

China's grassland policies and the Inner Mongolian grassland system

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Min Liu

Thesis

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Chapter 1

Introduction

1. INTRODUCTION

1.1 Problem statement

According to the FAO (2008), grasslands occupy 26% of the total land area of the world, and 70% of the global agricultural area. They are among the largest ecosystems in the world, contribute to the livelihoods of more than 800 million people, and are important as feed sources for livestock, as habitats for wildlife, for environmental protection efforts and for the in situ conservation of plant genetic resources (FAO, 2008). The sustainable use and management of grasslands have been of great concern to academic circles, policy-makers and NGOs, especially in the context of the growing worldwide problem of grassland degradation, rapid increases in demand for livestock products and the high level of poverty among pastoralists (Suttie et al., 2005; FAO, 2015). As defined by Fernandez-Gimenez and Swift (2003), sustainable grassland use and management should maintain the productive and adaptive capacity of grassland ecosystems while providing for the wellbeing of human communities. In practice, various policy interventions have been made, devoted to balancing the conservation of grasslands, production of livestock and the livelihoods of local households in agricultural societies.

However, the validity of existing policy interventions must be examined in terms of sustainable grassland use and management, given that grassland degradation is still ongoing and rural poverty is deepening in some pastoral areas (Hua and Squires, 2015). In this regard, research in multiple disciplines has found that some government policies accelerate resource destruction (Ostrom, 2009). To date, the policy interventions for grassland use and management widely employed by governments around the world aim to transform traditional pastoralism, by developing sedentarisation instead of nomadism, privatisation¹ of grasslands instead of communal tenure, intensification instead of extensive grazing, commercialisation of livestock systems instead of subsistence, deterministic stocking rates instead of opportunistic strategies with stochastic stocking rates, and so forth (Homewood, 1995; Li and Zhang, 2009). Traditional pastoralism is subsistence-oriented and has been perceived as primarily mobile, manifesting in such diverse forms as nomadism, transhumance, and migration as well as pasture rotation on a large scale (Fernandez-Gimenez and Le Febre, 2006). Widespread sentiments identify traditional pastoralism as an irrational, ecologically destructive and economically inefficient system (Homewood, 1995; Ho and Azadi, 2010). So, what are the actual outcomes of the government-driven

¹ Note that privatisation generally refers to the assignment of property rights over an asset or resource from the public (state) to the private sector. In the case of China, formal property rights over grasslands have remained in the public (or collective) sector. Hence, when we use the term privatisation for grasslands in the Chinese context we include the transfer of use rights over grasslands from the public to the private sector rather than the assignment of formal property rights.

transformation of traditional pastoralism? In other words, what are the impacts of the prevailing policy interventions on the grassland system²?

The global tendency of grassland policy in the twentieth century has been toward privatisation (or individualisation, subdivision, etc.), which transforms the property rights or use rights over grassland resources from communities to individuals (Mwangi, 2007; Li and Zhang, 2009; Li and Huntsinger, 2011). Since then, grassland privatisation has become the most significant transformation of traditional pastoralism, and the result of establishing physical boundaries between grassland parcels has led to a series of considerable changes in traditional modes of livestock production and household livelihoods (Sheehy et al., 2006; Li and Zhang, 2009). In general, land-ownership regimes other than private property are considered by many economists to be inefficient and prone toward the overuse of resources (Mwangi, 2007). Particularly, the conventional wisdom based on the concept of “the tragedy of the commons” (Hardin, 1968) and the theorem of property rights (Coase, 2013), contends that the open access to resources and unclear property rights of traditional pastoralism would inevitably cause overuse and increasing grassland degradation (Li et al., 2014). As such, grassland privatisation has been long embraced as the solution for achieving the sustainable use and management of grasslands (Mwangi, 2007; Ybarra, 2009). The development of grassland privatisation has also been influenced by the changes in the economy, such as technological innovation, changes in relative factor scarcities especially with increasing populations and the creation of new markets (Demsetz, 1967; Mwangi, 2007). However, an increasing number of scholars have explored some of the unintended effects of grassland privatisation, challenging the conventional wisdom of the “the tragedy of the commons”.

