in the text.

<b>Nitrogen pathway</b>	<b>HG (2-4 sheep/ha)</b>	<b>MG (1 sheep/ha)</b>	<b>UG79 (no grazing)</b>
Wet deposition	+0.3 – +0.5	+0.3 – +0.5	+0.3 – +0.5
Capillary rise, drainage	± 0	± 0	± 0
Lateral flow, surf. runoff	-0.05	± 0	± 0
Dry deposition	-0.2 – -0.6	-0.1 – +0.2	+0.3 – +1.0
N <sub>2</sub> -fixation	+0.1 – +0.2	+0.1 – +0.3	+0.1 – +0.2
N <sub>x</sub> O <sub>x</sub> fluxes	< -0.02	< -0.02	< -0.02
NH <sub>3</sub> emission	-0.1	-0.03	0
Hay	0	-1.0	0
Sheep meat	-0.1	-0.03	0
Sheep wool	-0.1	-0.03	0
Excrements	-0.5	-0.2	0
<b>Amount output (-)</b>	<b>1.0 – 1.5</b>	<b>1.3 - 1.4</b>	<b>± 0</b>
<b>Amount input (+)</b>	<b>0.4 – 0.7</b>	<b>0.4 – 1.0</b>	<b>0.7 – 1.7</b>
<b>Balance</b>	<b>-0.3 - -1.1</b>	<b>-0.3 - -1.0</b>	<b>0.7 – 1.7</b>

Nevertheless, high rainfall intensities during summer could exceed the infiltration rate of soils resulting in considerable surface runoff at heavily grazed areas, where soil is compacted by trampling (Krümmelbein 2007) and aboveground biomass is very low (Gao 2007). In order to consider these events, we assumed this process to happen with 10% of the total rainfall and the corresponding N amount in the rainwater ( $0.05 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) as loss through surface runoff.

The relevance of N gains via dry deposition and losses through wind erosion were surprisingly high compared to some other investigated processes. Hoffman et al. (2007) analysed the N concentration of dry deposition and estimated the



seasonal soil transport. They found considerable differences between both, sites and years. Ungrazed areas were usually sinks for dust and soil particles from atmospheric transport in the range of  $0.3 \text{ g N m}^{-2} \text{ yr}^{-1}$  in 2005 and  $1.0 \text{ g N m}^{-2} \text{ yr}^{-1}$  in 2006. At sites of medium grazing intensity these processes were balanced in both years. Heavy grazing areas lost approximately  $0.6 \text{ g N m}^{-2} \text{ yr}^{-1}$ .

Atmospheric  $\text{N}_2$  fixation through legumes or biological soil crusts is common in semi-arid and arid ecosystems. Biological soil crusts consisting of cyanobacteria, mosses, lichens and fungi were discontinuously observed at site WG and HG. At site UG79 soil crusts were rare due to dense vegetation cover. However, some legumes were identified at the ungrazed site (*Caragana microphylla*). Estimates of N inputs via biological soil crusts in the south-western US range from  $0.14$  to  $0.9 \text{ g N m}^{-2} \text{ yr}^{-1}$  (Belnap 2002).  $\text{N}_2$  fixation within semi-arid grassland of Inner Mongolia is assumed to be of minor importance since the proportion of legumes at grazed sites is very low and biological soil crusts are scattered and discontinuous. Biological soil crusts are sensitive to trampling. We assume that N inputs are balanced between grazed areas with more abundance of biological soil crusts and ungrazed areas hosting more legumes. Thus, at site UG79 and HG the N gain is calculated with  $0.1\text{--}0.2 \text{ g N m}^{-2} \text{ yr}^{-1}$ . However, at light and moderate grazing intensities the low trampling disturbance might support abundance and activity of biological soil crusts and  $\text{N}_2$  fixation. Here we assumed an annual input of (up to)  $0.3 \text{ g N m}^{-2} \text{ yr}^{-1}$ .

$\text{N}_2\text{O}$  and  $\text{NO}_x$  losses to the atmosphere were measured at the different sites by Holst et al. (2007) and were generally lower than  $0.02 \text{ g N m}^{-2} \text{ yr}^{-1}$  and, therefore, more or less negligible for the total N budget. The  $\text{NH}_3$  emissions from urine patches were estimated with 30% of the urea-N applied (Schimel et al. 1986, Ruess et al. 1988), which equals  $0.1 \text{ g N m}^{-2} \text{ yr}^{-1}$  (see nitrogen cycling).

Hay making is a common land-use practise on moderate to light grazing areas, where enough dry matter is left over from animal consumption. Also remote grassland areas with long distance to water resources and villages are commonly used for haymaking. Usually, the hay is cut one time the year in august down to a height of approximately 5 cm. Around 50% of the annual ANPP is harvested by hay making which equals  $50 \text{ g dry matter m}^{-2}$  with an average N concentration of 2%. From these data we calculated a loss of  $1.0 \text{ g N m}^{-2} \text{ yr}^{-1}$  from a hay making area.

The loss of N through sheep excrements (urine and faeces) is estimated as  $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$  at site HG and  $0.2 \text{ g N m}^{-2} \text{ yr}^{-1}$  at site MG. This amount corresponds to the approximately 40% of faeces and urine not redistributed to the grazing area since sheep are kept in paddocks overnight (Figure 2).

The N stored in sheep bodies is estimated as 3% of the live body weight and sheep with an average body weight of 25 kg contain around 750 g N. At a heavily grazed site (2 - 4 sheep per ha) we estimate an annual amount of  $0.23 \text{ g N m}^{-2} \text{ yr}^{-1}$  stored in sheep bodies (calculated with average stocking rate of 3 sheep per ha and 750 g N per sheep). Under the assumption that at least one third to half of the heard is sold every year, we calculate an N loss of  $0.1 \text{ g N m}^{-2} \text{ yr}^{-1}$ . The N concentration of dry wool was around 15% and approximately 2 kg of wool is harvested from every mother sheep per year, which results in N loss of about  $0.1 \text{ g N m}^{-2} \text{ yr}^{-1}$  at site HG. Together by harvesting meat and wool, a heavily grazed area loses approximately  $0.2 \text{ g N m}^{-2} \text{ yr}^{-1}$ .

### *Nitrogen cycling*

The system's internal N fluxes and compartments are illustrated in a flow chart (Figure 1). All values are given in  $\text{g N m}^{-2} \text{ yr}^{-1}$ .

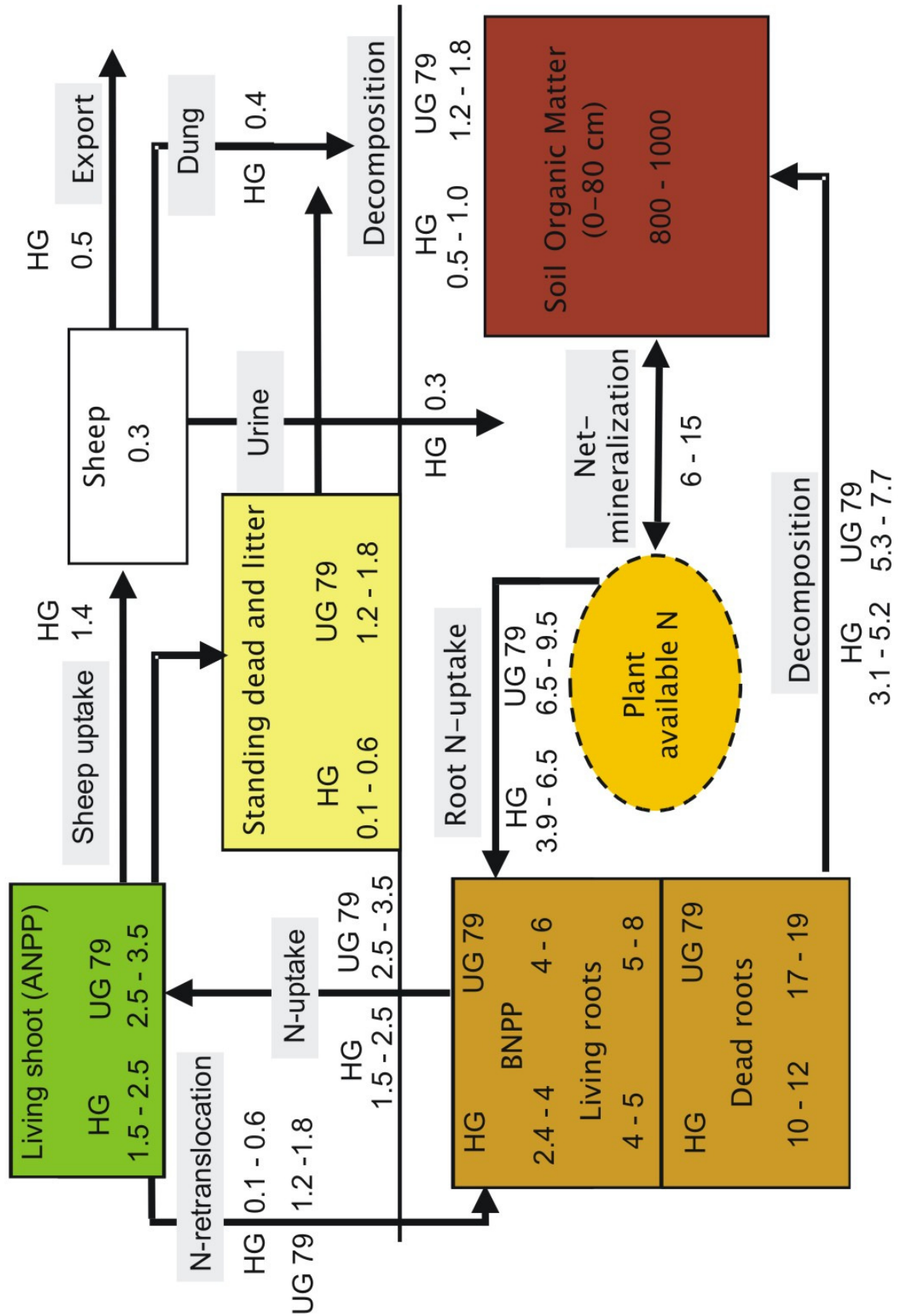
The annual mean aboveground net primary productivity (ANPP) of the semi-arid grasslands in Inner Mongolia is highly depending on rainfall variability (Bai et al. 2004). Since the fodder basis changes from year to year, the proportion of the ANPP consumed by grazers is variable under the assumption of a constant stocking rate. Using the moving cages method, ANPP was between 75 and  $120 \text{ g m}^{-2} \text{ yr}^{-1}$  (average  $87 \text{ g m}^{-2} \text{ yr}^{-1}$ ) during 2004–2006 on site HG. ANPP of the long-term ungrazed site was significant higher in all sampled years with an average of  $125 \text{ g m}^{-2} \text{ yr}^{-1}$ . Site MG exhibited an annual average ANPP of  $105 \text{ g m}^{-2} \text{ yr}^{-1}$  (Gao 2007). During the growing season, the average plant aboveground N content (leaves and stems) was between 2 and 2.5% N (Gao 2007, Schiborra et al. 2006). The total amount of N located to shoot during a growing season was, therefore, around  $1.5\text{--}2.5 \text{ g N m}^{-2} \text{ yr}^{-1}$  at site HG and  $2.5\text{--}3.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ , estimated according to the fluctuations in productivity and shoot N concentrations.

The ration of belowground net primary productivity (BNPP) to ANPP was estimated as 0.6 (Gao 2007) and, thus, 60% of the total annual net primary productivity (NPP) is allocated to the soil. The annual belowground N uptake

allocated to the roots was estimated to vary from 2.4–4 g N m<sup>-2</sup> yr<sup>-1</sup> (HG) and 4–6 g N m<sup>-2</sup> yr<sup>-1</sup> (UG79). Accordingly, the total plant N uptake was calculated as 3.9–6.5 g N m<sup>-2</sup> yr<sup>-1</sup> at site HG and 6.5–9.5 g N m<sup>-2</sup> yr<sup>-1</sup> at site UG79.

Belowground organs of perennial grasses can live several years and the considerable quantity of standing root biomass can store nutrients translocated from the aboveground parts during the winter. Belowground living and dead root N was calculated from belowground biomass measurements assuming 1.0% and 2.0% N concentration in dead and living roots, respectively. Belowground live root N was therefore estimated as 8 and 10 g N m<sup>-2</sup> for site HG, 10–12 g N m<sup>-2</sup> for site MG and 10–16 g N m<sup>-2</sup> at UG79. The total belowground root biomass N including the dead parts were calculated to be 21 g N m<sup>-2</sup> at site HG, 25 g N m<sup>-2</sup> at MG and 32 g N m<sup>-2</sup> at site UG79 (Gao 2007).

The average daily dry matter consumed per sheep can be assumed to be 0.7 kg (Glindemann 2007, pers. communication) or 0.07 g m<sup>-2</sup> at a stocking rate of 1 sheep/ha. Based on the average forage N content of 2 - 2.5% during the growing season, the average daily N uptake per sheep was calculated as 14–17.5 g N per day. A grazing time of 120 days (June–September) was estimated for the growing season, another 120 days was classified as winter grazing time from October to January. During this time the N uptake is reduced due to limited fodder quality. The N content of standing dead was between 0.5 and 0.9 % (see chapter 2). For the rest of the year, sheep are assumed to stay in the paddock. Thus, under heavy grazing around 50 g m<sup>-2</sup> grassland dry matter is consumed per year during the growing season (120 days) and winter grazing (120 days), which equals the proportion of almost 60 % of an average ANPP during the sampling years 2004–2006. For the N balance we calculated the annual uptake at site HG by sheep with 1.42 g N m<sup>-2</sup> yr<sup>-1</sup> resulting from 120 days grazing during growing season and 120 days winter grazing. During the rest of the year sheep are fed with hay originating from other grassland areas, outside the balanced system.



**Figure 4:** Flow chart of the N-cycle of semi-arid grassland of Inner Mongolia as influenced by heavy grazing (HG: 2 – 4 sheep ha stocking rate) and long-term grazing enclosure (UG 79). N-fluxes and pool sizes calculated under *steady state* conditions in  $g\ N\ m^{-2}\ yr^{-1}$ . Based on data sampled between 2004 and 2006.

From the annual N uptake by sheep at site HG ( $1.42 \text{ g N m}^{-2} \text{ yr}^{-1}$ ), roughly 10% is used to build up the sheep body mass, the bulk is released as faeces and urine. The average N amount stored in sheep bodies at site HG is calculated as approximately  $0.3 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Sheep are usually enclosed overnight, since the grazing area is mostly not fenced and sheep might be stolen (Figure 4). This behavior is an important point to address since faeces and therefore considerable N amounts are not redistributed to the grazing area. Approximately 40% of the faeces are dropped into the paddock and 60% remain on the grassland (Glindemann 2007, pers. communication). Faeces from the paddock are collected and piled on a heap (Figure 3) and later on used as fuel for heating and cooking. The average amount of faeces per sheep and day is 600 g dry matter with an average N content of 10 g (Glindemann 2007, pers. communication). The average amount of urine was estimated as 1 Liter per day and sheep with an N content of  $10 \text{ g l}^{-1}$ . N losses through volatilization from urine patches depend on vegetation and soil urease activity, pH, moisture, cation exchange capacity, and texture and was estimated to vary from 10 to 40% of urea-N (Schimel et al. 1986, Ruess et al. 1988). Annual volatilization losses (calculated with 25% of urine-N) from grazing area and paddock are estimated as  $0.18 \text{ g N m}^{-2} \text{ yr}^{-1}$  and a total amount of N lost from the area to the paddock of  $0.49 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Under consideration of seasonal grazing times, day-night differences and volatilization losses, the annual N redistribution to the grazing area via urine was calculated to be  $0.32 \text{ g N m}^{-2} \text{ yr}^{-1}$  and via faeces  $0.42 \text{ g N m}^{-2} \text{ yr}^{-1}$ .

