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DISSERTATION

DYNAMICS AND SUSTAINABILITY OF THE INNER MONGOLIA TYPICAL STEPPE – ECOSYSTEM RESPONSES TO GRAZING AND CLIMATE

Submitted by

Lindsey Christensen

Graduate Degree Program in Ecology

In partial fulfillment of the requirements

For the degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Fall 2001

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WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY LINDSEY CHRISTENSEN ENTITLED DYNAMICS AND SUSTAINABILITY OF THE INNER MONGOLIA TYPICAL STEPPE – ECOSYSTEM RESPONSES TO GRAZING AND CLIMATE BE ACCEPTED AS FULFILLLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

Committee on Graduate Work

Co-advisor

Michael B. Coughenour

Advisor

Department Head/Director

ABSTRACT OF DISSERTATION

DYNAMICS AND SUSTAINABILITY OF THE INNER MONGOLIA TYPICAL STEPPE – ECOSYSTEM RESPONSES TO GRAZING AND CLIMATE

The purpose of this study is to examine interactions of historic, present, and future climate patterns, vegetation patterns, and grazing practices to better comprehend and predict the sustainability and resilience of the Mongolian steppe grassland ecosystem. This study will utilize SAVANNA (a dynamic, spatially extensive ecosystem model) to understand, interpret, and predict how future grazing, soils, vegetation, and climate patterns interact to affect ecosystem sustainability and resilience.

Field results suggest that production of these grasslands has declined with increased grazing intensity. This information was incorporated in the model structure to examine future stability and resilience of this vegetation system. This analysis also used Savanna to simulate both vegetation and livestock production, which gave further insight into potential effects management, climate, and grazing on the sustainability of grassland systems.

Simulations that examined the effects of livestock density on vegetation showed a moderate level of grazing (~50% of vegetation removed) was sustainable for this particular system. This region was resilient to light and moderate levels of grazing; however, higher grazing intensities, in combination with low precipitation events, resulted in decreased herbaceous net primary production and increased shrub net primary

production. Once initiated, these patterns continued even by an after-the-fact removal of grazing from this region.

When predicted climate change scenarios were utilized in model simulations, results showed this grazing system was most sensitive to changes in precipitation patterns; however combinations of precipitation, temperature, and CO_2 had synergistic effects on herbaceous production. Greater increases in temperature put a larger stress on the sustainability of herbaceous vegetation, which resulted in unsustainable herbaceous biomass-alleviated only by a reduction in grazing intensity. Only increased precipitation, CO_2, or the combination of the two lessened the effects of a large increase in temperature. Simulations with climate change indicate that grazing increases the vulnerability of sustainable grassland systems to climate change. Therefore management must change grazing rates when net primary production is decreased to maintain grassland sustainability.

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SYNOPSIS

This work is a compilation of five individual research papers, each contributing to the main theme of understanding the sustainability of the Mongolian steppe grasslands in relation to land use and climate patterns. A simulation model was the main tool used for analyses, but the preparation work was also a critical component of this research. The papers in this dissertation incorporate both the preparation work and simulation analyses.

As set forth below, I outline each of the papers and how they contribute to the overall theme of this research project. I begin with **Chapter One**: a background and introduction, which provides a historical perspective of the area and a description of the ecological questions that face this region.

Chapter Two details the fieldwork research and analysis of remote sensing data at the Baiinxile Farm in Inner Mongolia, China. The purpose of this research was to determine if degradation had occurred on the grassland as a result of intensified livestock grazing, which stemmed from the hypothesis that increased livestock numbers and management changes have decreased grassland production. Fieldwork was conducted during the summer of 1997 to examine the native vegetation and determine how it differs between heavily and lightly grazed sites. Ten years of Normalized Difference Vegetation Index data, an indication of green vegetation biomass derived from Global Area Coverage satellite imagery, was used in combination with climate, elevation, soil, and livestock data to examine ecological trends in this region.

Chapter Three utilizes fieldwork from chapter one, ancillary data, and Savanna, an ecosystem model, to explore effects of livestock management on the typical steppe grasslands. Simulation runs were conducted to examine which livestock intensities were sustainable to this ecosystem to maintain grassland production that has existed for thousands of years, and which livestock intensities were unsustainable, with the potential of a vegetation state change to a shrub dominated system. This chapter also explores the resilience of this system is to grazing, and the possible vegetation state change due to shrub growth and heavy grazing on grass vegetation.

Chapter Four takes the analysis in chapter four one step further and examines how this system could potentially be affected by climate change. Grazing and grazing lands constitute a significant role in ecosystem processes; therefore it is important to understand how climate change may affect these systems. The object of this study was to use the Savanna ecosystem model to analyze potential effects of climate change and grazing on the typical steppe grasslands of the Mongolian steppe. I used a modeling approach to study the effects of increased minimum and maximum temperatures, increased CO_2, increased or decreased precipitation, and grazing on vegetation production.

Chapter 1

Overview

Background

Human managed grazing systems have existed for thousands of years yet problems still occur with their management (Noy-Meir 1975; Holling 1998). Many grazing systems around the globe are currently experiencing overuse and degradation despite the vast amount of knowledge on grazing management (van de Koppel et al. 1997). Continual mismanagement can lead to future problems in the sustainability and resilience of a system, which can be detrimental to the vegetation, livestock, and people using the land. In trying to understand proper grazing management, it is essential to understand the ecological structure and function of the grazing system in question. This is because ecosystems respond differently to grazing. Responses can differ based on ecological factors such as management type, history, and intensity; climate patterns; soil and vegetation type; topography; and nutrient dynamics of the system (Hobbs 1996; Holechek et al. 1995; Snyman 1998; Walker 1988). The purpose of this study was to examine the integration of these components to better understand the sustainability and resilience of grassland ecosystems in relation to grazing management.

This project focused on the typical steppe grasslands, defined as a bunch-grass steppe with many or few forbs, in a semi-arid climate (Lavrenko and Karamysheva 1993; Zhu 1993), where recent changes in land use patterns and land use intensity have raised questions about the sustainability and resilience of these ecosystems. It is an excellent case study because of its long evolutionary history of grazing and economic importance to the region. Nomads used this region for thousands of years for grazing but traditional nomadic patterns of rangeland use have recently been replaced with sedentary management practices (Zhu 1993) as a result of changes in the political orientation of the

Chinese government (MPR 1990). Restricted land use, along with similar or increased numbers of livestock apparently resulted in overuse and degradation of forage resources. To avoid further degradation, it is important to examine the relation between management, climate, soil, vegetation, nutrient, and topographical patterns. Because these factors are all important in determining ecosystem function, they need to be considered when trying to improve our understanding of grazing management. But it is not clear how these factors interact in grazing systems (Baars and Jeanes 1997; Behnke et al. 1993), which led to this research.

Grasslands fall into a region of variable climate patterns and are highly susceptible to degradation by overuse (Parton et al. 1996). A small shift in climate patterns may increase vulnerability to degradation. This project modeled the effects of both grazing management and climate change on grasslands to better understand how they affect ecosystem function.

Introduction

Grazing management is a form of applied plant ecology that optimizes plant growth, quality, and offtake within a defined time period (Walker 1988; Westoby et al. 1989; Snyman 1998). This definition is limited because it does not include long-term effects on the system. To better understand grazing management, it is necessary to understand the interaction of ecological characteristics in the system, such as soil, topography, vegetation, management, herbivory, and climate patterns, and how they relate to the resilience and sustainability of the system. These factors together create a range ecosystem. An ecosystem model is an appropriate tool to integrate these factors

and allow for a better understanding of how this system functions in terms of grazing management, and its relation to resilience and sustainability.

Overuse and degradation can have serious consequences on the sustainability and resilience of typical steppe grasslands. This could be devastating for the people who rely on this land. When discussing sustainability, it is important to determine its focus: whether it is for livestock, vegetation, to support current human populations, or to allow for increased human populations. Because China has an overall policy of zero human population growth (although exceptions exist), this project focused on the sustainability of vegetation and livestock health at current human population levels. Sustainable management is defined as the non-degradive use of ecosystems that maintains system health for present and future generations (Lele and Norgaard 1996; Lubchenco et al. 1991; Cairns 1997). Understanding how managers can graze these grasslands and maintain herbaceous production to support the influx of people in this region will help to identify a sustainable use of this system (Dasgupta et al. 2000, Hodgson and Silva 2000). Local herders in the Inner Mongolia region fear they are experiencing a marked reduction in grassland productivity, which threatens the sustainability of their land (personal communication with herders 1997). A reduction in productivity would be surprising due to the long history of grazing in this area. Reduced productivity could mean fewer numbers of livestock could be supported on these lands or the health of livestock would decrease.

When examining the sustainability of a system, it is important to understand how resilient the system is to disturbances. Resilience is defined as "a measure of the persistence of systems and their ability to absorb change and disturbance and still

maintain the same relationships between populations or state variables" (Holling 1973). The grasses in this region have evolved with and should be able to withstand grazing (Coughenour 1985; Zhu 1993). Therefore, there should be high resilience to grazing as a disturbance. But the resilience of this system has been questioned due to changes in management. Has the shift in grazing management caused a decrease in the resilience of the system? If so, this could have major consequences not only on the vegetation and livestock, but also on the people because of the economic importance of livestock production (Williams 1996a; Gao and Zhang 1997). A decrease in resilience could leave the system vulnerable to degradation due to heavy grazing, and cause a change in vegetation structure, such as an increase in bare ground. In the following paragraphs, I define the development of grazing ecology and how this information has led to our present understanding of grazing systems.

Most grazing systems have been and are traditionally managed with the Clementsian model of equilibrium dynamics (Clements 1936; Walker 1988). This assumes vegetation systems move towards a climax state when undisturbed. Grazing, within the theme of Clementsian science, is considered a disturbance that inhibits vegetation from reaching this final stage in development. By removal of grazing pressure, vegetation is able, and should continue to grow towards the climax state. This view led to the equilibrium model of plant-herbivore dynamics, which suggests plants and herbivores are balanced around an ecological equilibrium, as first described in simple predator prey systems (Rosenzweig and MacArthur 1963). The model of equilibrium dynamics remained the dominant theory behind most management until the middle of the 1970's.

Holling (1973) challenged the equilibrial view of natural systems by describing systems as being transient, not necessarily following equilibrium dynamics. He suggested systems have certain resilience to disturbances, but resiliency differs for different systems. When resiliency is low, disturbance can shift a system to another 'state' or condition that would differ from the ecological response of an equilibrium system. Wiens (1974) contributed to the idea that all systems are not in equilibrium with empirical evidence from a study on bird communities. He found abiotic factors have an overriding influence on some systems and systems were not necessarily affected by density dependent factors. He suggested equilibrium systems may exist, but large fluctuations in environmental processes may shift the system into non-equilibrium (Wiens 1977). He pointed out that systems do not have a tight coupling between predator and prey, but respond to different factors such as severe climate events or stages in population growth. Noy-Meir (1975) further explored Holling's ideas about departure from true equilibrial systems by examining interactions of plants and herbivores, and found some grazing systems to have a discontinuous stability. He used graphical representations of grazing systems, pointing out the existence of stable, multiple stable, and non-stable states due to different levels of plant and herbivore densities. DeAngelis and Waterhouse (1987) found numerous studies of systems that showed nonequilibrium dynamics due to nonlinear feedbacks, time lags in the system recovery process, and stochastic forcing from the environment. These foundational papers initiated a change in the way scientists viewed ecological systems.

The foundational papers found ambiguity with the current paradigm of systems in equilibrium, and future research turned to understanding the mechanisms behind these

systems. This involved understanding abiotic factors and their relation to ecosystem function. Shepherd and Caughley (1987) discussed causes behind equilibrium and non-equilibrium systems. They noted that variation in interannual precipitation patterns could help determine the equilibrial state of a system. A high coefficient of variation in interannual precipitation (CV > 0.3) would define a non-equilibrium state because the system is not in equilibrium due to time lags in the recovery process. A low CV in interannual precipitation (CV < 0.3) would characterize an equilibrium system. Ellis and Swift (1988) found arid and semi-arid ecosystems in Africa did not show equilibrial conditions, and were controlled more by stochastic perturbations and abiotic factors, rather than biotic factors. They suggested a paradigm shift to recognize non-equilibrium dynamics and how systems should be managed with this in mind, rather than the equilibrium view which focuses on the climax paradigm. Walker (1988) expanded this idea and discussed the importance of including autecological factors, community processes, climate, fire, and livestock management to determine rangeland dynamics because the equilibrium view and Clementsian succession were not always appropriate ways to view rangeland ecosystem dynamics. Westoby et al. (1989) proposed a new model for range management for non-equilibrium systems named the State-and-Transition Model. It recognized the dynamics of rangelands by describing different "states" of vegetation, and the importance of understanding the transitions between states, caused by fire, weather and management.

Although non-equilibrium patterns have been found in ecological systems, ecosystems do not follow exact patterns, which leaves questions in understanding how they function. It is well established that systems can fall into equilibrium or non-

equilibrial patterns based on abiotic and biotic factors (Behnke et al. 1993; Illius and O'Connor 1999). Illius and O'Connor (1999) found equilibrial and non-equilibrial patterns exist in all systems, depending on the scale that is being studied. Factors such as grazing intensity, topography, and soil type can cause dis-equilibrium patterns in equilibrium systems or cause equilibrium patterns in non-equilibrium systems (van de Koppel et al. 1997; Illius and O'Conner 1999).

This brings us back to the equilibrium/nonequilibrium view in relation to ecosystem function. It has yet to be determined or defined how system function relates to equilibrium and non-equilibrium dynamics. Systems are never at a "true" equilibrium due to variation in abiotic and biotic patterns, nor can they be non-equilibrium, because positive feedbacks exist that link systems abiotically and biotically (Bonan 1997). This brings us back to Wiens' idea that systems fall on a continuum between equilibrium conditions with strong biotic interactions and non-equilibrium conditions with strong abiotic controls (Wiens 1977). A goal of this project is to define at the landscape scale where certain grazing and climate regimes fall on this continuum to better understand the effects of grazing on ecosystems over a hundred year grazing period.

Grazing history plays a role in ecosystem function. Some grasses have evolved with grazing and are more able to withstand grazing pressure (Coughenour 1985); thus time, as well as location, has an effect on ecosystem function. Ecosystems can also respond differently based on grazing history and moisture patterns. Milchunas et al. (1989) found systems with a long evolutionary history of grazing were better able to withstand higher grazing intensities, which is also dependent on different moisture

regimes. This suggests history, as well as climate and vegetation patterns, is important when trying to understand system function in response to grazing.

There are also many other factors in grazing management besides history, vegetation, and climate, and it is important to understand how these factors interact in a system. Other factors include soils, topography, management history, and grazing patterns (Coughenour 1985; Ellis and Swift 1988; Milchunas et al. 1988; van de Koppel et al. 1997). It is difficult to create grazing management plans at the regional scale because grazing systems each have their own dynamics. Therefore, it is important to focus on particular systems and the different factors that create the system.

This project focuses on the typical steppe grassland in Inner Mongolia, China. Inner Mongolia has 300,000 km^2 of grasslands, with 210,000 km^2 of typical steppe located within 42°N – 48° N in latitude and 108°E and 119°E in longitude. This region of grassland lies south of Siberia, 2160-km inland from the eastern coast, and cradles the eastern side of Mongolia. Elevation ranges from 800 m to 1400 m. The three main soil types in this region are kastanozems, chernozems, and eolian sandy soils (FAO 1991). Dominant vegetation is composed of perennial herbaceous xerophilous and meso-xerophytic species (Zhu 1993). Dominant grasses include *Stipa grandis, Leymus chinensis,* and *Cleistogenes squarossa* while the dominant forb is *Artemisia frigida* (Li 1978; Liu 1989). *Caragana microphylla* is the dominant shrub (Li 1978).

The typical steppe has a temperate/semi-arid climate, with precipitation values ranging from 180 mm to 500 mm annually. The area receives its precipitation from the summer monsoon coming from the East. Air currents, which bring moisture from the ocean, are warm. This gives the region a warm wet climate during the growing season.

Seventy percent of total precipitation is received during May through August (Xiao et al. 1997). Because the majority of precipitation occurs during the growing season, biomass production is high as compared to other regions with similar amounts of precipitation but with smaller proportions received during the growing season.

Temperatures range from –27°C in winter to 28.6°C in the summer, with an annual mean temperature of –0.4° C (Xiao et al. 1997). Cold winters are a result of the jet stream moving southwestward from Siberia. The Daixinganling Mountains that border the eastern boundary of the grassland block the cold front from continuing eastward, so cold air lingers and winters remain long. Temperatures begin to rise in March, but the high winds and minimal precipitation from March to May postpone vegetation growth until the rainy season (Zhu 1993).

Herders have coped with these extreme weather patterns for thousands of years by leading a nomadic lifestyle. Following this lifestyle, herders move their herds from area to area depending on location of the best resources. When resources were depleted, they were freely able to move their herds to more plentiful areas. This grazing pattern decreased the amount of grazing pressure on specific areas because livestock could freely move over large areas. Livestock numbers not only fluctuated over these areas, but numbers also varied depending on climate conditions. In mild years, herds increased in size, while extremely cold or dry years caused numbers to decrease.

In 1949, the Chinese government took administrative control of the Inner Mongolia area and changed management strategies and land use patterns (Thwaites et al. 1996a). The Chinese government forced migrations of Han Chinese to Inner Mongolia and mandated grazing mixed with cropping land use practices (National Research

Council 1992; Williams 1996a). They also began to establish collective communities, abolishing the traditional management system of the Mongolian people. One goal of the collectives was to gain access for farming in pastoral areas (Yenhu 1996). This resulted in the building of permanent homes, decreasing available land for livestock, and increasing livestock numbers to support the increase in population. Control of livestock shifted from the nomadic people to an established administrative system (Yenhu 1996). Land that was once assigned to a family unit became an area for all communal members to use. Territorial boundaries were organized where traditional nomads previously moved freely. Because of the influx of people less educated in pastoralism and the change in administrative rule, there was higher proportion of people who lacked the traditional knowledge of taking care of these communal areas.

In 1978 the government dismantled the collectives and divided communal property between households because of increasing degradation (Williams 1996a; Zhang 1996). This change in policy was called 'Responsibility System' and was implemented to shift the level of administration to a more local scale (Yenhu 1996). This included a change in strictly government owned livestock to both private and government owned herds. Overall consequences from these political changes were replacement of traditional patterns of nomadic rangeland use with sedentary practices and new forms of grazing management (Zhu 1993). Restricted land use along with similar or increased numbers of livestock apparently resulted in overuse and degradation of forage resources. Consequently, the steppe's long-term viability under current management strategies is questionable and susceptible to degradation.

This study focused on the Baiinxile Livestock Farm within the typical steppe. It is located within the Xilingol River Basin, Inner Mongolia. The farm's main purpose is the production of wool, although horses, cows, and camels graze in the area and are used for market purposes. The farm is 3,680 km^2, divided into an administrative village and 12 branches. There are many levels of management at the livestock farm, which include national, provincial, Baiinxile Farm, branch, and family levels. China implemented the "Double Contract Household Production Responsibility System" in this region during the 1978 reforms (Thwaites et al. 1996a). This privatized livestock that was once owned by the collectives, but some animals were kept under government ownership. Privatization allowed families the freedom to regulate herd size. One result of this change in management was increased herd size driven by increases in living standards, personal wealth, and market values (Thwaites et al. 1995).

The typical steppe was selected because it is undergoing land use change (National Research Council 1992) and is vulnerable to climate change (Chase 2000). The threat of degradation (National Research Council 1992; Chen and Xiao 1993; Thwaites et al. 1996b; Williams 1996b; Gao and Zhang 1997) in a system that is highly adapted to grazing (Zhu 1993) shows the need to better understand ecosystem functions in response to grazing and climate. Grazing and climate effects can be defined as biotic and abiotic effects. The degree or combination of these effects on this ecosystem are unknown, but could play a large role in understanding the ecosystem's sustainability in relation to grazing management.

Because changes in climate patterns are occurring (IPCC 1996; Singh et al. 1998; IPCC 2001), it is necessary to understand how systems will respond to climate change. A

change in climate can significantly alter ecosystem function and structure (Xiao et al. 1995; Coffin and Lauenroth 1996; Gao and Zhang 1997; Luscher and Nosberger 1997) and could change the amount of livestock that can be supported on a sustainable basis. Because grasslands fall into a region of variable climate patterns, they are highly susceptible to degradation when overused (Parton et al. 1996). If degradive management patterns prevail, a change in climate could have compounding results and increase the rate of degradation. Projected climate changes are increases in temperature and CO_2 and changes in temperature and precipitation variability (Mearns 1993; Ojima et al. 1993; Ojima 1995; Riha et al. 1996; Semenov et al. 1996; Reddy et al. 1997). Simulated results from climate change scenarios are increased water use efficiency due to more efficient use of CO_2, shifts from C3 to C4 communities, and either increases or decreases in simulated vegetation yields depending on climate and soil type (Riha et al. 1996; Reddy et al. 1997; Gao and Yu 1998; Hall 2000).

Land use and climate change are occurring (Keeling et al. 1995; IPCC 1996; Watson et al. 1996; Diaz et al. 1998; IPCC 2001), so it is imperative that we gain a better understanding of grassland ecosystem responses to land use and climate by integrating climate, vegetation, management, and soil processes and patterns. Because these factors are all important in determining ecosystem function, they need to be considered when trying to determine proper grazing management. But it is not clear how these factors interact in grazing systems (Behnke et al. 1993; Baars and Jeanes 1997), which led to this research. For proper grazing management, it is necessary to develop a quantitative understanding of abiotic controlling factors and their interactions with biotic components.

The effects of grazing must be better understood to avoid an increase in degradation that is occurring today.

