

DISSERTATION

EFFECTS OF GRAZING AND COMMUNITY-BASED MANAGEMENT ON
RANGELANDS OF MONGOLIA

Submitted by

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ABSTRACT

EFFECTS OF GRAZING AND COMMUNITY-BASED MANAGEMENT ON RANGELANDS OF MONGOLIA

Rangelands are a crucial renewable resource and wealth for Mongolians, who have a long history of sustainable livestock herding over their vast territory, which is one of the largest intact temperate rangelands on Earth. Recent studies suggest that both livestock and climate change have strong effects on Mongolian grasslands at both broad and local scales. It is not clear if these changes represent degradation and the relative role of livestock and climate in causing change. Local communities and their donor supporters have responded to these changes by establishing community-based rangeland management (CBRM) initiatives, which have grown rapidly in number since consecutive years of natural disasters (*dzud* and drought) occurred in Mongolia in the late 1990s. Now there are over 2000 such initiatives across Mongolia but there have been no broad-scale, well-replicated studies of the ecological outcomes of these initiatives.

The overarching goal of this dissertation was to deepen our understanding of the effects of winter grazing and community-based management on rangelands of Mongolia by expanding the scope and scale of previous work to the national scale and by linking field and remote sensing data on vegetation changes. At the field level, I used broad-scale data to look at the effects of livestock grazing around piospheres or grazing gradients created around 143 winter shelters in four ecological zones of Mongolia. In order to understand long-term winter grazing and climate effects, I quantified trends in MODIS NDVI, livestock grazing intensity and climate variables to analyze change from 2000-2013. Lastly I quantified the effects of community-based

rangeland management (CBRM) on rangeland vegetation and soils across four ecological zones in Mongolia. In this thesis, grazing includes livestock trampling and urine and feces defecation besides actual grazing.

Winter grazing-induced changes were largest in the steppe (170 mm rainfall), least in the eastern steppe (258 mm rainfall) and moderate in the mountain and forest steppe (239 mm rainfall) and desert steppe (131 mm rainfall). In the mountain and forest steppe, sedge and shrub cover were greater in intensely grazed areas close to winter shelters. At the finer resolution species level, the grazing tolerant and dominant sedge *Carex duriuscula* was more abundant in heavily grazed sites and this species is recognized in Mongolia as an indicator of grazing-induced vegetation change. In the steppe zone, heavily grazed pastures had lower grass, higher forb and abundant annual weed cover, and wider open gaps between perennial plant bases, indicating the effects of grazing. Grazing affected the distribution of palatable forbs and grasses where cover was greater in lightly grazed pastures far from winter shelters. Unlike other ecological zones, we found very few grazing-induced changes in the eastern steppe and they were unexpected. Here, grass was more abundant and forbs less abundant in heavily grazed pastures close to winter shelters. We speculate this occurred because of abundant fires in this zone. In the steppe, lower grass cover and higher forb cover and abundant annual unpalatable weeds in the heavily grazed areas near winter shelters indicated effects of grazing. Overall, grazing created larger gaps between vegetation indicating grazing-induced changes in the steppe zone. In the desert steppe, heavily grazed pastures near winter shelters supported less grass, shrub and litter cover and more total and annual forb cover, similar to the patterns observed in the steppe ecological zone.

Winter grazing did not affect NDVI consistently from year to year in the last 14 years; the winter grazing effect was strong and significant in some years and was not detectable in other years, likely because of interactions between the levels of growing season rainfall and livestock grazing. NDVI showed winter grazing gradients, where NDVI was lower in heavily grazed pasture and higher in lightly grazed pastures in some years and not others. These grazing gradients appeared after periods of low livestock grazing, especially as measured by current and previous season NDVI. The NDVI patterns, compared to precipitation and forage use patterns, showed an apparent shift from precipitation-dominated vegetation dynamics in the early 2000s to livestock-dominated vegetation dynamics in the late 2000s especially in mountain and forest steppe and steppe. Our study also showed that NDVI in the mountain and forest steppe and steppe tracked winter grazing more closely than NDVI in the desert steppe, as predicted by non-equilibrium rangeland dynamics theory.

We used the coefficient of variation of NDVI as a measure of resistance of rangeland NDVI, with more resistance shown by a low average inter-annual CV over time. We found that NDVI in heavily grazed pastures was less resistant and more variable over time than lightly grazed pastures. We also used the speed of recovery after dry periods as a measure of resilience or elasticity of pastures grazed at different intensities. In only one of our six study areas, lightly grazed pastures recovered faster and were more resilient than heavily grazed pastures.

Our study is the first of its kind to compare the effects of CBRM across Mongolia's major ecological zones using many locations in each zone. Our results suggest that CBRM initiatives have neither many nor major impacts on any aspect of winter pastures across ecological zones. But there were some *subtle* effects of CBRM on pastures. In the mountain and forest steppe, there was less cover of the increaser grass species, *Cleistogenes squarrosa*, more

connected plant patches and less erosion in CBRM than non-CBRM pastures. In the eastern steppe, CBRM pastures had greater litter biomass and less cover of both annual plants and *Carex duriuscula*. *C. duriuscula* is grazing tolerant and increases in abundance with moderate to heavy grazing. In the steppe, there was more litter, shrub and standing dead plant biomass in CBRM pastures than non-CBRM pastures, suggesting that CBRM may improve condition of pastures in the steppe. Our results suggest very little difference between CBRM and non-CBRM pastures in the desert steppe, except less connected plant patches and more erosion in CBRM pastures, and more abundance of the annual grass, *Eragrostis minor*, in non-CBRM pastures.

The results of this dissertation provide an increased understanding of current level of grazing-induced changes, the combined effect of grazing and climate over time and the impacts of CBRM on rangelands in different ecological zones of Mongolia. We believe also that findings of this research provide comprehensive baseline information for the implementation of future rangeland monitoring and for better policy development in the future in Mongolia.

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DEDICATION

To my son: Ravdannyam

For his great and bright future

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CHAPTER ONE: EFFECTS OF GRAZING AND COMMUNITY-BASED MANAGEMENT ON RANGELANDS OF MONGOLIA

1.1. Introduction

The overarching goal of this dissertation is to deepen our understanding of the effects of winter grazing and community-based management on the rangelands of Mongolia. I¹ do this by expanding the scope and scale of previous work to the national scale and by linking field-level observations of vegetation and soils to remote sensing observations of vegetation changes. At the field level, I used the broad-scale data collected by the large ecological teams of the Mongolian Rangeland Resilience project (Fernandez-Gimenez et al. 2009) to look at the effects of livestock grazing along grazing gradients created by livestock around winter shelters in four ecological zones of Mongolia. In order to understand long-term grazing and climate effects, I used instrumental observations from remote-sensing covering winter shelters in selected counties (*soums*) in the three ecological zones. Lastly I wanted to understand the effects of community-based rangeland management (CBRM) on rangeland vegetation and soils in Mongolia. I chose to study all four ecological zones in Mongolia to understand how these effects differ across a climatic gradient but also to make a broad test of the predictions of the equilibrium and non-equilibrium rangeland dynamics theory.

This first chapter starts with some brief background information about Mongolian rangelands, rangeland dynamics, and recent changes in Mongolian rangelands. Then this chapter reviews the more general literature on: 1) the effects of livestock grazing on vegetation (plant cover, biomass, functional groups, plant palatability, and species richness/diversity), and the

¹ Note that I use the pronoun ‘I’ in my first and last chapter and use ‘We’ for all my data chapters (2-4), because all those who contributed will be co-authors when we publish the data chapters.

soil's surface, 2) equilibrium and non-equilibrium rangelands dynamics, 3) the concepts and measurement of rangeland resilience and resistance and 4) the effects of community-based rangeland management on resource/rangeland condition. Lastly, the chapter introduces my research questions and hypotheses that motivate each chapter.

1.2. Background

1.2.1. Mongolian rangelands

Rangelands are a crucial renewable resource and wealth for Mongolians, who have a long history of sustainable livestock herding over their vast territory. Mongolia is an upland country with a territory of 1.56 million km², with about 85 percent of its land area above 1000 m in elevation (ASL) and most lying between 1000 and 1500 m (Mongolian Society for Range Management 2010). In Mongolia, 72 percent of the land, or 1.12 million km² (112.8 million hectares), is categorized as a rangeland which supports the livelihoods of over 160,260 herder families (NSO 2013). The rangeland falls into five ecological zones, namely high mountain, forest steppe, steppe, desert steppe and desert with markedly different terrain, climate, flora and fauna (Ulziikhutag 1989).

Mongolia has a continental climate with extreme fluctuations in temperature, both daily and annually. July is the warmest month, with mean temperatures of 15°C in the mountains and 20 to 30°C in the southern desert steppe and desert. The lowest temperatures are recorded in January, when monthly temperature averages below -15°C and minimum temperatures fall as low as -40°C (Mongolian Society for Range Management 2010). Precipitation is generally low, varying within the different ecological zones. It ranges from less than 50 mm annually in the extreme desert in the south (Gobi desert region) to about 500 mm in the north in the forest steppe

(NHDR 2011). Average countrywide precipitation is about 230 mm annually. Most precipitation (95%) falls in the summer with less than 3% of total precipitation falling in the winter (MARCC 2009).

Grasslands in Mongolia have a very short growing season, limited by cold temperatures and variable precipitation. Pasture vegetation growth begins in mid-May and mostly ends in mid-August, with growth lasting generally from 80 to 100 days. The growing season is longer in the ecological zones of desert steppe and desert areas where it is warmer than it is in other zones (Batjargal 1997).

1.2.2. Recent changes in Mongolian rangelands

Climate is changing in Mongolia. According to the records at 48 meteorological stations, which are distributed evenly throughout the country, the annual mean temperature of Mongolia increased by 2.14⁰C over the last 70 years (MARCC 2009). Precipitation has also changed geographically and seasonally. Annual precipitation has decreased by 8.7-12.5% in the central and desert regions and increased by 3.5-9.3% in the eastern and western regions in the last 65 years (Mongolian Society for Range Management 2010). Also winter precipitation has increased and warm season precipitation has decreased slightly. Another indication of precipitation change is the change in its intensity, with more thunderstorms, flash floods, hail and other events occurring during the growing season in the past 2 decades than previously (MARCC 2009).

Rivers in Mongolia are also drying. A water inventory conducted in 2007 by Ministry of Nature and Environment revealed that the following water sources have dried up: 852 rivers and streams out of a total of 5,128; 2,277 springs out of 9,306; 1,181 lakes and ponds out of a total of 3,747 (MARCC 2009). There could be many contributing causes of this drying besides climate

change, such as water use for irrigation, mining, and changing water retention capacity of the soil. This drying could also be caused by human activities in watersheds such as deforestation or change in vegetation cover, but there is no detailed study of the reasons for the loss of surface water. Also studies of changes in river water flow demonstrate inconsistent results (Batjargal 1997, MARCC 2009).

Plant species richness, diversity and productivity are also in decline. According to Lkhagvajav's (2006) study, from 1961 to 2006, the number of plant species decreased by 23.6% in the desert steppe and 50% in the forest steppe. During the same period, pastureland productivity decreased by 28.6% in the desert and 52.2% in the steppe. Another long-term field study, from 1995-2013, showed that species richness and diversity declined significantly in the mountain and forest steppe zone due to increased grazing pressure and warming temperatures (Khishigbayar et al. 2015). The other study conducted in the mountain and forest steppe concluded that rangeland biomass decreased from the pre-collective period and this decrease was significantly correlated with changes in grazing management and increased stocking density (Sankey et al. 2009).

Changes in satellite-derived NDVI vegetation data over last the 27 years in Mongolian rangelands showed that there has been a visible difference in the rate and direction of change in vegetation cover comparing among ecological zones (Erdenetuya 2006). In general, the northern mountain, forest steppe and steppe regions have more vegetation than arid southern desert steppe and steppe regions, as would be expected based on rainfall. In the more mesic mountain, forest steppe and steppe zones, vegetation cover increased from 1985 to 1997. But since 1997, there has been a sharp decline in vegetation cover, especially in the forest steppe, which coincided with exceptionally severe droughts and strong temperature increases. The forest steppe recovered

quickly after disturbance, while mountain and steppe regions recovered more slowly, with recovery rates possibly affected by heavy livestock grazing. Also even where vegetation has recovered, it often does not support the same species diversity or palatability. Further ground monitoring data will be valuable for clarifying the relative role of climate change and anthropogenic influences on rangeland vegetation (Mongolian Society for Range Management 2010).

The Mongolian livestock herd has changed strongly in the last few decades in response to political change and natural disasters. The privatization of the country's livestock collectives in 1991-1993 provided an incentive for many Mongolians to turn to herding; livestock numbers increased from 23 million to 33 million by 1999 (Fernandez-Gimenez and Allen-Diaz 1999, Mau and Chantsallkham 2006). Three-years of severe *dzuds*², or winter storms, from 1999-2002, caused livestock numbers to decrease by 30%. Since then, livestock numbers have peaked at 44 million in 2009 and declined again by 10 million in the 2009-2010 *dzud* (NSO 2013). As of December 2014, livestock numbers recovered quickly after the 2010 *dzud* and has reached 50 million.

1.2.3. Effects of livestock on Mongolia rangelands

Recent studies suggest that both livestock and climate change have strong effects on Mongolian grasslands at both broad and local scales (Hilker et al. 2013, Liu et al. 2013, Khishigbayar et al. 2015). Recent research suggests that livestock grazing and climate are causing widespread decline in vegetation greenness (Hilker et al. 2013, Liu et al. 2013), but the extent and relative impact of these causes is unclear (Sankey et al. 2009, Wesche et al. 2010,

² *Dzud* occurs in winter and is usually caused by deep snow, extreme cold, and strong snow storm. But in desert and desert steppe, extreme cold and heavy wind with no snowfall can occur and Mongolians call this a black *dzud*.

Addison et al. 2012). Even in pastures where species changes have occurred due to grazing, as observed by both scientists and herders, herders sometimes say there is no change in the suitability of those pastures for livestock grazing (Kakinuma et al. 2008). Long-term empirical studies suggest that wetter Mongolian rangelands may be at a tipping point, with livestock grazing starting to affect rangelands in subtle ways over the last 20 years, but these changes may be largely reversible at this time (Khishigbayar et al. 2015). In drier rangelands, long-term empirical studies suggest similar subtle changes, but that recovery is not reversible over 6 years of exclusion of grazing (Sasaki et al. 2013). These conflicting results point to the need for more, rigorous field-based studies on this issue, coupled with remote sensing studies over time, comparing impacts across ecological zones.

1.3. Literature Review

1.3.1 Effects of livestock grazing on rangelands

Rangelands occupy approximately 40% of terrestrial land surface, occur mostly in arid and semi-arid lands, and support livestock production and many other activities (Garcia et al. 2014, Reid et al. 2014). Rangelands usually support low vegetation, and include grasslands, shrublands, desert, steppe, marshes, tundras, alpine communities (Allen et al. 2011). Rangelands usually have dry or cold climates (or both) and usually have low human populations. While they do support significant livestock production, rangelands are also used by wildlife and by people for mining, cropping, energy production and other activities (Herrick et al. 2012). Globally, most rangelands are owned in common by groups of people, with significant rangelands areas held privately in the US, Australia and elsewhere (Reid et al. 2014).

In these arid environments, grazing effects are not the same from one place to another at the landscape level. Livestock have more impacts around places where they congregate, like around water points, pastoral settlements and salt licks (Tolsma et al. 1987, Andrew 1988, Landsberg et al. 2003). Concentrations of livestock create ‘grazing gradients’ or ‘piospheres’, with high impacts close to concentration points and low impacts farther away. These grazing gradients have been used to assess the impact of livestock on rangelands (Tolsma et al. 1987, Landsberg et al. 2002).

Also at the landscape scale, livestock graze in different parts of the landscape at different intensities, determined by interactions with natural factors such as vegetation, soil and landform (Milchunas et al. 1988, Bailey et al. 1996, Adler and Hall 2005). Because of this, rangeland scientists use the concept of ‘ecological’ site to understand how grazing differs according to the pattern and organization of plants, animals and soils on a particular landscape. These ecological sites respond in predictable ways to grazing and natural disturbances and thus knowledge of ecological sites is crucial for appropriate management decisions and scenarios of the future effects of grazing (Bestelmeyer and Brown 2010). An ecological site is defined as “a distinctive kind of land with specific physical characteristics that differs from other kind of lands in its ability to produce a distinctive kind and amount of vegetation, and in its ability to respond to management actions and natural disturbances” (Bestelmeyer and Brown 2010).

The US Natural Resources Conservation Service uses the concept of ecological site (Brown 2010) to create spatial groupings on landscapes based on soil properties and processes within a climate zone to predict the dynamics of vegetation and related resources. Soil properties include soil texture or soil depth. Soil processes include a series of actions in the soil, for example, water percolating into the soil that determines soil moisture content (Duniway et al.

2010). These groupings allow scientists to understand the inherent ecological potential of the site to support distinct plant communities across the landscape (Brown 2010). There are 3 major linkages from soil to vegetation including attachments of plants to the land surface, plant available water, and soil nutrients. Plants attach to the land surface via rooting, which is affected by the depth of soil to the bedrock or to the water table. Plant available water is affected by the capability of soil to absorb, store and release water to plants. Thus soil water holding capacity, which differs by soil texture, is an essential component for plant available water. Soil nutrients, such as calcium, phosphorus and potassium, are derived from chemical weathering soil minerals. Also nitrogen available for plants is created through the N fixation in the soil as well as through atmospheric input. Also chemical properties of the soil, such as pH and salinity exert control on vegetation.

At a more local scale, livestock have effects on vegetation through grazing and trampling and dung / urine deposition (Schlesinger et al. 1990, Moleele and Perkins 1998, Fernandez-Gimenez and Allen-Diaz 2001). Heavy grazing removes plant biomass and thus can cause a decrease in plant biomass and cover and an increase in bare ground. Heavy grazing can cause change in composition of functional groups, species and palatability of vegetation. If livestock herds are dominated by grazers (like cattle, yaks, horses, sheep) rather than browsers (like camels, goats), heavy grazing can cause a shift in functional groups from herbaceous to woody plants (Schlesinger et al. 1990, Moleele and Perkins 1998). Conversely, heavy browsing can remove woody plants, giving a competitive advantage to herbaceous plants. Sometimes grazing causes a decline in shrubs, like in Mongolia's dry desert steppe (Sasaki et al. 2013). Grazing can also cause the replacement of palatable plants by unpalatable species (Ellison 1960) due to herbivore preferences for palatable species, which alters the competitive balance in favor of

unpalatable plants. With grazing, the relative abundance of tolerant species may increase, often called ‘increasers’ (Anderson and Briske 1995), and sensitive species may decrease, often called ‘decreasers’ (Dyksterhuis 1949). At the species level, plants with high reproductive rates like annual forbs may also increase with heavy grazing, and plants with slow reproductive rates, like perennials, may decrease (Diaz et al. 2007). For example, the replacement of perennials by annuals along grazing gradients has been found in several studies in Mongolia. (Hilbig 1995, Sasaki et al. 2008a, Hoshino et al. 2009). Trampling and grazing can encourage the spread of weedy or ruderal species (Grime 1977).

Long-term grazing in semi-arid grasslands leads to an increase in the spatial heterogeneity of water, nitrogen, and other soil resources. Dung/urine deposition can also increase soil nutrients around piospheres (Tolsma et al. 1987, Moleele and Perkins 1998, Fernandez-Gimenez and Allen-Diaz 2001). Also, this heterogeneity in soil resources can promote invasion by desert shrubs, which leads to further localization of soil resources under shrub canopies (Schlesinger et al. 1990). Although this has not been documented in Mongolia – in fact most of the evidence points to the opposite trend-increasing grazing causes a decline in shrubs.

1.3.2. Equilibrium and non-equilibrium rangeland dynamics

The two main rangeland vegetation dynamics concepts, equilibrium and non-equilibrium dynamics, have been reassessed and re-interpreted during the last several decades (Ellis and Swift 1988, Briske et al. 2003, Vetter 2005). Rangelands that exhibit equilibrium dynamics are semi-arid or sub-humid, and show a strong density-dependent link between livestock and vegetation. In these situations, the concept of carrying capacity, or an upper limit on the number

of livestock that vegetation in a rangeland can support, can be a useful and appropriate concept. The carrying capacity at a given time is determined by range condition. Range condition is assessed by biomass quantity, cover and species composition (Vetter 2005). Under equilibrium conditions, vegetation often responds to grazing linearly and reversibly (Briske et al. 2003). Where grazing intensity decreases, then vegetation parameters increase. In these types of rangelands, continuous intense grazing causes vegetation changes from palatable to unpalatable, replacement of perennial grasses by annuals, and higher to lower standing biomass and basal cover (Fernandez-Gimenez and Allen-Diaz 1999).

By contrast, in most arid systems, stochastic abiotic factors dominate and extreme climatic events, such as consecutive drought years and severe winter storms, reduce available forage and cause livestock numbers to fall (Briske et al. 2003). Thus, this non-equilibrium system is more dynamic, less predictable and dominated by density-independent abiotic factors. Ellis and Swift (1988) stated that non-equilibrium dynamics likely occur where the coefficient of variation of annual precipitation is greater than 33% and mean annual precipitation is less than 250 mm. But it is likely that this boundary is ‘fuzzy’, with no exact boundary implied. In their view, plant biomass, cover and species composition are driven by climatic factors rather than herbivory and thus plant–herbivore interactions are loosely coupled in non-equilibrium systems.

Illius and O’Connor (1999) emphasized the effects of spatial heterogeneity in non-equilibrium rangeland theory. They proposed that key resources exhibit density-dependent equilibrium dynamics within arid non-equilibrium systems. Key resources are confined to relatively small areas and include small wet areas within a landscape, such as lowlands along rivers, run-on areas, swamps and oases. This part of landscape has relatively stable forage resources because of concentrated ground and surface water (despite the variable rainfall) and

are used by livestock during the droughts. Because key resources are small in size and thus can only sustain low numbers of animals, they exhibit density-dependent characteristics during drought years (Illius and O'Connor 1999).

As Illius and O'Connor suggest, broader theoretical evidence suggests that both equilibrium and non-equilibrium dynamics can operate in the same ecosystem to influence vegetation dynamics at various temporal and spatial scales (Briske et al. 2003). Briske et al (2003) asserted that in rangeland ecology, vegetation dynamics need to be interpreted along an equilibrium and non-equilibrium continuum. Attributes of equilibrium and non-equilibrium systems are based on different levels of internal regulation and the corresponding response to external disturbances (see Table. 1).

Table 1.1. Summary of attributes of equilibrium and non-equilibrium systems (adapted from Briske 2003, page 603, which was modified from Ellis and Swift 1988).

	Equilibrium systems	↔	Non-Equilibrium systems
Abiotic patterns	Relatively constant		Stochastic/variable
Plant-herbivore interactions	Tight coupling Driven biotically		Weak coupling Driven abiotically
Population patterns	Density dependent Populations track carrying capacity		Density independent Dynamic carrying capacity limits population tracking
Community/ecosystem characteristics	Competitive structuring of communities Internal regulation		Competition not expressed External drivers

There are attempts to clarify the applicability this theory. These include emphasizing the importance and relevance of temporal and spatial scale (Briske et al. 2003, Oba et al. 2003, Vetter 2005, Zemmrich 2007) and the need for consistent variables/indicators to quantitatively determine the level of support for this model (Fernandez-Gimenez and Allen-Diaz 1999, Briske et al. 2003, von Wehrden et al. 2012). von Wehrden et al (2012) developed a global rainfall

variability map and reviewed 58 published studies, and concluded that areas away from water sources and key resource areas display non-equilibrium rangeland dynamics with loose density dependence and thus little livestock induced vegetation change. Whereas, rangelands with relatively stable annual precipitation, a low coefficient of variation in precipitation, key resources or nearby to water points display equilibrium dynamics and strong density dependence. Thus grazing-induced change in vegetation is most likely in the rangelands with equilibrium dynamics.

1.3.3. Resilience and resistance of rangelands, concept and measurement

Resilience and resistance are important concepts to understand the complexity and non-linearity of ecological systems. Ecological resilience is defined by Washington - Allen (2008) as "... the degree, manner, and the pace of the restoration of vegetation attributes after a disturbance". Resistance is the inertia of the system to change (Westman 1978).

Scholars have been attempting to measure and quantify resilience and resistance (Westman and O'Leary 1986, Tilman and Downing 1994, Shinoda et al. 2014). They use these concepts for broad-scale analyses of land degradation (Wessels et al. 2007, Washington-Allen et al. 2008) by using different data sources including field vegetation data and remote sensing image data. In field studies, Westman and O'Leary (1986) developed 4 measures of resilience including elasticity, amplitude, malleability and damping by estimating the responses of various plant functional types within a coastal sage scrub plant community. Elasticity is the rate of recovery from a disturbance; amplitude is the threshold beyond which the system has crossed into an irreversible state; malleability is how easily a system remains in a new state without returning to its original state; damping is the extent to which the new system oscillates after

disturbance. In another field study, Tilman and Downing (1994) used measured resistance to drought by calculating the relative change in plant community biomass from the year before a drought to its peak biomass in the drought year.

In remote sensing studies, Washington-Allen et al. (2008) calculated ecological resilience of a Bolivian agro-pastoral community's drylands to multiyear drought by using Landsat satellite imagery from 1972 to 1987 and then assessed land degradation. They used image differencing to calculate the four types of resilience defined by Westman and O'Leary (2004). Differences in the mean and variance of vegetation indices from the pre-drought initial condition year (reference site) to subsequent years were used as proxies for change in vegetation cover and as an indicator of land degradation. In other words, they plotted the mean-variance of vegetation images as a similar approach to using multivariate ordination methods to determine the state and threshold changes in plant species composition in response to disturbance.

Some studies have combined both field and remote sensing data to study resilience of rangelands. Shinoda et al. (2014) calculated drought resilience indices for rangelands in Inner Mongolia and Mongolia by using both satellite imagery and ground data. Resilience was calculated from the recovery of phytomass from drought to post-drought years. They showed that the south side of Khangai Mountains in western Mongolia and the nearby Chinese agro-pastoral region of the Ordos Plateau are both non-resilient. It is important for Mongolia to understand resilience and resistance concepts and to measure them to better understand the response of Mongolian rangelands to the interacting effects of changes in climate and grazing.

1.3.4. Effects of community-based resource management on rangelands

Community-based resource management (CBRM) is an approach to manage natural resources, where resource users agree on norms and rules that determine the access, control and sustainable use of resources (Agrawal and Gibson 2001). In many pastoral settings, unwritten culturally embedded norms have governed rangeland management instead of clearly stated strict rules (Reid et al. 2014). CBRM is a process of planning how to use resources collaboratively, transparently and equitably, how to monitor with less cost; and how to plan for uncertainty in a way to build resilience and adaptation (Fernandez-Gimenez et al. 2012b).

The vigorous theoretical and empirical studies about commons and community-based natural resource management were started after Hardin's famous article 'The Tragedy of the Commons' was published in 1968 (Hardin 1968). Garret Hardin concluded that privatization or government regulation is the only way to overcome the individual temptation to maximize one's own use from shared common natural resources and thus over-use often occurs on the commons. In response to his conclusion, researchers of the commons, including Ostrom (1990), presented a theoretical basis for the commons and eight design principles of successful common property management systems, based on the analysis of 14 case studies of commons across different continents. She described that, despite substantial differences that exist among the common property resource settings, they also share fundamental similarities that underpin her design principles. These design principles describe successful commons as those where: 1) clearly defined boundaries exist which individuals or groups with rights, 2) local resource users balance exploitation vs protection 3) most users are affected by operational rules and thus they participate in developing and modifying these rules, 4) resource users are responsible and actively involved in monitoring resource condition, 5) there are mechanisms among users to assess and punish

those who violate rules depending on the level of violation, 6) there is a mechanism to resolve conflicts locally, 7) there is flexibility in changing resource use rules by users with little involvement and restriction from local government, and 8) multiple layers or nested enterprises allow implementation of complex common property resources.

There are many empirical studies conducted following Ostrom's design principles on common property resources and emphasizing institutional and social relationships of the commons (Feeny et al. 1990, Agrawal and Gibson 1999, Cleaver 2000, Agrawal 2003, Agrawal and Chhatre 2006). The challenges in designing and implementing collaborative community based institutions have been documented. For instance, Cleaver (2000) emphasized that the incentives for individuals to be CBNRM members are not straightforward and simple, because individual incentives are of a bounded nature and interdependent of people's lives. For example, resource access is affected by many other influencing factors like family, social relations, mutual relations of kinship and reciprocity (the 'moral economy'). Incentives to participate differ among people according to their place in the social structure, their personal history, or characteristics other than simply rational economic goals (Cleaver 2000). Thus many initiatives fail to achieve equal participation due to the challenges in overcoming pre-existing power dynamics.

Individuals from powerful social groups take control of community institutions and use community institutions for their own benefit (Dressler et al. 2010). Also it is not always the case that collaboration increases social capital and trust (Rudeen et al. 2012). Cleaver (2000) stated that institutions are intermittent, partial, indeed often invisible, being located in daily interactions of ordinary lives. There are criticisms of donor-facilitated CBRM projects that have top down approaches, disregarding local context and social dynamics (Turner 2011). As a result after project cessation, many donor-initiated groups stop and are unsustainable (Dressler et al. 2010).

Developing optimum institutions is a continuous and evolutionary process. In rangeland systems, it is difficult to sustain and implement management with rigidly defined boundaries and membership, and can be unsound ecologically and create conflicts (Fernandez-Gimenez 2002). In this system, mobility is an essential strategy (Fernandez-Gimenez and Le Febre 2006) and flexible movement allows pastoralists to exploit spatially and temporally heterogeneous resources which provides the basis for a sustainable pastoral system (Turner 2011). Also other common pool resources like mineral licks and natural water sources cannot be given or allocated to specific groups or individuals. Thus, in arid rangeland systems, community-based management needs to be at a broad-scale to coordinate long distance between broader administrative boundaries (Reid et al. 2014).

The potential benefits of CBRM can be found in 3 main areas, including ecological, social and economic, which influence and shape one another. On the other hand, resource management outcomes depend on resource characteristics and biophysical variables that form the context of the socio-political, economic, and institutional variables (Agrawal and Chhatre 2006). The few studies that have measured ecological outcomes show that CBRM groups on common land can improve plant production and cover (Leisher et al. 2012), reduce illegal hunting (Mbaiwa et al. 2011), increase fish biomass and coral cover (Cinner and McClanahan 2015), and contribute to keeping the land intact by preventing fragmentation (Reid et al. 2008). Despite the above studies, there is no broad scale, well-replicated studies on the ecological outcomes of community-based management, especially in Mongolia.

1.4. Rationale for This Study

While Mongolia appears to be one of the most intact temperate rangelands on earth (Asner et al. 2004), it has recently experienced strong socio-economic and climate change. This suggests that this is a crucial time for Mongolia to have more spatially extensive, long-term studies to understand the interacting effects of climate and grazing on rangelands. Especially important is to understand how changes in rangeland governance and management have affected the resilience and resistance of the Mongolian rangelands. With this in mind, this dissertation attempts to contribute the following to our knowledge of Mongolian rangelands. Chapter 2 describes our observational study to test the impact of livestock grazing by sampling at different grazing intensities around the pastures of winter shelters across four ecological zones. This study gives a broad-scale assessment of the impacts of winter grazing on vegetation on different soils or ecological sites. Chapter 3 describes a 14-year study of changes in winter grazing intensity and its effects on vegetation, using satellite-derived vegetation data. In Chapter 4, the aim is to evaluate the ecological outcomes of community-based rangeland management (CBRM) in Mongolia and if and how these outcomes differ by ecological zone. Chapter 5 concludes with the main findings from this study and proposes management implications and future research.

Below are the theoretical and practical questions and research hypotheses that motivate each chapter. The results chapters of this dissertation (Chapters 2-4) are written in manuscript form in which each chapter includes an introduction, methods, results and discussion. A manuscript format has advantages in that each chapter stands alone and can therefore be read and understood in the absence of the other chapters.

1.4.1. Research Questions and Hypotheses

Chapter 2. Impacts of livestock winter grazing across ecological zones of Mongolian rangelands

The aim of this study was to complete the first very broad-scale field study of the effects of livestock on winter pasture vegetation, forage quality, and soils in Mongolia, comparing vegetation dynamics among the different ecological zones: desert steppe, steppe, mountain and forest steppe and the eastern steppe. We chose winter pastures as the focus of study since they are not grazed in the growing season, thus providing the equivalent of a ‘utilization cage’ for current season grazing.

Research Question 2.1:

Do livestock create grazing gradients around winter shelters in Mongolia and how long are those gradients? Do these winter grazing gradients differ for different livestock species or by ecological zones?

Research Question 2.2:

What are the long-term effects of winter grazing on vegetation, forage quality and soil surface indicators in the rangelands of Mongolia?

Research Question 2.3:

Do effects of winter grazing differ in different ecological zones? What are the implications of these differences for equilibrium vs non-equilibrium rangeland dynamics?

Research Question 2.4:

Within ecological zones, does livestock winter grazing have different effects on vegetation, forage quality and soils on different ecological sites?

Research Hypothesis 2.1:

Based on theory and past research findings, we predict that in the wetter pastures of the mountain and forest and eastern steppe, heavily grazed pastures close to winter shelter will have lower total and grass cover/biomass and greater forb and weedy annual cover/biomass than the lightly grazed pastures farther away from winter shelters. There will not be grazing-induced total and functional group cover/biomass changes along the grazing gradients in the pastures of dry desert steppe zone, but there will be some changes in total, grass, forb and weedy annual cover/biomass along the grazing gradients in the steppe ecological zone. Forage quality and palatability will increase as grazing intensity declines with increasing distance from winter shelters.

Research Hypothesis 2.2:

The size of gaps between perennial plants will decrease as grazing intensity declines and the connectivity between these plants will increase with distance from winter shelters. There will be little erosion and deposition by wind or water in plots farther away from winter shelters.

Research Hypothesis 2.3:

Grazing will have more effect on vegetation and soils in loamy and clay ecological sites than on vegetation in sandy and rocky ecological sites.

Research Hypothesis 2.4:

All the above effects will be greatest in the mountain and forest steppe and least in the desert steppe and mixed in the steppe and eastern steppe, following the predictions of theory of equilibrium and non-equilibrium rangeland dynamics.

Chapter 3. Mongolian rangeland changes and resilience to livestock winter grazing over time

The overarching goal of Chapter 3 is to understand how Mongolian rangelands grazed at different intensities have changed recently, how they respond to stress, and how to best measure this change across a range of different ecological regions, from the deserts in the south of the country to the forests and mountains in the north. We do this by assessing change and response to inter-annual changes in climate and grazing along a grazing gradient from intensively grazed pastures near herder winter shelters to lightly grazed pastures far from these same shelters.

Research Question 3.1:

As seen in chapter 2, we carefully selected small-scale field plots along winter grazing gradients, controlling for soils and landform (or ecological sites). If we overlay coarser resolution MODIS remote sensing data for the same season on these plots, do they show the same patterns of vegetative response to grazing? (How strong is the relationship between field measures of vegetation cover and biomass and remote sensing measures of NDVI?)

Research Question 3.2:

Are the winter grazing gradients we found in Chapter 2 (based on field sampling in 2011 and 2012) maintained over time or are they present only in certain years between 2000 and 2013?

Research Question 3.3:

How do the trends in vegetative greenness (NDVI) compare in winter pastures grazed at different intensities over time?

Research Question 3.4:

Do these winter grazing effects differ over time by ecological zone? Is precipitation or grazing a better predictor of NDVI over time?

Research Question 3.5:

How resilient are these rangelands to inter-annual changes in climate and livestock winter grazing? Does vegetation greenness, as measured by NDVI, in heavily grazed pastures near winter shelters recover more slowly after *dzud*/drought than vegetation in lightly used pastures far from winter shelters?

Research Hypothesis 3.1:

The NDVI data along the grazing gradient will show the same effects of grazing as the field vegetation data sampled in the same season and in the same locations.

Research Hypothesis 3.2:

The winter grazing gradient will be maintained over time and will become stronger in years when livestock densities (SFU/km²) in surrounding pastures are higher, and weaker when livestock densities in the surrounding pastures are lower.

Research Hypothesis 3.3:

The winter grazing gradient will be strongest in the mountain and forest steppe, moderate in the steppe and weakest in the desert steppe following the predictions of the theory of equilibrium and non-equilibrium rangeland dynamics.

Research Hypothesis 3.4:

More heavily grazed pastures near winter shelters will recover more slowly after *dzud*/drought than more lightly grazed vegetation far from winter shelters.

Research Hypothesis 3.5:

Winter grazing will have strongest effects on NDVI in the wetter compared to drier zones, and climate will have the opposite pattern. This means that grazing will have its strongest effects on NDVI in the mountain and forest steppe, moderate effect in the steppe and weakest effects in the desert steppe.

Chapter 4. Ecological outcomes of community-based rangeland management in Mongolia

In chapter 4, we sought to conduct the first country-wide assessment of the effects of CBRM groups on rangeland vegetation and soils in Mongolia across four ecological zones: the mountain and forest steppe, eastern steppe, steppe and desert steppe.

Research Question 4.1:

Are pastures managed by formal community-based rangeland management (CBRM) groups in better condition than pastures managed by traditional neighborhoods or non-CBRM groups?

Research Question 4.2:

Does CBRM management have different effects on vegetation, forage quality and soils in different ecological sites within ecological zones?

Research Question 4.3:

Do effects of CBRM management differ in different ecological zones?

Research Hypothesis 4.1:

Pastures managed by CBRM groups will have higher cover, biomass, and species richness than the pastures managed by the non-CBRM, traditional neighborhood groups.

Research Hypothesis 4.2:

Forage quality and palatability will also higher in CBRM-managed pastures than non-CBRM-managed pastures.

Research Hypothesis 4.3:

The size of gaps between perennial plants in pastures managed by CBRM groups will be smaller than in pastures managed by non-CBRM groups. There will be little erosion and deposition by wind or water in CBRM-managed pastures.

Research Hypothesis 4.4:

All the above effects will be greatest in the mountain and forest steppe and smallest in the desert steppe and mixed in the steppe and eastern steppe, following the predictions of equilibrium and non-equilibrium rangeland dynamics.

Chapter 5: Conclusions

This dissertation ends with a brief conclusions chapter, bringing together all of the results in the three middle chapters, plus a discussion of the practical implications of this work.

CHAPTER TWO: IMPACTS OF LIVESTOCK WINTER GRAZING ACROSS ECOLOGICAL ZONES OF MONGOLIAN RANGELANDS.

2.1. Introduction

There is a global debate about the status and causes of rangeland change. This is true in Mongolia, where it is commonly stated that more than 70% of Mongolian rangelands show negative change or degradation (UNEP 2002). This figure is often stated in the non-peer reviewed academic publications and donor reports (Batjargal 1997, UNEP 2002, Erdenetuya 2006). Recent studies suggest that both livestock and climate change have strong effects on Mongolian grasslands at both broad and local scales (Hilker et al. 2013, Liu et al. 2013, Khishigbayar et al. 2015). Recent research suggests that livestock grazing and climate are causing widespread degradation (Hilker et al. 2013, Liu et al. 2013), but the extent and relative impact of these causes are unclear (Sankey et al. 2009, Wesche et al. 2010, Addison et al. 2012).

The field studies in Mongolia have focused on vegetation responses to a wide range of environmental gradients and the relative importance of abiotic and biotic factors on vegetation. In Mongolia, like other rangelands around the world, the amount and variability of precipitation strongly influences the effects of livestock grazing on rangeland health (Fernandez-Gimenez and Allen-Diaz 1999, Stumpp et al. 2005, Sasaki et al. 2009, Wesche et al. 2010). In arid rangelands with non-equilibrium dynamics, rainfall and its variability are thought to have an overriding impact, much more than livestock grazing (Ellis and Swift 1988, Briske et al. 2003, Vetter 2005). In Mongolia, these rangelands occur in the desert steppe of the southern Gobi region (Fernandez-Gimenez and Allen-Diaz 1999, von Wehrden et al. 2012). In contrast, in wetter rangelands with

more predictable rainfall and equilibrium dynamics, grazing can have strong impacts on vegetation (Fernandez-Gimenez and Allen-Diaz 1999, von Wehrden et al. 2012).

Empirical studies show conflicting results in relation to these theories in Mongolia. In wetter rangelands, like mountain and forest steppe, rangelands appear to be driven by grazing as predicted (Fernandez-Gimenez and Allen-Diaz 1999, Van Staaldunen et al. 2007). However, the results from arid rangelands of desert steppe are less clear. Some studies show little response of these rangelands to grazing (Fernandez-Gimenez and Allen-Diaz 1999, Stumpp et al. 2005, Zemmrich 2007), but others show that pastures with high grazing intensity close to water points and herders' settlements are affected by grazing (Sasaki et al. 2013). Sasaki and colleagues show that vegetation recovery did not occur even after following 5 years of removal of grazing and the authors concluded that this indicated vegetation had crossed a critical threshold. These differing findings require reexamination of the design, methodology and spatio-temporal scale of studies to understand how complex rangeland systems in Mongolia respond to grazing and climatic variability.

At local scales, livestock have effects on vegetation through grazing and trampling and dung/urine deposition (Schlesinger et al. 1990, Moleele and Perkins 1998, Fernandez-Gimenez and Allen-Diaz 2001). Thus, in this chapter, grazing includes livestock trampling and urine and feces defecation besides actual grazing. Heavy grazing removes plant biomass and thus can cause a decrease in plant biomass and cover an increase in bare ground (Dyksterhuis 1949, Fernandez-Gimenez and Allen-Diaz 1999, Wesche et al. 2010). Heavy grazing can cause change in composition of functional groups, species and palatability of vegetation (Diaz et al. 2007). If livestock herds are dominated by grazers (like cattle, yak, horses, sheep) rather than browsers (like camels, goats), heavy grazing can cause a shift in functional groups from herbaceous to

woody plants (Schlesinger et al. 1990, Moleele and Perkins 1998). Conversely, heavy browsing can remove plants, giving a competitive advantage to herbaceous plants. Sometimes grazing causes a decline in shrubs, like in Mongolia's dry desert steppe (Sasaki et al. 2013). Grazing can also cause the replacement of palatable plants by unpalatable species (Ellison 1960) due to herbivore preference for palatable species, which alters the competitive balance in favor of unpalatable plants. With grazing, the relative abundance of tolerant species may increase, often called 'increasers' (Anderson and Briske 1995), and sensitive species decrease, often called 'decreaser'. At the species level, plants with high reproductive rates like annual forbs may also increase (Fernandez-Gimenez and Allen-Diaz 1999, Diaz et al. 2007). For example, the replacement of perennials by annual along grazing gradients has been found in several studies (Hilbig 1995, Sasaki et al. 2008a, Hoshino et al. 2009). Trampling and grazing can encourage the spread of weedy or ruderal species (Grime 1977).

It is important to control for the confounding effects of soils, landforms and other landscape characteristics when designing studies on the effects of grazing. Range scientists use the concept of ecological site to achieve this control, by sampling in places where edaphic and landscape characteristics are similar. The current definition of ecological site (Brown 2010) is groupings of a landscape based on soil properties and processes within a climate zone to predict the dynamics of vegetation and related resources. Soil properties include soil texture or soil depth. Soil processes are a series of actions in the soil, for example, water percolating into the soil that determines soil water (Duniway et al. 2010). These groupings allow understanding of the inherent ecological potential of the site to support distinct plant communities across landscape (Brown 2010). Knowledge of ecological sites is crucial for appropriate management decisions and scenarios of the future effects of grazing (Bestelmeyer and Brown 2010).

The aim of this study was to complete the first very broad-scale field study of the effects of livestock on winter pasture vegetation, forage quality, and soils in Mongolia, comparing vegetation dynamics among the different ecological zones: desert steppe, steppe, mountain and forest steppe and the eastern steppe. Our goal was to measure the long-term effects of grazing, so we focused our work on winter pastures. The main reasons for choosing winter pastures were that: 1) winter shelters create strong gradients without (or with) less chance of confounding gradients in ground water table levels than gradients from water points, 2) these pastures are mostly ungrazed in summer (Banzragch and Davaajamts 1970), and thus standing crop biomass at the start of winter is a reasonable estimate of total biomass with little offtake, and 3) species identification was more reliable in the absence of grazing. We also wanted to measure grazing intensity in our design, so our study uses ‘grazing gradients’ as a space-time substitution for grazing intensity around winter livestock shelters (and a few water points) in this winter pastures. In addition, we used the concept of ecological site carefully select our sampling sites along these grazing gradients.

Research questions

1. Do livestock create grazing gradients around winter shelters in Mongolia and how long are those gradients? Do these grazing gradients differ for different livestock species or by ecological zones?
2. What are the long-term effects of grazing on vegetation, forage quality and soils in the rangelands of Mongolia? What kinds and levels of effects will indicate “degradation”?
3. Do effects of grazing differ in different ecological zones? What are the implications of these differences for equilibrium vs non-equilibrium rangeland dynamics?

4. Within ecological zones, does livestock grazing have different effects on vegetation, forage quality and soils on different ecological sites?

Research hypotheses

1. Based on theory and past research findings we predict that in the wetter pastures of the mountain and forest and eastern steppe, heavy grazed pastures close to winter shelter will have lower total and grass cover/biomass and greater forb and weedy annuals cover/biomass than the lightly grazed pastures farther away from winter shelters. There will not be grazing induced total and functional group cover/biomass changes along the grazing gradients in the pastures of dry desert steppe zone, but there will be some changes in total and grass, forb and weedy annual cover/biomass along the grazing gradients in the steppe ecological zone. Forage quality and palatability will increase as grazing intensity declines with increasing distance from winter shelters.
2. The size of gaps between perennial plants will decrease increase as grazing intensity declines and the connectivity between these plants will increase with distance from winter shelters. There will be little erosion and deposition by wind or water in plots farther away from winter shelters.
3. Grazing will have more effect on vegetation and soils in loamy and clay ecological sites than on vegetation in sandy and rocky ecological sites.
4. All above effects will be greatest in the mountain and forest steppe and smallest in the desert steppe and mixed in the steppe and eastern steppe, following the predictions of the theory of equilibrium and non-equilibrium rangeland dynamics.

2.2. Methods

The central continental position of Mongolia, far from oceanic influences and mountainous terrain, shapes its climate. Mountain barriers in the north and western part of Mongolia intercept atmospheric flows carrying moisture from the Atlantic side, while Mongolia is virtually open to the dry Central Asian desert winds from the South. Mongolia's winters are cold and dry, summers are warm and wet. Precipitation is distributed unimodally throughout the country with peak rainfall occurring in second half of the summer (Gunin et al. 1999).

Ecologically, the country is divided into six ecoregions from north to south including high mountains, taiga, forest steppe, steppe, desert steppe, and desert (Hilbig 1995). This study focused on only four of the above mentioned ecological ecoregions or zones including mountain and forest steppe, eastern steppe, steppe and desert steppe. Mean annual temperature and precipitation is -2.2°C and 239 mm in the mountain and forest steppe region, 0.2°C and 258 mm in the eastern steppe region, -0.09°C and 170 mm in the steppe region and 2.6°C and 131 mm in the desert steppe (Hilbig 1995, Hijmans et al. 2005, Chen et al. 2008).

We sampled vegetation and soils in the mountain and forest steppe, steppe, eastern steppe and desert steppe, which can be described by their location along a continuum of ecosystems from equilibrium to non-equilibrium according to the amount and variability of precipitation across this gradient (see Figure 2.1, a map of zones, *soums*, and winter shelters sampled). Ellis and Togtokhyn (1993) predicted that non-equilibrium ecosystems in Mongolia exist where mean annual precipitation is less than 250 mm and the coefficient of variation of inter-annual precipitation is more than 33%. Coefficients of variation for precipitation range from 23% in mountain and forest steppe, 32% in steppe, 30% in eastern steppe and 34% in desert steppe region (von Wehrden et al. 2012), thus we classify only the desert steppe as a non-equilibrium

system and the other three zones as equilibrium systems. More than 80% of precipitation falls between May and September in all zones. Summer is short and hot, with the hottest month in July when mean average temperature range between 18 and 26⁰C. Winter is cold and dry with the coldest month in January, when mean average temperature ranges from -35⁰C in the mountainous regions and -10⁰C in the desert steppe. All four ecological zones are dominated by perennial grasses (Gunin et al. 1999) (See Table 2.1 for a summary of site characteristics).

All study areas have been grazed by domestic livestock under a nomadic and transhumance pastoral use for at least 1000 years and possibly for several millennia (Johnson et al. 2006). The main types of livestock are cow/yaks, horse, sheep and goats in the mountain and forest steppe; horses, sheep, cow/yaks and goats a camels in the eastern steppe; sheep, goats, horses, cow/yaks and camels in the steppe; and goats, sheep, camels, horses and cow/yaks in the desert steppe. The basic pattern of livestock use is seasonal across all four ecological zones, with herder families spending summers in the vicinity of rivers, lakes, and water wells, moving to other camps in the fall. Families and their livestock spend winter in sheltered places, facing south, usually locating their winter shelters on the warmer, leeward side of mountains or hills. In the spring, families move to lower more open areas. Traditionally, herders graze different pastures in the four seasons, particularly avoiding summer grazing in winter grazing grounds to preserve critical pastures for winter grazing. These ‘preserved’ winter grazing lands cover about third to half of the annual grazing orbit (ALAGAC 2010).

2.2.1. Study design

Our larger objective for this research was to measure the effects of community-based rangeland management of Mongolian grasslands (See Chapter 4). Thus, our winter shelters were

located in pairs of counties (*soums*) with and without formal community-based rangeland management (CBRM) groups. In this chapter, we focus our analysis on each winter shelter and its surrounding grazing gradient, and thus we will not distinguish between areas with and without these groups in this chapter.

We sampled in a total 36 *soums* (counties) of Mongolia in a total of 143 winter shelters (we did sampling at a very few water points instead of winter shelters) or sites. Sampling was completed in July and August of 2011 and 2012 (Figure 2.1).

We chose to sample shelters in winter rather than summer pastures because winter pastures provided us a natural ‘utilization cage’ from current season or summer grazing. Most herders avoid grazing livestock in winter pastures around any winter shelters (their or their neighbors) during the summer time, although some non-winter grazing sometimes occurs. This allowed us to measure the effects of long-term rather than short-term seasonal grazing on rangelands, and saved the time and resources that would have been required to build exclosures in summer pastures. Because it is a winter pasture thus dormant season grazing occur and in sometimes early spring grazing could occur on winter pastures as well.

We then measured vegetation, forage quality, and soils along a ‘grazing gradient’ or ‘piosphere’ around each winter shelter. Our direct measure of grazing in dung density, thus we did not measure grazing directly, and thus our ‘grazing’ gradients are really ‘livestock use’ gradients. However, for the purposes of this dissertation, I will use the term ‘grazing gradient’ because of its common usage. Livestock density and grazing pressure are usually highest close to livestock concentration areas, like water points, salt licks and corrals, and decrease with distance away from them (Valentine 1947). Grazing gradients have been used to measure the effects of grazing around the world (Andrew 1988) as well as in Mongolia (Fernandez-Gimenez

and Allen-Diaz 2001, Sasaki et al. 2008a). We also measured the grazing gradient itself through the frequency and density of dung of different livestock species, which has rarely been done in other studies in Mongolia.