In order to consider the N flux by shoot decomposition, N retranslocation to perennial root organs is relevant in semi-arid grassland systems. We calculate the N retranslocation into perennial organs at the end of the growing season with 50% based on measurement of seasonal tissue N content of shoot and standing dead material. Thus, at the ungrazed site the N amounts entering the pool representing shoot decomposition were reduced to  $1.2\text{--}1.8 \text{ g N m}^{-2} \text{ yr}^{-1}$ . The amount of litter available for decomposition at site HG is strongly depending on the herbivore consumption. Assuming an aboveground consumption of 60% ANPP, the N amount entering the shoot decomposition process at site HG is reduced to  $0.2\text{--}0.4 \text{ g N m}^{-2} \text{ yr}^{-1}$ .

The N content of the soil organic matter pool in temperate semi-arid grassland is large compared to the annual N fluxes. Titlyanova and Bazilevich (1979)

calculated 1400 g N m<sup>-2</sup> stored in the soil organic matter of Siberian steppe. According to Steffens (2007) the total soil total N content is between 800–1000 g N m<sup>-2</sup> in this grassland system. Significant differences in topsoil (0-4 cm) N content were detected between the grazing sites. The N content of sites UG79 and WG with 120 g m<sup>-2</sup> was significantly higher compared to that of site HG with 90 g m<sup>-2</sup> (Steffens et al. 2007). Net microbial N turnover rates [(microbial ammonification + nitrification) – microbial immobilization] within soil depth layer 0-10 cm at site WG were 6.6 g N m<sup>-2</sup> yr<sup>-1</sup> in 2004. Measurements in the dry year 2005 indicate a pronounced effect of year as well as difficulties to predict real N turnover rates at the three differently managed sites. Net microbial N turnover with 15 g N m<sup>-2</sup> yr<sup>-1</sup> was very high at site UG 79, while no net mineralization was measured at sites HG and site WG (Holst et al. 2007).

## Discussion

### *N gains and losses and grazing management options*

According to the balance of gains and losses of N from different grazing and land-use types, ungrazed sites represent N sinks with an annual N accumulation between 0.7–1.7 g N m<sup>-2</sup> yr<sup>-1</sup>, while heavy grazed areas are N sources with losses up to 1.5 g N m<sup>-2</sup> yr<sup>-1</sup>. Areas of moderate grazing only are N sources if combined with haymaking. The land-use form of haymaking causes highest N losses per area.

Dry deposition was identified as the main pathway contributing to N accumulation with more than 1 g N m<sup>-2</sup> yr<sup>-1</sup>. Concurrently, wind erosion counts for high N losses with up to 0.6 g N m<sup>-2</sup> yr<sup>-1</sup>. Wet deposition is of second highest relevance for N input with up to 0.5 g N m<sup>-2</sup> yr<sup>-1</sup>. At ungrazed areas, the aboveground biomass accumulation is characteristically very high (Bakker 1997), which increases surface roughness and reduces the wind speed near soil surface (Facelli and Pickett 1991). Dust and sand particles transported in the lower atmosphere are deposited at a particle size dependant threshold value of wind speed. On the other hand, with increasing grazing intensity the surface roughness decreases and the resulting higher wind speed keeps dust particles in the air or even is able to take up particles from the surface. This situation is realized at moderate and heavy

grazed areas, where dry deposition is balanced or respectively negative. Wind erosion is one important factor making heavy grazed areas to N sources. Bare surfaces also increase rainwater runoff during high intensity precipitation events (Kurz et al. 2006). N losses through surface runoff at heavy grazed areas might be much higher than calculated ( $0.05 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) because organic matter at the soil surface is washed out from the area as well.

Grazing as a land-use form contributes to considerable N losses due to the common management practice to keep the sheep in paddock overnight. If the sheep dung would be redistributed to the area, the annual losses could be reduced by 20–30% or the amount of  $0.3 \text{ g N m}^{-2} \text{ yr}^{-1}$  at site HG. N losses due to meat and wool harvest are comparable small and would be sustainable as soon as stocking rate is reduced to a level where the remaining aboveground biomass prevents wind erosion. Hay making was identified as the land-use form causing highest N losses to the grassland through biomass export. At these areas, productivity will most likely decrease due to limited nutrients in the long run. We suggest rotational grazing systems, where between the years grazing is alternated with hay-making.

A general problem of grazing and land management is rainfall variability between the years and within one growing season. The amount of hay harvested is varying according to the total annual grassland productivity, and therefore adapted to the environmental changes. However, as soon as a more or less fixed number of sheep graze throughout the growing season the amount of grassland consumption is fixed and fluctuations of productivity are not respected. In a drought year sheep can consume almost all biomass produced with severe consequences for the area with respect to wind and water erosion and finally for the N balance. Furthermore, excessive grazing during a drought year will have extended effects to the following growing seasons by the altered plant species competition. These effects between growing seasons complicate the finding of optimal stocking rates. To adjust the sheep number dynamically to grassland productivity usually can't be realized without financial losses to the farmer.

The extreme drought during the growing season 2005 had obviously prolonged effects into the next growing season. Although the precipitation during the growing season 2006 was comparable to the amounts of the year 2004 according to long term meteorological data recorded by IMGERS (Inner Mongolia Grassland

Ecosystem Research Station), the ANPP was significantly lower in 2006. This suggests that previous years' precipitation pattern affect next growing seasons grassland productivity. September rain is stored in the soil and transpiration losses are limited due to plant senescence and the low temperatures of the season. The frost period starts in October and lasts until next spring end of April beginning of Mai. The stored last years late season rain is used for early seasons plant production. This basically offers management options since last years precipitation pattern might be used as indicator to find accurate the stocking rates of the following growing season.

### *N-cycling*

Since annual processes characterize the aboveground ecosystem dynamic, the tracking of N fluxes and pools is relatively simple compared to the complex belowground N pathways. By the measurement of ANPP under grazing conditions on the one hand (Gao, 2007) and the detailed analysis of N uptake and release by sheep on the other hand (Glindemann 2007, pers. communication), we achieved a relatively complete knowledge about N-pathways from the consumer site. Under fixed stocking rates, the proportion of consumed ANPP is depending on system productivity, which is in turn affected by the precipitation pattern. Accordingly, N retranslocation is highly variable at site HG and belowground N storage for the next growing season or during drought periods is strongly reduced. As a consequence, plant communities at these areas are much more sensitive to natural disturbances since regeneration potential is limited. The grazing impacts are partly compensated by adapted, grazing tolerant plant communities (see chapter 5), which on the other hand might be of lower fodder quality.

Under the current grazing management practices, roughly about half of the consumed N by sheep from the grazing site is returned. The possible acceleration of N cycling through dung and urine deposition by herbivores (Holland and Detling 1990, McNaughton, et al. 1997; Frank et al. 2000) was not confirmed by compensatory growth (Gao 2007) or increased N availability under grazing (see chapter 3). This could partly be explained by the general management practice whereby the sheep are kept in the paddock at night and thus the amount of N redistributed to the grazing site is reduced.



Detailed analysis of the litter decomposition process revealed neither a grazing impact on the plant litter decay rates via litter quality nor location effects (see chapter 2). However, grazing reduces the aboveground biomass (standing dead and litter) and therefore the amounts of N returned through litter decomposition.

Analysing the system internal N cycling with regard to belowground fluxes and pool sizes, the compartment of the soil organic matter (SOM) is outstanding with enormous N contents of about  $1000 \text{ g N m}^{-2}$ . This in relation to all other N pools and fluxes huge belowground N pool dominates the N dynamics of this grassland ecosystem. Under the conservative estimation of 1% annual net mineralization rate of SOM ( $8\text{--}10 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) the total N net uptake requirements by plants estimated with  $6.5\text{--}9.5 \text{ g N m}^{-2} \text{ yr}^{-1}$  are already covered. In fact, the net mineralization rates measured for microbial turnover processes of  $6\text{--}15 \text{ g N m}^{-2} \text{ yr}^{-1}$  are calculated to the upper 0-10 cm of soil (Holst et al. 2007). Thus, already the topsoil mineralization rates would fulfill the requirements of total annual NPP. Adding the mineralized N from deeper horizons, where process intensities are reduced but not completely inhibited, this grassland ecosystem should hardly be N limited. Even under the assumption of continuous severe N losses caused by over-grazing and attributed consequences, the average total N pool stored in the soil should be able to support grassland productivity for decades. But obviously potential ecosystem fertility not necessarily guarantees plant available N. Mineralization rates of the SOM pool interact with water availability and temperatures (Fang et al. 2005). Furthermore, during some measuring periods of the growing season, net mineralization rates were strongly reduced due to high microbial N immobilization rates (Holst et al. 2007), indicating complex interactions between different soil N compartments with varying results for plant N availability. Ion exchange measurements and soil N extractions confirm the high fluctuations in N availability (see chapter 3). Also, the above- and belowground decomposition, a key process in N cycling (Aerts et al. 2003) shows pronounced disparities with significant consequences for N dynamics (see chapter 2). In fact, the processes of N mineralization and N availability in semi-arid grassland in interaction with rainfall variation and wet-dry cycles are poorly understood (Yahdjian et al. 2006). Nevertheless the large N stocks in the soil, representing high potential ecosystem fertility, might play an important role for ecosystem restoration and regeneration. As long these N stocks are not lost through land degradation processes, options

for a sustainable land management might still be available for natural Inner Mongolia grassland ecosystems.

Some data reported in this study were not measured on all sites, let alone all years. This certainly reduces the up-scaling possibilities. This disadvantage is more affecting the internal N cycling than the balance of gains and losses. On the other hand the experimental sites were carefully selected and N data derived from sheep, dry and wet deposition are less sensitive to the selected sites. Within the balance uncertainties exist with regard to the contribution of biological N<sub>2</sub> fixation. During our studies we analyzed three growing season of average and very low precipitation, we did not experience the situation of a comparable wet growing season which most likely affect matter fluxes in relation to the grazing intensity and land-use practice. Due to the extreme dry growing season in 2005, we collected a data set representing the 'bottom' of N dynamics in a semi-arid grassland ecosystem, which is of a certain value as well. In this situation, grazing and land-use systems are definitely more critical to the ecosystem processes since resources are more limited. In this way we were able to study these processes most probably contributing to the increasing degradation observed in northern Chinese grassland ecosystems.

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## How grazing affects plant communities and environmental parameters in a semi-arid grassland of Inner Mongolia

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### Abstract

Grazing effects on ecosystem functions are variable. We ask for environmental parameters as altered by grazing and their effects on plant community structure in a semi-arid grassland ecosystem extensively affected by excessive overgrazing.

Five grazing intensities were analysed at totally 450 sample positions, including a 26 year long-term enclosure, intermediate and heavy grazing sites. Canonical correspondence analysis (CCA) was applied to link plant species composition, abundance and cover to variables of nutrient availability (N, P, K), topsoil water content, aboveground biomass (AGB), organic carbon (OC), total soil N and S concentration, pH and bulk density.

The long-term enclosure showed highest N availability, highest OC and total N concentration, lowest bulk density and high AGB. The heavy grazed site was characterised by low topsoil water content and reduced N availability as well as low OC and total N concentration, high bulk density and very low AGB. At both

sites dicotyledonous species showed high abundances. Graminoids dominated intermediate grazing sites. Here K availability was low, while other soil parameters ranked medium. Differences in species cover, abundance and composition among grassland sites were best explained by topsoil water content in combination with N and K availability. Plant species were classified as responding to specific environmental conditions.

The grazing effect on water availability was surprisingly high and might be generally underestimated. The results provide a basis for the development of land-use strategies within conflicting interests of live-stock farming, grassland regeneration and conservation.

**Key-words**

CCA, environmental parameters, land-use, species composition, steppe

## Introduction

The temperate steppe ecosystems of northern China have been exposed to a significant population growth and distinct land-use changes in the recent decades. The traditional extensive, semi-nomadic sheep grazing system changed to seriously intensified stationary livestock farming. For the autonomous region of Inner Mongolia a continuous overstocking in relation to ecological resources was reported (Yu et al. 2004). Around one third of China's total natural grasslands (ca. 2.8 Mio km<sup>2</sup>) is nowadays affected by land degradation (Zhou et al. 2002), most visible as changes of vegetation, soil properties and erosion processes (Tong et al. 2004; Su et al. 2005). Dust storms from central Asia might be known as most prominent off-site effects. In consideration of these far-reaching problems, consolidated knowledge about grazing and ecosystem interaction of Inner Mongolia grasslands is needed to improve land-use systems and develop conservation and rehabilitation strategies.

Different and partly controversial results from natural grass- and rangelands led to discussions and modelling approaches about the variable ways how grazing alters ecosystem functions and vegetation dynamics (e.g. Milchunas et al. 1988, Olf & Ritchie 1998). Particularly for arid and semi-arid ecosystems, the grazing effect often interferes with highly variable climate factors (Varnamkhasti et al. 1995, Biondini et al. 1998, Fernandez-Gimenez & Allen-Diaz 1999). Studies about long-term site developments are required (Bai et al. 2004).

In general, a change of species composition (*sensu* Grime 1977, Bakker 1998) is considered to be a fast response to grazing, while changes of productivity or soil nutrient parameters represent intermediate, slow- or long-term effects (Milchunas & Lauenroth 1993). Nutrient dynamics altered by grazing are often the focal point of studies (Mazancourt et al. 1998, Proulx & Mazumder 1998, Le Roux et al. 2003, Bakker et al. 2004, Loiseau et al. 2005). On a local scale, grazing accelerates or decelerates nutrient cycling. Plant nutrient availability was either found to be increased (Holland & Detling 1990; McNaughton, et al. 1997; Frank et al. 2000) or decreased (Biondini et al. 1998, van Wijnen et al. 1999). Since particularly N limits the productivity of many terrestrial ecosystems (Vitousek 1982), nutrient dynamics altered by grazing might also function as a key long-term factor affecting the vegetation of Inner Mongolia grasslands.

The consequences of overgrazing in Inner Mongolia are studied only fragmentary with partly contrasting results: Su et al. 2006 found increasing grazing intensity accompanied by increased soil compaction while soil organic matter and concentrations of total N and OC decreased. Available N and P decreased, while total and available K was not affected in a study of Xie & Wittig (2004). Contrarily, Barger et al. (2004) found increased N mineralisation with increasing grazing intensity. Fernandez-Gimenez & Allen-Diaz (2001) found soil P and K concentrations in Mongolia grasslands positive correlated with grazing pressure and concurrently the driving variables of ordination analysis. Plant communities responded to grazing with reduced abundance of characteristic perennial grasses like *Stipa grandis*, *Leymus chinensis* and *Koeleria cristata* (Chen et al. 2005, Xie & Wittig 2003), while dicotyledonous species increased (Fernandez-Gimenez & Allen-Diaz 1999). *Artemisia frigida* and *Potentilla acaulis* as well as the C<sub>4</sub> grass *Cleistogenes squarrosa* were identified as important grazing increasers in degraded Inner Mongolia grasslands (Li 1989; Liang et al. 2002; Barger et al. 2004).

Our study area, the Xilin River Basin represents a typical grassland ecosystem of central Asia (Bai et al. 2004). Here the land degradation processes are proved on more than 70% of the area (Tong et al. 2004). To investigate these effects we analysed five sites of different grazing intensity, including a 26 year long-term enclosure and a more than 30 years heavy grazed site. We applied canonical correspondence analysis (CCA), to link plant species composition and cover with variables of nutrient availability, aboveground biomass (AGB), organic carbon (OC), total soil N and S concentration, topsoil water content, pH and bulk density. Special attention was paid to nutrient availability of N, P and K, which entered the CCA as variables measured as time-integrated nutrient fluxes recorded by resin capsules. If nutrient availability is a major long-term factor altered by grazing, we hypothesize that a substantial part of species variability between the sites should be explained by different nutrient availability. If not, what parameters can be identified instead?