An effective way to understand the combination of these factors is to construct an explanatory and predictive ecosystem model and then use it to conduct analyses, syntheses and to make predictions (Agren et al. 1991; Burke et al. 1991; Riebsame et al. 1994; Hall and Day 1997). The model can be used to understand interactions between abiotic and biotic factors to determine which grazing practices and intensities of use might be appropriate to avoid mismanagement (Wiens 1984; Caughley et al. 1987; Ellis et al. 1993). Many models have successfully been used to study grasslands (Hunt et al. 1991; Coffin and Lauenroth 1996; Gao et al. 1996; Parton et al. 1996; Pickup 1996; Coughenour and Parton 1997) while most integrate vegetation, soils, and climate, few consider ungulate dynamics and ecosystem responses to grazing. The SAVANNA model (Coughenour 1993) was developed to simulate both vegetation and ungulate dynamics based on climate, topography, soils, nutrients, vegetation, and grazing patterns. The model was used to examine the integration of these components to better understand the sustainability and resilience of grassland ecosystem in relation to grazing management and climate change.

The SAVANNA model was developed to simulate a wide range of mixtures of grasses, shrubs, and trees, including vegetation growth and ungulate herbivory. It is a spatially explicit, dynamic, process-oriented model that can simulate ecosystem processes at both a local and regional scale. Ecosystem processes are modeled by using soils, topography, climate data, and livestock density to affect vegetation production and cover, livestock forage utilization, livestock production, and human offtake. SAVANNA

provides a realistic representation of ecosystem processes because it is process-based, representing flows of biomass, nitrogen, and organisms and subsequent rates of change in system states. Its one-week time step allows for large spatial and temporal runs. These dynamic capabilities allow it to simulate interactive responses to grazing and climate change.

Global Objective

The objective of this study is to achieve a better understanding of ecosystem dynamics in relation to climate and grazing by integrating grazing, vegetation, and climate patterns on the typical steppe with a spatial dynamic ecosystem model. The development of spatially extensive models for grazing ecosystems is critical for understanding, interpreting, and projecting system change.

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Chapter 2

Field and Remote Sensing Analysis of Grazing Effects on the Typical Steppe Grassland of Inner Mongolia

Abstract

Vegetation analyses using field experiments and remote sensing data were conducted to examine the effects of livestock grazing on the typical steppe grassland in Inner Mongolia. Fieldwork was conducted at the Baiinxile Farm, Inner Mongolia during the summer of 1997 to examine vegetation cover and biomass, and how it differs between heavily and lightly grazed sites. Field research showed effects of intensified grazing practices: percent grass canopy cover was significantly higher in light compared to heavy grazed plots ($R^2 = 0.45$, $P = 0.033$) and percent bare ground was significantly higher in heavy grazed plots ($R^2 = 0.81$, $P = 0.0079$). On lightly grazed plots compared to heavy grazed plots, grass biomass was significantly higher ($R^2 = 0.59$, $P = 0.0096$).

Eleven years (1982-1992) of Normalized difference vegetation index (NDVI) data, used as an indication of above ground live biomass, was used in combination with climate, elevation, and livestock data to analyze trends in vegetation at the same site. Analysis of satellite imagery showed the adjusted NDVI (NDVI * precipitation^{-1} (mm)) declined with increasing livestock numbers ($R^2 = 0.78$, $P = 0.049$). Based on field and remote sensing analyses, this system is subject to biomass reduction in response to increased livestock despite grasses being well adapted to grazing.

Introduction

Inner Mongolia contains about 30% of China's grasslands (Ellis 1992) and is economically important to the people of China (Zhu 1993). Recent grazing management changes and increased human population levels have caused increased grazing pressure and degradation of these rangelands (Sneath 1998, Chen 1996). Research for this study

was conducted at the Baiinxile livestock farm, which lies within the typical steppe grassland of China. The typical steppe grasslands, defined as a bunch-grass steppe with many or few forbs in a semi-arid climate, have supported and sustained grazing as the dominant form of land use for over 2,000 years (Zhang and Skarpe 1996).

Within the last 50 years, the grasslands at the Baiinxile Farm have gone through many changes due to major shifts in both livestock management and governmental policy. Restricted land use, along with similar, or increased numbers of livestock occurred in the late 1940's, which apparently resulted in overuse and degradation of forage resources (Chen 1996; Thwaites et al. 1996). Local people claim increasing livestock levels have caused a decrease in vegetation biomass and a change in species composition. Consequently, the area's long-term viability under current management strategies is questionable and susceptible to degradation. The objective of this study was to use field and remote sensing data to gain a better understanding of the effects of grazing management on the grasslands in this region.

Methods

Study Area

The study was conducted at the Baiinxile Livestock Farm in Inner Mongolia, China (43.5° North Latitude and 116.5° East Longitude). The primary agricultural product at the livestock farm is wool production, yet cattle and horses are also raised for market purposes. The livestock farm covers 3600 km^2, and is composed of 12 branches and a livestock farm headquarters (LF) (Fig. 2.1). A branch is an administrative division of land, ranging in size from 79 km^2 to 655 km^2 (Table 2.1) with an average elevation of

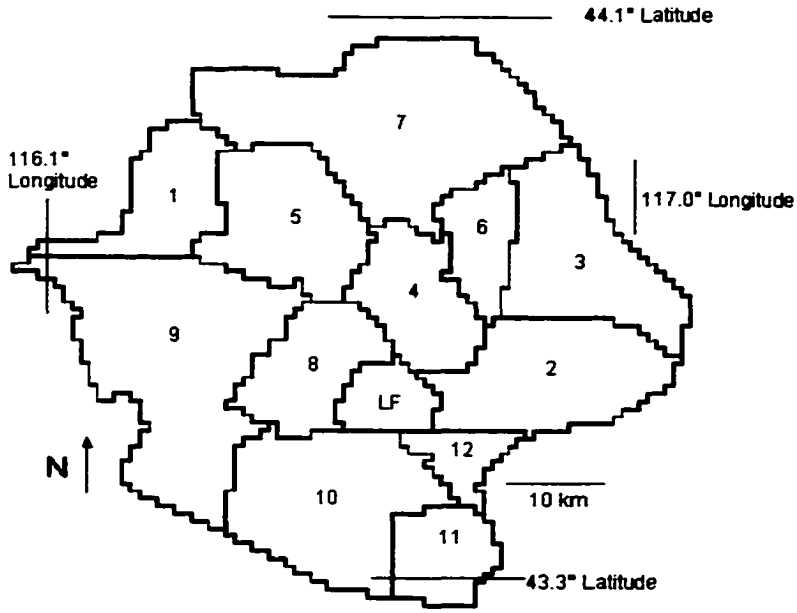

Figure 2.1. Spatial layout of the Baiinxile Livestock Farm, divided into 12 branches and a livestock farm headquarters (1 cell = 1 km^2).

Table 2.1: Summary statistics of Baiinxile livestock farm.

Branch	Area (km2)	Elevation (m)
1	196	1182
2	334	1314
3	323	1314
4	212	1217
5	301	1186
6	130	1324
7	655	1225
8	208	1212
9	609	1128
10	412	1229
11	141	1311
12	79	1240

1200 m (EDC 1996). Elevation data, at a 30 arc second (approximately 1 km^2) resolution, was averaged for each branch.

Vegetation in this region is dominated by *Leymus chinenses*, *Stipa grandis*, *Artemisia frigida*, and *Caragana sp.* (Li 1978; Liu 1989). Dominant soils are kastanozems, chernozems, and eolian sandy soils (FAO 1991). The temperatures range from –27.0° C to 28.6° C, with a mean annual temperature of -0.4° C (Li 1989). Mean annual precipitation is 360 mm but varies between 180 and 500 mm (Zhu 1993).

Field Data

Three of the livestock farm's branches were sampled in a randomized incomplete block design with two treatments. The treatments consisted of heavily and lightly grazed sites (3 replicates for the lightly grazed and 6 replicates for the heavy grazed). Lightly grazed sites were located in permanent exclosures (in existence for 17 years, yet animals occasionally grazed) or areas reserved for hay cutting in the fall (2 year old fenced area, also reserved for hay cutting) while heavily grazed sites were located in areas within two kilometers of households where heavy grazing existed. Vegetation biomass and percent cover were sampled to determine vegetation condition and biomass on heavily and lightly grazed sites. Plots were 1000 m^2, randomly located in defined areas. Within each plot, twelve 20 cm x 50 cm frames were randomly located to estimate percent cover (Daubenmire 1959). Ocular estimates were used to determine percent cover of grass, forbs, shrubs, and bare ground. Four of the twelve frames were clipped to determine plant biomass. Plants were removed at base of stems for collection. Plant biomass samples were taken to the Inner Mongolia Grassland Ecosystem Research Station

(IMGERS), separated into functional groups (grasses, forbs, shrubs), oven dried, and weighed to obtain plant dry weight.

Soil samples were collected at each site and brought to the Natural Resource Ecology Laboratory, Colorado State University for analysis. Soil cores, twenty centimeters in depth, were taken from the four frames where vegetation biomass was collected. Soil samples from each plot were combined, air-dried, and analyzed for texture by the modified hydrometer method (Gee and Bauder 1986).

Satellite imagery

The Normalized Difference Vegetation Index (NDVI) derived from satellite imagery was used as an indication of above ground live biomass to analyze vegetation trends (Tucker et al. 1985; Tucker and Sellers 1986). Eleven years (1982-1992) of Advanced Very High Resolution Radiometer (AVHRR) satellite imagery data was obtained from the National Oceanic and Atmospheric Administration (NOAA) 7, 9, and 11 satellites (Justice et al. 1985). Daily 1 km^2 resolution satellite images were used to create global area coverage (GAC) data with an approximate 4 km^2 resolution (Malingreau 1986). NDVI bimonthly composites were derived from channel 1 (RED) and channel 2 (NIR) of the satellites (Tucker 1979; Tucker et al. 1985) using the equation:

$$NDVI=((NIR - RED)/(NIR + RED)). \quad 1.$$

Channel 1 is the visible channel (0.58-0.68 μm), over which chlorophyll absorbs much of the incoming radiation while channel 2 is the near-infrared channel (0.725-1.1 μm), over which much of incoming radiation is reflected (Tucker et al. 1983; Justice et al. 1985).

This ratio gives an index of greenness because near-infrared reflectance from green surfaces is always greater than the visible reflectance (Tucker 1979). The NDVI data were composited over time intervals of two weeks. The maximum NDVI value over this period was taken to create a biweekly composite. Benefits of using GAC data include minimized cloud cover effect, reduced atmospheric aerosol effect (Holben 1986), and long temporal coverage (Justice and Hiernaux 1986). NDVI was rescaled to the interval −1.0 to 1.0.

NDVI values were averaged for each branch of each satellite image (12 years, 2 images per year). Peak values of each year were chosen to represent peak live standing crop for each branch. Regression analysis was used to test if precipitation was significantly related to peak NDVI values (Myneni et al. 1998; Nicholson et al. 1998; Prince et al. 1998). Peak NDVI was divided by annual precipitation values to account for yearly fluctuations in precipitation over the 11-year period. Annual precipitation values were based on precipitation for the 12 months prior to peak NDVI, therefore monthly values were summed from September through August. Other studies have used combinations such as growing season NDVI (integrated from July to October) and annual precipitation (Myneni et al. 1998), growing season NDVI (integrated from July to October) and seasonal (June-October) precipitation (Nicholson, Tucker et al. 1998), and annual NDVI (integrated from June to May, growing season plus the following dry season) and average annual precipitation (Prince et al. 1998). Peak NDVI and annual precipitation (September through August) were used for this study because of: 1) varying time lags between precipitation events and plant growth, 2) the availability of only one

weather station, and 3) the complication of non-vegetative variables within NDVI data (Myneni, et al. 1998).

Analysis of variance (ANOVA) (Statistical Analyses Software SAS 1989) was used to test if adjusted NDVI increased or decreased over time, blocking on branches. Because elevation was averaged for each branch, blocks on branches could be equated to elevation. Regressions were also conducted with livestock numbers to better understand impacts of grazing on this ecosystem.

Results

In summer of 1997, grass biomass (g m^{-2}) was significantly higher in lightly compared to heavily grazed plots ($R^2 = 0.59$, $P = 0.0096$) (Fig. 2.2). There were no significant differences in forb and shrub dry weight biomass between grazing treatments. Percent grass canopy cover was significantly higher in lightly compared to heavily grazed plots ($R^2 = 0.45$, $P = 0.033$) and percent bare ground canopy cover was significantly higher in heavily grazed plots ($R^2 = 0.81$, $P = 0.0079$). There were no significant differences between heavily grazed and lightly grazed sites with respect to litter, forbs, or shrub percent cover (Fig. 2.3). Soil texture measurements showed no significant difference between grazing intensity.

Peak NDVI was significantly correlated with precipitation (summed from the previous 12 months) ($R^2 = 0.54$, $P < 0.0001$). The ANOVA indicated that the adjusted NDVI decreased significantly over time ($R^2 = 0.5$; $F = 10.06$; $P < 0.0001$) (Fig. 2.4). Further regressions of adjusted NDVI with livestock data showed adjusted NDVI significantly decreased as livestock numbers increased ($R^2 = 0.78$, $P = 0.049$) (Fig. 2.5).

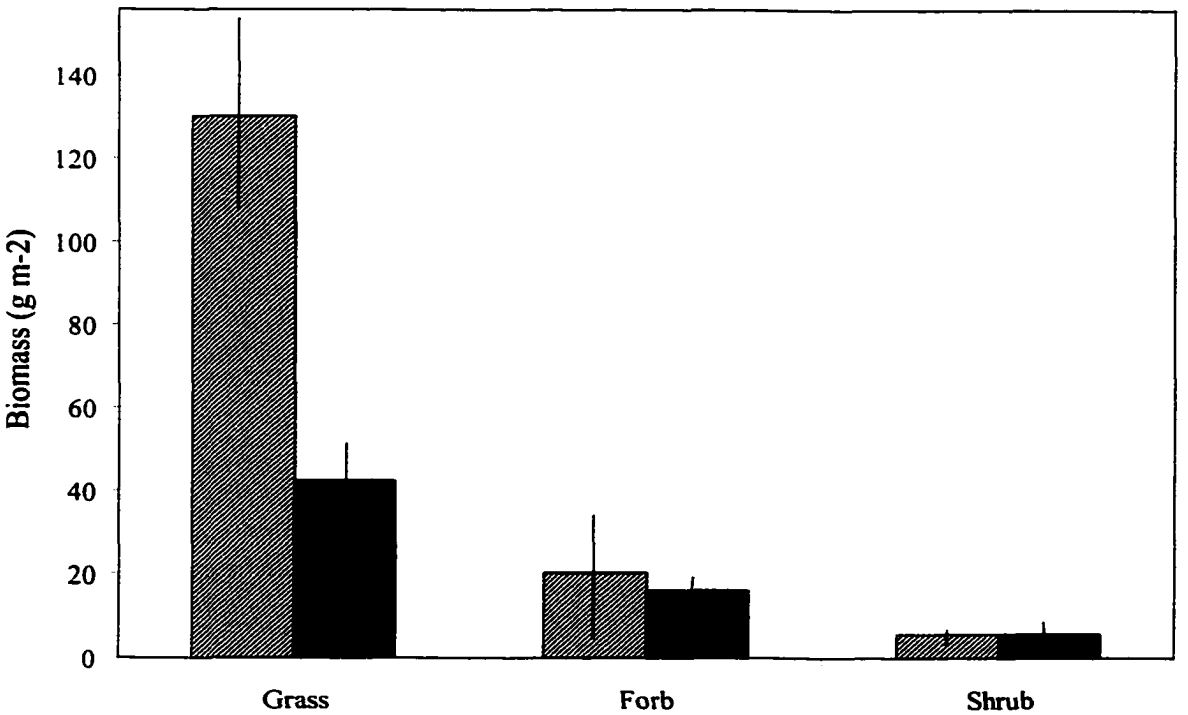

Figure 2.2. Dry biomass (g m^{-2}, ± SE) of three plant functional groups for light (diagonal lines) and heavy (checkered) grazing treatments.

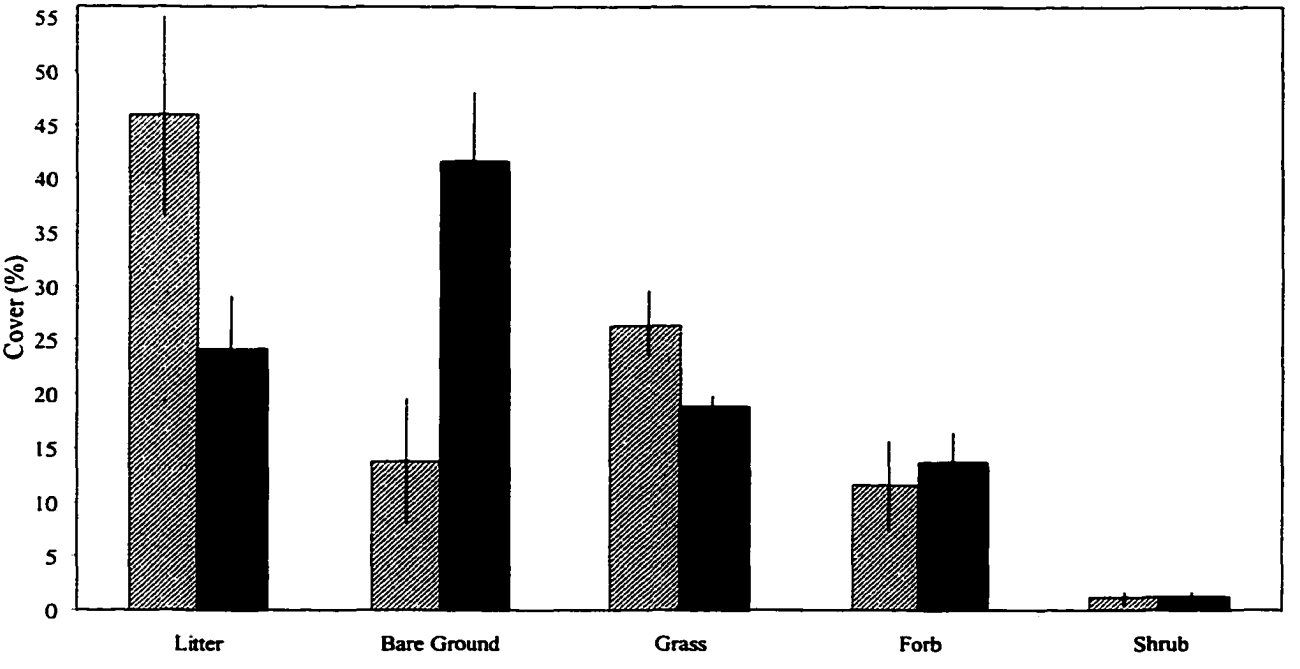

Figure 2.3. Percent canopy cover of litter, bare ground, grass, forb, and shrub (± SE) for light (diagonal lines) and heavy (checkered) grazing treatments.

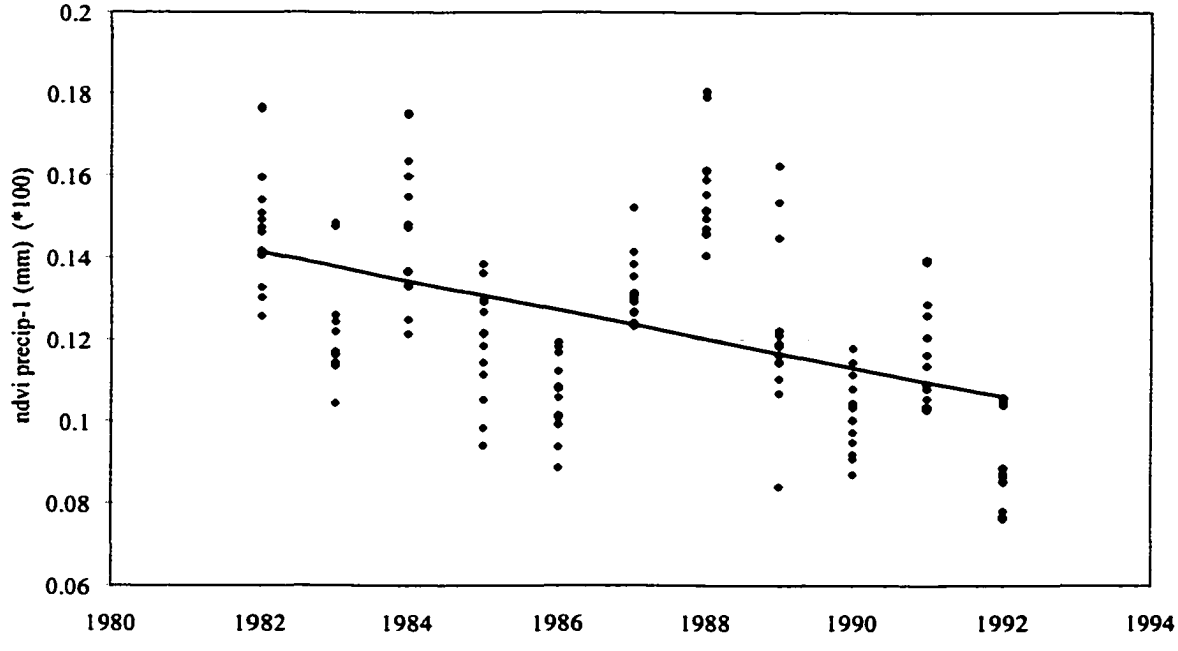

Figure 2.4. Adjusted NDVI [NDVI * precip^{-1} (mm)] (*100) plotted over time with linear trendline. Each point represents NDVI value from Baiinxile farm branches (12 per year). High adjusted NDVI values in 1988 occurred when livestock populations were at low levels.