Table 2.1 Site characteristics of the mountain and forest steppe, eastern steppe, steppe and desert steppe. Note that elevation range and dominant plant species listed in this table are based on the data we collected around winter shelters in these zones.

	Mountain and forest steppe	Eastern steppe	Steppe	Desert steppe
Mean annual precipitation (mm)*	239	258	170	131
Coefficient of variation of precipitation**	23	30	32	34
Mean warmest temperature†	24.6	18.7	23.7	23.0
Mean coldest temperature‡	-25.7	-27.9	-25.8	-24.9
Elevation range (m)	860-2700	600-970	1150-2500	900-1960
Dominant plant species	Grasses: <i>Agropyron cristatum</i> , <i>Cleistogenes squarrosa</i> , <i>Elymus chinensis</i> , <i>Stipa krylovii</i> , <i>Festuca lenensis</i> , <i>Koeleria macrantha</i> , <i>Poa attenuate</i> Forbs: <i>Allium bidentatum</i> , <i>Allium senescens</i> , <i>Arenaria capillais</i> , <i>Artemisia dracunculus</i> , <i>Artemisia glauca</i> , <i>Asplenium altajense</i> , <i>Bupleurum bicaule</i> , <i>Cymbaria</i>	Grasses: <i>Agropyron cristatum</i> , <i>Bromus pumpellianus</i> , <i>Cleistogenes squarrosa</i> , <i>Elymus chinensis</i> , <i>Stipa krylovii</i> Forbs: <i>Allium bidentatum</i> , <i>Allium odorum</i> L., <i>Bupleurum bicaule</i> , <i>Fillifolium sibiricum</i> , <i>Leuzea uniflora</i> , <i>Stellaria dahurica</i> , <i>Medicago ruthenica</i>	Grasses: <i>Achnatherum splendens</i> , <i>Agropyron cristatum</i> , <i>Cleistogenes squarrosa</i> , <i>Elymus chinensis</i> , <i>Stipa krylovii</i> , <i>Stipa gobica</i> Forbs: <i>Allium anisopodium</i> , <i>Allium mongolicum</i> , <i>A.polyrrhizum</i> , <i>Convolvulus ammannii</i> , <i>Chenopodium album</i> , <i>Salsola</i>	Grasses: <i>Cleistogenes songorica</i> , <i>Stipa gobica</i> , Forbs: <i>Artemisia pectinata</i> , <i>Salsola collina</i> , <i>Allium polyrrhizum</i> , <i>A. mongolicum</i> , <i>Convolvulus ammannii</i> , <i>Scorzonera</i>

	<i>dahurica</i> , <i>Galium</i> <i>verum</i> , <i>Iris tigrida</i> , <i>Polygonum</i> <i>agustifolium</i> , <i>Potentilla bifurca</i> , <i>Pulsatilla</i> <i>Turczaninovii</i> , <i>Thalictrum simples</i> Subshrubs and Shrubs: <i>Artemisia adamsii</i> , <i>Artemisia</i> <i>commutate</i> , <i>Artemisia frigida</i> , <i>Artemisia laciniata</i> , <i>Dasiphora</i> <i>fruticosa</i>	Subshrubs and Shrubs: <i>Artemisia frigida</i>	Subshrubs and Shrubs: <i>Artemisia</i> <i>adamsii</i> , <i>Artemisia frigida</i> , <i>Caragana</i> <i>leucoploea</i> , <i>Caragana</i> <i>microphylla</i> , <i>Caragana</i> <i>stenophylla</i> , <i>Kochia prostrate</i>	<i>preudodivarcata</i> , Subshrubs and Shrubs: <i>Ajania</i> <i>achileoides</i> , <i>Anabasis</i> <i>brevifolia</i> , <i>Artemisia</i> <i>xerophytica</i> , <i>Asparagus</i> <i>gobicus</i> , <i>Caragana</i> <i>leucoploea</i> , <i>Caragana</i> <i>stenophylla</i>
Annual standing crop yield (kg/ha)***	100-400	200-700	200-700	320-370

* - Source: (Chen et al. 2008); ** - Source: (von Wehrden et al. 2012)

† , ‡ - Source: (Fan and Van den 2008); *** Source: (Jigjidsuren and Johnson 2003)

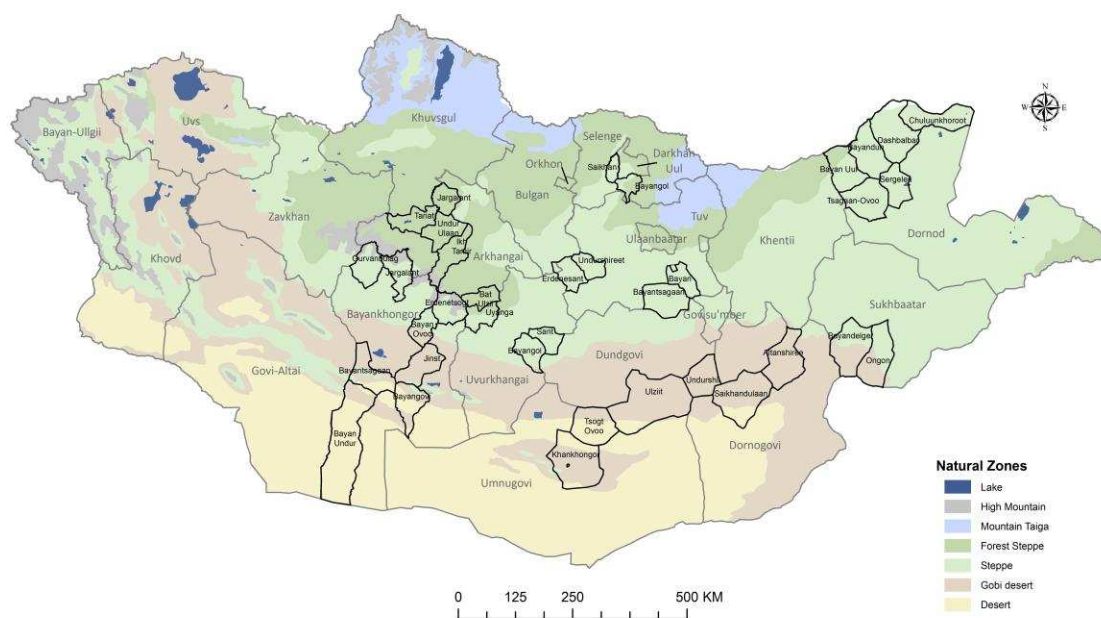


Figure 2.1. Study sites in the four ecological zones of Mongolia

2.2.2. Sampling methods

At each winter shelter, we sampled vegetation and soils in 3 plots located at 3 distances from the winter livestock shelter (100 m, 500 m and 1000 m), as measured from the gate of the livestock corral at the shelter, to examine the effects of livestock grazing pressure on vegetation. We selected these specific distances because of previous work showing that the effects of livestock grazing are greatest close to livestock grazing impact points such as livestock camps or water points and largely minimal farther than 1000m from the impact (Sasaki et al. 2008a, Sasaki et al. 2011). We selected plots at the three distances on about the same landform (e.g. hill, fan/piedmont, terrace, or plain), hillslope profile position (summit, shoulder, backslope, footslope, toeslope etc.), aspect, and, if possible, soils. This allowed us to sample a grazing gradient that falls on the same ecological site, which is defined by landform, soils, and climate, which potentially produces similar kinds and amounts of vegetation and responds similarly to natural disturbances, drivers and management (Bestelmeyer et al. 2009). Because winter shelters are often located at the head of a valley, we selected sites along a gradient following a contour along the edge of the valley rather than dipping downslope onto different soil at the valley bottom. This meant that most sites were in specific upland positions, not in riparian areas deep in valley bottoms, unless the winter shelter was located in this landscape position. Thus, our sample is not a random sample of grazing effects in winter grazing areas, but rather the specific effects of grazing on the types of ecological sites selected by herders for their winter shelters. In this way, we avoided confounding distance from the winter shelter or ‘impact point’, intended as an index of grazing intensity, with a gradient in soil moisture or water table depth, and associated changes from riparian to upland vegetation. We selected the plots, with only a few exceptions, so that they were located at least as far away from any *other* livestock camps and water points, as

they were from the focal impact point (winter shelter or water point). In other words, we made sure that the 1000-m plot was at least 1000-m away from any other livestock camps or water point as well as from the selected campsite. We did not use the appearance of the vegetation to select plots that are on similar ecological sites, only landform and soils, to limit confounding, since grazing can impact vegetation. An almost equal number of replicate plots were selected in each distance from winter shelters in each ecological zone. At our total of 143 winter shelters/water points, we sampled 117 plots in desert steppe, 122 plots in steppe, 33 plots in eastern steppe and 156 plots in mountain and forest steppe ecological zones. At one site we skipped sampling a plot at 100 m distance because there was no location at the distance that matched the ecological sites of the other distances at that site.

Each plot consisted of a 50 x 50 m plot with 5 systematically spaced 50-m transects (Figure 2.2). Transects originated at the 0 point, 12.5, 25, 37.5 and 50 meters along the baseline. If the plot was on a slope, we oriented it so that transects ran up the slope, to incorporate variability within each transect. The origin point (at 0 m) and baseline were always on the downhill side of the plot with transects running uphill from the baseline.

We obtained livestock numbers at the *soum* (county) level from National Statistical Office of Mongolia (2013), whereas livestock numbers at the winter shelter level was provided from the social team survey of the Mongolian rangeland resilience project (MOR2). When reporting all the species together, we converted different species into the same relative measure of sheep forage units (SFU's), where one camel=5 SFU, one horse=7 SFU, once cow/yak=6 SFU, one goat=0.9SFU and one sheep=1 SFU (NSO 2013).

We quantified livestock use in our plots as a direct measure of grazing along the gradient away from the winter shelters in 3 of 4 ecological zones (not the eastern steppe). We made these

measurements in a subset of our plots by sampling 27 plots in the desert steppe, 156 plots in the mountain and forest steppe, and 92 plots in the steppe. At each plot, we recorded the frequency and density of sheep/goat pellets and frequency pellets of horse and camel dung and the frequency of piles of cow/yak. We used a 50 x 50 cm quadrat for counting presence and absence of sheep/goat pellets, a 1x1m quadrat frame for counting presence and absence of cow/yak, horse and camel dung pellets or piles. We used a smaller quadrat for sheep and goat pellets because these pellets were so evenly spread that the smaller quadrat size allowed us to capture some quadrats with no dung (the larger quadrat often registered 100% presence for all our plots). We placed the frame to the right of the transect tape every 5 meters at the 0, 5, 10, 15, 20, 25, 30, 35, 40 and 45 meters along each of the 5 transects. In total, presence/absence or density of dung was recorded in 50 quadrat frames in each plot

Standing crop biomass, if ungrazed, is an estimate of vegetation production at the site, which is an important indicator of the health of the site and its value for livestock production. We separated out the biomass samples by seven functional groups, including grasses, forbs, shrubs, sedges, litter, standing dead, and the large grass, *Acnatherum splendens*. Standing crop biomass of herbaceous plants was clipped in 5 quadrats in each plot at the base of the plant for grasses, forbs, sedges and *A. splendens*. We separated out *A. splendens* because it was very patchy and when present, usually of very high biomass. For shrubs and subshrubs, we used the representative branch method and or collected shrub leaves and current year's growth of twigs within at 3-D projection of the plot frame, regardless of whether the shrub was rooted inside or outside the frame (Bonham and Ahmed 1989). For litter and standing dead, we clipped or picked up all detached pieces inside the quadrat frame. We determined the size of the quadrat by the amount of biomass in the ecological zone, using a 50 x 50 cm quadrat in the mountain and forest

steppe, eastern steppe and steppe and 1 x 1 m quadrat in the desert steppe. All samples were dried in a drying oven at 60°C for 48 hours in the laboratory and then weighed to an accuracy of +/- 0.01 grams.

To measure forage quality, crude protein and acid detergent fiber (ADF) analyses were carried out on a subset of the functional group samples (ADF analysis sample size: mountain and forest steppe=41, eastern steppe=12, steppe=33, desert steppe=38; Crude protein analysis sample size: mountain and forest steppe=145, eastern steppe=24, steppe=119, desert steppe=111) at the Feed Evaluation Laboratory of the Research Institute of Animal Husbandry in Mongolia. The ANCOM technology was used for acid detergent fiber analysis, whereas the Kjel-Foss automated macro-Kjeldahl method was used for crude protein analysis. All 5 samples of functional groups from each of the 5 biomass quadrats sampled in each plot were mixed and ground before analysis. Both crude protein and ADF analysis were run in duplicate. If there was a large difference between the duplicates then the analysis was repeated until repeat measures were nearly identical.

Plant foliar and basal cover by species were measured using the line point intercept (LPI) method with points dropped every meter along each of the five 50-m transects for a total of 250 points per plot (Herrick et al. 2009). Foliar cover was measured as the area of ground covered by vegetation leaves. Small openings in the canopy and intraspecific overlap were excluded and thus foliar cover is always less than canopy cover, since the later sums up the overlap in different layers of the canopy. All nomenclature in this study follows Grubov (1982).

Species richness data were collected by searching for all species within the entire 50x50 m plot. This was done by walking zig zag through the plot and recording all species observed. Each species was scored on the datasheet according to their functional or life form group

(perennial grass, annual grass, perennial forb, annual forb, perennial sedge, shrub and sub-shrub). Any additional species found during LPI measurements but not in the species search were added to the total species list for the plot.

The gap between perennial plants bases was measured along transects 2 and 4 using the basal gap intercept method. We only recorded gap sizes that were larger than 20 cm between perennial plant bases to capture the larger gaps efficiently. A gap was defined as the distance between perennial plant bases with a minimum base size of a single perennial plant stem (1 mm). In the desert steppe (Gobi), crowns of apparently dead plants (e.g. *Stipa*, *Allium*) that were buried under the soil were counted a perennial plant base. These were detected by running one's fingers along the soil at the edge of the transect.

In the analysis, we tested the effects of grazing on the palatability of forage species. We use the classification of Damiran (2005) for the palatability of species in the dormant season (winter) when livestock are present at the winter shelter. The palatability classes include preferred, desired, consumed but undesirable, not consumable and toxic.

To understand the effect of grazing on soil surface conditions and plant patterns, we recorded soil resource retention and soil redistribution classes (Burkett et al. 2012). The resource retention class, which describes the spatial patterning and connectivity of persistent vascular plant patches and inter-patches across plot, was recorded for the whole plot from most to least connected in the following classes: **1**-Interconnected persistent plant cover or dense bunchgrasses and surrounding round interpatch areas < 30cm, **2**-Persistent plants interconnected and surrounding round/oval interpatch areas > 30cm, **3**-Persistent plant patches fragmented by elongated interpatch areas that are bounded in the plot, **4**-Persistent plant patches fragmented by elongated interpatch areas that cross through the plot, **5**-Interpatch areas interconnected and

crossing the plot in several directions; isolated plant patches, **6**-Interpatch areas interconnected; scattered or no persistent plants.

Lastly, soil redistribution class, which describes the extent and severity of erosion and deposition on a plot, was recorded from least to most redistribution in these classes: **0**-No evidence of erosion deposition, **1**-Very slight soil redistribution, **2**-Patchy, slight (< 5 cm) soil loss and deposition, **3a**-Extensive, moderate soil loss (< 10 cm), **3b**-Extensive, moderate soil redistribution (< 10 cm), **4a**-Extensive, severe erosion (> 10 cm); little deposition, **4b**-Extensive, severe erosion (> 10 cm) with patchy sediment deposition, **4c**-Extensive, severe sediment deposition (> 10 cm). Six ordinal class values of soil redistribution were used and higher class values represented greater extent and severity of soil redistribution and its visual appearance around the base of plants.

After sampling, we developed an ecological site key for all our plots and classified each plot to an ecological site. We then used ecological site as a variable in the analysis to test the effects of site type on soils and vegetation, and to uncover any interactions between grazing and ecological site.

2.2.3. Data analysis

Data were analyzed with the statistical package SAS 9.3 for Windows. We corrected the non-normality of the data using a $\log(y+1)$ transformation on biomass and arcsine transformation on cover data, sheep/goat pellets and gap data. When these transformations did not achieve normality, we ranked the data and analyzed the ranks. Some of the cover data contained large numbers of zeros; here we transformed the original data into binary codes (=presence/absence) when more than half the values were zero. Once the transformations were complete, we used a

model type III ANOVA to assess the effects of ecological zone, ecological site, and distance from winter shelter on standing crop biomass, forage quality, foliar and basal cover, species richness, plant palatability, cover of dominant species and open gaps at the plots. Because the effects of ecological zone were so large, we ran all subsequent analyses separated by ecological zone including desert steppe, steppe, eastern steppe, and mountain and forest steppe. We report least squares means of untransformed data when transformation was not necessary and if test results of both transformed and untransformed data were the same. If test results between untransformed and ranked and binary data were different we used the untransformed results, but we still reported the least square means to correct for the main effects. Two-way ANOVAs followed by Tukey-adjusted multiple tests were used for multiple comparisons of vegetation variables among the grazing distances and ecological sites. We used a Chi-square to test for differences in the distributions of soil surface characteristics across three grazing distances and among the ecological sites. When there were few plots in certain categories of the resource retention and soil redistribution variables, then we re-grouped them into similar classes to achieve sufficient sample sizes.

We report p-value of ≤ 0.05 as ‘significant’, to avoid both type I and II errors, which are both significant in our analysis. On the one hand, we do not want to over-report significance (Type II error) to ensure reliability; on the other hand, we also do not want to under-report significance (Type I error) in order to avoid artificially reducing the significance of this research.

2.3. Results

2.3.1. Livestock densities and grazing gradients from the winter shelters

At the *soum* level, total livestock densities by sheep forage unit (SFU/ha) were 3 times higher in the mountain and forest steppe than in the eastern steppe and the steppe, and 5 times that of the desert steppe.

Table 2.2 Average livestock number by each livestock species type in four ecological zones at the *soum* level (SFU=livestock number in sheep forage unit)

Ecological zone	N of <i>soums</i>	Average livestock density (average number/ha) in SFU					
		Sheep	Goat	Cow/Yak	Horse	Camel	Average total
Mountain and forest steppe	11	0.27	0.18	0.47	0.28	0.00	1.2
Eastern steppe	6	0.09	0.04	0.09	0.13	0.00	0.36
Steppe	9	0.15	0.12	0.06	0.13	0.01	0.47
Desert steppe	10	0.06	0.07	0.03	0.05	0.02	0.22

We also separated out sheep and goats, since we found that these species were responsible for the grazing gradient (see below). Sheep/goat densities were four times higher in the mountain and forest steppe than the eastern steppe and desert steppe, and double that in the steppe (Figure 2.2).

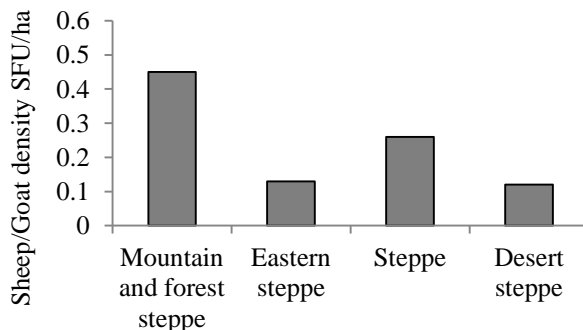


Figure 2.2. Sheep/goat density per hectare by SFU at the *soum* level in four ecological zones

At the winter shelters that we sampled, the composition of herds differed by ecological zone. Cows and yaks dominated herds in the mountain and forest steppe, horses dominated in the

eastern steppe, sheep dominated in the steppe and goats dominated in the desert steppe (Table 2.2). Overall, sheep and goats together dominated herds in most zones, where proportion of sheep and goats in the total herd was 40-60% for households. Exception was the eastern steppe, where horses dominated herds and sheep and goats made up less than 30% of the herds (Figure 2.3).

Table 2.3. Estimated* average livestock number (in SFU) by each livestock species at winter shelters sampled in four ecological zones.

Ecological zones	Average livestock number in SFU					
	Sheep	Goat	Cow/Yak	Horse	Camel	Total
Mountain and Forest steppe	385	236	720	470	0	1811
Eastern steppe	1427	322	652	3422	163	5986
Steppe	481	237	148	319	26	1211
Desert steppe	233	232	89	209	107	870

*We estimated average livestock numbers by multiplying livestock number we recorded from the winter shelter owner by the number of households that shared the same winter shelter

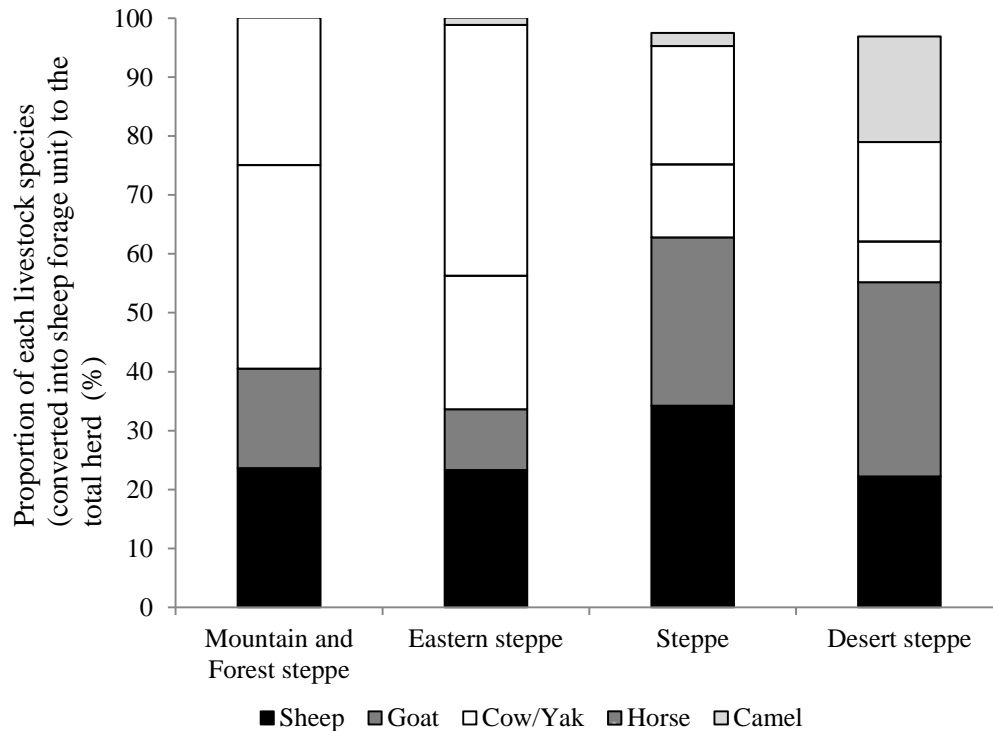
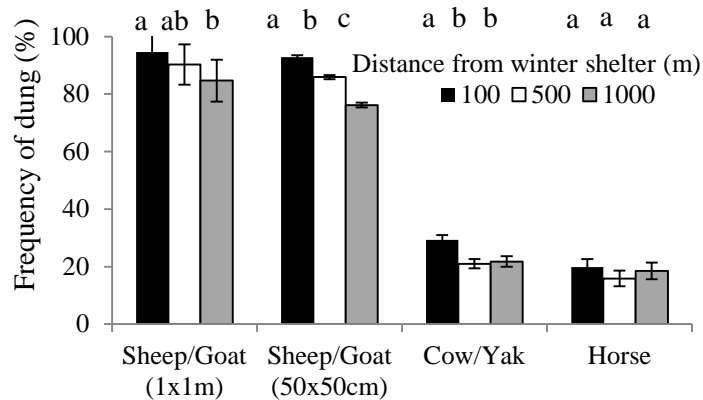


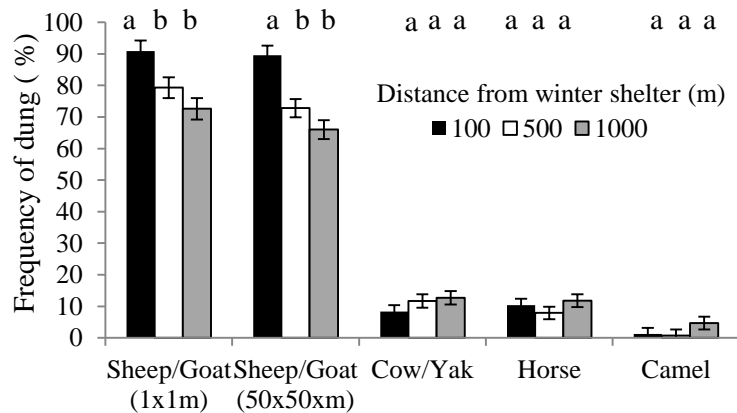
Figure 2.3. Percentage of livestock species (in SFUs) in the total herd of households sharing the same winter shelter at the 131 winter shelters in four ecological zones.

Dung frequency measurements showed that distance from winter shelter can be used as a proxy for livestock activity but only for sheep and goats in the mountain and forest steppe, steppe and desert steppe, and cows/yaks in the mountain and forest steppe (Figure 2.4). For horses and camels, across all ecological zones, there was no decline in dung frequency with increasing distance from winter shelters. Thus, the grazing gradients around winter shelters are principally created by sheep and goats, with no real effect of horses and camels. We did not measure dung in the eastern steppe ecological zone.

a) Mountain and forest steppe



b) Steppe



c) Desert steppe (note change in quadrat frame size below)

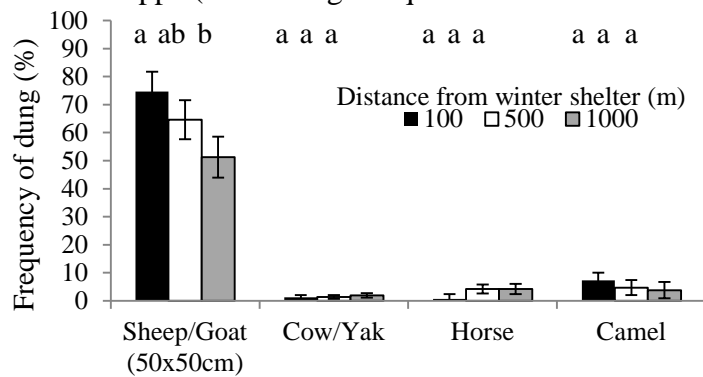


Figure 2.4. Gradients in livestock dung frequency from winter shelter in the a) mountain and forest steppe, b) steppe, and c) desert steppe ecological zones. Bars with different letters above them are significant at $p \leq 0.05$.

We also measured the density of sheep and goat pellets to get a more accurate picture of how they used the pastures around winter shelters. Here we see much heavier use by sheep and goats around winter shelters in the steppe and moderate use in the mountain and forest steppe and lowest in the desert steppe (Figure 2.5). This contrasts strongly with the *soum*-level sheep and goats dung densities (Figure 2.2), where the density of sheep and goats at this level are greater in the mountain and forest steppe than the steppe. In addition, it appears that the grazing gradient declines most sharply from 100 to 500m in the steppe, followed by the mountain and forest steppe and then the desert steppe.

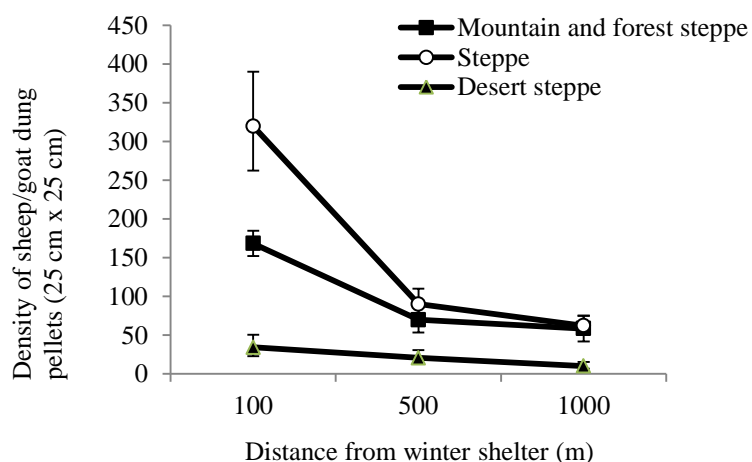


Figure 2.5. Density of sheep/goat dung pellets in the mountain and forest steppe, steppe and desert steppe.

2.3.2. Vegetation results

Ecological zone significantly affected both biomass and cover, thus we completed the analysis separately by zone. Both distance and ecological site significantly affected vegetation variables, but the effects were different for each variable (see ANOVA table in Appendix 2.1). We did not distinguish ecological sites in the eastern steppe, due to low sample size in the two contrasting ecological sites we sampled. In the results below, we first present the interaction effects between the grazing and ecological site and then the main effects in the mountain and forest steppe, steppe and desert steppe zones. The main effect means are adjusted by the other main effects and interactions by LSMEANS in SAS.

2.3.2.1. *Standing crop biomass of all plants and functional groups*

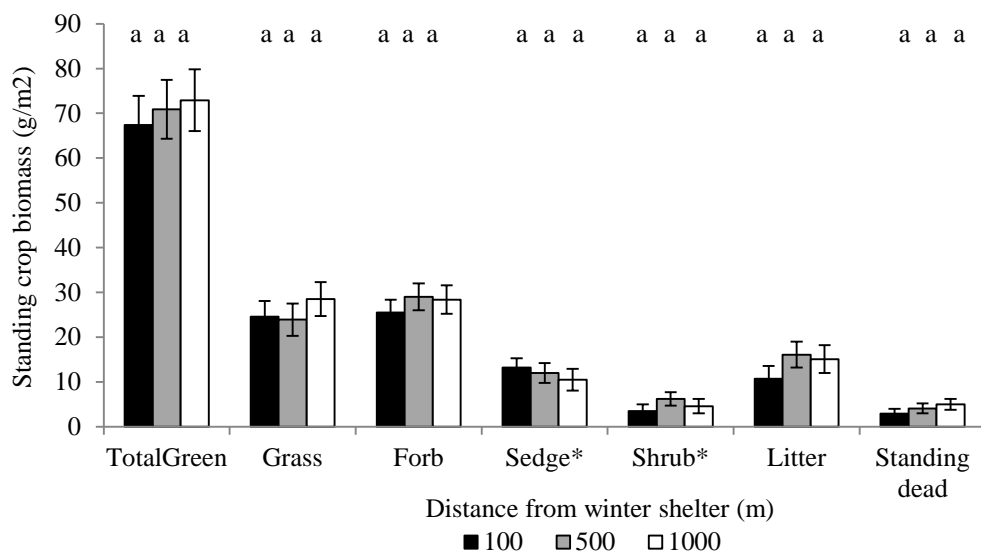
There were no significant interactions between grazing and ecological sites in all ecological zones. The levels of standing crop biomass differed in different ecological zones. The eastern steppe (about 110g/m²) had six times more standing crop biomass than the desert steppe (about 22g/m²). The amount of standing crop biomass in the mountain and forest steppe (about 70g/m²) and steppe (about 54g/m²) fell between the above two. Both grazing and ecological sites were important for vegetation standing crop biomass in the steppe, but ecological site was least important for vegetation standing crop biomass both in the mountain and forest steppe and desert steppe.

The grazing gradient had no effect on biomass of any functional group in the mountain and forest steppe, moderate effects in the eastern steppe and desert steppe and the largest effects in the steppe (Figure 2.6). In the eastern steppe, grazing only affected forb biomass ($p=0.0074$), which was significantly greater in the lightly grazed 1000 m plots than the more heavily grazed

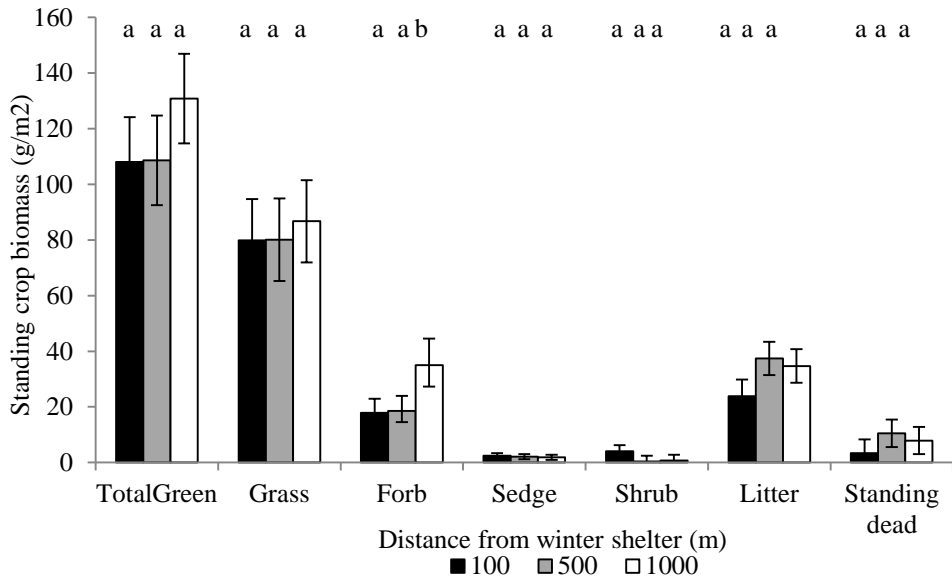
100m and 500m plots (Figure 2.6b). In the steppe, grass ($p=0.02$), litter ($p=0.0007$) and standing dead biomass ($p<0.0001$) were lower in heavily grazed pastures near winter shelters; by contrast, forb biomass ($p=0.02$) showed the opposite pattern. Total green, sedge and shrub biomass did not differ with distance from winter shelter in the steppe (Figure 2.6c). In the desert steppe, only grass standing crop biomass ($p=0.04$) increased with increasing distance from the winter shelter (Figure 2.6d).

Ecological site had limited effect on functional group biomass (no figures presented). In the mountain and forest steppe, ecological site only affected sedge biomass ($p=0.04$) where it was higher in the high water table sites than in the rocky hill shallow sites. In the steppe, both forb ($p=0.01$) and litter ($p=0.04$) biomass were lower in the high water table sites than in the rocky hill shallow sites (See ANOVA table in Appendix 2.1). In the desert steppe zone, total green biomass ($p=0.05$) was lower in the high water table ecological site than in the other ecological sites.

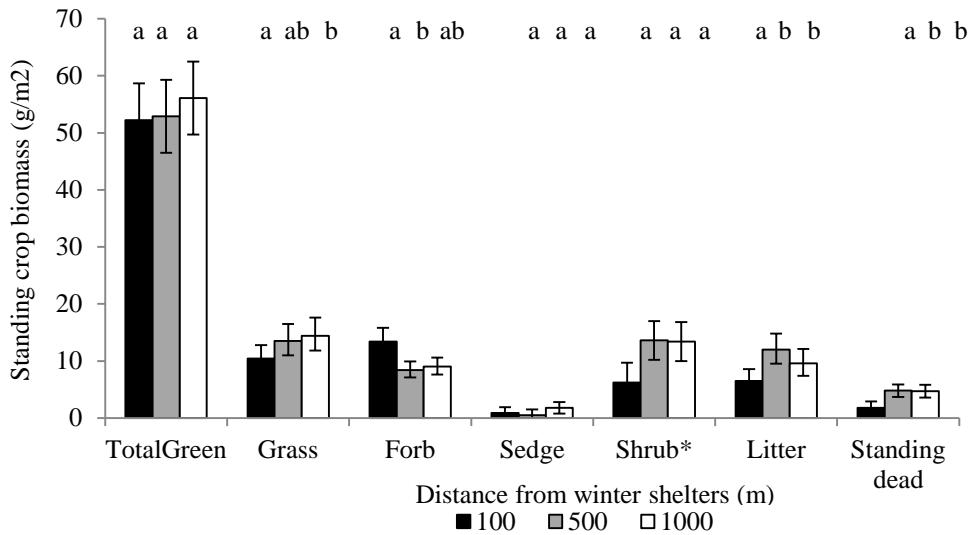
a) Mountain and forest steppe



b) Eastern steppe



c) Steppe



d) Desert steppe

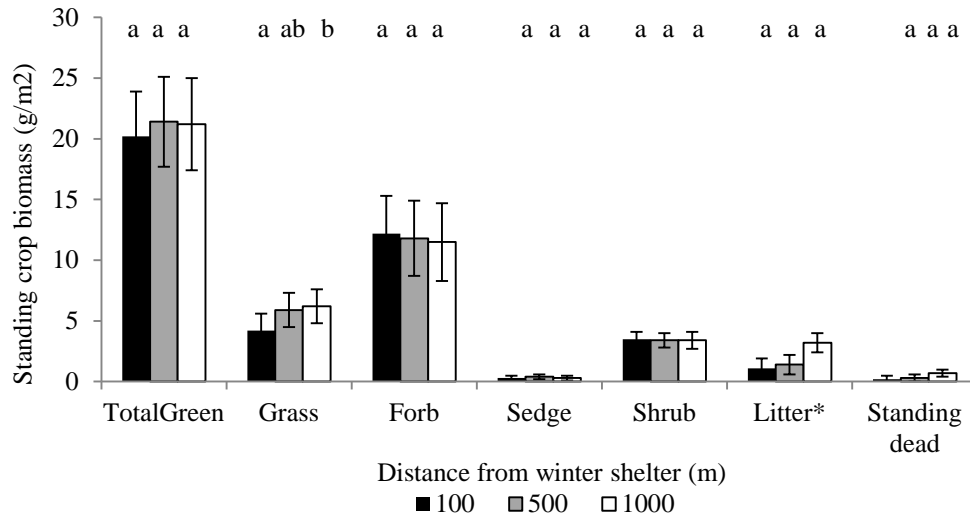


Figure 2.6. Standing crop biomass with distance from winter shelter in the a) mountain and forest steppe, b) eastern steppe, c) steppe and d) desert steppe. Bars with different letters above them are significant at $p \leq 0.05$. *- represents variables were significantly different using a ranked transformation, but are represented here on the original scale.

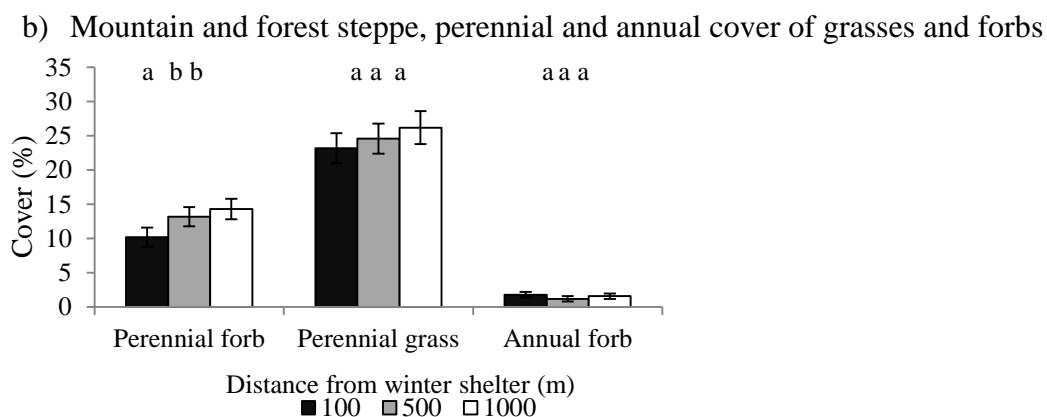
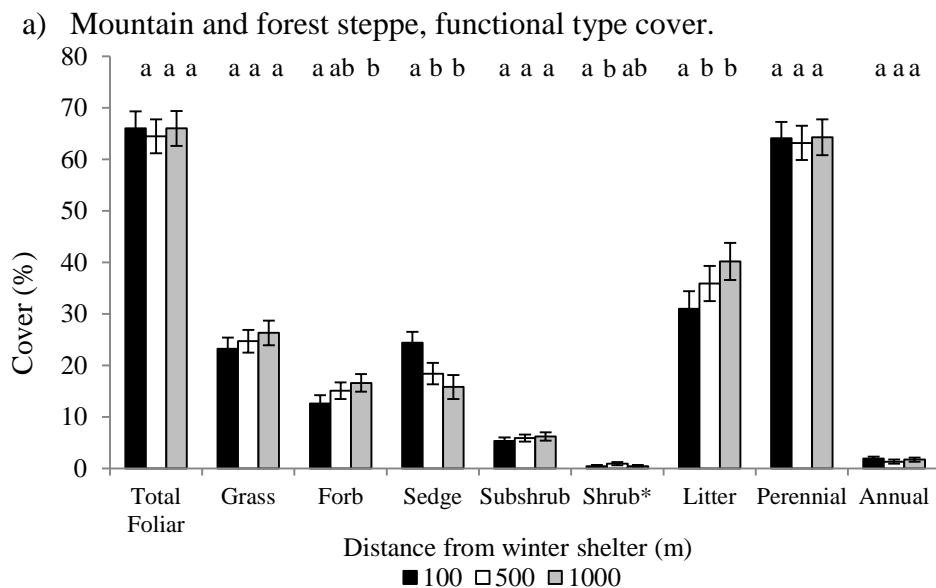
2.3.2.2. Plant functional type cover

It appears that cover was a more responsive measure than biomass to different levels of grazing (Figure 2.7). There were no significant interactions between grazing and ecological sites in the mountain and forest steppe and desert steppe. In the steppe, there were significant interaction effects between grazing and ecological sites in grass ($p=0.04$) and perennial plant ($p=0.02$) cover. These interactions are described in the steppe zone section below.

Overall, there was very little cover of annuals, shrubs and sedges in any zone. In the mountain and forest steppe, there was no effect of grazing on total foliar, grass and sub-shrub cover. In this zone, there was more forb (mostly perennial, $p=0.02$), shrub ($p=0.02$) and litter cover ($p=0.0002$) farther from the winter shelter, but the opposite was true for sedge cover ($p=0.0004$, Fig 2.7a and b). Statistical significance ($p=0.02$) for shrub cover was tested using a binary data (shrub cover present and absent) and the actual amount was small compared to the

cover of other functional types. There were no grazing impacts on the rest of functional groups measured in the mountain and forest steppe.

Ecological site affected plant and bare soil cover in the mountain and forest steppe (Figure 2.7c). Total foliar cover, sedge and perennial plant cover were lower in rocky/hill/shallow sites with lower soil water retention than in the loam, clay and high water table ecological sites with higher soil water retention. By contrast, there was less bare soil cover in the loam ecological site than the rocky/hill/shallow site. There were no significant interactions between grazing and ecological sites for functional type cover in this zone.



c) Mountain and forest steppe, functional type cover in different ecological sites

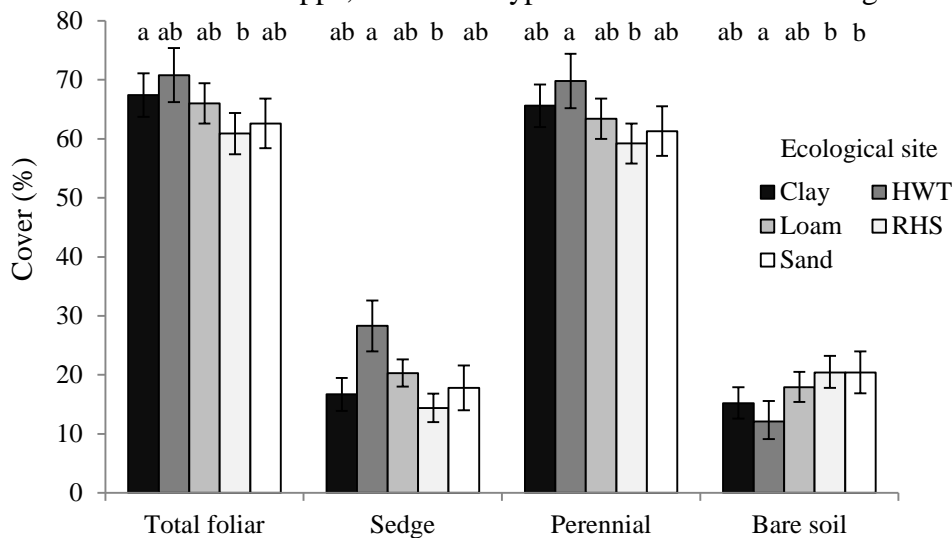
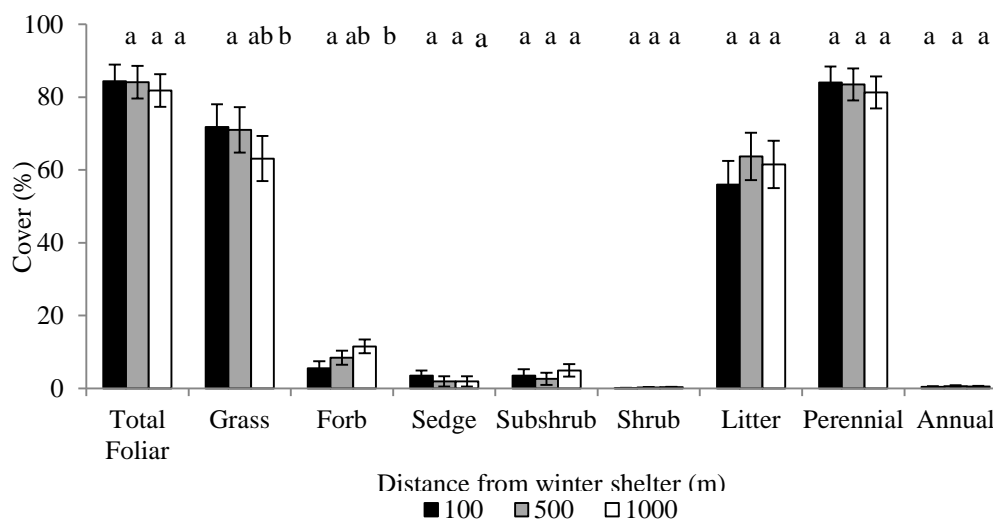


Figure 2.7. Functional type cover a,b) along the grazing gradient and c) across the ecological sites, in the mountain and forest steppe. Bars with different letters above them are significant at $p \leq 0.05$. HWT-High Water Table, RHS-Rocky/Hill/Shallow; *- represents variables that were significantly different using binary (plots with shrubs and plots with no shrub cover) data, but are represented here on the original scale.

In the eastern steppe, we were surprised to find that grass cover was lower and forb cover was higher where grazing was less intense, far from winter shelters. Total foliar, sedge, subshrub, shrub, litter and annual forb cover did not differ along the grazing gradient (Figure 2.8a and b).

a) Eastern steppe, functional type cover



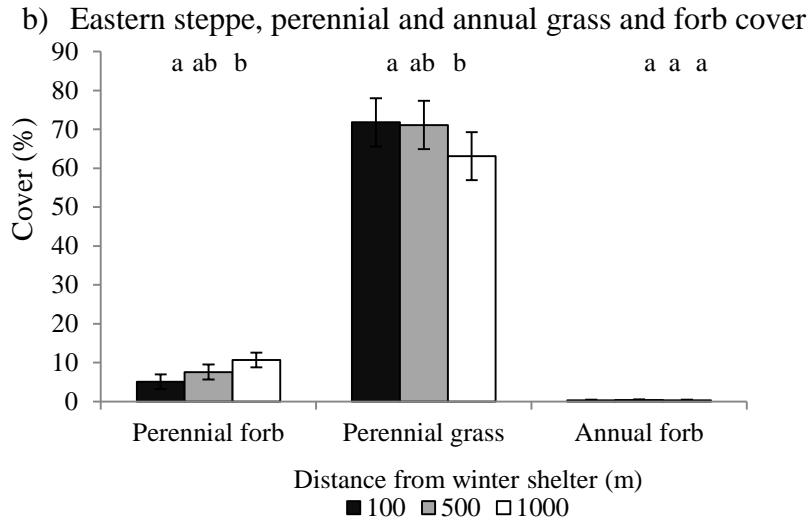


Figure 2.8. Effects of grazing on functional type cover in the eastern steppe: a) functional types, b) perennial vs annual grass and forb cover. Bars with different letters above them are significant at $p \leq 0.05$.

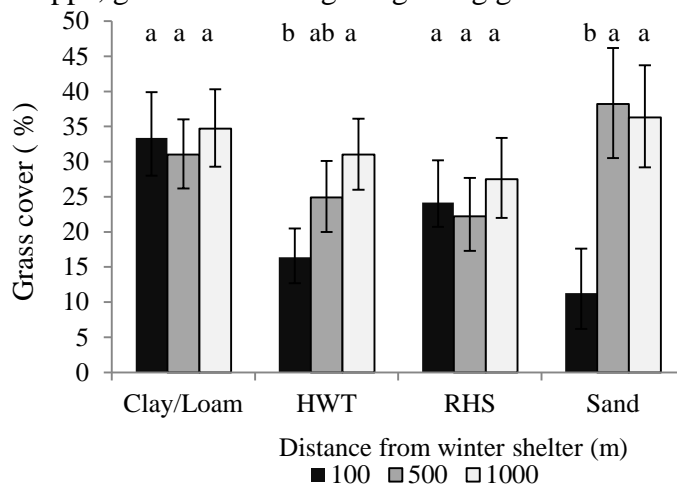
In the steppe, there was a significant interaction effect between ecological site and grazing for grass ($p=0.04$), perennial grass ($p=0.04$) and perennial plant cover ($p=0.02$). Grazing reduced grass and perennial grass cover on sandy and HWT sites, but not on clay loam and rocky hill shallow sites. (Figure 2.9a). For perennial plant cover, the pattern was similar as shown in grass and perennial grass cover where there was no difference in cover along the grazing gradient in clay loam and rocky hill shallow sites, but there was less cover at the 100 m plots in the sandy ecological sites (Figure 2.9b).

There were many significant effects of grazing in the steppe zone, and these effects conformed to the classic predicted effects of grazing on vegetation. Total foliar cover at 500 m plot was lower than the plot at 1000 m. But there was no difference in the cover between the plots at 100m and at the other two (plots at 500m and 1000m). This pattern could be created by abundant weedy plants at 100m plots. In contrast to the eastern steppe, forb cover increased with distance to the winter shelter (Figure 2.9c). There was more shrub, litter, perennial plant and perennial grass cover far than near the winter shelters, but the opposite was true for annual plant

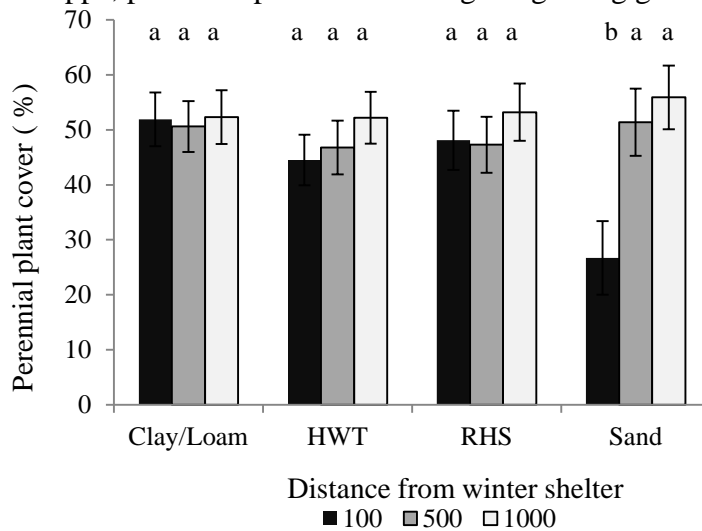
and annual forb cover. Sedge, perennial forb and subshrub cover did not differ along the grazing gradient.