In this study of a semi-arid grassland ecosystem affected by degradation we (1) analyse how grazing alters environmental parameters, (2) show how plant communities respond to these changed environmental conditions and (3) finally



discuss consequences for land-use and future research within conflicting interests of live-stock farming, grassland regeneration and conservation.

## **Material and Methods**

### *Study area and experimental sites*

Detailed information about the study area and experimental sites are given in the general introduction part. During this study, we investigated five sites characterized by different grazing intensities. A long-term grazing enclosure (hereafter: UG79, position 43°33'10"N; 116°40'33"E), a short-term grazing enclosure (UG99, position 43°33'05"N; 116°40'15"E), a moderate winter grazing site (WG, position 43°32'57"N; 116°40'04"E), a continuous grazed site (CG, position 43°33'09"N; 116°40'02"E) and a long-term heavily grazed site (HG, position 43°34'54"N; 116°40'37"E).

### *Sampling design*

Data were collected from June to September of 2004. Beside a delayed begin of summer rains, this year offered average climatic conditions according to long-term meteorological data recorded at IMGERS. Altogether 548 sampling points were installed in a grid design at the sites. Sample areas covered between 1.4 ha represented by 100 sample points in 5 to 15 m distance at sites HG, UG99 and UG79 and 16.5 ha represented by 124 sample points in 10 to 50 m distance at sites WG and MG.

### *Vegetation, soil, water and nutrient analysis*

Vegetation was analysed by the combined recording of individual numbers and cover proportion for every species (cover abundance classes: r, +, 1, 2a, 2b, 3, 4, 5) on all sample points within 1 x 1 m plots. The results were transformed into average values of cover percent (0.001, 0.5, 2.5, 10, 20, 37.5, 62.5, 87.5). AGB was sampled at peak time on 0.25 m x 0.25 m plots at all grid points in August 2004.

Soil samples from the upper 4 cm at each grid position were analysed for bulk density, total C, total N, total S and pH. Total C, N and S concentrations were determined by dry combustion on a Vario Max CNS elemental analyser

(Elementar Analysensysteme GmbH, Hanau). All samples were free of carbonate so that the total C concentration equals the organic carbon (OC) concentration. Soil pH was determined in a 1:2.5 soil to 0.01 M CaCl<sub>2</sub> solution.

Average topsoil water content (0-6 cm) of each grid point was measured with capacitance sensors (Theta probe) at 12 dates during growing season 2004 (29.06., 03.07., 07.07., 14.07., 20.07., 27.07., 12.08., 23.08., 02.09., 14.09., 16.09. and 21.09.). Sampling dates were designed to include seasonal dry periods, rainfall events and their dry-off dynamics.

Nutrient availability was measured by ion exchange resin capsules (PST-1, UNIBEST, Bozeman Montana, USA) at all grid points (248) of sites WG and MG and 80 grid points at sites HG, UG79, and UG99. Mixed bed resins act like a strong sink for cations and anions by the exchange of H<sup>+</sup> and OH<sup>-</sup>-ions into the soil solution. The nutrient flux into the capsule is determined by ion concentration of soil solution, diffusion rates and capsule surface area (11.4 cm<sup>2</sup>). Capsules were installed into the main rooting zone at 15 cm depth and remained at the sites from early June to early September measuring cumulative nutrient fluxes. We followed instructions of UNIBEST to analyse the amount of ammonium, nitrate, phosphorus and potassium exchanged by capsules (see chapter 3).

#### *Data analysis*

CCA was carried out using PC-ORD 4.0 software based on those 450 sample points, which exhibit data of all ten variables. We chose CCA to explain the community structure related to environmental parameters. In the ordination, we used biplot scaling by optimizing for species and 'LC' scores, the axis are formed by linear combinations of environmental variables. The main (species) matrix included 53 grassland species showing abundance higher than 5. The second matrix comprised variables of topsoil bulk density, pH-value, OC, total N, and total S concentration (0-4 cm), average water content (0-6 cm), nutrient availability of N, P and K and AGB.

Statistical analysis were performed by using SPSS Version 13. One way ANOVA and Tukey HSD at  $p = 0.05$  was used for multiple comparison of means.

## Results

### *Grazing intensities and environmental variables*

AGB increased with decreasing grazing intensity, and was highest in UG99 (Table 1). With decreasing grazing intensity, OC, total N, total S concentrations and water contents increased and bulk density decreased. Topsoil total N and resin N were highest at UG79. P availability was higher at grazed sites. While high values of K availability were found at sites HG and UG79, lower values were recorded at MG, WG and UG99. Topsoil pH ranged between 6.6 and 6.8.

**Table 1:** Mean values (ANOVA results, Tukey HSD,  $p = 0.05$ ) of parameters sampled at different grazing intensities in summer 2004 at long-term enclosure since 1979 (UG79) and 1999 (UG99), winter grazing (WG), moderate grazing (MG) and heavy grazing (HG) of Inner Mongolia grassland ecosystem.

site	n	OC <sup>a</sup>	N <sub>tot</sub> <sup>a</sup>	S <sub>tot</sub> <sup>a</sup>	AGB <sup>b</sup>	Bulk-Dens. <sup>a</sup>	Water (0-6 cm) <sup>c</sup>	Resin-N <sup>d</sup>	Resin-P <sup>d</sup>	Resin-K <sup>d</sup>	pH <sup>a</sup>
		(mg/g)			(g/m <sup>2</sup> )	(g/cm <sup>3</sup> )	(m <sup>3</sup> /m <sup>3</sup> )	(mg m <sup>-2</sup> d <sup>-1</sup> )			
HG	75	16.8 <sup>a</sup>	1.7 <sup>a</sup>	0.27 <sup>a</sup>	132 <sup>a</sup>	1.28 <sup>a</sup>	0.15 <sup>a</sup>	1.71 <sup>a</sup>	0.27 <sup>c</sup>	3.43 <sup>c</sup>	6.6 <sup>a,b</sup>
MG	103	22.8 <sup>b</sup>	2.4 <sup>b</sup>	0.29 <sup>a</sup>	164 <sup>b</sup>	1.17 <sup>b</sup>	0.18 <sup>b</sup>	2.85 <sup>a</sup>	0.25 <sup>c</sup>	1.13 <sup>a</sup>	6.6 <sup>a</sup>
WG	119	25.9 <sup>c</sup>	2.7 <sup>c</sup>	0.37 <sup>b</sup>	168 <sup>b</sup>	1.09 <sup>c</sup>	0.19 <sup>b</sup>	1.69 <sup>a</sup>	0.26 <sup>c</sup>	0.79 <sup>a</sup>	6.7 <sup>b,c</sup>
UG99	77	25.5 <sup>c</sup>	2.6 <sup>c</sup>	0.35 <sup>b</sup>	399 <sup>d</sup>	1.09 <sup>c</sup>	0.21 <sup>c</sup>	2.51 <sup>a</sup>	0.09 <sup>a</sup>	0.73 <sup>a</sup>	6.8 <sup>c</sup>
UG79	76	31.2 <sup>d</sup>	3.2 <sup>d</sup>	0.42 <sup>c</sup>	313 <sup>c</sup>	0.94 <sup>d</sup>	0.21 <sup>c</sup>	6.20 <sup>b</sup>	0.13 <sup>b</sup>	2.32 <sup>b</sup>	6.6 <sup>a</sup>

<sup>a</sup> values from topsoil 0-4 cm, OC = organic carbon.

<sup>b</sup> aboveground biomass at 1 cm, incl. dead parts.

<sup>c</sup> topsoil water content 0-6 cm, mean values of 12 measurements.

<sup>d</sup> nutrient flux into resin capsules (see material and methods for unit explanations).

### *CCA Analyses*

The three interpreted axis formed by the variables which entered CCA explain together 15.7% (9.2, 4.9 and 1.6%) of the data variance, with a total variance ("inertia") in the species data of 3.28. All three axis passed Monte Carlo permutation test at  $p = 0.001$  for significant eigenvalues as well as species environment correlations.

Analysing the variance explained by a single variable, surprisingly topsoil water content is able to account for 7.7% of data variance (Table 2), followed by BD (6.3%), N<sub>tot</sub> (5.8%), OC (5.6%) and AGB (4.0%). The variables of nutrient availability only account for 3.8% (K), 3.6% (N) and 2.4% (P) of data variance. The variables of S<sub>tot</sub> (2.4%) and pH (0.9%) only explain minor variance. The best

combination of variables however is realized by water and nutrient availability of K or N, explaining 10.8% respectively 10.5% of data variance, all three variables together cover already 12.5% of the 15.7% variance explained by all 10 variables. Intraset correlations (ter Braak 1986) for the three interpreted axis are shown in Table 3. For the two illustrated axis in the biplot (Fig. 1 and 2), axis 1 has highest correlation to water content followed by bulk density, OC and total N concentration. The second axis was mainly formed to represent the nutrient availability of different sites and correlates most to N and K availability.

**Table 2:** Canonical correspondence analysis (CCA): results for % of variation in species data explained by selected variables combinations at  $p < 0.01$  (W = water content, BD = bulk density, OC, N<sub>tot</sub>, S<sub>tot</sub> = organic carbon, total nitrogen or total sulphur of topsoil, AGB = aboveground biomass, rN, rK, rP = nutrient availability by resin capsules for nitrogen, potassium and phosphorous, pH = topsoil pH-value).

One variable	% of variance explained	two variables	% of variance explained	Other combinations	% of variance explained
<b>W</b>	7.7	<b>W, rK</b>	10.8	<b>BD, rK, rN</b>	11.5
<b>BD</b>	6.3	<b>W, rN</b>	10.5	<b>W, BD, rN</b>	12.0
<b>N<sub>tot</sub></b>	5.8	<b>BD, rK</b>	10.0	<b>W, rK, rN</b>	12.5
<b>OC</b>	5.6	<b>W, BD</b>	9.7	<b>W, BD, rK, rN</b>	13.5
<b>AGB</b>	4.0	<b>N<sub>tot</sub>, rK</b>	9.5	<b>W, BD, rK, rN, N<sub>tot</sub></b>	14.0
<b>rK</b>	3.8	<b>W, N<sub>tot</sub></b>	9.3	<b>W, BD, rK, rN, N<sub>tot</sub>, AGB</b>	14.8
<b>rN</b>	3.6	<b>W, rP</b>	8.6		
<b>rP</b>	2.4	<b>BD, rN</b>	8.4	<b>all 10 variables</b>	15.7
<b>S<sub>tot</sub></b>	2.4*	<b>BD, N<sub>tot</sub></b>	7.5		
<b>pH</b>	0.9				

\*  $p = 0.02$

**Table 3:** Canonical correspondence analysis (CCA): Intraset correlations of ter Braak (1986) between the three interpreted axis and the environmental variables. Axis 1 and 2 are shown in the biplot (Fig. 1 and 2). Further unit explanations see Table 1.

	OC	N <sub>tot</sub>	S <sub>tot</sub>	AGB	Bulk-Dens.	Water	Resin-N	Resin-P	Resin-K	pH
<b>Axis 1</b>	0.72	0.75	0.43	0.55	-0.74	0.90	0.29	-0.38	-0.45	0.08
<b>Axis 2</b>	-0.37	-0.31	-0.33	-0.28	0.48	-0.28	-0.70	0.36	-0.62	0.29
<b>Axis 3</b>	-0.10	-0.15	-0.03	0.66	0.04	0.16	-0.45	-0.35	-0.05	0.40

This general orientation of the dataset illustrated by the ordination axis is also represented by correlations of sampled parameters shown in Table 4. On the one hand, correlations are generally positive between water content, OC and total N concentration as well as AGB and negative between bulk density and the previously mentioned variables. On the other hand, considerably less, respectively almost no correlations of the parameters mentioned above were found with N and K availability (e.g. water and resin-N had only a weak correlation of 0.30), which underlines the relative independent position of nutrient variables within the sites. Summarising the CCA results, the water variable explains most variance of the species dataset. Together with nutrient availability of N and K, a large proportion of the maximum explainable variance is covered.

**Table 4:** Canonical correspondence analysis (CCA): Correlations of selected parameters sampled at different grazing intensities in 2004 of Inner Mongolia grassland ecosystem, China. Further unit explanations see Table 1.

	<b>N<sub>tot</sub></b>	<b>S<sub>tot</sub></b>	<b>AGB</b>	<b>Bulk-Dens.</b>	<b>Water</b>	<b>Resin-N</b>	<b>Resin-P</b>	<b>Resin-K</b>	<b>pH</b>
<b>OC</b>	0.988	0.603	0.356	-0.929	0.577	0.383	-0.269	-0.023	0.147
<b>N<sub>tot</sub></b>		0.605	0.341	-0.921	0.584	0.374	-0.260	-0.065	-0.199
<b>S<sub>tot</sub></b>			0.228	-0.587	0.368	0.242	-0.175	-0.003	-0.104
<b>AGB</b>				-0.430	0.632	0.175	-0.591	-0.122	0.147
<b>Bulk-Dens.</b>					-0.617	-0.403	0.325	0.033	0.147
<b>Water</b>						0.297	-0.462	-0.191	0.157
<b>Resin-N</b>							-0.192	0.311	-0.218
<b>Resin-P</b>								0.227	-0.034
<b>Resin-K</b>									0.082

The ordination biplot in Fig. 1 illustrates the relation between variables, shown as radiate vectors in ordination space, and 450 sample positions representing our different sites. The respective variable influence at a sample point is high when the distance to the ordination centre is large. Most obviously distinguished are sample points of sites HG and UG79, representing the extremes of grazing intensities. The moderate grazing intensities at sites MG, and WG as well as UG99 form a third

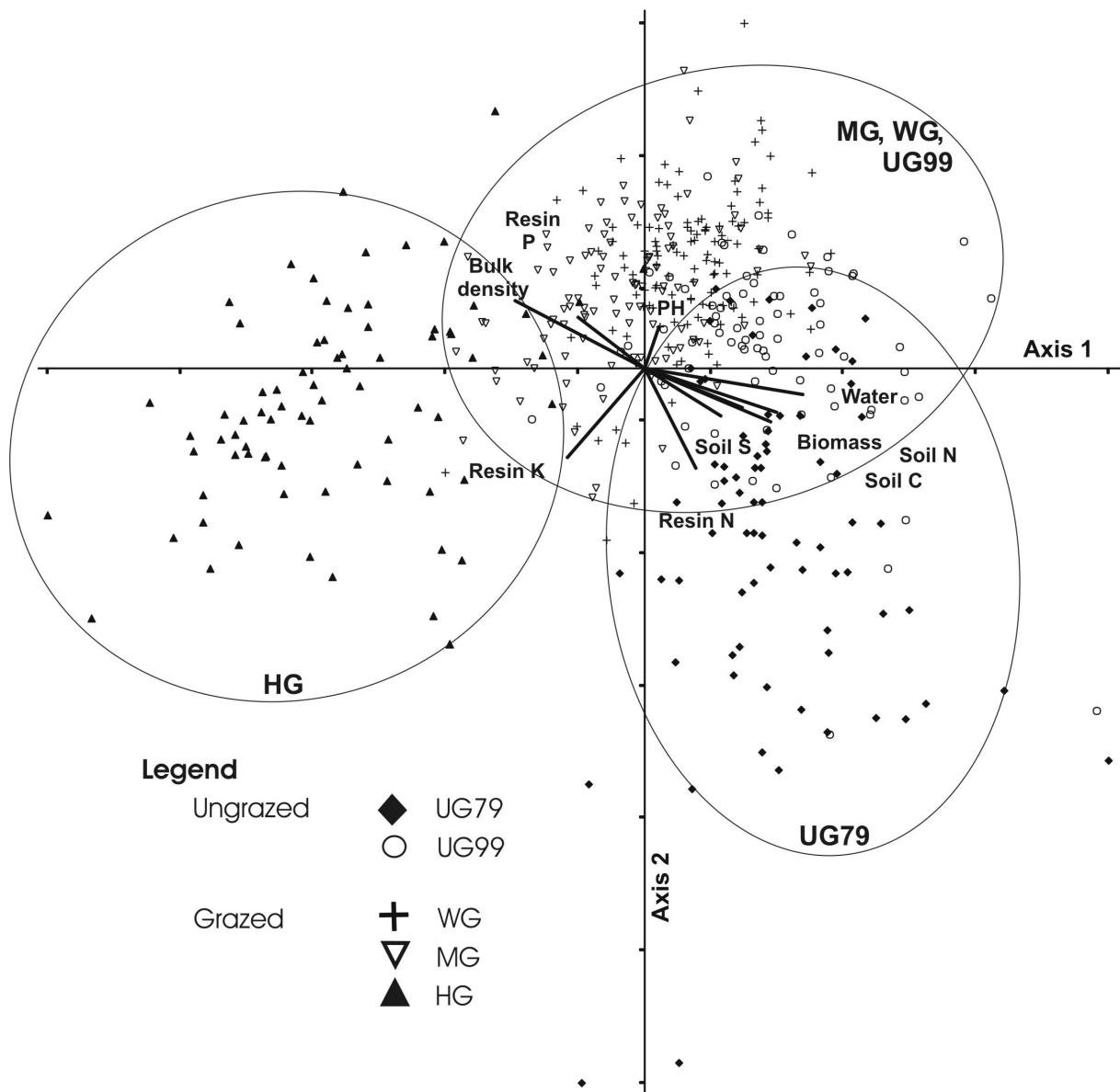
widely spread cluster of sample units, where UG99 has a considerable overlapping to UG79. From the results of CCA and ANOVA analysis, the environmental parameter combination defining the habitat of vegetation at different grazing sites is summarized in Table 5.