After the period of implementation of grassland privatisation, various environmental policies and programs have been introduced, influenced by growing attention to grassland ecosystems and worldwide grassland degradation in recent decades. The policy interventions for grassland conservation mainly target at controlling livestock grazing, because overgrazing is believed to be one of the main causes of grassland degradation by most of policy makers, such as in South Africa, South American Campos, China etc. (Suttie et al., 2005; Miao et al., 2015; Liu et. al., 2016). These environmental policies and programs further prompted pastoralists to transform their traditional lifestyles, particularly developing the intensification of animal husbandry, in place of extensive, grazing practices (Cao et al., 2013b).

Nevertheless, traditional pastoralism had prevailed for centuries. Increasing literature has criticised existing policy interventions as impairing the strategies of traditional pastoralism, reducing mobility,

² The grassland system in this research includes both the grassland condition (grassland area and quality) and livestock production. This is an adaptation of the resource system as defined by Ostrom (2009) in her framework of social-ecological systems.

flexibility and reciprocity of grassland use (Fernandez-Gimenez and Le Febre, 2006). It has been advocated that the government-driven transformation of traditional pastoralism broke the balance between the grassland ecosystem and livestock production, increased vulnerability to natural disasters and threatened pastoral livelihoods. In particular, privatisation and enclosure in arid and semi-arid pastoral areas seriously threatens the integrity of ecological systems because these areas are characterised by temporal variability and spatial heterogeneity (Mwangi, 2007). It is necessary to examine whether a tragedy of privatisation may have occurred, rather than the well-known tragedy of the commons. In addition, some scholars have implied that grassland degradation as a result of overgrazing is not possible in traditional pastoralism, because animals would be moved to new places for available forage before the vegetation of one area is significantly impacted by grazing (Li and Huntsinger, 2011). Furthermore, the restriction of livestock grazing inevitably causes economic loss to herders, which may impede the implementation of the environmental policies and programs. As such, the environmental policies and programs do not seem to be effective for preventing grassland degradation, especially those focussed on controlling livestock grazing.

What are the outcomes of these prevailing policy interventions, i.e. the outcomes of the government-driven transformation of traditional pastoralism? Are they valid in light of the sustainable use and management of grasslands? In this dissertation, these questions will be examined by researching the following issues: what is the impact of grassland privatisation on grassland condition and livestock production? What is the impact of the environmental policies and programs on grassland conservation? Although some academic studies and governmental reports have presented the results of policy interventions on grassland systems, the existing literature lacks quantitative analysis, based on large-scale areas and long-term observations, in order to dissect the policy effectiveness, and has thus far fallen short of thorough and systematic discussion on the outcomes of government-driven transformation of traditional pastoralism (Lu et al., 2015). Therefore, this research employs empirical analysis based on panel data involving extensive grasslands of China and spanning several decades, with a focus on the effects of the transformation of pastoralism on both grassland condition and livestock production.

1.2 Background information

1.2.1 Grasslands of China

China has around 392 million hectares of grasslands, the second largest area in the world after Australia, accounting for 12% of the world's grasslands and 41.7% of the national land area (Fan et al., 2008). Nearly 80% of these grasslands are in arid and semi-arid regions, which are characterised as most vulnerable to degradation, desertification and salinisation (Feng et al., 2008; National Bureau of

Statistics of China, 2009). Approximately 17 million people maintain their livelihoods on the grasslands of China (Li et al., 2014). The extensive grasslands of China are concentrated in six provinces and autonomous regions: Xinjiang, Tibet, Qinghai, Sichuan, Gansu and Inner Mongolia. They account for 75% of grasslands and accommodate 70% of the grazing livestock of China (Suttie et al., 2005). These areas maintained traditional pastoralism over hundreds of years, before undergoing thorough land tenure reform since the 1980s and a series of Ecological Construction Programs since 2000. These policy interventions have been implemented widely in favour of transforming traditional pastoral practices.