In the steppe, ecological site only affected cover for grasses and perennial plant basal cover (Figure 2.9e). Grass cover was greater in clay loam ecological site than high water table ecological site, interestingly; perennial plant basal cover in high water table was lower than in rocky hill shallow ecological site.

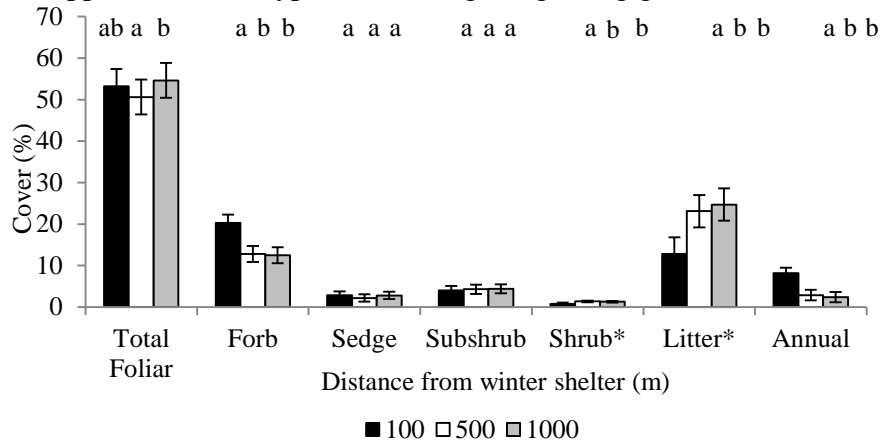
a) Steppe, grass cover along the grazing gradient in different ecological sites



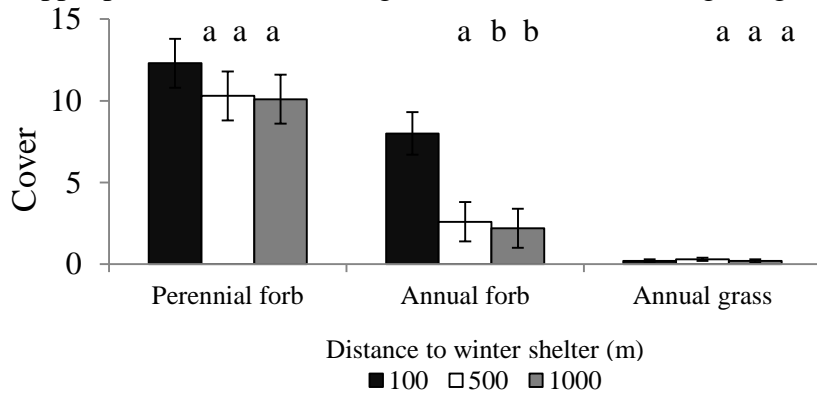
b) Steppe, perennial plant cover along the grazing gradient in different ecological sites



c) Steppe, functional type cover along the grazing gradient



d) Steppe, perennial and annual grass and forb cover along the grazing gradient



e) Steppe, grass and perennial plant basal cover in different ecological sites

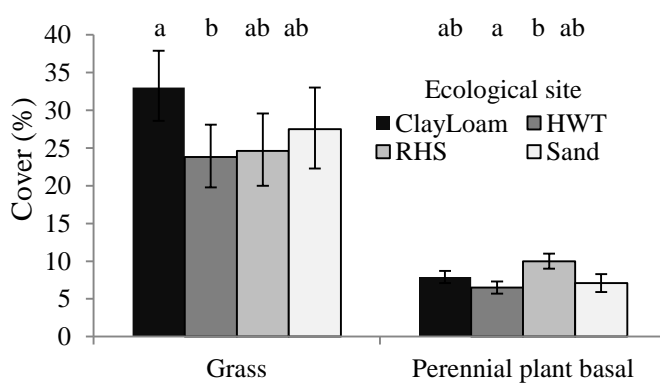
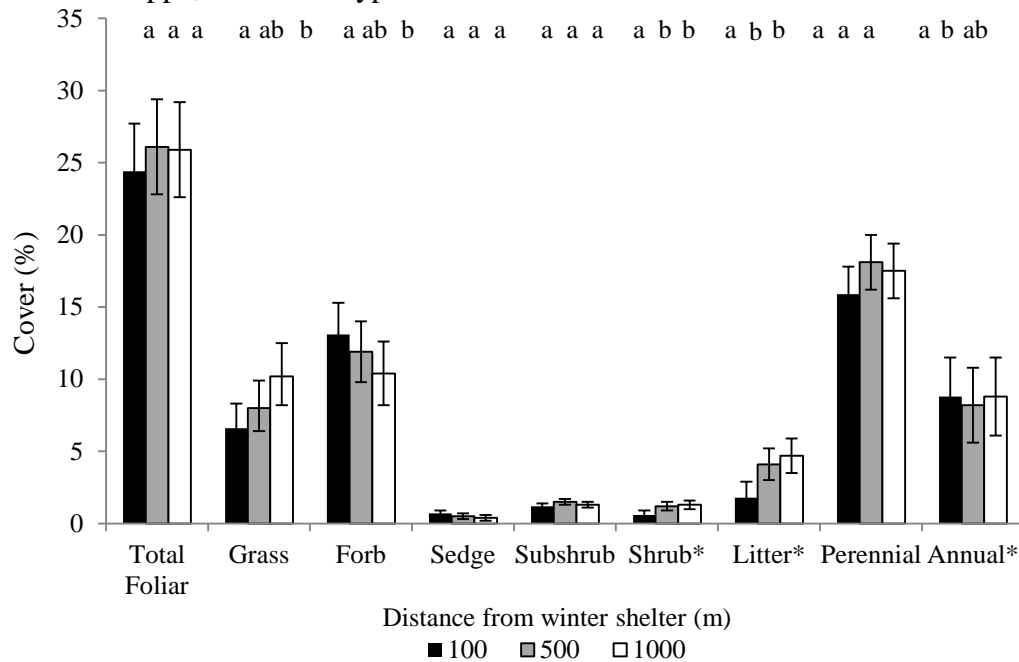


Figure 2.9. Vegetative cover in the steppe ecological zone, including: a) grass and b) perennial plant cover along the grazing gradient and in different ecological sites, and c) functional type and, d) perennial vs annual grass and forb cover along the grazing gradient, and e) grass and perennial plant basal cover across different ecological sites (HWT-High Water Table, RHS-Rocky Hill Shallow). Bars with different letters above them are significant at $p \leq 0.05$. *-represents variables where statistical significance was tested using ranked data.

In the desert steppe, there was no significant interaction effect between ecological site and grazing for all variables. There was more cover of grass ($p=0.0017$), perennial grass ($p=0.0028$), shrub ($p<0.0001$) and litter cover ($p<0.0001$) farther from winter shelters and this pattern was opposite for annual ($p=0.03$) and perennial forb ($p=0.01$) cover (Figure 2.10a). Annual plant cover at 100 m plots was higher than at 500m plots. Grazing had no effect on total foliar, sedge, subshrub, perennial plants and annual grass cover. Ecological site affected only total green biomass ($p=0.04$), where it was greater in the clay loam rocky hill shallow sand site than the high water table site.

a) Desert steppe, functional type cover



b) Desert steppe, perennial and annual grass and forb cover

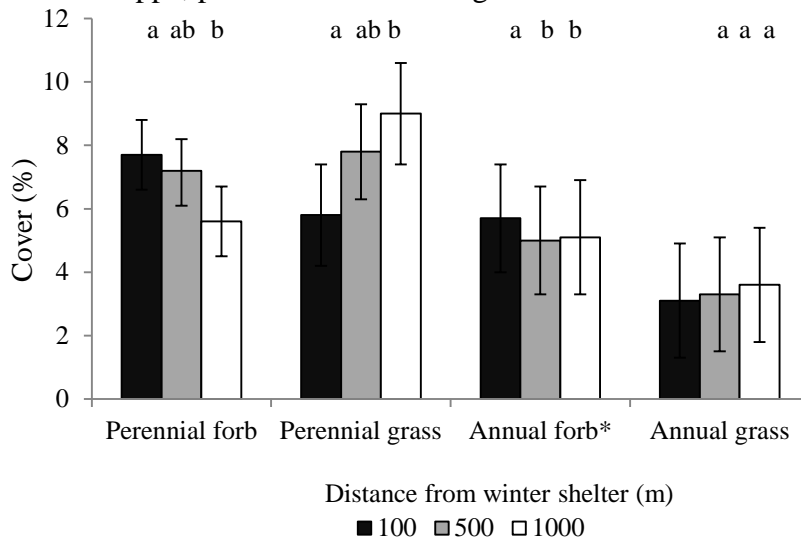


Figure 2.10. Vegetative cover with distance from winter shelter for a) functional types and b) perennial vs annual grasses and forbs. Bars with different letters above them are significant at $p \leq 0.05$. *- represents variables where statistical significance was tested using ranked data.

2.3.2.3. Species cover and species richness

We analyzed the species level cover only for dominant species, which were those that occurred on more than 10% of plots in each ecological zone in our sample. We identified 28 dominant species in the mountain and forest steppe, 15 dominant species in the eastern steppe, 18 dominant species in the steppe and 16 dominant species in the desert steppe. For these species, we completed the same ANOVA tests as above for other variables.

In the mountain and forest steppe, we found no interactions between ecological site and grazing for any individual species. Ecological site affected only for the cover of *Pulsatilla turczaninowii* ($p=0.05$) where the cover was greater in clay rocky hill shallow and sand ecological sites than in high water table site. The cover of only two species was affected by grazing (intensity) around winter shelters. The grazing tolerant sedge, *Carex duriuscula*, was more abundant ($p=0.04$) near than far from winter shelter, but opposite was true for the perennial

grass, *Agropyron cristatum* ($p=0.04$). In the eastern steppe, there was only one dominant species affected by grazing; *Cleistogenes squarrosa* was more abundant ($p=0.01$) far than near the winter shelters.

In the steppe, there was a significant interaction between ecological site and grazing for the cover of *Allium polyrrhizum* ($p=0.02$) and *Salsola collina* ($p=0.03$). The cover of the perennial forb *Allium polyrrhizum* was greater close to winter shelter in clay loam and high water table ecological sites, but was greater farther from winter shelter in the rocky hill shallow and sand ecological sites. The cover of the annual forb, *Salsola collina*, was greater close to winter shelter in high water table ecological site but was lower far from the winter shelter in the rocky hill shallow sand.

In total, 5 out of 18 dominant species were influenced by the main effect of grazing. *Chenopodium album*, a disturbance associated species, was most abundant ($p<0.0001$) close to winter shelters but the pattern was opposite for the drought-tolerant steppe grass, *Stipa krylovii* ($p=0.001$). The perennial shrub, *Caragana microphylla*, covered more ground ($p=0.009$) far from winter shelters, and the cover of *Caragana stenophylla* was greater ($p=0.05$) at 500m than 1000 m plots. The large perennial grass, *Achnatherum splendens*, often associated with ground water, was more abundant ($p=0.02$) near than far from winter shelters. Ecological site affected only the cover of the perennial grass, *Agropyron cristatum* ($p=0.0007$), which was greater in clay loam rocky hill shallow and sand ecological sites than the high water table sites.

In the desert steppe, there were no significant interactions effects for individual species. But, two out of 16 dominant species were affected by the main effect of grazing. The perennial forb, *Allium mongolicum* ($p=0.05$), and the perennial grass, *Stipa gobica* ($p=0.05$), were both more abundant far away from winter shelters.

In all ecological zones, species richness was not affected by grazing. In average there were 37 plant species in our study plots in the mountain and forest steppe, 38 plants in the eastern steppe, 25 species in the steppe and 21 species in the desert steppe. In the steppe only, species richness was affected by ecological site, where it was greater in the rocky hill shallow site than the high water table site.

2.3.2.4. Bare soil, basal cover and vegetation gaps

We used different attributes to assess if grazing or ecological site affected bare soil, basal perennial plant cover or the size of vegetative gaps. There was no significant interaction between ecological site and grazing on these attributes in any ecological zone. Also, grazing had no effect on either basal cover or bare ground in any of the four ecological zones (Figure 2.11).

There were, however, some effects of grazing on gap size. In the eastern steppe, average gap size between the bases of perennial plants was smaller ($p < 0.0001$) in heavily used pastures near winter shelters, but the opposite was true for the steppe, where average gap ($p = 0.01$) was smaller in lightly used pastures far from winter shelters. Average gap size was not affected by grazing in the mountain and forest and desert steppe zones (Figure 2.12a-d).

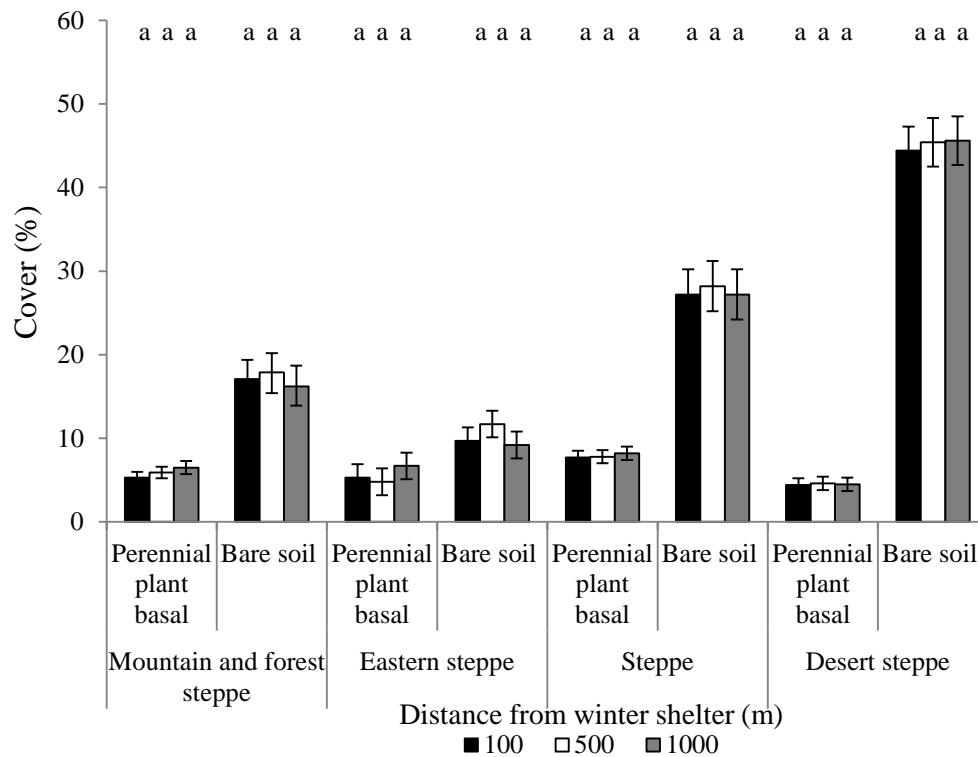
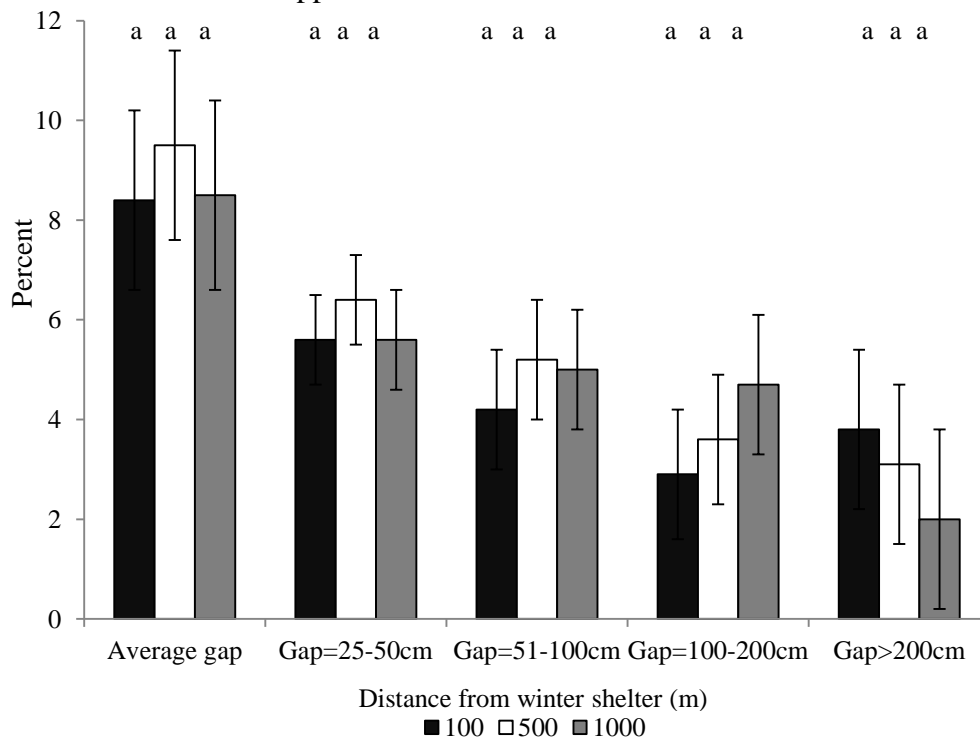
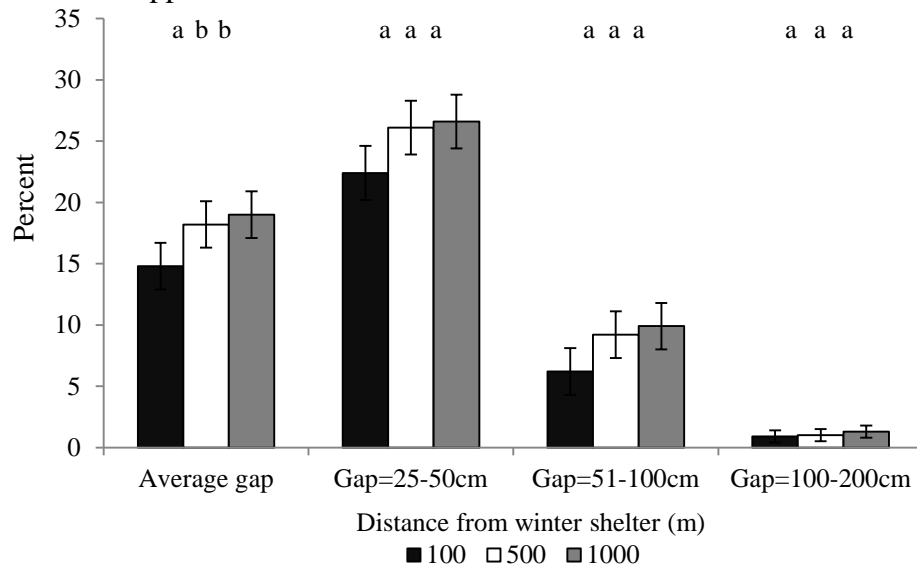


Figure 2.11. Perennial plant basal cover and bare soil cover with distance from winter shelter in the four ecological zones. Bars with different letters above them are significant at $p \leq 0.05$.

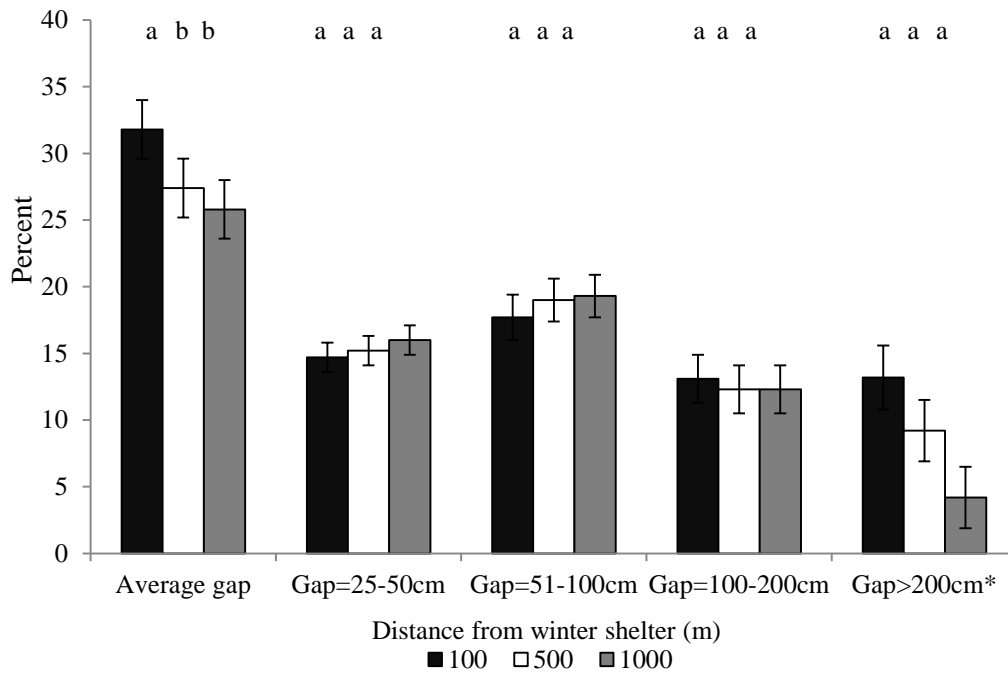
a) Mountain and forest steppe



b) Eastern steppe



c) Steppe



d) Desert steppe

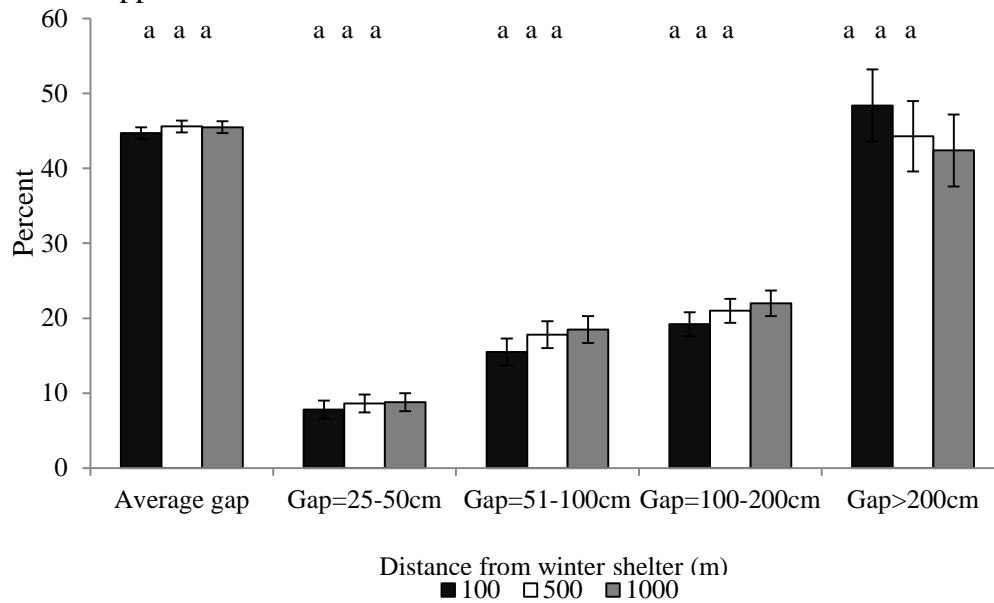
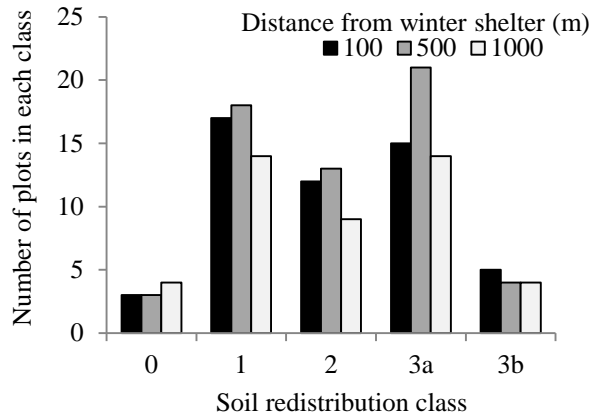
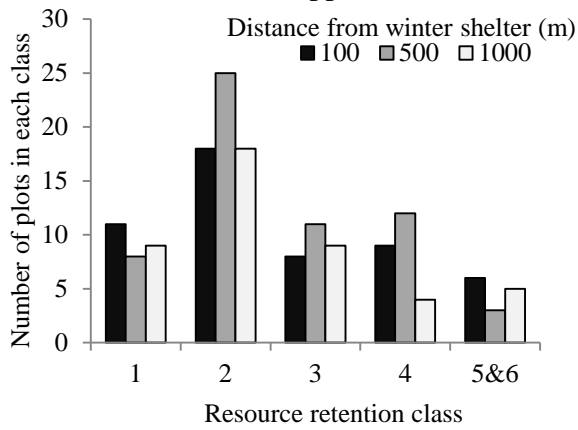


Figure 2.12. The effects of grazing on the percent of the transect line covered by gaps of different sizes and their average, measured between the bases of perennial plants, in the a) mountain and forest steppe, b) eastern steppe, c) steppe, d) desert steppe. Bars with different letters above them are significant at $p \leq 0.05$. *- represents variables where statistical significance was tested using a binary data.

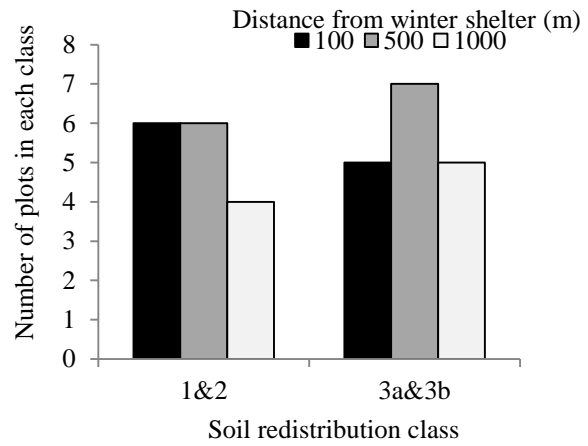
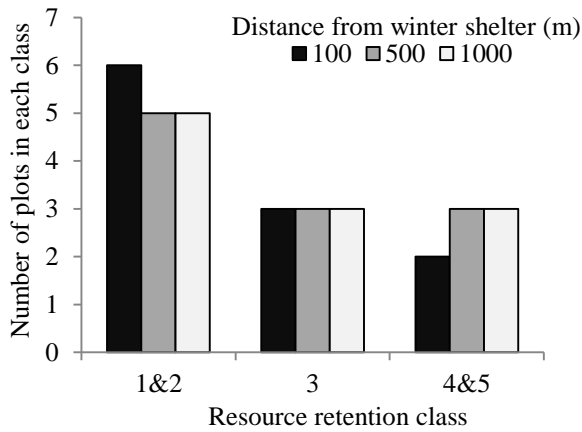
2.3.2.5. Soil surface characteristics

To understand if grazing affected the spatial pattern of bare ground and plant patches, we measured the (resource retention) size and connectivity of persistent vascular plant patches and inter-patch areas across the plot and the extent and severity of soil redistribution processes (soil erosion and deposition by wind and water). Low numbers in soil resource retention classes indicate smaller open patches and greater connectivity of vascular plant patches and the opposite for high numbers with more fragmented and isolated patches. For soil redistribution classes, low numbers indicate less soil movement, whereas high numbers indicate more extensive soil movement. None of the Chi-square tests showed significant effects of grazing intensity on either resource retention and soil redistribution in any of the ecological zones (Figure 2.13).

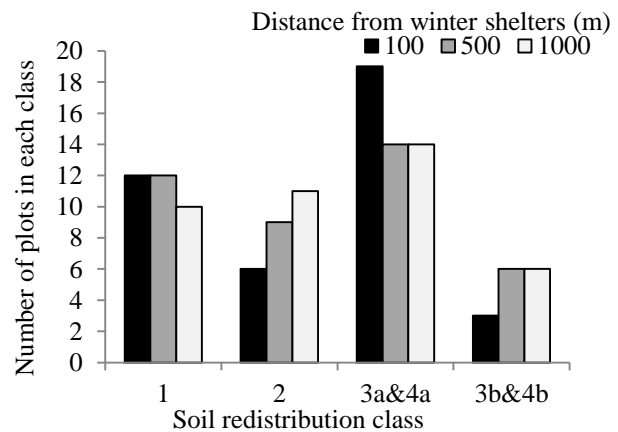
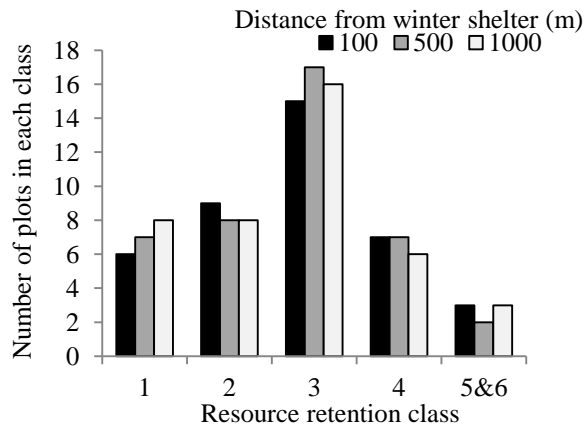
a) Mountain and forest steppe



b) Eastern steppe



c) Steppe



d) Desert steppe

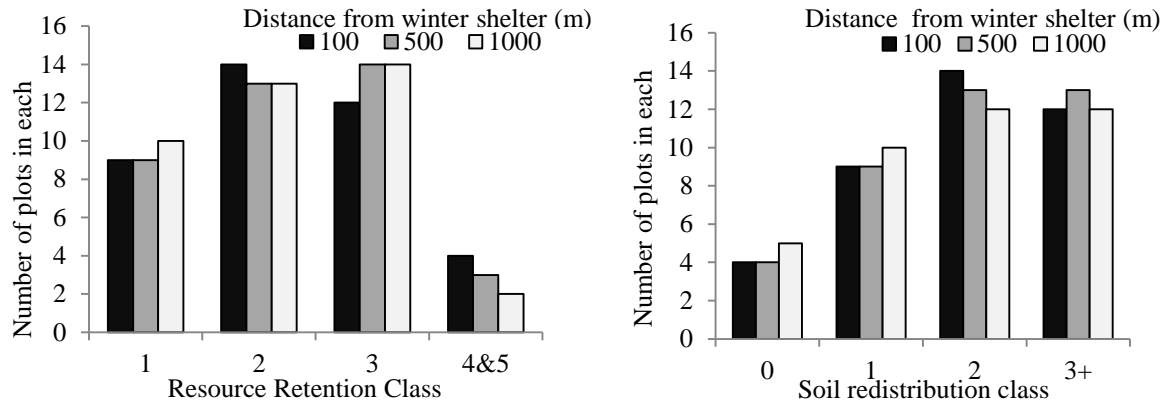


Figure 2.13. The effects of grazing on soil surface characteristics (resource retention class and soil redistribution class) in the a) mountain and forest steppe, b) eastern steppe, c) steppe and d) desert steppe.

2.3.2.6. Forage quality and vegetation palatability

Grazing only influenced crude protein in the steppe ecological zone, with higher levels of crude protein ($p=0.0008$) in forage near winter shelters, where grazing pressure was high (Figure 2.14). Similarly, ADF was lower ($p=0.0088$) in the forage most heavily used next to the winter shelters in the desert steppe, but not in other ecological zones.

In the mountain and forest steppe, perennial forbs and grasses that are palatable for sheep and goats during the growing season were less abundant ($p=0.03$ for forb, $p=0.04$ for grass) in the heavily grazed pastures close to winter shelters. Perennial forbs that are unpalatable for sheep and goats were also abundant ($p=0.02$) farther from winter shelters in both the growing and dormant seasons (Figures 2.15a and 2.16a).

In the eastern steppe, unpalatable perennial forb cover was lower ($p=0.005$) near than far from winter shelter in both growing and dormant season. Palatable and unpalatable perennial grass, unpalatable annual forb did not vary in both growing and dormant seasons with distance from winter shelters (Figures 2.15b and 2.16b).

In the steppe, there was a significant distance x ecological site interaction effect on the cover of palatable perennial grasses ($p < 0.0001$). The interaction effect pattern was the same as shown above for perennial grass cover, where there was no difference in cover in response to grazing in clay loam and rocky hill shallow sites, but there was more cover farther from winter shelters in the sandy and high water table ecological sites. Also, heavy grazing near winter shelters reduced the cover of perennial grasses ($p < 0.0001$) that were palatable during both growing and dormant seasons (Figure 2.15c and 2.16c). Dormant season palatable annual forb cover ($p = 0.05$) increased with increasing distance from winter shelters, whereas, both growing ($p < 0.0001$) and dormant ($p < 0.0001$) season unpalatable annual forb cover was greater close to winter shelter. Dormant and growing season palatable and unpalatable perennial forbs did not differ with distance from winter shelter (Figures 2.15c and 2.16c).

In the desert steppe, growing season palatable perennial forbs were greater ($p = 0.05$) at 500m plots than plots at 100m and 1000m. Growing season unpalatable annual forbs decreased ($p = 0.02$) with increasing distance from winter shelters (Figures 2.15d and 2.16d).

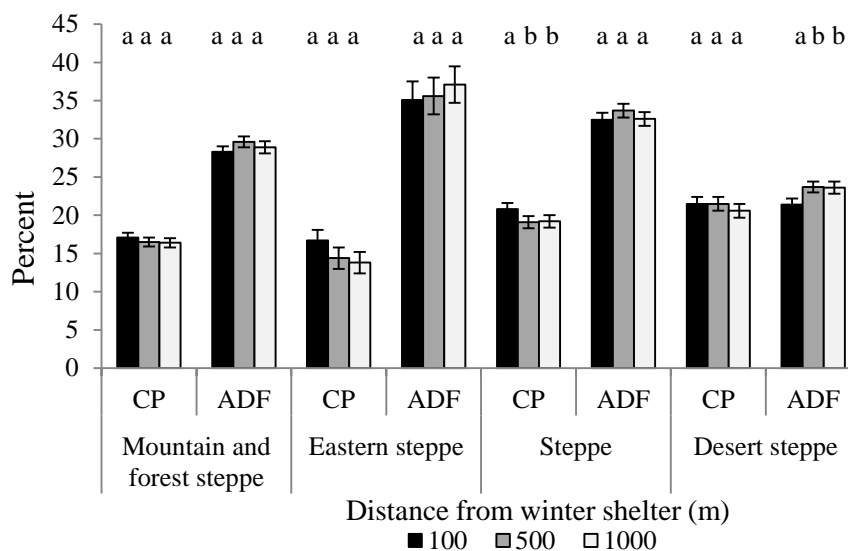
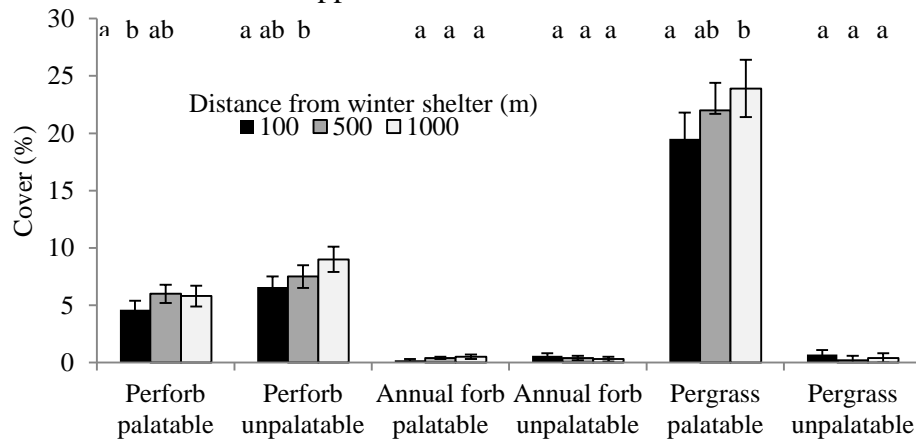
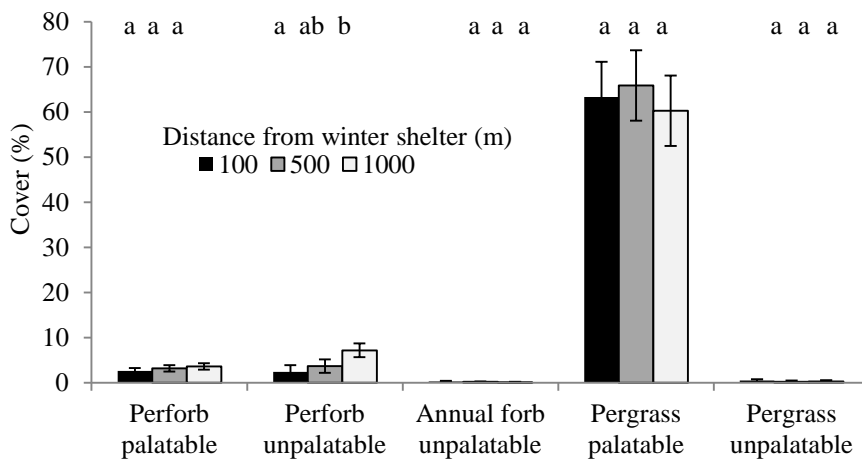


Figure 2.14. The effects of grazing on crude protein (CP) and acid detergent fiber (ADF) in forage in the four ecological zones. Bars with different letters above them are significant at $p \leq 0.05$.

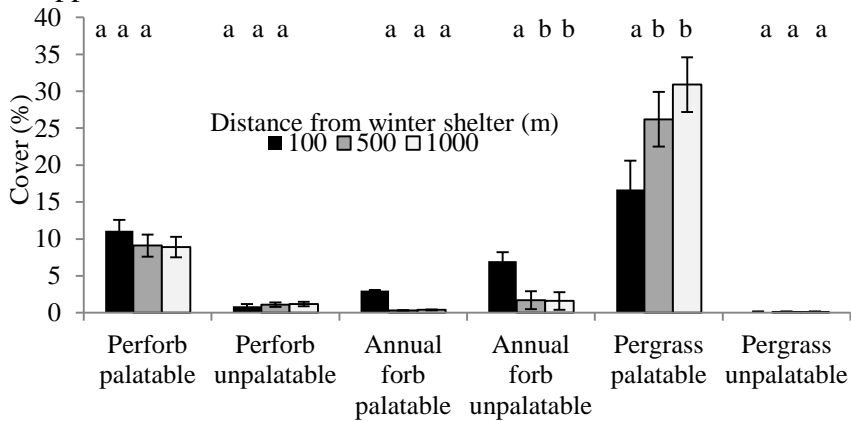
a) Mountain and forest steppe



b) Eastern steppe



c) Steppe



d) Desert steppe

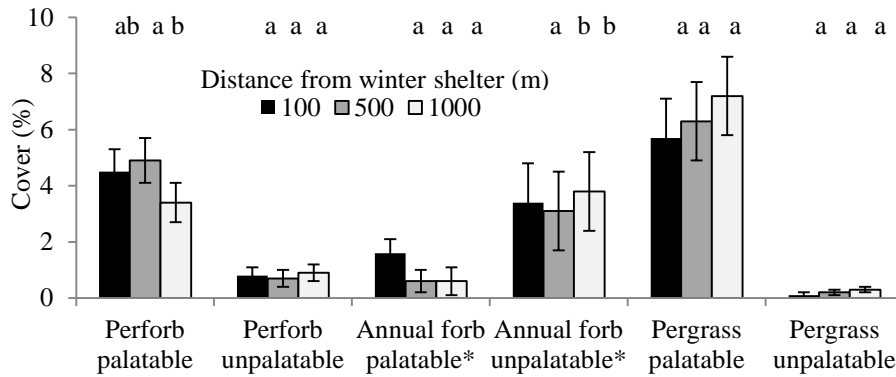
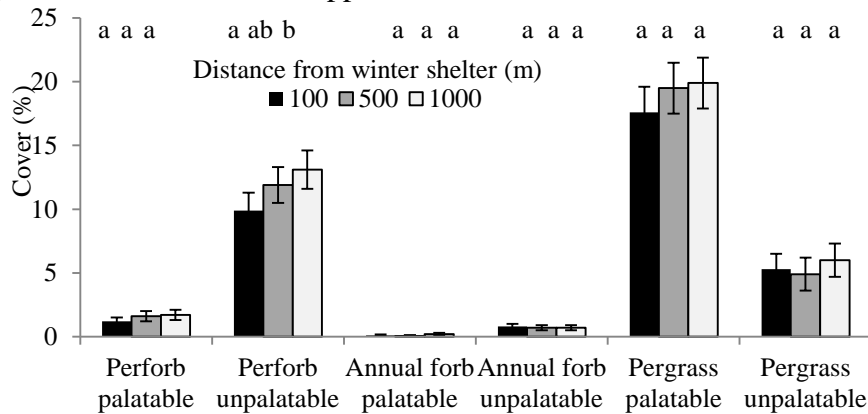
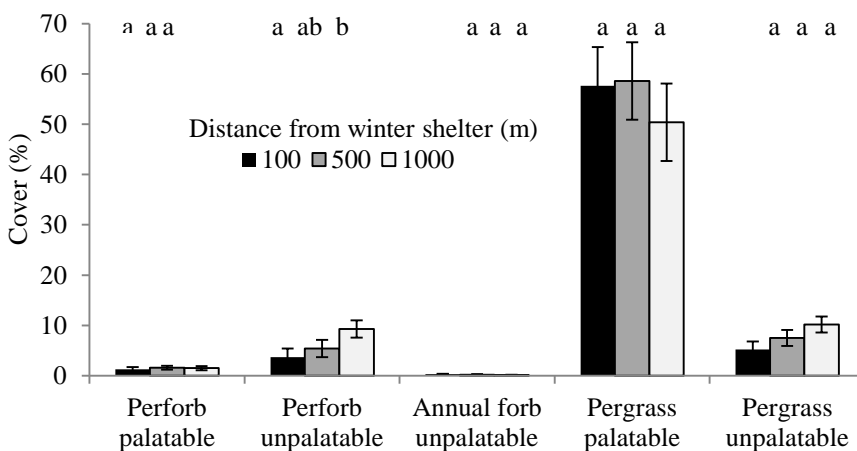


Figure 2.15. Changes in palatable and unpalatable grass and forb cover during the growing season with distance from winter shelter in the a) mountain and forest steppe, b) eastern steppe, c) steppe and d) desert steppe ecological zones. Bars with different letters above them are significant at $p \leq 0.05$. *- represents statistical significance found in the ranked data. Perforb=perennial forb, pergrass=perennial grass.

a) Mountain and Forest steppe



b) Eastern steppe



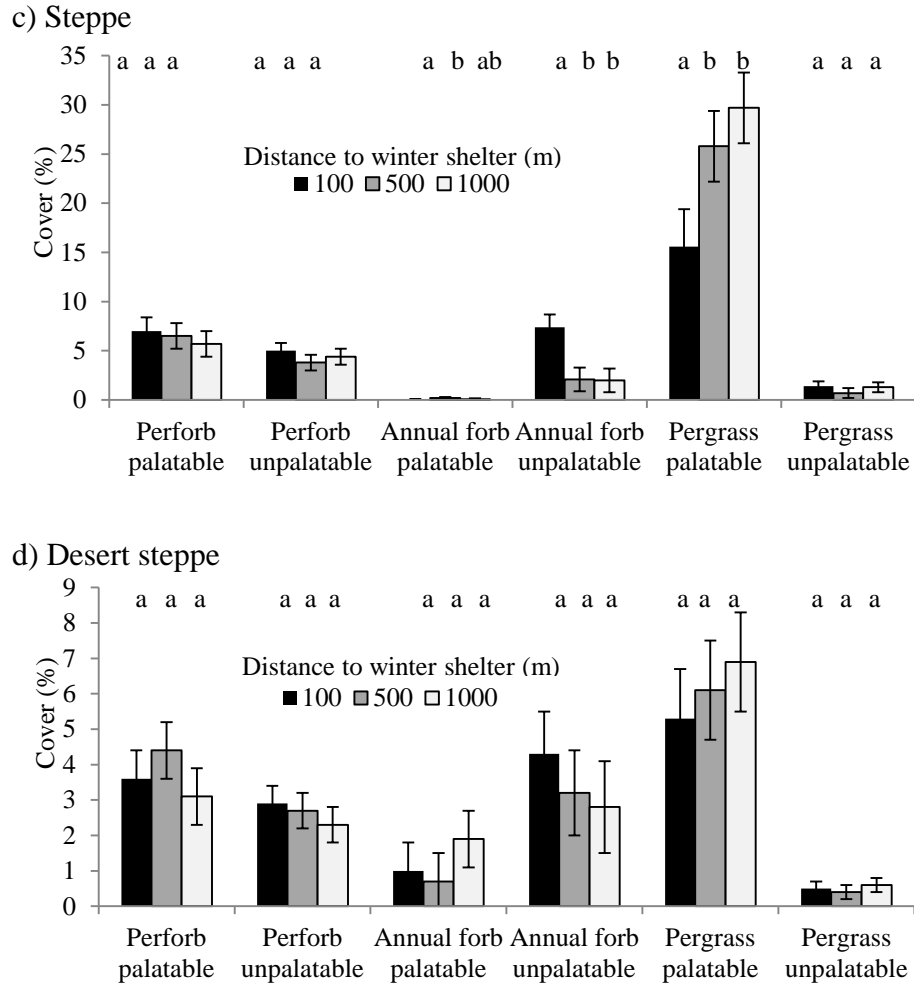


Figure 2.16. Changes in palatable and unpalatable grass and forb cover during the dormant season with distance from winter shelter in the a) mountain and forest steppe, b) eastern steppe, c) steppe and d) desert steppe ecological zones. Bars with different letters above them are significant at $p \leq 0.05$. Perforb=perennial forb, pergrass=perennial grass.

2.4. Discussion

This study focused on the effects of livestock grazing on dormant season pastures, or winter pastures, which can be different than the effects of livestock grazing on growing season pastures. The main effects of livestock on dormant season pastures are trampling and grazing on senescent vegetation which affect soil hydrology and physical quality thus impacting its capacity to function (Stavi et al. 2011). Trampling can also increase soil compaction and reduce infiltration capacity, thus reducing soil moisture content. Lower soil moisture content decreases

herbaceous colonization, litter production and decomposition (Stavi et al. 2011). Trampling damage to the crowns of the plants in winter reduces the leaf surface area and volume in the following growing season. Livestock trampling can have a particularly strong effect on certain species if their buds are lower due to heavy grazing in the previous growing season, and these can be damaged in winter, when there are few tillers. Less litter results in increased raindrop impact on the soil, reduces water infiltration and also increases evaporative losses of moisture from the soil (Bromley et al. 1997, Stavi et al. 2008) In this study we did not measure compaction and infiltration, but we did measure the cover (biomass) of litter and standing dead which are important attributes for understanding the direct and indirect effects of livestock on the winter pastures.

Overall, grazing induced changes along grazing gradients were larger in the steppe ecological zone than any other ecological zone, with more subtle effects in the desert steppe, and little change in the mountain and forest steppe and eastern steppe. In the steppe, there was lower grass and litter cover and higher forb and annual weedy unpalatable vegetation cover in the heavily grazed pastures near winter shelters. This is a pattern that is supported by high dung density in the pastures of 41 winter shelters in this zone. In this zone, perennial species were replaced by annuals and this created open gaps in heavily used, pastures proximate to winter shelters.

In the desert steppe, there was less grass and shrub and litter cover and more total and annual forb cover near winter shelters, similar to the patterns observed in the steppe. However there was no evidence of grazing induced soil changes in the 39 winter shelters we studied in this zone.

In the mountain and forest steppe, we found greater sedge and lower litter cover close to winter shelters indicating more subtle grazing induced changes. Here, there was no indication of changes in the spatial patterning, bare ground cover, soil resource retention or soil redistribution associated with grazing. On the other hand there was no grazing-associated soil erosion and deposition thus there is a weak indication of the processes that affects plants directly via disturbances and indirectly via the addition or loss of nutrients on the pastures of 52 winter shelters we studied. In the eastern steppe, grazing reduced forb cover and biomass in the intensely grazed pastures close to winter shelters. There was no evidence of grazing-induced changes in spatial patterning of vegetation and open bare ground or erosion/deposition in the pastures of 11 winter shelters we sampled in this zone.

2.4.1. Livestock grazing patterns in winter pastures

Only sheep and goats (and cows and yaks in the mountain and forest steppe) created grazing gradients around winter shelters. Sheep and goats are brought to the winter shelters each night, but horses and camels are not, and graze away from the winter shelter. Cows and yaks are brought in for milking every day in winter and thus they do contribute to this grazing gradient especially in the mountain and forest steppe. Our dung data confirm a strong grazing gradient created by sheep and goats in our three sampled zones, with a gradient contributed by cows and yaks only in the mountain and forest steppe.

We had a classic piosphere effect in all ecological zones, but its strength differed by zone (Figure 2.5). The piosphere effect was strongest in the steppe and moderate in the mountain and forest steppe and weakest in the desert steppe. We offer three hypotheses to explain why grazing intensity near the winter shelters was greater in the steppe than in the mountain and forest steppe

(Figure 2.5). First, marginally more sheep and goats numbers in the steppe zone could contribute to the higher grazing intensity near the winter shelters. Second, our field observation suggest that in some areas sheep and goats spend more time close to the winter shelters in the steppe than in the mountain and forest steppe. This is because in the mountain and forest steppe, rough terrain around the winter shelters may causes sheep and goats to move away quickly from winter shelters perhaps because milking cows and yaks trample and graze heavily around winter shelters. Third, there are more cows and yaks (35% of herds) in the herds of families in the mountain and forest steppe, and these species have larger grazing orbits than sheep and goats. These larger grazing orbits may have caused the relatively ‘flat’ grazing gradients in this region compared to the steppe. These flat grazing gradients may explain the relatively weak impacts of grazing in this zone.

The piosphere effect was weak in the desert steppe, partly because the desert steppe supports a lower density of sheep and goats ($p=0.08$) which creates weak grazing gradients. Sheep and goats have a spatially extensive grazing orbit in this zone which also contributes to the weak piosphere effect. This extensive grazing may explain the relatively weak impacts of grazing in this zone.

2.4.2. Effects of winter grazing on vegetation, forage quality and the soil surface

We examined a wide range of vegetation variables to assess the effects of winter grazing on vegetation in different ecological zones, ranging in resolution from very coarse, like total biomass and total foliar cover, to medium, like functional group cover to fine individual species cover. Coarser resolution variables were less sensitive to grazing than medium or fine resolution variables. For example, winter grazing had no effect on total biomass or total foliar cover in any

ecological zone. But winter grazing did affect the cover of functional groups including grasses, forbs, shrubs and sedges. Winter grazing also influenced the cover of fine resolution variables measured at the species level. Khishigbayar et al. (2015) found similar impacts of grazing on vegetation when measuring changes in cover between 1994 and 2013 in central Mongolia.

Mountain and forest steppe

In the mountain and forest steppe, we found no changes in coarse measures of vegetation including total foliar cover, total standing crop biomass and potentially slow changing variables like the size of open gaps between perennial plants. At the medium resolution, functional group level, sedge cover was high and shrub cover was greater in intensely grazed areas close to winter shelters, mirroring results found Khishigbayar et al. (2015) when measuring vegetation around water points in the mountain and forest steppe zone of central Mongolia. At the finer resolution species level, the grazing tolerant and dominant sedge, *Carex duriuscula*, was more abundant in heavily grazed sites and this species is recognized in Mongolia as an indicator of grazing induced vegetation change (Hilbig 1995, Gunin et al. 1999, Fernandez-Gimenez and Allen-Diaz 2001).