**Table 5:** Parameters measured at five sites as of different grazing intensities from long- and short-term enclosure (UG79, UG99) to winter- moderate- and heavy grazing of Inner Mongolia grassland ecosystem in 2004).

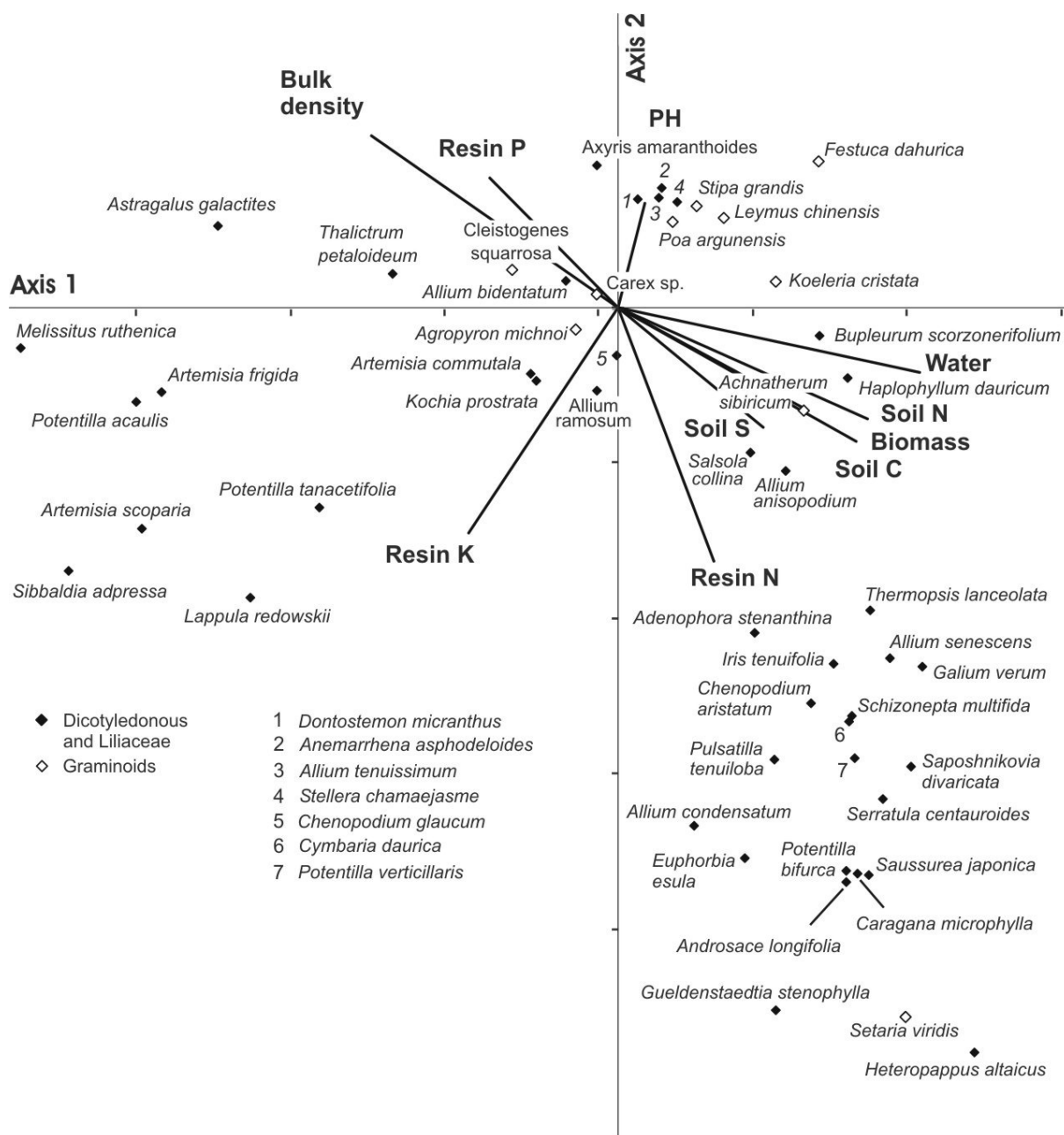
<b>Name of site</b>	<b>Site as characterized by environmental parameter</b>
1979 (UG79)	high N- availability, high biomass, high topsoil OC- and N-stocks, very low bulk density and medium to high topsoil water content.
1999 (UG99)	average N-, low P- and K availability, very high biomass with high topsoil OC- and N-stocks, low bulk density and high topsoil water content.
Winter grazing (WG)	low N- and K-, high P availability, average biomass, topsoil OC- and N-stocks, bulk density and topsoil water content.
Moderate grazing (MG)	average N-, slightly raised K- and high P availability, average to low biomass, low topsoil OC- and N-stocks, high bulk density and low topsoil water content.
Heavy grazing (HG)	low N-, high K- and P availability, very low biomass and topsoil OC- and N-stocks, high bulk density and very low topsoil water content.

#### *Grazing intensities and plant species response*

In Fig. 2, totally 53 plant species are distinguished by ordination according to environmental variables. Species with similar requirements to their environment are placed close to each other, which enables a classification into groups. The chosen ordination offers direct comparison between species position in Fig. 2 and sample point positions in Fig. 1. According to the ordination results and additionally by examination of species frequencies and cover proportions illustrated for selected species in Fig. 3, the plant community is classified into four units of species response groups: (1) light to moderate grazing decreaseers and/or competition strategists, (2) light to moderate grazing tolerant species, (3) generalists and (4) heavy grazing increaseers. The attached number (1-4) to the species name in Fig. 4 indicates the classification into species response group.



**Fig. 1:** Canonical correspondence analysis (CCA): Biplot of 450 sample points in relation to environmental variables at five grazing intensities (heavy grazing [HG], continuous grazing [CG], winter grazing [WG], grazing enclosure since 1999 [UG99], and grazing enclosure since 1979 [UG79]) of Inner Mongolia grassland ecosystem in July/August 2004.



**Fig. 2:** Canonical correspondence analysis (CCA): Biplot of plant species recorded at five grazing intensities in July/August 2004 of Inner Mongolia grassland ecosystem in relation to environmental variables.



*Group 1, light to moderate grazing decreaseers and/or strong competitors*

This group comprises species with decreasing relative frequency and cover proportions at already light to moderate grazing intensities. If grazing pressure is reduced or stopped, competition between species for resources (primary water, nutrients and light) is enhanced. Grazing intolerant species and/or strong competitors are consequentially mainly found in the long-term enclosure (UG79) and in the short-term enclosure (UG99) respectively. One large group of around 30 mostly herbaceous species was identified by CCA under environmental conditions primary found at UG79 (Fig. 1, 2). With together more than 11% cover *Potentilla bifurca*, *Serratula centauroides*, *Heteropappus altaicus*, *Pulsatilla tenuiloibia*, *Cymbaria daurica* and *Galium verum* contribute significant (beside the grasses) to the total cover in decreasing order. Species with minor cover but high relative frequency are *Allium senescens*, *A. anisopodium*, *Chenopodium aristatum*, *Potentilla verticillaris* and *Schizonepeta multifida*. A characteristic large bunch grass of this plant community is *Achnatherum sibiricum* with cover of almost 12% in UG79 and 7.5% in UG99. Similar to *A. sibiricum*, also *Festuca daturica* loses cover proportions continuously with increasing grazing intensity and was not found at HG (Fig. 3).

*Group 2, light to moderate grazing tolerant, heavy grazing decreaseers*

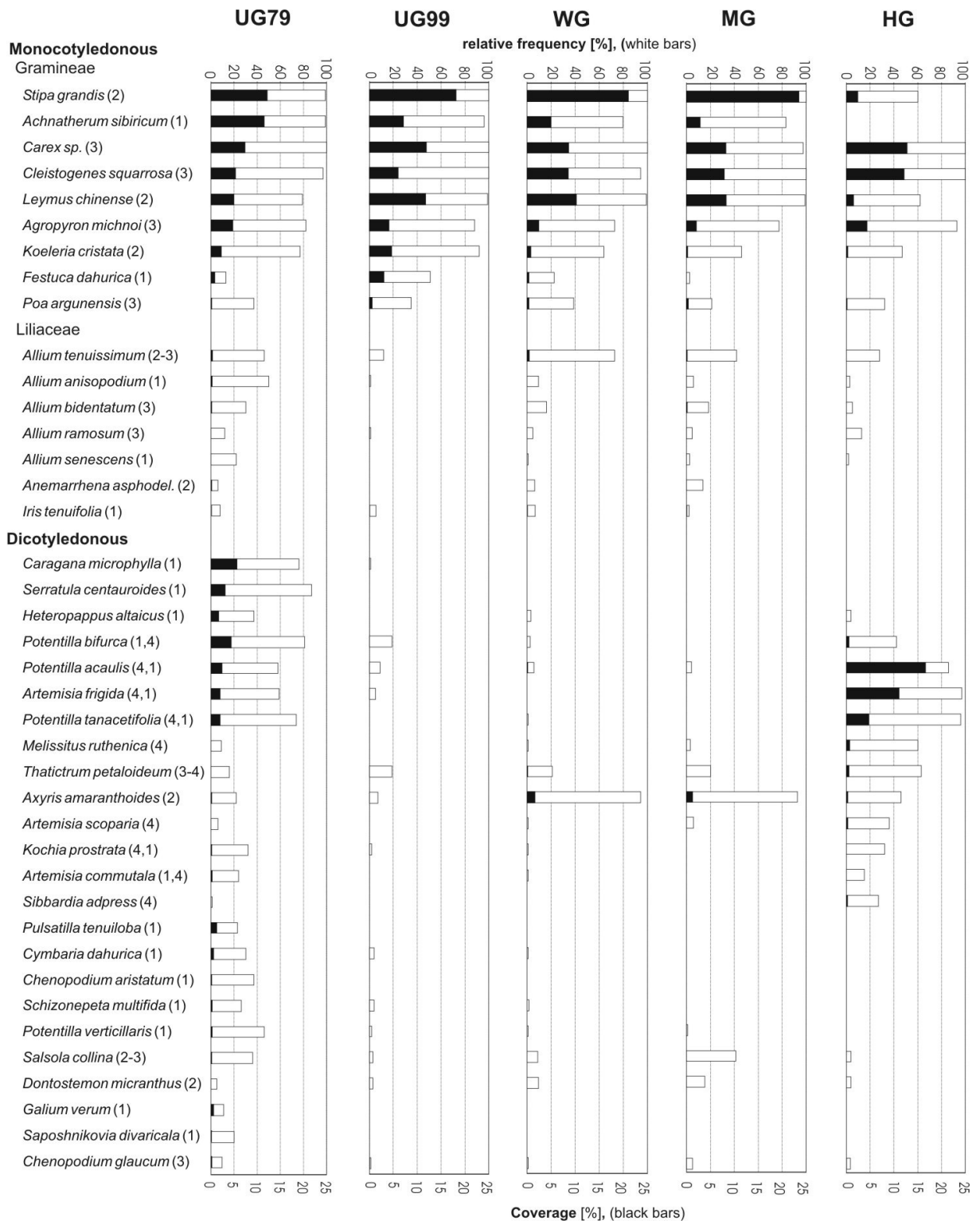
In contrast to the first classified group, species mentioned here are to a certain extend adapted to grazing, they actually benefit from reduced competition. The plant community is dominated by grasses like *Stipa grandis*, *Leymus chinensis* and *Koeleria cristata*, further *Allium tenuissimum* and *Anemarrhena asphodeloides* as non gramineous species. Dicotyledonous have generally problems to establish with exception of *Dontostemon micranthus*, *Salsola collina* and *Axyris amaranthoides*. *Stipa grandis* shows highest cover and very high abundance in all treatments except site HG, which underlines the dominant position of the large bunch grass in this ecosystem. The cover increases continuously from UG79, UG99, WG to MG up to more than 20% (Fig. 3). *Stipa* is not selected in favour by sheep and has thus advantages under moderate grazing due to reduced competition. As preferentially selected species become rare, *Stipa* is increasingly consumed, which explains the weak appearance at site HG with 2.4% cover. *Leymus chinensis* is preferentially selected by sheep. The cover decreases from

UG99, WG and MG continuously from 11.8% to 8.3%. At site HG this rhizome grass is strongly reduced to only 1.5% cover. The low cover of 5% at site UG79 may indicate reduced competition ability. *Koeleria cristata* is an inherent part of the plant community under all grazing intensities underlined by a relative frequency not below 40%. But under high grazing pressure at site CG and HG, the cover values of this small bunch grass are strongly reduced.

### *Group 3, generalists*

Generalists are present with either considerable cover proportions or high relative frequencies within all investigated sites and show hardly a preference to any environmental variable. In the CCA biplot, these species are usually placed in the ordination centre (Fig. 2). Here we find the sedge *Carex* sp. and the rhizome grass *Agropyron michnoi* as well with some restrictions the bunch grass *Poa argrunensis*. *Carex* sp. shows highest frequency values of all species. This rhizome species appears at all sites with remarkable high cover between 7% and 12% (Fig. 3). Within the plant communities, the two *Carex* species of *C. korshinskyi* and *C. duriuscula* are present. They are difficult to distinguish and can be aggregated from their ecological function. *Agropyron michnoi* shows neither sensitivity to grazing nor acts as a weak competitor. This rhizome grass has a cover between 2% and 5% at all sites. *Poa argrunensis* has generally low cover values but a relative frequency higher than 20% at all sites.