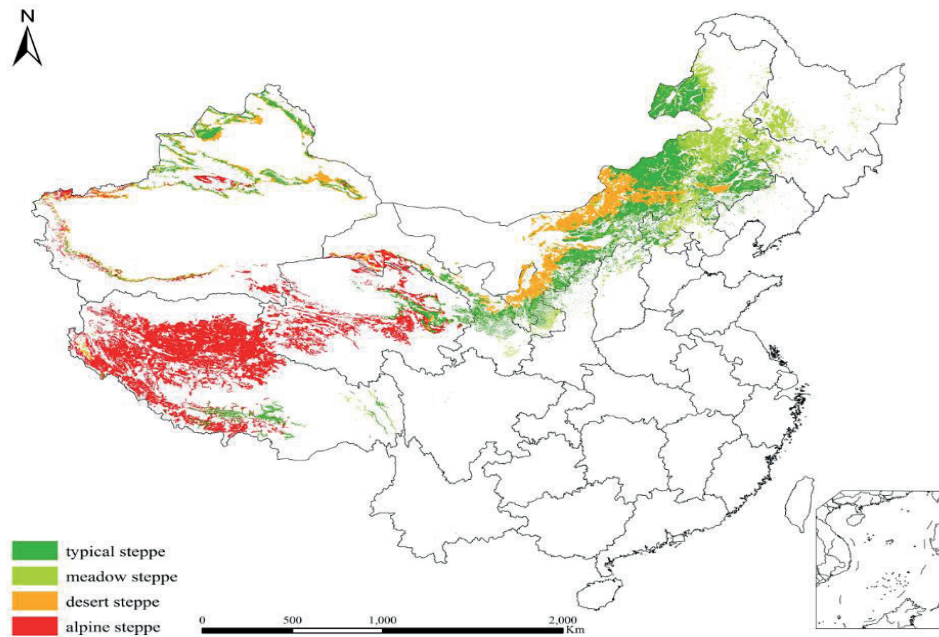


Figure 1.1 The distribution of grasslands in China

As shown in Figure 1.1, the grasslands of China are mainly distributed on the Inner Mongolia Plateau, the Loess Plateau and the Qinghai-Tibetan Plateau. The extensive grasslands are classified into four major types based on the analysis of plant community in phytology (Wu et al., 1980; Kang et al., 2007), including meadow steppes, typical steppes, desert steppes and alpine steppes. These four types of grasslands have a combined distribution ranging from the northeast plain adjacent to Mongolia to south of the Tibetan Plateau (Kang et al., 2007). The meadow steppes and typical steppes are the most commonly used grasslands for grazing and other economic activities related to livestock production, and most of them are located on the Inner Mongolian Plateau (Kang et al., 2007).

China's grasslands play crucial roles in livestock production, household livelihoods and global ecosystems (Kang et al., 2007). Northern China contains over 50% of China's grasslands, which are

called the traditional pastoral areas (Ho and Azadi, 2010), and which were home to the world's largest population of sheep and goats and the fourth largest concentration of cattle in the 1990s (Zhang and Yang, 1990; National Research Council, 1992; Ho and Azadi, 2010). They are considered to be the major areas for livestock production in China, with a long history of producing meat, milk, wool and fur. Moreover, local animal husbandry provides livelihoods to millions of people, with a majority existing as traditional self-sufficient enterprises and a few commercial grazing businesses (Li et al., 2014). The ecological function of China's grasslands is being emphasized increasingly, such as biodiversity conservation, carbon sequestration, soil retention, soil fertility maintenance, and catchment protection (Suttie et al., 2005). It was shown that the grasslands of northern China significantly impact both regional climates and the global carbon cycle (Ni, 2002). In addition, the majority of pastoral areas are populated by ethnic minorities, such as Mongolians and Ewenkis, thus protecting the grasslands of China also has a social function of maintaining cultural diversity and social stability (Kang et al., 2007). These socio-economic and environmental functions of grasslands have invoked the increasing attention of central and local governments to adopt policy interventions for sustaining the productivity of—in addition to protecting—the grassland ecosystem.