Winter grazing affected the distribution of palatable and unpalatable plants in the mountain and forest steppe especially those preferentially grazed during the growing season, with more cover of palatable forbs and grasses far from winter shelters. This is similar to findings around the world (Sternberg et al. 2000, McIntyre and Lavorel 2001, Sasaki 2008).

There was little effect of winter grazing on the spatial patterning and level of bare ground and in general, pastures in this zone can retain water and nutrient resources and can prevent erosion. There was also no grazing-associated soil erosion and deposition thus there is no

indication that grazing affects plants directly or indirectly through soil loss or via the addition or loss of nutrients.

The results from mountain and forest steppe suggest that grazing is having weak impacts on the production and abundance of plant functional groups and species in winter pasture areas. Here, basic ecosystem functions of primary production appear to be largely intact, the soil surface is protected from soil loss and erosion, and it does not appear these pastures are degraded at this time.

Eastern steppe

In the eastern steppe, winter grazing affected 5 out of the 57 vegetation and soils variables that we measured. Unlike other ecological zones, grass was more abundant and forbs less abundant close to winter shelters. More than 60% of the grass cover was made up of drought tolerant *Stipa krylovii* and rhizomatous *Elymus chinensis*, although we did not find statistical significance along the grazing gradient, there was a trend in higher cover of both species in more intensely grazed pastures close to winter shelters. Rather, the greater abundance of grass far from winter shelters was made up of the less abundant bunch grass species, *Cleistogenes squarrosa* and *Agropyron cristatum*. These are species can increase in abundance in response to heavier grazing pressure (Zhang et al. 2014). Liu et al (2013) suggested that the abundant fires in the eastern steppe might be causing changes in grazing management. These fires would burn more intensely in the less grazed vegetation far from winter shelters, perhaps causing this unusual pattern of less grass in highly grazed pastures. We can only speculate about the grazing intensity in these winter pastures, since we did not measure dung abundance in these plots. But the relative abundance of cover of *C. squarrosa* and *A. cristatum* to the total grass cover was low, 11.8% together. This is compared to the 63% of total grass cover of both the rhizomatous, increaser

species *Elymus chinensis* and the drought and grazing tolerant *Stipa krylovii*. Low grass cover far from winter shelters may also explain why gaps between perennial plants were larger in those locations.

Steppe

Our results from the steppe zone showed the largest winter grazing-induced changes of any ecological zone in our study. Lower grass cover and higher forb and annual unpalatable cover in the heavily grazed areas near winter shelters indicate strong effects of grazing, which is supported by high dung density in these locations. These findings are similar to those of previous studies conducted in Mongolia (Tserendash and Erdenebaater 1993, Fernandez-Gimenez and Allen-Diaz 1999, Sasaki et al. 2008b). However, the interpretation of changes in grass cover and grass standing crop biomass was confounded by a significant interaction between grazing and ecological site. Grass cover was lower near than far from winter shelters in the sandy and high water table ecological sites. Intense grazing and livestock trampling close to winter shelters increased susceptibility of sandy soils to the erosion and loss of protective grass cover. Soil surface disturbance by livestock trampling close to winter shelters could break up soil aggregates and compact soils on the fine textured high water table ecological sites, thus reducing infiltration rates and increasing evaporative losses and contributed to lower grass cover.

Perennial species were replaced by annuals, creating open gaps in intensely grazed sites near winter shelter (Figure 2.12c). But soil surface characteristics indicate that soil loss and erosion have not occurred. Abundant annual weeds, especially *Chenopodium album*, close to winter shelter may contribute to high crude protein (Marten and Andersen 1975) near than far from winter shelters.

Overall, winter grazing affected coarse, medium and finer resolution vegetation variables and created larger gaps in the vegetation, indicating moderate grazing-induced changes in the steppe with decreasing distance from winter shelters. This may suggest that grazing is starting to push these sites towards a tipping point of over-use, as suggested by Khishigbayar et al (2015). But it is unclear if these pastures are degraded now; only experimental removal of grazing could answer this question.

Desert steppe

In the desert steppe zone, we found no winter grazing-induced changes in broad resolution variables, like total biomass and cover, some subtle changes in the medium resolution variables, like functional group cover, and very few changes in finer resolution variables, like species cover. There was less grass, shrub and litter cover and more total and annual forb cover in the heavily grazed areas near winter shelters, similar to the patterns observed in the steppe ecological zone.

Winter grazing did not affect the palatability of plants except growing season unpalatable annual forbs, which were abundant, and growing season palatable perennial forbs, which were less abundant near the winter shelters. Greater cover of unpalatable annual plants close to winter shelters is consistent with high livestock disturbance. Dormant season unpalatable annuals did not differ along the grazing gradient. First, most plants are withered by the winter when grazing occurs and second, there may have been changes in chemical compounds in the standing dead, turning them into available food for livestock during the cold, long dormant season. The increase in palatable perennial forbs with distance from winter shelter is explained by greater cover of onions, *Allium polyrrhizum* and *Allium mongolicum* along the grazing gradient. There was greater acid detergent fiber (ADF) in vegetation samples in the little grazed areas far from winter

shelters and this could be attributed that greater grass standing crop biomass farther from winter shelters. Grass cell walls have abundant fiber composition (Heitschmidt and Stuth 1991).

The results from desert steppe suggest that there have been some subtle grazing-induced vegetation along the grazing gradient, yet primary production appears to be intact and the soil surface appears to be protected from soil loss and erosion. It does not appear that these pastures are degraded at this time.

Thus, are Mongolian rangeland degraded? In our study of winter pastures, we find only moderate evidence of grazing effects on winter pasture areas of Mongolia, with little effects in the mountain and forest steppe and the desert steppe. If our plots are representative of all winter pasture areas of Mongolia, this may suggests that these pastures that make up a third of all Mongolian rangelands are relatively healthy.

2.4.3. Effects of ecological sites

In Mongolia, the ecological site concept is comparatively new and there are attempts to classify ecological sites in some regions. We classified ecological sites after sampling by using the information we collected in the field including landform, slope, aspect and soil. Vegetation was not used as a primary ecological site criterion, because vegetation can be manipulated easily by natural and human-caused disturbances and defining ecological sites by vegetation would not allow testing the effects of these disturbances.

We found more significance in the ecological site effect than the grazing in the mountain and forest steppe and steppe with less significance in the desert steppe zone. Thus, ecological site was more important than grazing as a determinant of vegetation characteristics in the mountain

and forest steppe zone and was important for steppe and least important in the desert steppe ecological zone.

In the mountain and forest steppe, total foliar cover and the cover of sedge and perennial plants and sedge standing crop biomass were lower in rocky hill shallow sites than in the other ecological sites. This is likely caused by the lower productivity of the soils in these sites. High water table sites, with fine soils, supported 72% cover of the grazing tolerant sedge, *Carex duriuscula* (Hilbig 1995). Our field observations suggest that these sites sustain heavier grazing, since they are more productive and flatter. This is similar to findings of other studies, where the Mongolian botanist, Dashnyam (1974) showed that the disturbance indicator species *C. duriuscula* increases in depressions and lowland areas with clayey soils due to increased grazing pressure. This is also similar to the findings of Kakinuma et al. (2014) (Dashnyam 1974, Kakinuma et al. 2014).

In the steppe ecological zone, perennial plant basal cover, forb biomass and species richness were higher in rocky hill shallow sites than high water table ecological site, opposite the pattern in the mountain and forest steppe where total foliar cover, perennial plant cover sedge biomass were greater in high water table site than the rocky hill shallow site. Steppe sites are much less rocky and steep than the mountain and forest steppe, which may lead these sites to be relatively more productive. Here again, relatively higher winter grazing pressure on flatter, high water table sites may cause the lower amount of grass. Livestock trampling on high water table sites with fine soils, could also change water regime factors such as infiltration and percolation (Zemmerich et al. 2010) thus causing less forb biomass, and lower species richness than in rocky hill shallow sites. Our high water table sites also often support halophytic plants. Sasaki et al. (2008) showed that grazing impact on vegetation floristic and functional composition on

salinized soils is almost negligible, even though trampling removes significant vegetation in these areas.

In the desert steppe, we compared high water table sites to a combination of all other sites and found few differences. In this zone, ecological site apparently has little influence on vegetation, although our lumping technique may have influenced these results.

2.5. Implications and future research

2.5.1. Implications for equilibrium and non-equilibrium dynamics

An understanding of the consequences of equilibrium and non-equilibrium dynamics for rangeland change and policy development in Mongolia is crucial. Equilibrium rangeland dynamics emphasize the impacts of livestock is a major driver of rangeland vegetation change in systems with these dynamics. Increased livestock grazing in equilibrium rangelands leads to decreases in perennial grasses and increases in weedy annuals (Dyksterhuis 1949, Fernandez-Gimenez and Allen-Diaz 1999, Cheng et al. 2011). Rangeland condition is reversible upon removal of grazing, thus range management centers on estimating carrying capacity, controlling stocking rates in relation to carrying capacity, and monitoring range condition (Vetter 2005). In contrast to the conventional equilibrium rangeland dynamics, non-equilibrium rangeland dynamics predict that stochastic rainfall events override livestock grazing impacts in semi-arid and arid rangelands. Thus, grazing-induced degradation is low in rangelands with relatively variable precipitation (Ellis and Swift 1988). Most of the studies conducted in Mongolia that have tested non-equilibrium rangeland dynamics in dry regions suggest that these systems are largely driven by climate as predicted (Fernandez-Gimenez and Allen-Diaz 1999, Stumpp et al. 2005, Zemmrich 2007, Cheng et al. 2011). However, a 7 year-long enclosure experiment

conducted in the desert steppe showed slight but significant differences between grazed and un-grazed treatments (Wesche et al. 2010). Recent studies suggest that equilibrium and non-equilibrium are extremes along a continuum and that many systems encompass both (Briske et al. 2003, Vetter 2005).

We expected that wetter and less climatically variable, mountain and forest steppe sites would display characteristics of equilibrium rangeland dynamics, eastern steppe and steppe zones would display mixed equilibrium and non-equilibrium characteristics, while desert steppe would exhibit non-equilibrium dynamics. In the mountain and forest steppe, contrary to expectations, there were little impacts of grazing on vegetation, perhaps because of the flat grazing gradients described above. In this zone, village (*soum*) level livestock density is high and compared to other ecological zones, many households hold livestock and live at same winter shelter. Also the distance between winter and summer pastures is less, so there are more opportunities for out of season grazing (Ulambayar 2015). Our team's social data show that about four herder families live at each winter shelter we sampled (although we have only the livestock number of households sampled that have official certificates to use the winter shelter). More families live at the winter shelters in the mountain and forest steppe than any of the other ecological zones. Thus, we think the flat grazing gradient we see in this zone may be caused by overlapping sheep, goat, cow and yak grazing orbits.

Our results from the steppe zone showed that with increasing grazing pressure, grasses decrease (especially *Stipa krylovii* $p=0.001$) and forbs and unpalatable annual forbs increase especially weedy disturbance indicator species like *Chenopodium album*. This suggests that grazing plays a more important role in determining vegetation composition in this zone

compared to other zones. Thus, the pattern of vegetation change in the steppe zone fits classic equilibrium dynamics.

Unlike Fernandez-Gimenez and Allen-Diaz (1999), we found that vegetation dynamics in the arid desert steppe of southern Mongolia responded slightly to grazing in an equilibrical way where perennial grass decreased and unpalatable annual forb increased with increasing grazing pressure. Recent researchers are attempting to clarify the applicability of non-equilibrium rangeland dynamics and suggesting the importance and relevance of temporal and spatial scales (Briske et al. 2003, Oba et al. 2003, Vetter 2005, Zemmrich 2007) and consistent variables/indicators to measure quantitatively whether the model is supported or refuted (Fernandez-Gimenez and Allen-Diaz 1999, Briske et al. 2003, von Wehrden et al. 2012). Our findings from the desert steppe support the findings of Oba et al (2003) and Zemmrich (2007) that suggest that grazing impacts become evident as spatial scale decreases in the arid and semi-arid areas. Zemmrich and colleagues, working in the desert steppe of western Mongolia, found that it was difficult to detect grazing effects at a landscape scale, because they are masked by variations in soils, but they found marginally statistically significant correlations between grazing intensity and vegetation parameters at the community level. At the population level of *Artemisia xerophytica*, their results clearly reflect the equilibrical grazing effects with greater effects closer to winter shelters with high livestock activities. Also, desert steppe is a mosaic of different ecological sites; the wetter sites may exhibit more equilibrium dynamics than others. That may explain why we found grazing effects in the wet, high water table sites (which constituted of 78% of ecological sites) within the desert steppe (Kakinuma et al. 2013).

Overall, 14.7% of vegetation variables we tested (11 out of 75) displayed statistically significant changes along the grazing gradient in the mountain and forest steppe. This included

greater cover of total forb, perennial forb, growing season palatable perennial forb, both growing and dormant season unpalatable perennial forb shrub, litter growing season perennial grass and *Agropyron cristatum*, less total sedge, *Carex duriuscula*, near than far from winter shelter. . In the eastern steppe, 9 (14.7%) variables out of 61 variables displayed statistically significant changes along the grazing gradient. There were greater total grass cover especially perennial grass, less forb biomass, total forb cover especially perennial forb and average gap between perennial plant bases, cover of *Cleistogenes squarrosa*, both growing and dormant season unpalatable perennial forb near than far from winter shelters. In the steppe, , 25 (38.5%) variables out of 65 variables displayed significant changes along the grazing gradient. This included the cover of total foliar, both perennial and annual grasses, shrub, perennial plant and litter, standing crop biomass of grass, litter and standing dead were less near than far from winter shelter, but opposite was true for cover of annual plant, especially annual forb, both for total forb cover and biomass and crude protein. Three out of 5 dominant species including *Chenopodium album*, *Achnatherum splendens* and *Caragana stenophylla* were greater near than far from winter shelter, whereas 2 species including *Stipa krylovii* and *Caragana stenophylla* displayed the opposite pattern. There were greater average open gaps between perennial plant bases and less connectivity of persistent vascular plants and interpatch in the pastures close to winter shelter than the farther away. In the desert steppe, 14 (22.2%) variables out of 62 variables displayed statistically significant changes with increasing grazing pressure. This included less grass cover especially perennial grass and grass biomass, cover of shrub, litter and ADF, dominant species including *Allium mongolicum*, *Stipa gobica*, greater forb cover including both perennial and annual forb near than far from winter shelter. From our results conducted in four different ecological zones along the precipitation gradient, we can conclude that rangelands in

Mongolia occur along a continuum of equilibrium and non-equilibrium rangeland dynamics. An important question for rangeland management is, when should we apply equilibrium and non-equilibrium rangeland dynamics for our spatially heterogeneous and temporally variable rangelands in order to disentangle selective grazing effects from non-selective intermittent climate effects? Thus, specific variables chosen and spatial and temporal scale of study influence for making inferences on rangeland dynamics and following the management.

2.5.2. Implications for management and policy

Our study shows that there are moderate grazing-induced changes in winter pastures in the steppe and more subtle changes in the desert steppe and quite slight changes in the mountain and forest steppe. In the mountain and forest steppe, because the grazing gradients seem to be flat, it is not possible to conclude that grazing has no effect, only that our grazing gradient was weak. For the steppe and the desert steppe, these grazing-induced changes are not strong, implying that our sampled winter pastures are relatively healthy. If our results represent winter pastures across Mongolia, which may represent about a third of all Mongolian rangelands, then degradation here does not appear widespread. This finding seems to contradict the widely cited (in the non-peer reviewed academic publications and NGO reports) statistic that about 70% of Mongolian rangelands are degraded and the main cause is livestock overgrazing. But maybe most of the degradation is on the other 70% of rangelands that grazed in the spring, summer and fall.

At the moment degradation is highly subjective concept among researchers in Mongolia and it is very important to consider the inherent potential of rangelands and rangeland ecosystem dynamics including equilibrium and non-equilibrium dynamics in developing a set of indicators

to detect degradation. It is important to consider fluctuations of number and type of livestock that graze on the rangelands and inter-annual variability in precipitation in the arid and semi-arid non-equilibrium rangelands that drive fluctuations in vegetation parameters. Thus there is a need for a commonly agreed upon and clear definition of degradation and identification of a set of variables to distinguish between reversible and potentially temporary changes in biotic communities and degraded or irreversible, permanent changes or loss of productive potential associated with soil loss and soil chemistry and hydrology of the system (Khishigbayar et al. 2015). It is also important to note that degradation effects can accumulate over time with continuous heavy grazing, and grazing interacts with climate. These both could lead the system to suddenly shift or cross over a threshold before management decisions can be made.

Thus, consistent and long-term monitoring by a team of government personnel (land manager and climate station worker), NGOs and both experienced and young herders at the district level is needed to detect and prevent rangelands from irreversible degradation. Monitoring indicators should include both slow and rapidly changing vegetation, soil and climate (temperature, precipitation) variables at coarse, medium and fine resolutions that are sensitive enough to detect the possible changes due to grazing and climate. Also some estimate of actual grazing pressure (utilization and actual grazing densities) will be essential and practical in understanding and interpreting the changes that are occurring in rangelands. This is the government's role to support and ensure the continuity of these monitoring activities with rangeland scientists involved throughout the monitoring process. The results of this ground level monitoring activity will be very essential for evidence-based management in the future and for policy development.

2.5.3. Future research

We found that natural grazing gradients or piospheres have limitations in detecting the effects of grazing on Mongolian rangelands. In some instances (mountain and forest steppe) the gradients were weak and thus grazing effects hard to detect, and in other instances (eastern steppe) fire may be more important than grazing. Thus, there is a need to conduct widespread, experimental studies of grazing across ecological zones. There is some research needed to understand future grazing-induced changes and interaction of these changes with changing climate. Our study focused on winter grazing pastures, but there is a need for more focus on summer (and spring and fall) pastures. Well replicated long-term exclosure studies in summer, spring and fall grazing areas will help us to understand the level of current-season grazing and livestock impacts on rangelands. Long-term exclosures placed at different grazing intensities with enough replicates in different ecological zones will allow not only study of grazing effects but also separation of the effects of grazing and climate across ecological zones. In our interviews with herders (Chantsallkham. J, unpublished data) in all ecological zones, they almost all agree that summer pasture conditions are deteriorating, thus it will be very important to involve herders from the very beginning in the designing new studies. Second, manipulative controlled experiments of the grazing effects of different livestock at different grazing intensities on rangelands in different ecological zones will allow us to understand the vegetation response of coarse to medium to fine resolution variables and application of equilibrium and non-equilibrium rangeland concepts.

CHAPTER THREE: RANGELAND CHANGES AND RESILIENCE OF MONGOLIAN RANGELANDS TO LIVESTOCK GRAZING OVER TIME

3.1. Introduction

Mongolia's steppes are one of the largest intact temperate grasslands in the world (Asner et al. 2004), and directly support the livelihoods of one third of all Mongolians, and indirectly more than half of the population (Mongolian Society for Range Management 2010). Currently, Mongolian rangelands are changing because of changing climate (MARCC 2009), and possibly from changing livestock management (Hilbig 1995, NSO 2013, Khishigbayar et al. 2015). To sustain these vast grasslands, we require a deeper understanding of ecosystem response to grazing, how climate modifies this response, the resilience of the grasslands to current and future stress, and how best to monitor these responses to stress.

Livestock populations have fluctuated strongly since the transition to a market economy and privatization of formerly state-owned livestock in 1992 (NSO 2013), livestock populations increased rapidly until 1999 and during 3 years of consecutive drought and *dzud* (extremely cold and snowy winters) from 1999 and 2002, the national herd declined by 30% and recovered and peaked (40 million) again by 2009. During the harsh winter of 2009-2010, livestock populations dropped again by 20%. Since then, livestock numbers have recovered and reached 51 million head as of December 2013³. In this study, we consider these fluctuations as a stressor, and measure the resilience of grasslands to this stress.

Climate is also changing in Mongolia and affecting these grasslands. Temperatures and winter precipitation are increasing and summer precipitation is falling (MARCC 2009,

³ Note that these increases in numbers of livestock likely mean changes in livestock grazing intensity, but because of shifts in the type of livestock during this time, this is not entirely clear. In this study, we will convert these livestock numbers to standard sheep forage units (SFU's) to better approximate grazing intensity.

Mongolian Society for Range Management 2010). Local people report that rangeland biomass and production is declining (Bruegger et al. 2014), similar to field studies (Sankey et al. 2009) and remote sensing studies (Liu et al. 2013) conducted in different parts of Mongolia.

Recent studies suggest that both livestock and climate change have strong effects on Mongolian grasslands (Hilker et al. 2013, Liu et al. 2013, Khishigbayar et al. 2015). The equilibrium/nonequilibrium theory of rangeland dynamics would suggest that the effects of grazing should be stronger than climate in wetter, northern zones of Mongolia with equilibrium dynamics, while the effects of climate should be stronger than grazing in the drier, southern zones of Mongolia with non-equilibrium dynamics (Ellis and Swift 1988, Fernandez-Gimenez and Allen-Diaz 1999, von Wehrden et al. 2012).

These studies assume that the density of livestock, averaged over broad regions, is a good proxy for grazing. In this chapter, grazing includes livestock trampling and urine and feces defecation besides actual grazing. This approach first requires the conversion of livestock density of several species into the standard index of sheep forage units (SFUs) which is not always done. Second, there is a need for careful study of changes in vegetation productivity over time in pastures that have experienced different levels of grazing for extended periods of time. The design of such studies is important, and can use either a manipulative or natural experiment design. The manipulative experiment design would compare fenced or exclosed pastures that are not grazed with those that are grazed at different levels. This type of research will allow teasing apart climate effect and grazing effect and their relative contribution for the changes over time. Alternatively, the natural experiment approach would use pastures only used in the non-growing season, which are effectively, by common pastoral practice, 'exclosed' during the summer, and arraying measurements along a grazing intensity gradient. These designs allow us to assess the

long-term effects of grazing on winter pastures, but not the long-term effects of summer season grazing. We used the second approach in this study, by measuring pastures around winter shelters of pastoral families, which are usually used every winter over many years, but are not grazed during the summers (when our data were collected) to reserve the pastures for grazing for the next winter.

In addition, to our knowledge, few studies have measured how grazing affects the ability of grasslands to recover after climate stress (drought, *dzud*). Such studies give a measure of the ecological resilience and inertia of these grasslands in response to change. Ecological resilience is the ability of the system to return to its pre-stress or pre-disturbance level and maintain the structure and function of the system after this perturbation (Holling 1973, Gunderson 2000). Westman (1978) goes further to characterize resilience by different measures that include elasticity (rebound speed), malleability (stickiness in new state) and damping (of oscillations). Inertia (or resistance) is the ‘ability of a system to resist displacement in structure or function when subjected to a disturbing force’ (p. 705, Westman 1978). This can be considered part of (Holling 1973) or separate from the term ‘resilience’ (Westman 1978). Here, we measure the resilience characteristic, elasticity, by comparing the speed of rebound in vegetation productivity after disaster in adjacent pastures with different levels of grazing (faster rebound = more resilient); we measure inertia or resistance by the variability of vegetation in response to stress in these same pastures (less variable = more inertia or resistance).

We also need to understand how to best monitor the resilience of these grasslands in response to grazing. Ground measurements are costly; remotely sensed measures may provide a reliable alternative in the absence of repetitive on-the-ground measures of vegetation biomass and cover. Remote sensing also has the advantage of continuous spatial coverage vs discrete

point locations for on the ground monitoring (Butt 2010). Some remote sensing data, like MODIS, are collected frequently and are free, providing a low cost option for monitoring. The most common remotely sensed vegetation indicator is the Normalized Difference Vegetation Index (NDVI), sometimes called a greenness index, which is used as a proxy of vegetation productivity and vigor (Tucker 1979, Lillesand 2008). These data can be compared to field measurements of green biomass on grazing gradients to understand how these different data can be used to assess the effects of grazing in grasslands.

The overarching goal in this study is to understand how Mongolian rangelands grazed at different intensities have changed recently, how they respond to stress, and how to best measure this change across a range of different ecological regions, from the deserts in the south of the country to the forests and mountains in the north. We do this by assessing change and response to disasters along a grazing gradient from intensively grazed pastures near herder winter shelters to lightly grazed pastures far from these same shelters. We combine multi-temporal NDVI data with a ground-based vegetation study to contribute to our understanding of changes in Mongolian rangelands by answering the following questions:

Research questions

1. As seen in chapter 2, we carefully selected small-scale field plots along grazing gradients, controlling for soils and landform (or ecological sites). If we overlay coarser resolution MODIS remote sensing data for the same season on these plots, do they show the same patterns of vegetative response to grazing? (How strong is the relationship between field measures of vegetation cover and biomass and remote sensing measures of NDVI?)

2. Are the grazing gradients we found in Chapter 2 (based on field sampling in 2011 and 2012) maintained over time or are they present only in certain years between 2000 and 2013?
3. How do the trends in vegetative greenness (NDVI) compare in pastures grazed at different intensities from 2000 to 2013?
4. Do these grazing effects differ over time by ecological zone? Is precipitation or grazing a better predictor of NDVI over time?
5. How resilient are these rangelands to stress? How does grazing influence resilience to stress? Does vegetation greenness, as measured by NDVI, in heavily grazed pastures near winter shelters recover more slowly after *dzud*/drought than vegetation in lightly grazed pastures far from winter shelters?

Research hypotheses

1. The remotely sensed NDVI data along the grazing gradient will show the same effects of grazing as the field vegetation data sampled in the same season and in the same locations.
2. The grazing gradient will be maintained over time and will become stronger in years when livestock densities (SFU/km²) in surrounding pastures are high, and weaker when livestock densities in the surrounding pastures are lower.
3. The grazing gradient will be strongest in the mountain and forest steppe, moderate in the steppe and weakest in the desert steppe following the predictions of equilibrium and non-equilibrium rangeland dynamics.
4. NDVI at more heavily grazed pastures near winter shelters will recover more slowly (be less resilient or less elastic) and vary more (have more inertia or be less resistant) after *dzud*/drought than more lightly grazed vegetation far from winter shelters.

5. Grazing will have strongest effects on NDVI in the wetter compared to drier zones, and climate will have the opposite pattern. This means that grazing will have its strongest effects on NDVI in the mountain and forest steppe, moderate effects in the steppe and weakest effects in the desert steppe.

3.2. Methods

3.2.1. Study Areas

We sampled vegetation using field and remotely sensed data in pastures grazed at different intensities around 8 winter shelters in 2 *soums* (counties) in the mountain and forest steppe, 9 winter shelters in 2 *soums* in the steppe and 9 winter shelters in 2 *soums* in the desert steppe. Most of the study sites in the mountain and forest steppe and steppe zones were located in areas with mountainous terrain, whereas in the desert steppe the areas were generally flat. Annual rainfall averages between 120-270 mm from desert steppe to mountain and forest steppe, the coefficient of variation of precipitation is 21-36% along this gradient.

All study areas have been grazed by domestic livestock under nomadic and transhumance pastoral use for at least 1000 years and possibly for several millennia (Johnson et al. 2006). The main livestock types are cow/yaks, horse, sheep and goats in the mountain and forest steppe; sheep, goats, horses, cow/yaks and camels in the steppe; and goats, sheep, camels, horses and cow in the desert steppe. Within each ecological zone, herders move seasonally with their livestock through a series of seasonal pastures. Herder families spend summers in the vicinity of water sources (rivers, lakes, and water wells), move to fall camps, often in the open steppe, and then spend winters in sheltered places, facing south, usually locating their winter shelters on the warmer, leeward side of mountains or hills. In the spring, families move to lower more open

areas (Fernandez-Gimenez 1999). Traditionally, herders graze different pastures in the four seasons, particularly avoiding summer grazing in winter grazing grounds to reserve critical pastures for winter grazing (Banzragch and Davaajamts 1970). These ‘reserved’ winter grazing lands cover about one third to one half of the annual grazing orbit (ALAGAC 2010).

3.2.2. Sampling methods and data processing

To test the effects of grazing on vegetation we used a grazing gradient approach, where we sampled in heavily grazed pastures near winter shelters compared with more lightly grazed pastures farther from shelters. We collected field and remote sensing data at 100m, 500m and 1000m from each of 26 winter shelters in the 3 ecological zones. Vegetation sampling was completed in July and August of 2011 and 2012. We compared these data for questions 1 and 2 with season-long MODIS NDVI data for the same years for the years 2000-2013.

3.2.2.1. Field data sampling

At each winter shelter, we sampled vegetation and soils in 3 plots located at 3 distances from the winter livestock shelter (100 m, 500 m and 1000 m), as measured from the gate of the livestock corral at the shelter, to examine the effects of livestock grazing pressure on vegetation. We selected these specific distances because of previous work showing that the effects of livestock grazing are greatest close to livestock grazing impact points such as livestock camps or water points and largely minimal farther than 1000m from the impact (Sasaki et al. 2008a, Sasaki et al. 2011). We selected plots at the three distances on about the same landform (e.g. hill, fan/piedmont, terrace, or plain), hillslope profile position (summit, shoulder, backslope, footslope, toeslope, etc.), aspect, and, if possible, soils. This allowed us to sample a grazing

gradient that falls on the same ecological site, which is defined by landform, soils, and climate, which potentially produces similar kinds and amounts of vegetation and responds similarly to natural disturbances, drivers and management (Bestelmeyer et al. 2009). Because winter shelters are often located at the heads of valleys, we selected sites along a gradient following a contour along the edge of the valley rather than dipping downslope onto different soil at the valley bottom. This meant that most sites were in specific upland positions, not in riparian areas of deep valley bottoms, unless the winter shelter was located in this landscape position. Thus, our sample is not a random sample of grazing effects in winter grazing areas, but rather the specific effects of grazing on the types of ecological sites selected by herders for their winter shelters. In this way, we avoided confounding distance from the winter shelter or ‘impact point’, intended as an index of grazing intensity, with a gradient in soil moisture or water table depth, and associated changes from riparian to upland vegetation. We selected the plots, with only a few exceptions, so that they were located at least as far away from any *other* livestock camps and water points, as they were from the focal impact point (winter shelter or water point). In other words, we made sure that the 1000-m plot was at least 1000-m away from any other livestock camps or water point as well as from the selected campsite. We did not use the appearance of the vegetation to select plots that are on similar ecological sites, only landform and soils, to limit confounding, since grazing can impact vegetation. An almost equal number of replicate plots were selected in each distance from winter shelters in each ecological zone. At our total of 26 winter shelters, we sampled 24 plots in mountain and forest steppe, 27 plots in steppe and 27 plots in desert steppe ecological zones.

Each plot consisted of a 50 x 50 m plot with 5 systematically spaced 50-m transects (Figure 2.2). Transects originated at the 0 point, 12.5, 25, 37.5 and 50 meters along the baseline.

If the plot was on a slope, we oriented it so that transects ran up the slope, to incorporate variability within each transect. The origin point (at 0 m) and baseline were always on the downhill side of the plot with transects running uphill from the baseline.

Standing crop biomass, if ungrazed, is an estimate of vegetation production at the site, which is an important indicator of the health of the site and its value for livestock production. We purposely sampled in winter grazing areas, where there was limited grazing during the growing season, so our measures of biomass largely quantified production and not grazing offtake in these natural ‘utilization cage’. We separated out the biomass samples by seven functional groups, including grasses, forbs, shrubs, sedges, litter, standing dead, and the large grass, *Acnatherum splendens*. Standing crop biomass of herbaceous plants was clipped in 5 quadrats in each plot at the base of the plant for grasses, forbs, sedges and *A.splendens*. We separated out *A.splendens* because it was very patchy and when present, usually produced very high biomass. For shrubs and subshrubs, we used the representative branch method and or collected shrub leaves and current year’s growth of twigs within at 3-D projection of the plot frame, regardless of whether the shrub was rooted inside or outside the frame (Bonham and Ahmed 1989). For litter and standing dead, we clipped or picked up all detached pieces inside the quadrat frame. We determined the size of the quadrat by the amount of biomass in the ecological zone, using a 50 x 50 cm quadrat in the mountain and forest steppe, eastern steppe and steppe, and 1 x 1 m quadrat in the desert steppe. All samples were dried in a drying oven at 60°C for 48 hours in the laboratory and then weighed to an accuracy of +/- 0.01 grams.

Plant foliar cover by species was measured using the line point intercept (LPI) method with points dropped every meter along each of the five 50-m transects for a total of 250 points per plot (Herrick et al. 2009). Plant functional type cover was calculated by adding species cover

within particular functional types and plant total cover was calculated by summing the cover of all functional groups. Foliar cover was measured as the area of ground covered by vegetation leaves. Small openings in the canopy and intraspecific overlap were excluded and thus foliar cover is always less than canopy cover, since the later sums up the overlap in different layers of the canopy. All nomenclature in this study follows Grubov (1982).

3.2.2.2. Remote sensing data sampling and processing

For all our questions, we used MODIS Normalized Difference Vegetation Index (NDVI) data at 250 m spatial resolution (Mod13a1) from 7 MODIS tiles for the years between 2000 and 2013. NDVI is calculated from red and near-infra-red spectral reflectance measurements which determine the amount of absorbed and reflected photosynthetically active radiation from the density of chlorophyll in leaf structures and density of green leaves (Lillesand 2008). For question 1 and 2, we used only data from 2011 and 2012 to match the years we collected field data, for the other questions we used the entire time series of MODIS data. For all questions, we used the TIMESAT software (Eklundh and Jonsson 2009) to calculate the integrated NDVI (iNDVI) for each of the 14 years between 2000-2013. Integrated NDVI (iNDVI) is a proxy for vegetation biomass. TIMESAT calculates a small integral NDVI value, which is the current, whole growing season greenness; it also calculates a large integral NDVI value which is the current, whole season greenness plus the residual greenness from previous years. Here, we call the small integral ‘current season NDVI’ (abbreviated as CS NDVI) and the large integral ‘current and previous year NDVI’ (abbreviated as CPS NDVI).

We overlaid each MODIS 250m x 250m grid cell (pixel) on top of Google earth map accessed for late May, 2013, where our 3 study plots and winter shelter were located. During

field work we visited every candidate 250 x 250 pixels that overlaid each 50 x 50 m plot and decided if the candidate pixel was sufficiently similar to the 50 x 50 plot to make a good comparison. The criteria for pixel selection included, first, biophysical similarity, where we excluded several pixels because they were located on surrounding hills while the actual 50 x 50 plots were located on sloping alluvial fans between the hills. Thus sometimes we selected an adjacent pixel that made a better comparator, which still had the same grazing intensity as the target pixel. Second, grazing intensity, where we maintain the distance from winter shelter. For example, we excluded several pixels that fell between 500m and 1000m plots because they fell at the wrong distance from the winter shelter.

3.2.3. Data analysis

Data were analyzed with the statistical package SAS 9.3 for Windows. We corrected the non-normality of the data using a log (y+1) transformation for plant biomass and arcsine transformation for plant cover data. To test if field and remote sensing data can be used interchangeably to assess the effects of grazing on vegetation (question 1), we ran regressions on: 1) the CPS NDVI (current and previous season NDVI) and total plant biomass, 2) the CS NDVI (current season NDVI) and total plant cover and total green biomass (separately) in each ecological zone. We did not separate out the data by different distances along the grazing gradient in this analysis. Also we created two new variables that included both sources of data (field and NDVI); total cover, total green biomass and total biomass as source=field and only current season NDVI and current + previous season NDVI as type=NDVI. In order to compare if these two sources of data display similar patterns across the grazing distances we tested for an interaction between source and grazing distance. NDVI (CS and CPS) data were scaled to have

variances similar to the field data. A randomized block 2-way ANOVA was run with source, distance and the source by distance interaction as fixed effects. Winter shelter (*uvuljuu* in the Mongolian language) was included as a random block effect and distance was included in the model as a repeated measures effect to allow for spatial correlation. For questions 3-5, we used only NDVI data from 2000-2013, with a similar model as above, where 2-way ANOVA was run with year, distance and the year by distance interaction as fixed effects and added precipitation as a covariate in the model.

For question 4, we used visual interpretation of several sources of data to understand the relative effects of growing season precipitation and livestock density/percent forage use on pastures. We then compared these changes to the total growing season precipitation and the *soum*-level livestock density (SFU/ha) and percent forage use. Percent forage use can be used to estimate grazing intensity over time. Percent forage use from 2000 to 2013 was created by calculating the ratio of forage demand ($\text{SFU ha}^{-1} \text{ year}^{-1}$) to available forage (Gao et al. 2015). Forage availability was estimated by using a linear regression relationship between herbaceous biomass and the 250m MODIS NDVI (J. Angerer, pers comm, (Khishigbayar et al. 2015).

It is important to distinguish our two measures of grazing at two different scales. First, at the winter shelter scale, our NDVI values are from pixels falling along a grazing gradient, which measures places with different (but unmeasured) levels of grazing around the winter camps. Second, at the *soum* scale, our livestock and forage use data measure the actual, broad-scale grazing intensity across each *soum*, which we assume to apply to the particular winter shelter sampled with the NDVI data.

To measure the resistance and resilience of pastures in response to grazing (question 5), we used two simple measures. For resistance, we used the coefficient of variation (standard

deviation/mean*100) over the 14 years from 2000-2013 as a proxy for the resistance (or inertia) of vegetation to change, with greater resistance indicated by a lower coefficient of variation of NDVI. We then measured the speed of recovery of NDVI (Tilman and Downing 1994, Harris et al. 2014) after disturbance (low rainfall or heavy grazing) as a proxy for the resilience (elasticity) of pastures grazed at different intensities. We did this for each distance at each winter camp by selecting out all annual sequences of NDVI that contained declining NDVI and counted the number of years that elapsed before the NDVI returned to the same level as before the decline in NDVI began. Here, we define more resilient pastures as those that recover more rapidly after low rainfall periods than less resilient pastures. We expect CS NDVI to be a better indicator of resilience by recovering more rapidly and to greater magnitude than CPS NDVI.

In the desert steppe, due to erratic rainfall events, the TIMESAT program was not able to estimate a growth curve because the NDVI signal was too erratic. Thus we have missing CS and CPS NDVI data for many of our grazing gradients in the desert steppe. For the calculation of question 5 we excluded the winter shelters that have missing values for more than 2 consecutive years. We included 3 out of 5 winter shelters in the Ulziit *soum* and one out of 4 winter shelters in the Undurshil *soum* in the desert steppe, so that only 4 grazing gradients were remaining in this part of the analysis.

3.3. Results

3.3.1. Relationship between field and remote sensing data

There are only moderate relationships between the plant cover and biomass data, measured in the field, and NDVI data, measured from satellite (Table 3.1). There was a positive linear relationship between current season NDVI (CS NDVI) and total cover in the mountain and

forest steppe ($p=0.004$, $r^2=0.32$), but there was no relationship between CS NDVI and total green biomass or CPS NDVI and total biomass. In the steppe, there was a positive relationship between CS NDVI and total green biomass ($p=0.01$, $r^2=0.19$), but no relationship between the other two variable pairs. In the desert steppe, we found no significant relationships between the remotely sensed and field data.

Table 3.1. Relationship (p-value and r^2) of field and remotely sensed variables of vegetation at each grazing distance across 3 ecological zones in Mongolia.

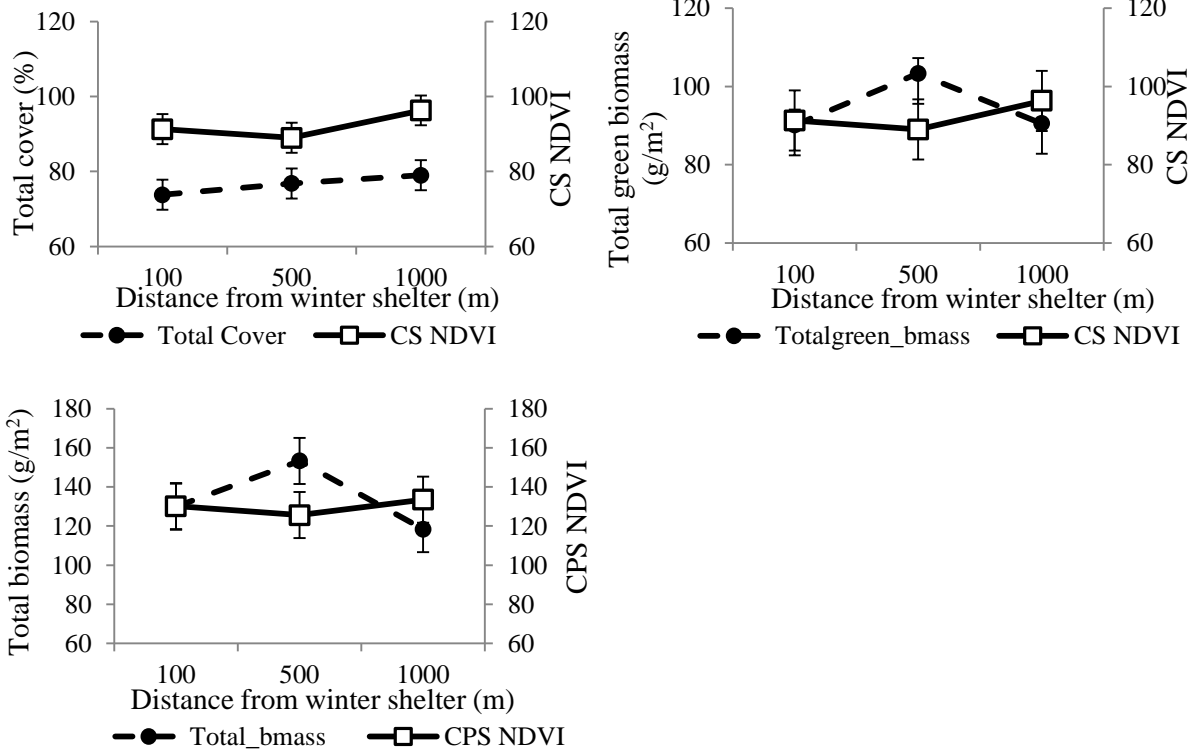
Field variable	Remote sensing variable	Mountain and Forest Steppe	Steppe	Desert steppe
Total green biomass	CS NDVI	$p=0.448$, $r^2=0.03$	$p=0.012$, $r^2=0.19$	$p=0.458$, $r^2=0.02$
Total cover	CS NDVI	$p=0.004$, $r^2=0.32$	$p=0.192$, $r^2=0.05$	$p=0.101$, $r^2=0.10$
Total biomass	CPS NDVI	$p=0.413$, $r^2=0.03$	$p=0.053$, $r^2=0.12$	$p=0.549$, $r^2=0.01$

CS = Current season NDVI (= small integral in TIMESAT)

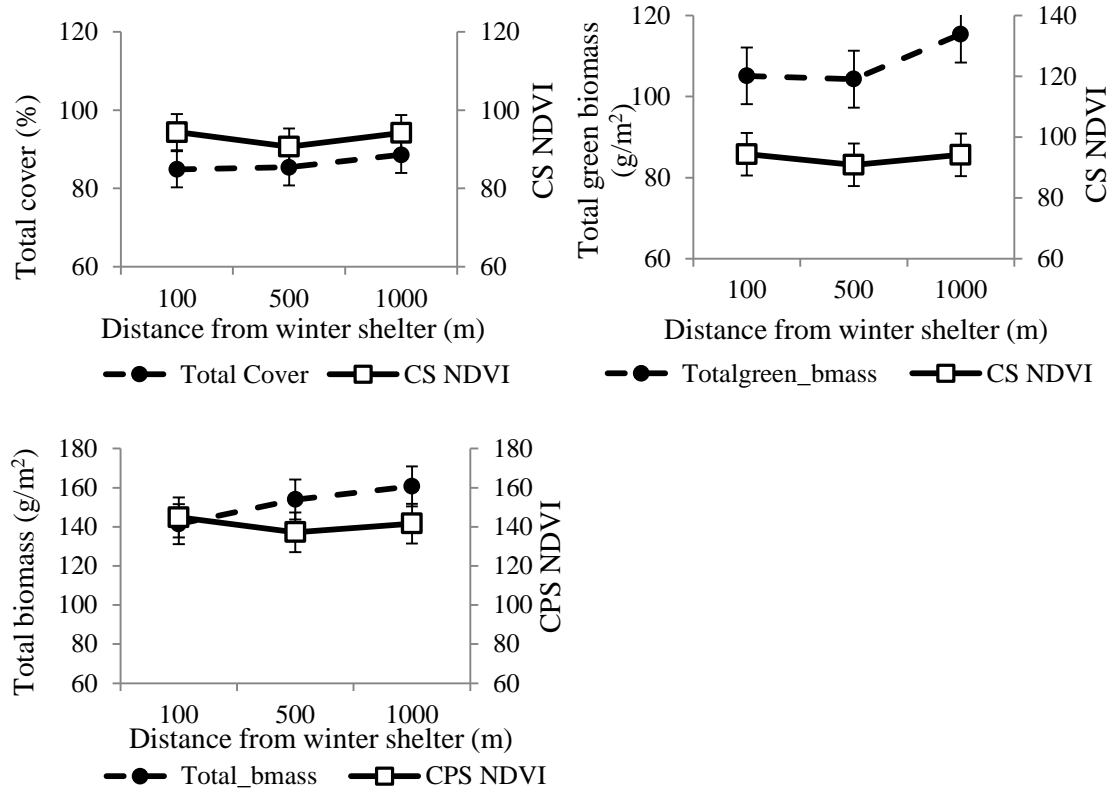
CPS = Current and previous season NDVI (= large integral in TIMESAT)

For question 2, we compared the significance of the same one-way ANOVA model for all dependent variables from the field and remote sensing, using distance to the winter shelter as the independent variable (100, 500 and 1000 m), which is a proxy for grazing intensity (Figure 3.1). For all ecological zones, none of the ANOVA models were significant, which suggests that field and remote sensing data show a similar response to grazing (see more information in the table in appendix 3.1).

a) Mountain and forest steppe



b) Steppe



c) Desert steppe

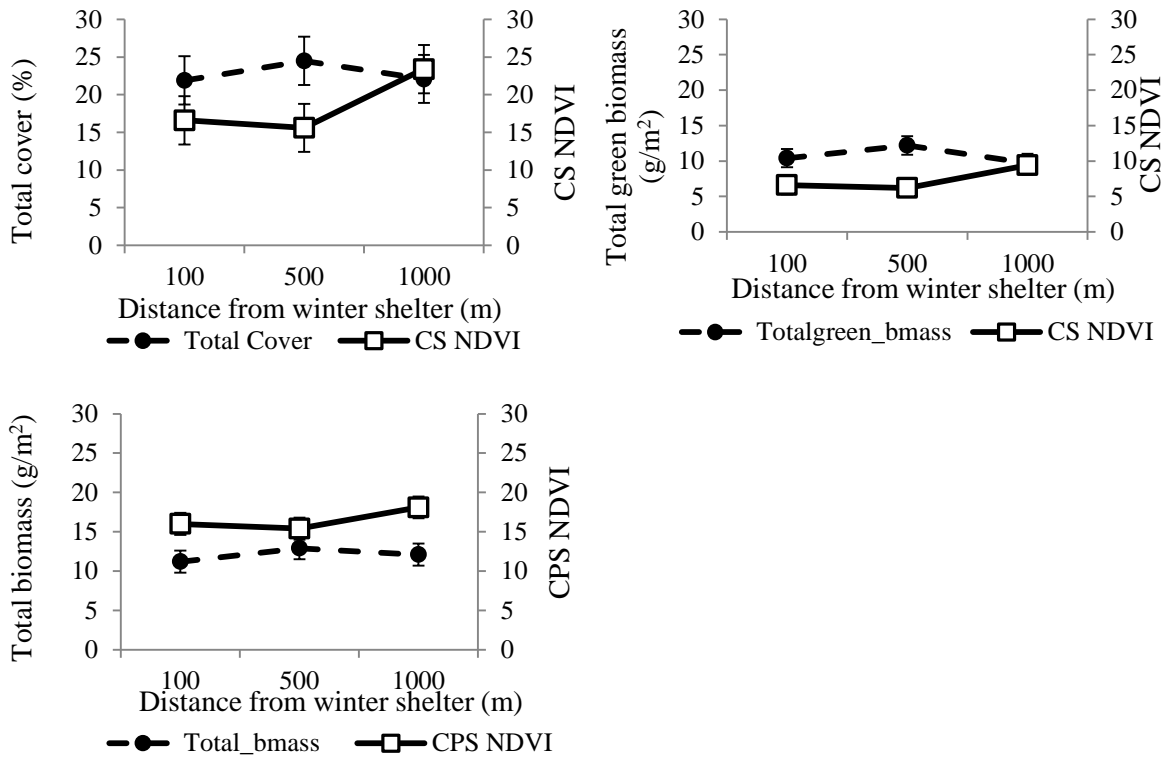


Figure 3.1. Comparison of field and remote sensing data along the three grazing distances from winter shelters in the a) mountain and forest steppe, b) steppe and c) desert steppe.

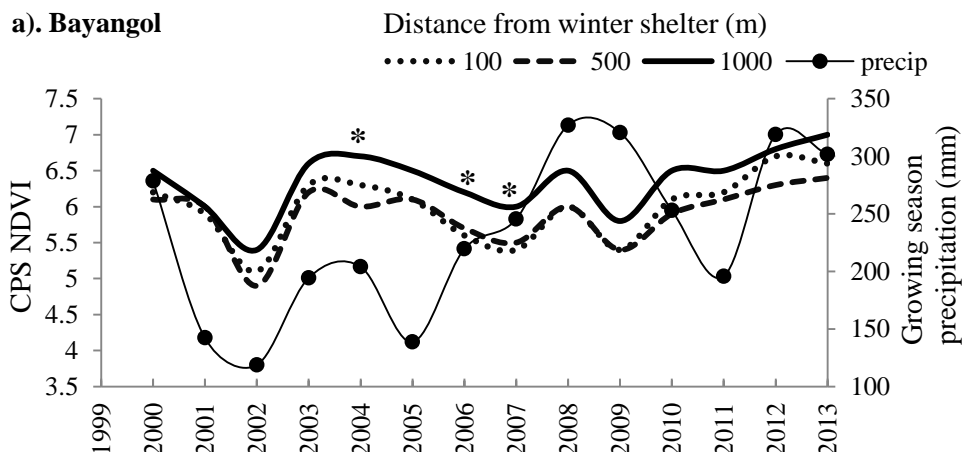
In the all zones we found no significant differences in total cover, total green and total biomass and remote sensing data (CS NDVI and CPS NDVI) among the 3 grazing distances (Figure 3.1c).