Including *Cleistogenes squarrosa* (C<sub>4</sub> grass) and *Allium bidentatum*, *A. ramosum* and *Thatictrum petaloideum* to this group, we recognise a tendency to higher bulk density and reduced water content (Fig. 2). This species certainly benefit from increasing grazing intensity, which would have also justified a classification into the grazing increaser group, especially for *Cleistogenes* with increasing cover from 5.3% at site UG79 to 12% at site HG (Fig. 3). But, although the composition of environmental factors is variable among the sites, the mentioned species and in particular *Carex*, *Agropyron* and *Cleistogenes* underline their outstanding position in this ecosystem by constantly high proportion of cover and relative frequency > 70% for *Agropyron* and even > 90% for *Carex* and *Cleistogenes* within the all investigated plant communities, which finally classifies them as generalists.



**Fig. 3:** Cover and relative frequency of 40 selected plant species recorded at five grazing intensities of Inner Mongolia grassland ecosystem in July/August 2004. Species response groups are attached to species name: (1) light to moderate grazing decrease and/or competition strategists, (2) light to moderate grazing tolerant species, (3) generalists and (4) heavy grazing increasers. The following 36 species are not listed in the figure, but present at site UG79<sup>(a)</sup>, UG99

<sup>(b)</sup>, WG <sup>(c)</sup>, MG <sup>(d)</sup> and HG <sup>(e)</sup>: *Adenophora stenanthina* <sup>(ab)</sup>, *Allium condensatum* <sup>(acd)</sup>, *Androsace longifolia* <sup>(ac)</sup>, *Artemisia sieversiana* <sup>(ac)</sup>, *Astragalus galactites* <sup>(acde)</sup>, *Buplerum scorzonerifolium* <sup>(abc)</sup>, *Cannabis sativa* <sup>(a)</sup>, *Convolvulus arvensis* <sup>(a)</sup>, *Echinops latifolius* <sup>(ae)</sup>, *Euphorbia esula* <sup>(abe)</sup>, *Filifolium sibiricum* <sup>(a)</sup>, *Gentiana dahurica* <sup>(acd)</sup>, *Gueldenstaedtia stenophylla* <sup>(abcd)</sup>, *Haplophyllum dauricum* <sup>(ab)</sup>, *Lappula redowskii* <sup>(acde)</sup>, *Leontopodium leontopodioides* <sup>(a)</sup>, *Lilium pumilum* <sup>(acd)</sup>, *Limonium bicolor* <sup>(ae)</sup>, *Melandrium apricum* <sup>(ac)</sup>, *Melilotus suaveolens* <sup>(ab)</sup>, *Orostachys malacophyllus* <sup>(ad)</sup>, *Oxytropis myriophylla* <sup>(ace)</sup>, *Phlomis dentosa* <sup>(ad)</sup>, *Plantago asiatica* <sup>(ae)</sup>, *Polygonum divaricatum* <sup>(a)</sup>, *Potentilla betonicaefolia* <sup>(d)</sup>, *Ptilotrichum canescens* <sup>(abc)</sup>, *Saussurea japonica* <sup>(a)</sup>, *Scutellaria scordiifolia* <sup>(ae)</sup>, *Setaria viridis* <sup>(a)</sup>, *Silene jenissensis* <sup>(acd)</sup>, *Spiraea aquilegifolia* <sup>(a)</sup>, *Stellera chamaejasme* <sup>(abc)</sup>, *Tephrosia kirilowii* <sup>(ae)</sup>, *Thermopsis lanceolata* <sup>(a)</sup> and *Thymus serpyllum* <sup>(e)</sup>.

#### Group 4, heavy grazing increasers

Into this group we classified species clearly having their main occurrence under high grazing pressure. Here, we notice again a pronounced occurrence of dicotyledonous species like *Artemisia frigida* and *A. scoparia* as well as *Potentilla acaulis*, *P. tanacetifolia*, *Sibbaldia adpressa*, *Kochia prostrata* and *Melissitus ruthenica*. As shown by CCA, the species must be adapted to drought and limited nitrogen availability in addition to the mechanical grazing and trampling impact (Fig. 2). Underlined by our results, *P. acaulis* and *A. frigida* are very important increasers under heavy grazing. *P. acaulis* grows dense and patchy and accumulates large parts of the aboveground biomass within 1-2 cm above soil surface to avoid consumption by herbivores. At site HG, *P. acaulis* was measured with highest cover (16%) but with less abundance compared to some other grasses. Despite *A. frigida* is preferentially consumed by sheep, we measured a cover of 11.1% at site HG (Fig. 3).

Some of the dicotyledonous species classified as heavy grazing increasers surprised us with noticeable results. We found *Potentilla acaulis* and *P. tanacetifolia* and *P. bifurca* as well as *Artemisia* species (*A. frigida*, *A. commutala* and *A. scoparia*) and *Kochia prostrata* with considerable cover, respectively relative frequencies at site UG79, while those species were of minor relevance at intermediate grazing areas. At site UG79 *A. frigida* exhibit a cover of 2%, while *P. acaulis* and *P. tanacetifolia* together achieved more than 4% of cover (Fig. 3).

## Discussion

### *The effect of grazing on environmental parameters*

The results confirmed that grazing in Inner Mongolia grasslands affects nutrient availability. This agrees in principle with the current state of knowledge, regarding the nutrient availability as one focal point to explain grazing impact on ecosystems (e.g. Olf & Ritchie 1998, Proulx & Mazumder 1998). However, we underestimated the influence of water availability affected by grazing on the local scale, the variable explaining the highest proportion of variance in the species data. The importance of water is mainly known in relation to climate gradients (e.g. Milchunas et al. 1988) or climate variability (Biondini et al. 1998).

In general, large stocks of organic matter in Chernozem soils of northern Chinese steppe ecosystem should provide sufficient or at least well balanced nutrient availability by mineralisation process. However, this process can be limited by slow turnover rates due to dry and/or cold climate conditions which define most of the year in this ecosystem. This situation of a potential N deficiency can be ameliorated by grazing effects as a result of enhanced nutrient cycling by dung and urine redistribution. But, as reported for some other grazed ecosystems (Holland & Detling 1990; McNaughton, et al. 1997; Frank et al. 2000), we cannot confirm an increased N availability under grazing. Instead, we found significantly higher N availability in the long-term enclosure, which opposed results of increased N mineralisation with increasing grazing intensity measured by Barger et al. (2004). We further measured increased P and K availability under increased grazing intensity, which did not confirm the results of Xie & Wittig (2004), but agrees with those of Fernandez-Gimenez & Allen-Diaz (2001). We argue, that total aboveground biomass and specially standing dead and litter increases characteristically with decreasing grazing intensity (Milchunas & Lauenroth 1993). As a result, because more material remains at the area, OC- and total N concentration of topsoil increases (Su et al. 2005), what we also measured for our enclosures (Steffens unpublished data).

Concurrently, as a long-term effect, grazing is associated with nutrient loss through biomass export. Because less organic matter is returned through litter, soil total C and N declines (Holland & Detling 1990; Holland et al. 1992; Dormaar & Willm 1998). This process is enhanced in the Inner Mongolia grassland, as sheep are generally kept in paddocks overnight, reducing the redistribution of dung to

grazing areas. Therefore, *in-situ* net soil N mineralisation decreases (Biondini et al. 1998) which, in turn will effect plant available N (van Wijnen et al. 1999). According, Kurz et al. (2006) reported increasing nutrient losses on grazed areas by surface runoff as a result of reduced macro soil pores, increasing bulk density and water penetration resistance. The increased topsoil bulk density results from trampling and is used as an indicator of intensified grazing (Tate et al. 2004). Grazing can reduce topsoil water content (Naeth et al. 1991), but on the other hand increased transpiration losses can lead to decreased soil water content at ungrazed areas, as it might be the case at site UG79. High amounts of standing dead and litter, characteristically for short time enclosures (UG99), however reduces dry-off processes by changing the microclimate (Facelli & Pickett 1991). These two effects together (surface runoff and dry-off dynamics) explain on the other hand the dry environment situation at site HG, which in turn has large influence on plant community, promoting drought tolerant species.

#### *The effect of grazing on plant species composition*

Beside physical damage, low water and N availability concurrently affects the vegetation at site HG, resulting in a strong reduction of characteristic grassland species, mainly perennial grasses. Species being sensitive to heavy grazing, drought or N limitation were compensated by the adapted group of heavy grazing increasers, which are mainly dicotyledonous, but noteworthy totally consist of native grassland species (pers. comm., IMGERS 2004). For Chinese grasslands a general increase of dicotyledonous species under grazing was also reported by Li (1989); Xie & Wittig (2003), Barger et al. (2004), Chen et al. (2005).

At medium grazing intensities, species benefit from reduced competition for resources as long as they can compensate the loss of assimilating tissue. Here perennial grasses, like *Stipa grandis* and *Leymus chinensis*, are obviously promoted, the cover proportion and abundance of dicotyledonous species is however extremely reduced.

The long term enclosure (UG79) can certainly be considered as an extreme, since grazing ever occurred in Chinese natural grasslands. Nevertheless, we can transfer some valuable results to understand how grazing alters ecosystem functions. The environmental conditions under long-term grazing enclosure generate a relative high N availability, but also increase competition. An increase

of tall growing, competitive gramineous species in enclosures was reported by Rodriguez et al. (2003), what we confirm with regard to *Achnatherum sibiricum*. But also dicotyledonous plants, including creeping and low growth forms, increased significantly within the long-term enclosure. Thus, light shortage at ground level could only partly be a limiting factor. We assume, water availability becomes limiting when comparably high photosynthetic active biomass causes considerable transpiration losses, which in turn again benefits drought tolerant species even being characteristic for heavy grazed areas like *Potentilla acaulis* and *Artemisia frigida*. The results from Zhao, Y. (unpubl.) support this hypothesis, showing the highest soil water contents within site UG99 and not at site UG79, which is generally characterized by higher spatial heterogeneity compared to the other sites (Steffens, M., Schneider, K. unpubl.). A livestock exclusion can increase the spatial heterogeneity of vegetation (Adler & Lauenroth 2000). This situation might also promote the occurrence of *Caragana microphylla* (and also *Spiraea aquilegifolia*) thorn shrubs, recorded at UG79 with remarkable 5.7% cover (Fig. 3). At this point we must raise questions about grassland succession of enclosures in this ecosystem. The coexistence of shrubs (deep rooting system) and grasses (shallow rooting system) in semi-arid ecosystems indicates a vertical allocation of water resources, usually as a result of texture effects on evaporation and percolation (Walter 1979; Dodd et al. 1998). Originally *Caragana* species were mainly found on deep drained sandy soils of arid environments. Because these shrubs are also well known to increase under heavy grazing in distinct areas (Chen et al. 2005), we assume changed competition for water by decreased abundances of gramineous species under long-term enclosures or heavy grazing might be beneficial for *Caragana*. As one reason we expect the incomplete precipitation absorption in topsoil by less continuous developed grass species root system. In the ordination we found the shrub in strong dependence of N availability, which *Caragana* as a legume probably influences by itself (Fig. 2). If areas protected from grazing or under heavy grazing will generally shift to an undesired equilibrium plant community dominated by thorn shrubs is not finally proven, but some evidence suggest that grazing interacts with *Caragana* appearance. The example of *Caragana* underlines the importance of water availability as environmental factor altered by grazing in this ecosystem.

*Conclusions and applications*

Grazing affects vegetation of Inner Mongolia grasslands by altering water- and nutrient availability. Water as an environmental factor influenced by grazing might be generally underestimated on a local scale. As a result of changed environmental conditions, the plant community was classified into four grazing response groups.

Although our study area is considered to be representative for central Asia steppe ecosystems (Bai et al. 2004), we only carefully draw conclusions for land management strategies. Due to intensive sampling pattern, our study could not cover all established land-use variations of Inner Mongolia. As a general trend we observed an increasing number of farms to change over from sheep to stock their land with cattle, which opens new land-use options. We further emphasize to intensify research on hay-making and light-grazing areas, traditionally located in remote areas with distance to settlements and water resources. These relicts of natural grasslands might function as important resources for native plant species pools, from where degraded areas might be colonized again.

In degraded grasslands grazing is mainly based on the species groups classified as grazing increasers like *Artemisia frigida* and *Potentilla acaulis* (among others) and the generalists (e.g. *Agropyron michnoi*, *Cleistogenes squarrosa* and *Carex* sp.). An important question is how these species compensate the reduction of characteristic perennial grasses like *Leymus chinensis* with respect to fodder quality and sward productivity but also ecosystem functions.

The N deficiency caused by grazing could be compensated by applying small amounts of organic or mineral fertilizer. But with regard to the high natural variability of water availability in this semi-arid ecosystem, precipitation regimes will always be dominant. Therefore, long-term studies are needed to investigate climate variability and resultant effects on ecosystem matter fluxes and plant community.

In contrary to soil nutrient status, water availability altered by grazing cannot be considered as long-term effect. It is dependent on factors like aboveground biomass which is changeable in short-term, even on degraded areas. Consequently, adapted land management should be able to control soil water content within categories predetermined by precipitation and thus reduce drought



enhanced by grazing as superior factor affecting the plant community, which in turn could be the first step of grassland rehabilitation.

The results, about the effects of grazing on ecosystem parameters and plant community are of value for further analysis of semi-arid grassland ecosystems under changing land-use, for rehabilitation and conservation measures as well as for studies of climate variability or change.

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**General discussion**

Grazing effects on ecosystem functions are discussed contrarily and an agreement exists by accepting the multiple ways and levels how grazing alters ecosystem processes with different consequences. However, studies comparing different semi-arid ecosystems regarding matter fluxes and diversity, have illustrated the general importance of natural precipitation gradients as well as the basic ecosystem fertility as key variables how different grasslands respond to grazing (Olf and Ritchie 1998, Milchunas et al. 1988).

During our studies, we experienced precipitation dynamics on a temporal scale by the highly variable rainfall pattern of semi-arid Inner Mongolia grasslands, resulting in three very different growing seasons with an extreme drought in the year 2005. Further, within a growing season, alternating dry and wet cycles influenced the grassland on small temporal scales. Finally, the majority of the results were directly or indirectly affected by the seasonal and annual rainfall variability. Thus, we have to ask for the universal validity and transferability of our observations and conclusions. Did we sample extreme situations, or are these extremes more or less normal fluctuations of an ecosystem, which is finally characterized by a high variability on multiple scales?

The discussion comes to the point, where we question, how this ecosystem reacts generally to disturbances. How can we assess the balance between ecosystem resources and land-use? These aspects are currently intensively debated in the context of equilibrium and disequilibrium theories (Illius 2005, Gillison and Hoffmann 2007, Hambler et al. 2007). Are grasslands, as driven by density dependent effects in competition to key resources, changing to alternative stable states or regarding larger time scales even in a long-term stable state (equilibrium)? Or are environmental fluctuations continuously changing the ecosystem functions und matter fluxes, so that equilibrium is never reached (disequilibrium). These basic ecological considerations are of importance to reflect the project conclusions and the message finally to be communicated to the society.

During the three years of field studies in the Inner Mongolia steppe, we experienced a broad variety of grassland areas as affected by different types of natural or anthropogenic disturbances. On the one hand the influence of different constant natural environmental factors as well as long-term land-use effects,

clearly induced the development of different grassland varieties as illustrated for example by plant species composition, productivity and various soil properties. On the other hand these grasslands are characterized by highly dynamic continuously processes, interactively driven by land-use and climate variability in particular. This continuously progressing development of the grassland ecosystems takes place on multiple temporal and spatial scales und thus forms a continuum of equilibrium and disequilibrium states, which in our view suggest the integration of both theories as postulated by Briske et al. (2003).