1.2.2 Grassland degradation in China

Grassland degradation, a worldwide problem, is particularly severe in China (Ho and Azadi, 2010). Grassland degradation can manifest itself in the lowering of grassland productivity, reduction in soil fertility, soil compaction, increased presence of unpalatable species of grass, or a combination thereof (Li et al., 2013). Although lacking clear documentation and differing in specifics regarding the extent of China's grassland degradation during the last several decades, it is unanimous among herders, scholars and governments that the grasslands of China have experienced degradation, with significant regional variation (Harris, 2010; Waldron et al., 2010). An often-cited figure is that about 90% of China's grasslands were degraded to various extents by the 2000s (Unkovich and Nan, 2008; Waldron et al., 2010), and that degradation was increasing at a rate of two million hectares per year (Ren et al., 2007; Harris, 2010; Cao et al., 2013b). The grassland areas shrunk significantly by the beginning of the twenty-first century. In particular, the northern grassland boundary had moved about 200 km southward and the western boundary about 100 km eastward compared to the previous hundred years (Liu 2008; Cao et al., 2013b).

A series of natural disasters in China in the late 1990s are to some extent responsible for degradation and desertification of the grasslands in northern China (Kang et al., 2007; Wu et al., 2015), for instance, the Yangtze River floods that killed thousands of residents, the Yellow River droughts that cost billions of US dollars in economic losses, and the frequent dust-storms and sand-storms that blanketed the sky of Beijing and affected the health and economic wellbeing of millions of city-dwellers (Harris, 2010).

It has been noticed that grassland degradation does not only impact herders who rely on the grasslands for survival and their livelihood, but also threatens the ecological security of the whole country (Harris, 2010; Huang et al., 2013).

Plenty of politicians and scholars have raised concerns over the causes of grassland degradation in China. Overgrazing and unsuitable grassland reclamation for cultivation have been accepted as the principal reasons (Ho, 2001). It was indicated that the total number of livestock in China increased by 594 million from 1981 to 2006 (Cao et al., 2013b), and an estimated 67 million hectares of high-quality grasslands have been converted to the cultivation of grains since 1949 (Ho and Azadi, 2010). However, other scholars have suggested that overgrazing only exists and leads to degradation in specific areas, with little evidence from national data to suggest that overgrazing per se has been responsible for widespread grassland degradation across China's pastoral grazing areas (Cao et al., 2013b). As such, one of the most controversial and debated aspects of research about grasslands is the existence and extent of overgrazing and its impact on grassland degradation (Suttie et al., 2005). Other anthropogenic interventions, such as over-excavation of medicinal herbs and mining, are thought to be responsible for at least some of the grassland degradation (Ren et al., 2007). Climate-related factors are an ongoing concern as a significant driver of grassland degradation in arid and semi-arid areas. For example, the distribution of precipitation is discussed as a principle factor of the changes in grassland condition (Harris, 2010; Cao et al., 2013b), while temperature is concluded to be a driver for changes in the biomass production of grasslands (Piao et al., 2006). Last but not least, the policy interventions that attempt to transform traditional pastoralism have been suspected to be significant causes of grassland degradation in China (e.g. Li and Zhang, 2009; Harris, 2010; Cao et al., 2013b).