3.3.2. Relative effect of growing season precipitation and winter grazing on NDVI

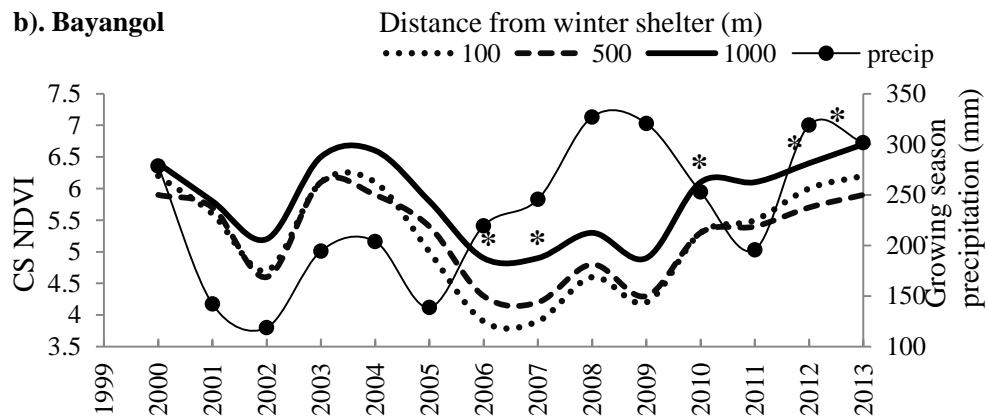
To answer research question 3, “How do the trends in vegetative greenness (NDVI) compare in pastures grazed at different intensities over time?”, we compared the differences in CS and CPS NDVI across our grazing gradient in each of 14 years from 2000-2013 and looked at the consistency in their patterns (Figures 3.2 to 3.4). In general, grazing had stronger effects on CS and CPS NDVI in some years than others over the last 14 years. Also note that the y-axes

in the following figures are sometimes different in different *soums* to allow the patterns in NDVI to be seen.

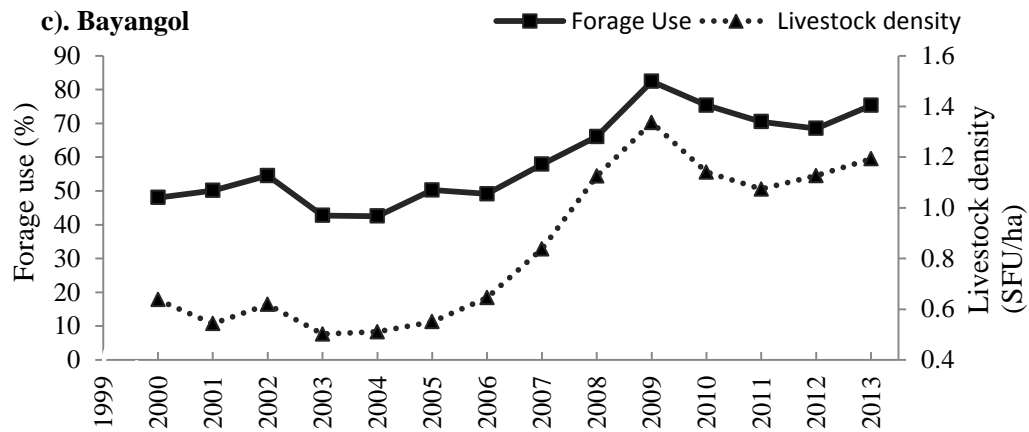
Grazing did not affect NDVI consistently from year to year in the last 14 years; the grazing effect was strong and significant in some years and was not different in other years. In Bayangol *soum* in the mountain and forest steppe, CPS NDVI was significantly greater in lightly (1000m) grazed than moderately (500m) grazed pastures in 3 out of 14 years ($p=0.02$ in 2004, $p=0.04$ in 2006, $p=0.05$ in 2007; Figure 3.2a). These were years with from 204-245mm of growing season rainfall (Figure 3.2a) and 42-58% forage use (Figure 3.2c). There were significant differences in CS NDVI along the grazing gradients in 5 out of 14 years ($p=0.03$ in 2006, $p=0.01$ in 2007, $p=0.01$ in 2010, $p=0.05$ in 2012, $p=0.02$ in 2013; Figure 3.2b). These were years with 219-318 mm rainfall and 49-75% forage use (Figure 3.2a, c). In Saikhan *soum* in the mountain and forest steppe, both CS and CPS NDVI were greater in lightly grazed pastures (1000m) than in heavily grazed pastures (100m) only in 2008 ($p=0.04$ in 2008 for both CPS and CS NSVI) (Figure 3.2d,e). This was the highest rainfall year in our 14 year period (327mm) and forage use was 56%. Also note that Saikhan had consistently lower percent forage use by livestock compared to Bayangol even though use was reasonably high in both *soums*, especially after 2006.



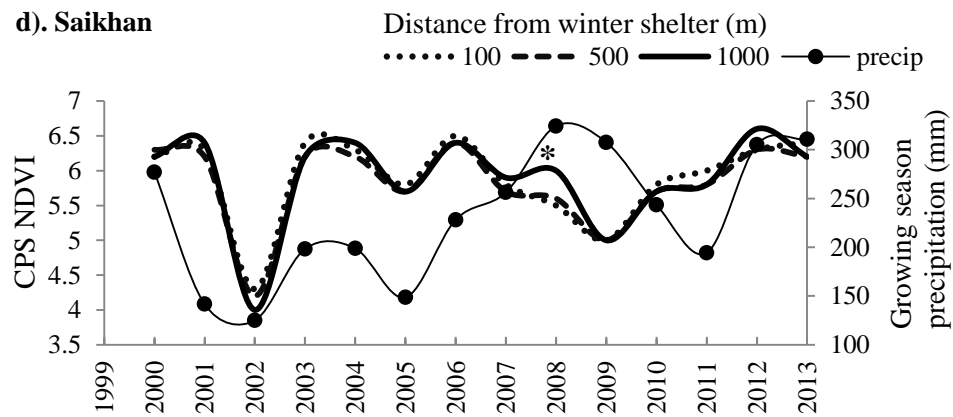
b). Bayangol



c). Bayangol



d). Saikhan



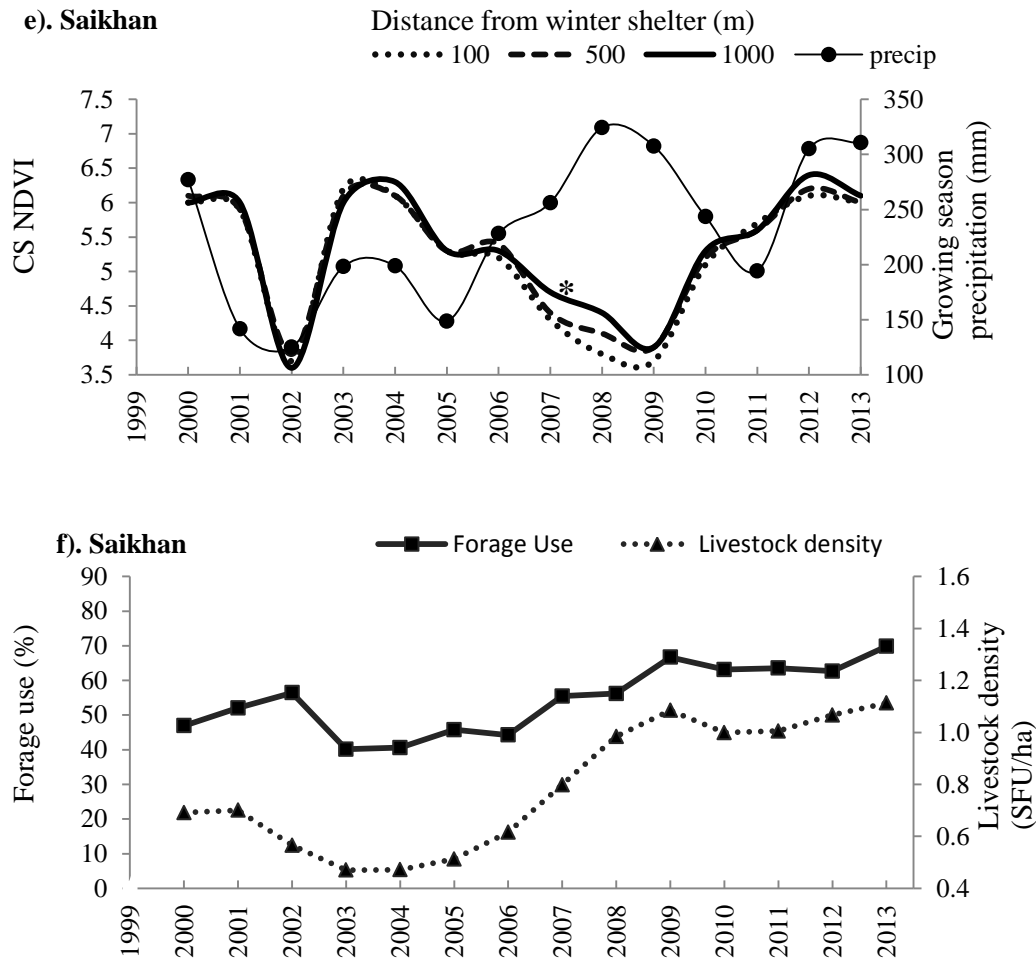
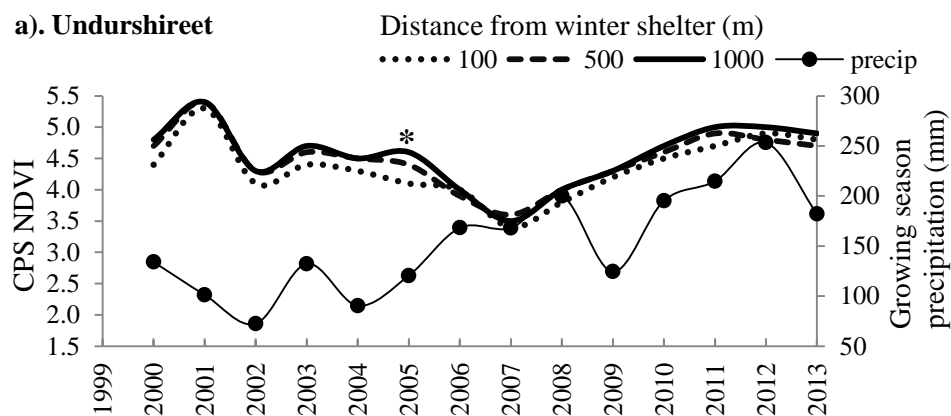


Figure 3.2. CPS NDVI, CS NDVI, forage percent use and livestock density (SFU/ha) between 2000 and 2013 in the a-c) Bayangol *soum* d-f) Saikhan *soum* of the mountain and forest steppe. SFU=sheep forage units.

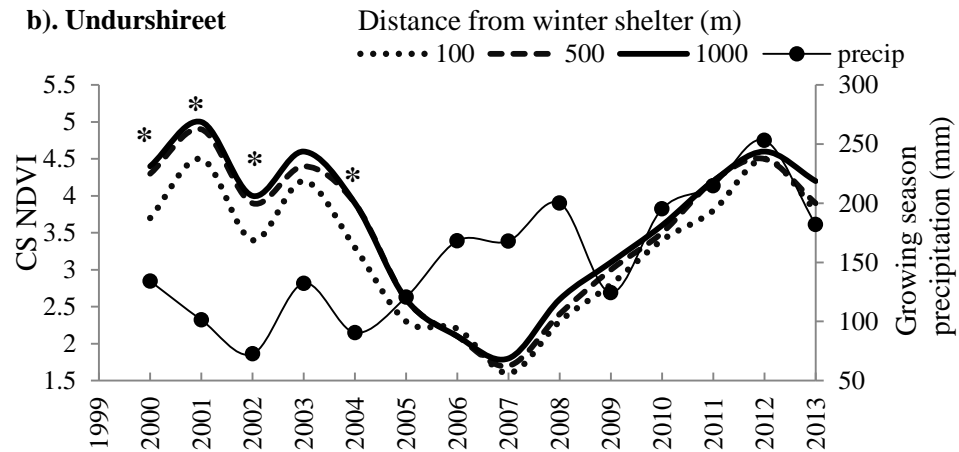
The two *soums* in the steppe had contrasting patterns of NDVI in response to grazing. In general, grazing gradients appeared in CPS and CS NDVI when there was moderate to low rainfall and low livestock densities. In Undurshireet *soum* of the steppe zone, CPS NDVI was greater ($p=0.03$) in lightly grazed pastures (1000m) than heavily grazed pastures (100m), but only in 2005 (Figure 3.3a). This was a year with moderately low rainfall (120 mm) and relatively low percent use and livestock densities, after five years with low livestock densities (Figure 3.3c). In this *soum* CS NDVI was greater in moderately and lightly grazed pastures (500, 1000m) than heavily grazed pastures (100m) in 2000 ($p=0.002$), 2001 ($p=0.04$), 2002 ($p=0.01$) and 2004

($p=0.006$) (Figure 3.3a, b). These were years with moderately low rainfall, and low livestock densities during and after the 1999-2002 *dzud*.

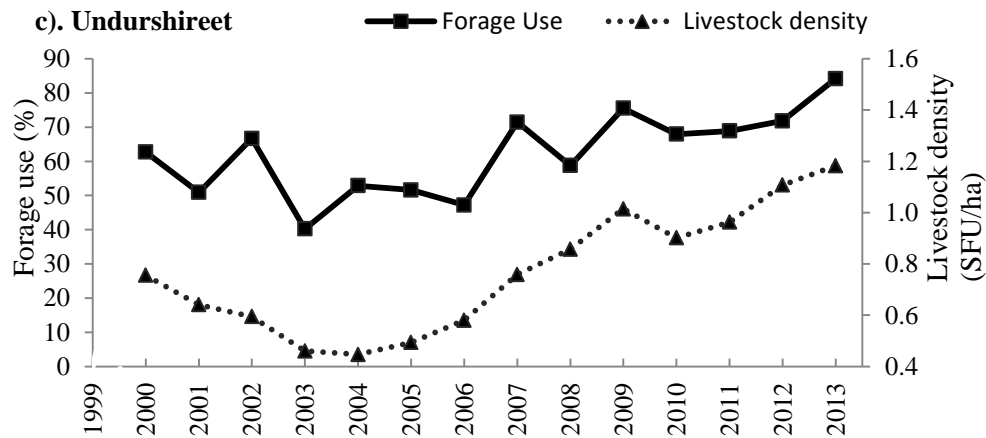
Unlike Undurshireet, NDVI in Erdenesant *soum* was greater in heavily (100m) than moderately grazed (500m) in 2008 (CPS NDVI, $p=0.003$; CS NDVI, $p=0.04$) and 2012 (CPS NDVI, $p=0.04$; Figure 3.3d, e). These were years when growing season rainfall was the highest during our study period (above 200 mm) and livestock forage use was relatively high (59 and 72% respectively, Figure 3.3f). A different pattern appeared in 2010, when both lightly (1000 m) and heavily (100 m) grazed pastures had higher CS and CPS NDVI than moderately (500 m) grazed pastures (CPS NDVI, $p=0.007$; CS NDVI $p=0.0001$). This was a moderately high rainfall year (195 mm), right after livestock populations fell after the *dzud* of 2009 with 69% forage use (Figure 3.3f). Note the percent use in Erdenesant was generally higher than in Undurshireet, across most of the study years. In fact, in 2013, percent forage demands (driven by the number of animals) greatly exceeded the forage available. Although we have no data to verify this, we suspect that herders moved their livestock to other areas to avoid a forage shortage.



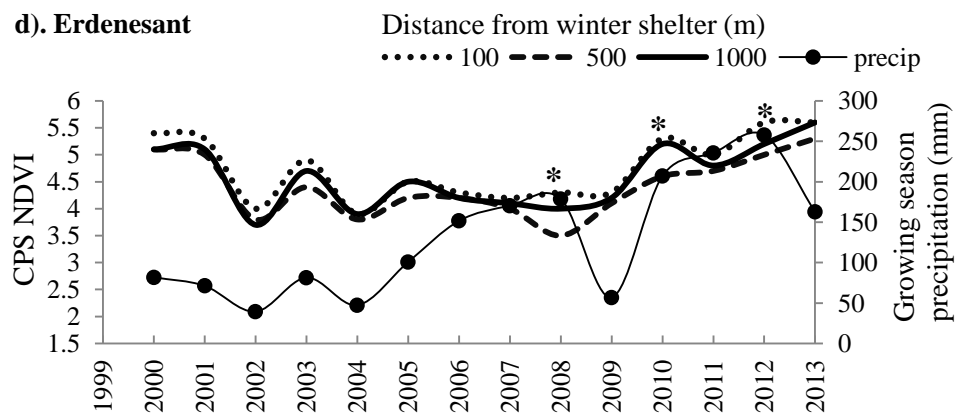
b). Undurshireet



c). Undurshireet



d). Erdenesant



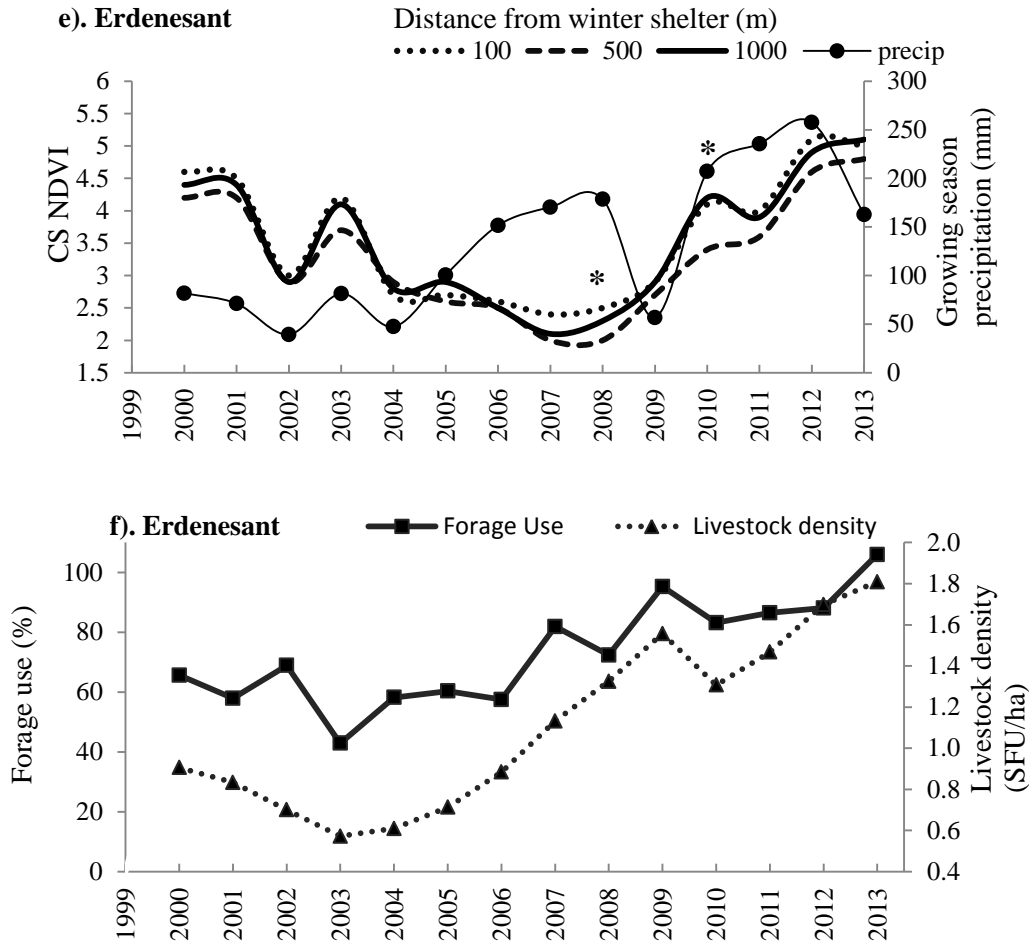


Figure 3.3. CPS NDVI and CS NDVI and forage percent use and livestock density (SFU/ha) between 2000 and 2013 in the a-c) Undurshireet *soum* and d-f) Erdenesant *soum* of steppe.

In the desert steppe, the grazing gradients were generally strongest in the early years of the study period during and after the 1999-2002 *dzud*. Interestingly, these are periods of generally low livestock numbers or forage use in either *soum*.

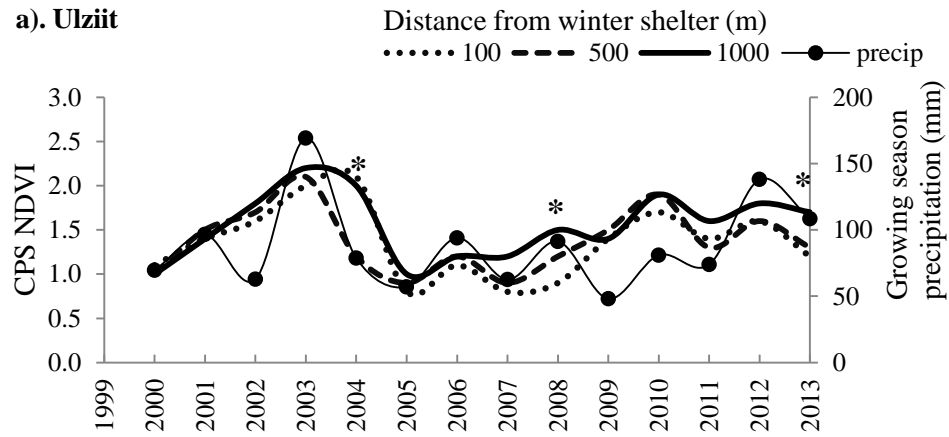
In the Ulziit *soum* of the desert steppe, there were grazing gradient differences only in 3 years out of 14 years in both CPS and CS NDVI (Figure 3.4a, b). In addition, NDVI patterns along the grazing gradient were inconsistent from year to year. In 2004, heavily (100m) and lightly (1000m) grazed pastures had greater CPS NDVI than the moderately grazed pastures (500m) ($p < 0.0001$). In 2008 and 2012, in contrast, distant pastures (1000 m) had higher CPS

NDVI than pastures closer to winter shelters (100 m, 500 m; 2008, $p=0.01$; 2012, $p=0.0048$). These were years with moderate to high rainfall (79-138 mm) and moderate forage use (27-34%). Generally, the expected grazing gradient, with greater NDVI in lightly grazed pastures, was strongest in the moderate to high rainfall years of 2008 (91 mm) and 2012 (138 mm).

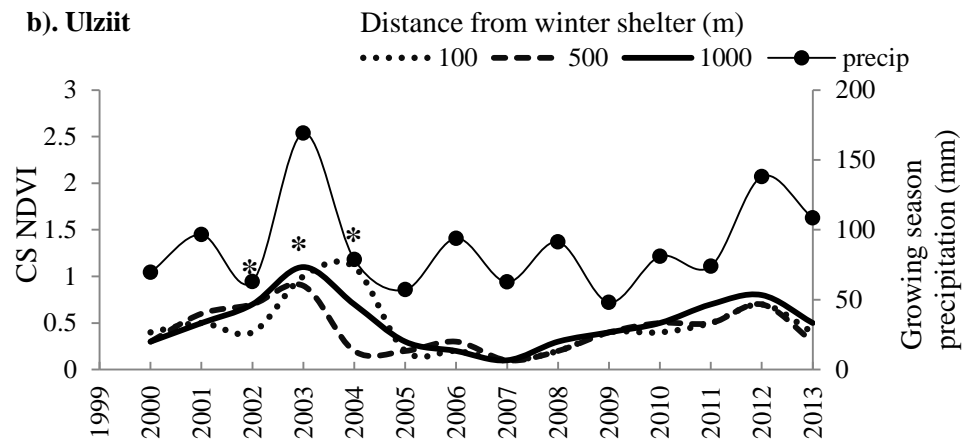
For CS NDVI in Ulziit in 2002, moderately grazed pastures (500 m) had greater CS NDVI than heavily grazed pastures at 100 m ($p=0.03$). In 2003, these same moderately grazed pastures had lower CS NDVI than lightly grazed pastures at 1000 m from winter shelters ($p=0.02$). Interestingly, in 2004, heavily grazed (100 m) and lightly grazed (1000 m) pastures had greater CS NDVI than plots in moderately grazed pastures (500 m; $p<0.0001$ Figure 3.4a, b). For CS NDVI results, these included low to high rainfall years (67-169 mm) at the end and immediately following the *dzud* in 1999-2002, when livestock use was low to moderate (29-34%) compared to other years in our study period.

In contrast to Ulziit, Undurshil *soum* of desert steppe only had grazing gradients in the direction predicted by our hypothesis (Figure 3.4 d, e). Both CPS ($p=0.03$ in 2002, $p=0.0013$ in 2003) and CS NDVI ($p=0.0032$ in 2003, $p=0.01$ in 2004) were greater in lightly grazed pastures at 1000 than those closer to the winter shelter (100m and 500m) grazed pastures in 2 out of 13 years (Figure 3.4e-d).

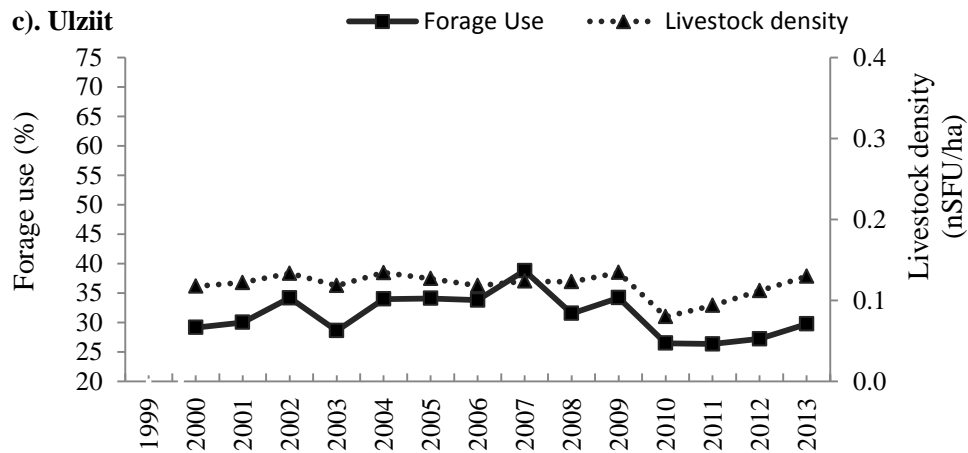
a). Ulziit



b). Ulziit



c). Ulziit



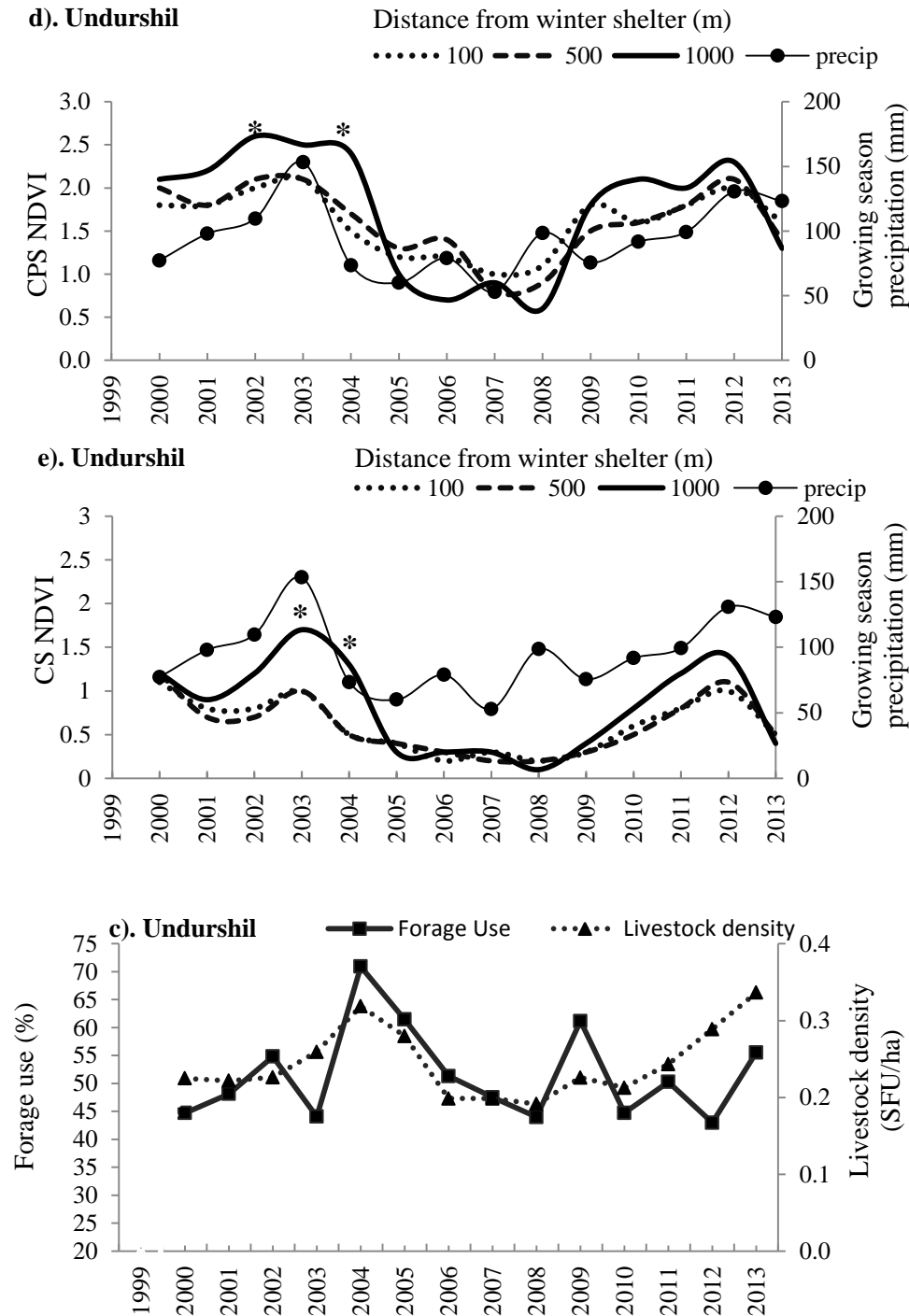


Figure 3.4. CPS NDVI and CS NDVI and forage percent use and livestock density between 2000 and 2013 in the a-c) Ulziit *soum* and d-f) Undurshil *soum* of desert steppe.

Significant grazing gradients appeared in equilibrium systems in the mountain and forest steppe and steppe after the winter disaster *dzud* of 2009 and livestock numbers decreased. CPS

NDVI in Bayangol *soum* of the mountain and forest steppe and both CPS and CS NDVI in Erdenesant *soum* of the steppe displayed significant grazing gradients. The percent forage use of the Bayangol *soum* was 82% and in Erdensant *soum* was 95% just before the 2009 winter disaster (Figure 3.2c and 4.4c).

In summary, comparing across ecological zones, there were the same number of years with significant grazing gradients in each of the three ecological zones, the mountain and forest steppe, steppe and desert steppe. Overall, 16-23% of study years had grazing gradients as measured by CPS and CS NDVI in the mountain and forest steppe, 16-46% in the steppe. But grazing affected significance of both CS and CPS NDVI for the same number of years for each measure in the desert steppe and were 19% of total study years.

In general across our all study sites, when livestock densities and forage use were low, CS and CPS NDVI corresponded with changes in rainfall (Figure 3.2-4.4). But when livestock densities and forage use increased, CS and CPS NDVI started to vary independent of rainfall, becoming more affected by livestock density and use (but not necessarily more than climate) over the last 14 years. The years that NDVI mirrored the patterns of rainfall or livestock density and use varied between the ecological zones and even among the *soums* within an ecological zone.

In both Bayangol and Saikhan *soums* of the mountain and forest steppe, CPS and CS NDVI were coupled with growing season rainfall early in our study period (between 2000 and 2004 in the Bayangol *soum*; between 2000 and 2006 in the Saikhan *soums*; Figure 3.2a-b and d-e), when livestock densities were low and percent forage use was about 50% or less (Figure 3.2 c, f). As livestock densities increased and percent forage use exceeded 80%, between the 2006 and 2009, CS and CPS NDVI began to de-couple from patterns of rainfall and couple more

closely with the amount of livestock use. After the *dzud* during the winter of 2009 - 2010, livestock density and forage use decreased slightly.

In Undurshireet *soum* in the steppe zone, unlike the mountain and forest steppe, there was fast and altering correspondence (or correlation) of rainfall and livestock density and use with NDVI. The NDVI increased in response to decreased livestock use even when precipitation was low in 2001, whereas the low NDVI response in 2002 displayed the combined effects of decreased rainfall and increased livestock number. When livestock use was comparatively low, CS and CPS NDVI patterns were closer to those of rainfall between 2002 and 2005 and started to respond to increased livestock densities/use between 2006 and 2007. After the *dzud* in the winter of 2009-2010, livestock densities and use decreased slightly and NDVI patterns more closely mirrored those of rainfall (Figure 3.3 a-c). In Erdenesant *soum*, both CPS and CS NDVI patterns were close to those of rainfall between 2000 and 2005 when livestock densities and use were low. NDVI patterns appeared to respond to livestock densities and use sharply since 2005. This was especially clear between 2005 and 2009, when livestock densities and use were rising. NDVI was low even though rainfall was high. Since 2009, after the *dzud*, livestock densities and use decreased slightly and growing season rainfall increased tremendously between 2009 and 2012, when NDVI patterns more resembled that of rainfall (Figure 3.3 e-f).

Overall, the NDVI patterns in the desert steppe *soums* more closely followed the patterns of rainfall than those of livestock between 2000 and 2013 (Figure 3.4a,b d, e). As we saw under question 3, there appeared to be greater livestock effects on vegetation in the years with the highest rainfall. Also, the highest rainfall event occurred in 2003 over the last 13 years in these 2 desert *soum* with the greatest NDVI response.

3.3.3. Resistance and resilience to disturbance of pastures grazed at different intensities

Here, we predicted that the NDVI in more intensively grazed pastures (100 m) will vary more widely (higher coefficient of variation, less inertia and resistance) from year to year than the NDVI in less intensively grazed pastures (500 and 1000 m, Table 3.2). NDVI variability was either higher or showed no difference at 100 m plots than 500 or 1000 m plots for all *soums* except Erdenesant in the steppe. In Erdenesant, NDVI was higher in the 100 m plots (see Figure 3.2 above) and NDVI variability was lower in these same plots (Table 3.2).

Table 3.2. Coefficient of variation (%) of CS and CPS NDVI at 100, 500 and 1000 plots in each *soum*, averaged across all years from 2000 to 2013.

<i>Soum</i>	NDVI measure	100 m	500 m	1000 m
Bayangol (Mountain and forest steppe)	CPS	15.0	13.6	10.9
	CS	25.0	21.2	19.0
Saikhan (Mountain and forest steppe)	CPS	13.6	13.8	13.6
	CS	21.2	18.9	18.9
Undurshireet (Steppe)	CPS	18.6	17.8	17.8
	CS	36.4	37.1	36.4
Erdenesant (Steppe)	CPS	20.8	18.2	21.7
	CS	38.9	39.4	45.7
Ulziit (Desert steppe)	CPS	28.6	28.6	25.0
	CS	75	75	66.7
Undurshil (Desert steppe)	CPS	37.5	35.3	36.8
	CS	83.3	83.3	77.8

In addition, we predicted NDVI would recover after significant declines more rapidly in more lightly grazed pastures (500 and 1000 m) than heavily grazed pastures (100 m; Table 3.3). In general, NDVI recovery times did not support our expectations, except for CPS NDVI in the

steppe *soum* of Undurshireet. In the mountain and forest steppe, NDVI recovery time was longer in more lightly grazed (500 and 1000 m) than heavily grazed pastures (100 m). The other *soum* in this zone, Saikhan, showed no difference in recovery times in response to grazing. In Erdenesant, NDVI recovery times were longest in moderately grazed pastures (500 m) and shorter in both heavily (100 m) and lightly grazed (1000 m) pastures. Recovery times in the desert steppe were inconsistent with recovery times for CPS NDVI with longer times in more lightly grazed pastures and shorter times in heavily grazed pastures for CS NDVI.

Table 3.3. Mean recovery time (years) of CPS and CS NDVI after either low rainfall or heavy grazing in pastures 100, 500 and 1000 m from winter shelters in 6 study *soums* of the mountain and forest steppe (MFS), steppe (S) and desert steppe (DS).

<i>Soum</i>	NDVI measure	100 m	500 m	1000 m
Bayangol (Mountain and forest steppe)	CPS (\pm StDev)	3.5 (\pm 2.6)	3.9 (\pm 2.4)	4.1 (\pm 2.6)
	CS (\pm StDev)	3.3 (\pm 2.1)	4.0 (\pm 2.6)	4.2 (\pm 2.2)
Saikhan (Mountain and forest steppe)	CPS (\pm StDev)	3.5 (\pm 1.4)	3.3 (1.0)	3.4 (\pm 1.4)
	CS (\pm StDev)	3.8 (\pm 1.9)	3.9 (\pm 2.6)	3.8 (\pm 2.2)
Undurshireet (Steppe)	CPS (\pm StDev)	5.0 (\pm 3.1)	4.8 (\pm 2.5)	4.1 (\pm 1.9)
	CS (\pm StDev)	4.9 (\pm 2.6)	5.0 (\pm 2.8)	4.4 (\pm 2.3)
Erdenesant (Steppe)	CPS (\pm StDev)	5.1 (\pm 3.8)	4.6 (4.0)	5.7 (\pm 3.7)
	CS (\pm StDev)	5.0 (\pm 3.2)	3.7 (\pm 3.5)	5.2 (\pm 3.5)
Ulziit and Undurshil (Desert steppe)	CPS (\pm StDev)	3.3 (\pm 2.1)	4 (\pm 1.5)	4.5 (\pm 2.1)
	CS (\pm StDev)	3.5 (\pm 0.6)	3.2 (\pm 1.1)	3.3 (\pm 1.8)

3.4. Discussion

3.4.1. Relationship between field and remote sensing data

Our comparison of field and NDVI data showed only moderate relationships between the two sets of data. When there was significance, it supported our hypothesis that greater biomass or cover was matched by greater NDVI values, as shown commonly elsewhere (Tucker and

Sellers 1986, Kawamura et al. 2005a). Field and remote sensing data were significantly correlated in the mountain and forest steppe and steppe, but not the desert steppe. Field measures of vegetation taken at one point in a season may have a poor relationship with total season biomass accumulation (like NDVI), and this is especially true in drier, more climatically variable areas than in wetter areas (Elmore et al. 2000, Wessels et al. 2012). The greater variability of the desert steppe vegetation means it is very unlikely our field data sampling occurred at the height of the growing season, causing the poorer correlation in this dry zone to full season NDVI.

These results suggest that remotely sensed and field data have different strengths for assessing change in rangelands, as found elsewhere (Hunt et al. 2003, Booth et al. 2005). NDVI, the most commonly used remote sensing-derived measurement, provides spatially extensive and temporally extensive observations of rangeland change which are usually not available through-ground measured data because of limitations of time and financial resources. Although remotely sensed NDVI data can be very helpful for detecting changes over time, these data still cannot detect important fine-scale information like functional group biomass, cover and species richness, which are best collected by on-the-ground sampling.

3.4.2. Effects of winter grazing gradients on NDVI

Generally, when there is a grazing gradient, lightly grazed pastures had higher NDVI values than more heavily grazed pastures, as we expected based on other studies (Kawamura et al. 2005b, Butt 2010, Bradley and O'Sullivan 2011, Sha et al. 2014). We did not expect, however, to find the opposite pattern, where NDVI was higher in heavily than lightly grazed pastures, but this was the case in the steppe site of Erdenesant, that there were just as many years

with significant grazing gradients in each of our ecological zones came as a surprise. We expected to see many more years with significant grazing gradients in wetter zones than drier zones, as would be predicted by the non-equilibrium theory of rangeland dynamics. This theory predicts that climate has more effect on rangelands in drier zones, while livestock has more effect on rangeland vegetation in wetter zones (Ellis and Swift 1988).

CPS and CS NDVI showed different responses to grazing. CS NDVI responded quickly to the season-by-season changes in growing season precipitation and livestock densities/forage use. By contrast, grazing gradients only appeared in CPS NDVI after longer term periods of low livestock densities/forage use. Also, CS NDVI showed more years with significant grazing gradients than CPS NDVI across our study sites. This may suggest that the standing dead vegetation, which is only part of the CPS NDVI measures, buffers the effects of grazing and production variations. This could have important management and policy implications.

Our data show that grazing significantly affects NDVI only in some years and not others, and this appears in the interaction between the levels of growing season rainfall and livestock grazing. For example, in both Bayangol and Saikhan in the mountain and forest steppe, grazing gradients did not appear in either CPS or CS NDVI in years with rainfall below 200 mm. In these years, livestock may be more spread out to obtain sufficient forage, so the grazing gradient is weaker. Grazing gradients also appeared after periods of low livestock grazing, especially as measured by CPS NDVI. When there are few livestock, there is plenty of forage and animals can concentrate near winter camps because they have no need to spread out to get enough to eat. For example, in Bayangol, grazing gradients in CPS NDVI occurred after a 4 –year period of low livestock use during and after the 1999-2002 *dzud*. In 2007, the same patterns occurred: elevated precipitation and low livestock densities from preceding years created a significant grazing

gradient in CPS NDVI, likely showing the accumulation of dried biomass from year to year. Similarly, in the Saikhan *soum*, a grazing gradient appeared in 2008, when growing season precipitation peaked as livestock densities were steadily rising.

The steppe sites showed a contrasting pattern of grazing gradients appearing in response to interactions between rainfall and livestock densities and use. In Erdenesant *soum*, steppe, heavily grazed pastures close to winter shelters had high CPS and CS NDVI when both precipitation and livestock use were relatively high, like the mountain and forest steppe. By contrast, in the Undurshireet *soum*, the grazing gradient appeared in CPS NDVI in 2005 after 6 years of low livestock densities, showing biomass accumulation. But, unlike Bayangol, the grazing gradient in Undurshireet appeared in CS NDVI during the years with low to moderate precipitation. This is similar to the Sahel, where grazing had more impact in years with low rainfall, not high rainfall (Hein 2006).

Another interesting pattern occurred in Erdenesant, probably because this *soum* had higher livestock densities and forage use than our other study *soums* (Figure 3.3c). We think this caused a reversal of the grazing effects on NDVI, where NDVI was higher in heavily than lightly grazed pastures. Here, we suspect that high cover of annual forbs and weedy plants in heavily grazed pastures close to winter shelters may be causing this pattern. Previous studies found that that heavy use by livestock can create directional changes in vegetation composition with increased unpalatable forb and weedy annual forbs in heavily grazed pastures (Fernandez-Gimenez and Allen-Diaz 1999, Cheng et al. 2011, Okayasu et al. 2012, Sasaki et al. 2013). Forbs have a high spectral reflectance because of their broad leaf structure and greater greenness values (Karnieli et al. 2013). In Chapter 2, we found a high abundance of annual forbs near

winter shelters across many steppe sites, including Erdenesant. We also suggest this is the explanation for the greater NDVI found near water points found by Sternberg (2012).

In the desert steppe, there were no consistent patterns in the grazing gradients, comparing among lightly to moderately to heavily grazed pastures. This would be expected in rangelands where climate is the over-riding factor driving vegetation dynamics. However, the expected grazing gradient, where NDVI was greater far from winter shelters, did appear, but only appeared in higher rainfall years (but did not appear in all higher rainfall years). As rainfall increases in the desert steppe, the effects of livestock may become more apparent, as predicted by equilibrium and non-equilibrium theory.

Finally, another pattern appeared in our data in relation to Chapter 4 on community-based management groups (CBRM). In Chapter 4, we compare *soums* with and without active community-based management. In this chapter, our six *soums* include 3 *soums* (Bayangol, Undurshireet and Ulziit) with this active management and 3 *soums* without. We found here that there were more years with significant grazing gradients in *soums* with active, formal CBRM groups than in *soums* with more informal traditional neighborhoods (see Ch 4 for a description of these groups). This was especially true for gradients measured with current season (CS) NDVI. These patterns could be the result of improved grazing practices by CBRM groups (Ulaambayar 2015) and / or better enforcement of protection of winter pastures from summer grazing.

3.4.3. Relative effects of growing season precipitation and grazing on NDVI

The NDVI patterns, compared to precipitation and forage use patterns, showed a shift from precipitation-dominated vegetation dynamics in the early 2000s to livestock-dominated vegetation dynamics in the late 2000's especially in our study *soums* in mountain and forest

steppe and steppe. These patterns were much less clear in the desert steppe, where the dynamics appear to follow precipitation more closely than in the wetter zones. Remote sensing studies on Mongolian rangelands disagree about whether rangeland dynamics are driven primarily by precipitation (Liu et al. 2013) or by livestock grazing (Hilker et al. 2013) or both depending on the precipitation gradient (Wang et al. 2013). Our study, like Khishigbayar et al. (2015), suggests that Mongolian rangelands may be approaching a tipping point where grazing becomes more important than precipitation in driving rangeland dynamics.

Our study also showed that NDVI in the mountain and forest steppe and steppe tracked grazing more closely than NDVI in the desert steppe. Even so, there were just as many years with significant grazing gradients in the desert steppe as the other zones. These two conflicting results suggest some, but not total, support for the predictions of equilibrium and non-equilibrium rangeland dynamics where drier systems are driven more by precipitation and wetter systems are driven more by livestock grazing (Ellis and Swift 1988, Fernandez-Gimenez and Allen-Diaz 1999).

3.4.4. Resistance and resilience of pastures grazed at different intensities to disturbance

We predicted that heavy grazing will cause the NDVI of pastures to be more variable over time, and thus be less resistant or have low inertia (Westman 1978, Washington-Allen et al. 2008) to grazing. Inertia (or resistance) is the ‘ability of a system to resist displacement in structure or function when subjected to a disturbing force’ (p. 705, Westman 1978). We used the coefficient of variation of NDVI as a measure of resistance, with more resistance shown by a low average inter-annual CV over time. In addition, we also predicted that the NDVI of heavily

grazed pastures will recover more slowly than the NDVI in lightly grazed pastures. For resistance, our predictions were largely supported, with heavily grazed pastures showing more variability in NDVI over time and lightly grazed pastures showing less variability over time.

We predicted that heavily grazed pastures would take longer to recover from disturbance than lightly grazed pastures. This recovery time is measure of resilience or elasticity (Westman 1978, Washington-Allen et al. 2008) of pastures grazed at different intensities. The results of one study *soum* (Undushireet *soum*, steppe zone) out of 6 *soums* supported our hypothesis. However, one of the study *soums* showed the opposite of our hypothesis. In the Bayangol *soum* of mountain and forest steppe, lightly grazed pastures had a longer recovery period than the more heavily grazed pastures. This could be a real pattern showing that grazing had different effects on resilience in different places. We offer two other explanations. First, the NDVI of heavily grazed pastures may rebound quite quickly, since these pastures have little accumulated biomass from year to year. Second, our study period may have been too short to detect the differences in resilience caused by grazing (Washington-Allen et al. 2008).

3.4.5. Future research

To understand system resilience, we need to continue tracking NDVI, forage use and climate year by year and even season by season. It is also essential to have ground level vegetation data or to ground-truthing to be able fully interpret NDVI trends. Because livestock number and or forage use have been increasing in all study sites, it will be essential to track any cumulative livestock effects to understand if they are reducing system resilience, combined with stochastic climate events (drought and *dzud*). We need to develop robust indicators of important, threshold crossing change, meaning like the lack of recovery to pre-stress NDVI levels, or long

delays in recovery. Further it will be essential to conduct studies to understand agreement between the changes in plant communities (medium scale=functional groups and fine scale=individual species) and satellite remote sensing data, thus it will be useful for future rangeland monitoring to detect the cause or trigger of future level of rangeland changes due to predicted climate change and their interaction with changing grazing pressure. It will be essential to include local herders in interpreting the study results and application for their pasture management as well.

CHAPTER FOUR: ECOLOGICAL OUTCOMES OF COMMUNITY-BASED RANGELAND MANAGEMENT IN MONGOLIA

4.1. Introduction

In the early 1990's, many efforts began around the world to devolve natural resource management to the local or community level (Agrawal 2003). Since then, in different parts of the world, community-based resource management (CBRM) institutions have been emerging on state, common and private lands with the goal to achieve environmentally sustainable management of resources, livelihoods, and social relationships in the face of rapidly changing political, economic and social-ecological conditions (Reid et al. 2014).

There is a little evidence about the ecological outcomes of community-based institutions; of these few studies, there are more in forests (Topp-Jorgensen et al. 2005, Van Rijsoort and Jinfeng 2005, Chhatre and Agrawal 2008, Brooks et al. 2013) and coastal systems (Campbell and Salus 2003, Cinner et al. 2012, Cinner and McClanahan 2015) and very few in rangeland systems (Leisher et al. 2012, Addison et al. 2013). Most of the information about the ecological outcomes of CBRM is hypothetical or based on the perceptions, attitudes and observations of community members and other stakeholders about resource condition managed by the community groups. There is a little direct measurement of changes in ecological conditions managed by community groups.

The few studies that have directly measured ecological outcomes show that CBRM groups sometimes do and sometimes do not improve these outcomes. On the positive side, these efforts can improve plant production and cover (Leisher et al. 2012), reduce illegal hunting (Mbaiwa et al. 2011), increase fish biomass and coral cover (Cinner and McClanahan 2015) and contribute to keeping the land intact and preventing fragmentation (Reid et al. 2008). On the

other hand, some studies have shown no effect of CBRM (Addison et al. 2013). Despite the above studies, there are no broad-scale, well-replicated studies about the ecological outcomes of community-based management.

Like elsewhere in the world, in Mongolia, CBRM initiatives grew rapidly in the late 1990s. From 1999-2002 there were 3 consecutive years of drought and *dzud* (harsh winter), when about 30% of all livestock herds perished and many herder households lost livestock and as well as their livelihoods (ReliefWeb 2010, Fernandez-Gimenez et al. 2015). After a Mongolian government appeal, international donors started implementing projects to provide support for rural herders. Donors saw herder groups as beneficial institutions to support because of their local nature and close kinship relations. Herders then started forming more groups as a response to donor project agendas, with over 2000 groups established throughout the country since then (Mau and Chantsallkham 2006). Most herder groups were initiated ‘top down’ as part of outside donor projects, but some others were formed by a ‘bottom-up’ community approach, largely as a grassroots response to the ‘community vacuum’ left by collapse of the socialist system that occurred in the early 1990s (Mau and Chantsallkham 2006).

Before these new CBRM institutions were established, herders in Mongolia practiced traditional neighborhood cooperation at three different scales (Russell 2005). At a small scale cooperation occurred among members of a herding camp or *khot ail*, comprised of up to 6 households sharing the same herding campsite. At a medium scale, herders cooperated in *neg golynkhon* (people from one river area) and or *neg jalgynkhan* (people from one valley), comprised of 20-30 *khot ails*. Large-scale cooperative are called *neg nutgiinkhan* (people from one place or one district). This system was in place for several hundred years (Russell 2005) although it was influenced and forced to change during the socialist state collective periods. The

main difference between traditional neighborhoods (hereafter called non-CBRM groups) and CBRM groups is that the latter received donor project support and various trainings and are more recent.

Herders in CBRM groups are more likely to be prepared for winter disasters and adopt innovative practices more often than herders in traditional neighborhoods (Fernandez-Gimenez et al. 2015). CBRM herders are more likely to reserve spring pastures, cull unproductive animals in the fall, cut hay and hand fodder than herders in traditional neighborhoods. They also adopt newer agricultural practices like fencing pastures, hayfields and water sources; using irrigation; planting gardens and taking part in environmental monitoring (Fernandez-Gimenez et al. 2015). All these practices are assumed to contribute to ecological outcomes of CBRM, yet there is no study that connects community groups' practices with ecological outcomes.