In the context of convergent and divergent ecosystem processes, the understanding carbon and nitrogen pools, fluxes and budgets is still limited. Recent studies emphasise the importance of natural climate variability, especially the precipitation patterns and consequences on semi-arid ecosystem functions (Knapp et al. 2002, Austin et al. 2004, Heisler and Weltzin 2006, Potts et al. 2006, Anderson et al. 2007). In the frame of climate change we will experience even more extremes of climate parameters (Easterling et al. 2000) and the task to predict matter fluxes for grassland ecosystems will be a challenge. However, these findings are in line with the general conclusions of our Inner Mongolia field campaign concerning the basic variability of the ecosystem processes. We can answer our prior questions by the conclusion, that our measurements represent (at least) the natural fluctuations for semiarid grassland ecosystems. The analysed interaction of these variability with land-use management practices is therefore of considerable value for the scientific discussion on semi-arid grassland ecosystem matter fluxes.

However, many aspects of the recent land-use changes in northern China are still unclear. The relation between grazing and degradation is a controversial discussion at all. (Gillison and Hoffmann 2007, Hambler et al. 2007). One central question is, when does overgrazing and degradation starts, respectively? How can we distinguish variability from potentially long-term degradation? And, how can we first define and later on realize a situation of sustainable land use management for the Inner Mongolia grassland? Anyway, with respect to central Asian steppe ecosystems a general agreement exists about the negative consequences of overgrazing. According to the basic ecosystem dynamics, the challenge will be the introduction of flexibility into the existing management systems. Finally the societies must decide subjectively and explicitly which changes in the environment

are acceptable (Hamblen et al. 2007). Research results from the central Asian steppes are highly requested to continue and enrich the scientific discussion on the grazing-ecosystem interaction of the world's natural grasslands.

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## Summary and Conclusions

In 2004 the Sino-German cooperation project MAGIM (**M**atter fluxes of **G**rasslands in **I**nnner **M**ongolia as influenced by grazing) was created in the context of the dramatic land use changes in the Northern Chinese semi-arid natural grasslands. Decades of overgrazing were followed by land degradation including considerable decrease in grassland productivity and fodder quality, loss of soil organic matter and nutrients as well as severe land erosion processes.

Although central Asian grasslands exceed the size of other natural grasslands by far, these ecosystems were much less considered and therefore underrepresented in scientific research. From this point of view, the effects of grazing on the ecosystems carbon and N cycling are of fundamental importance for the regional and global matter flux analysis and discussion. The present work addresses in particular the nitrogen (N) dynamics, since this macronutrient currently dominates the discussion of grazing effects on nutrient dynamics.

### *Effects of grazing and variable rainfall on root and shoot decomposition*

The decomposition of plants was investigated as a key process of carbon and N cycling in ecosystems. The decay process was analysed separately for shoot and root with the “litter bag” method at sites of different grazing intensity. To distinguish the effects of local environmental factors and litter quality, as altered by grazing, common and site-specific material was incubated for three and two growing seasons. Growing seasons of average and very low precipitation allowed us to study the effect of inter annual rainfall variability. Local environmental factors as altered by different grazing intensities had no effect on decay rates of shoot and root dry mass. Also differences in litter quality between the grazing sites were not reflected by root decomposition dynamics. In contrast to the expected grazing effects, we found a pronounced impact of rainfall variability on the decomposition dynamics. In the year of average precipitation (2004, 325 mm) root decomposition was faster than shoot decay, while under dry condition (2005, 166 mm) the opposite was observed. Fitting a simple exponential decay function ( $x_0 = x_1 e^{-kT}$ ), we found soil moisture storage was the best parameter explaining the dynamics of root decomposition. Contrarily, shoot decomposition was unaffected by annual rainfall variability. Net N immobilization was not observed during the decay process of shoot and root at all sites.



Consequently, a separate analysis of shoot and root decay dynamics is required in order to describe carbon and N cycling in this semi-arid grassland. When root decomposition is strongly reduced in dry periods, shoot decomposition becomes relatively more important for plant nutrient supply. Further, a stronger N limitation for plant productivity at grazed sites results from general low aboveground biomass and decreased shoot N release, since sheep are kept in paddocks over night and thus considerable amounts of dung and urine are not returned to the grazing site. Under average rainfall conditions differences between grazing sites with respect to nutrient supply through decomposition are less pronounced because of more active belowground processes. The grazing impact on C and N fluxes through decomposition of plant material likely exhibits a strong interaction with seasonal rainfall pattern.

*N availability as affected by grazing and variable rainfall*

We ask for the impact of grazing and rainfall variability on plant available inorganic N. Ecosystem net primary productivity (NPP) and plant N uptake was linked to inorganic N availability. The availability of inorganic N-form ( $\text{NO}_3^-$ ,  $\text{NH}_4^+$ ) was analysed in the context of seasonal and inter-annual rainfall variability.

We combined time-integrated analysis of N fluxes measured with ion-exchange capsules and conventional soil analysis of mineral N to analyse inorganic N availability for plants. Three grazing sites, a long-term grazing enclosure (UG79), a moderate winter grazed site (WG) and a long-term heavily grazed site (HG) were included to cover different land-use variations.

Grazing had comparable minor impact on the N availability compared to seasonal and annual rainfall variability. In one of three analysed growing seasons (2004) the long-term grazing enclosure showed significant higher N availability compared to site WG and site HG. Annual differences of N availability were much more pronounced as a result of rainfall variability. Precipitation pattern controlled soil water content and as a result the dynamics of N availability.

In general, grassland NPP correlated positive to inorganic N availability as measured by ion exchange resin and soil extractions. Since productivity in semi-arid grasslands occurs to mayor part below ground, the root production needs to be taken into account if inorganic N availability is correlated to NPP. A temporal asynchrony was not observed between N availability and ecosystem productivity

pattern as reported from several other semi-arid grassland ecosystems. Ion exchange resins could indicate the plant N uptake dynamic for the different growing seasons, as well as partly for the different grazing sites. However, during growing seasons of comparable low N availability as a result of low rainfall, grazing site-specific N dynamics were not reflected.

Rainfall controlled the  $\text{NO}_3^-$  -  $\text{NH}_4^+$  ratio in the soil solution by altering soil water content (SWC), respectively the soil water potential in dependence of soil texture. Below 8 vol.% SWC ammonium dominated the available inorganic N-form, until 13 vol.% SWC the ratio is variable while under more moist conditions >13 vol.% the nitrate proportion of inorganic N was at least higher than 40%.

Climate seasonality and variability finally triggers N-availability processes in semi-arid ecosystems by altering the soil water potential. Positive correlations between N availability and PPN respectively plant N-uptake indicates ecosystem N limitation if water is sufficiently supplied. We conclude that in times of relatively dry conditions, plants that preferentially take up ammonium have a competitive advantage in the semi-arid grasslands of Inner Mongolia. These findings suggest intensifying the research on interactions of precipitation pattern and semi-arid ecosystems functions, which is also of current interest in the frame of the prognoses towards climate change. The knowledge about the role of herbivores in this frame is still fragmentary, which demands for further research. Since climate obviously dominates basic processes of available inorganic N, grazing effects are most likely limited as long intensity is controlled. Decreased soil moisture with increasing grazing intensity however, might possible alter the dynamic of plant available N (and the relation of N-forms) and by this important processes of central Asian semi-arid grassland carbon and N cycling, with consequences for regional and global balances of matter fluxes.

#### *N cycling and balance*

The understanding of the complex pathways of N semi-arid grassland ecosystems under grazing is fragmentary. This demands for a balance of the different N pools and fluxes as well as N gains and losses as influenced by grazing. The identification of key processes within the N dynamics of grazed Inner Mongolia grasslands might indicate possible options for the application of adapted and sustainable land-use management practices.

We defined a functional unit of a typical grassland farm as our system boundary for N balancing. Ungrazed sites represent N sinks ( $0.7$  and  $1.7 \text{ g N m}^{-2} \text{ yr}^{-1}$ ), while heavy grazed areas are N sources with losses up to  $1.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Areas of moderate grazing only are N sources if combined with haymaking, since this land-use type causes highest ( $\pm 1.0 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) N losses per area.

Dry deposition was identified as the main pathway contributing to N accumulation with more than  $1 \text{ g N m}^{-2} \text{ yr}^{-1}$  at ungrazed areas. Concurrently wind erosion counts for high N losses up to  $0.6 \text{ g N m}^{-2} \text{ yr}^{-1}$  at heavy grazed sites. The wet deposition counts for N input with up to  $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Heavy grazing causes N losses ( $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) due to the common management practice to keep sheep in paddock overnight (dung not returned). N losses due to meat and wool harvest are comparable small ( $0.2 \text{ g N m}^{-2} \text{ yr}^{-1}$ ).

High potential ecosystem fertility (N stored in soil organic matter  $800 - 1000 \text{ g N m}^{-2}$ ) indicated considerable regeneration potential even of long-term heavy grazed areas. The complex interactions between climate variability and ecosystem productivity however demand for flexible grazing systems. Grazing would be compensated in the N-balance as soon as stocking rate is reduced to a level where the remaining aboveground biomass prevents wind erosion and the involved N losses. A rotational system between hay and grazing sites, as well as growing season rainfall amounts as indicator for next season stocking rates could be a management option.

#### *How grazing affects plant communities and environmental parameters*

We ask for environmental parameters as altered by grazing and their effects on plant community structure.

Five grazing intensities were analysed at totally 450 sample positions. Canonical correspondence analysis (CCA) was applied to link plant species composition, abundance and coverage to variables of nutrient availability (N, P, K), topsoil water content, aboveground biomass (AGB), organic carbon (OC), total soil N and S concentration, pH and bulk density.

The long-term enclosure showed highest N availability, highest OC and total N concentration, lowest bulk density and high AGB. The heavy grazed site was characterised by low topsoil water content and reduced N availability as well as low OC and total N concentration, high bulk density and very low AGB. At both

sites dicotyledonous species showed high abundances, while Graminoids dominated intermediate grazing sites. Here K availability was low, while other soil parameters ranked medium. Differences in species coverage, abundance and composition among grassland sites were best explained by topsoil water content in combination with N and K availability. Plant species were classified as responding to specific environmental conditions and grazing intensities.

The grazing effect on water availability was surprisingly high and might be generally underestimated. By linking environmental variables to plant species composition as affected by grazing the results provide a basis for the development of land-use strategies within conflicting interests of live-stock farming, grassland regeneration and conservation.

*Slope aspect has effects on productivity and species composition*

In order to investigate the productivity of hilly grassland of the Xilin River Basin as affected by slope aspect (North versus South), aboveground green biomass (AGB) and species composition were studied at four hill sites in 2004 and 2005. Additionally, a detailed investigation combining vegetation parameters, physical and chemical soil characteristics was carried out at different slope positions (base, middle, and top) of north and south slope direction on one of the hills in 2004 and 2005.

Northern slopes exhibited a higher productivity and species diversity compared to southern slopes. Plant available soil K, P and N were not limiting the productivity at south- relative to north-oriented slopes, whereas soil available water was the primarily limiting factor for plant productivity, which was supported by results of leaf carbon isotope discrimination of main species. A negative correlation between productivity and relative abundance of C<sub>4</sub> plants indicates that the relative biomass of C<sub>4</sub> plants can be used as a water-stress indicator.

## **Zusammenfassung und Schlussfolgerungen**

Im Jahr 2004 wurde das deutsch-chinesische Forschungsprojekt MAGIM (**M**atter fluxes of **G**rasslands in **I**nnner **M**ongolia as influenced by grazing – (Einflüsse von Beweidung auf Stoffströme in Grassländern der Inneren Mongolei) vor dem Hintergrund dramatischer Änderung der Landnutzung in den Nordchinesischen Steppengebieten gegründet. Nach Jahrzehnten der Überweidung sind weite Teile der Inneren Mongolei geprägt von Landdegradation, die sich in verminderter Grassland Produktivität und Futterqualität, Verlust an organischer Bodensubstanz und Nährstoffen, als auch in schweren Erosionsprozessen zeigt.

Obwohl die Ausmaße der zentralasiatischen Steppen andere natürliche semi-aride Grasländer bei weitem übertreffen, wurden diese Ökosysteme vergleichsweise wenig beachtet und erscheinen somit unterrepräsentiert in der wissenschaftlichen Forschung. Von diesem Standpunkt ist die Erforschung von Beweidungseffekten auf Kohlenstoff- und Stickstoffkreisläufe dieser Grasländer von grundlegender Bedeutung für die Bilanzierung regionaler und globaler Stoffströme. Die vorliegende Arbeit widmet sich insbesondere der Stickstoffdynamik, da dieses Makronährelement gegenwärtig die wissenschaftliche Diskussion um die Auswirkungen von Beweidung auf Nährstoffkreisläufe dominiert.

### *Effekte von Beweidung und variablen Regenfällen auf den Wurzel- und Sprossabbau*

Die Zersetzung von Pflanzenmaterial gilt als Schlüsselprozess innerhalb des Kohlenstoff- und Stickstoffkreislaufs von Ökosystemen. Auf Flächen unterschiedlicher Beweidungsintensität wurde der Abbauprozess für Wurzel und Spross mit der „litter-bag“ Methode untersucht. Beweidung beeinflusst die Zersetzung des Pflanzenmaterials durch Veränderung lokaler Umweltfaktoren, als auch durch die geänderte Qualität des Materials aufgrund selektivem Konsum hochwertiger (stickstoffreicher) Pflanzenarten. Um diese Effekte zu unterscheiden wurde einheitliches und flächenspezifisches Pflanzenmaterial für zwei bzw. drei Vegetationsperioden ausgebracht. Jahre mit durchschnittlichen und sehr niedrigen Regenmengen erlaubten uns den Effekt von Niederschlagschwankungen zu untersuchen.

Durch Beweidung veränderte lokale Umweltfaktoren hatten keinen Einfluss auf die Abbaurate der Spross und Wurzeltrockenmassen. Auch Unterschiede in der Materialqualität zwischen den Beweidungsflächen hatten keine signifikanten

Auswirkungen auf die Abbaudynamik. Im Gegensatz zu den erwarteten Beweidungseffekten fanden wir einen ausgeprägten Einfluss der Regenvariabilität. Bei durchschnittlichen Regenmengen (2004, 325 mm) war der Wurzelabbau schneller als der Sprossabbau, hingegen wurde unter trockenen Bedingungen (2005, 166 mm) das Gegenteil beobachtet. Durch die Anpassung einer simplen Exponentialfunktion ( $x_0 = x_1 e^{-kT}$ ) an den Abbauverlauf fanden wir, dass der Bodenwassergehalt die Dynamik des Wurzelabbaus am Besten beschreibt. Im Gegensatz hierzu war der Sprossabbau von der jährlichen Regenvariabilität unabhängig. Eine netto Stickstoffimmobilisierung wurde innerhalb der Abbauprozesse von Wurzel und Spross auf allen untersuchten Flächen nicht festgestellt. Insgesamt erschließt sich die Notwendigkeit einer getrennten Analyse von Wurzel und Sprossabbau, um Kohlenstoff und Stickstoffkreisläufe in semi-ariden Grasländern zu beschreiben. Ist der Wurzelabbau in Trockenperioden stark reduziert, so steigt der relative Beitrag der Sprosszersetzung an der Mineralstoffrückführung. Insbesondere in Trockenperioden schließen wir auf eine stärkere Stickstofflimitierung der Pflanzenproduktivität auf beweideten Flächen, resultierend aus der generell geringen oberirdischen Biomasse und entsprechend reduzierter Stickstofffreisetzung, da durch die nächtliche Einpferchung der Schafe beträchtliche Mengen an Kot und Urin den Flächen nicht zurückgeführt werden. Unter normalen Regenbedingungen sorgen aktive Bodenabbauprozesse hinsichtlich der Nährstoffversorgung für deutlich weniger ausgeprägte Unterschiede zwischen den Beweidungsintensitäten. Der Einfluss von Beweidung auf die Kohlenstoff- und Stickstoffkreisläufe über die ober- und unterirdischen Abbauprozesse weist somit eine starke Interaktion zu den saisonalen Regenfällen auf.