1.2.3 Livestock production in the pastoral areas of China

China's livestock systems can be classified as traditional grazing, industrial production or mixed systems (Cao et al., 2013b). Traditional grazing systems are characterised by the direct consumption of permanent pastures (Waldron et al., 2007). The majority of production in traditional grazing systems provides subsistence to local residents, and therefore their livestock production usually includes several species of multipurpose animals—producing meat, milk, hides, fibre, transport, and manure for fuel (Suttie et al., 2005). These regions are called pastoral areas, which are geographically vast but sparse in population. They are mainly located in the west, north and northwest of China, including the Mongolian and Tibetan plateaus and Xinjiang Uighur Autonomous Region (Cao et al., 2013b). Industrial production systems aim to produce commodities for market sale, such as milk and meat. Their livestock production usually concentrates on one or two species of animals and relies also on crop products and sown pastures. Industrial production systems are mainly located in the southeast (Guangdong Province and Guangxi Zhuang Autonomous Region), north (Shanxi Province, Hebei Province and Beijing and

Tianjin Municipalities) and northeast (Liaoning Province, Jilin Province and Heilongjiang Province) (Cao et al., 2013b). Mixed systems are the interface between traditional grazing and industrial production.

China is a major player in the international livestock sector and has experienced impressive increases in livestock production³ but it has been observed that the increases in livestock production are predominantly associated with industrialised, rather than traditional, grazing systems in China (Cao et al., 2013b). The intensification of livestock production in China started in the late 1980s and has advanced progressively ever since, which resulted in significant changes in livestock production structures as well as the geographical distribution and composition of the livestock commodities of China (FAO, 2016). In addition, with the increasing environmental concerns about the ecosystem of natural grasslands, livestock production increasingly relies on the utilisation of crop stalks, bran, and other byproducts of grain, instead of natural pastures (FAO, 2015). In the 1990s, the central government called for moving cattle and sheep production from the traditional grazing region to the grain-producing provinces, because it was believed that overgrazing of pastoral areas was the main reason for grassland degradation. This policy strategy was strengthened after a series of environmental disasters had occurred in the 2000s. What followed has been a steady increase in the total livestock outputs in China but a decrease in the share of livestock production in the pastoral areas (Li, 2009). In short, the market share of livestock products in China has experienced a geographic shift from pastoral areas to crop-farming areas, in accordance with changes in production technology and national policy strategies in recent years (Li et al., 2008).

Livestock production in the pastoral areas provides livelihoods to millions of people and has considerable capacity to alleviate the increasing stress of livestock product demands that have risen with growing population, urbanisation and rising disposable incomes in China (FAO, 2015). Nevertheless, few studies have discussed the livestock production of the pastoral areas clearly and thoroughly. One important reason is the questionable accuracy of Chinese livestock statistics, especially the data about the pastoral areas (Waldron et al., 2007). Statistical data collection on the livestock population is extremely difficult in China because small rural households raise livestock at a small scale, and supply chains are dominated by countless small traders and processors, unlike the centralised slaughter and auction systems that facilitate statistical data collection in developed countries (Waldron et al., 2007). China's method of collecting these data has been a system of bottom-up reporting, which is vulnerable

³ Based on the FAOSTAT database, China's meat output accounted for 10.8% of the world total in 1980, and the share rose to 28.7% in 2004 (Li, 2009). The average growth rate in meat production between 1980 and 2004 was 7% in China, compared with only 3% for the world (Waldron et al., 2007).

to misreporting from herders and local officials hoping to respond to the policy orientation of higher-level governments (Waldron et al., 2007).

1.2.4 The overview of China's grassland policies

For a long time, China had no comprehensive long-term policy for grasslands (Ho, 2000). In the past, grasslands were owned by princes, lamaseries, landlords, or clans, and remained in common use by tenant herders. After that, in the collectivist period of New China, the grassland was owned by the production team or People's commune and the grassland was still managed and used communally by all local herders (Hua and Squires, 2015), while local communities had their own institutional structures for the use and management of grasslands (Ho, 2000). When the People's commune was abolished and grassland degradation was impending, there was an urgent need to fill the institutional vacuum for grassland use and management (Ho, 2000). Subsequently, there are two major policies that have been widely employed for the grassland use and management of China, including the land tenure reform during the 1980s-1990s and a series of Ecological Construction Programs after 2000. The former aims at privatising use rights of grasslands to individual households and displacing traditional common use (Banks et al., 2003) and the latter mainly concerns the grassland conservation as well as the maintenance of local household livelihoods (Liu et al., 2016).