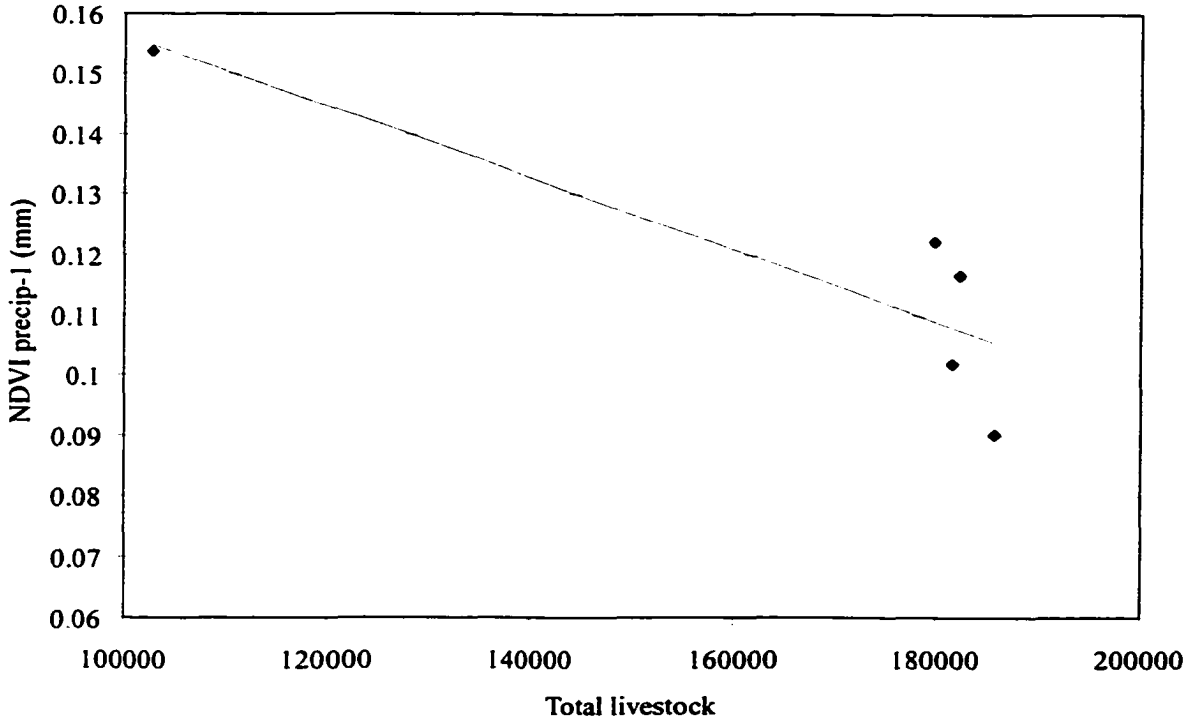

Figure 2.5. Adjusted NDVI [NDVI * precip^{-1} (mm)] (*100) compared to increasing livestock numbers ($R^2 = 0.78$, $P = 0.049$).

Average NDVI in 1982 was 0.44 with 283 mm precipitation and 102,858 head of livestock as compared to 0.41 in 1992 with 455 mm precipitation and 185,829 head of livestock.

The significant relationship between NDVI and elevation helped to explain some of the variation in adjusted NDVI over time. Because elevation was averaged for each branch, blocks on branches could be equated to elevation. Regression of adjusted NDVI on elevation also had a positive significant correlation, but was less significant than peak NDVI ($R^2 = 0.14$; $P < 0.0001$).

Discussion

Degradation can be interpreted as "a grazing-induced reduction in the amount of vegetation cover likely to be present after the best growth conditions experienced within a reasonable time" (Bastin et al. 1993). Results from this study showed a significant increase in the percentage of bare ground cover at heavily grazed sites with a decrease in the amount of grass canopy cover. Plant biomass was also lower with heavy grazing. The vegetation differences were not attributed to soil texture because it did not differ between treatments. Because samples were not taken throughout the growing season following release from grazing, it is premature to conclude that the decrease in grass biomass observed in vegetation plots is a form of degradation. There could be more production at these sites because of growth compensation (McNaughton 1979). On the other hand, the increased bare ground cover and decreased grass canopy cover in grazed relative to ungrazed plots is noteworthy because it represents a physical change that has occurred due to grazing and are indicative of a decrease in plant potential production. Field data suggest changes are due to heavy grazing.

The simple regression showed adjusted NDVI significantly decreased over time, but further analyses indicated the increase in livestock attributed to this decrease. The increase in livestock numbers in Figure 2.5 coincide with time, whereas low livestock numbers occurred in 1982, and increased with time until livestock numbers peaked at 185,829 in 1992. Farm livestock data was limited for this analysis, unavailable from 1983 to 1988, therefore a large gap exists in the data. However, it is important to note the temporal trend of both increasing livestock and a decrease in adjusted NDVI, which signifies a system under change. Although these changes are not necessarily permanent, the increase in grazing intensity as a result of increasing livestock numbers could increase the system's vulnerability to degradation (Sneath 1998).

From field results and NDVI data on the typical steppe, it appears that although the system has a long history of grazing, recent management practices have had an effect on the vegetation. This supports the notion that rangeland overuse and degradation by livestock grazing is occurring in this area (National Research Council 1992; Chen and Xiao 1993; Williams 1996; Gao and Zhang 1997). This information will be further used in an ecosystem model to study ecosystem responses to land use by integrating vegetation, soil, climate, and grazing patterns. Land use change is at the forefront of ecological concern and it is imperative that we gain a better understanding of grassland ecosystems to learn the effects of this change.

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Chapter 3

A Systems Approach to Grazing Management on the Typical Steppe

Abstract

The sustainability and resilience of a typical steppe grazing system was assessed by determining thresholds and stable states with an ecosystem simulation model. This analysis used SAVANNA to simulate spatial climate, vegetation, and livestock grazing dynamics, at 14 different livestock densities (5.5 – 59.8 AU km^{-2}). Grazing effects on vegetation were assessed, including effects on primary production, vegetation composition, and root biomass. Simulations were run for 100 years: 50 years to examine sustainability and 50 years to examine resilience of the system. Results showed that a grazing intensity (1-g/u; g = biomass in grazed area, u = biomass in ungrazed area) of 0.49 was sustainable for this particular system. This region was resilient to grazing up to the intensity of 0.49, where the system remained dominated by herbaceous production. Grazing intensities higher than 0.49, in combination with low precipitation events, resulted in decreased herbaceous net primary production and root biomass, and increased shrub net primary production and root biomass. Herbaceous vegetation was unable to gain a competitive advantage over shrubs in areas where grazing intensities were above 0.49; consequently, the system shifted to a stable shrub-dominated state that could not return its original composition even without further grazing.

Introduction

The long-term sustainability and resilience of typical steppe grassland systems are under pressure as a result of overuse and mismanagement of livestock due to changes in management structure over the last 50 years (Chen and Xiao 1993; Chen 1996; Thwaites et al. 1996; Williams 1996b; Renzhong and Ripley 1997; van de Koppel et al. 1997;

Hiernaux 1998; Snyman 1998). Prior to 1949, herders led a nomadic lifestyle, moving their herds from area to area based on location of resources and climate patterns. This traditional management style existed when both human and livestock populations were small and people followed cultural traditions for management decisions. The grassland behaved as a non-equilibrium system (Holling 1973; Noy-Meir 1975; Wiens 1977; DeAngelis and Waterhouse 1987; Ellis and Swift 1988) in that it was not strongly regulated by density dependent factors. Instead, herd sizes decreased and increased in response to variable climate patterns. For example, if a drought or extremely cold winter occurred, herd sizes were reduced due to culling or starvation (Chen 1996).

In 1949, the Chinese government took administrative control of the Inner Mongolia area, forced migrations of Han Chinese into Inner Mongolia, and mandated land use practices of grazing mixed with cropping (National Research Council 1992; Thwaites et al. 1996; Williams 1996b). They also established livestock collectives, abolishing the traditional management system of the Mongolian people. This was not successful so in 1978 the government dismantled the collectives and installed the Individual Responsibility System, which shifted the level of administration to a local scale (Williams 1996a; Yenhu 1996; Zhang 1996). A system of strictly government owned livestock was converted to one in which there was both private and government owned herds. These politically driven changes in management resulted in increased livestock numbers, increased land degradation, and herders with a lack of traditional knowledge of grazing management. Management was based on economic gain, with an incentive to maximize the number of animals per unit of land. Consequently, this region has recently seen increases in livestock populations (Fig. 3.1) (Liu 1989 and data from

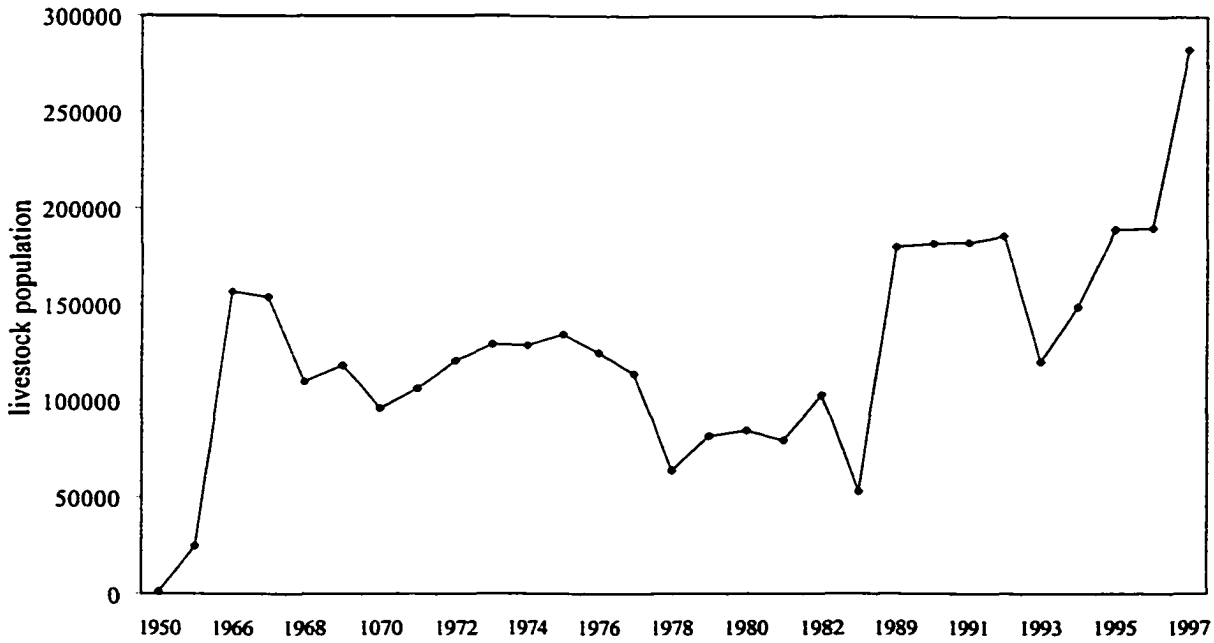

Figure 3.1. Total livestock populations at the Baiinxile Farm, Inner Mongolia during time of political and management change from the late 1950's to the late 1990's. Establishment of the Farm occurred in the 50's (recorded livestock numbers equaled 1,019), therefore increases in first 10 years represent increases in Farm area. Livestock decreases in 1968, 1977, and 1988 occurred during severe storms and droughts (Source: Liu 1989 and data from Inner Mongolia Grassland Research Station).

Inner Mongolia Grassland Research Station). It has in effect been managed as an equilibrium system, in which livestock numbers have been held constant, or have been allowed to gradually increase from year to year. This is now recognized as inappropriate management for this region due to observation of increased shrubs and grassland degradation by the local herders.

Noy-Meir (1975) first discussed how grassland systems could shift among multiple stable states due to threshold effects in grazing systems. He applied Rosenzweig and MacArthur's (1963) theory on the stability of predator-prey systems to grazing systems in an attempt to understand stocking rates, productivity, and stability of grazing systems. The importance of understanding multiple stable states and thresholds in grazing management has been stressed in the literature (Laycock 1991; Rietkerk and van de Koppel 1997; Allen-Diaz and Bartolome 1998; Augustine et al. 1998) but these theories are difficult to apply in complex ecosystems. Friedel (1991) defined a threshold as a boundary between two vegetation states where shifts are irreversible without forced intervention. She stressed the importance of thresholds in grazing management.

The recent shifts in vegetation and increases in livestock populations on the typical steppe demonstrate a lack in understanding of grazing management and an urgent need to determine appropriate grazing regimes (National Research Counsel 1992). For proper grazing management, it is necessary to develop a quantitative and inclusive understanding of abiotic controlling factors and their interactions with biotic components at large spatial and temporal scales (Pickup et al. 1994). An effective way to understand the combination of these factors is to construct an explanatory and predictive ecosystem model and then use it to conduct analyses and syntheses, and to make predictions (Agren

et al. 1991; Burke et al. 1991; Riebsame et al. 1994; Hall and Day 1997). The model can be designed to examine both theoretical and experimental questions at a scale conducive to both. It can be used to understand interactions between abiotic and biotic factors to determine which grazing practices and intensities of use are sustainable and do not lead to degradation (Wiens 1984; Caughley et al. 1987; Ellis et al. 1993).

Many models have been used successfully to study grasslands (Hunt et al. 1991; Coffin et al. 1996; Gao et al. 1996; Parton et al. 1996; Pickup 1996; Coughenour and Chen 1997). While most models integrate vegetation, soils, and climate, few consider ungulate dynamics and ecosystem responses to grazing. The SAVANNA model (Coughenour 1993) was developed to simulate both vegetation and ungulate dynamics based on climate, topography, soils, nutrients, vegetation, and grazing patterns. SAVANNA is a spatial and dynamic model that represents ecosystem processes at both local and regional scales. Ecosystem processes are modeled by using data on soils, topography, climate, and livestock densities to affect vegetation production and cover, livestock forage utilization, livestock production, and human offtake. SAVANNA provides a realistic representation of ecosystem processes in that it represents flows of biomass, nitrogen, and organisms, and subsequent rates of change in system states. Its one-week time step enables large-scale spatial and long-term temporal simulations. These dynamic capabilities allow it to simulate interactive responses to grazing in order to understand the sustainability and resilience of semi-arid grasslands.

The objective of this study was to use SAVANNA to locate thresholds and causes of shifts among stable states and determine the implications this has for sustainability and resilience of the typical steppe grassland. Multiple simulations were conducted to

analyze the sustainability of herbaceous production under different livestock densities. Output from each run was used to initialize a second set of model runs to examine the system's ability to recover from different grazing densities. The SAVANNA model simultaneously examined different ecosystem responses, such as biomass, above ground net primary production (ANPP), and root biomass dynamics, to various levels of grazing intensity.

Methods

Study area

This study was focused on the Baiinxile Livestock Farm, on the typical steppe grasslands of Inner Mongolia, China (43.5° north latitude and 116.5° east longitude). The typical steppe is defined as a bunch-grass steppe with many or few forbs, in a semi-arid climate (Lavrenko and Karamysheva 1993; Zhu 1993). This landscape consists of large rolling hills, with elevation ranging from 963 m to 1569 m. Long term mean annual precipitation is 360 mm but varies between 180 mm and 500 mm. Seventy percent of total annual precipitation occurs from May through August (Xiao et al. 1997) from the summer monsoon coming from the east.

Mean annual temperature is -0.4° C but ranges from –27.0° C to 28.6° C (Li 1989; Xiao et al. 1997). Temperatures rise in March, but high winds and minimal precipitation from March to May postpone vegetation growth until the rainy season (Zhu 1993). Warm air currents from the Pacific Ocean give this region its warm-wet climate during the growing season. Because the majority of precipitation occurs during the growing season, biomass production is high as compared to other regions with similar amounts of

rainfall but with smaller proportions received during the growing season (Ellis and Galvin 1994).

The three main soil types in this area are kastanozems, chernozems, and eolian sandy soils (FAO 1991). Dominant grasses include *Stipa grandis* and *Leymus chinensis*, which all have a C3 photosynthetic pathway (Li 1978). Grass height ranges from 15 cm to 60 cm depending on species and growth conditions. Dominant forbs include *Artemisia frigida* and *Potentilla sp.*, while the primary shrubs are *Caragana microphylla* and *Astragalus melilotoides* (Li 1978).

The Baiinxile Livestock Farm's main purpose is wool production from sheep, although horses, cows, and on rare occasion camels graze in the area and are used for market purposes. The farm is 3,680 km^2, divided into an administrative village and 12 branches (Fig. 3.2). A branch is an administrative division of land, ranging in size from 79 km^2 to 655 km^2. Livestock have been supported in this region for a minimum of a thousand years (Xu et al. 1993). In the past, when both livestock and human populations were small, herders were nomadic, moving their livestock in response to seasonal vegetation dynamics and community traditions (Ellis 1992; Fernandez-Gimenez 1993; Williams 1996a, b). Management has since changed with the influence of the Chinese government, increased human population, and increased cultivation (Yenhu 1996). Herders now lead a sedentary lifestyle, and instead of moving livestock from area to area, grazing occurs in smaller areas with reserve pastures set aside for winter fodder (personal communication with local herders 1997; Thwaites et al. 1995; Yenhu 1996). These changes are thought to be the main cause of degradation in East Asia (Sneath 1998).

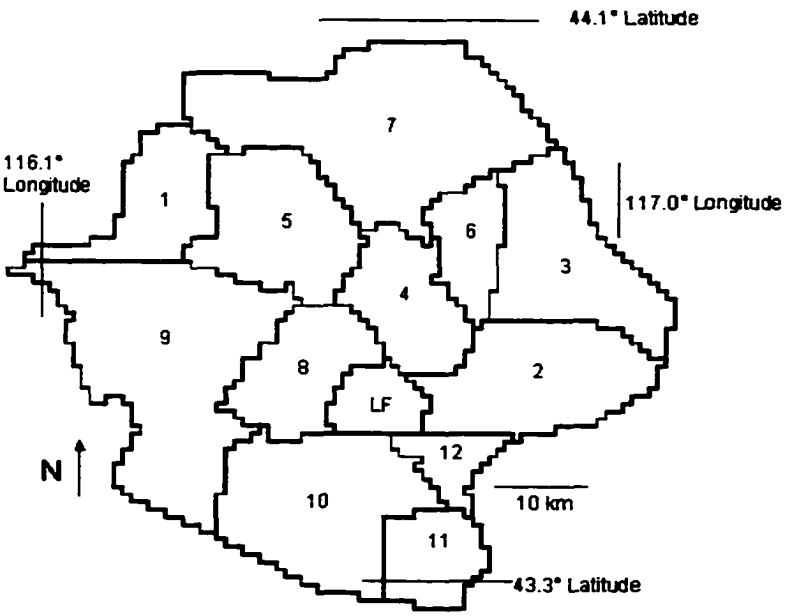

Figure 3.2. Map of Baiinxile Farm with layout of 12 branches and livestock farm (LF) headquarters.

Model description

SAVANNA (Coughenour 1993) is a spatially explicit model that simulates ecosystem processes using data on soils, topography, climate, and herbivore densities to affect vegetation production and cover, forage utilization, and herbivore population and production (Fig. 3.3). It has successfully been parameterized, validated, and used to simulate vegetation and ungulate production in a wide variety of ecosystems such as Rocky Mountain National Park, Colorado (Weisberg 2000); Yellowstone National Park, Wyoming (Coughenour and Singer 1996a; Coughenour and Singer 1996b); Kajiado District, Kenya (Boone et al. 2001); Ngorongoro Conservation Area, Tanzania (Boone et al. in revision); and Kruger National Park, South Africa (Kiker 1998).

SAVANNA has a three-scaled hierarchical spatial structure. The largest scale is the landscape level with a resolution of one grid-cell. Grid-cells are typically 100-1000 m wide. Spatial analyses and model output occur at this scale. The second scale is the subarea, which is a proportion of the grid-cell in a particular topographic position (run-on or run-off areas). Each sub-area is composed of patches, or vegetation types, dominated by herbaceous plants, shrubs, or trees. Plant growth, competition, herbivory offtake, and soil water budgets are modeled at this scale. This information is scaled to the grid-cell by multiplying results by the proportions of grid-cell covered by each patch type.

The model uses monthly precipitation (mm) and average maximum and minimum temperatures (°C). Data from multiple weather stations are interpolated to derive weather patterns over the gridded area. Stochastic weather is generated by randomly picking years from weather files and adding stochastic variation. The model also uses mean monthly values of daily global short-wave radiation, relative humidity, and wind speed.

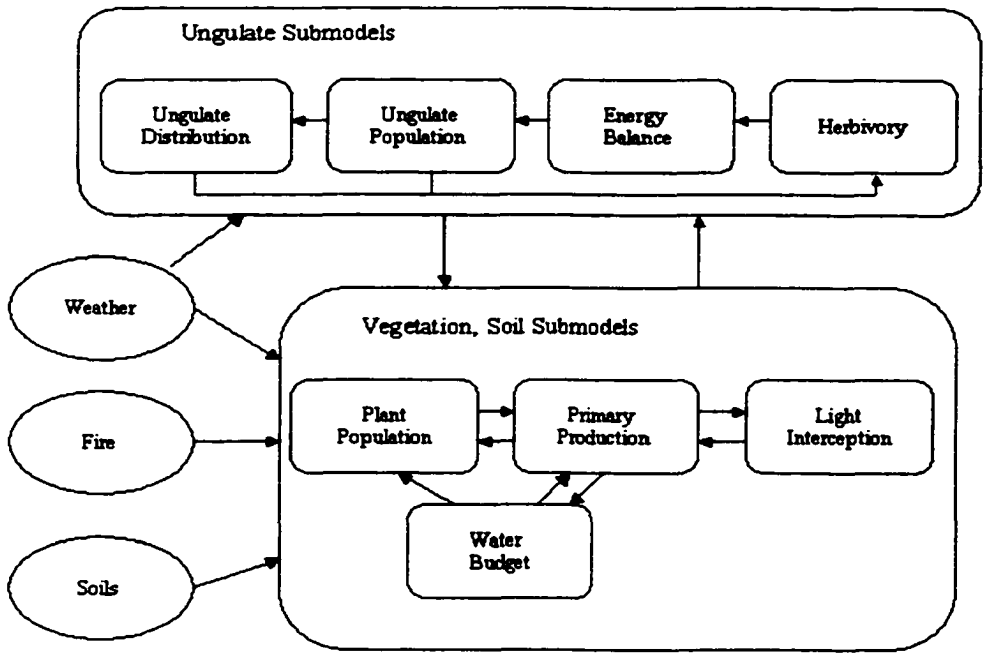

Figure 3.3. SAVANNA model structure (Coughenour 1993).