In Mongolia, the few field studies (Leisher et al. 2012, Addison et al. 2013) that have measured ecological outcomes of these improved practices of community-based rangeland management groups in the southern and dry part of the country and show contradictory results. Leisher et al. (2012) concluded that CBRM implementation had beneficial effects on pasture condition by using a remote sensing approach over a period of 10 years, focusing on 6 of 50 *soums* (or counties) in the dry Gobi region. They found that CBRM pastures had a longer growing season and higher peak plant growth than non-CBRM pastures. In contrast, Addison et al (2013), working in 14 Gobi *soums*, showed that there was little difference in rangeland condition between CBRM and non-CBRM pastures, based on ecological field measurements. They used vegetation and soil variables such as percent plant cover; perennial vegetation patch; litter cover; existence, severity and type of erosion features; and vegetation utilization during the time of their study.

In this study, we sought to conduct the first, country-wide assessment of the effects of CBRM groups and traditional neighborhoods on rangeland vegetation and soils in Mongolia across four ecological zones: the mountain and forest steppe, eastern steppe, steppe and desert steppe. We sought the answers to the following research questions:

Research questions

1. Are pastures managed by formal community-based rangeland management (CBRM) groups in better condition than pastures managed by traditional neighborhood or non-CBRM groups?
2. Do CBRM and traditional neighborhood management have different effects on vegetation, forage quality and soils in different ecological sites within ecological zones?
3. Do the effects of CBRM and traditional neighborhood management differ in different ecological zones?

Research hypotheses

1. Pastures managed by CBRM groups have higher cover, biomass, and species richness than the pastures managed by the non-CBRM, traditional neighborhood groups.
2. Forage quality and palatability are also higher in CBRM-managed pastures than non-CBRM-managed pastures.
3. The size of gaps between perennial plants in pastures managed by CBRM groups is smaller than in pastures managed by non-CBRM groups. There will be little erosion and deposition by wind or water in CBRM-managed pastures.
4. All the above effects will be greatest in the mountain and forest steppe and smallest in the desert steppe and mixed in the steppe and eastern steppe, following the predictions of equilibrium and non-equilibrium rangeland dynamics.

4.2. Methods

4.2.1. Study Areas

To achieve broad coverage across ecological zones, we selected study areas in 36 *soums* (counties) of 10 *aimags* (provinces) across 4 ecological zones in the mountain and forest steppe, eastern steppe, steppe and desert steppe (Figure 4.1).

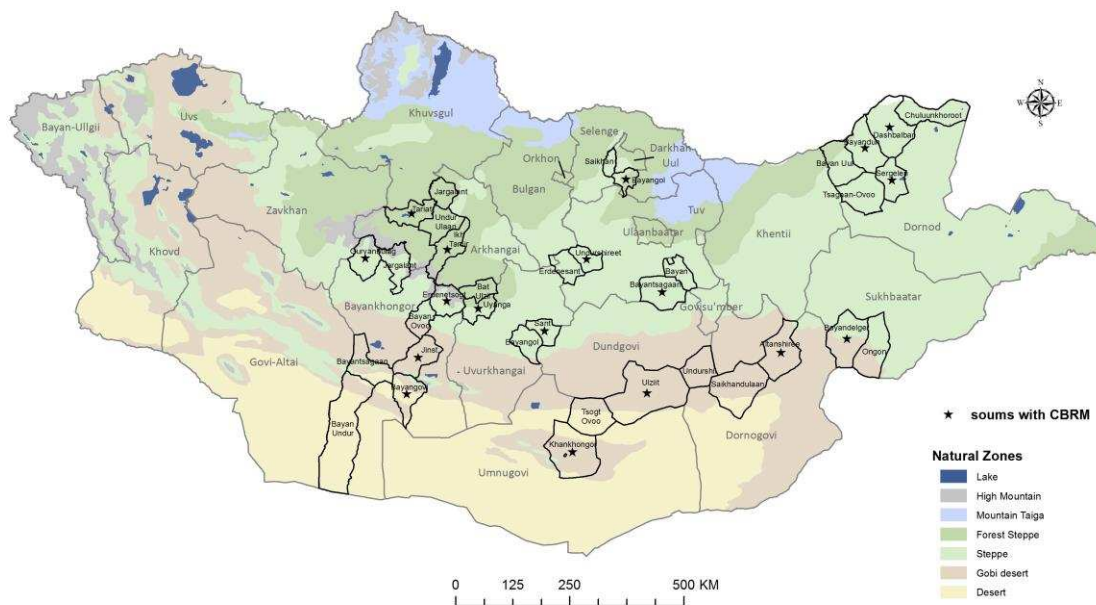


Figure 4.1. Study sites. Villages with star sign are those with community-based rangeland management groups

We selected pairs of *soums* in each *aimag* (province), with one *soum* of each pair having donor-supported community-based (CBRM) groups, and the other *soum* with traditional neighborhoods (non-CBRM groups), without any donor financial support. The CBRM *soums* (and groups) were supported by one of four donor projects, including the Green Gold Pasture Ecosystem Management project funded and implemented by the Swiss Agency for Development and Cooperation (SDC), the Sustainable Grassland Management project funded and implemented by the United Nations Development Program (UNDP), the Conservation and

Sustainable Management of Natural Resources project funded by German Technological Cooperation and implemented by New Zealand Nature Institute (NZNI), and the Daurian Steppe Sustainable Conservation Approaches for Priority Ecosystems project implemented by the Wildlife Conservation Society (WCS). One (SDC) out of four donor projects is still ongoing, the other 3 projects ended in 2006 (NZNI), in 2012 (UNDP) and in 2014 (WCS). Two of the donor projects (SDC and UNDP) operated in three ecological zones (mountain and forest steppe, steppe and desert steppe), whereas NZNI operated only in the desert steppe and WCS operated only in the eastern steppe (Ullambayar 2015).

Mean annual temperature and precipitation is -2.2°C and 239 mm in mountain and forest steppe, 0.2°C and 258 mm in eastern steppe, -0.09°C and 170 mm in steppe, and 2.6°C and 131 mm in desert steppe (Hijmans et al. 2005, Chen et al. 2008). More than 80% of precipitation falls between May and September in all zones. Summer is short and hot, with the hottest month in July when temperatures range between 18 and 26°C in all zones. Winter is cold and dry with the coldest month in January when average temperatures are -35°C in the mountainous region and -10°C in the desert steppe region. All four ecological zones are dominated by perennial grasses (Gunin et al. 1999) (See Table 2.1 for a summary of ecological zone characteristics in Chapter 2).

All study areas have been grazed by domestic livestock under nomadic and transhumance pastoral use for at least 1000 years and possibly for several millennia (Johnson et al. 2006). The main types of livestock are cattle/yaks, horses, sheep and goats and camels. Herd composition varies with ecological zone with few to no camels in the mountain and forest steppe and few cattle and no yaks in the desert steppe. In all four ecological zones, herders move livestock seasonally through different pasture areas. Herds spend summers near water sources including

rivers, lakes and water wells. Families then move to fall camps, usually in open plains, and then spend the winter in sheltered places, facing south, usually locating their winter shelters on the warmer, leeward side of mountains or hills. In the spring, families move to lower elevation, more open areas where snow melts out early. Traditionally, herders graze different pastures in the four seasons, particularly avoiding summer grazing in winter grazing grounds to preserve critical pastures for winter grazing. These ‘preserved’ winter grazing lands cover about third to half of the annual grazing orbit (ALAGAC 2010).

4.2.2. Sampling methods

We focused our sampling on winter pastures for three reasons. First, winter reserves are a critical limiting factor for Mongolian livestock production systems. Second, winter shelters are usually used year-after-year, and thus livestock grazing impacts accumulate here over the long term, so grazing impacts on pastures around winter shelters should be some of the greatest in Mongolian pastures. Third, winter pastures are usually not grazed during the growing season, and thus our samples represent measurements of ungrazed, peak annual production, somewhat like a natural ‘utilization cage’ experiment.

In each *soum* with a CBRM organization, we sampled pastures around five randomly selected winter shelters; in non-CBRM *soums*, we sampled pastures around four winter shelters. At each winter shelter, our goal was to control for the effects of grazing by measuring vegetation and soils in pastures with different levels of grazing. We did this to avoid confounding the effects of CBRM and grazing. To control for grazing, we sampled along a grazing gradient away from each winter shelter in 3 plots located at 3 distances from the winter livestock shelter (100 m, 500 m and 1000 m), as measured with a GPS from the gate of the livestock corral at the

shelter. We selected these specific distances because of previous work showing that the effects of livestock grazing are greatest close to livestock grazing impact points such as livestock camps or water points and largely minimal farther than 1000 m from the impact point (Sasaki et al. 2008a, Sasaki et al. 2011). We used this grazing gradient variable to as a covariate in the analysis and to adjust for grazing in estimated least square means, and thus do not present its main or interaction effects, because this is not the focus of this chapter.

We selected plots at the three distances on the same landform (e.g. hill, fan/piedmont, terrace, or plain), hillslope profile position (summit, shoulder, backslope, footslope, toeslope etc.), aspect, and, if possible, soils. This allowed us to sample a grazing gradient that falls on the same ecological site, which is defined by landform, soils, and climate, which potentially produces similar kinds and amounts of vegetation and responds similarly to natural disturbances, drivers and management (Bestelmeyer et al. 2009). Because winter shelters are often located at the head of a valley, we selected sites along a gradient following a contour along the edge of the valley rather than dipping downslope onto a different soil at the valley bottom. This meant that most sites were in specific upland positions, not in riparian areas or deep in valley bottoms, unless the winter shelter was located in this landscape position. Thus, our sample is not a random sample of winter grazing areas, but rather a sample of the specific ecological sites selected by herders for their winter shelters. In this way, we avoided confounding distance from the winter shelter or ‘impact point’, intended as an index of grazing intensity, with a gradient in soil moisture or water table depth, and associated changes from riparian to upland vegetation. We selected the plots, with only a few exceptions, so that they were located at least as far away from any *other* livestock camps and water points, as they were from the focal impact point (winter shelter or water point). In other words, we made sure that the 1000-m plot was at least

1000-m away from any other livestock camp or water point as well as from the selected campsite. We did not use the appearance of the vegetation to judge if our selected plots were on similar ecological sites, only using landform and soils, to limit confounding, since grazing can impact vegetation. We sampled a total of 143 winter shelters/water points, with 117 plots in the desert steppe, 122 plots in the steppe, 33 plots in the eastern steppe and 156 plots in the mountain and forest steppe. At six winter camps, we could not find a 1000 m site and thus sampled two 500 m plots instead; and at one site we could not find a 100 m site and thus sampled only two plots (500 m and 1000 m) at this camp.

While we located our three plots on a similar ecological site for each winter shelter, we did not sample all winter shelters on the same ecological site across our study. Thus this study also tests for the main and interaction effects of ecological site on vegetation and soils, in relation to CBRM (but not grazing).

Each plot was 50 x 50 m in size with 5 systematically spaced 50-m transects (Figure 2.2). Transects originated at 0, 12.5, 25, 37.5 and 50 meters along the baseline. If the plot was on a slope, we oriented it so that transects ran up the slope, to incorporate variability within each transect. The origin point (at 0 m) and baseline were always on the downhill side of the plot with transects running uphill from the baseline.

We obtained *soum*-level livestock numbers from National Statistical Office of Mongolia (2013), whereas livestock numbers at the *uvuljuu* level were collected through household surveys of the *uvuljuu*'s (or winter camp's) traditional "owner" (Umbayar 2015). When reporting all the livestock species together, we converted different species into the same relative measure of sheep forage units (SFUs), where one camel=5 SFU, one horse=7 SFU, one cow/yak=6 SFU, one goat=0.9 SFU and one sheep=1 SFU (NSO 2013).

We quantified livestock use in our plots as an indirect measure of grazing along the gradient away from the winter shelters in 3 of 4 ecological zones (not the eastern steppe). We made these measurements in a subset of our plots by sampling 27 plots in the desert steppe, 156 plots in the mountain and forest steppe, 92 plots in the steppe. At each plot, we recorded the frequency of sheep/goat pellets, horse and camel dung pellets and cow/yak dung piles. We used a 50 x 50 cm quadrat to record presence and absence of sheep/goat pellets, a 1x1m quadrat frame to record presence and absence of cow/yak, horse and camel dung pellets or piles. We used a smaller quadrat for sheep and goat pellets because these pellets were so evenly spread that the smaller quadrat size allowed us to capture some quadrats with no dung (the larger quadrat often registered 100% presence for all our plots). We placed the frame to the right of the transect tape every 5 meters at the 0, 5, 10, 15, 20, 25, 30, 35, 40 and 45 meters along each of the 5 transects. In total, presence/absence of dung as well were recorded in 50 quadrat frames in each plot

Standing crop biomass, if ungrazed, is an estimate of vegetation production at the site, which is an important indicator of the health of the site and its value for livestock production. We purposely sampled in winter grazing areas, where there was limited grazing during the growing season, so our measures of biomass largely quantified production and not grazing offtake in these utilization cage. We separated biomass samples into seven functional groups, including grasses, forbs, shrubs (including subshrubs), sedges, litter, standing dead, and the large grass, *Acnatherum splendens*. Standing crop biomass of herbaceous plants was clipped in 5 quadrats in each plot at the base of the plant for grasses, forbs, sedges and *A.splendens*. We separated out *A. splendens* because it was very patchy and when present, usually produced very high biomass. For shrubs and subshrubs, we used the representative branch method or collected shrub leaves and current year's growth of twigs within at 3-D projection of the plot frame, regardless of

whether the shrub was rooted inside or outside the frame (Bonham and Ahmed 1989). For litter and standing dead, we clipped or picked up all detached pieces inside the quadrat frame. We determined the size of the quadrat by the amount of biomass in the ecological zone, using a 50 x 50 cm quadrat in the mountain and forest steppe, eastern steppe and steppe and 1 x 1 m quadrat in some of the steppe sites and in the all desert steppe. All samples were dried in a drying oven at 60° C for 48 hours in the laboratory and then weighed to an accuracy of +/- 0.01 grams.

We measured forage quality, crude protein and acid detergent fiber (ADF) on a subset of the functional group samples (ADF analysis sample size: mountain and forest steppe-41, eastern steppe-12, steppe-33, desert steppe-38; Crude protein analysis sample size: mountain and forest steppe-145, eastern steppe-24, steppe-119, desert steppe-111) at the Feed Evaluation Laboratory of Research Institute of Animal Husbandry in Mongolia. The ANCOM technology was used for acid detergent fiber analysis, whereas Kjeld-Foss automated macro-Kjeldahl method was used for crude protein analysis. All 5 samples of functional groups from each of the 5 biomass quadrats sampled in each plot were mixed and ground before analysis. Both crude protein and ADF analyses were run in duplicate. If there was a large difference between the duplicates then the analysis was repeated until repeat measures were nearly identical.

Plant foliar and basal cover by species were measured using the line point intercept (LPI) method (Herrick et al. 2005) with points dropped every meter along each of the five, 50-m transects for a total of 250 points per plot. Foliar cover was measured as the area of ground covered by vegetation leaves. Small openings in the canopy and intraspecific overlap were excluded and thus foliar cover is always less than canopy cover, since the latter sums up the overlap in different layers of the canopy. All nomenclature in this study follows Grubov (1982). Species richness data were collected by searching for all species within the entire 50 x 50 m plot.

This was done by walking zig-zag through the plot and recording all species observed. Each species was scored on the datasheet according to their functional or life form group (perennial grass, annual grass, perennial forb, annual forb, perennial sedge, shrub and sub-shrub). Any additional species found during LPI measurements but not in the species search were added to the total species list for the plot.

The gap between perennial plants bases was measured along transects 2 and 4 using the basal gap intercept method (Herrick et al. 2009). We only recorded gap sizes that were larger than 20 cm between perennial plant bases to capture the larger gaps efficiently. A gap was defined as the distance between perennial plant bases with a minimum base size of a single perennial plant stem (1 mm). In the desert steppe (Gobi), crowns of apparently dead plants (e.g. *Stipa*, *Allium*) that were buried under the soil were counted perennial plant bases. These were detected by running one's fingers along the soil at the edge of transect.

In the analysis, we tested the effects of CBRM management on the palatability of forage species. For palatability, we used the classification of Damiran (2005) for the palatability of species in the dormant season (winter) when livestock are present at the winter shelter. The palatability classes include preferred, desired, consumed but undesirable, not consumable and toxic. For the ease of analysis and interpretation we re-grouped the above classes into two classes as palatable and unpalatable. The palatable group includes both preferred and desired classes, and the unpalatable group includes undesirable, not consumable and toxic classes. We focused on plant palatability during the dormant season to measure the effect of livestock selectivity on plants in these pastures in this season.

To understand the effect of grazing on soil surface conditions and plant-gap patterns, we recorded soil resource retention and soil redistribution classes (Burkett et al. 2012). Resource

retention class, which describes the spatial patterning and connectivity of persistent vascular plant patches and inter-patches across the plot, was recorded for the whole plot from most to least connected in the following 6 classes: **1**-Interconnected persistent plant cover or dense bunchgrasses and surrounding round interpatch areas < 30 cm wide, **2**-Persistent plants interconnected and surrounding round/oval interpatch areas > 30 cm wide, **3**-Persistent plant patches fragmented by elongated interpatch areas that are bounded in the plot, **4**-Persistent plant patches fragmented by elongated interpatch areas that cross through the plot in one direction, **5**-Interpatch areas interconnected and crossing the plot in several directions or isolated plant patches, and **6**-Interpatch areas interconnected; scattered or no persistent plants.

Soil redistribution class, which describes the extent and severity of erosion and deposition on a plot, was recorded from least to most redistribution in these 8 classes: **0**-No evidence of erosion deposition, **1**-Very slight soil redistribution, **2**-Patchy, slight (< 5 cm) soil loss and deposition, **3a**-Extensive, moderate soil loss (< 10 cm), **3b**-Extensive, moderate soil redistribution (< 10 cm), **4a**-Extensive, severe erosion (> 10 cm); little deposition, **4b**-Extensive, severe erosion (> 10 cm) with patchy sediment deposition, **4c**-Extensive, severe sediment deposition (> 10 cm).

After sampling, we developed an ecological site key for all our plots and classified each plot to an ecological site (Reid and Chantsallkham unpublished). This resulted in the following ecological sites for our analysis: Clay, HighWaterTable, Loam, Rocky, Hill, Shallow and Sand ecological sites. We then used ecological site as a variable in the analysis to test the effects of site type on soils and vegetation, and to uncover any interactions among our two main effects of CBRM and ecological site.

4.2.3. Data analysis

Data were analyzed with the statistical package SAS 9.3 for Windows. We corrected the non-normality of the data using a log ($y+1$) transformation on biomass and an arcsine transformation on cover data, sheep/goat pellets and gap data. When log and arcsine transformation did not achieve normality, we used ranking and analyzed the ranks. Some of the cover data contained large numbers of zeros; here, we transformed original data into binary codes (=presence/absence) when more than half the values were zero.

We used a model type III ANOVA to assess the effects CBRM and ecological site, and the interaction effects of these two variables. Our dependent variables were standing crop biomass, forage quality, foliar and basal cover, species richness, plant palatability, cover of dominant species and open gaps at the plots. Two-way ANOVAs followed by Tukey-adjusted multiple tests were used for multiple comparisons of vegetation variables among the CBRM and ecological sites. Because the effects of ecological zone were so large, we ran analyses separately for each ecological zone.

We used the p-values generated during the tests on transformed data whenever the transformation was warranted. We report least squared means of untransformed data when transformation was not necessary and if test results of both transformed and untransformed data were similar. If test results between untransformed and log, square root and arcsine transformed data were different, we used least square means of back-transformed data in the graphics. If test results between untransformed and ranked and binary data were different we used least square means of untransformed data in the graphics.

For soil surface characteristics, we used a Chi-square to test for differences in soil surface characteristics comparing between the pastures of 2 management types. We did lump some

classes for the resource retention and soil redistribution variables because of small sample sizes. Our rule for regrouping classes was to ensure there were at least $n=3$ plots in each cell in the analysis. Based on the similarity in landform and possible infiltration of rain water we did lump the clay ecological site with high water table site and lumped the rocky-hill-shallow and sand sites.

We report a p-value of ≤ 0.05 as ‘significant’, to balance Type I and Type II errors.

4.3. Results

4.3.1. Livestock densities and dung frequency in CBRM and non-CBRM winter pastures

Livestock numbers (in SFUs) herded by families in CBRM groups differed greatly from those herded by families in traditional neighborhoods. In the mountain and forest steppe, there were 54% more SFUs of all species and double the number of sheep and goats in CBRM pastures than non-CBRM pastures. In the eastern steppe there were 13% more SFUs and 69% more horses in CBRM than non-CBRM pastures. In the steppe, there were 135% more SFUs and 3 times as many cattle in CBRM than non-CBRM pastures. In contrast, there were 30% fewer SFUs in CBRM than non-CBRM pastures in the desert steppe, partly because there were so few horses in CBRM pastures (Table 4.1).

Table 4.1. Mean (\pm SD) livestock numbers, presented in Sheep Forage Units (SFUs) by species at winter shelters sampled in this study by management type in the four ecological zones.

Ecological zones and community based management groups		Average SFUs					
		Sheep	Goat	Cattle	Horse	Camel	Total
Mountain and forest steppe	CBRM	501 (± 507)	277 (± 282)	861 (± 1085)	509 (± 465)	0	2148 (± 469)
	Non-CBRM	243 (± 319)	186 (± 223)	547 (± 712)	422 (± 569)	0	1398 (± 1460)
Eastern steppe	CBRM	993 (± 1367)	320 (± 313)	517 (± 421)	4183 (± 8350)	293 (± 570)	6306 (± 10852)

	Non-CBRM	1970 (±3688)	325 (±313)	820 (±1053)	2471 (±3157)	0	5586 (±7931)
Steppe	CBRM	694 (±1180)	338 (±363)	236 (±251)	476 (±558)	10 (±29)	1754 (±476)
	Non-CBRM	299 (±528)	150 (±109)	72 (±89)	184 (±268)	40 (±147)	746 (±873)
Desert steppe	CBRM	193 (±223)	222 (±195)	96 (±205)	153 (±252)	74 (±119)	738 (±829)
	Non-CBRM	288 (±518)	246 (±273)	78 (±132)	284 (±387)	151 (±170)	1047 (±1205)

At the *soum* level, there was little or no difference in the density of sheep forage units of all livestock species in CBRM vs non-CBRM *soums* except in the mountain and forest steppe (Figure 4.1). In contrast to the patterns at the winter shelter level, there were 27% more SFU/ha in the non-CBRM managed *soums* than the CBRM-managed *soums* in the mountain and forest steppe. In the eastern steppe and steppe zones, CBRM managed *soums* had slightly more SFU/ha than non-CBRM *soums*. In the desert steppe, there was no difference in SFUs between *soums*.

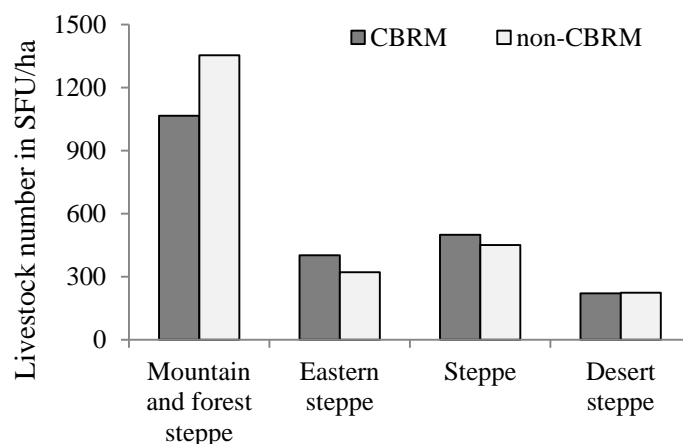


Figure 4.2. Sheep forage units (SFU) per hectare across all livestock species in our study *soums* in four ecological zones.

4.3.2. Effects of CBRM management on vegetation and soils and interactions with ecological sites

The relative significance of CBRM and ecological site differed by ecological zone, and most of the interactions between these two main effects, although few, occurred in the desert steppe. In the mountain and forest steppe, vegetation and soils were affected more by ecological site (12.2% out of total 74 tests were significant) than CBRM (2.7% out of total 74 tests were significant). In the eastern steppe, we tested CBRM alone, not ecological site. CBRM here was significant in 3.5% out of 57 tests. In the steppe, ecological site was more important (14.5% out of total 62 tests were significant) than the CBRM (8% out of total 62 tests were significant) or the CBRM and ecological site interaction (3.2% out of total 62 tests were significant). In the desert steppe, the CBRM and ecological site interaction effect were more important (5.3% out of 57 total tests were significant) than the main two effects (8.8% out of total 57 tests were significant for both effects).

Variables with significant interactions between CBRM and ecological site were as follows. In the mountain and forest steppe, medium gaps (101-200cm) between perennial plant bases, presence of individual species such as *Thalictrum simplex* showed significant interactions between CBRM and ecological site. There was a significant ($p=0.03$) CBRM*ecological site interaction for medium size gap between perennial plant bases. The medium size gap between perennial plant bases was not affected by ecological site in CBRM pastures, but was greater in RHSS than ClayHWT and Loam sites in non-CBRM managed pastures (Figure 4.3).

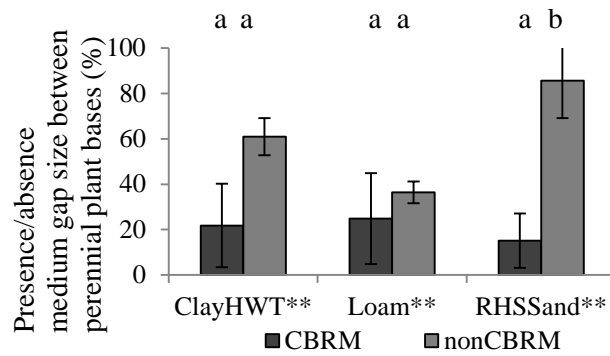


Figure 4.3. Effects of ecological site on a) medium gap size (101-200 cm) between perennial plant bases in pastures managed by CBRM and non-CBRM groups in the mountain and forest steppe. Bars with different letters above them were significant at $p \leq 0.05$. Variables marked with a ** were analyzed as binary variables, and are presented here on a binary scale.

The effect of CBRM was greater ($p=0.04$) on the presence *Thalictrum simplex*, perennial forb, on high infiltration soils (RockyHillShallowSand) than low infiltration soils (ClayHighWaterTable and Loam) compared to the other management type (Figure 4.3a).

In the steppe, the cover of dormant season unpalatable subshrubs and the cover of dominant species, *Stipa gobica* showed significant interactions between these two main effects. The cover of *Stipa gobica* was greater ($p=0.03$) on high infiltration RHSS site in CBRM pastures than in non-CBRM pastures (Fig 4.4b). The cover of unpalatable subshrub during dormant season ($p=0.02$) was greater on HWT site than ClayLoam site in CBRM pastures. But the opposite was true where the cover of unpalatable subshrubs was greater on the ClayLoam site than on the HWT site in non-CBRM pastures.

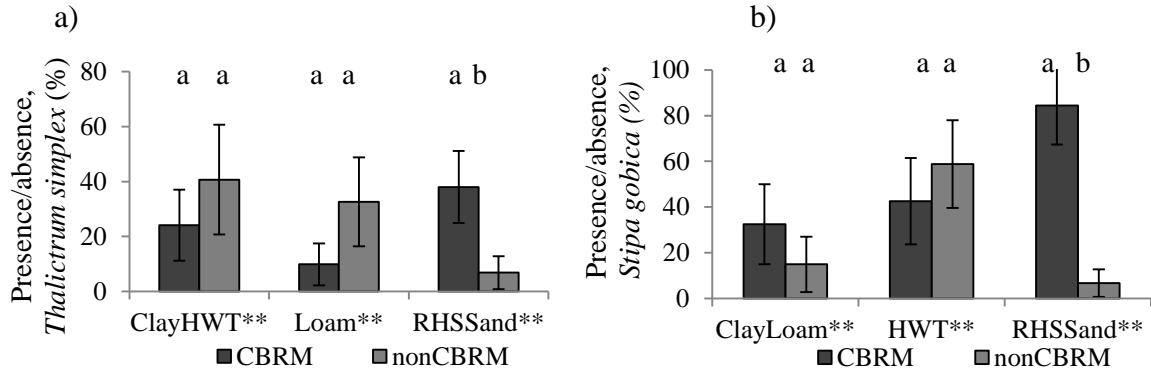


Figure 4.4. Effects of ecological site effect on dominant species. a) perennial forb *Thalicttrum simplex* in the mountain and forest steppe, b) perennial grass *Stipa gobica* in the steppe in comparing pastures managed by CBRM and non-CBRM groups. Bars with different letters above them are significant at $p \leq 0.05$. ClayHWT=Clay HighWaterTable, RHSSand=Rocky Hill Shallow Sand. Variables marked with a ** were analyzed as binary variables and are presented here on a binary scale.

In the desert steppe, total foliar cover, cover of grass and perennial plants had significant interaction effects. The cover of grasses ($p=0.02$) and perennial plants ($p=0.02$) were not affected by ecological site for the CBRM-managed pastures, but was greater in ClayLoamRHSS than HWT sites in non-CBRM-managed pastures (Figure 4.5a,b).

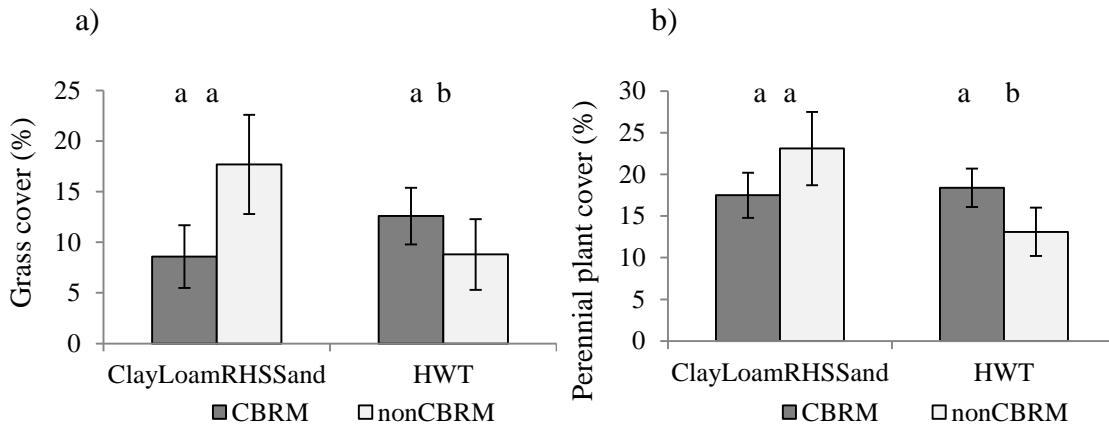


Figure 4.5. Interactions between CBRM and ecological site effects on a) grass and b) perennial plant cover in the desert steppe. Bars with different letters above them are significant at $p \leq 0.05$.

4.3.2.1. Standing crop biomass

CBRM had no significant effect on total green standing crop biomass in any ecological zone (mountain and forest steppe ($p=0.80$), eastern steppe ($p=0.87$), steppe ($p=0.0.80$) and the desert steppe ($p=0.89$) (Figure 4.6).

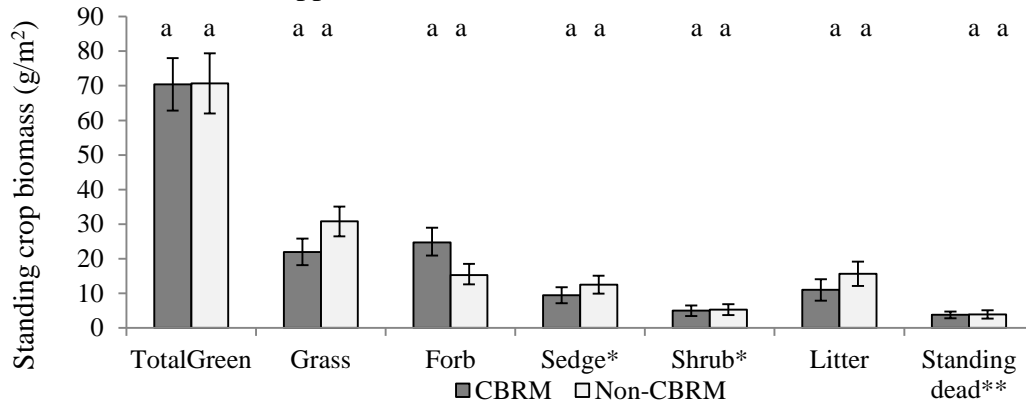
There was no significant difference in the six functional types of standing crop biomass comparing between the two management types in the mountain and forest steppe. These results are consistent with dung frequency results and do not support our *pasture condition hypothesis* that CBRM pastures will be in better condition than in non-CBRM pastures.

Similarly, in the eastern steppe, standing crop biomass by plant functional type did not differ between the management types, except for litter biomass which had greater ($p=0.02$) in the pastures managed by CBRM than non-CBRM groups (Figure 4.6b). This weakly supports our *pasture condition hypothesis* of improved management in CBRM vs non-CBRM-managed pastures.

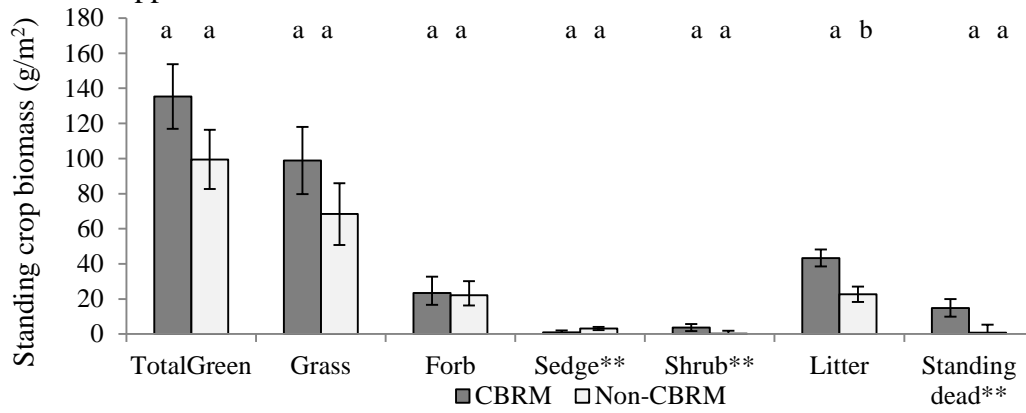
In the steppe zone, shrub ($p=0.01$), litter ($p=0.03$) and standing dead ($p=0.03$) biomass was significantly greater in CBRM than non-CBRM pastures despite more dung in CBRM than non-CBRM pastures (Figure 4.6c). This is weak support for our *pasture condition hypothesis*, but also the strongest support for our hypothesis in any ecological zone.

There were no interactions between CBRM and ecological site for any biomass variables in any zone.

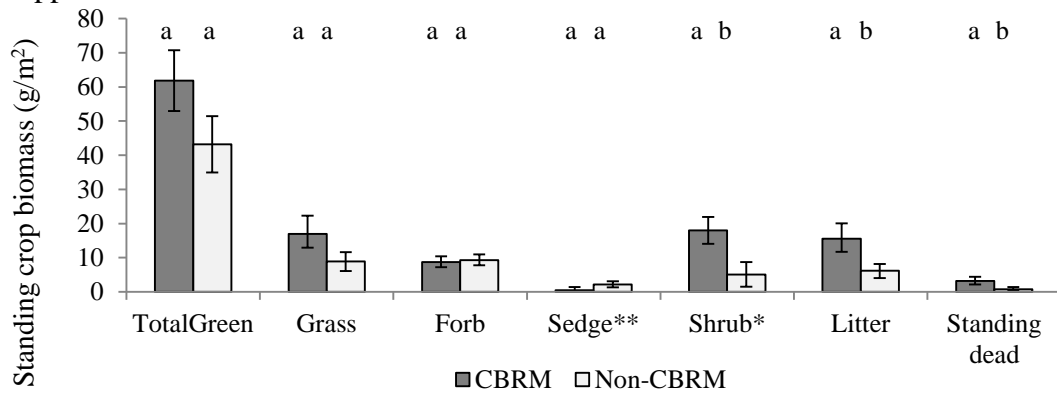
a) Mountain and forest steppe



b) Eastern steppe



c) Steppe



d) Desert steppe

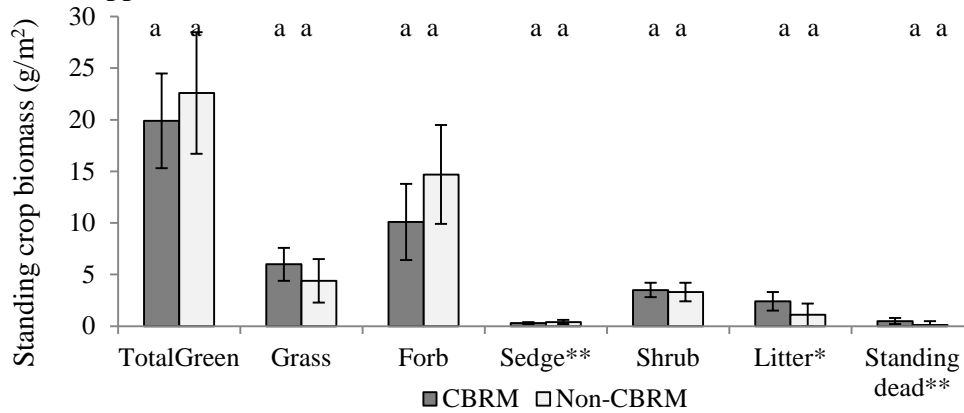


Figure 4.6. Standing crop biomass in pastures managed by CBRM and non-CBRM groups in the a) mountain and forest steppe, b) eastern steppe, c) steppe and d) desert steppe. Bars with different letters above them are significant at $p \leq 0.05$. * - represents variables were significantly different using a ranked transformation, ** - represents variables were significantly different using binary data, but are represented here on the original scale

4.3.2.2. Plant total and functional type cover

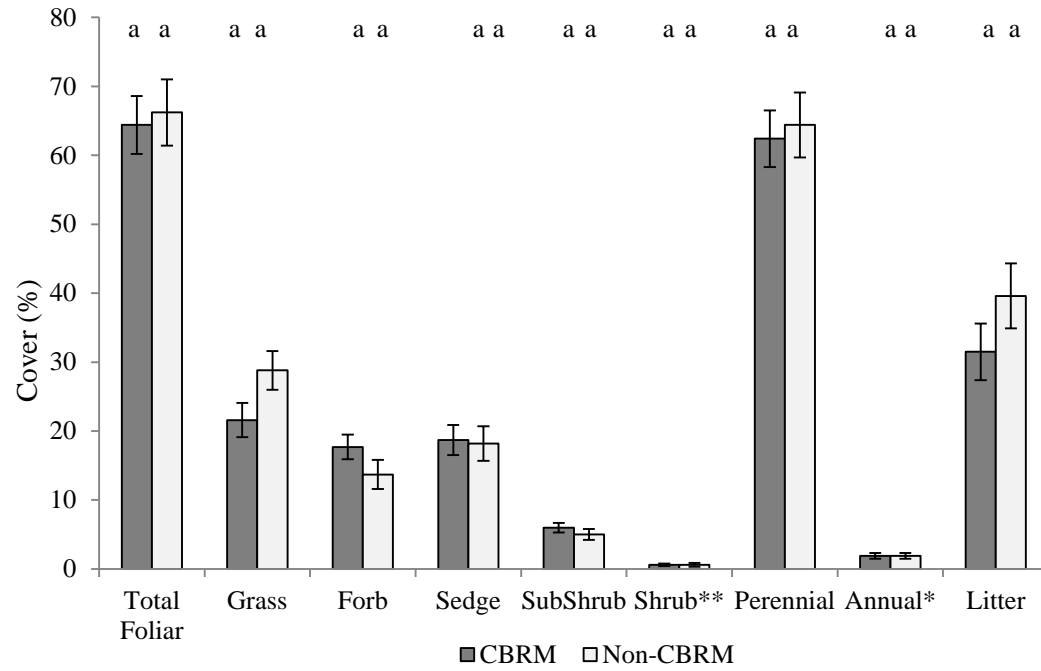
There were no significant and few significant differences in plant cover comparing between CBRM and non-CBRM managed pastures in any ecological zone (Figure 4.7). In the mountain and forest steppe, there were no differences in total or functional type cover comparing between the pastures managed by the CBRM and the non-CBRM groups (Figure 4.7a).

In the eastern steppe, annual plant cover ($p=0.01$) was greater in the pastures managed by non-CBRM than the CBRM groups (Figure 4.7b).

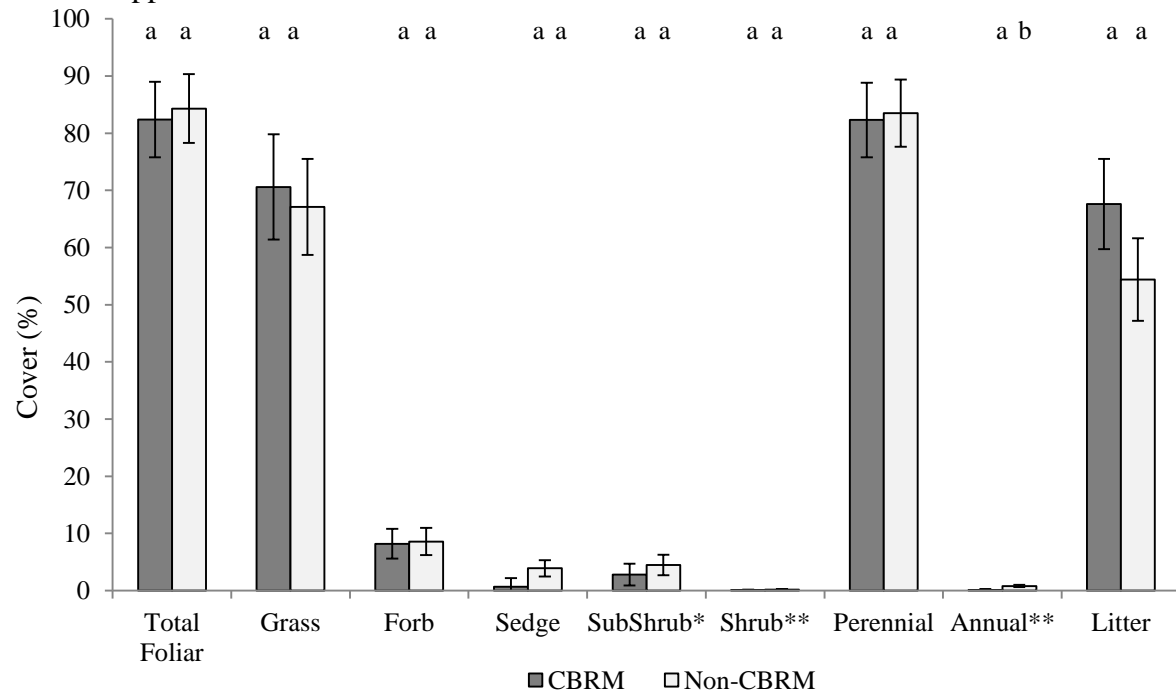
In the steppe, CBRM managed pastures had greater litter cover ($p=0.02$) than the non-CBRM managed pastures, but no other differences (Figure 4.7c).

In the desert steppe, there were no differences in total vegetation cover or functional type cover between the CBRM and non-CBRM managed pastures (Figure 4.7d).

a) Mountain and forest steppe



b) Eastern steppe



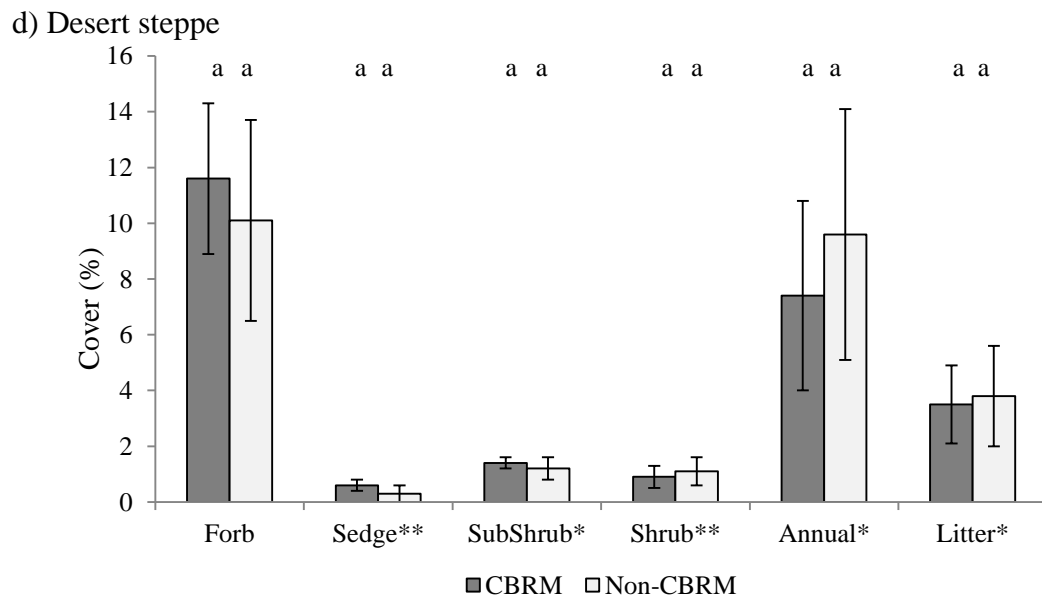
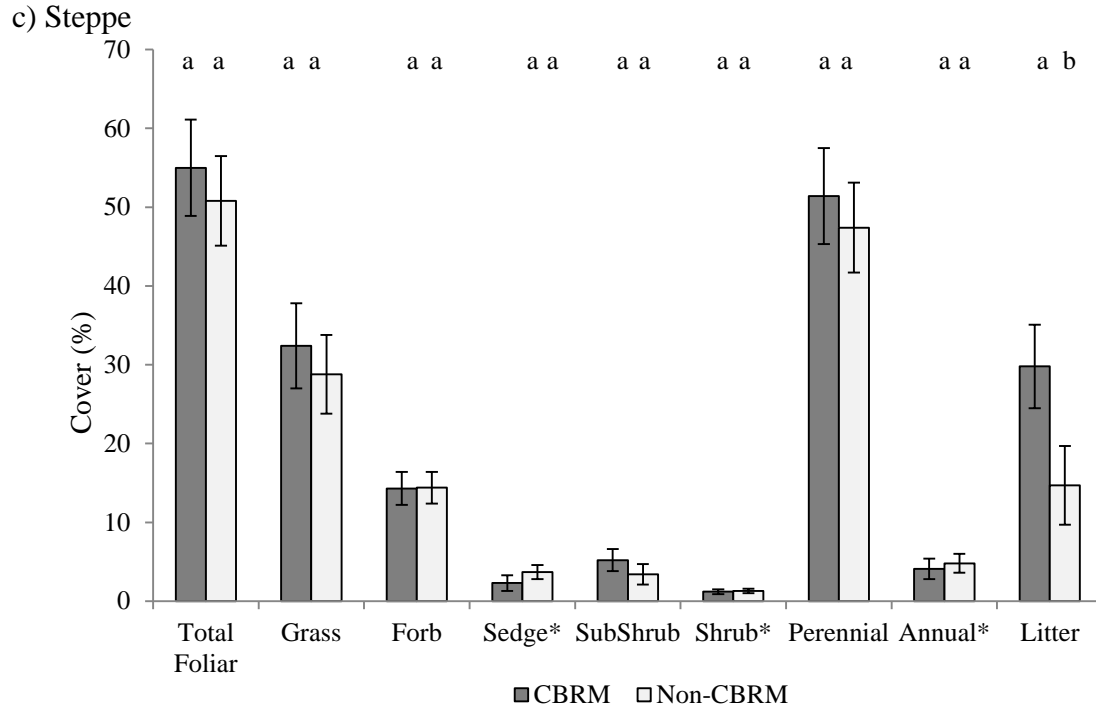


Figure 4.7. Total and functional type cover in pastures managed by CBRM and non-CBRM groups in the a) mountain and forest steppe, b) eastern steppe, c) steppe, d) desert steppe. Bars with different letters above them are significant at $p \leq 0.05$. Variables marked with a * were analyzed as ranked variables, with a ** were analyzed as binary variables, but are presented here on their original scale.

4.3.2.3. Species cover and species richness

Of the 28 dominant species we tested in the mountain and forest steppe, management type only significantly affected the cover of one grazing tolerant grass, *Cleistogenes squarrosa*; otherwise there were no significant effects of management on other species. Here, there was significantly more *Cleistogenes squarrosa* in non-CBRM pastures than CBRM pastures ($p=0.01$).

In the eastern steppe, we found only one significant difference in the cover of dominant species comparing between the pastures managed by CBRM and non-CBRM groups out of 16 dominant species we tested. The cover of the grazing-tolerant sedge, *Carex duriuscula*, was lower in the pastures managed by CBRM than the pastures managed by non-CBRM groups.

In the steppe, of the 18 dominant species we tested, CBRM affected the cover of one perennial forb, *Kochia prostrata*, which was greater in the pastures managed by CBRM groups than by the non-CBRM pastures.

Of the 16 dominant species we tested in the desert steppe, CBRM affected (main effect) only the cover of the annual grass *Eragrostis minor*, with more cover of this species ($p=0.04$) in the pastures managed by non-CBRM than by the CBRM groups. We did not find any interaction effect between CBRM and ecological site for all dominant species we tested.

4.3.2.4. Bare soil, basal cover and vegetation gaps

We found no differences in the cover of perennial plants bases and bare soil between pastures managed by CBRM and non-CBRM groups in any ecological zone (Figure 4.8). And only in the desert steppe did CBRM pastures have proportionately smaller average gaps ($p=0.02$)

and also fewer large gaps ($p=0.04$) between perennial plant bases than in the non-CBRM pastures (Figure 4.9d).