More specifically, land tenure reform started with the Household Production Responsibility System in the cropland areas of China in the late 1970s and spread to the pastoral areas in the early 1980s. The land tenure reform in the pastoral areas was implemented by local governments through local regulations, such as the Double Contracting System ("cao xu shuang cheng bao") in Inner Mongolia. The ownership of livestock was assigned to households on the basis of the size of the household and labour force (Banks et al., 2003). The use rights to grasslands were informally allocated to small groups of households, and use rights to hayfields and croplands were allocated to individual households. Meanwhile, the marketing system for most livestock products was simultaneously liberalised and only wool remained under state monopoly control (Banks et al., 2003). The first Grassland Law of China, initiated in 1985, stipulated that the property rights over grasslands are owned either by the state or the collective but that use rights to the grasslands could be contracted out to either households or collectives. Moreover, the users (individual households) of the grasslands are regulated to control their livestock numbers according to a stocking rate in order to avoid overuse of grasslands (Ho, 2000). Then, the assignment of grassland use rights to individual households was strengthened by Chinese governments in the 1990s, which greatly fuelled the progress of grassland privatisation and was regarded as the second round of grassland privatisation in China (Li and Zhang 2009). In 2002, the revised Grassland Law reaffirmed the shift from state- and collective-oriented land use rights to those of individual households (National People's Congress of the People's Republic of China, 2002). The State Council

emphasised the specific allocation of the contracted items (e.g. grassland plots, areas, issuing user-right certificates) to households to ensure the long-term stability of contract relations in 2011 (Li, 2012). According to the current Law on the Contracting of Rural Land, grassland use rights can be contracted to individual households for 30 to 50 years.

Since the severe natural disasters of the late 1990s (Harris, 2010), the central government has hastened to engage in ecosystem protection and therefore a series of Ecological Construction Programs have been initiated, especially after 2000 (Wu et al., 2015). For example, the Conversion of Cropland to Forest and Grassland Program (also known as the Grain for Green Program) was mainly implemented between 2001 and 2010, and covered 25 provinces of China with a total budget of 225 billion RMB. It involved returning sloping or deserted cropland to forest or artificial grassland (Wang and Bennett, 2008; Liu, Lu and Yin, 2010). The Returning Grazing to Grassland Program was launched in 2000. It is one of the largest environmental programs on the grasslands of China. This program aimed at grassland conservation by sowing grass on those grasslands experiencing serious degradation, as well as restricting grazing in all program areas, involving grazing bans (either permanently or seasonally), rotational grazing and setting a deterministic stocking rate (Li et al., 2014; Hua and Squires, 2015). Apart from these two major programs in the 2000s, there were other programs that employed similar policy measures for grassland conservation, such as the Program to Combat Desertification around Beijing and Tianjin. These Ecological Construction Programs were implemented at household-level, with or without some compensation in cash, grain or grass seeds for the herders' economic loss (Suttie et al., 2005). Since 2011, another successive Ecological Construction Program, the Subsidy and Incentive System for Grassland Conservation (SIGSC), has been initiated in eight pastoral provinces of China, including Inner Mongolia, Xinjiang, Tibet, Qinghai, Sichuan, Gansu, Ningxia and Yunnan. It aims at conserving the condition of grassland and meanwhile improving local herders' livelihoods and livestock production through intensification (State Council of the People's Republic of China, 2010). The central government promised a budget of 13.4 billion RMB per year for its implementation between 2011 and 2015. The major regulations on grazing still include the permanent grazing ban and the deterministic stocking rate, but they are supported with more subsidies or incentives to compensate the herders' economic loss.