Precipitation flows into soil layers based on soil depth and soil water holding capacity using a tipping bucket model. Water is drained through soil layers based on field capacity. SAVANNA uses a soil profile of three layers. Bare soil evaporation is taken from the top layer. Herbaceous plants use water in the top 2 layers, and woody plants use water in all 3 layers. Water removal depends on potential evapotranspiration (PET) and plant uptake. Available water to plants is soil water at field capacity minus that at wilting point.

Net primary production (NPP) is simulated as the outcome of light interception, transpiration, nitrogen uptake, leaf growth, and biomass allocation. The light interception submodel uses radiation data to determine mean incident light in the tree, shrub, and herbaceous plant canopy layers. Light penetration is modeled both between plant canopy layers and within canopy layers using Beer's Law.

Simulated radiation on each canopy layer is used in combination with temperature, available nitrogen, and the ratio of available water to potential evapotranspiration (AWT:PET) to affect maximum potential photosynthetic rates of each plant functional group. The effect of light (ANETPAR) is computed from:

$$anetpar = alpha * parx / sqrt(1. + (alpha * parx / pnetmx)**2) \quad (1.)$$

where ALPHA is the initial slope of the light response curve (m^2 s $mmol^{-1}$), PARX is photosynthetically active radiation (mmol m^{-2} s^{-1}), and PNETMX is the maximum photosynthetic rate (umol CO_2 m^{-2} s^{-1}). ANETPAR is divided by maximum photosynthesis rate (PNETMX) to assess the effect of light (EFLP):

$$eflp = anetpar / pnetmx. \quad (2.)$$

Other environmental factors are then introduced to determine the final photosynthetic rate:

$$anet = pnetmx * amin(efwp, eflp) * eftp * efco2 * amin(efnp, efsrp) \quad 3.$$

where the minimum value of the effect of soil water (EFWP) and light (EFLP) is multiplied by the effect of temperature (EFTP), the effect of CO_2 (EFCO2), and the minimum value of the effect of nitrogen (EFNP) and the effect of shoot/root ratio (EFSRP).

Temperature has a curvilinear effect on photosynthesis. Maximum, minimum, and optimum temperatures are specified for each plant functional group. Photosynthesis is maximal at the optimum temperature, and decreases as temperature moves towards minimum and maximum values. Respiration is represented as a Q_{10} factor of temperature (Q_{10} value of 2 is used); therefore an increase in temperature will result in an increase in respiration. Respiration is proportional to the amount of nitrogen in the plant (Ryan 1991). The potential photosynthetic rates may be reduced by competition for water and nitrogen. When total demands for water or N exceed supply, the available resource is allocated among plant functional groups in proportion to demands.

Stomatal conductance is calculated as:

$$cs = csfixa + csfixb\ (anet * relh / CO2\ ppm) \quad 4.$$

which is derived from the Ball and Berry equation (Ball 1988) where CSFIXA is the intercept, which specifies a minimum stomatal conductance. CSFIXB is the slope of the line, which differs markedly between plant species having C3 or C4 photosynthesis. RELH is the relative humidity and CO_2 ppm is the concentration of CO_2 in the atmosphere. Atmospheric and stomatal responses are then used in the Penman-Monteith

equation (Penman 1953; Monteith 1965) along with radiation, humidity, vapor pressure deficit to calculate transpiration (T):

$$T = [slope * (radnet + rho * cp * vpd/rab)]/[lambda * (slope + gamma * (1. + (rs/rab))] \quad 5.$$

where SLOPE is slope of the saturated vapor pressure versus temperature (kPa $°C^{-1}$), RADNET is net ratiation (J m^{-2} s^{-1}), RHO is the density of air (kg m^{-3}), CP is the heat capacity of air (J kg^{-1} deg^{-1}), VPD is the vapor pressure deficit (kPa), RAB is the atmospheric resistance (s m^{-1}), LAMBDA is the latent heat (J g^{-1}), GAMMA is the phychrometer constsant (kPa $°C^{-1}$), RS is stomatal resistance (s m^{-1}) and RAB is the atmospheric resistance (s m^{-1}). Actual transpiration rates are subsequently limited by total available water and competition from other plant functional groups. Water use efficiency (g m^{-2} mmH_2O^{-1}) is then calculated as:

$$wue = anet / transp. \quad 6.$$

If water supply (AVLWATER) is less than transpiration demand, then actual water uptake equals the potential. Net primary production (NPP) is then determined by:

$$NPP = avlwater * wue. \quad 7.$$

Plants are modeled at the functional group level. Net primary production of herbs, shrubs, and trees is allocated to root and shoot growth. Growth initiation is affected by water and temperature, and temperature affects the phenological stage of the plant. Herbs, shrubs, and trees compete for water and nitrogen as well as light. When total soil resources cannot meet total demand, the resource is allocated among competitors based on relative demands, and rooting distributions.

The death rates of leaves, stems, and roots determine flows from live plants to litter. Shoot tissues are transferred to "standing dead" and then "litter". Root tissues are transferred to belowground litter according to the root turnover rate. Decomposition of litter is affected by available water and temperature. As available water increases, death rates decrease, while an increase in temperature causes an increase in decomposition rates. Litter is further partitioned into metabolic and structural components, in response to the nitrogen to lignin ratio of the plant tissues. Decomposed structural and metabolic material flows into the labile and recalcitrant soil organic matter pools, and is then decomposed based on temperature, water, and nitrogen to carbon ratios.

Simulated forage biomass and nutrient content are used to predict forage offtake and ungulate distribution across the landscape. Forage intake then affects ungulate energy balance, which subsequently affects ungulates population dynamics, which is modeled separately for different types of livestock. Forage offtake is affected by forage quantity and availability, ungulate diet composition and preference, maximal species intake rates and stocking rates. Ungulate weight dynamics are derived by modeling herbivore energy intake and expenditure, and resultant changes in body mass. An animal condition index is then calculated, and can be used to affect ungulate population dynamics. Condition index is a number between 0.0 and 1.0, defined as:

$$cond = (cwtkg - bwmn) / (bwmx - bwmn) \qquad 8.$$

where CWTKG is the simulated body weight of the animal, BWMN is the minimum body weight, and BWMX is the maximum body weight. Ungulate numbers may or may not be simulated. For this analysis sheep, horses, and cattle per km^2 were entered as model input.

Although initial input is mean animal number per km^2, SAVANNA redistributes animals spatially, based on available vegetation and other factors. Therefore, some areas may have more than the specified mean animal density while other areas have less. SAVANNA distributes ungulate populations based on a habitat suitability index (HSI) locations of suitable forage and management practices. The HSI is affected by available forage, water, and tree cover, and is defined as:

$$HSI = pforage(tforage) * phys * prefar * pwatr \quad 9.$$

where HSI is the initial function value, PFORAGE(TFORAGE) is a 0-1 suitability index for forage abundance, phys is a 0-1 suitability index of the physical habitat, PREFAR is an index for a preferred area, and PWATR is a 0-1 suitability index for water. HSI is then normalized over all grid cells to determine animal distribution. The fraction of animals in a grid-cell 'i' is:

$$f_i = HSI / \sum HSI. \quad 10.$$

Herbivory in return impacts vegetation by increasing plant mortality. Grazing increases physical damage to plants due to trampling or uprooting, which causes plants to die. Therefore animal density is related to plant mortality, which is defined as:

$$Hkill = Pkill * Hdens * 0.0192 \quad 11.$$

where HKILL is the fraction of herbaceous or shrubby plants killed per week, PKILL is the fraction per animal per km^2 per year, and HDENS is animal density. Herbivory affects plant growth by removing plant tissues, reducing transfers to litter, and reducing nitrogen retranslocation. Leaves are unable to translocate nitrogen to the plant before senescence and are removed by herbivores and recycled as urine and fecal matter. Overall, herbivory reduces the amount of carbon relative to nitrogen in the decomposer

pathways, which enhances net nitrogen mineralization and thus plant nitrogen uptake. A small fraction of plants are assumed to die due to trampling or uprooting.

Management strategies are incorporated into the model by limiting and restricting livestock movements with forcing and boundary maps created in a geographic information system (GIS). This model therefore has the ability to simulate ecosystem responses to alternative management practices and corresponding ungulate population densities.

SAVANNA's structure includes rasterized GIS maps of vegetation, soil, digital elevation model (DEM), slope, and aspect of the Baiinxile farm. Spatial analyses and model output are provided at the scale designated by the grid-cell size. A 1 km^2 grid-cell size was used for this analysis, which was an appropriate scale for model validation, theoretical questions, and to address local concerns about livestock management.

Model parameterization/verification

Fieldwork was conducted in Inner Mongolia from June through September of 1997 to obtain data needed to parameterize the vegetation and soil portions of the model (see Chap. 2). Vegetation biomass and percent cover data from lightly grazed plots were used in SAVANNA initialization parameter files to describe the vegetation at the Baiinxile farm. The purpose was to use the model to simulate vegetation in a non-disturbed state. Both percent cover and vegetation biomass of each functional group were used in model initialization files. Vegetation data from heavily grazed plots were used to compare with SAVANNA output data. Other vegetation information required by SAVANNA parameter files, such as plant physiological characteristics, was taken from Yang (1987). The base

vegetation map of the model was derived from Landsat TM imagery. The original TM image was classified into 56 vegetation types by Xiao et al. (1997) and then geo-rectified in Arc/Info (ESRI 1997). These vegetation types were condensed into 9 vegetation types, which was an appropriate resolution for a 1 km^2 model analysis (Table 3.1). The TM image was then rescaled from 30 m^2 to 1 km^2 in Arc/Info (ESRI 1997).

Field soil texture data from soil samples was used to parameterize soil water holding capacity and water infiltration rates. A digitized soil map of China (scale 1:4 million) was used as the soil base vegetation map.

Ungulates used in model simulations were sheep, horses, and cattle, the primary grazers at the livestock farm. Information needed for model parameters such as livestock height, weight, and diet were gathered from livestock data specific to the typical steppe (Yang 1987). On the typical steppe, areas of grassland are reserved for hay cutting and winter grazing. To reproduce this spatial effect in model simulations, force maps were used. Force maps block specified areas of vegetation during summer months to exclude grazing, and are then removed during winter months so vegetation is available. Vegetation cells classified as meadow steppe were blocked from grazing during April through November to simulate vegetation areas reserved for fodder growth during the growing season. Grid cells classified as meadow steppe were chosen because these areas were more productive, higher in elevation, and further away from water holes, therefore reserved for fodder growth (personal communication with local herders). Areas classified as sand dunes were reserved for winter grazing because of the shelter they provided against winter climate events.

Table 3.1. Vegetation categories used in SAVANNA simulation runs, simplified from Xiao (1997).

Vegetation classification	
1	meadow steppe
2	typical steppe
3	Artemisia frigida
4	desertificated land
5	saline alkaline land
6	wetland
7	cropland
8	fallow cropland
9	sand dunes with Ulmus sp.

Climate data from the IMGERS station, centrally located in the Baiinxile Farm, and six surrounding weather stations were used in SAVANNA simulations. Maximum and minimum temperatures, precipitation, wind speed, humidity, and radiation data collected at the IMGERS station were used in model analyses. Data were available for 19 years. Random years were chosen from the original 19 to create a weather data set for 50 years. Precipitation data from six surrounding weather stations were used in the weather interpolation submodel of SAVANNA.

Verification analyses were conducted to help parameterize the model to the typical steppe (Jorgensen 1994). Xiao et al. (1996) conducted a long-term field experiment where vegetation biomass data were collected from 1980 to 1989 in an enclosed plot at the Baiinxile Farm. Biomass was sampled at the end of each month during the growing season, May through September. Model outputs from the two closest branches to the study site, branches 10 and 11, were compared to these field data in verification analysis.

Model validation

Model validations compared satellite imagery, not used in model parameterization, to final model output to determine model simulation accuracy (Jorgensen 1994). Eleven years, 1982-1992, of Normalized Difference Vegetation Index (NDVI) bimonthly composites derived from satellite imagery (Tucker 1979, Tucker et al. 1983, Justice et al. 1985, Tucker et al. 1985, Justice and Hiernaux 1986, Malingreau 1986) were used to validate SAVANNA simulation output (see Chap. 2). NDVI data was analyzed in Idrisi (IDRISI 1997), a GIS software package. NDVI data were extracted

and averaged over all grid-cells in each image and each branch. These trends were plotted and compared to SAVANNA green vegetation biomass output, which was also averaged over cells for each branch. SAVANNA green vegetation biomass output included summation of green leaf mass of herbaceous, shrubs, and trees and green herbaceous stem biomass. The eleven-year cyclic trend in NDVI was used to compare against model output instead of converting NDVI values to biomass values for comparison. This conversion has been done in some studies so that biomass values from satellite imagery and biomass values from model output could be compared (Tucker 1979; Tucker et al. 1983; Tucker and Sellers 1986; Box et al. 1989; Todd et al. 1998). But errors exist in time series AVHRR data due to satellite orbital drift and volcanic aerosols (Asner accepted; Malmstrom 1997; Privettet 1995). This results in added error in NDVI conversions to biomass values (Asner accepted). Due to these complications, NDVI conversions were not used for this analysis.

NDVI data are from a region that is grazed, but there is difficulty in removing the effect of grazing in satellite imagery; therefore it was necessary to reflect grazing activity in the SAVANNA model runs for an accurate comparison. A light grazing treatment of 14.65 animal units (AU) (Holecheck et al. 1998) per km^2 was included in simulation runs for the validation analysis to reflect realistic grazing activity on the grasslands.

Model experiments

The first goal of this analysis was to determine through model experiments, those grazing densities which were sustainable on the typical steppe grassland. This was accomplished by modeling the effects of different livestock densities on vegetation. A

density analysis is only one of many different types of management that could be tested, but was chosen based on the scale and type of this analysis. Fourteen different livestock densities were chosen for model runs, ranging from extremely low to extremely high densities (Table 3.2). Proportion of animals in each livestock species group were based on typical herd sizes in that region. Therefore there were proportionally more sheep than cattle, and very few horses. Population numbers of sheep, horses, and cattle were converted into AU km^{-2} for analysis (Table 3.2). Each model run was 50 years in length.

To examine the resilience of the system, animal populations were removed from model runs to simulate grasslands in recovery from grazing. State variable outputs from each density run were saved and used to initialize a set of "resilience" runs. Grazing intensity was set at zero for each of the 14 resilience experiments and vegetation production was modeled for 50 years following removal of grazing. Fifty years or more are considered adequate time for a system to recover from overgrazing (Burke et al. 1995). The same weather file was used in this analysis as the previous set of runs.

Model output of herbaceous and shrub above ground net primary production (ANPP); grass, forb, and shrub live roots; and percent vegetation offtake from each stocking density were used in grazing intensity analyses to examine the sustainability and resilience of the typical steppe grassland system. Temporal output from model runs was examined to determine the effect of grazing density on the simulated system. Density runs, conducted to examine the sustainability of the system, were analyzed separately from model runs examining the resilience of the system.

Table 3.2. Livestock composition of each density and its total animal unit (AU) equivalent (Holechek et al. 1998) used in model experiments.

	sheep	horse	cow	au
dens 1	5	1	3	5.55
dens 2	7	2	4	8.65
dens 3	10	2	6	11.10
dens 4	15	3	7	14.65
dens 5	20	4	8	18.20
dens 6	23	5	12	24.45
dens 7	25	6	15	29.55
dens 8	27	7	17	33.65
dens 9	30	8	20	38.90
dens 10	33	8	22	41.35
dens 11	35	8	25	44.65
dens 12	40	8	30	50.40
dens 13	43	9	32	54.65
dens 14	45	10	35	59.75

Results

Verification analysis

SAVANNA simulations of green vegetation biomass from branches 10 and 11 followed a trend close to biomass data collected in model verification analyses (R^2 = 0.40, $P < 0.0001$; $R^2 = 0.42$, $P < 0.0001$ respectively) (Fig. 3.4). Plots of simulated biomass versus field biomass samples show SAVANNA correctly simulated peaks in seasonal trends 60% of the time while 20% of the time the model predicted greater biomass than observed (years 87 and 88) and 20% of the time it predicted less biomass than observed (years 84 and 89). The major difficulty with model predictions was in timing of vegetation green up and leaf senescence. SAVANNA predicted an earlier green up and senescence than field biomass data showed. Beginning 1985, SAVANNA's peak in green vegetation biomass was one month earlier than field data. When peaks in simulated biomass were shifted one month later (starting in year 1985) to match biomass plots, the correlation between observed and simulated biomass data increased ($R^2 = 0.65$, $P < 0.0001$, and $R^2 = 0.63$, $P < 0.0001$ for branches 10 and 11 respectively).

Validation analysis

Monthly NDVI images were compared to SAVANNA green vegetation biomass output for the growing season, May through October, to validate vegetation biomass model predictions. This NDVI data set was not used in model calibration so a true validation comparison could be made to final model output. For analyses, NDVI images and SAVANNA output were averaged for each branch and plotted over time (Fig. 3.5). Accuracy in SAVANNA's predictions of green vegetation biomass varied depending on

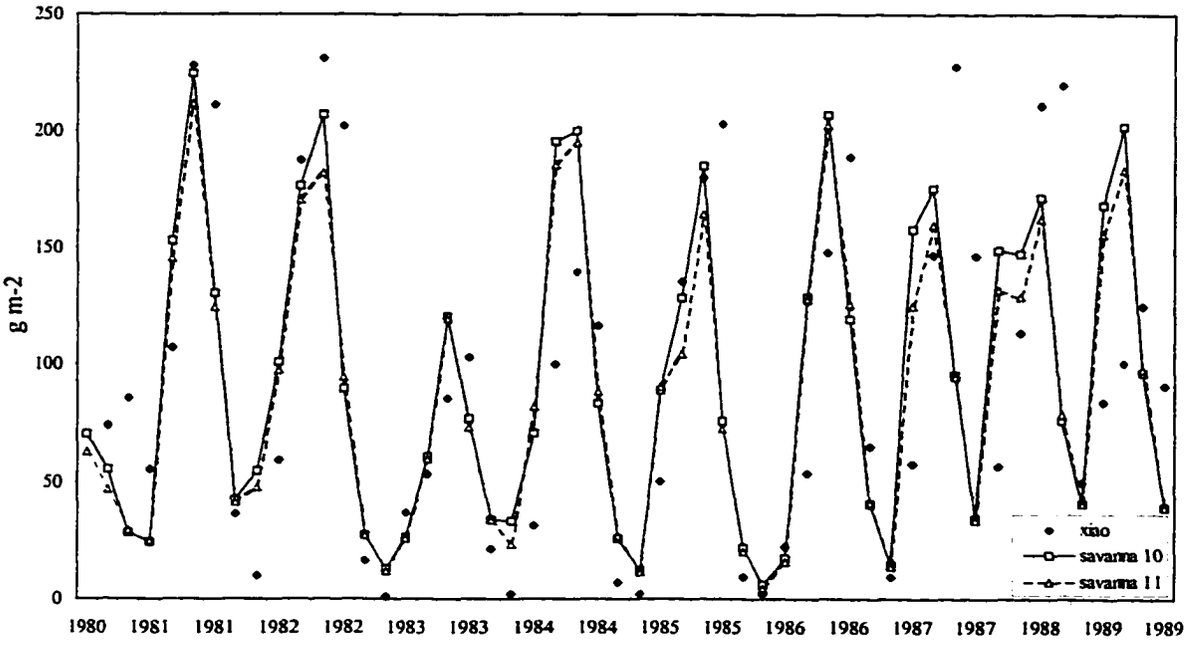

Figure 3.4. Verification output of field biomass data collected during the growing season (May – October), years 1980-1989, by Xiao et al. (1997) compared to SAVANNA output from branches 10 and 11. Field and simulated data includes a light grazing treatment.

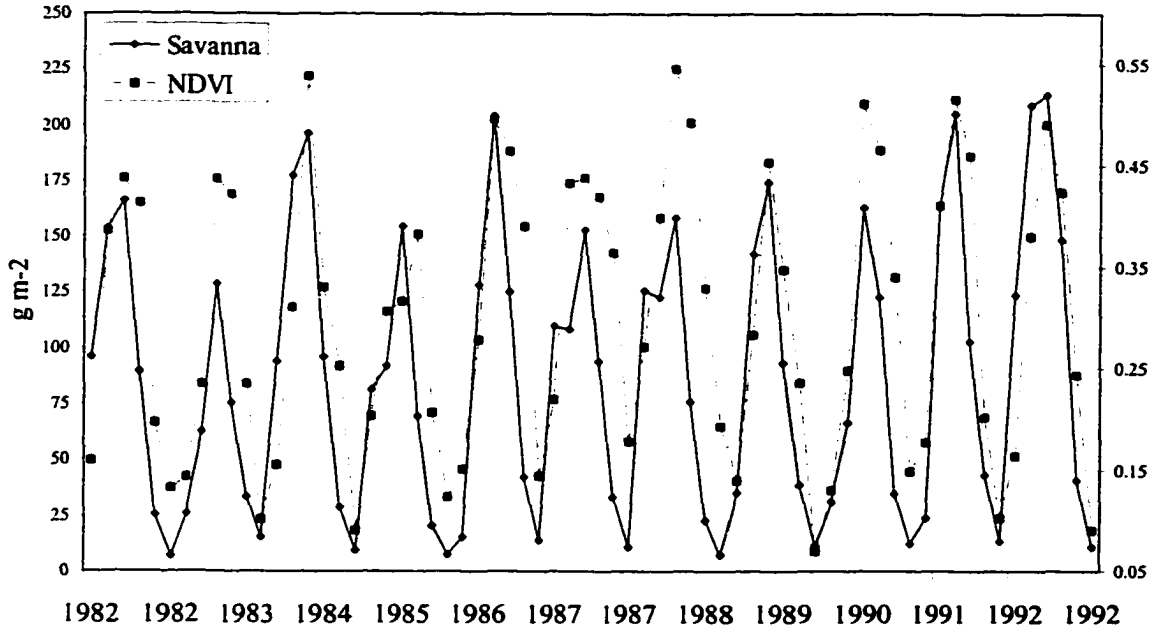

Figure 3.5. Validation data of SAVANNA output from branch 3. Continuous line represents biomass data (g m^{-2}) from SAVANNA, dotted line represents NDVI data, rescaled to the interval –1 to 1. Values from May to October are shown on graph.

branch. Correlations between predictions and NDVI data ranged from $R^2 = 0.34$ ($P = 0.00021$) at branch 9 to $R^2 = 0.57$ ($P < 0.0001$) at branch 3 (Table 3.3). Contrary to verification analysis, simulated biomass in verification analysis more accurately predicted green-up when compared to NDVI data. However, simulated senescence was sometimes earlier than NDVI data, which can be interpreted as a shortened growing season in model simulations.