### *Stickstoffverfügbarkeit unter Beweidung und variablen Regenfällen*

Wir fragen nach dem Einfluss der Beweidung und variablen Regenfällen auf den pflanzenverfügbaren anorganischen Stickstoff (N). Zusätzlich wurde die Netto-primärproduktion des Ökosystems (NPP) und die N-Aufnahme der Pflanzen mit der anorganischen N-Verfügbarkeit korreliert. Wir analysierten die Verfügbarkeit der anorganischen N-Formen ( $\text{NO}_3^-$ ,  $\text{NH}_4^+$ ) im Kontext der saisonalen und jährlichen Regenvariabilität.

Wir kombinierten die zeitintegrierte Analyse von N-Flüssen durch Ionenaustauscher Kapseln mit konventionellen Bodenextraktionen von mineralischem N.

Drei Beweidungsintensitäten (eine langfristig unbeweidete Fläche (UG79), eine moderat winterbeweidete Fläche (WG) und eine langfristig stark beweidete Fläche (HG) wurden in die Untersuchungen einbezogen, um verschiedenen Landnutzungsvarianten abzudecken.

Beiweidung hatte einen vergleichsweise geringen Einfluss auf die N-Verfügbarkeit im Vergleich zu den saisonalen und jährlichen Schwankungen der Regenfälle. In einer von drei Vegetationsperioden (2004) zeigte die langfristig unbeweidete Fläche (UG79) eine signifikant höhere N-Verfügbarkeit gegenüber den Flächen WG und HG. In Abhängigkeit zu den Regenfällen waren die jährlichen Unterschiede der N-Verfügbarkeit weit höher ausgeprägt. Aus Niederschlägen resultierende höhere Bodenwassergehalte führten zu steigender N-Verfügbarkeit. Grundsätzlich korrelierte die Grasland-NPP positiv zu der anorganischen N-Verfügbarkeit, wie sie durch die Ionenaustauscher und Bodenextraktionen bestimmt wurde. Der Anteil unterirdischer Produktivität in semi-ariden Grasländern, erfordert die Beachtung der Wurzelproduktion bei der Korrelation von anorganischer N-Verfügbarkeit zu der Ökosystem-NPP. Eine temporäre Asymmetrie zwischen der N-Verfügbarkeit und Ökosystem Produktivitätszyklen, wie von einigen semi-ariden Grassländern berichtet, wurde nicht beobachtet. Die Ionenaustauscher waren in der Lage, die jährliche Dynamik, als auch teilweise die Beweidungsunterschiede in der N-Aufnahme der Pflanzenbestände abzubilden. Jedoch waren in trockenen Vegetationsperioden mit insgesamt geringer N-Verfügbarkeit beweidungsspezifische Unterschiede der N-Verfügbarkeit nicht zu messen.

Niederschläge kontrollierten das Nitrat-Ammonium-Verhältnis ( $\text{NO}_3^-$ - $\text{NH}_4^+$ ) in der Bodenlösung durch Veränderungen im Bodenwassergehalt (Bodenwasserpotential) in Abhängigkeit der Bodentextur. Unter einem Bodenwassergehalt von 8 vol.% wurde die verfügbare anorganische N-Form von Ammonium dominiert, zwischen 8 und 13 vol.% war das Verhältnis variabel, während unter feuchten Bedingungen >13 vol.% der Nitratanteil mindestens bei 40% lag.

Die klimatische Saisonalität und Variabilität bestimmt über den Einfluss auf Bodenwasserpotentiale auch Grossteile der N-Verfügbarkeit in semi-ariden Graslandökosystemen. Positive Korrelationen zwischen N-Verfügbarkeit und Produktivität bzw. N-Aufnahme der Pflanzen deuten auf eine N-Limitierung des Ökosystems hin, sobald genügend Wasser verfügbar ist. Wir folgern, dass

Pflanzen, die eine Präferenz für die Ammonium-Aufnahme zeigen, in Zeiten relativer Trockenheit einen Konkurrenzvorteil in semi-ariden Graslandökosystemen der Inneren Mongolei haben. Die Ergebnisse suggerieren auch im Kontext der prognostizierten Klimaveränderungen eine Intensivierung der Forschung zur Interaktion zwischen Niederschlagsvariabilität und Ökosystemprozessen. Das Wissen über die Rolle der Herbivoren in diesem Zusammenhang ist nach wie vor lückenhaft und bedarf weiterer Nachforschungen. Da das Klima die Prozesse der N-Verfügbarkeit dominiert, erscheinen Auswirkungen der Beweidung eher gering solange die Intensität und damit die Landdegradation kontrolliert ist. Ein mit zunehmender Beweidung reduzierter Bodenwassergehalt, könnte jedoch die Dynamik des pflanzenverfügbaren Stickstoffs (sowie das Verhältnis der N-Formen) verändern und dadurch wichtige Prozesse des Kohlenstoff- und Stickstoffkreislaufes zentralasiatischer Steppenökosysteme beeinflussen, mit weitreichenden Konsequenzen für regionale und globale Stoffkreislaufbilanzen.

### *Stickstoffkreislauf und Bilanzierung*

Das Verständnis der komplexen Wege des Stickstoffs (N) in semi-ariden Graslandökosystemen unter Beweidung ist lückenhaft. Dies verlangt nach einer Bilanz der verschiedenen N-Poolgrößen und -Flüsse sowie der -Einträge und -Verluste unter Einfluss der Beweidung. Ziel ist es, aus der Identifizierung von Schlüsselprozessen innerhalb der N-Kreisläufe in beweideten Grasländern, Handlungsoptionen für eine angepasste und nachhaltige Bewirtschaftungsform abzuleiten.

Als Systemgrenze der N-Bilanzierung wurde die funktionale Einheit einer typischen innermongolischen Farm definiert. Während unbeweidete Flächen N-Senken darstellen ( $0.7$  and  $1.7 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) müssen stark beweidete Flächen als N-Quellen mit Verlusten bis zu  $1.5 \text{ g N m}^{-2} \text{ yr}^{-1}$  gelten. Flächen moderater Beweidung zählen nur zu den N-Quellen, wenn sie gleichzeitig als Heufläche genutzt werden, da diese Landnutzung sehr hohe N-Verluste bewirkt ( $\pm 1.0 \text{ g N m}^{-2} \text{ yr}^{-1}$ ).

Auf unbeweideten Flächen wurde die Trockendeposition als größter N-Eintrag identifiziert mit mehr als  $1 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Gleichzeitig sorgt Winderosion auf stark beweideten Flächen für Verluste in Höhe von bis zu  $0.6 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Naßdeposition zählt als N-Eintrag mit bis zu  $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ . Starke Beweidung



verursacht N-Verluste ( $0.5 \text{ g N m}^{-2} \text{ yr}^{-1}$ ) aufgrund der verbreiteten Praxis die Schafe über Nacht einzupferchen (Dung wird nicht zurückgebracht). N-Verluste aufgrund von Fleisch und Wollernten sind vergleichbar gering ( $0.2 \text{ g N m}^{-2} \text{ yr}^{-1}$ ). Eine hohe potentielle Ökosystemfruchtbarkeit (N-Vorräte in der organischen Bodensubstanz zwischen  $800 - 1000 \text{ g N m}^{-2}$ ) deutet ein beträchtliches Regenerationspotential, sogar nach langfristig starker Beweidung an. Die komplexen Interaktionen zwischen Klimavariabilität und Ökosystem Produktivität verlangen jedoch nach flexiblen Beweidungssystemen. Eine Beweidung ist hinsichtlich der N-Bilanz ausgeglichen, sobald die Bestockung auf ein Maß reduziert ist, bei der die verbleibende oberirdische Biomasse die Winderosion und die damit verbundenen N-Verluste unterbindet. Ein Rotationssystem zwischen Heu- und Weideflächen zur Reduzierung der N-Verluste, als auch die Verteilung und Menge der Vorjahresniederschläge als Indikator zur Bemessung der diesjährigen Bestockungsintensität könnte als Ansatz für die Ableitung von Bewirtschaftungsoptionen dienen.

### *Wie Beweidung Pflanzengesellschaften und Umweltparameter beeinflusst*

Wir fragen nach Auswirkungen der Beweidung auf Umweltparameter und Konsequenzen für die Struktur der Pflanzengesellschaft.

Fünf Beweidungsintensitäten wurden auf insgesamt 450 Beprobungspunkten analysiert. Eine Kanonische Korrespondenzanalyse (CCA) wurde genutzt, um die Artzusammensetzung, Artabundanzen und Bedeckung der Vegetation mit Variablen der Nährstoffverfügbarkeit (N, P, K), Oberbodenwassergehalt, oberirdische Biomasse (AGB), organischer Kohlenstoff (OC), Boden N- und S-Gehalte, pH-Wert und Lagerungsdichte zu korrelieren.

Die langfristig unbeweidete Fläche zeigte die höchste N-Verfügbarkeit, höchste OC- und N-Gehalte, niedrigste Lagerungsdichte und eine hohe AGB. Die langfristig stark beweidete Fläche war charakterisiert durch geringe Oberbodenwassergehalte reduzierte N-Verfügbarkeit sowie geringe OC- und N-Gehalte, hohe Lagerungsdichte und sehr geringe AGB. Auf beiden Flächen zeigten dicotyledone Arten hohe Abundanzen, während Graminoide die moderat beweideten Flächen dominierten, wo die K-Verfügbarkeit gering war und die anderen Parameter Mittelmaß zeigten. Unterschiede in der Artenzusammensetzung, Bedeckung und Abundanz konnten am besten mit dem Oberbodenwassergehalt in Kombination

mit der N- und K-Verfügbarkeit erklärt werden. Entsprechend der Reaktion auf Beweidung und Umweltfaktoren konnten die Pflanzenarten in fünf Kategorien von beweidungs-sensitiv bis extrem beweidungstolerant klassifiziert werden.

Die Bedeutung der Wasserverfügbarkeit im Zusammenhang mit Beweidungseffekten auf die Vegetationsstruktur war überraschend groß und könnte allgemein unterschätzt sein. Die gezeigte Verknüpfung von Umweltvariablen und Artzusammensetzung in Abhängigkeit von Beweidungsintensitäten liefert Erkenntnisse für die Entwicklung von Landnutzungsstrategien im Spannungsfeld von Nutztierbewirtschaftung, Grasslandregeneration und Naturschutz.

### *Hangexposition und Effekte auf Produktivität und Artzusammensetzung*

Um die Produktivität und Artzusammensetzung der semi-ariden Graslandökosysteme in Abhängigkeit der Hangexposition (Nord-Süd) zu analysieren wurden oberirdische Biomasse an vier Standorten mit jeweiligen Nord- und Süd-Hängen in 2004 und 2005 untersucht. Eine zusätzliche Untersuchung erfasste die Vegetation sowie physikalische und chemische Bodenparameter an unterschiedlichen Hangpositionen (Hangfuß, -mitte, -spitze).

Nordexponierte Hänge zeigten höhere Produktivität und Artendiversität im Vergleich zu südexponierten Hängen. Pflanzenverfügbares K, P und N limitierten nicht die Produktivität an süd- im Verhältnis zu nordexponierten Hängen. Dies wurde unterstützt durch Ergebnisse der Blattkohlenstoffisotopendiskriminierung im Blatt der wichtigsten Pflanzenarten. Durch eine negative Korrelation zwischen Produktivität und relativer Abundanz von C<sub>4</sub>-Pflanzen, kann die relative Biomasse von C<sub>4</sub>-Pflanzen als Indikator für Wasserstress genutzt werden.

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Marc Giese

Ottendorf, im September 2007

**Table 1:** Species list of plants recorded at the different grassland sites (UG79, UG99, WG, CG, HG, see introduction) at Inner Mongolia Ecosystem Research Station (IMGERS, 43° 38' N and 116° 42' E) in July and August 2004<sup>1</sup>.

No.	Species Name	Chinese Name	Family	Lifeform (Raunkiaer)
1	<i>Achnatherum sibiricum</i>	西羽茅	Poaceae	Hemicryptophyte
2	<i>Adenophora stenanthina</i>	长柱沙参	Campanulaceae	Hemicryptophyte
3	<i>Agropyron michnoi</i>	米冰草	Poaceae	Geophyte
4	<i>Allium anisopodium</i>	矮葱	Liliaceaea	Geophyte
5	<i>Allium bidentatum</i>	双齿葱	Liliaceaea	Geophyte
6	<i>Allium condensatum</i>	黄花葱	Liliaceaea	Geophyte
7	<i>Allium ramosum</i>	野韭	Liliaceaea	Geophyte
8	<i>Allium senescens</i>	山葱	Liliaceaea	Geophyte
9	<i>Allium tenuissimum</i>	细叶葱	Liliaceaea	Geophyte
10	<i>Androsace longifolia</i>	长叶点地梅	Primulaceae	Hemicryptophyte
11	<i>Anemarrhena asphodeloides</i>	知母	Liliaceaea	Geophyte
12	<i>Artemisia commutala</i>	变蒿	Asteraceae	Chameaphyte
13	<i>Artemisia frigida</i>	冷蒿	Asteraceae	Chameaphyte
14	<i>Artemisia scoparia</i>	黄蒿	Asteraceae	Chameaphyte
15	<i>Artemisia sieversiana</i>	大籽蒿	Asteraceae	Therophyte
16	<i>Astragalus galactites</i>	乳白花黄芪	Leguminoseae	Hemicryptophyte
17	<i>Axyris amaranthoides</i>	轴藜	Chenopodiaceae	Therophyte
18	<i>Bupleurum scorzonerifolium</i>	红柴胡	Apiaceae	Hemicrypt./Geop.
19	<i>Cannabis sativa</i>	野大麻	Cannabaceae	Therophyte
20	<i>Caragana microphylla</i>	锦鸡儿	Leguminoseae	Nanophanerophyte
21	<i>Carex korshinskyi</i>	黄囊苔草	Cyperaceae	Geophyte
22	<i>Chenopodium aristatum</i>	刺藜	Chenopodiaceae	Therophyte
23	<i>Chenopodium glaucum</i>	灰绿藜	Chenopodiaceae	Therophyte
24	<i>Cleistogenes squarrosa</i>	糙隐子草	Poaceae	Hemicryptophyte
25	<i>Convolvulus arvensis</i>	田旋花	Convolvulaceae	Geophyte
26	<i>Cymbaria daurica</i>	莘芭	Scrophulariaceae	Hemicryptophyte
27	<i>Dontostemon micranthus</i>	小花花旗竿	Brassicaceae	Therophyte
28	<i>Echinops latifolius</i>	蓝刺头	Compositae	Hemicryptophyte
29	<i>Euphorbia esula</i>	乳浆大戟	Euphorbiaceae	Hemicrypt/Geop.
30	<i>Festuca dahurica</i>	羊茅	Poaceae	Hemicryptophyte
31	<i>Filifolium sibiricum</i>	线叶菊	Compositae	Hemicryptophyte
32	<i>Galium verum</i>	蓬子菜	Rubiaceae	Geophyte
33	<i>Gentiana dahurica</i>	达乌里龙胆	Gentianaceae	Hemicryptophyte
34	<i>Gueldenstaedtia stenophylla</i>	狭叶米口袋	Leguminoseae	Hemicryptophyte
35	<i>Haplophyllum dauricum</i>	草芸香	Rutaceae	Hemicryptophyte
36	<i>Heteropappus altaicus</i>	阿狗	Asteraceae	Hemicryptophyte
37	<i>Iris tenuifolia</i>	细叶鸢尾	Liliaceaea	Geophyte
38	<i>Kochia prostrata</i>	木地肤	Chenopodiaceae	Chameaphyte
39	<i>Koeleria cristata</i>	落草	Poaceae	Hemicryptophyte
40	<i>Lappula redowskii</i>	鹤虱	Boraginaceae	Therophyte
41	<i>Leontopod. leontopodioides</i>	火绒草	Asteraceae	Hemicryptophyte
42	<i>Leymus chinensis</i>	羊草	Poaceae	Geophyte
43	<i>Lilium pumilum</i>	细叶百合	Liliaceaea	Geophyte
44	<i>Limonium bicolor</i>	二色补血草	Plumbaginaceae	Hemicryptophyte
45	<i>Melandrium apricum</i>	女娄菜	Caryophyllaceae	Therophyte
46	<i>Melilotus suaveolens</i>	草木樨	Leguminoseae	Therophyte

**Table 1, continued:** Species list of plants recorded at the different grassland sites (UG79, UG99, WG, CG, HG, see introduction) at Inner Mongolia Ecosystem Research Station (IMGERS, 43° 38' N and 116° 42' E) in July and August 2004<sup>1</sup>.