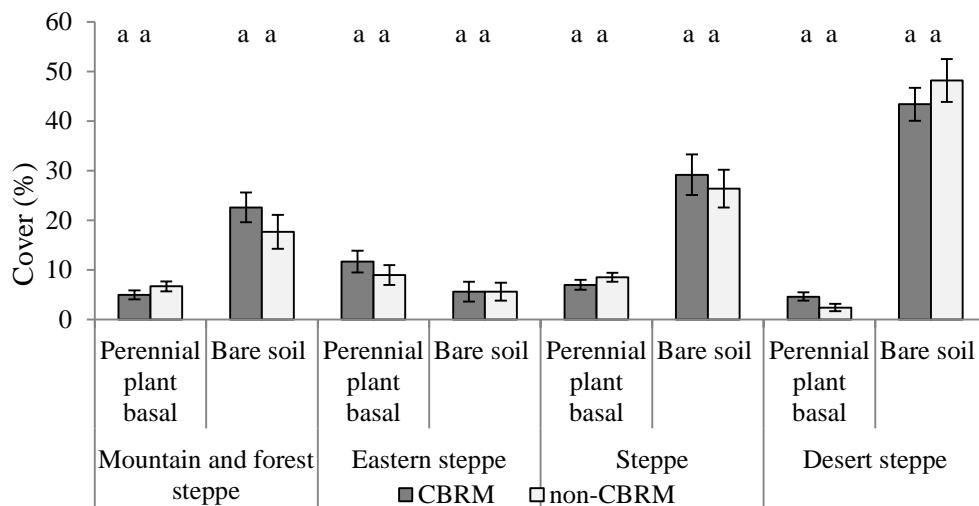
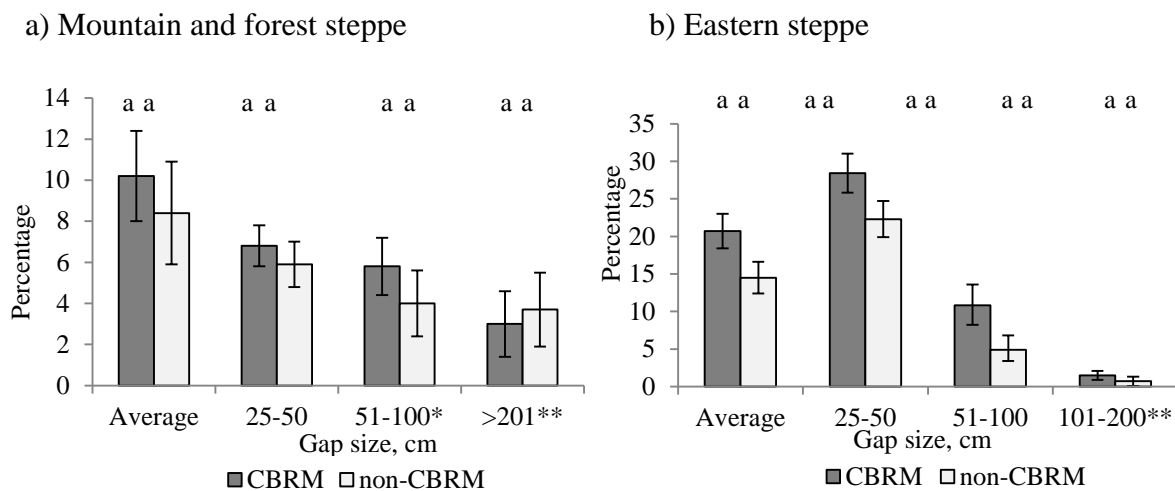


Figure 4.8. Cover of perennial plant bases and bare soil in the pastures managed by CBRM and non-CBRM groups in four ecological zones. Bars with different letters above them were significant at $p \leq 0.05$.



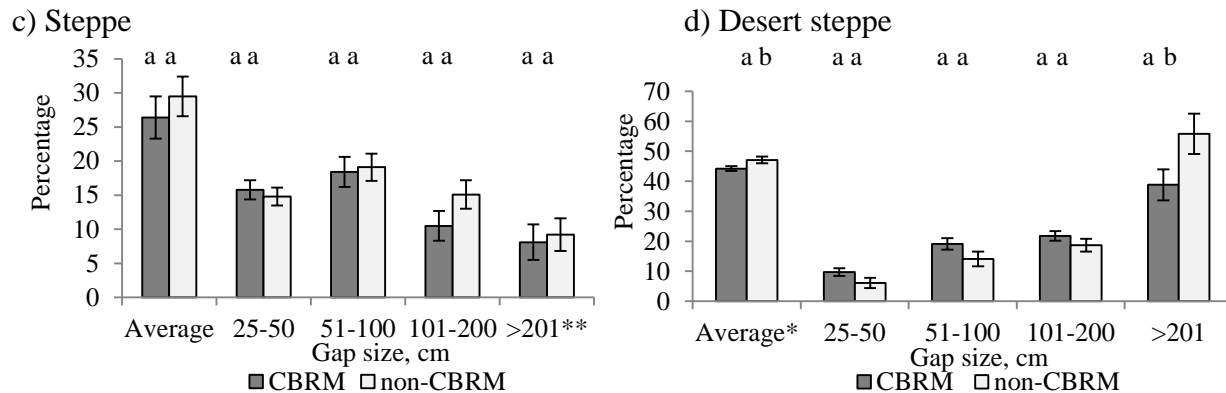
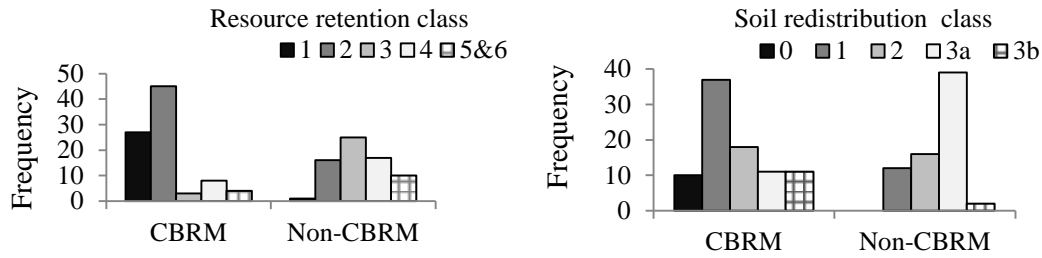


Figure 4.9. The effects of CBRM and non-CBRM management on the average gap and gap size classes, measured between the bases of perennial plants, in the a) mountain and forest steppe, b) eastern steppe, c) steppe, d) desert steppe. Bars with different letters above them were significant at $p \leq 0.05$. Variables marked with a * were analyzed as ranked variables, with a ** were analyzed as binary variables, but are presented here on their original scale. Note that for the mountain and forest steppe, there was a significant interaction between CBRM and ecological site for medium gap sizes (101-200 cm) as shown in figure 4.3. For the eastern steppe, there were no gaps bigger than 201 cm.

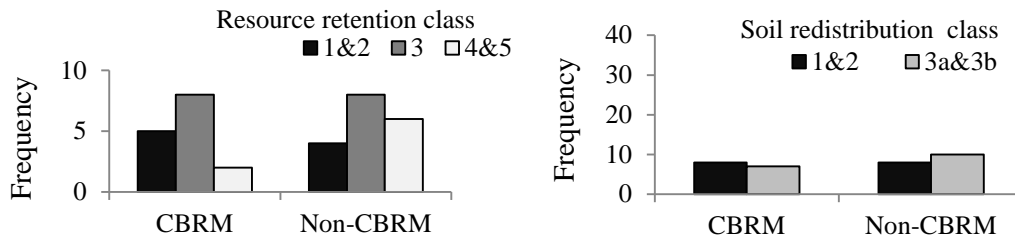
4.3.2.5. Soil surface characteristics

To understand if CBRM management affected both plant patch pattern and soil erosion, we used a Chi-square to test the differences in the (resource retention) size and connectivity of persistent vascular plant patches and inter-patch areas across a plots and the extent and severity of soil redistribution processes (soil erosion and deposition by wind and water). Low numbers in soil resource retention classes indicate smaller open patches and greater connectivity of vascular plant patches, with higher numbers indicating fragmented and isolated patches. For soil redistribution classes, low numbers indicate healthy conditions with less soil movement, whereas high numbers indicate extensive and severe movements of soil.

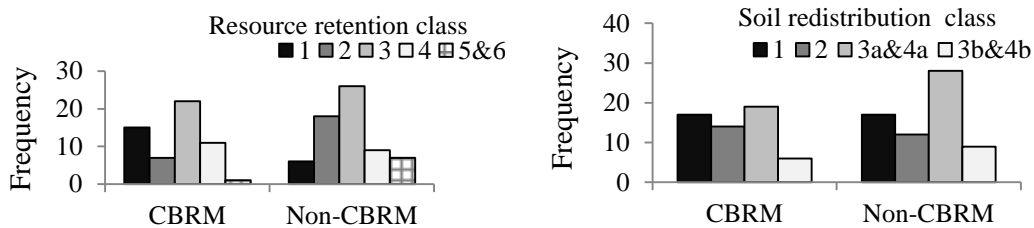
a) Mountain and forest steppe



b) Eastern steppe



c) Steppe



d) Desert steppe

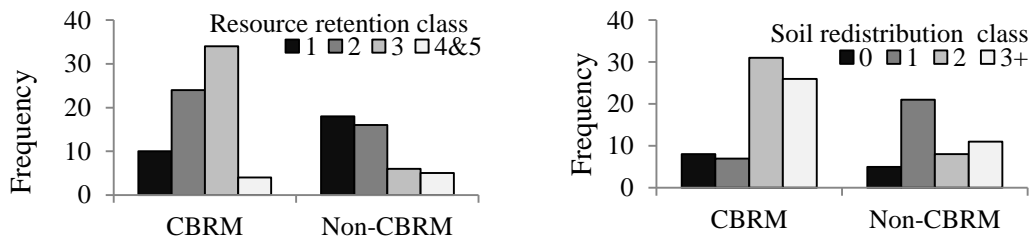


Figure 4.10. The effects of CBRM vs non-CBRM management on soil surface characteristics (resource retention class and soil redistribution class) in the a) mountain and forest steppe, b) eastern steppe, c) steppe and d) desert steppe.

In the mountain and forest steppe, persistent vascular plant patches were highly connected ($p < 0.0001$) and soil movement was low ($p < 0.0001$) in CBRM compared with non-CBRM pastures. In the eastern steppe, there was no CBRM effect on either measure (resource retention $p = 0.40$; soil redistribution, $p = 0.61$). In the steppe zone, CBRM pastures had more highly connected plant patches than non-CBRM pastures (resource retention, $p = 0.01$), but there was no significant difference in soil movement (soil redistribution, $p = 0.64$). In contrast, in the desert steppe, CBRM pastures had less connectivity between plant patches (resource retention, $p = 0.0004$) and more soil erosion (soil redistribution, $p < 0.0001$) than non-CBRM pastures.

4.3.2.6. Forage quality and vegetation palatability

We found no significant difference in crude protein and acid detergent fiber or in the palatability of plant species between the CBRM and non-CBRM pastures in any ecological zone (Figures 4.11 and 4.12), except in the steppe where ecological site had effect ($p = 0.02$) on the presence of dormant season unpalatable subshrub in both CBRM pastures and in non-CBRM pastures (Figure 4.12d). The effect of non-CBRM was greater for the presence of dormant season unpalatable subshrubs on ClayLoam and RHSS site compared to CBRM management. But the pattern was opposite on HWT site where the effect of CBRM was greater than non-CBRM management.

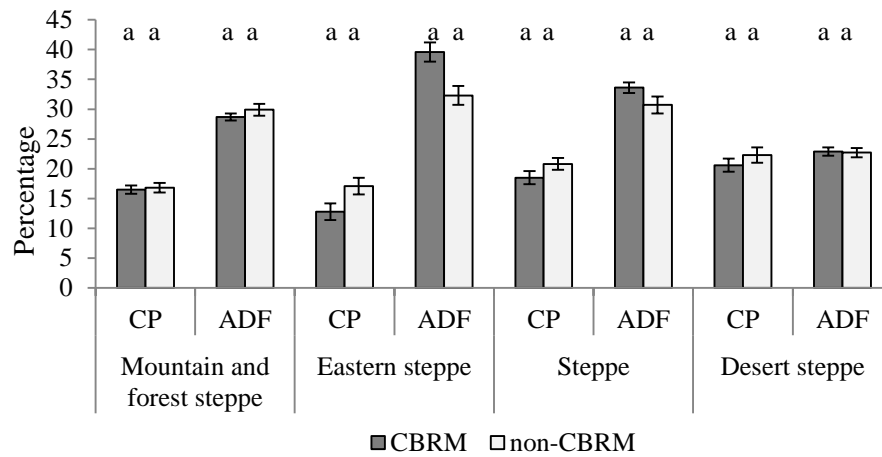
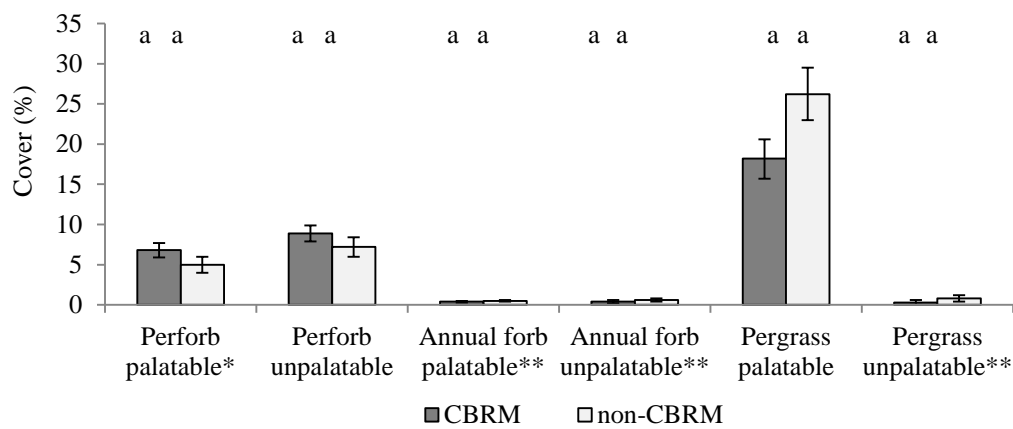
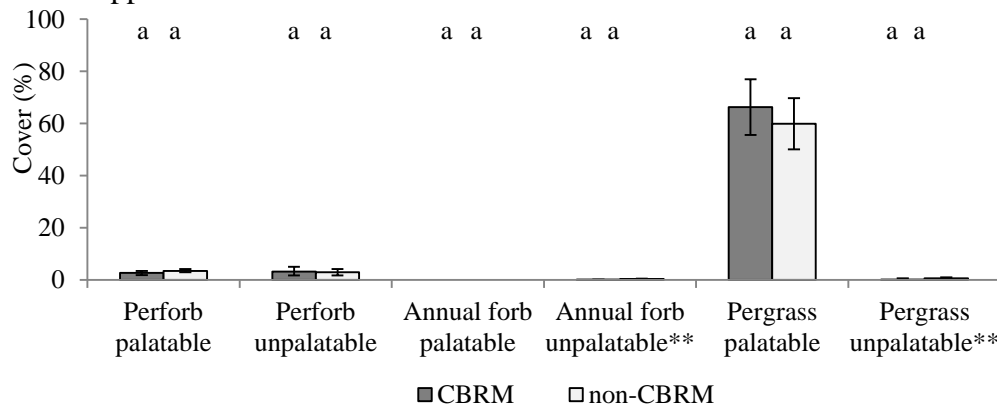


Figure 4.11. The effects of community-based management on crude protein (CP) and acid detergent fiber (ADF) in forage in the four ecological zones. Bars with different letters above them are significant at $p \leq 0.05$.

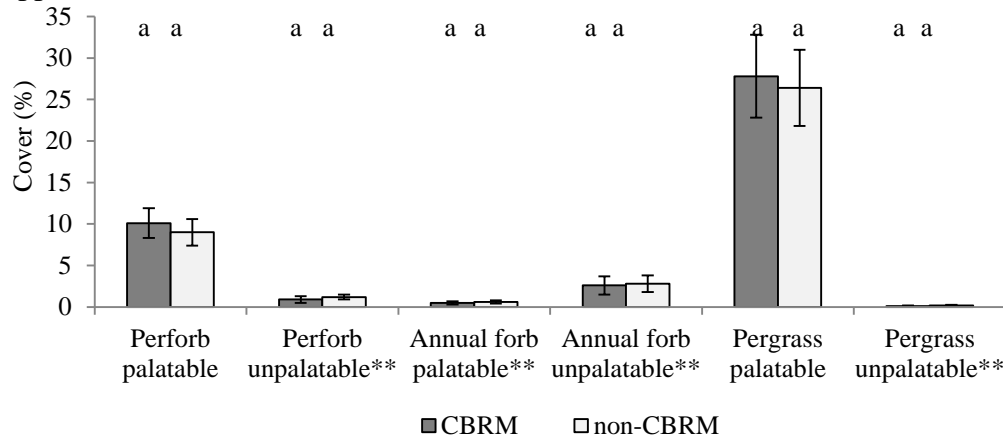
a) Mountain and forest steppe



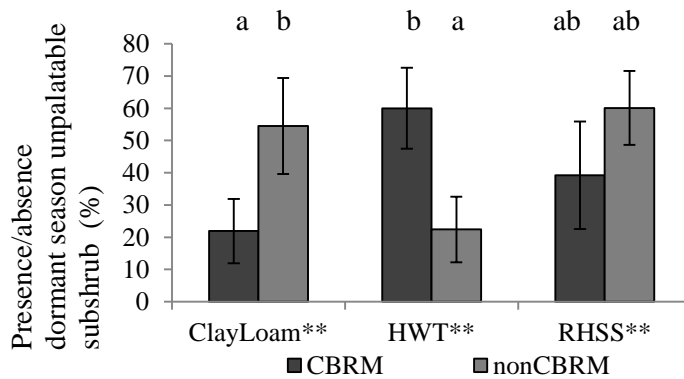
b) Eastern steppe



c) Steppe



d) Steppe: CBRM and ecological site interaction effect for dormant season unpalatable subshrubs



e) Desert steppe

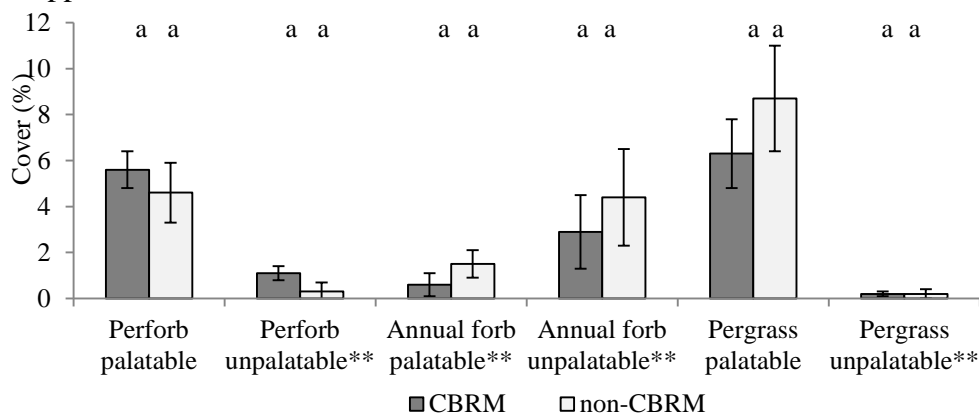


Figure 4.12. Differences in cover of grasses and forbs that are palatable and unpalatable during the growing season in pastures managed by CBRM and non-CBRM groups in the a) mountain and forest steppe, b) eastern steppe, c) steppe e) desert steppe and) CBRM * ecological site interaction effect for dormant season unpalatable subshrub . Bars with different letters above them are significant at $p \leq 0.05$. Variables marked with a * were analyzed as ranked variables, with a ** were analyzed as binary variables, but are presented here on their original scale. Perforb=perennial forb, pergrass=perennial grass.

4.3.3. Effects of ecological sites on vegetation and soils

In general, ecological site was relatively important in all 3 ecological zones where we used this main effect in the analysis (see results in Appendix Tables 4.1 and 4.2). In the mountain and forest steppe, total foliar cover ($p=0.002$), forb cover ($p=0.02$), perennial plant cover ($p=0.005$), litter cover ($p=0.01$) and bare soil cover ($p=0.005$) were greater in the ClayHWT ecological site than the RHSS site (see appendix 4.1). Cover of unpalatable perennial forbs in both the growing ($p=0.03$) and dormant ($p=0.02$) seasons was greater in the loam site than in the RHSS site. By contrast, small (25-50cm) and medium (51-100cm) gap sizes ($p=0.008$ and $p=0.006$ respectively) between perennial plants were greater in the RHSS site than the ClayHWT site.

In the steppe, the effects of ecological site were more significant, where grass biomass ($p=0.03$), cover of perennial plant bases ($p=0.05$) and species richness ($p=0.04$) were greater in RHSS sites than HWT sites. Forb biomass ($p=0.02$) was greater in RHSS sites than ClayLoam sites, whereas grass cover ($p=0.03$) was greater in ClayLoam sites than HWT sites. Palatable perennial grass cover in both the growing ($p=0.003$) and dormant seasons ($p=0.008$) was greater in the ClayLoam than in the Rocky Hill Shallow Site. Cover of the dominant grass, *Agropyron cristatum*, ($p=0.002$) was greater in the Clayloam, Rocky Hill Shallow Sites than the HWT site.

In the desert steppe, ecological site was less significant than in the steppe and mountain and forest steppe zones. Overall, the HWT table site was less productive than the ClayLoamRHSS site, where the cover of perennial plants ($p=0.04$) and total green biomass ($p=0.05$) were all lower in the HWT site than the ClayLoamRHSS site. Note that we did not compare different ecological sites in the eastern steppe.

4.4. Discussion

4.4.1. Livestock densities in CBRM and non-CBRM winter pastures and across *soums*

Overall, herders kept more livestock at their winter shelters in CBRM than non-CBRM-managed pastures in 3 of the 4 ecological zones: all except the desert steppe. In this driest region, herders kept more livestock in non-CBRM than CBRM-managed pastures.

Generally, where there were more livestock at winter shelters, there were also more livestock at the wider *soum* level, with the exception of the mountain and forest steppe. Here, livestock densities (in SFUs) were much higher at the winter pastures managed by CBRM groups than the winter pastures managed by non-CBRM groups. In addition, at the *soum* level, there were fewer livestock in CBRM *soums* than non-CBRM *soums*. This implies that herders in CBRM pastures may have the opportunity to herd their livestock (particularly their abundant sheep and goats) farther away from winter shelters in the CBRM *soums* with fewer livestock populations than in the more populated non-CBRM *soums*.

Herding families at winter shelters in the CBRM-managed pastures hold 2-3 times more livestock as families in non-CBRM managed pastures. But, CBRM *soums* have about the same density of livestock as the non-CBRM *soums*. This again implies that our randomly sampled families may be unusual in CBRM *soums*.

4.4.2. Effects of CBRM management on vegetation, forage quality and soil surface characteristics

Our study is the first of its kind to compare the effects of CBRM across Mongolia's major ecological zones. Our results suggest that CBRM initiatives are not having *major* impacts

on any aspect of winter pastures across ecological zones. But these results are somewhat deceptive. There are more livestock at winter camps in CBRM than non-CBRM pastures in the mountain and forest steppe, eastern steppe and steppe, and yet these CBRM pastures near these camps were as healthy as winter camps with smaller livestock herds managed by non-CBRM herders.

But there were some *subtle* significant effects of CBRM on pastures. In the mountain and forest steppe, we found only four significant differences between CBRM and non-CBRM managed pastures. There was less cover of increaser grass species, *Cleistogenes squarrosa*, more connected plant patches and less erosion and less unpalatable subshrub during the dormant season in CBRM than non-CBRM pastures. This appears to show the very beginning impacts of CBRM pasture management on rangelands, although these effects are weak.

In the eastern steppe, we found two significant differences where CBRM pastures had greater litter biomass and less cover of annual. Like the mountain and forest steppe, these results indicate the beginning of impacts of CBRM rangeland management. Unlike other zones, CBRM did not affect the soil surface indicators in the eastern steppe.

In the steppe, there was more significant litter cover, litter, shrub and standing dead plant biomass in CBRM pastures than non-CBRM pastures, suggesting that CBRM may improve condition of pastures in the steppe, which is remarkable considering that winter camps managed by CBRM herder families have 2.4 times more livestock than winter camps managed by non-CBRM herder families. There is less erosion and more highly connected plant patches in the CBRM compared to non-CBRM pastures in the steppe. This occurs despite the larger livestock herds at the CBRM winter camps.

In the desert steppe alone, there are other studies with which to compare our results. Our study results are more similar to those of Addison et al. (2013), and contradict those of Leisher et al. (2013). Addison et al. (2013) used field studies and showed no effects of CBRM while Leisher used remote sensing and showed 11% increase in time-integrated NDVI in CBRM pastures vs non-CBRM pastures within each *soum*. Our results suggest very little difference between CBRM and non-CBRM pastures in the desert steppe, except less connected plant patches and more erosion in CBRM pastures, and more abundance of the annual grass, *Eragrostis minor*, in non-CBRM pastures. This is surprising because CBRM winter camps had fewer livestock than non-CBRM winter camps in the desert steppe. We acknowledge the differences in study sites and methods used and thus we need to be careful in making broad-scale comparisons of these studies. Study sites of Addison's team were 1 km away from water points, similar to our 1000 m plots, and they used vegetation and soil surface-based indicators. Study sites of Leisher's team were at the *soum* level (areas averaged over all seasonal pastures) and they used a remote sensing approach to detect growing season vegetation growth and length of growing seasons in CBRM vs non-CBRM *soums* in the desert steppe.

Here, in the desert steppe, there is some evidence of the advantage of CBRM management. Plant patches were more connected in CBRM than non-CBRM pastures and the cover of the annual grass, *Eragrostic minor*, was less. This species usually emerges after summer rains in the gaps between the shrubs (Hilbig 1995).

We found that 5-9% of our tests were significant comparing between CBRM and non-CBRM pastures out of 64-71 tests in each ecological zone. There are several possible explanations for this. We would expect that: a) it takes time for these systems to respond to a

change in management, and b) rainfall and the *dzud* effects may have had a greater impact than management differences on grazing pressure in the years immediately prior to our sampling.

Work by the social science part of our team shows that CBRM initiatives, compared to non-CBRM groups, display better information access and knowledge exchange and coping and adaptive capacity through their collective action (Fernandez-Gimenez et al. 2012a, Fernandez-Gimenez et al. 2015, Ulambayar 2015). As described in the introduction, they adopt a number of specific practices, only some of which will likely improve the environment.

The practices that improve winter pasture condition include reserving winter pasture, reducing the amount of time of winter grazing by coordinated long distance (*otorotor*) movement of herds and increased accessibility of remote pastures (Table 4.2). If families in the CBRM groups move regularly, and if they do not use winter pastures in the growing season, this reduces out of season grazing of winter pastures. Allowing winter pasture to rest for a growing season enables individual plants and plant communities to recover and accumulate stored carbohydrates which leads to healthy and productive pastures. Coordinated fall long-distance (*otorotor*) movement is a critical adaptive strategy for disaster preparation for formal CBRM organizations. Survival rate of the herd during the harsh winter increases, if the herd has enough fattening from the fall *otor* grazing (Fernandez-Gimenez et al. 2015). My field observations and communication with herders suggest that CBRM groups worked together to access remote pastures by repairing existing and digging new wells and by building bridges and roads for fall *otor* use. These initiatives helped CBRM herders to extend the duration of fall *otor* and to reduce the amount of time of spent grazing on winter pastures, thus contributing to their health. Donor projects have been supporting CBRM groups to improve livestock health and quality with the *assumption* that this will reduce grazing pressure when herders keep fewer livestock of good quality (Green Gold

project report 2010). Yet, these practices are not widespread and there are no studies that have tested the relationship between livestock health and pasture condition improvement. However, when CBRM herders fence hayfields, plant forage plants, they may be fragmenting land with unintended negative consequences (Galvin et al. 2008). On the positive side, CBRM groups participate in monitoring more often than herders in traditional neighborhoods (Fernandez-Gimenez et al. 2015), and this learning can lead to better management.

Based on our study results (Chapter 2), reducing winter grazing results in more grass biomass and fewer weedy annual forbs in the following growing season. Other studies show that reducing winter grazing increases early spring regrowth (Clark et al. 1994) and especially early spring grass regrowth (Black 1975). Thus, the length of grazing during the dormant season grazing is essential for determining rangeland health. Short periods of grazing during the winter months minimizes walking and trampling damage and allows quick regrowth in spring (Newton and Jackson 1985). Even though the social impacts of these CBRM initiatives are significant (Ulbayar 2015, Fernandez-Gimenez et al 2015), it appears that translating CBRM initiatives into ecological outcomes may take many years.

The second explanation for the minor impacts of CBRM is that abiotic disaster may be overwhelming the effects of CBRM management. In 2000 and 2009, since CBRM initiatives started, there were two nationwide natural disasters (or *dzud*, extremely cold winter combined with drought) and each caused a decline in the national livestock herd of 20-30%. A relatively short return interval between *dzud* events in Mongolia reduces grazing pressure on rangelands (Fernandez-Gimenez et al. 2012a); and in general exposure to *dzud* for both CBRM and non-CBRM groups is similar.

Table 4.2. Different practices implemented by CBRM groups that lead to different ecological outcomes.

Herding practices	Processes that lead to ecological outcomes	Ecological outcome
Reserve winter pastures	No out of season grazing → opportunity for grasses to regrow during the growing season → healthier root systems and more stored carbohydrates	More total and diverse vegetation cover and biomass of perennial grasses; less bare ground; more soil moisture.
Coordinated fall long-distance (<i>otor</i>) movement	Reduces the amount of time grazing on winter pastures → more residuals or litter cover → less snow blowing by wind and more accumulation of snow on the ground → more spring vegetation emergence (thermal and moisture regulation)	More total vegetation cover and cover and biomass of perennial grasses; less bare ground; more soil moisture.
Use un-used pastures by repairing broken wells or digging new wells or building and repairing road access (Pasture are un-used due to water shortage for both for people and livestock. Usually remote pastures.)	Reduces the amount of time of winter grazing → more residuals or litter cover → less snow blowing by wind and more accumulation of snow on the ground → more spring vegetation emergence (thermal and moisture regulation)	More total vegetation cover and cover and biomass of perennial grasses; less bare ground; more soil moisture.
Purchase of hay and fodder (Usually poor and lactating livestock (race horses) are fed by hay and fodder)	1. Displacing pasture grazing to hay fields; 2. More trampling close to winter shelters → more breakage of vegetation stolons and growing buds 3. Better disaster preparedness → reduced vulnerability to <i>dzud</i>	Low total vegetative cover, more weedy, unpalatable vegetation and bare ground.
Growth of forage plants/fencing areas for hay collecting	More Fragmented land → increased spatial heterogeneity → decreased productivity of rangeland	Low total vegetative cover, more weedy, unpalatable vegetation and bare ground.
More animal sales to reduce herd size; slaughter of non-milking, male livestock	Reduced winter grazing pressure → more residuals or litter cover → less snow blowing by wind and more snow accumulation on the ground → better spring vegetation emergence (thermal and moisture regulation)	Opportunity for grasses to regrow during the growing season → healthier root systems and more stored carbohydrates
Improved livestock health (vaccination, deworming, treating livestock for external parasites) Improving livestock quality (purchase breeding stock)	Improved quality of livestock → herders have an incentive to keep fewer livestock of good quality → reduced winter grazing pressure → more residuals or litter cover → less snow blowing by wind and more snow accumulation on the ground → better spring vegetation emergence (thermal and moisture regulation)	Opportunity for grasses to regrow during the growing season → healthier root systems and more stored carbohydrates

This similar exposure occurs because there is an institutional vacuum to support pasture use and management, especially poor regulation of long distance (*otor*) movement during the *dzud* and drought in both vertical (CBRM and government) and horizontal (between CBRM and non-CBRM areas) scales and this could lead to similar *dzud* losses in CBRM and non-CBRM *soums*.

Even so, CBRM groups display better information access, knowledge exchange, coping and adaptive capacity by their collective action during *dzud* (Fernandez-Gimenez et al. 2012a, Fernandez-Gimenez et al. 2015).

A third explanation for the minor effects of CBRM is our selection of study pastures. We chose to sample winter rather than summer pastures because winter pastures provided us a natural ‘utilization cage’ for summer grazing. This allowed us to measure the effects of long-term rather than short-term seasonal grazing on rangelands, and saved the time and resources that would have been required to build exclosures in summer pastures. Winter pastures are an essential resource during the critical time of the year for Mongolian herders and most herders avoid grazing livestock in winter pastures around any winter shelters (their or their neighbors) during the summer time (Banzragch and Davaajamts 1970, Nachin 1984). It may be, as indicated above, that traditional herding practices in such cases are strong among Mongolian herders and that is why the winter pasture condition does not differ immensely between the two types of management, regardless of additional donor support for the newer CBRM groups. However, CBRM herders in our study are more likely not to graze their winter pastures out of season (Ulbayar 2015). It is quite possible that CBRM management makes a bigger difference in summer than winter pastures. This is implied by Leisher’s results, where their *soum*-level measures of total season productivity includes pastures used in all seasons. Another explanation in relation to sampling and method, we did not have a before-after, control-impact design. One

implication is the need for more rigorous monitoring to accompany development experiments such as CBRM, to be able to attribute cause and effect.

4.4.3. General effects of ecological site and how it modified the effects of CBRM

In general, ecological site was more often significant than CBRM across all ecological zones. This points to the importance of physical characteristics of the landscape and soils in determining vegetation, as is common in rangelands around the world.

In the mountain and forest steppe and steppe, we found more ecological site effects than CBRM effects, whereas in the desert steppe effects of CBRM and ecological site were about the same. Generally, in our wettest site, the mountain and forest steppe, several plant measures including total foliar cover, perennial plant cover, forb cover, unpalatable perennial forb cover in both growing and dormant seasons and litter cover were higher in ecological sites with low infiltration and seasonally standing water (Clay and High Water Table sites) than in rocky and steep, high infiltration sites. In this zone, the high infiltration, steep rocky and shallow sites have more open gaps between perennial plant bases, indicating the nature of these sites which are usually sparsely vegetated.

And in the driest zone, the desert steppe, ecological site was less important, but had some interesting patterns. Here the seasonally inundated, high water table sites were less productive than the better drained sites, similar to the steppe. These patterns follow the inverse texture hypothesis, which predicts that productivity will be higher in coarse than finer textured soils in drier sites, and the opposite will occur in wetter sites (Lane et al. 1998). In arid regions, sand content in coarse textured soil shows less water loss through bare-soil evaporation compared to fine textured soils. Precipitation percolates to a greater depth in the coarse textured soil and top

layer of coarse textured soil dries out quickly and forms a barrier that prevents conductance and evaporation of water from deeper soil. In contrast, in humid regions, fine textured soils may have greater water availability than coarse textured soils, because of higher water holding capacity (Lane et al. 1998). Thus in arid regions, coarse soil usually support denser perennial vegetation than finer soil. The same amount of productivity can be detected in coarse soils in dry regions and in fine soils humid regions (Noy-Meir 1973). Noy-Meir (1973) stressed that coarser textured soils had higher productivity than finer soils and this relationship between soil texture shifts at about 300 to 500m elevation, which is just above the rainfall level of our mountain and forest steppe sites.

Overall, we found a few significant interactions of CBRM and ecological site in all ecological zones. In the wetter mountain and forest steppe, the presence of *Thalictrum simplex*, a mesophytic, perennial forb used by herders as a medicinal plant, was greater in rocky hill sites than in the clay and high water table sites in CBRM pastures. Medium gaps were more abundant on steep rocky hill shallow sites than in clay and high water table site in the non-CBRM pastures. In the steppe, the presence of *Stipa gobica*, a highly palatable xerophyte, was greater in rocky hill shallow site than in the clay loam sites. There may be two explanations for these patterns. First, the degree of slope affects both vegetation productivity and use by range animals. As slope increases, vegetation productivity declines per unit of precipitation because less water enters the soil and more runs off as overland flow (Holechek 2004). Second, lowland pastures on clay and high water table sites are grazed more compared to pastures on rocky hill shallow sites in CBRM pastures (Dashnyam 1974).

4.5. Implications and future research

4.5.1. Implications for management and policy

We found only subtle ecological impacts of CBRM management on winter pastures. At least in winter pastures, it appears that herders in both CBRM and non-CBRM areas still practice customary herding strategies, by avoiding out of season grazing especially for winter pastures, essential during this critical time of the year. Mongolian rangelands have been grazed by domestic livestock for at least 1000 years, and thus appear to be ecologically resilient and adapted to the grazing. On the other hand cyclical *dzud* events reduce livestock numbers and this is an opportunity for the rangeland to rest and recover from intense grazing from preceding *dzud* years in both CBRM and non-CBRM pastures. Recently, livestock numbers quickly recovered from the last natural disaster (*dzud*) to reach the highest peak in livestock numbers in the last century (Mongolian Society for Range Management 2010). In this situation, grazing pressure might exceed the system's resilience with additional stress from a changing climate and this could cause rapid changes to pass a tipping point (or threshold) from reversible vegetation changes to irreversible degradation in these rangelands (Khishigbayar et al. 2015).

Customary herding management often depends on dispersed and overlapping social networks over large landscapes especially to reduce the vulnerability during times when forage is reduced due to *dzud* and harsh weather. Thus, cross boundary and cross-level governing institutions play essential roles in successful CBRM pasture management, by regulating movements and coordinating CBRM initiatives within the *soum* and between *soums* especially during the *dzud* and drought times (Fernandez-Gimenez et al. 2012a). Mongolia is still in the process of development of governance institutions to implement sustainable herding in the face of the current rapid socio-economic and ecological changes. Most of the CBRM groups in

Mongolia established with the donor initiation or top-down approach and most of them are no longer active after project cessations. Our social team's study results indicate that there is a wide range in the amount of their adoption of CBRM rules and practice from group to group. On the other hand some CBRM groups do better than others, and thus there is wide scope for many groups to improve pasture condition more strongly (Ulambayar 2015). Therefore, formalizing CBRM groups could be important for the success and sustainability of CBRM implementation to handle and at same time it is really important to develop appropriate and pertinent cross-level (*soum* and aimag level) governing institutions that guide CBRM implementation and contribute to resolving the pastoral paradox issue (Fernandez-Gimenez 2002), which means securing the land and providing flexibility to access resources.

4.5.2. Future research

Our study, conducted on winter pastures comparing between CBRM and non-CBRM managed *soums*, points to several research opportunities that could support more informed policy decisions in relation to CBRM management. First, there is need for more long-term monitoring in controlled, experimental exclosure studies on plant communities and soil surface indicators in summer pastures both managed by CBRM and non-CBRM groups in different ecological zones. This is expensive, but seems to be the only way to control for grazing and climate, and understand their interactions, in summer pastures. Second, there is an opportunity to bring together local and scientific knowledge through collaborative rangeland monitoring by including school children, herders, government officials and scientists. Herders' long-term observations (Bruegger et al. 2014) on rangeland change should be integrated into the monitoring activities to develop a deeper understanding of Mongolian rangelands. This will have mutual benefits such

as encouraging and increasing responsibilities of herders and preserving and transferring local ecological knowledge to the next generation and ensuring that scientific research is more targeted and needs based. Another important benefit of collaborative rangeland monitoring will be increased collaboration among herders, local government and mining companies for pasture improvement. Integrating herder observations and ecological knowledge into monitoring activities will be essential for managing wildlife, rare plants and other precious natural resources in the face of mining development and related infrastructure such as road and other development. Also herders could use their rangeland monitoring results to ensure their rangelands remain intact or are restored after mining activities. At the moment in Mongolia, this is a big gap and herders and local government are lacking the capacity to do this. Third, long-term trend analysis of rangeland condition, using a remote sensing approach, while controlling for climate and livestock numbers, is needed in all ecoregions of Mongolia.

CHAPTER FIVE: CONCLUSIONS

5.1. Introduction

The overarching goal of this thesis was to deepen our understanding of the effects of grazing and community based management on rangelands of Mongolia by expanding the scope and scale of previous work at a national scale and by linking field-level observations of vegetation to remote sensing of vegetation changes. At the field level, we used the broad-scale data collected by the large ecological teams of the Mongolian Rangeland Resilience project to look at the effects of livestock grazing around piospheres created around winter shelters in four ecological zones of Mongolia. In order to understand the long-term grazing and climate effects, we used instrumental observations from remote-sensing on the selected counties (*soums*) of the four ecological zones. Lastly, we wanted to understand the effects of community-based rangeland management (CBRM) on rangeland vegetation and soils in Mongolia.

Below I summarize the empirical findings of each data chapter that contributes to reach the overarching goal of this dissertation. Then I will present a cross-chapter synthesis, which will link together the results of all the chapters, and lastly I will provide a brief summary of implications for stakeholders.

5.2. Impacts of livestock grazing on Mongolian rangelands across ecological zones

The dung data confirmed a strong grazing gradient created by sheep and goats in our three sampled ecological zones, with a gradient contributed by cows and yaks only in the mountain and forest steppe. We had a classic piosphere effect in all ecological zones, but its

strength differed by zone. The piosphere effect was strongest in the steppe, moderate in the mountain and forest steppe and weakest in the desert steppe.

Grazing had no effect on the broad resolution variables of total biomass and total cover in any ecological zone. Rather, grazing had effects on medium resolution variables like the cover of functional groups including grasses, forbs, shrubs and sedges and influenced the cover of fine resolution variables measured at the species level across all ecological zones. There was less grass, and more abundant annual forbs, weedy plants and grazing tolerant increaser species in heavily than lightly grazed pastures. The effect of livestock grazing was greatest in the steppe, moderate in the desert steppe and mountain and forest steppe and was least in the eastern steppe. Grazing affected the distribution of unpalatable and palatable plants, but this differed by ecological zones. In the steppe and desert steppe, unpalatable annual forbs were more abundant close to winter shelters which are consistent with heavy livestock grazing. In the mountain and forest steppe and eastern steppe, palatable grass and forbs were more abundant far from winter shelters. This is the commonly found elsewhere around the world (Sternberg et al. 2000, McIntyre and Lavorel 2001, Sasaki 2008). In addition, grazing-induced soil loss and erosion have not occurred in all ecological zones.

We expected that the wetter, less climatically variable mountain and forest steppe would display characteristics of equilibrium rangeland dynamics, the eastern steppe and steppe would display mixed equilibrium and non-equilibrium characteristics, while desert steppe would exhibit non-equilibrium dynamics. Unexpectedly, we did not find much effect of grazing in the mountain and forest steppe, probably because these areas had weak grazing gradients. Our results from steppe showed that the pattern of vegetation change fits the classic equilibrium dynamics. We found that vegetation dynamics in the arid desert steppe of southern Mongolia

responded, in some variables, to grazing in equilibrium way where perennial grass decreased and unpalatable annual forbs increased with increasing grazing pressure. Our findings from desert steppe study supported the findings of Oba et al (2003) and Zemmrich (2007) who suggest that grazing impact becomes evident as spatial scale decreases in the arid and semi-arid areas. Also, the desert steppe is a mosaic of different ecological sites (citation) and some of which may exhibit more equilibrium dynamics than others. This may explain why we found grazing effects in the high water table communities (which constituted of 78% of the ecological sites) within the desert steppe.

Ecological site was more important than grazing as a determinant of vegetation characteristics in the mountain and forest steppe zone and was important for steppe and least important in the desert steppe ecological zone. As we expected, vegetation response in same ecological sites in different ecological zones differed. Interacting soil forming factors including parent material, climate, topography, biota and time are basis of properties of soil (Duniway et al. 2010).

5.3. Mongolian rangeland changes and resilience to livestock grazing over time

Our comparison of field and NDVI data showed only moderate relationships between the two sets of data. Comparing across ecological zones, field and remote sensing data were correlated in the mountain and forest steppe and in the steppe, but not in the desert steppe.

Between 2000 and 2013, there were more years with no evident grazing gradients in the NDVI data than years with clear grazing gradients in all ecological zones. Our data show that grazing significantly affects NDVI only in some years and not others, and this appears by an interaction between the levels of growing season rainfall and livestock grazing. Generally, when

there is grazing gradient, lightly grazed pastures display more NDVI than more heavily grazed pastures, except in the Erdenesant *soum* of steppe zone where we found opposite pattern with higher NDVI in heavily than lightly grazed pastures. This latter pattern may be caused by vigorous forb growth in heavily grazed pastures of this *soum*.

Current season (CS) NDVI responded quickly to the season-by-season changes in growing season precipitation and livestock densities/forage use. This could be explained that in lower rainfall years, livestock spread out more to obtain sufficient forage, so the gradient could be muted. By contrast, a grazing gradient only appeared in current and previous season (CPS) NDVI after long periods of low livestock densities/forage use. This may occur when biomass residuals from the previous year (litter and standing dead) buffer the grazing gradient. In the desert steppe, there were no consistent patterns in the grazing gradients, comparing among lightly to moderately to heavily grazed pastures. This would be expected in rangelands where climate is the over-riding factor driving vegetation dynamics. However, the expected grazing gradient, where NDVI was greater far from winter shelters, did appear, but only appeared in higher rainfall years (but did not appear in all higher rainfall years).

The NDVI patterns, compared to precipitation and forage use patterns, showed a shift from precipitation-dominated vegetation dynamics in the early 2000s to livestock dominated vegetation dynamics in the late 2000s especially in our study *soums* in mountain and forest steppe and steppe. In high rainfall years in the desert steppe, the effects of livestock became more apparent, as predicted by equilibrium and non-equilibrium theory.

We did find that in all 6 study *soums*, heavily grazed pastures had more variability in NDVI or were less resistant than lightly grazed pastures over time. The results of one study *soum* (Undurshireet *soum*, steppe zone) out of 6 *soums* showed that heavily grazed pastures had

low resilience (or longer recovery periods) than the lightly grazed pastures. However, Bayangol *soum* in the mountain and forest steppe, showed the opposite pattern, where lightly grazed pastures had a longer recovery period or lower resilience than the heavily grazed pastures. Some of these results may be confounded by the high spectral reflectance of weedy broad leaf forbs in heavily grazed pastures close to the winter shelters.

5.4. Ecological outcomes of community-based rangeland management in Mongolia

Our results suggest that CBRM initiatives are not having *major* impacts on any aspect of winter pastures across ecological zones. Only 5-9% of our tests were significant and significant comparing between CBRM and non-CBRM pastures out of 64-71 tests. We did find CBRM effects on soil surface indicators and cover of dominant species. In the mountain and forest steppe and steppe, CBRM showed positive effects on the connectedness of plant patchiness and soil erosion, but the pattern was opposite in the desert steppe, where CBRM pastures had less connected vegetation patches and soil movement than the non-CBRM pastures had. We did not see differences in soil surface variables between the pastures of the two management types in the eastern steppe. In the mountain and forest steppe CBRM-managed pastures had less cover of the increaser grass species, *Cleistogenes squarrosa*. In addition, in the eastern steppe, there was less cover of annual plants in the CBRM than non-CBRM managed pastures.

5.5. Cross chapter synthesis

The dung data we collected in 2011 and in 2012 showed that there are strong grazing gradients created by sheep and goats in all ecological zones and by cow and yak in the mountain and forest steppe. Field vegetation data supported the grazing gradients. There were more

grazing-induced changes in heavily than in lightly grazed pastures and moderate grazing induced changes have been occurring in the steppe, while subtle changes have occurred in the desert steppe and mountain and forest steppe. We did not see these grazing gradients consistently over time in the remotely sensed vegetation data. But when there was a significant grazing gradient, lightly grazed pastures had more NDVI than more heavily grazed pastures, as we expected based on other studies (Kawamura et al. 2005b, Butt 2010, Bradley and O'Sullivan 2011, Sha et al. 2014).

In general, out of three variables we analyzed, the grazing (or distance) effect was strongest (most significant) and the CBRM effect was weakest (least significant) with the effects of ecological site falling in between these variables across all ecological zones. It is logical that small scale and intensive grazing near winter shelters would affect grassland vegetation more than broader CBRM management rules. It also points to the importance of physical characteristics of the landscape in determining vegetation and soil effects, as is common in rangelands around the world.

In the mountain and forest steppe and steppe zone, both field vegetation and remotely sensed data display characteristics of equilibrium rangeland dynamics. Field data showed that with increasing grazing pressure, grasses decrease and forbs increase. Weedy and unpalatable annual forbs, like *Chenopodium album* (in the steppe) and grazing tolerant *Carex duriuscula* (in the mountain and forest steppe) increase under heavy grazing, which suggests that grazing plays an important role in determining vegetation composition. These grazing effects may be increasing over time, as our remotely sensed data showed a de-coupling of NDVI from precipitation in the last decade in this zone.

Several lines of evidence suggest that these rangelands may be approaching a tipping point (Bestelmeyer 2014, Khishigbayar et al. 2015), where grazing effects may become more pronounced in the future. NDVI trends suggest greater effects of grazing recently. Livestock populations appear to recover quickly from winter disasters, returning to high pre-disaster levels. In addition, our data are from winter pastures, where impacts of grazing may be lighter than summer camps that are grazed during the growing season.

5.6. Practical implications

It is critical time for Mongolia to have commonly agreed upon and clearly identified set of indicators to distinguish between reversible and potentially temporary changes in biotic communities and irreversible and permanent changes or loss of productive potential associated with changes in soils, hydrology and vegetation of rangeland systems. At the moment, degradation is highly subjective concept among researchers in Mongolia and it is very important to consider the inherent potential of rangelands and rangeland non-equilibrium ecosystem dynamics in developing a set of indicators to detect the degradation. It is probable that after several good moisture years, the pasture condition could reach to the not degraded stage which was considered as a degraded pasture and assessed. It will be essential to have commonly agreed indicators that are distinguished by its response behavior to different stresses and disturbances such as climate driven vs grazing driven vegetation and soil indicators.

We hope our study results provide substantial information on the current rangeland condition in 4 ecological zones and will be used as baseline for rangeland monitoring activities in the future. It is important to anticipate future change, especially interactions between climate and grazing, to ensure that we avoid rapid shifts in rangeland conditions over thresholds. Thus,

consistent and long term monitoring that uses both field and remote sensing approaches will be essential to understand grazing-induced changes and interaction of these changes with changing climate. This information can be used to predict possible future condition of the Mongolian rangelands, which is essential for evidence-based policy development.

Although we did find small changes in ecological impacts of CBRM on winter pastures, it may take longer for these institutions to strongly affect rangeland conditions. Thus, findings from this dissertation work should be introduced to herders both in CBRM and non-CBRM districts, and possible continuation of monitoring with herders should be explored. New monitoring could expand into other seasons' pastures and other *soums* that are not involved in this study. These new efforts will be essential to detect the impact of CBRM on rangelands and to develop effective policy for Mongolian rangelands.

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APPENDICES

Appendix 2.1. Results from the ANOVA of the effects of grazing distance from winter shelter and ecological sites on vegetation variables. In the eastern steppe, results from ANOVA are solely the effect of grazing from winter shelter (no number in the ecological site row). ANOVA results of variables that used rank transformations are stated under each variable by ecological zone and variables that used binary transformations appears in Appenix 2.2 below and appear in this table highlighted by grey. *-indicates the significance at $p < 0.05$. ANOVA results of dominant species included in this table are only those that were significant by each ecological zone. NS – not significant.