Livestock intensity analysis

Grazing intensities were used to describe effects of animal units on vegetation. McNaughton (1979) described grazing intensity as:

$$GI = 1 - (g/u) \qquad 13.$$

where g = biomass in grazed area and u = biomass in ungrazed areas. Grazing intensity provided a more accurate definition of the relationship between animal numbers and vegetation than livestock density in SAVANNA simulation runs for the following reasons: 1) per-animal intake rate varies among species, mainly due to body weight differences, and 2) SAVANNA allocates animals among cells in proportion to forage distribution. Therefore, animal population numbers were an average of the entire area being simulated and animal populations for each cell varied greatly from month to month depending on available forage (Fig. 3.6). 3) The grazing intensity scales biomass of the grazed to that of the ungrazed treatment so NPP differences among sites were removed.

Table 3.3. Regression statistics for branches 1-12 comparing simulated biomass from May through October to 11 years of NDVI Global Area Coverage (4 km^2) data (n=65).

Branch	1	2	3	4	5	6	7	8	9	10	11	12
Area (km^2)	196	334	323	212	301	130	655	208	609	412	141	79
Adjusted R^{2***}	0.343	0.532	0.574	0.385	0.410	0.560	0.467	0.397	0.340	0.431	0.429	0.543
Standard Error	0.082	0.090	0.090	0.092	0.086	0.091	0.092	0.081	0.081	0.082	0.090	0.078

*** Significant at the 0.001 level.

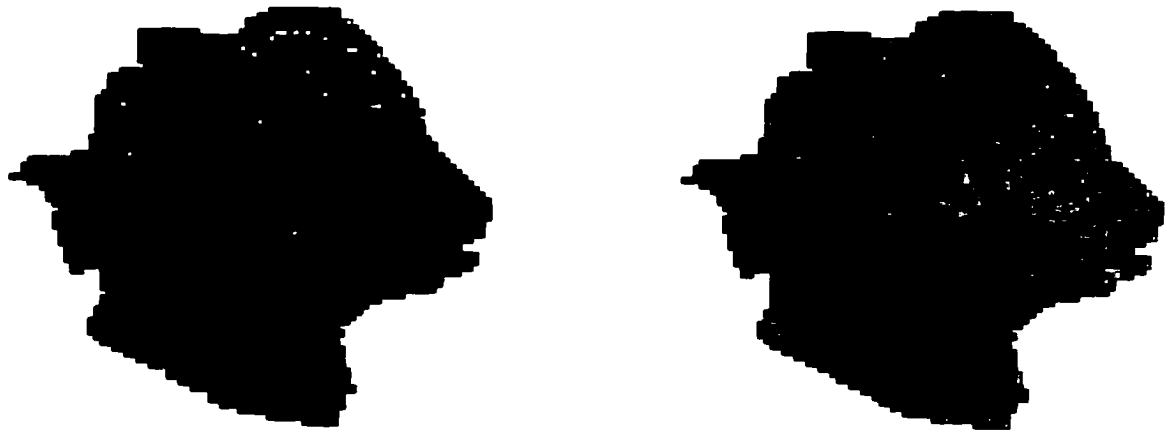

Figure 3.6. Spatial description of livestock distribution at Baiinxile Farm for months of May and June, respectively. Darker values represent areas with lower densities of livestock.

Sustainability

Grazing intensity data was averaged for the growing season May through October for analysis. Mean seasonal grazing intensity increased as grazing density increased, with a minimum intensity of 0.085 with density 1 (5.55 AU km^{-2}) and a maximum intensity of 0.902 with density 14 (59.75 AU km^{-2}) (Fig. 3.7). Each increment in density (averaging 3.2 AU km^{-2}) increased the grazing intensity by 0.035 with densities 1-5, whereas densities 6-9 (averaging 4.8 AU km^{-2}) increased the grazing intensity by 0.084 with each density. Higher grazing densities (10-14, each increasing by an average of 4.6 AU km^{-2}) averaged a 0.065 increase in grazing intensity.

Effects of grazing intensity on annual herbaceous ANPP ($ANPP_h$) varied (Fig. 3.8). $ANPP_h$ decreased slightly as grazing intensity increased from 0.085 to 0.49 (livestock densities 1-9). When $ANPP_h$ dropped below 100 g m^{-2} y^{-1} with grazing intensity 0.49, annual shrub ANPP ($ANPP_s$) increased to values greater than 65 g m^{-2} y^{-1} as compared to 18.5 g m^{-2} y^{-1} at low grazing intensities (Fig. 3.9). This grazing intensity did not suppress herbaceous biomass to the point that shrubs were able to gain a competitive advantage. Grazing intensities 0.085 to 0.23 (densities 1-5) had the least effect on $ANPP_h$ where trends did not differ from $ANPP_h$ with no grazing. Grazing intensities 0.3 to 0.49 (densities 6-9) caused a proportionally higher decrease in $ANPP_h$ than lower grazing intensities, but still followed a similar trend as $ANPP_h$ with no grazing.

A grazing intensity of 0.64 and higher (densities 10-14) was not sustainable and caused a catastrophic decline in $ANPP_h$ subsequently followed by an increase in $ANPP_s$, which can readily be seen when mean $ANPP_h$ is plotted for each grazing intensity (Fig.

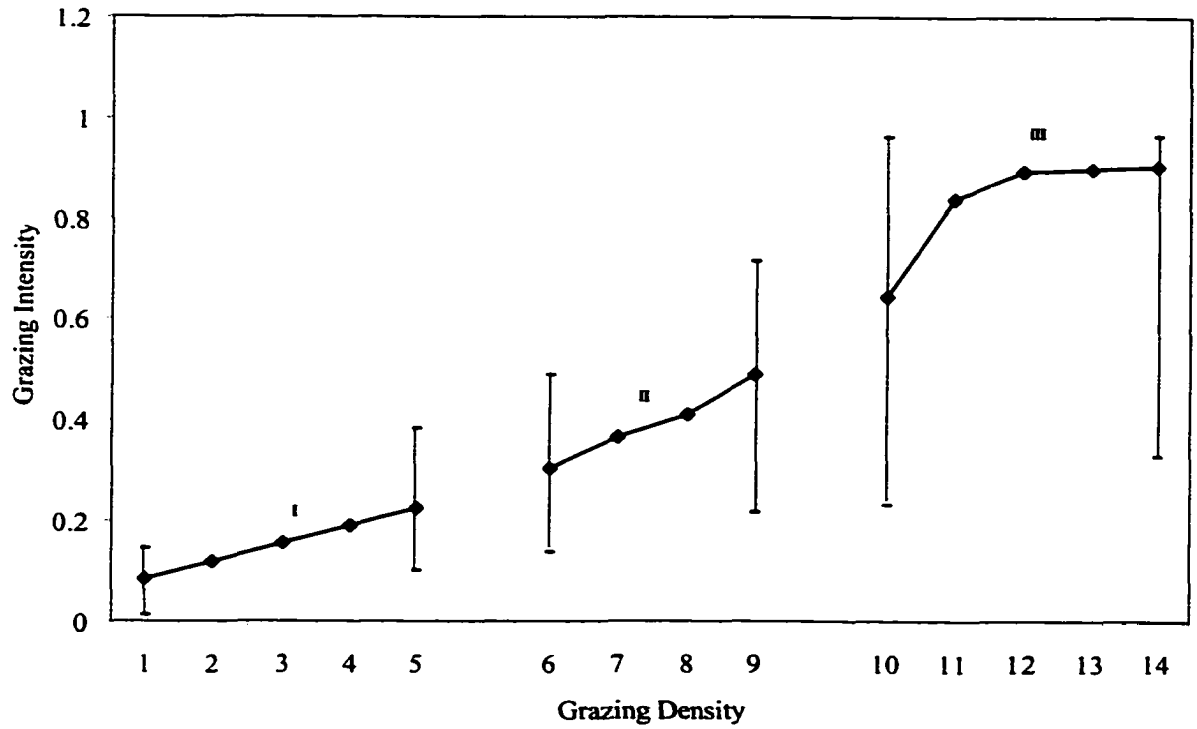

Figure 3.7. Relationship between grazing density (AU km^{-2}) and grazing intensity. Line I represents densities 1-5 (light grazing), line II represents densities 6-9 (moderate grazing), and line III represents densities 10-14 (heavy grazing). Each line has bars representing minimum and maximum intensity values for each density.

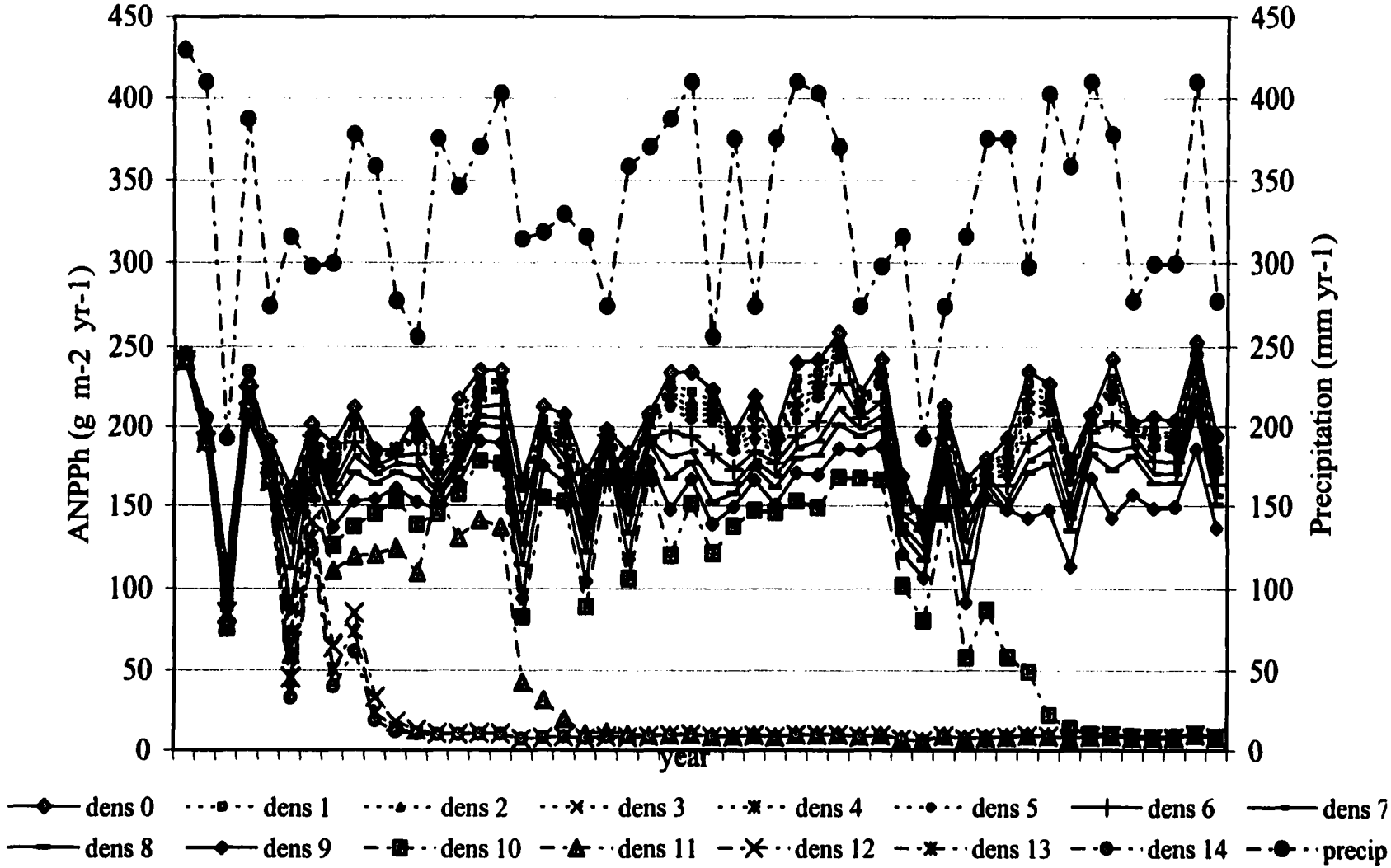

Figure 3.8. Annual herbaceous ANPP for 14 different grazing densities (densities 1-14) and annual precipitation data from 50-year simulation runs.

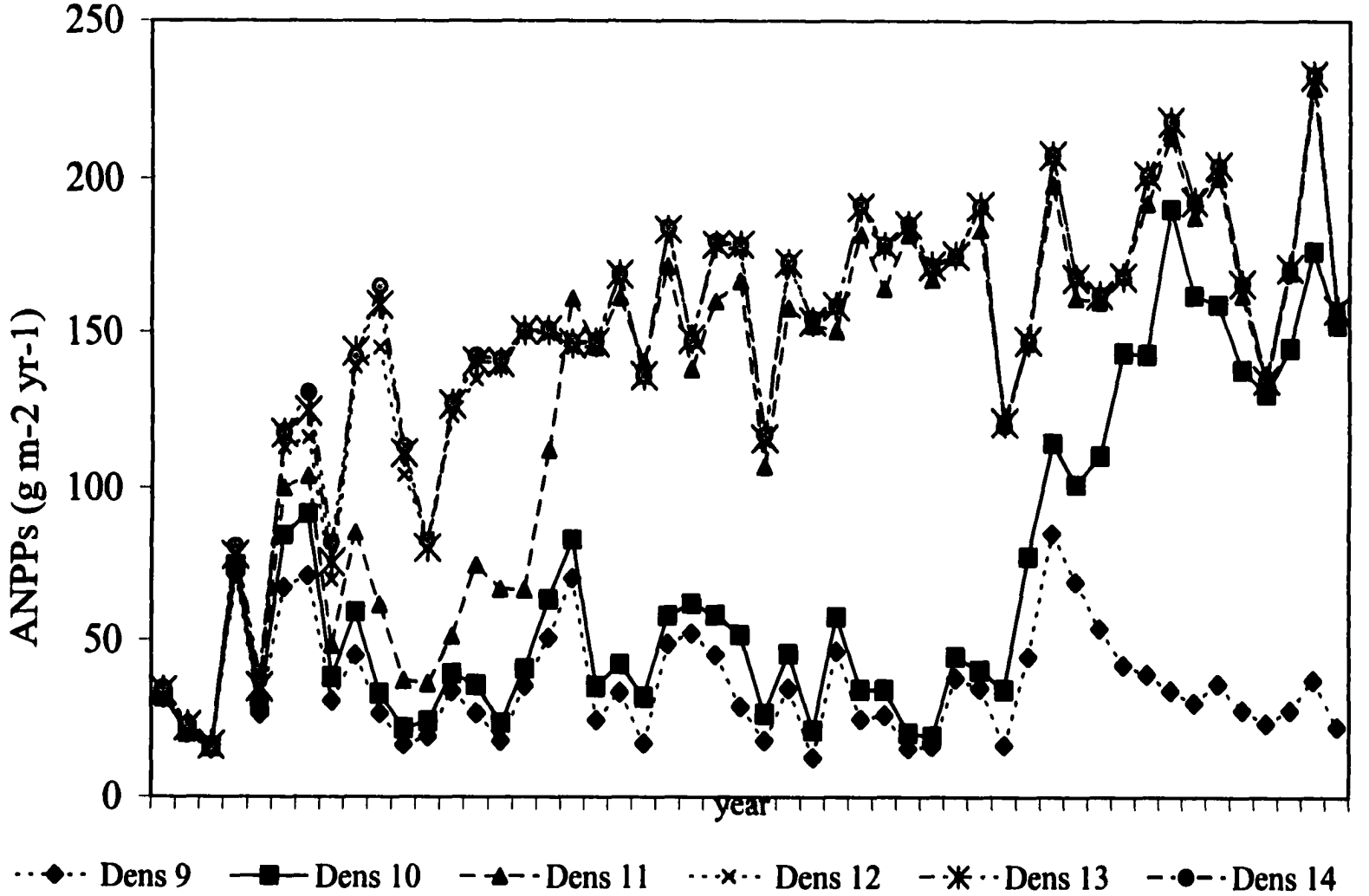

Figure 3.9. Annual shrub ANPP in areas grazed with intensities of 0.49 to 0.9 (densities 9-14) from 50-year simulation runs.

3.10). After 43 years of grazing with intensity 0.64 a threshold was reached and $ANPP_h$ dropped to an average of 10 g m^{-2} y^{-1} (Fig. 3.8). This threshold was reached more rapidly with higher grazing intensities, where $ANPP_h$ was reduced to an average of 9 g m^{-2} y^{-1} after 18 years with intensity 0.84 and after 11 years with intensity 0.89-0.9. The drop in $ANPP_h$ for grazing intensity of 0.64 and higher resulted in an increase in $ANPP_s$ (Fig. 3.9). This switch from grass-dominated vegetation to shrub-dominated vegetation demonstrates a key state change with the new system being unsustainable for livestock. There was a rapid replacement of the herbaceous community by a shrub-dominated one due to shrub's access to water in deep soil layers and lack of browsing on shrub vegetation.

Biomass output data from grazing intensity 0.64 was compared to field data to check model accuracy in a heavy grazing simulation. Herbaceous biomass collected in the field during the month of August 1997 ranged from 26.5 g m^{-2} to 98.1 g m^{-2} in heavily grazed plots, with an average of 63.85 g m^{-2}. These values closely match modeled herbaceous biomass data (45.2 g m^{-2}) from areas grazed with intensity 0.64, during August of the same year. However, field data (see Chap. 2) in the high-density grazed areas did not show the increase in shrubs as modeled output. Possible explanations are that the fieldwork was completed in one season, while the increase in shrubs in the modeling experiment occurred over a 38 year period. Also, livestock numbers have only just increased in the field, whereas the modeling results are a prediction of what could happen after many years of heavy grazing.

Root biomass was averaged from May through October, and then averaged in 10-year increments (Fig. 3.11). Grass root biomass ranged from 328.0 g m^{-2} to 476.0 g m^{-2}

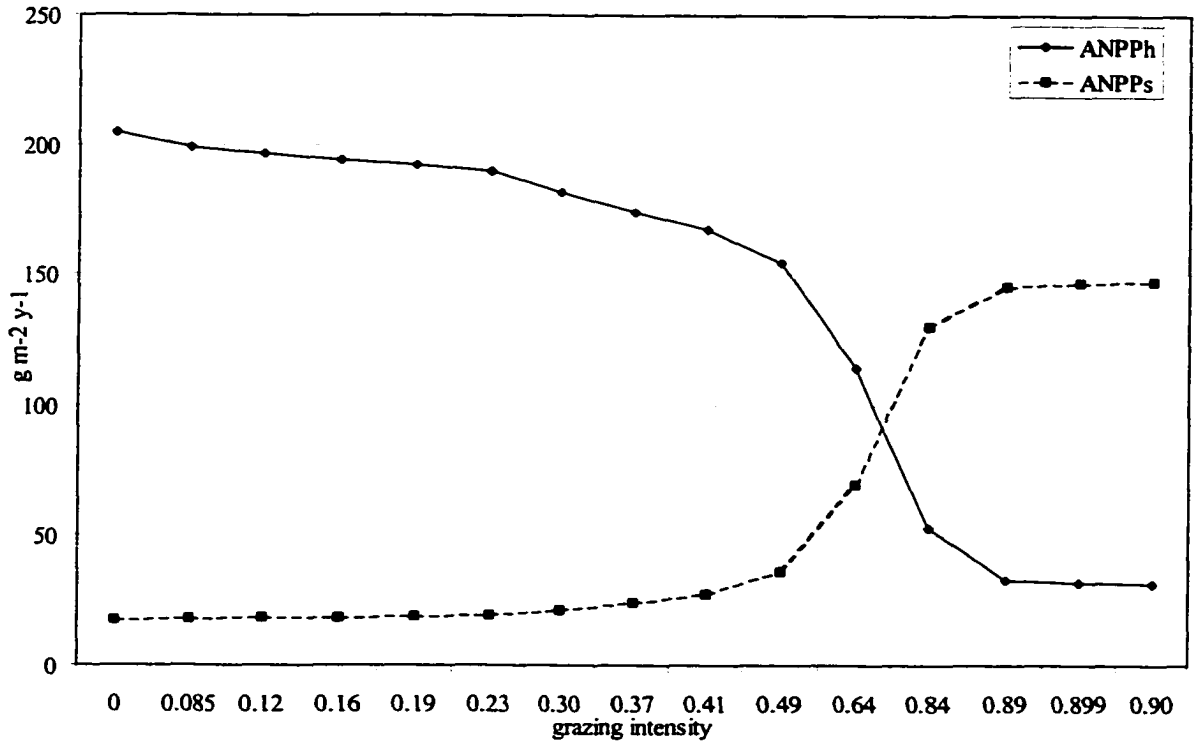

Figure 3.10. Mean annual $ANPP_h$ and $ANPP_s$ averaged over 50-year simulation period. Grazing intensity is defined as 1-g/u; g = biomass in grazed area, u = biomass in ungrazed area.