1.2.5 Research region: the pastoral areas of Inner Mongolia

Inner Mongolia, a province⁴ located in the arid and semi-arid areas of northern China, has 118.3 million hectares of land with a permanent population of 25 million in 2014. It contains 21.7% of the area of China's natural grasslands. Approximately 67% of the total land in Inner Mongolia is classified as

⁴ The full name is Inner Mongolia Autonomous Region.

grassland (Angerer et al., 2008). (Angerer et al., 2008). It has played an important role in the supply of livestock products and the ecosystem of China. Official statistics showed that the animal husbandry of Inner Mongolia accounted for 45% of the province's gross output value of agriculture in 2013. Inner Mongolia accounted for 18% of the population of sheep in China in 2013; the mutton output accounted for 22% of total output in China; 6% of cattle were raised in Inner Mongolia, and 8% of beef was produced there.

As a traditionally pastoral region populated by Mongolians, the vast majority of local people maintain their livelihoods by grazing on the grasslands (Angerer et al., 2008). Inner Mongolia has been the major and prior region for implementing the grassland policies in China because of its typical position in China's grassland resource and livestock production (Li and Huntsinger, 2011). However, it was reported that 90% of the natural grasslands of Inner Mongolia had been degraded to some extent by 2000 and grassland degradation remains problematic since (Waldron et al., 2010; Briske et al., 2015). With society's growing attention to the grassland ecosystem and continuing grassland degradation, Inner Mongolia has been increasingly emphasised as an ecological barrier of northern China instead of the previous status it had as the main production area of livestock products (Wu et al., 2015). Moreover, the provincial government of Inner Mongolia has put emphasis on the development of livestock production in crop farming areas, rather than the pastoral areas, since 2001. For instance, 70% of livestock production in Inner Mongolia was located in the crop farming area in 2007 (Su, 2010).

According to the current administrative divisions of Inner Mongolia, there are 102 counties, including 33 pastoral counties, 21 semi-pastoral counties and 48 counties dominated by crop farming or urban districts. Pastoral counties are characterised by traditional grazing systems and natural grassland is the dominant land type. In semi-pastoral counties, both natural grassland and cropland are the dominant land types, while traditional grazing systems exist and mixed crop-livestock systems also occur. The pastoral counties and semi-pastoral counties are usually regarded as the pastoral areas of Inner Mongolia and represent a typical region to investigate the grasslands of China (e.g. Taylor, 2006; Li and Zhang, 2009). Figure 1.2 illustrates the location of Inner Mongolia in China and its three types of counties.