No.	Species Name	Chinese Name	Family	Lifeform (Raunkiaer)
47	<i>Melissitus ruthenica</i>	扁蓊豆	Leguminosae	Hemicryptophyte
48	<i>Orostachys malacophyllus</i>	钝叶瓦松	Crassulaceae	Therophyte
49	<i>Oxytropis myriophylla</i>	多叶棘豆	Leguminosae	Hemicryptophyte
50	<i>Phlomis dentosa</i>	尖齿糙苏	Lamiaceae	Hemicryptophyte
51	<i>Plantago asiatica</i>	车前	Plantaginaceae	Hemicryptophyte
52	<i>Poa argunensis</i>	早熟禾	Poaceae	Hemicryptophyte
53	<i>Polygonum divaricatum</i>	叉分蓼	Polygonaceae	Geophyte
54	<i>Potentilla acaulis</i>	星毛委陵菜	Rosaceae	Geophyte
55	<i>Potentilla betonicaefolia</i>	三出委陵菜	Rosaceae	Geophyte
56	<i>Potentilla bifurca</i>	二裂委陵菜	Rosaceae	Geophyte
57	<i>Potentilla tanacetifolia</i>	菊叶委陵菜	Rosaceae	Hemicryptophyte
58	<i>Potentilla verticillaris</i>	轮叶委陵菜	Rosaceae	Hemicryptophyte
59	<i>Ptilotrichum canescens</i>	燥原芥	Brassicaceae	Hemicryptophyte
60	<i>Pulsatilla tenuiloba</i>	细裂白头翁	Ranunculaceae	Hemicryptophyte
61	<i>Salsola collina</i>	猪毛菜	Chenopodiaceae	Therophyte
62	<i>Saposhnikovia divaricata</i>	防风	Apiaceae	Hemicryptophyte
63	<i>Saussurea japonica</i>	翼茎风毛菊	Asteraceae	Therophyte
64	<i>Schizonepeta multifida</i>	多裂叶荆芥	Lamiaceae	Therophyte
65	<i>Scutellaria scordiifolia</i>	并头黄芩	Lamiaceae	Hemicryptophyte
66	<i>Serratula centauroides</i>	麻花头	Asteraceae	Hemicryptophyte
67	<i>Setaria viridis</i>	狗尾草	Poaceae	Therophyte
68	<i>Sibbaldia adpressa</i>	伏毛山莓草	Rosaceae	Hemicryptophyte
69	<i>Silene jenissensis</i>	早麦瓶草	Caryophyllaceae	Hemicryptophyte
70	<i>Spiraea aquilegifolia</i>	耧斗叶绣线菊	Rosaceae	Nanophanerophyte
71	<i>Stellera chamaejasme</i>	狼毒	Thymelaeaceae	Hemicryptophyte
72	<i>Stipa grandis</i>	大针茅	Poaceae	Hemicryptophyte
73	<i>Tephrosia kirilowii</i>	狗舌草	Asteraceae	Hemicryptophyte
74	<i>Thalictrum petaloideum</i>	瓣蕊唐松草	Ranunculaceae	Geophyte
75	<i>Thermopsis lanceolata</i>	披针叶黄华	Leguminosae	Hemicryptophyte
76	<i>Thymus serpyllum</i>	百里香属	Lamiaceae	Chameaphyte

<sup>1</sup> sample plot size: 1 x 1 m arranged in a grid design, distances 5-10 m for sites UG79, UG99, HG (n = 100), distances 10-50 m for sites WG and CG (n = 124).

**Table 2:** Species cover (%) recorded at the different grassland sites (UG79, UG99, WG, CG, HG, see introduction) at Inner Mongolia Ecosystem Research Station (IMGERS, 43° 38' N and 116° 42' E) in July and August 2004 <sup>1</sup>.

No.	Species Name	UG79	UG99	WG	CG	HG
1	<i>Achnatherum sibiricum</i>	11,52	7,16	5,01	2,84	
2	<i>Adenophora stenanthina</i>	0,02	0,03			
3	<i>Agropyron michnoi</i>	4,72	4,12	2,46	2,05	4,34
4	<i>Allium anisopodium</i>	0,25		0,05		
5	<i>Allium bidentatum</i>	0,15		0,12	0,14	0,04
6	<i>Allium condensatum</i>	0,04				
7	<i>Allium ramosum</i>	0,10		0,01	0,02	0,09
8	<i>Allium senescens</i>	0,10				
9	<i>Allium tenuissimum</i>	0,32	0,01	0,47	0,19	0,10
10	<i>Androsace longifolia</i>	0,05				
11	<i>Anemarrhena asphodeloides</i>	0,18		0,06	0,10	
12	<i>Artemisia commutala</i>	0,23				0,07
13	<i>Artemisia frigida</i>	1,98	0,04			11,10
14	<i>Artemisia scoparia</i>	0,04				0,35
15	<i>Artemisia sieversiana</i>	0,20				
16	<i>Astragalus galactites</i>					0,03
17	<i>Axyris amaranthoides</i>	0,18	0,04	1,63	1,30	0,34
18	<i>Bupleurum scorzonerifolium</i>	0,02				
19	<i>Cannabis sativa</i>					
20	<i>Caragana microphylla</i>	5,67	0,10			
21	<i>Carex korshinskyi</i>	7,34	11,98	8,65	8,23	12,80
22	<i>Chenopodium aristatum</i>	0,17				
23	<i>Chenopodium glaucum</i>					
24	<i>Cleistogenes squarrosa</i>	5,26	6,00	8,57	7,88	12,18
25	<i>Convolvulus arvensis</i>	0,04				
26	<i>Cymbaria daurica</i>	0,62	0,01			
27	<i>Dontostemon micranthus</i>		0,01	0,01	0,03	
28	<i>Echinops latifolius</i>	0,18				
29	<i>Euphorbia esula</i>	0,02				
30	<i>Festuca dahurica</i>	0,85	3,02	0,37		
31	<i>Filifolium sibiricum</i>	0,15				
32	<i>Galium verum</i>	0,61				
33	<i>Gentiana dahurica</i>			0,02		
34	<i>Gueldenstaedtia stenophylla</i>	0,01				
35	<i>Haplophyllum dauricum</i>	0,03				
36	<i>Heteropappus altaicus</i>	1,65				
37	<i>Iris tenuifolia</i>	0,04	0,04			
38	<i>Kochia prostrata</i>	0,15		0,02		0,11
39	<i>Koeleria cristata</i>	2,21	4,68	0,81	0,27	0,31
40	<i>Lappula redowskii</i>					0,04
41	<i>Leontopod. leontopodioides</i>					
42	<i>Leymus chinensis</i>	4,97	11,81	10,20	8,27	1,51
43	<i>Lilium pumilum</i>					
44	<i>Limonium bicolor</i>	0,03				
45	<i>Melandrium apricum</i>					
46	<i>Melilotus suaveolens</i>	0,01				

**Table 2, continued:** Species cover (%) recorded at the different grassland sites (UG79, UG99, WG, CG, HG, see introduction) at Inner Mongolia Ecosystem Research Station (IMGERS, 43° 38' N and 116° 42' E) in July and August 2004<sup>1</sup>.

No.	Species Name	UG79	UG99	WG	CG	HG
47	<i>Melissitus ruthenica</i>					0,69
48	<i>Orostachys malacophyllus</i>	0,01				
49	<i>Oxytropis myriophylla</i>					
50	<i>Phlomis dentosa</i>	0,05				
51	<i>Plantago asiatica</i>					
52	<i>Poa argunensis</i>	0,17	0,56	0,39	0,42	0,14
53	<i>Polygonum divaricatum</i>	0,03				
54	<i>Potentilla acaulis</i>	2,41	0,12	0,19	0,05	16,67
55	<i>Potentilla betonicaefolia</i>				0,03	
56	<i>Potentilla bifurca</i>	4,38	0,12			0,53
57	<i>Potentilla tanacetifolia</i>	1,97				4,72
58	<i>Potentilla verticillaris</i>	0,28				
59	<i>Ptilotrichum canescens</i>	0,02	0,01			
60	<i>Pulsatilla tenuiloba</i>	1,23				
61	<i>Salsola collina</i>	0,13	0,01	0,01	0,10	
62	<i>Saposhnikovia divaricata</i>	0,02				
63	<i>Saussurea japonica</i>	0,04				
64	<i>Schizonepeta multifida</i>	0,29	0,02			
65	<i>Scutellaria scordiifolia</i>	0,03				
66	<i>Serratula centauroides</i>	3,10				
67	<i>Setaria viridis</i>	0,06				
68	<i>Sibbaldia adpressa</i>					0,26
69	<i>Silene jenissensis</i>	0,03				
70	<i>Spiraea aquilegifolia</i>	0,03				
71	<i>Stellera chamaejasme</i>	0,03	0,07	0,04		
72	<i>Stipa grandis</i>	12,20	18,08	21,01	23,47	2,38
73	<i>Tephrosieris kirilowii</i>	0,01				
74	<i>Thalictrum petaloideum</i>	0,03	0,03	0,13	0,04	0,56
75	<i>Thermopsis lanceolata</i>	0,10				
76	<i>Thymus serpyllum</i>					0,03

<sup>1</sup> sample plot size: 1 x 1 m arranged in a grid design, distances 5-10 m for sites UG79, UG99, HG (n = 100), distances 10-50 m for sites WG and CG (n = 124).



**Table 3:** Relative abundance of species recorded at different grassland sites (UG79, UG99, WG, CG, HG, see introduction) at Inner Mongolia Ecosystem Research Station (IMGERS, 43° 38' N and 116° 42' E) in July and August 2004 <sup>1</sup>.

No.	Species Name	UG79	UG99	WG	CG	HG
1	<i>Achnatherum sibiricum</i>	99	80	96	83	
2	<i>Adenophora stenanthina</i>	8		3		
3	<i>Agropyron michnoi</i>	82	73	88	77	93
4	<i>Allium anisopodium</i>	50	10	1	6	3
5	<i>Allium bidentatum</i>	30	16		19	5
6	<i>Allium condensatum</i>	13	3		1	
7	<i>Allium ramosum</i>	12	5	1	5	13
8	<i>Allium senescens</i>	22	1		2	2
9	<i>Allium tenuissimum</i>	46	73	12	42	28
10	<i>Androsace longifolia</i>	11	1			
11	<i>Anemarrhena asphodeloides</i>	6	6		14	
12	<i>Artemisia commutala</i>	24	1			15
13	<i>Artemisia frigida</i>	59		5		97
14	<i>Artemisia scoparia</i>	6	1		6	36
15	<i>Artemisia sieversiana</i>	2	1			
16	<i>Astragalus galactites</i>	1	2		1	3
17	<i>Axyris amaranthoides</i>	22	94	7	93	46
18	<i>Bupleurum scorzonerifolium</i>	6	1	2		
19	<i>Cannabis sativa</i>	1				
20	<i>Caragana microphylla</i>	76	1	1		
21	<i>Carex korshinskyi</i>	100	100	100	98	100
22	<i>Chenopodium aristatum</i>	37				
23	<i>Chenopodium glaucum</i>	9	1	1	5	3
24	<i>Cleistogenes squarrosa</i>	97	94	100	100	100
25	<i>Convolvulus arvensis</i>	4				
26	<i>Cymbaria daurica</i>	30	1	4		
27	<i>Dontostemon micranthus</i>	5	10	3	15	4
28	<i>Echinops latifolius</i>	4				1
29	<i>Euphorbia esula</i>	5		1		1
30	<i>Festuca dahurica</i>	13	23	51	2	
31	<i>Filifolium sibiricum</i>	5				
32	<i>Galium verum</i>	11				
33	<i>Gentiana dahurica</i>		1		1	
34	<i>Gueldenstaedtia stenophylla</i>	1	1	2	2	
35	<i>Haplophyllum dauricum</i>	9		2		
36	<i>Heteropappus altaicus</i>	37	3			4
37	<i>Iris tenuifolia</i>	7	6	5	2	
38	<i>Kochia prostrata</i>	32	1	2		32
39	<i>Koeleria cristata</i>	77	64	92	46	47
40	<i>Lappula redowskii</i>	1	1		2	6
41	<i>Leontopod. leontopodioides</i>	1				
42	<i>Leymus chinensis</i>	79	99	99	99	62
43	<i>Lilium pumilum</i>	1	1		2	
44	<i>Limonium bicolor</i>	1				1
45	<i>Melandrium apricum</i>	2	1			
46	<i>Melilotus suaveolens</i>	4		1		

**Table 3, continued:** Relative abundance of species recorded at the different grassland sites (UG79, UG99, WG, CG, HG, see introduction) at Inner Mongolia Ecosystem Research Station (IMGERS, 43° 38' N and 116° 42' E) in July and August 2004<sup>1</sup>.

No.	Species Name	UG79	UG99	WG	CG	HG
47	<i>Melissitus ruthenica</i>	9	1		3	60
48	<i>Orostachys malacophyllus</i>	1			1	
49	<i>Oxytropis myriophylla</i>	1	1			3
50	<i>Phlomis dentosa</i>	4			1	
51	<i>Plantago asiatica</i>	1				1
52	<i>Poa argunensis</i>	37	39	35	21	32
53	<i>Polygonum divaricatum</i>	2				
54	<i>Potentilla acaulis</i>	58	6	9	4	86
55	<i>Potentilla betonicaefolia</i>				3	
56	<i>Potentilla bifurca</i>	81	2	19		42
57	<i>Potentilla tanacetifolia</i>	74	1			96
58	<i>Potentilla verticillaris</i>	46	1	2	1	
59	<i>Ptilotrichum canescens</i>	3	1	3		
60	<i>Pulsatilla tenuiloba</i>	23				
61	<i>Salsola collina</i>	36	9	3	41	4
62	<i>Saposhnikovia divaricata</i>	18				2
63	<i>Saussurea japonica</i>	10				
64	<i>Schizonepeta multifida</i>	26	2	4		
65	<i>Scutellaria scordiifolia</i>	2				1
66	<i>Serratula centauroides</i>	87				
67	<i>Setaria viridis</i>	6				
68	<i>Sibbaldia adpressa</i>	1				27
69	<i>Silene jenissensis</i>	1	2		1	
70	<i>Spiraea aquilegifolia</i>	1				
71	<i>Stellera chamaejasme</i>	1	2	5		
72	<i>Stipa grandis</i>	99	100	100	100	60
73	<i>Tephroses kirilowii</i>	2				1
74	<i>Thalictrum petaloideum</i>	16	21	19	20	63
75	<i>Thermopsis lanceolata</i>	8				
76	<i>Thymus serpyllum</i>					1

<sup>1</sup> sample plot size: 1 x 1 m arranged in a grid design, distances 5-10 m for sites UG79, UG99, HG (n = 100), distances 10-50 m for sites WG and CG (n = 124).

## Lebenslauf

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Kiel, den 12. September 2007

Marcus Giese