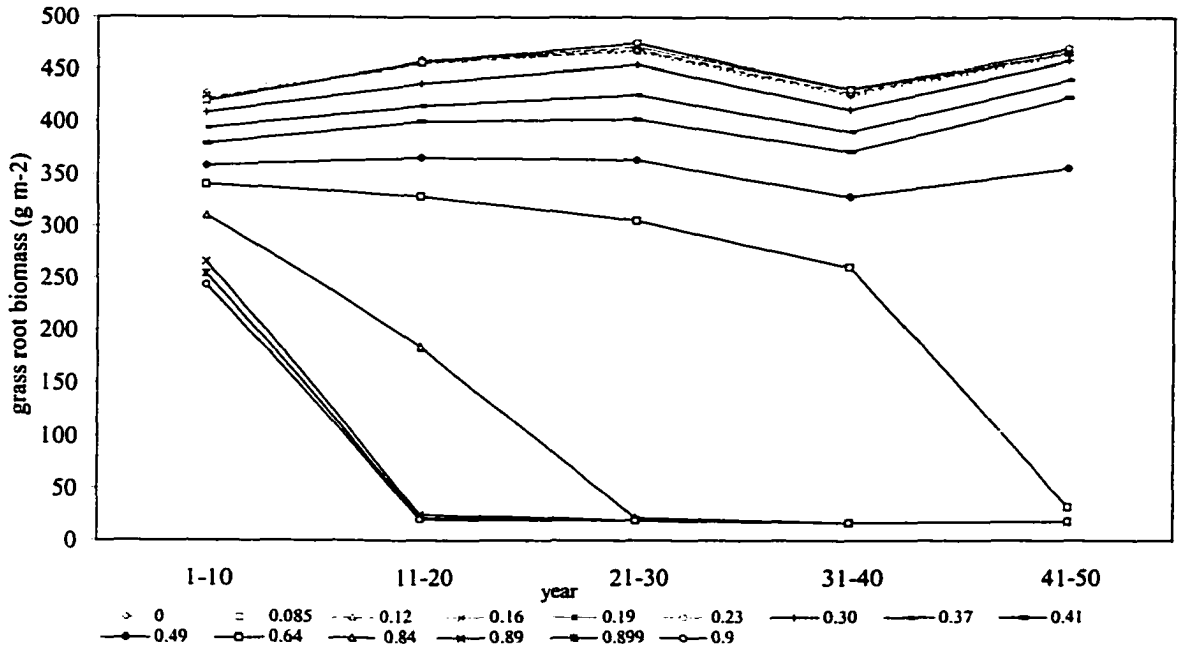

Figure 3.11. Root dynamics from grass vegetation (g m^{-2}) averaged for the growing season May through October, then averaged in 10-year increments. Each line represents different grazing intensities (grazing densities 1-14).

with grazing intensities 0.085 to 0.49, but contrary to ANPP, root biomass increased as grazing intensity increased with grazing intensities 0.085 – 0.23. Root biomass in areas grazed with intensities 0.085 to 0.23 were similar, differing only by an average of 1.4 g m^{-2} with a maximum root density of 476.0 g m^{-2} with intensity 0.23 (density 5). A threshold was reached at this grazing intensity, and higher grazing intensities resulted in decreased root biomass.

Herbaceous root biomass decreased as grazing intensity increased from 0.30 to 0.49 (densities 6-9), averaging a 24 g m^{-2} decrease with each intensity increment of 5 AU km^{-2}. Although root biomass decreased, their dynamics followed the same trend as roots with lighter grazing intensities. A second threshold was reached at grazing intensity 0.49, and higher grazing intensities resulted in a crash of grass root biomass. Grass root biomass dropped to 31.9 g m^{-2} in the last 10 years of simulation with intensity 0.64 (density 10), while grazing intensity of 0.84 (density 11) decreased grass root biomass to 20.9 g m^{-2} after 30 years of simulation. Higher grazing intensities (0.89-0.90, densities 12-14) caused an immediate drop in grass root biomass, which fell to an average of 22 g m^{-2} after 10 years.

Shrub root biomass with grazing intensities 0.085 to 0.49 remained stable, between 12.9 g m^{-2} and 19.8 g m^{-2} (Fig. 3.12). These values are low because within the model they are expressed as g per total area in the steppe. Shrubs cover only 12% of typical steppe vegetation (dominant vegetation) therefore reported shrub biomass is lower than data that is expressed as g m^{-2} within the shrub-covered areas. Shrub root biomass in areas grazed with intensity 0.64 remained below 57.6 g m^{-2} for the first 40 years, then gained the competitive advantage over herbaceous root biomass and increased to 179.8 g

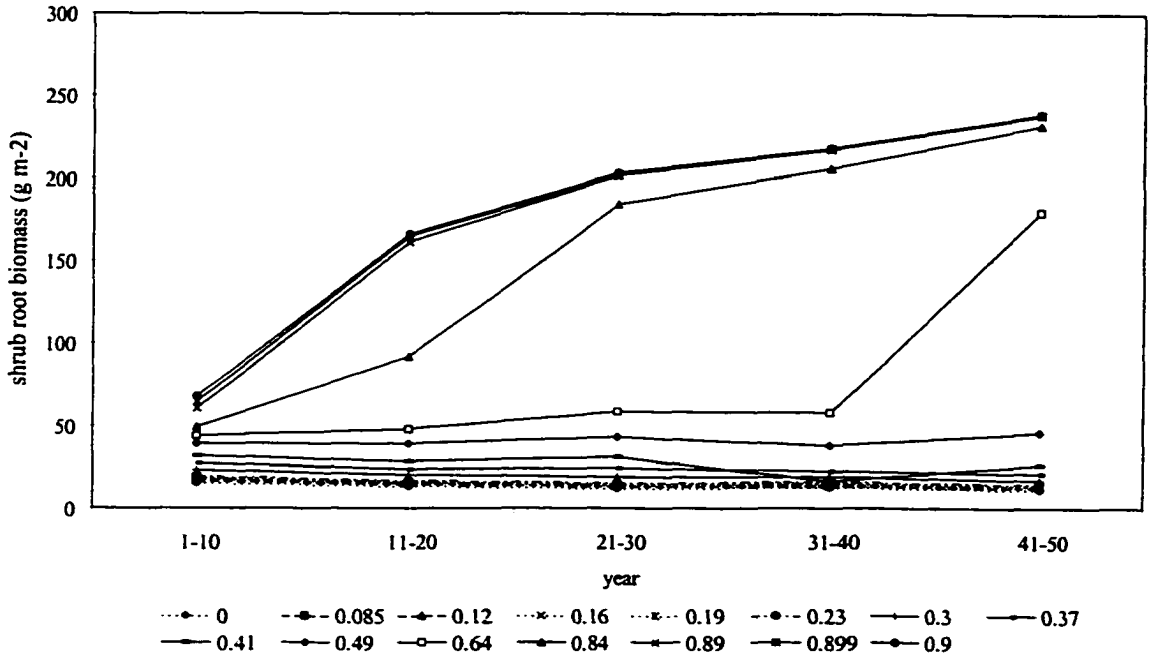

Figure 3.12. Shrub root dynamics from grass vegetation (g m-2) averaged for the growing season May through October, then averaged in 10-year increments. Each line represents different grazing intensities (grazing densities 1-14).

m^{-2}. Shrub root biomass in areas grazed with intensities 0.84 and higher gained a competitive advantage in the first 10 years of simulation and increased to biomass levels of 238.7 g m^{-2}.

Based on results from SAVANNA output, grazing intensities were categorized into three groups: light, moderate, and heavy (Fig. 3.7). A light grazing intensity included grazing intensities of 0.085-0.23 (densities of 6 – 18 AU km^{-2}). Light grazing intensity did not have a large effect on root biomass dynamics as annual average root biomass dynamics differed only slightly from root biomass dynamics with no grazing. Root biomass increased as grazing intensities increased from 0.085 to 0.23. There was a slight decrease in $ANPP_h$ as grazing intensity increased, but $ANPP_h$ followed closely the trend of $ANPP_h$ without grazing. Light grazing intensity did not have a large effect on vegetation dynamics, so there was in effect, no disturbance, and resilience was irrelevant.

Moderate grazing intensities occurred at livestock grazing intensities of 0.0085 to 0.49 (densities of 24 – 39 AU km^{-2}) where the herbaceous system was sustained. Averaged annual root biomass was less than root biomass in the ungrazed areas, and decreased as grazing intensity increased. A threshold in herbaceous production was reached at a moderate grazing intensity, above which the system was both unsustainable and non-resilient.

Heavy grazing included grazing intensities of 0.64 to 0.9 (densities 41 to 60 AU km^{-2}), corresponding to an intensity of 0.64 to 0.9. The decrease in herbaceous biomass allowed shrubs to grow to their maximum size. Once grazing was removed, herbaceous vegetation could not compete with the large shrubs for resources. This system was not resilient at this grazing intensity but experienced a permanent state shift.

The catastrophic drop in $ANPP_h$ and root biomass with heavy grazing was initiated during years with below average precipitation. Average annual precipitation at the Baiinxile farm for these simulations was 334.7 mm. In the 5th year of simulation, annual precipitation dropped to 274.3 mm and $ANPP_h$ began to decrease as grazing intensity increased. The following three years, annual precipitation was below average (315.6 mm, 297.6 mm, and 299.2 mm respectively) and $ANPP_h$ grazed at intensities 0.89 to 0.9 (densities 12-14) dropped to an average of 8.6 g m^{-2} y^{-1}. Subsequently shrub production increased and became dominant. In the 12^{th} year of simulation, $ANPP_h$ at grazing intensity 0.84 (density 11) dropped, but the following year had above average precipitation and $ANPP_h$ recovered. Five years later, there were three consecutive years with below average precipitation and $ANPP_h$ dropped to an average of 9.0 g m^{-2} y^{-1}. The drop in $ANPP_h$ with grazing intensity 0.64 (density 10) occurred after simulation years 36-38, all of which had below average annual precipitation (192.5 mm, 274.3 mm, and 315.6 mm respectively). While herbaceous vegetation was dominant and grazing intensity was light, shrubs remained small. When herbaceous vegetation was removed by heavy grazing, coinciding with low precipitation years, shrubs were able to grow to their maximum size.

Resilience

Results from simulation runs that examined resilience of the system were similar to results from sustainability runs. The $ANPP_h$ in areas previously grazed with intensities 0.085 to 0.49 (densities 1-9) returned to levels similar to runs with no grazing. The $ANPP_h$ associated with a grazing intensity of 0.49 took 46 years to recover to the same

biomass levels as with no grazing, but the reduction in $ANPP_h$ was small, therefore the system was resilient with this grazing intensity. The $ANPP_h$ that was grazed with intensities 0.64 to 0.9 was unable to recover, and $ANPP_s$ remained high. Herbaceous vegetation grazed with intensity 0.64 slightly increased $ANPP_h$ to 32.5 g m^{-2}, but was unable to compete with shrubs and did not return to the original biomass level.

Root biomass from simulation runs examining resilience of the system was analyzed in a similar fashion as sustainability runs: biomass was averaged for the growing season, May through October, and then averaged in 10 year increments. Grass root biomass ranged from 405 g m^{-2} to 467 g m^{-2} in runs that had previously been grazed with intensities 0.085 to 0.49, densities 1-9. Grass root biomass decreased as grazing intensity increased from 0.085 to 0.49 (densities 1-9) and followed a trend similar to grass root biomass with no grazing. Shrub root biomass was not affected by the removal of grazing. Shrub root biomass in areas that had been grazed with intensities 0.085 to 0.49 remained stable between 11.5 g m^{-2} and 20.4 g m^{-2}. Shrub root biomass in areas previously grazed with intensity 0.64 (density 10) ranged from 201.9 g m^{-2} to 220.8 g m^{-2} and shrub roots ranged from 238.3 g m^{-2} to 253.7 g m^{-2} in areas grazed with intensities 0.83 to 0.9 (densities 11-14). Shrub root biomass maintained the competitive advantage over grass root biomass in areas that had grazing intensities of 0.64 and higher.

Discussion

Grazing systems are in a constant state of change due to shifts in human and livestock population levels, changes in climate, changes in land use from grazing to agriculture, alteration of management technologies, and changes in management such as the change from nomadic to sedentary grazing practices (Archer et al. 1995; Walker and

Abel 1999). These changes have occurred in the Inner Mongolia region without an understanding of their consequences and therefore have left the system vulnerable to change. The Baiinxile Farm, in Inner Mongolia, China, is a semi-arid grassland region with variable climate patterns, which falls into the category of a non-equilibrium system (Holling 1973; Noy-Meir 1975; DeAngelis and Waterhouse 1987; Ellis and Swift 1988). Semi-arid grasslands are inherently at risk to shifts in stable states due to high climate variability. Because precipitation can be both low and variable in semi-arid regions, areas that can support certain levels of livestock during average precipitation years are vulnerable to overgrazing in low precipitation years when biomass is decreased (Sheehy 1993). Disturbances during low precipitation years reach a threshold that results in a shift to alternate stable states (Holling 1973), and grazing systems should be managed with this in mind (Ellis and Swift 1988; Ellis et al. 1993). It is vital to recognize these thresholds in order to maintain sustainable management.

In the past, herders on the typical steppe coped with extreme climate events by 1) maintaining low herd sizes; therefore, grassland use was not maximized, 2) maintaining a nomadic lifestyle, which enabled herders to more freely move herds to ungrazed areas, and 3) borrowing reserved grazing areas from other herders (Sheehy 1993; Chen 1996; Williams 1996b; Fernandez-Gimmenez 2000). Often, herders were unable to cope with these extreme events and livestock would die due to starvation (Chen 1996). Grazing management has changed in the last 50 years and consequently the grassland has shown signs of degradation. Nomadic grazing patterns have been replaced with more sedentary management and livestock grazing intensity has increased (Bilik 1996; Williams 1996a). One consequence is decreased winter reserves due to a shortage of land for higher

livestock numbers. The intensification of grazing has also resulted in changes in vegetation composition and increased bare ground, which demonstrates that livestock density had an impact. Changes in vegetation and bare ground were a result in changes in livestock density. As a result, grassland degradation is increasing (Li 1989; Sheehy 1993; Chen 1996) and the number of livestock die-offs in increasing (Chen 1996). At low and moderate livestock population levels, this system is regulated by density independent factors. But as livestock population levels increase, a threshold is reached and density dependent factors start to take affect (Illius and O'Connor 1999).

The idea of thresholds first surfaced in scientific literature when Holling (1973) challenged equilibrial views of natural systems by describing systems as being transient. He suggested systems have a certain resilience to disturbances, but if resiliency is low, disturbance can shift a system to another state or condition that differs from the traditional response of an equilibrium system (Holling 1973). This modeling analysis has shown the typical steppe region's susceptibility to shifts in vegetation states when heavy grazing is combined with years of below average precipitation, yet present day herders maintain an equilibrium style management where livestock numbers are kept constant. Westoby et al. (1989) proposed a new model for range management for non-equilibrium systems named the State-and –Transition Model. This theoretical model recognized the dynamics of rangelands by describing different vegetation states, and the importance of understanding the mechanisms causing transitions between states.

These 'transitions' occur when thresholds in a particular stable state are passed and the system shifts to an alternate stable state. Grazing systems are vulnerable to shifts in stable states when subject to poor management or the combination of intensified

grazing pressure and extreme climate events (Friedel 1991; Laycock 1991; Ellis et al. 1993; Rietkerk and van de Koppel 1997). Yet it is difficult to apply this knowledge when managing grazing systems, whether because information is too theoretical, non-specific, or herders simply do not have access to appropriate management information.

For these reasons, a systems approach was used to more specifically address the problems of grazing management. From my results, I hypothesize this system as having the potential to switch from a grass to a shrub dominated stable state when heavy grazing is combined with below average precipitation events. In model simulations, lower growth rates were not sufficient for herbaceous plants to respond from grazing and shrubs were able to capitalize on available resources once the herbaceous vegetation was suppressed (Archer 1996). Because shrubs were not browsed they displaced the overgrazed herbaceous plants and became the dominant functional group. Therefore shrub roots were able to capitalize on soil water and nutrients, and grow to their maximum size (Angell 1997).

The SAVANNA model determined what combinations of grazing intensities, precipitation, and vegetation patterns were sustainable for the Xilingol region and found where thresholds and stable states exist. Results of the model helped to explain how heavy grazing intensities, above 0.49, in combination with two or more years of below average precipitation events surpassed the threshold of sustainability and caused a catastrophic decline in grass vegetation. Angell (1997) conducted a four-year study on the effects of grazing on grass and shrub vegetation. His results were similar to this study in that grass standing crop decreased in years two, three, and four with heavy grazing, and low precipitation events limited grass regrowth after grazing. Standing

biomass in heavily grazed sites (1.2 AU ha^{-1}) was reduced by 43% as compared to continuously grazed (0.6 AU ha^{-1}) sites. He also found an increase in juvenile sagebrush survival due to heavy grazing on herbaceous vegetation. Gillen et al. (1998) also found decreases in standing crop due to heavy as compared to light grazing, decreasing by as much as 50 percent during one growing season. Although these were short-term studies, they show a similar trend to the first 10 years of model simulations in this analysis.

SAVANNA's capabilities allowed an analysis of the effect climate patterns and livestock densities have on grazing systems because the model includes livestock growth and feeding characteristics, plant growth and population characteristics, soil water dynamics, and climate patterns, all of which are important components of semi-arid grassland systems (Rietkerk and van de Koppel 1997). A simulation modeling approach to grazing management as opposed to a ground-based experiment was beneficial because it simultaneously incorporated abiotic and biotic environmental factors. SAVANNA also allowed for an analysis of multiple grazing patterns over a 3,680-km^2 area in a relatively short time period, which would have been extremely difficult with live animals in real time. However, this model analysis was at a 1 km^2 scale and did not include interspecific plant competition within herbaceous plants, nor within woody plants, which could impact grazing results. Model simulations included only grazers due to a system dominated by sheep, horses, and cattle. If goats, which browse shrubs, were simulated, then results from model simulations would likely be different.

Figure 3.13 is a graphical representation of the dynamics of this system in regard to thresholds and stable states (Fig. 3.13). The figure describes two steady states systems divided by a threshold zone where section I is dominated by grass vegetation and section

II is dominated by shrubby vegetation. The horizontal line in section I represents a grass dominated steady state system. As grazing intensity increases (moving left to right on the chart) the stable state line decreases to where shrubs dominate the system (region II). The angled line between the two stable state regions represents a transitional system. As grazing intensity increases towards the threshold isocline (grazing intensity < 0.49) the proportion of shrubs increases. But when grazing is removed, the system is resilient to these levels of grazing and is capable of returning to a grass steady state (represented by arrows) despite precipitation being below average. If grazing intensity increases past the threshold isocline (grazing intensity = 0.49-0.64) the system shifts to a shrub dominated stable state (region II). When grazing intensity ranges from 0.64 to 0.89 during below average precipitation events, the grassland system is not resilient. These levels of grazing will eventually result in a shrub dominated system, represented by the vertical line in region II. Time also factors into this graph. High grazing intensities tend to shift the vegetation to a shrub-dominated state, but this will happen in shorter time periods as grazing intensity increases.

Soil information was also limited at the scale used, which could potentially impact on shrub growth and/or effects of grazing (Rietkerk and van de Koppel 1997; van de Koppel et al. 1997). More information on soil and plant species accomplished by fieldwork would allow the model to be run at a smaller scale due to differences in water infiltration rates, which affect water availability to plants, and plant competition. This would create more detailed output accounting for the effects of soil patterns on shrub growth.

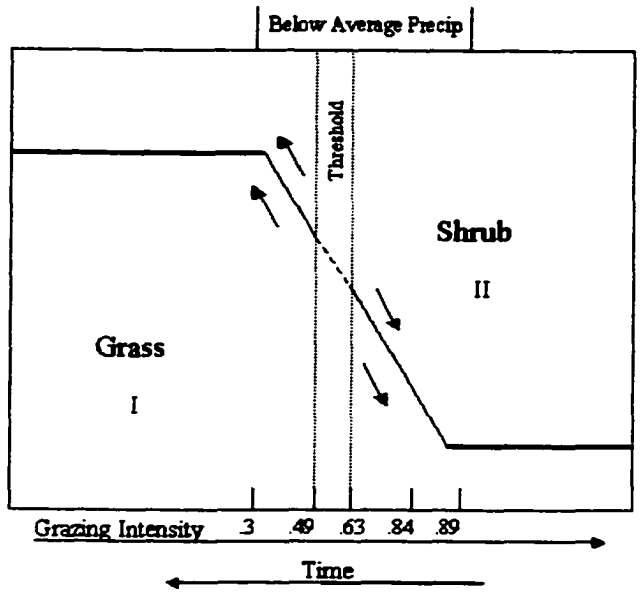

Figure 3.13. Grass/shrub stable state system, one dominated by grass (region I) and the other shrubs (region II). Horizontal lines (above Grass and below Shrub) represent systems in a stable states. As grazing intensity (GI) increases, the system increases in shrubby biomass (horizontal line shifts diagonally). The system is resilient if grazing intensity is below threshold zone (GI 0.49-0.63) and will return to a grass stable state if grazing is removed (arrows) despite precipitation being below average. If GI is higher than threshold zone (GI > 0.63) and precipitation is below average, the grassland system is not resilient to grazing and will gradually move towards a shrub stable state (arrows). The higher the GI, the shorter amount of time it takes for the system to move towards a shrub stable state.

The SAVANNA model determined potential thresholds in a particular system and the effects of climate on these thresholds; to better understand its sustainability and resilience for future successful management. Understanding thresholds has important policy implications in the Inner Mongolian region. Because of a non-equilibrium climate pattern, it is important to maximize grazing areas or maintain grazing reserves. This means limiting the amount of land being converted to cropping in order to maintain suitable areas for grazing. This simulation analysis as well as other research has shown the negative impact of high grazing intensities. To avoid negative impacts on this grassland region, policyholders must limit the amount of livestock or limit high levels of grazing intensity in certain areas to maintain sustainable livestock production.