Variable	Source	Mountain and forest steppe (MFS)				Eastern steppe (ES)				Steppe (S)				Desert steppe (DS)			
		df	MS	F	P	df	MS	F	P	df	MS	F	P	df	MS	F	P
Total Foliar cover	distance	2	.008	.99	.50	2	.008	1.10	.35	2	.03	5.26	.006*	2	.006	2.22	.11
	ecological site	4	.015	1.94	.03*					3	.004	.84	.70	1	.00001	.00	.85
	error (main effect)	97	.008	.	.	20	.007	.	.	76	.005	.	.	74	.003	.	.
	distance x ecological site	8	.003	.43	.82					6	.001	.28	.94	2	.003	1.07	.37
	error (with interaction)	89	.008	.	.					70	.005	.	.	72	.003	.	.
Grass cover	distance	2	.030	3.04	.08	2	.004	4.42	.03*	2	.13	9.95	.0002*	2	.04	6.73	.002*
	ecological site	4	.01	.78	.48					3	.02	1.91	.14	1	.00007	.01	.81
	error (main effect)	97	.01		.	20	.008	.	.					74	.0006	.	.
	distance x ecological site	8	.009	.87	.70					6	.03	2.49	.03*	2	.0004	.06	.95
	error (with interaction)	89	.001	.	.					70	.01	.	.	72	.006	.	.
Forb cover	distance	2	.03	4.13	.02*	2	.03	8.36	.002	2	.14	10	.0001*	2	.02	5.33	.008
	ecological site	4	.02	2.03	.06					3	.02	1.55	.21	1	.0001	.02	.85
	error (main effect)	97	.0008	.	.	20	.003	.	.	76	.01	.	.	74	.005	.	.
	distance x ecological site	8	.005	.60	.65					6	.01	.94	.47	2	.007	1.59	.26
	error (with interaction)	89	.008	.	.					70	.01	.	.	72	.004	.	.
Subshrub cover (Ranked-	distance	2	.003	.33	.68	2	46.9	.89	.43	2	33.02	.05	.99	2	217.5	.36	.66
	ecological site	4	.001	.16	.92					3	149.	.22	.50	1	581.5	.97	.55

ES, S and DS)										11							
	error (main effect)	97	.008	.	.	20	52.8	.	.	76	685.6	.	.	74	599.3	.	.
	distance x ecological site	8	.01	1.63	.10					6	213.7	.29	.94	2	49.6	.81	.47
	error (with interaction)	89	.08	.	.					70	725.9	.	.	72	602.3	.	.
Sedge cover (Ranked-ES and S)	distance	2	.12	7.72	.0004*	2	3.34	.22	.80	2	13.65	.26	.79				
	ecological site	4	.03	1.82	.03*					3	96.57	.19	.62				
	error (main effect)	97	.02	.	.	20	15.2	.	.	76	502.1	.	.				
	distance x ecological site	8	.02	1.07	.31					6	189.88	.36	.83				
	error (with interaction)	89	.02	.	.					70	528.8	.	.				
Shrub cover (Ranked- S and DS)	distance									2	2243.1	4.70	.01*	2	4411.9	13.22	<.0001*
	ecological site									3	387.5	.8	.53	1	652.6	1.96	.21
	error (main effect)									76	477.7	.	.	74	333.7	.	.
	distance x ecological site									6	787.9	1.75	.06	2	395	1.19	.29
	error (with interaction)									70	451.1	.	.	72	332	.	.
Perennial plant cover	distance	2	.004	.47	.79	2	.007	1.04	.37	2	.11	14.5	<.001*	2	.01	2.80	.06
	ecological site	4	.01	1.77	.04*					3	.01	1.43	.24	1	.0001	.03	.34
	error (main effect)	97	.008	.	.	20	.007	.	.					74	.004	.	.
	distance x ecological site	8	.004	.44	.81					6	.02	2.61	.02*	2	.005	1.12	.40
	error (with interaction)	89	.008	.	.					70	.08	.	.	72	.004	.	.
Annual plant cover (Ranked-	distance	2	1019.6	1.22	.24					2	7715	19.84	<.0001*	2	688.2	3.68	.03*

MFS, S and DS)	ecological site	4	1287.8	1.51	.11					3	637.9	1.64	.08	1	9.05	.05	.69
	error (main effect)	97	833.9	.	.					76	388.9	.	.	74	187	.	.
	distance x ecological site	8	499.6	.58	.87					6	33.9	.84	.48	2	106.8	.56	.62
	error (with interaction)	89	863	.	.					70	393.8	.	.	72	189.2	.	.
Litter cover (Ranked-ES and DS)	distance	2	.16	9.60	.0003	2	.01	.49	.62	2	2929.7	23.03	<.0001*	2	5065.7	17.1	<.0001*
	ecological site	4	.02	1.11	.10					3	96.1	.76	.55	1	45.5	1.52	.48
	error (main effect)	97	.02	.	.	20	.03	.	.	76	127	.	.	74	296.9	.	.
	distance x ecological site	8	.01	.70	.58					6	99	.76	.65	2	378.6	1.28	.27
	error (with interaction)	89	.02	.	.					70	129.63	.	.	72	294.6	.	.
Basal cover	distance	2	.007	2.38	.12	2	.005	1.97	.17	2	.0004	.14	.91	2	.002	.75	.44
	ecological site	4	.001	.41	.86					3	.001	.47	.04*	.000	.00	.00	.80
	error (main effect)	97	.003	.	.	20	.003	.	.	76	.003	.	.	74	.002	.	.
	distance x ecological site	8	.002	.59	.79					6	.004	1.40	.32	2	.0007	.32	.56
	error (with interaction)	89	.003	.	.					70	.003	.	.	72	.002	.	.
Bare soil cover	distance	2	.009	1.10	.51	2	.007	1.24	.31	2	.001	.20	.78	2	.002	.25	.81
	ecological site	4	.01	1.50	.05*					3	.006	.88	.49	1	.005	.69	.16
	error (main effect)	97	.008	.	.	20	.006	.	.	76	.07	.	.	74	.007	.	.
	distance x ecological site	8	.007	.88	.53					6	.01	1.56	.16	2	.005	.67	.60
	error (with interaction)	89	.008	.	.					70	.007	.	.	72	.007	.	.
Total green standing crop biomass	distance	2	.59	.36	.71	2	.02	1.07	.36	2	.01	.51	.64	2	.02	.78	.44
	ecological site	4	.29	.18	.90					3	.02	.76	.42	1	.04	2.21	.05*
	error (main effect)	97	1.63	.	.	20	.02	.	.	70	.02	.	.	74	.02	.	.
	distance x ecological site	8	.96	.57	.89					6	.04	2.08	.08	2	.02	1.05	.42
	error (with interaction)	89	1.69	.	.					70	.02	.	.	72	.02	.	.
Grass	distance	2	.12	2.05	.24	2	.002	.06	.94	2	.19	4.2	.02	2	.16	3.42	.04

biomass											7	*				*	
	ecological site	4	.006	.10	.99					3	.06	1.24	.19	1	.02	.45	.98
	error (main effect)	97	.06	.	.	20	.03	.	.	76	.05	.	.	74	.05	.	.
	distance x ecological site	8	.08	1.36	.35					6	.06	1.47	.24	2	.09	1.94	.14
	error (with interaction)	89	.06	.	.					70	.04	.	.	72	.04	.	.
Forb biomass	distance	2	.01	.23	.72	2	.27	6.34	.007	2	.34	3.85	.03*	2	.05	1.13	.38
	ecological site	4	.05	.90	.44					3	.13	1.48	.01*	1	.0006	.01	.40
	error (main effect)	97	.05	.	.	20	.04	.	.	76	.09	.	.	74	.04	.	.
	distance x ecological site	8	.06	1.08	.41					6	.04	.47	.86	2	.04	1.00	.44
	error (with interaction)	89	.05	.	.					70	.09	.	.	72	.04	.	.
Sedge biomass (Ranked-MFS)	distance	2	1359.9	1.62	.15												
	ecological site	4	1453.9	4.71	.04*												
	error (main effect)	97	838.1	.	.												
	distance x ecological site	8	591	.69	.63												
	error (with interaction)	89	86.3	.	.												
Shrub biomass (Ranked-S)	distance					.	.	.23	.80	2	1418.3	2.85	.06	2	.007	.15	.85
	ecological site									3	193.7	.39	.25	1	.09	1.93	.16
	error (main effect)									76	497.12	.	.	74	.05	.	.
	distance x ecological site									6	735.2	1.54	.16	2	.06	.13	.92
	error (with interaction)									70	476.71	.	.	72	.05	.	.
Litter biomass (Ranked-DS)	distance					2	4.50	2.08	.15	2	8.12	8.03	.0007*	2	201.5	.43	.64
	ecological site									3	1.99	1.97	.04*	1	876.2	1.88	.16
	error (main effect)					20	2.16	.	.	76	1.01	.	.	74	467.3	.	.
	distance x ecological site									6	.49	.47	.85	2	59.2	.12	.85
	error (with interaction)									70	1.06	.	.	72	478.6	.	.
Standing	distance					.	.	.29	.75	2	5.32	12.	<.0	2	.	2.97	.06

dead biomass (binary-east, DS)												05	001*				
	ecological site									3	.50	1.14	.43	1	.	.53	.47
	error (main effect)									76	.44	.	.				
	distance x ecological site									6	.58	1.36	.24	2	.	.96	.39
	error (with interaction)									70	.43	.	.				
Species richness	distance	2	.008	1.45	.18	2	17.5	1.45	.26	2	28.15	1.54	.24	2	1.9	.18	.78
	ecological site	4	.008	1.48	.18					3	7.60	.42	.04*	1	8.2	.76	.09
	error (main effect)	97	.006	.	.	20	12.1	.	.	76	18.3	.	.	74	1.7	.	.
	distance x ecological site	8	.003	.043	.86					6	27.82	1.59	.13	2	6.4	.59	.47
	error (with interaction)	89	.008	.	.					70	17.5	.	.	72	1.84	.	.
Crude protein	distance	2	6.89	1.35	.28	2	19.04	3	.08	2	35.28	8.1	.0008*	2	9.4	1.58	.18
	ecological site	4	6.35	1.25	.33					3	1.81	.41	.38	1	.2	.03	.62
	error (main effect)	87	5.09	.	.	14	6.36	.	.	73	4.36	.	.	70	6	.	.
	distance x ecological site	8	5.29	1.04	.31					6	5.49	1.29	.27	2	7.6	1.28	.37
	error (with interaction)	79	5.03	.	.					67	4.26	.	.	68	5.9	.	.
Acid detergent fiber (ADF)	distance	2	5.35	2.86	.10	2	4.41	1.05	.40	2	4.98	.90	.44	2	21	6.06	.009*
	ecological site	4	.69	1.37	.63					3	1.04	.19	.74	1	.2	.06	.98
	error (main effect)	20	1.87	.	.	6	4.19	.	.	17	5.51	.	.	22	3.5	.	.
	distance x ecological site	3	3.12	2.47	.09					5	1.99	.29	.88	2	2.8	.79	.63
	error (with interaction)	14	1.30	.	.					12	6.97	.	.	20	3.5	.	.
AGCR - <i>Agropyron cristatum</i> (Ranked-MFS)	distance	2	2354.3	3.24	.04*												
	ecological site	4	684.8	.94	.34												
	error (main effect)	97	725.7	.	.												
	distance x ecological site	8	913.4	1.29	.37												
	error (with interaction)	89	708.8	.	.												
STGB - <i>Stipa gobica</i> (Ranked-	distance													2	1038.3	2.82	.05
	ecological site													1	3.0	.08	.61
	error (main effect)													74	367.6	.	.

DS)	distance x ecological site													2	78.5	.21	.70
	error (with interaction)													72	375.7	.	.
ALLMG L - <i>Allium mongolicu m</i> (Ranked- DS)	distance													2	954.7	3.25	.05 *
	ecological site													1	293.5	1.00	.59
	error (main effect)													74	293.4	.	.
	distance x ecological site													2	20.3	.68	.47
	error (with interaction)													72	296	.	.
STKR - <i>Stipa gobica</i> (Ranked- MFS and S)	distance	2	133	.22	.82					2	.	7.6 0	.00 1*				
	ecological site	4	263.1	.44	.84					3	.	2.2 9	.08				
	error (main effect)	97	600	.	.												
	distance x ecological site	8	3208. 8	.65	.57					6	.	1.4 7	.20				
	error (with interaction)	89	617.9	.	.												
CXDU (arsin- MFS)	distance	2	.08	3.04	.04*												
	ecological site	4	.03	1.24	.15												
	error (main effect)	97	.02	.	.												
	distance x ecological site	8	.03	1.31	.39												
	error (with interaction)	89	.02	.	.												
KOMA - <i>Koeleria macrantha</i>	distance	2	.0009	.28	.82												
	ecological site	4	.006	2.07	.20												
	error (main effect)	97	.0003	.	.												
	distance x ecological site	8	.0008	.25	.98												
	error (with interaction)	89	.003	.	.												
CLSQ - <i>Cleistogene s squarrosa</i>	distance					2	.04	5.5 2	.01 *								
	ecological site																
	error (main effect)					20	.008	.	.								
	distance x ecological site																
	error (with interaction)																
Perennial forb cover	distance	2	.04	5.46	.004*	2	.03	8.0 1	.00 3*	2	.01	1.5 0	.19	2	.02	5.33	.01
	ecological site	4	.01	1.25	.20					3	.004	.42	.30	1	.0000 2	.00	.21
	error (main effect)	97	.008	.	.	20	.004	.	.	76	.009	.	.	74	.004	.	.
	distance x ecological site	8	.005	.66	.61					6	.008	.85	.50	2	.005	1.49	.23
	error (with interaction)	89	.008	.	.					70	.000	.	.	72	.004	.	.

										9.							
Perennial grass cover	distance	2	.03	2.98	.08	2	.04	4.37	.03	2	.13	1.3	.0002*	2	.04	6.10	.003*
	ecological site	4	.008	.77	.50					3	.02	1.92	.04*	1	.0006	.008	.52
	error (main effect)	97	.01	.	.	20	.008	.	.					74	.007	.	.
	distance x ecological site	8	.009	.87	.71					6	.03	2.48	.04*	2	.001	.19	.87
	error (with interaction)	89	.01	.	.					70	.01	.	.	72	.007	.	.
Annual forb cover (Ranked-MFS, DS)	distance	2	816.8	1.01	.30									2	1563.7	5.24	.006*
	ecological site	4	1305.2	1.62	.13									1	22.9	.08	.85
	error (main effect)	97	808.1	.	.									74	298.6	.	.
	distance x ecological site	8	545.4	.66	.81									2	31.1	.10	.87
	error (with interaction)	89	831.7	.	.									72	306	.	.
Palatable perennial forb during growing season (Ranked-MFS)	distance	2	2156.9	3.12	.03*	2	.004	1.29	.30	2	.01	1.05	.32	2	.02	3.55	.05*
	ecological site	4	958.9	1.39	.09					3	.006	.50	.40	1	.0000	.00	.24
	error (main effect)	98	69.5	.	.	20	.003	.	.	76	.01	.	.	74	.005	.	.
	distance x ecological site	8	404.2	.56	.79					6	.006	.50	.75	2	.01	2.72	.08
	error (with interaction)	90	716	.	.					70	.01	.	.	72	.005	.	.
Palatable perennial forb during dormant season	distance					2	.001	.93	.41	2	.002	.19	.88				
	ecological site									3	.007	.88	.23				
	error (main effect)					20	.002	.	.								
	distance x ecological site									6	.02	2.74	.02*				
	error (with interaction)									70	.008	.	.				
Unpalatable perennial forb during growing season (Ranked-MFS)	distance	2	.02	3.64	.02*	2	.03	7.08	.005*								
	ecological site	4	.01	1.79	.16												
	error (main effect)	98	.006	.	.	20	.005	.	.								
	distance x ecological site	8	.005	.74	.52												
	error (with interaction)	90	.006	.	.												
Unpalatable perennial forb during	distance	2	.02	3.55	.025*	2	.03	6.05	.009*	2	28.15	.07	.92	2	485.7	1.53	.25
	ecological site	4	.02	2.20	.06					3	107	.25	.25	1	440.5	1.39	.44

dormant season (Rank-S and DS)	error (main effect)	98	.007	.	.	20	.006	.	.	76	426.9	.	.	74	317.2	.	.
	distance x ecological site	8	.003	.47	.79					6	219	.49	.89	2	299.9	.94	.43
	error (with interaction)	90	.007	.	.					70	444.77	.	.	72	317.7	.	.
Palatable perennial grass during growing season	distance	2	.03	3.75	.04*	2	.01	.47	.63	2	.20	13.14	<.0001*	2	.01	1.11	.28
	ecological site	4	.01	1.07	.39					3	.12	8.11	<.0001*	1	.0003	.03	.75
	error (main effect)	98	.009	.	.	20	.03	.	.					74	.01	.	.
	distance x ecological site	8	.007	.78	.75					6	.07	4.59	.0007*	2	.002	.16	.82
	error (with interaction)	90	.009	.	.					70	.02	.	.	72	.01	.	.
Palatable perennial grass during dormant season	distance	2	.02	2.19	.14	2	.03	1.13	.34	2	.20	13.24	<.0001*	2	.01	1.26	.25
	ecological site	4	.01	.96	.32					3	.1	4.68	.0003	1	.002	.17	.89
	error (main effect)	98	.001	.	.	20	.02	.	.					74	.01	.	.
	distance x ecological site	8	.01	1.04	.48					6	.07	4.68	.0007*	2	.002	.15	.78
	error (with interaction)	90	.01	.	.					70	.02	.	.	72	.01	.	.
Unpalatable perennial grass during dormant season (Ranked-MFS)	distance	2	1277	2.62	.11	2	.03	3.22	.06								
	ecological site	4	853.8	1.75	.24												
	error (main effect)	98	488	.	.	20	.008	.	.								
	distance x ecological site	8	10.4	.19	.99												
	error (with interaction)	90	522.4	.	.												
Unpalatable annual forb during dormant season (Ranked-S)	distance									2	9003	22.5	<.0001				
	ecological site									3	523.37	1.31	.13				
	error (main effect)									76	399.66	.	.				
	distance x ecological site									6	405.	1.0	.30				

											23	2					
	error (with interaction)									70	399.17	.	.				
Palatable subshrub during dormant season (rank-DS)	distance													2	24.7	.05	.99
	ecological site													1	121.7	2.23	.29
	error (main effect)													74	541.9	.	.
	distance x ecological site													2	56.6	.10	.91
	error (with interaction)													72	555.4	.	.
Unpalatable subshrub during dormant season	distance	2	.002	.32	.68												
	ecological site	4	.005	.75	.43												
	error (main effect)	98	.007	.	.												
	distance x ecological site	8	.006	.82	.42												
	error (with interaction)	90	.007	.	.												
Average gap (Ranked-MFS)	distance	2	1122.7	2.30	.19	2	54.91	5.65	.01*	2	.04	14.2	<.0001*	2	.001	.94	.38
	ecological site	4	725.5	1.49	.06					3	.006	2.20	.24	1	.003	2.41	.46
	error (main effect)	97	488.1	.	.	20	9.72	.	.					74	.001	.	.
	distance x ecological site	8	516.5	1.06	.38					6	.01	4.54	.006*	2	.0004	.31	.67
	error (with interaction)	89	485.6	.	.					70	.003	.	.	72	.001	.	.
Gap (25-50cm)	distance	2	.009	1.70	.22	2	.01	2.50	.11	2	19.4	.97	.41	2	.005	1.00	.33
	ecological site	4	.02	3.29	.02*					3	11.7	.58	.23	1	.002	.35	.67
	error (main effect)	97	.005	.	.	20	.004	.	.	76	2.09	.	.	74	.005	.	.
	distance x ecological site	8	.005	.95	.51					6	45.065	2.51	.06	2	.01	2.14	.12
	error (with interaction)	89	.005	.	.					70	17.95	.	.	72	.005	.	.
Gap (51-100cm) (Rank-MFS)	distance	2	923.8	1.51	.35	2	.01	3.47	.05	2	.003	.55	.59	2	.02	1.93	.13
	ecological site	4	732.9	1.20	.06					3	.004	.59	.83	1	.007	.84	.20
	error (main effect)	97	61.7	.	.	20	.003	.	.	76	.006	.	.	74	.008	.	.
	distance x ecological site	8	488	.78	.47					6	.009	1.49	.26	2	.01	1.35	.35
	error (with interaction)	89	621.8	.	.					70	.005	.	.	72	.008	.	.

Gap (101-200cm)	distance									2	.01	1.1 1	.34	2	.01	1.63	.17
	ecological site									3	.010	1.0 4	.57	1	.005	.60	.69
	error (main effect)									76	.010	.	.	74	.008	.	.
	distance x ecological site									6	.009	.91	.53	2	.02	2.91	.12
	error (with interaction)									70	.010	.	.	72	.008	.	.
Gap (>201cm)	distance													2	.04	1.62	.18
	ecological site													1	.02	.69	.85
	error (main effect)													74	.03	.	.
	distance x ecological site													2	.02	.71	.49
	error (with interaction)													72	.03	.	.

Appendix 2.2. Results from the ANOVA of the effects of grazing distance from winter shelter and ecological sites on vegetation variables that were transformed into binary values (presence and absence). In the eastern steppe, results from ANOVA are solely the effect of grazing from winter shelter (no number in the ecological site row). ANOVA results of variables that were used other transformations appear in Appendix 2.1 and highlighted in grey in this table. *-indicates the significance at $p < 0.05$. ANOVA results of dominant species included in this table are only those that were significant by ecological zone.

Variable	Source	Mountain and forest steppe			Eastern steppe			Steppe			Desert steppe		
		df	F	P	df	F	P	df	MS	F	df	MS	F
Sedge cover	distance										2	1.27	.29
	ecological site										2	2.50	.12
	distance x ecological site											1.75	.18
Shrub cover	distance	2	3.98	.02*	2	1.11	.35						
	ecological site	4	.98	.42									
	distance x ecological site	8	1.11	.36									
Legume cover	distance	2	1.63	.20	2	.40	.68	2	.35	.70	Only 5 plots with .4-1.6% cover		
	ecological site	4	1.58	.19				3	1.40	.25			
	distance x ecological site	8	1.15	.35				6	.25	.96			
Annual plant cover	distance				2	.13	.88						
	ecological site												
	distance x ecological site												
Sedge biomass	distance				2	1.38	.27	2	.04	.96	2	.23	.80
	ecological site							3	.57	.64	1	1.50	.22
	distance x ecological site							6	.85	.53	2	.65	.52
Shrub biomass	distance	2	2.88	.06	2	.23	.80						
	ecological site	4	1.51	.20									
	distance x ecological site	8	1.73	.10									
Litter biomass	distance	2	.33	.72									
	ecological site	4	.37	.83									
	distance x ecological site	8	.32	.96									
Standing dead biomass	distance	2	1.28	.28	2	.29	.75				2	1.28	.28
	ecological site	4	.91	.46							4	.91	.46
	distance x ecological site	8	.36	.94							8	.36	.94
ACSP – <i>Achnatherum Splendens</i>	distance							2	4.38	.02*			
	ecological site							3	.84	.48			
	distance x ecological site							6	.53	.78			
CARST – <i>Caragana stenophylla</i>	distance							2	3.17	.05*			

	ecological site							3	.16	.92			
	distance x ecological site							6	2.04	.07			
CARMI- <i>Caragana microphylla</i>	distance							2	5.01	.009*			
	ecological site							3	.71	.55			
	distance x ecological site							6	.98	.44			
CHAL – <i>Chenopodium album</i>	distance							2	14.99	<.000 1*			
	ecological site							3	.86	.47			
	distance x ecological site							6	.93	.48			
Annual forb cover	distance				2	.05 8	.57	2	6.46	.003*			
	ecological site							3	.29	.83			
	distance x ecological site							6	.35	.90			
Palatable perennial forb during dormant season	distance	2	2.94	.06							2	.22	.80
	ecological site	4	1.00	.41							1	2.09	.15
	distance x ecological site	8	1.23	.29							2	.14	.87
Unpalatable perennial forb during growing season	distance							2	.19	.83	2	.08	.92
	ecological site							3	.84	.48	1	.89	.35
	distance x ecological site							6	.48	.82	2	.85	.43
Unpalatable perennial grass during growing season	distance	2	2.46	.09	2	.88	.43	2	.28	.76	2	.42	.66
	ecological site	4	1.91	.12				3	.04	.99	1	.33	.57
	distance x ecological site	8	.83	.58				6	.64	.70	2	.12	.89
Unpalatable perennial grass during dormant season	distance							2	.19	.83	2	.62	.54
	ecological site							3	.16	.92	1	1.88	.17
	distance x ecological site							6	1.04	.41	2	.54	.58
Palatable annual forb during growing season	distance	2	.31	.73				2	.41	.66	2	.06	.94
	ecological site	4	1.14	.34				3	.31	.82	1	3.0	.09
	distance x ecological site	8	.59	.78				6	1.60	.16	2	.58	.56
Palatable annual forb during dormant season	distance	2	.38	.68				2	3.03	.05*	2	.23	.79
	ecological site	4	2.20	.07				3	1.91	.14	1	1.90	.17
	distance x ecological site	8	.62	.76				6	1.16	.34	2	.08	.93
Unpalatable annual forb during growing season	distance	2	1.49	.23	2	1.4 8	.25	2	<.000 1		2	3.87	.02 *
	ecological site	4	.84	0.50				3	.66	.58	1	.96	.33
	distance x ecological site	8	.55	.81				6	.48	.82	2	.14	.87
Unpalatable annual forb during dormant season	distance	2	.03	.97	2	1.4 8	.25				2	2.21	.12
	ecological site	4	.31	.87							1	.60	.44
	distance x ecological site	8	.25	.98							2	.86	.42

Palatable subshrub during growing season	distance	2	.39	.68	2	.17	.84	2	.28	.76	2	.04	.96
	ecological site	4	1.11	.36				3	1.03	.38	1	.21	.65
	distance x ecological site	8	1.36	.22				6	2.27	.05	2	.96	.39
Palatable subshrub during dormant season	distance	2	.81	.45	2	.17	.84	2	.06	.94			
	ecological site	4	2.01	.10				3	.42	.74			
	distance x ecological site	8	.87	.55				6	.80	.57			
Unpalatable subshrub during growing season	distance	2	.91	.41	2	1.39	.27	2	.53	.59	2	.35	.71
	ecological site	4	2.14	.08				3	.74	.53	1	.44	.51
	distance x ecological site	8	1.15	.34				6	.21	.97	2	.83	.44
Unpalatable subshrub during dormant season	distance				2	1.39	.27	2	.37	.69	2	.75	.48
	ecological site							3	1.05	.38	1	.56	.46
	distance x ecological site							6	.79	.58	2	.30	.74
Gap (101-200cm)	distance	2	.35	.71	2	.47	.63						
	ecological site	4	.83	.51									
	distance x ecological site	8	.38	.93									
Gap (>201cm)	distance	2	.91	.41				2	.75	.45			
	ecological site	4	.36	.84				3	.24	.87			
	distance x ecological site	5	1.14	.34				6	.29	.94			

Appendix 4.1. Results from the ANOVA of the effects of CBRM and ecological site on vegetation variables. In the eastern steppe, results from ANOVA are solely the effect of CBRM (no number in the ecological site row). ANOVA results of variables that were analyzed using a rank transformation are indicated under each variable name by ecological zone and variables that were analyzed using a binary transformation appear in Appendix 4.2 and not this table but, instead, are highlighted in gray. *-indicates significance at $p \leq 0.05$. ANOVA results of dominant species included in this table are only those that were significant by ecological zone. NS – not significant.

Variable	Source	Mountain and forest steppe (MFS)				Eastern steppe (ES)				Steppe (S)				Desert steppe (DS)			
		df	MS	F	P	df	MS	F	P	df	MS	F	P	df	MS	F	P
Total Foliar cover	CBRM	1	.01	.06	.80	1	.004	.03	.87	1	.02	.06	.80	1	.001	.02	.90
	ecological site	2	.037	6.45	.002*					2	.004	.91	.68	1	.01	2.02	.16
	error (main effect)	101	.008	.	.	22	.07	.	.	79	.005	.	.				
	CBRM x ecological site	2	.01	.52	.59					2	.01	2.11	.24	1	.01	4.56	.04*
	error (with interaction)	99	.008	.	.					77	.005	.	.	75	.003	.	.
Grass cover	CBRM	1	.28	3.21	.08	1	.02	.09	.78	1	.06	.29	.60	1	.005	.10	.75
	ecological site	2	.007	1.21	.30					2	.04	2.29	.03*	1	.04	1.78	.018
	error (main effect)	101	.01	.	.	22	.01	.	.	79	.02	.	.				
	CBRM x ecological site	2	.02	.45	.64					2	.007	.40	.67	1	.04	6.10	.02*
	error (with interaction)	99	.01	.	.					77	.02	.	.	75	.006	.	.
Forb cover	CBRM	1	.18	2.49	.12					1	.001	.02	.99	1	.008	.07	.79
	ecological site	2	.04	3.81	.02*	1	.004	.10	.76	2	.04	2.19	.06	1	.00002	.14	.71
	error (main effect)	101	.008	.	.	22	.006	.	.	79	.02	.	.	76	.005	.	.
	CBRM x ecological site	2	.006	1.30	.28					2	.05	2.77	.59	1	.003	.59	.44
	error (with interaction)	99	.008	.	.					77	.02	.	.	75	.005	.	.
Subshrub cover (Ranked-ES, S, DS)	CBRM	1	.03	1.18	.28	1	7.82	.04	.85	1	43.4	.02	.92	1	411.9	.21	.65
	ecological site	2	.0003	.22	.80					2	196.3	.30	.52	1	447.8	.21	.64
	error (main effect)	101	.008	.	.	22	52.21	.	.	79	656.5	.	.	76	589.2	.	.
	CBRM x ecological site	2	.005	.81	.45					2	5137.1	2.43	.09	1	36.4	.27	.60
	error (with interaction)	99	.008	.	.					77	633.7	.	.	75	596.6	.	.

Sedge cover (Ranked-ES, S)	CBRM	1	.009	.16	.69	1	485.1	2.04	.19	1	1655.8	.67	.44				
	ecological site	2	.006	1.67	.19					2	136.6	.28	.64				
	error (main effect)	101	.02	.	.	22	14.11	.	.	79	486.3	.	.				
	CBRM x ecological site	2	.03	1.12	.33					2	213.4	.43	.61				
	error (with interaction)	99	.02	.	.					77	493.4	.	.				
Shrub cover (Ranked-S)	CBRM									1	2214.9	.94	.34				
	ecological site									2	186.9	.35	.72				
	error (main effect)									79	527.9	.	.				
	CBRM x ecological site									2	595.7	1.13	.71				
	error (with interaction)									77	526.2	.	.				
Perennial plant	CBRM	1	.01	.07	.79	1	.0004	.00	.96	1	.03	.13	.72	1	.0002	.05	.82
	ecological site	2	.03	5.54	.005*					2	.003	.27	.53	1	.01	4.63	.03*
	error (main effect)	101	.008	.	.	22	.007	.	.	79	.01	.	.				
	CBRM x ecological site	2	.01	.38	.69					2	.006	.56	.86	1	.03	5.45	.02*
	error (with interaction)	99	.008	.	.					77	.01	.	.	75	.004	.	.
Annual plant cover (Ranked-MFS, S, DS)	CBRM	1	127.6	.02	.90					1	65.77	.03	.98	1	64.4	.14	.71
	ecological site	2	1612.7	2.16	.12					2	1097.6	1.90	.08	1	25.01	.00	.96
	error (main effect)	101	848.7	.	.					79	577.8	.	.	76	200.2	.	.
	CBRM x ecological site	2	76.4	.88	.42					2	1351.8	2.42	.34	1	63.7	.14	.71
	error (with interaction)	99	864.3	.	.					77	557.7	.	.	75	202	.	.
Litter cover	CBRM	1	.16	.97	.33	1	121.3	.72	.42	1	1.45	5.87	.02*	1	659.3	.29	.59

(Ranked-ES, S, DS)	ecological site	2	.06	4.66	.01*					2	.009	.51	.56	1	357.8	.10	.76
	error (main effect)	101	.02	.	.	22	60.89	.	.	79	.02	.	.	76	422.4	.	.
	CBRM x ecological site	2	.02	.25	.78					2	.007	.37	.91	1	2913	2.30	.13
	error (with interaction)	99	.02	.	.					77	.02	.	.	75	389.2	.	.
Basal cover (Ranked-S)	CBRM	1	.097	2.04	.16	1	.002	.97	.35	1	.03	1.54	.23	1	.09	3.84	.06
	ecological site	2	.001	.09	.92					2	.001	.33	.05*	1	.00002	.05	.82
	error (main effect)	101	.003	.	.	22	.003	.	.	79	.003	.	.	76	.002	.	.
	CBRM x ecological site	2	.008	1.25	.29					2	.003	.96	.28	1	.01	3.59	.06
	error (with interaction)	99	.003	.	.					77	.003	.	.	75	.002	.	.
Bare soil cover (Ranked-S)	CBRM	1	.14	1.21	.28	1	.002	.07	.80	1	.04	.25	.62	1	.08	.91	.35
	ecological site	2	.03	5.52	.005*					2	.003	.36	.71	1	.006	1.91	.17
	error (main effect)	101	.008			22	.006	.	.	79	.007	.	.	76	.007	.	.
	CBRM x ecological site	2	.006	.05	.95					2	.007	.96	.35	1	.0002	.60	.44
	error (with interaction)	99	.008	.	.					77	.007	.	.	75	.007	.	.
Total green standing biomass	CBRM	1	.006	.00	.98	1	.14	2.56	.14	1	1.16	3.64	.06	1	.04	.10	.75
	ecological site	2	.495	.52	.60					2	.03	1.32	.19	1	.05	3.95	.05*
	error (main effect)	101	1.59	.	.	22	.02	.	.	79	.02	.	.	76	.02	.	.
	CBRM x ecological site	2	.92	.94	.39					2	.01	.54	.87	1	.0009	.03	.86
	error (with interaction)	99	158.4	.	.					77	.02	.	.	75	.02	.	.
Grass biomass	CBRM	1	.29	.79	.38	1	.21	1.42	.26	1	1.93	3.06	.09	1	.30	.62	.44
	ecological site	2	.005	.24	.79					2	.13	2.67	.03*	1	.02	.00	.96
	error (main effect)	101	.06	.	.	22	.03	.	.	79	.05	.	.	76	.05	.	.
	CBRM x ecological site	2	.073	.11	.90					2	.005	.11	.78	1	.20	2.45	.12
	error (with interaction)	99	.06	.	.					77	.05	.	.	75	.05	.	.
Forb biomass	CBRM	1	1.466	3.37	.07	1	.004	.01	.91	1	.05	.18	.75	1	.14	.23	.63
	ecological site	2	.085	.94	.39					2	.15	1.62	.02*	1	.006	1.03	.311
	error (main effect)	10	.05	.	.	22	.06	.	.	79	.10	.	.	76	.04	.	.

		1															
	CBRM x ecological site	2	.03	1.79	.17					2	.32	3.59	.45	1	.03	1.18	.28
	error (with interaction)	99	.05	.	.					77	.09	.	.	75	.04	.	.
Sedge biomass (Ranked-MFS)	CBRM	1	453.2	.13	.72												
	ecological site	2	1111.9	2.75	.07												
	error (main effect)	101	864.7	.	.												
	CBRM x ecological site	2	868.9	.37	.69												
	error (with interaction)	99	864.6	.	.												
Shrub biomass (rank-MFS, S)	CBRM	1	548.4	.17	.68					1	1527	7.58	.01*	1	.003	.003	.86
	ecological site	2	1949.2	2.35	.10					2	372.8	.72	.16	1	.09	1.95	.17
	error (main effect)	101	733.3	.	.					79	515.3	.	.	76	.05	.	.
	CBRM x ecological site	2	52.18	.82	.44					2	296.6	.57	.86	1	.03	1.36	.25
	error (with interaction)	99	747.0	.	.					77	520.9	.	.	75	.05	.	.
Litter biomass (Ranked-ES, DS)	CBRM	1	.47	.03	.86					1	65.9	4.93	.03*	1	170.8	.13	.72
	ecological site	2	.68	.09	.91					2	2.58	2.18	.06	1	3.38	.00	1.00
	error (main effect)	101	2.46	.	.					79	1.18	.	.	76	501.3	.	.
	CBRM x ecological site	2	7.05	1.38	.25	1	801.9	8.92	.02*	2	.65	.54	.89	1	354.2	.59	.44
	error (with interaction)	99	2.37	.	.	22	62.8	.	.	77	1.93	.	.	75	503.2	.	.
Standing dead biomass	CBRM									1	23.1	4.91	.03*				
	ecological site									2	.24	.43	.81				
	error (main effect)									79	.56	.	.				
	CBRM x ecological site									2	.87	1.57	.62				
	error (with interaction)									77	.55	.	.				
Species richness	CBRM	1	368.6	.56	.46	1	76.39	.61	.45	1	2.22	1.79	.23	1	.07	.07	.80

	ecological site	2	55.57	1.57	.21					2	.15	.79	.04*	1	.04	2.08	.15
	error (main effect)	101	29.76	.	.	22	12.58	.	.	79	.19	.	.	76	.13	.	.
	CBRM x ecological site	2	18.58	1.51	.22					2	.14	.72	.16	1	.17	.09	.76
	error (with interaction)	99	29.98	.	.					77	.019	.	.	75	.13	.	.
Crude protein	CBRM	1	4.63	.06	.80	1	110.3	4.74	.07	1	152.8	2.69	.11	1	78.7	1.10	.30
	ecological site	2	13.9	2.07	.13					2	2.43	.47	.42	1	.04	.02	.88
	error (main effect)	91	5.12			16	7.9	.	.	76	5.14	.	.	72	6.08	.	.
	CBRM x ecological site	2	3.00	.09	.91					2	8.3	1.64	.17	1	.94	1.32	.25
	error (with interaction)	89	5.17	.	.					74	5.06	.	.	71	6.16	.	.
Acid detergent fiber	CBRM	1	4.75	1.05	.32	1	157.7	10.28	.08	1	.22	.02	.86	1	1.26	.14	.71
	ecological site	2	1.80	.84	.44					2	.99	.18	.85	1	1.44	.05	.82
	error (main effect)	24	2.02	.	.	8	4.24	.	.	20	5.53	.	.	24	5.1	.	.
	CBRM x ecological site	2	1.74	.51	.60					2	9.86	1.95	.08	1	1.16	.01	.92
	error (with interaction)	22	2.04	.	.					18	5.05	.	.	23	5.25	.	.
Perennial forb cover	CBRM	1	.16	2.41	.13	1	.0002	.01	.94	1	.0008	.00	.96	1	.09	2.49	.12
	ecological site	2	.03	2.57	.08					2	.005	1.12	.33	1	.0007	2.25	.14
	error (main effect)	101	.008	.	.	22	.006	.	.	79	.009	.	.	76	.004	.	.
	CBRM x ecological site	2	.008	1.45	.24					2	.03	1.34	.26	1	.00004	.07	.79
	error (with interaction)	99	.008	.	.					77	.009	.	.	75	.004	.	.
Perennial grass cover	CBRM	1	.29	3.34	.07	1	.02	.08	.78	1	.05	.24	.62	1	.044	.71	.40
	ecological site	2	.007	1.17	.31					2	.04	3.64	.03*	1	.0004	.27	.60
	error (main effect)	101	.01	.	.	22	.01	.	.	79	.02	.	.	76	.008	.	.
	CBRM x ecological site	2	.02	.43	.65					2	.005	.39	.68	1	.04	3.60	.06
	error (with interaction)	99	.01	.	.					77	.02	.	.	75	.008	.	.
Annual forb cover (Ranked-	CBRM	1	210.6	.05	.83					1	247.4	.04	.85	1	454.1	.20	.66
	ecological site	2	1608.	2.11	.12					2	1294	2.5	.08	1	67.8	.10	.76

MFS, S, DS)			2							.7	6						
	error (main effect)	101	820.2	.	.					79	603.7	.	.	76	331.9	.	.
	CBRM x ecological site	2	92.6	1.12	.33					2	1345.3	1.01	.37	1	636.2	1.32	.25
	error (with interaction)	99	834.9	.	.					77	584.5	.	.	75	327.8	.	.
Palatable perennial forb during growing season	CBRM	1	9368.5	1.97	.17	1	.01	1.15	.31	1	.004	.10	.76	1	.03	1.41	.24
	ecological site	2	1776.8	2.02	.14					2	.008	1.11	.33	1	.001	1.95	.16
	error (main effect)	102	724.7	.	.	22	.003	.	.	79	.01	.	.	76	.005	.	.
	CBRM x ecological site	2	388.8	.77	.46					2	.04	1.46	.24	1	2.42	.36	.55
	error (with interaction)	100	731.4	.	.					77	.01	.	.	75	.005	.	.
Palatable perennial forb during dormant season (Ranked-DS)	CBRM					1	.02	2.86	.12	1	.003	.02	.89	1	2267.3	.89	.35
	ecological site									2	.02	2.44	.09	1	40.04	1.61	.21
	error (main effect)					22	.001	.	.	79	.009	.	.	76	439.3	.	.
	CBRM x ecological site									2	.02	.66	.52	1	1412.9	2.98	.09
	error (with interaction)									77	.009	.	.	76	426.3	.	.
Unpalatable perennial forb during growing season (Ranked-MFS)	CBRM	1	.09	1.96	.17	1	.0001	.00	.96								
	ecological site	2	.03	3.49	.03*												
	error (main effect)	102	.007	.	.	22	.007	.	.								
	CBRM x ecological site	2	.006	1.14	.32												
	error (with interaction)	100	.007	.	.												
Unpalatable perennial forb during	CBRM	1	.16	2.59	.11	1	.0008	.02	.89	1	.91	.00	.99	1	166.9	.06	.80
	ecological site	2	.035	3.93	.02*					2	156.8	2.03	.14	1	368.2	.51	.48
	error (main effect)	10	.007	.	.	22	.008	.	.	79	411.	.	.	76	321.7	.	.

dormant season (Rank-S and DS)		2								7							
	CBRM x ecological site	2	.007	1.32	.27					2	166.1	.19	.83	1	275	.00	.98
	error (with interaction)	100	.007	.	.					77	418.1	.	.	75	322.3	.	.
Palatable perennial grass during growing season	CBRM	1	.34	3.72	.06	1	.05	.21	.66	1	.02	.10	.76	1	.004	.03	.87
	ecological site	2	.009	1.36	.26					2	.11	6.20	.003*	1	.04	1.36	.25
	error (main effect)	102	.009	.	.	22	.02	.	.	79	.02	.	.				
	CBRM x ecological site	2	.02	.41	.66					2	.007	.34	.72	1	.04	3.87	.05*
	error (with interaction)	100	.009	.	.					77	.02	.	.	75	.01	.	.
Palatable perennial grass during dormant season	CBRM	1	.36	3.62	.06	1	.05	.25	.63	1	.04	.27	.61				
	ecological site	2	.006	.78	.46					2	.08	5.01	.008*				
	error (main effect)	102	.01	.	.	22	.02	.	.	79	.02	.	.				
	CBRM x ecological site	2	.02	.44	.64					2	.003	.13	.88				
	error (with interaction)	100	.01	.	.					77	.02	.	.				
Unpalatable perennial grass during dormant season (Ranked-MFS)	CBRM	1	7.52	.00	.97	1	.009	.05	.82								
	ecological site	2	981.9	1.51	.23												
	error (main effect)	102	513.6	.	.	22	.01	.	.								
	CBRM x ecological site	2	286.7	1.35	.26												
	error (with interaction)	100	518.1	.	.												
Unpalatable annual forb during dormant season (Ranked-S)	CBRM									1	40.2	.00	.98				
	ecological site									2	1029.4	2.22	.11				
	error (main effect)									79	622.4	.	.				
	CBRM x ecological site									2	1358.8	1.30	.28				
	error (with interaction)									77	603.3	.	.				

Palatable subshrub during growing season (Ranked-MFS)	CBRM	1	58.9	.02	.90												
	ecological site	2	1874.1	2.51	.08												
	error (main effect)	102	657.7	.	.												
	CBRM x ecological site	2	435.8	.65	.52												
	error (with interaction)	100	666.5	.	.												
Palatable subshrub during dormant season (Ranked-DS)	CBRM													1	599.4	.25	.62
	ecological site													1	1185.4	.	.
	error (main effect)													76	528.3	.	.
	CBRM x ecological site													1	144.9	.51	.47
	error (with interaction)													75	533.4	.	.
Unpalatable subshrub during dormant season	CBRM	1	.09	2.83	.10												
	ecological site	2	.007	1.63	.20												
	error (main effect)	102	.007	.	.												
	CBRM x ecological site	2	.006	.47	.63												
	error (with interaction)	100	.007	.	.												
Average gap (Ranked-DS)	CBRM	1	4.52	.56	.46	1	309.8	3.91	.08	1	.03	.39	.53	1	12993	5.91	.02*
	ecological site	2	1.1	2.58	.08					2	.003	.13	.88	1	544.7	.13	.71
	error (main effect)	101	.74	.	.	22	13.8	.	.	79	.004	.	.	76	429.5	.	.
	CBRM x ecological site	2	.47	.75	.48					2	.002	.16	.85	1	2611.1	2.03	.16
	error (with interaction)	99	.74	.	.					77	.004	.	.	75	400.4	.	.
Gap (25-50cm)	CBRM	1	.04	1.16	.29	1	.05	3.0	.12	1	.006.40	.53		1	.14	3.11	.09
	ecological site	2	.03	5.01	.008*					2	.004	1.87	.16	1	.002	.09	.76
	error (main effect)	101	.005	.	.	22	.004	.	.	79	.005	.	.	76	.005	.	.
	CBRM x ecological site	2	.009	1.31	.27					2	.0007	.27	.76	1	.02	1.29	.26
	error (with interaction)	99	.005	.	.					77	.005	.	.	75	.005	.	.

Gap (51-100 cm) (Ranked-MFS)	CBRM	1	3003.8	.79	.38	1	.10	3.60	.09	1	.008	.10	.76	1	.13	2.59	.12
	ecological site	2	1890.9	5.36	.006*					2	.006	.57	.57	1	.006	1.20	.28
	error (main effect)	101	608.8	.	.	2	.004	.	.	79	.006	.	.	76	.008	.	.
	CBRM x ecological site	2	1045	1.31	.27					2	.003	1.09	.34	1	.01	.50	.48
	error (with interaction)	99	599.9	.	.					77	.006	.	.	75	.008	.	.
Gap (101-200cm)	CBRM									1	.15	1.54	.22	1	.03	1.76	.19
	ecological site									2	.01	1.26	.29	1	.004	.41	.52
	error (main effect)									79	.01	.	.	76	.009	.	.
	CBRM x ecological site									2	.02	1.03	.36	1	.0003	.00	.99
	error (with interaction)									77	.01	.	.	75	.009	.	.
Gap (>200cm)	CBRM													1	1.24	4.73	.04
	ecological site													1	.02	.00	.96
	error (main effect)													76	.03	.	.
	CBRM x ecological site													1	.13	2.5	.14
	error (with interaction)													75	.03	.	.

Appendix 4.2. Results from the ANOVA of the effects of CBRM and ecological site on vegetation variables that were transformed into binary values (presence and absence). For binary analysis there is no Mean Square (MS) calculated. In the eastern steppe, results from ANOVA are solely the effect of CBRM (no number in the ecological site row). ANOVA results of variables with other transformations appear in Appendix 4.1 and are not included in this table but are highlighted in grey below. *-indicates the significance, $p < 0.05$. ANOVA results of dominant species included in this table are only those that were significant by ecological zone.

Variable	Source	Mountain and forest steppe			Eastern steppe			Steppe			Desert steppe		
		df	F	P	df	F	P	df	F	P	df	F	P
Sedge cover	CBRM										1	.42	.52
	ecological site										1	2.32	.13
	CBRM x ecological site										1	3.09	.08
Shrub cover	CBRM	1	.14	.71	1	.13	.73				1	.72	.40
	ecological site	2	.81	.45							1	.234	.13
	CBRM x ecological site	2	1.49	.23							1	.28	.60
Legume cover	CBRM	1	2.30	.14	1	.09	.80						
	ecological site	2	.74	.48									
	CBRM x ecological site	2	.29	.75									
Annual plant cover	CBRM				1	6.66	.03*						
	ecological site												
	CBRM x ecological site												
Sedge biomass	CBRM				1	.09	.77	1	4.03	.05*	1	.07	.80
	ecological site							2	.41	.67	1	1.47	.23
	CBRM x ecological site							2	.36	.70	1	.36	.55
Shrub biomass	CBRM				1	2.37	.16						
	ecological site												
	CBRM x ecological site												
Litter biomass	CBRM												
	ecological site												
	CBRM x ecological site												
Standing dead biomass	CBRM	1	.07	.80	1	1.92	.20				1	.04	.85
	ecological site	2	.58	.56							1	.32	.57
	CBRM x ecological site	2	.84	.43							1	.00	.99
CLSQ – <i>Cleistogenes squarrosa</i>	CBRM	1	6.75	.01*									
	ecological site	2	1.47	.24									
	CBRM x ecological site	2	.37	.70									
KOPR– <i>Kochia prostrata</i>	CBRM							1	6.74	.01*			
	ecological site							2	1.34	.27			
	CBRM x ecological site							2	.00	.10			

ERMI - <i>Eragrostis minor</i>	CBRM										1	4.30	.04*
	ecological site										1	.19	.66
	CBRM x ecological site										1	.08	.78
Annual forb	CBRM				1	3.61	.09						
	ecological site												
	CBRM x ecological site												
Palatable perennial forb during dormant season	CBRM	1	1.05	.31									
	ecological site	2	.38	.68									
	CBRM x ecological site	2	.94	.40									
Unpalatable perennial forb during growing season	CBRM							1	.00	.96	1	.97	.33
	ecological site							2	.69	.51	1	.53	.47
	CBRM x ecological site							2	.77	.46	1	.00	.98
Unpalatable perennial grass during growing season	CBRM	1	.29	.60	1	.18	.68	1	.01	.91	1	.36	.55
	ecological site	2	1.39	.25				2	.04	.96	1	.31	.58
	CBRM x ecological site	2	.57	.57				2	1.29	.28	1	.00	.97
Unpalatable perennial grass during dormant season	CBRM							1	.07	.79	1	.03	.87
	ecological site							2	.21	.81	1	.00	.96
	CBRM x ecological site							2	.86	.43	1	.03	.86
Palatable annual forb during growing season	CBRM	1	.20	.66				1	.15	.70	1	.30	.59
	ecological site	2	.74	.48				2	.88	.42	1	2.88	.09
	CBRM x ecological site	2	.55	.58				2	.96	.34	1	.09	.76
Palatable annual forb during dormant season	CBRM	1	.07	.79				1	2.04	.16	1	.22	.64
	ecological site	2	1.29	.28				2	1.36	.26	1	1.80	.18
	CBRM x ecological site	2	.66	.52				2	.00	.99	1	.00	.99
Unpalatable annual forb during growing season	CBRM	1	.27	.61	1	1.63	.22	1	.000	.99	1	2.18	.15
	ecological site	2	.88	.42				2	.46	.64	1	.97	.33
	CBRM x ecological site	2	.74	.48				2	1.06	.35	1	.35	.56
Unpalatable annual forb during dormant season	CBRM	1	.13	.72	1	1.63	.22				1	1.07	.31
	ecological site	2	.04	.96							1	.40	.53
	CBRM x ecological site	2	.80	.45							1	1.08	.30
Palatable subshrub during growing season	CBRM				1	.02	.88	1	1.71	.20	1	.00	.99
	ecological site							2	1.28	.28	1	.23	.64
	CBRM x ecological site							2	.94	.40	1	.00	.98
Palatable subshrub during dormant season	CBRM	1	5.5	.67	1	.02	.88	1	.27	.61			
	ecological site	2	.90	.70				2	.29	.75			
	CBRM x ecological site	2	.10	.90				2	.49	.61			
Unpalatable subshrub during growing season	CBRM	1	4.33	.04*	1	.02	.90	1	.58	.45	1	1.23	.27
	ecological site	2	2.04	.13				2	.57	.57	1	.17	.68

	CBRM x ecological site	2	.07	.93				2	1.87	.16	1	1.46	.23
Unpalatable subshrub during dormant season	CBRM				1	.02	.90	1	.19	.67	1	.28	.60
	ecological site							2	.47	.63	1	.03	.85
	CBRM x ecological site							2	4.31	.02*	1	1.22	.27
Gap (101-200cm)	CBRM	1	.00	.97	1	.37	.56						
	ecological site	2	2.33	.10									
	CBRM x ecological site	2	3.62	.03*									
Gap (>201cm)	CBRM	1	.03	.87				1	1.75	.19			
	ecological site	2	.15	.86				2	.23	.79			
	CBRM x ecological site	2	.10	.91				2	.64	.53			