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Chapter 4

Sensitivity Analysis of Typical Steppe Grassland Systems to Grazing and Climate Change

Abstract

This research examines the potential effects of climate change and grazing on the sustainability of grassland vegetation, Inner Mongolia, China. Grazing is an important form of land use in this region, yet there are uncertainties as to how it will be affected by climate change. A sensitivity analysis was conducted to study the effects of increased minimum and maximum temperatures, doubled CO_2, increased or decreased precipitation, and moderate grazing (39 animal units (AU) km^{-2}) on vegetation production. The model projected slight increases (3%) in herbaceous above ground net primary production ($ANPP_h$) with moderate grazing and small increases in temperature (+2° C and +0° C minimum and maximum temperature increase (T_{+2+0})) but doubled CO_2 (700 ppm) resulted in a 21% increase in $ANPP_h$ over a 50 year simulation period. The combination of increased temperature (T_{+2+0}) and precipitation (by 20%) increased $ANPP_h$ by 38%, while the addition of doubled CO_2 resulted in greater increases (56%). Doubled CO_2 caused a decrease in stomatal conductance and increased photosynthesis; therefore water use efficiency increased, which in turn increased plant growth. A 20% reduction in precipitation ($T_{+2+0}P_{80}$) with moderate grazing reduced $ANPP_h$. This grazing intensity could not be supported with reduced precipitation. Increasing CO_2 ($T_{+2+0}P_{80}C_i$) ameliorated the effect of lower precipitation but was not sufficient for system sustainability. A +5° C and +1° C minimum and maximum temperature increase (T_{+5+1}), and ambient or decreased precipitation with moderate levels of grazing were also unsustainable. Increased precipitation ($T_{+5+1}P_{120}$) and doubled CO_2 negated the effects of increased temperature and stabilized herbaceous production.

When grazing density was reduced (18 AU km^{-2}), climate change scenarios $T_{+5+1}P_{100}$ and $T_{+5+1}P_{120}$, that previously caused reductions in herbaceous biomass production with moderate grazing, were sustainable. Simulation runs with $T_{+2+0}P_{80}$, $T_{+2+0}P_{80}C_i$, and $T_{+5+1}P_{80}C_i$ climate patterns also resulted in a grazing system that was sustainable, but shrub biomass gradually increased over the 50-year run. These simulations demonstrate two important ecological points. First, grazing increases the vulnerability of grassland systems to climate change. Second, if net primary production is decreased by climate change, grazing rates must be reduced proportionately to available vegetation. This stresses the important role of management in maintaining grazing system sustainability.

Introduction

It is well recognized that atmospheric changes have occurred worldwide. These include doubled CO_2 (Keeling et al. 1995; IPCC 1996; Watson et al. 1996; Diaz et al. 1998; IPCC 2001), increased temperatures (Karl et al. 1993; IPCC 1996; Smit and Yunlong 1996; IPCC 2001), and changed precipitation patterns (Fu and Wen 1999; Smit and Yunlong 1996; Zhai et al. 1999; IPCC 2001). Although large-scale climate changes have occurred, changes are predicted to differ depending on regional location. For example, increased winter temperatures and decreased precipitation patterns are predicted in central and northern China (Smit and Yunlong 1996; Polley et al. 2000).

Multiple studies have been conducted to examine the effects of localized climate changes on vegetation processes, yet these effects, in combination with grazing, are less well known (Baker et al. 1993; Diaz et al. 1998). One reason is the difficulty of

conducting experiments at a large enough scale to ascertain grazing effects. Experiments have been conducted in chambers or sample plots to examine the interactive effects of climate change and grazing by insects or by clipping (Newton et al. 1995; Diaz et al. 1998), but few studies have explored the interactions of climate change and livestock grazing at larger scales (Baker et al. 1993). Grazing and grazing lands play significant roles in ecosystem processes (Watson et al. 2000), therefore it is important to understand how climate change may affect grazing ecosystems (Polley et al. 2000).

Grazing lands cover more than 40% of the earth's land surface (Allen-Diaz 1996; Polley et al. 2000), and are an important resource to those who rely on them. They exist due to their unique combination of temperature, precipitation, and vegetation appropriate for grazing. Native grasses of the typical steppe, defined as a bunch-grass steppe with many or few forbs, in a semi-arid climate (Lavrenko and Karamysheva 1993; Zhu 1993), are well adapted to herbivory and have supported grazing as the dominant land use for thousands of years. A change in climate could affect this tightly coupled system, disrupt the balance between grazers and vegetation, and have negative effects on the people who use them. Decreases in precipitation and/or increases in temperature may decrease plant production, thus reducing the number of livestock that could be supported. Decreased livestock numbers could result in economic hardship for pastoralists. For these reasons, it is important to better understand interactions between climate change and grazing, to prevent overgrazing and to maintain viable grazing systems. To best analyze climate-grazing interactions, it is vital to simultaneously consider ecosystem components, such as vegetation, soil, and livestock (Allen-Diaz 1996).

Consequently a systems approach was used in this analysis to examine the interactive effects of climate change and livestock on vegetation. Modeling is an appropriate tool to examine these effects because it allows an assessment of multiple factors over long time periods including multiple herbivore species and different climate scenarios, weather variability, spatial heterogeneity, and non-equilibrial assumptions. The SAVANNA model (Coughenour 1993) used in this analysis is a spatially explicit, dynamic, simulation model that represents ecosystem processes at both local and regional scales. Ecosystem processes are modeled by using data on soils, topography, climate, and livestock densities to affect vegetation production and cover, livestock forage utilization, livestock production, and human offtake. SAVANNA is process-based, representing flows of biomass, nitrogen, and organisms, and subsequent rates of change in system states. Its one-week time step enables large-scale spatial and long-term temporal simulations. These capabilities enable simulations of interactive responses to grazing.

The objective of this study was to assess potential effects of climate change and grazing on the typical steppe grasslands using the SAVANNA ecosystem model. A factorial experiment involving precipitation, temperature, and CO_2 was conducted to explore grazing system sustainability under climatic change scenarios.

Methods

Study area

The typical steppe region of Inner Mongolia, China extends across 41° to 47° north latitude and 109° to 117° east longitude. This study was focused on the Baiinxile

Livestock Farm located at 43.5° north latitude and 116.5° east longitude within the typical steppe. The farm is 3,680 km^2, divided into an administrative village and 12 branches. A branch is an administrative division of land, ranging in size from 79 km^2 to 655 km^2. This landscape consists of large rolling hills, with elevation ranging from 963 m to 1569 m. The typical steppe is dominated by C_3 grasses, which include *Leymus chinensis* and *Stipa grandis* (Li 1978). Important forb and shrub species include *Artemisia frigida* and *Caragana microphylla* (Zhu 1993). The grasses in this region have coexisted with grazers for thousands of years and are well adapted to grazing.

Mean annual temperature is –0.4° C but range from –17.0° C to 28.6° C (Li 1989; Xiao et al. 1997). Temperatures rise in March, but high winds and minimal precipitation from March to May postpone vegetation growth until the rainy season (Zhu 1993). Long term mean annual precipitation is 360 mm but varies between 180 mm and 500 mm. Seventy percent of total precipitation occurs in May through August (Xiao et al. 1997). Because the majority of precipitation occurs during the growing season, biomass production is high as compared to other regions with similar amounts of rainfall but with smaller proportions received during the growing season (Ellis and Galvin 1994).

Model description

SAVANNA is a spatially explicit model that simulates ecosystem processes using data on soils, topography, climate, and herbivore densities to affect vegetation production and cover, forage utilization, and herbivore population and production (Fig. 4.1). It has successfully been parameterized, validated, and used to simulate vegetation and ungulate

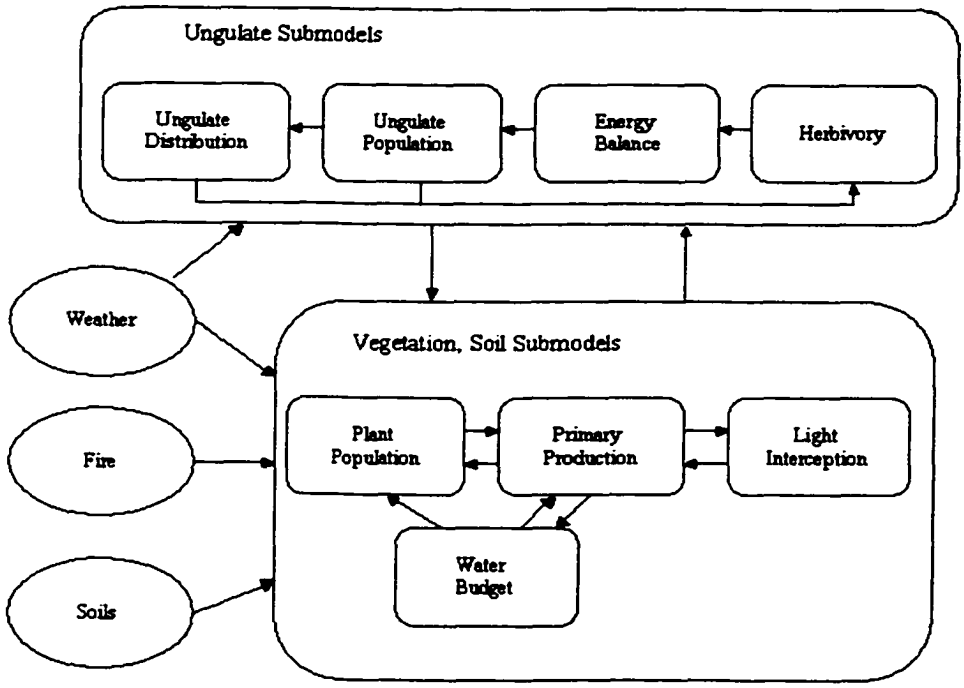

Figure 4.1. SAVANNA model structure (Coughenour 1993).

production in a wide variety of ecosystems such as Rocky Mountain National Park, Colorado (Weisberg 2000); Yellowstone National Park, Wyoming (Coughenour and Singer 1996a; Coughenour and Singer 1996b); Kajiado District, Kenya (Boone et al. 2001); Ngorongoro Conservation Area, Tanzania (Boone et al. in revision); Kruger National Park, South Africa (Kiker 1998); and Inner Mongolia, China (see Chap. 3).

The model was modified to represent CO_2 effects on photosynthesis and transpiration. Under doubled CO_2 (700 ppm) stomatal conductance and transpiration rates were reduced by 20%, while photosynthesis rates were increased by 20%. Conservative rather than maximum values were chosen based on field information (Jackson et al. 1994; Jackson et al. 1998; Wand et al. 1999; J. Morgan personal communication) because the modeled CO_2 effect was constant, and not dependent on time of day or season. The CO_2 response increased water use efficiency (WUE) and soil water, which could further affect plant growth (Knapp et al. 1996). Use of the Penman-Monteith equation (Penman 1953; Monteith 1965) to calculate canopy-scale transpiration, driven by temperature, radiation, and humidity allows for a plant physiological response to climate change because it uses heat, radiation, and vapor pressure deficit as well as stomatal conductance in predicting water loss by leaves. An increase in temperature increases vapor pressure deficit, which results in an increase in transpiration rate.

Experimental design

Three simulation experiments were conducted to examine temperature, precipitation, CO_2, and grazing effects on grassland vegetation. The first two

experiments examined the sustainability of a grassland system impacted with different climate change scenarios (CC) (Riedo et al. 1997) including increased temperature, precipitation changes, and moderate grazing. Moderate grazing was defined as 39 animal units (AU) km^{-2}. Moderate grazing, defined in Chapter 4 based on simulated vegetation impacts, ranged from 24 to 39 AU km^{-2}, whereas higher animal densities (> 39 AU km^{-2}) resulted in degradation. Thirty-nine AU km^{-2} is the maximum number of animals that could be supported by herbaceous vegetation while maintaining a sustainable system of grazing and herbaceous biomass production.

Experiment one was a sensitivity analysis, which used a factorial design with two temperature levels, three precipitation levels and moderate grazing. Minimum and maximum temperatures were increased 2°C and 0° C respectively (T_{+2+0}), then 5° C and 1° C (T_{+5+1}) respectively. Precipitation levels included 80%, 100%, and 120% of actual precipitation data (P_{80}, P_{100}, P_{120}). Experiment two was a factorial design, which followed the same design as experiment one, but with doubled CO_2 (700 ppm (C_{2x})).

A third experiment was conducted where moderate grazing was replaced with light grazing from the first two experiments. The purpose was to examine the importance of grazing management when the combination of moderate grazing and climate change resulted in vegetation decline. Light grazing was defined in Chapter 3 as 18 AU km^{-2}, the maximum number of animals per km^{-2} (within a range of 6 to 18 AU) that resulted in only slight decreases in herbaceous production.

Each experiment consisted of model runs 50 years in length. Eighteen years of local climate data were available from the area being simulated. To create 50 years of data, years were selected randomly from the 18 to create 32 more years. For climate

change scenarios, precipitation days and temperatures were altered uniformly within each year, i.e. climate was not varied intra-annually. Thus each monthly input of temperature was increased and each monthly input of precipitation data was increased or decreased by 20%. The output variables used in the analyses were herbaceous and shrub above ground net primary production ($ANPP_h$ and $ANPP_s$).

Modeling Results

Experiment 1

Combinations of ambient and increased temperatures (T_a, T_{+2+0}), ambient and increased precipitation (P_{100}, P_{120}), and moderate grazing had varying effects on $ANPP_h$, but all combinations maintained a sustainable level of herbaceous biomass. A T_{+2+0} increase in temperature increased $ANPP_h$ by 3%, from 154 g m^{-2} y^{-1} with ambient temperature and precipitation (T_aP_{100}) to 158 g m^{-2} y^{-1} (values are averaged for the 50 years simulation run) (Table 4.1). Adding a 20% increase in precipitation ($T_{+2+0}P_{120}$) resulted in a 38% increase in $ANPP_h$ compared to a 39% increase with T_aP_{120}. Climate change scenarios (CC) with a 20% reduction in precipitation, ambient and elevated temperatures (T_a, T_{+2+0}), and moderate grazing resulted in a decline in herbaceous productivity. The CC of ambient temperatures at 80% precipitation (T_aP_{80}) and increased temperature at 80% precipitation ($T_{+2+0}P_{80}$) reduced $ANPP_h$ by an average of 78% and 81% respectively (Table 4.1). Shrub above ground net primary production ($ANPP_s$) increased in response to decreased $ANPP_h$. Shrubby vegetation was not browsed, so it expanded as $ANPP_h$ declined and competition with herbs decreased (Fig. 4.2).

Table 4.1. Herbaceous above ground net primary production (g m^{-2} yr^{-1}) responses to ambient and elevated CO_2 (μmol mol^{-1}), increased temperature (°C), precipitation (Ppt) change (% of ambient), and moderate grazing (29 AU km^{-2}) from experiment 1 and 2. Values are averaged over 50-year simulation.

	Tempera-ture (°C)		350 μmol mol-1	700 μmol mol-1
Ppt (%)	ΔTmin	ΔTmax	CO2	CO2
80	0	0	34.6	130.7
	2	0	29.7	120.5
	5	1	22.3	34.4
100	0	0	153.9	186.9
	2	0	158.2	187.4
	5	1	30.8	183.3
120	0	0	214.0	239.4
	2	0	212.6	240.3
	5	1	142.0	211.2
			exp 1	exp 2

Combinations of T_{+5+1}, all precipitation levels, and moderate levels of grazing had greater impacts on $ANPP_h$ than the smaller temperature increase. A large temperature increase T_{+5+1} ($T_{+5+1}P_{100}$) in combination with moderate grazing reduced $ANPP_h$ catastrophically to an average of 30.8 g m^{-2} y^{-1} (Table 4.1). When precipitation was increased by 20% ($T_{+5+1}P_{120}$), $ANPP_h$ still decreased but much less, from 153.9 g m^{-2} y^{-1} to 142.0 g m^{-2} y^{-1}. Reduction in precipitation (CC of $T_{+5+1}P_{80}$) decreased $ANPP_h$ by an average of 86% from T_aP_{100}. There was a rapid replacement of the herbaceous community by a shrub-dominated one due to shrub access to water in deep soil layers and lack of browsing on shrub vegetation (Fig. 4.3).

Experiment 2

Elevated CO_2 with ambient precipitation, all temperature levels, and moderate grazing resulted in increases in $ANPP_h$. The $T_aP_{100}C_{2x}$, $T_{+2+0}P_{100}C_{2x}$, and $T_{+5+1}P_{100}C_{2x}$ scenarios resulted in 21%, 22%, and 19% increases in $ANPP_h$ respectively (Table 6.2). When precipitation was increased by 20%, there were much larger increases in $ANPP_h$. CC of $T_aP_{120}C_{2x}$, $T_{+2+0}P_{120}C_{2x}$, and $T_{+5+1}P_{120}C_{2x}$ resulted in 56%, 56%, and 37% increases in $ANPP_h$ respectively. A doubling of CO_2 did not ameliorate the effect of a 20% decrease in precipitation and moderate grazing on plant production, and $ANPP_h$ decreased in all CC scenarios. The $T_{+5+1}P_{80}C_{2x}$ scenario resulted in a 78% decrease in $ANPP_h$. CC of $T_aP_{80}C_{2x}$ and $T_{+2+0}P_{80}C_{2x}$ resulted in a gradual decrease in $ANPP_h$ but no shift in vegetation state (Fig. 4.4).

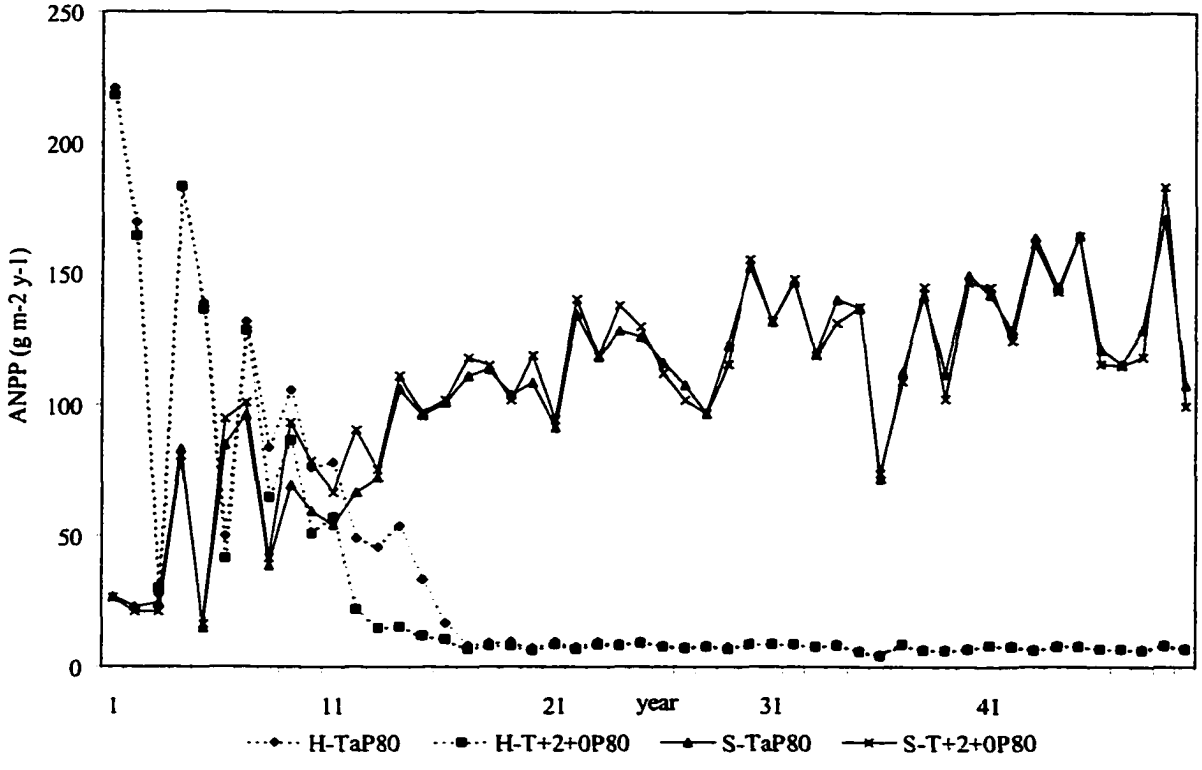

Figure 4.2. Herbaceous (H) and shrub (S) above ground net primary production from 50-year simulation runs with ambient and increased temperature (T_a, T_{+2+0}), and 80% precipitation (P_{80}), and moderate grazing.

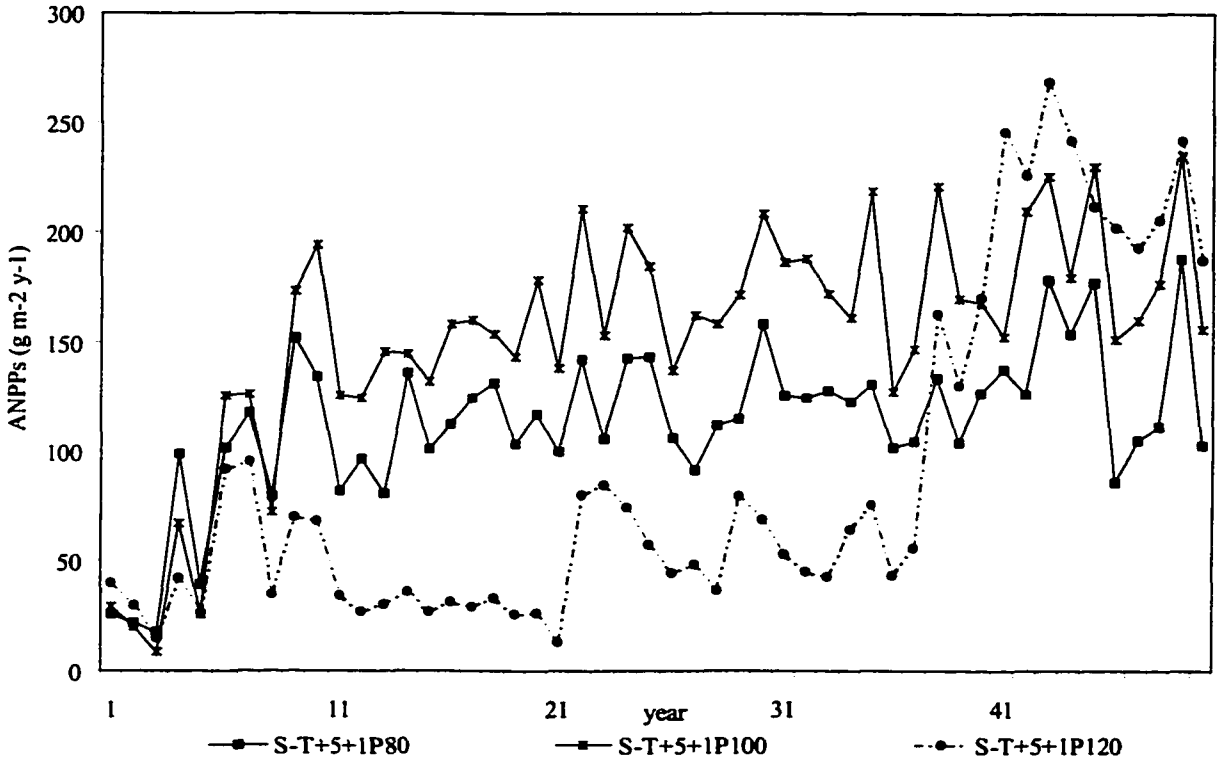

Figure 4.3. Shrub (S) above ground net primary production from 50-year simulation runs with increased temperature (T_{+5+1}), 80%, 100% and 120% ambient precipitation (P_{80}, P_{100}, P_{120}), and moderate grazing.

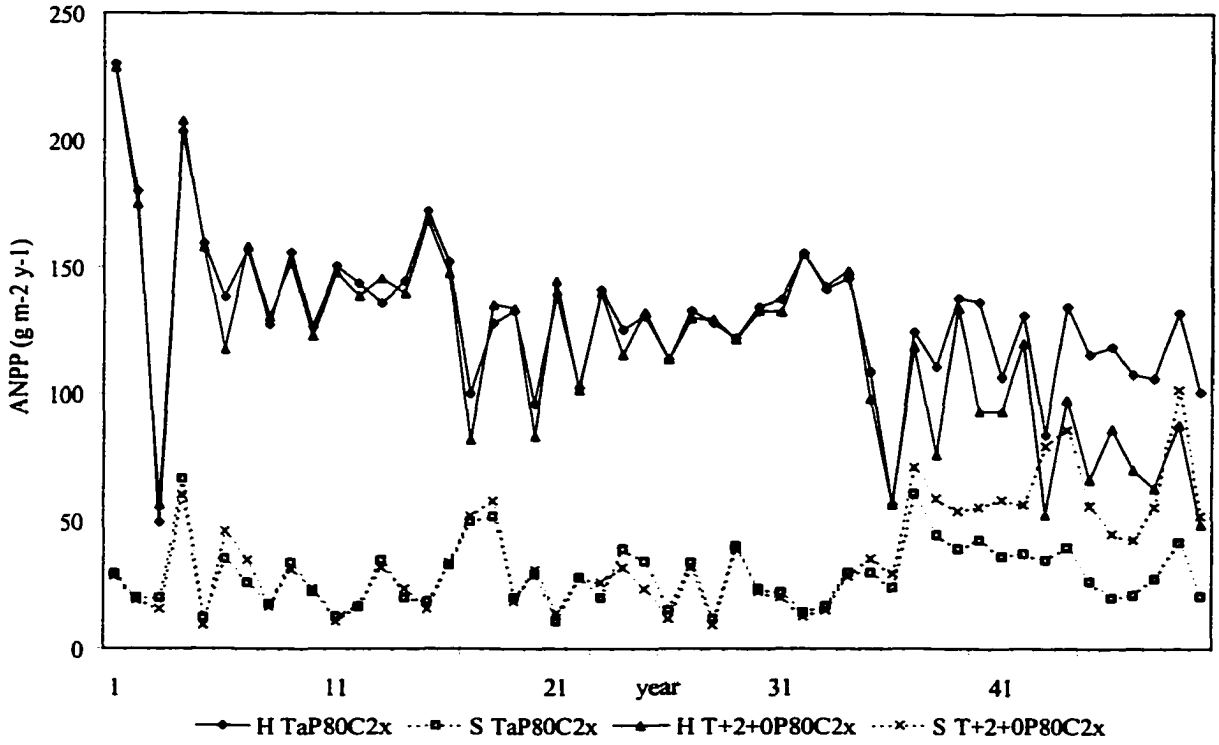

Figure 4.4. Herbaceous (H) and Shrub (S) ANPP from 50-year simulation runs with ambient and increased temperature (T_a, T_{+5+1}), 80% precipitation (P_{80}), elevated CO2 (C_{2x}), and moderate grazing.

Experiment 3

A decrease in grazer density reversed declines in $ANPP_h$ and maintained a sustainable level of herbaceous biomass production. When grazing was reduced from moderate to light with CC of $T_{+5+1}P_{120}$, average $ANPP_h$ increased to 228.2 g m^{-2} y^{-1} as compared to 142.0 g m^{-2} y^{-1} with a moderate grazing density (Table 4.2). The $T_{+5+1}P_{100}$ scenario and light grazing was also sustainable, with an average of 170.1 g m^{-2} y^{-1} (Fig. 4.5). Light grazing and $T_aP_{80}C_{2x}$, $T_{+2+0}P_{80}C_{2x}$, $T_{+5+1}P_{80}C_{2x}$, T_aP_{80}, and $T_{+2+0}P_{80}$ scenarios were sustainable for the 50-year simulation run with an average $ANPP_h$ of 151.3 g m^{-2} y^{-1}, 149.4 g m^{-2} y^{-1}, 135.9 g m^{-2} y^{-1}, 129.7 g m^{-2} y^{-1}, and 117.6 g m^{-2} y^{-1} respectively, but $ANPP_h$ values gradually decreased over the simulation period (Fig. 4.6). Light grazing in combination with CC $T_{+5+1}P_{80}$ was not sustainable and $ANPP_h$ production was reduced to 42.1 g m^{-2} y^{-1}. The system was unable to maintain herbaceous growth with decreased precipitation despite the reduction in grazing density.

Discussion

Simulations showed that $ANPP_h$ in this grazing ecosystem was most sensitive to changes in precipitation levels. However combinations of precipitation, temperature, and CO_2 had synergistic effects on herbaceous production. Increased temperature (T_{+2+0}), CO_2, and precipitation had positive effects on $ANPP_h$, which resulted in a 3-56% increase in above ground herbaceous growth with moderate grazing. Increases in precipitation had larger positive effects on $ANPP_h$ than a doubling of CO_2, while small increases in temperature only had minimal effects. Greater increases in temperature (T_{+5+1}) put a

Table 4.2. Herbaceous above ground net primary production (g m^{-2} yr^{-1}) responses to ambient and elevated CO_2 (μmol mol^{-1}), increased temperature (°C), precipitation (Ppt) change (% of ambient), and light grazing (18 AU km^{-2}) from experiment 3. Values are averaged over 50-year simulation.

	Temperature (°C)		350 μmol mol-1	700 μmol mol-1
Ppt (%)	ΔTmin	ΔTmax	CO2	CO2
80	0	0	129.7	151.3
	2	0	117.6	149.4
	5	1	42.1	135.9
100	0	0		
	2	0		
	5	1	170.1	
120	0	0		
	2	0		
	5	1	228.2	
			exp 1	exp 2

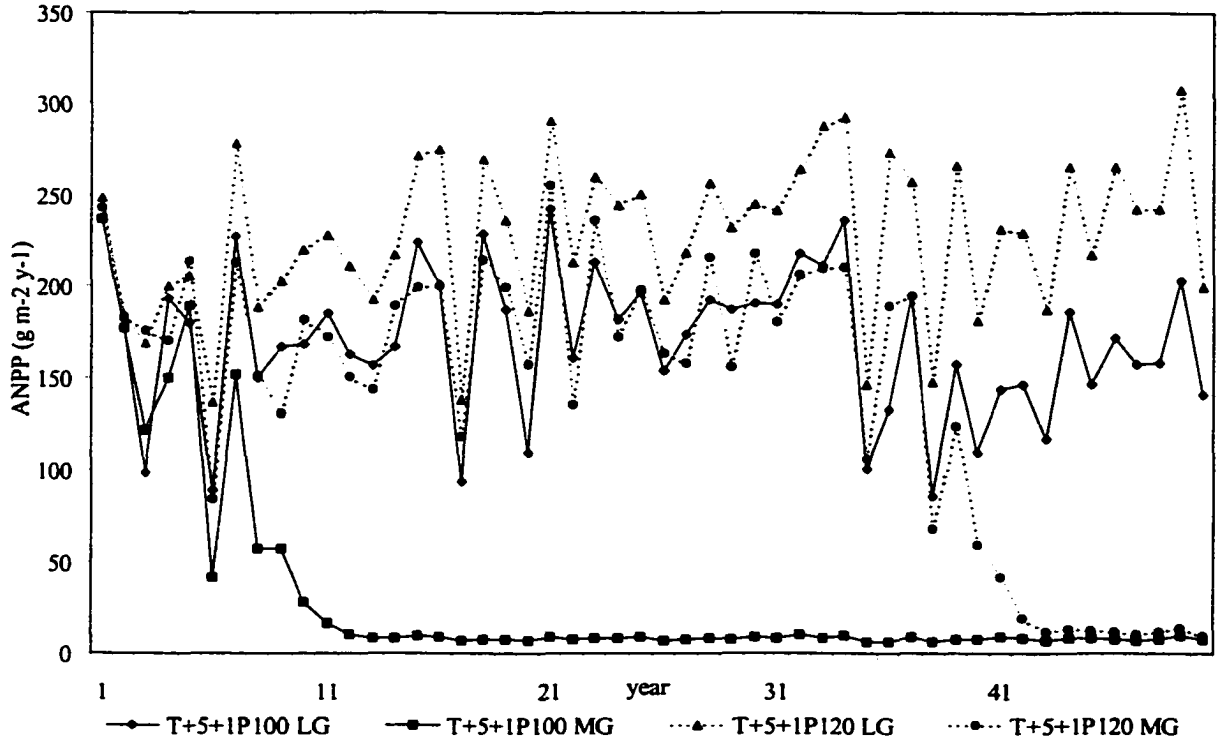

Figure 4.5. Herbaceous above ground net primary production ($ANPP_h$) in response to increased temperature (T_{+5+1}), and precipitation (P_{100}, P_{120}) change (% of ambient) with light grazing (LG = 18 AU km^{-2}) for 50-year simulation runs. $ANPP_h$ also reported for these climate change scenarios with moderate grazing (MG).

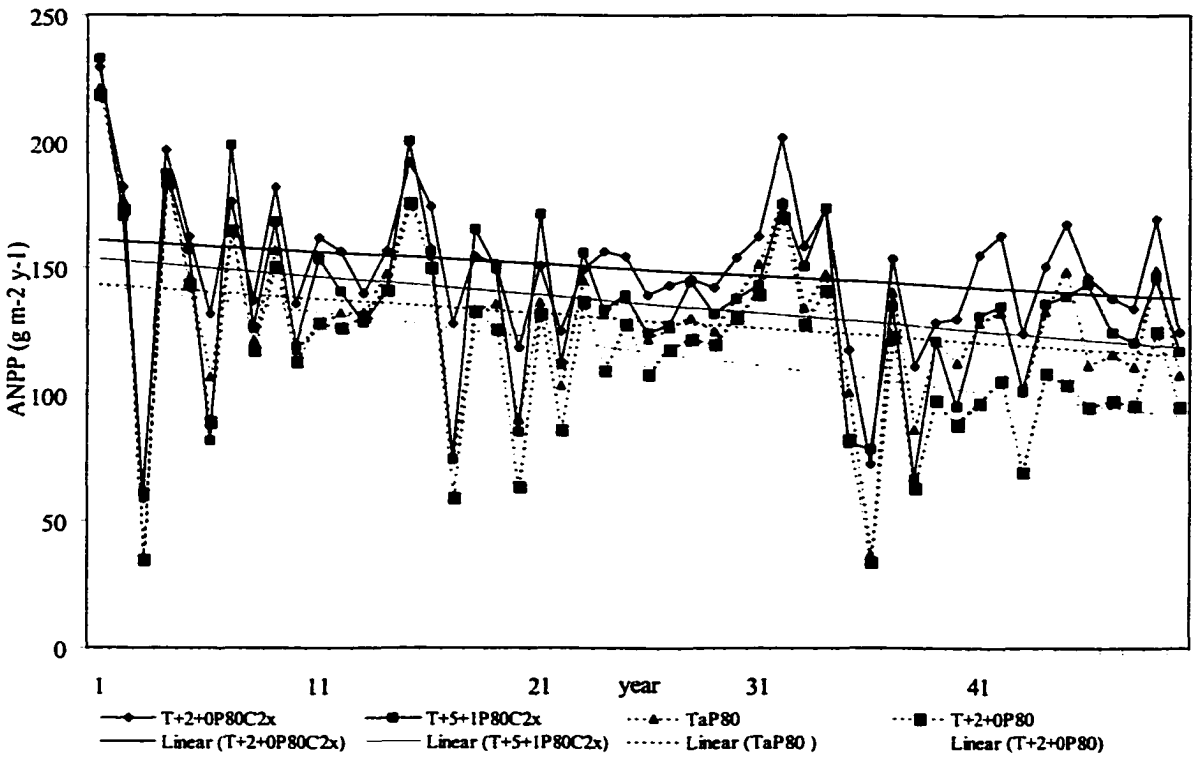

Figure 4.6. $ANPP_h$ in response to ambient and doubled CO_2 (C_{2x}), ambient and increased temperature (T_a, T_{+2+0}, T_{+5+1}), and precipitation (P_{80}, P_{120}) change (% of ambient) with light grazing (LG = 18 AU km^{-2}) for 50-year simulation runs. Linear regression trend lines from each climate change scenario are shown; only regression from climate change scenario $T_{+2+0}P_{80}$ was significant ($P < 0.05$).

larger stress on the plants, resulting in declining $ANPP_h$ unless increased precipitation, CO_2, or the combination of the two ameliorated the effects of the large increase in temperature. Because net primary production in this region is water limited, sufficient soil moisture was necessary for grasses to recover from grazing.

Reduced precipitation caused a decline in $ANPP_h$, so that moderate grazing of 39 AU km^{-2} could not be sustained in this modeling experiment. A decrease in precipitation resulted in lower soil water, which decreased plant growth rates. Lower growth rates were not sufficient for herbaceous plants to respond from grazing and shrubs were able to capitalize on available resources once the herbaceous vegetation was suppressed (Archer 1996). Because shrubs were not browsed they displaced the overgrazed herbaceous plants and became the dominant functional group. A doubling of CO_2 partially ameliorated the effects of a 20% decrease in precipitation as a result of increased WUE. A doubling of CO_2 caused a decrease in stomatal conductance and increased photosynthesis; therefore WUE increased, which in turn increased plant growth in this water-limited system. The CO_2 effects allowed grazed grasses to remain competitive relative to unbrowsed shrubs despite a 20% decrease in water.

Modeling experiments showed the system could not maintain a 50-year sustainable level of herbaceous biomass when subject to certain grazing and climate patterns; thus it was vulnerable to a shift from herbaceous to shrub-dominated vegetation. Shrub growth was limited because grasses limit water availability to shrubs with their intensive fibrous root structure (Köchy and Wilson 2000; Polley et al. 2000; Brown et al. 1998). Decreased precipitation in combination with grass removal from grazing and lack of shrub browsing allowed shrub encroachment. It is important to note this system was

dominated by grazers, including sheep, horses, and cattle. If goats, which browse shrubs, were simulated then the results would be different. These vegetation changes due to grazing and climate change are similar to threshold state changes typical of non-linear systems. It is important to recognize the potential of vegetation shifts with climate change because it can lead to permanent and sudden ecological changes with little forewarning. Managers need to monitor livestock offtake to reduce system vulnerability under such scenarios of climatic change.

Previous models have projected increased grassland production under elevated CO_2 (Parton et al. 1995; Coughenour and Chen 1997; Gao and Zhang 1997; Neilson et al. 1998; Polley et al. 2000). Coughenour and Parton (1997) used the mechanistic GRASS-CSOM model and found elevated CO_2 at ambient and increased precipitation levels resulted in a 20-70% increase in C_3 grassland production. Wand et al. (1999) in a meta-analysis concluded grass biomass on average increased by 44% with elevated CO_2. While biomass could not be directly related to production, the trends in both show increases. A simulation study conducted by Gao and Zhang (1997) predicted an overall 15% increase in vegetation production with doubled CO_2 in the northeast region of China. However, modeling analysis at a specific site differed with elevated CO_2, which resulted in a 26% and 61% increase in net primary production (NPP) in *Stipa baicalensis* steppe and *Aneurolipidium chinense* steppe grasslands. The combination of doubled CO_2 and increased precipitation and temperature caused a 24% decrease in vegetation production in the *Aneurolipidium chinense* steppe while it increased NPP by 44% in the *Stipa baicalensis* steppe. Decreases were due to larger temperature effects on the

Aneurolipidium steppe than the *Stipa* steppe(Gao and Yu 1998). SAVANNA was not parameterized to simulate differences in these species to elevated CO_2.

This modeling experiment explored the possible implications of temperature, precipitation, and CO_2 changes for grazing management of the grasslands in Northeast Asia. A small increase in temperature, with increased CO_2 and decreased precipitation is likely to occur in this region within the next hundred years (Keeling et al. 1995; Smit and Yunlong 1996; Giorgi et al. 1998; Polley et al. 2000; Watson et al. 2000; IPCC 2001). The simulation model estimates a 22% decrease in $ANPP_h$ under such a climate regime and moderate grazing. To maintain ecosystem states it would be necessary to decrease livestock numbers in proportion to the decrease in forage production (Coughenour and Chen 1997).

A second implication is the possible shift of vegetative state due to climatic change and grazing. Many rangelands exist in areas with variable climate patterns. Therefore, small shifts in climate could increase system vulnerability to vegetation state change (Rietkerk and van de Koppel 1997). Rietkerk and van de Koppel (1997) give a mechanistic explanation of grazing induced shifts in vegetative stable states. They used a model to show how a loss in plant growth with heavy grazing resulted in a decrease in soil water due to higher run-off and decreased percolation. Lower soil water subsequently decreased plant production, so resources were insufficient to sustain production while being grazed. They found semi-arid systems are vulnerable to threshold effects and are thus fragile, when resources have reached these low levels. Results here are consistent with that conclusion.

Shifts in temporal biomass patterns due to changed climate have potentially significant implications for management. The decrease in plant production due to increased temperatures and decreased precipitation show the need to reduce grazing levels so grass vegetation can maintain a competitive advantage over shrub growth. Biomass reached higher values earlier with increased precipitation, temperature, and CO_2 as compared to simulations with ambient climate. This shift in timing could affect optimal times for marketing animals and the timing of livestock movements. It is clear that management will need to be altered under changing climate to maintain system stability.

With expanding human populations, there are increased pressures on rangelands for their resources (Polley et al. 2000). The typical steppe region is particularly vulnerable because it is currently experiencing climate change and an increasing trend in human and livestock population densities coupled with increased sedentariazation. This intensified form of land-use could have disastrous effects with climate change (Archer 1995). Models have been used to address the effects of grazing and climate on land use processes (Parton et al. 1987; Hanson et al. 1988; Thornley and Verberne 1989; Hunt et al. 1991; Coughenour and Chen 1997) but there remain many unanswered questions (Nosberger et al. 2000). Studies are only beginning to analyze how livestock based agriculture will change due to shifts in climate patterns (Parsons et al. 2001).

Model simulations from this analysis give detailed insight as to which climate change scenarios had the greatest impact on changes in biomass. Grassland ecosystems are vulnerable to climate change and will likely experience increases or decreases in plant production. But more importantly, climate change can destabilize grazing ecosystems if

not managed properly. Systems that would otherwise remain unsustainable under increased temperatures and decreased precipitation can maintain their stability if livestock grazing densities are adjusted.

Nomenclature:

CC	climate change scenarios
T_aP_{100}	ambient temperature, ambient precipitation
$T_aP_{100}C_{2x}$	ambient temperature, ambient precipitation, doubled CO_2
T_aP_{80}	ambient temperature, 80% precipitation
$T_aP_{80}C_{2x}$	ambient temperature, 80% precipitation, doubled CO_2
T_aP_{120}	ambient temperature, 120% precipitation
$T_aP_{120}C_{2x}$	ambient temperature, 120% precipitation, doubled CO_2
$T_{+2+0}P_{100}$	increase minimum temperature by 2°C, max by 0 °C, ambient precipitation
$T_{+2+0}P_{100}C_{2x}$	increase minimum temperature by 2 °C, max by 0 °C, ambient precipitation, doubled CO_2
$T_{+2+0}P_{80}$	increase minimum temperature by 2 °C, max by 0 °C, 80% precipitation
$T_{+2+0}P_{80}C_{2x}$	increase minimum temperature by 2 °C, max by 0 °C, 80% precipitation, doubled CO_2
$T_{+2+0}P_{120}$	increase minimum temperature by 2 °C, max by 0 °C, 120% precipitation
$T_{+2+0}P_{120}C_{2x}$	increase minimum temperature by 2 °C, max by 0 °C, 120% precipitation, doubled CO_2
$T_{+5+1}P_{100}$	increase minimum temperature by 5 °C, max by 1 °C, ambient precipitation
$T_{+5+1}P_{100}C_{2x}$	increase minimum temperature by 5 °C, max by 1 °C, ambient precipitation, doulbed CO_2
$T_{+5+1}P_{80}$	increase minimum temperature by 5 °C, max by 1 °C, 80% precipitation
$T_{+5+1}P_{80}C_{2x}$	increase minimum temperature by 5 °C, max by 1 °C, 120% precipitation, doubled CO_2
$T_{+5+1}P_{120}$	increase minimum temperature by 5 °C, max by 1 °C, 120% precipitation

$T_{+5+1}P_{120}C_{2x}$ increase minimum temperature by 5 °C, max by 1 °C, 120% precipitation, doubled CO_2

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