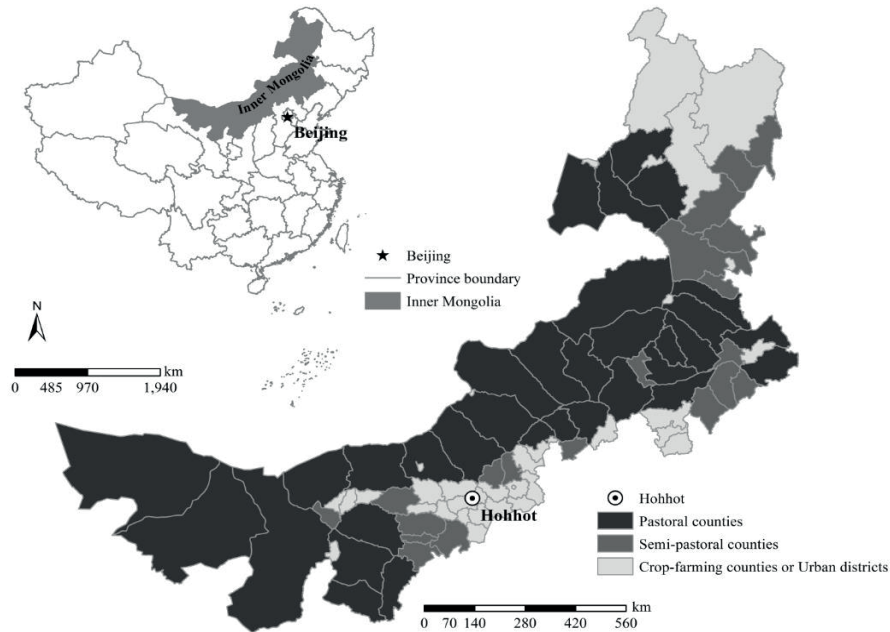


Figure 1.2 Inner Mongolia and its 102 counties

1.3 Objectives and research questions

Given that China's grasslands are confronted with the challenge of grassland degradation, the increasing demand for livestock products and the livelihoods of millions of households, an investigation into the sustainability of grassland use and management is urgently needed. Plenty of policies devoted to transforming traditional pastoralism have already been implemented to balance the productive and ecological functions of grasslands. However, grassland degradation and rural poverty in some pastoral areas are still worsening in China and there is no consistency within academic circles regarding how to transform traditional pastoralism under contemporary social, economic and political settings. As such, the objective of this research is to dissect the outcomes of China's policy interventions on its grassland system and further discuss the government-driven transformation of traditional pastoralism.

To fulfil this research objective, four research questions will be addressed in four chapters of this dissertation:

Question 1: What are the individual and social optima of rural land allocation by stakeholders: among cultivated land, grassland, forest and other rural land? This question is answered in Chapter 2. This chapter explores the different desires of stakeholders shaping grassland areas, which mirrors the different perceptions of grassland use under government-driven transformation of traditional pastoralism.

Question 2: What is the impact of land tenure reform on the grassland condition in the pastoral areas of China? This question is answered in Chapter 3. Chapter 3 investigates the ecological effect of land tenure reform on the grassland system, which presents the ecological outcome of the transformation of pastoralism with the abandonment of communal tenure.

Question 3: What is the impact of land tenure reform on livestock production in the pastoral areas of China? This question is answered in Chapter 4. This chapter explores the economic effect of land tenure reform on the grassland system, and presents the economic outcome of transforming pastoral areas, in terms of livestock changes in population as well as productivity.

Question 4: What is the impact of Ecological Construction Programs on the grassland condition in the pastoral areas of China? This question is answered in Chapter 5. This chapter investigates the effect of Ecological Construction Programs on the grassland system, which mirrors the ecological outcome of the transformation of pastoralism, through the use of deterministic stocking rates and grazing ban.

To summarise, the research objective is specifically addressed by evaluating the ecological and economic effects of the land tenure reform and Ecological Construction Programs that have been employed to intervene in the grassland use and management in China. The aim of this research is to contribute to the approach toward sustainable use and management of grasslands in China or other regions that have similar circumstances to China's, by providing empirical evidence for the academic arguments on the transformation of traditional pastoralism.

1.4 Materials and methods

1.4.1 Conceptual framework

All resources used by humans are embedded in complex social-ecological systems (SESs). Ostrom (2009) introduced a general framework for analysing sustainability of SESs, which integrates the different frameworks of different disciplines and provides a common method for dissecting the complex SESs. It helps to identify the potentially relevant variables and their subcomponents to design the data collection, to conduct the fieldwork, and to perform the analysis of findings about the sustainability of complex SESs (Ostrom, 2009). As such, this research employs Ostrom's framework as the starting point for conducting our study of linked SESs of grasslands.

Figure 1.3 illustrates the conceptual framework adapted from Ostrom's framework for analysing the SESs of grasslands. It shows the four first-level core subsystems of the SESs of grasslands, the relationships between subsystems and the link to social, economic, and political settings and related ecosystems. The four first-level core subsystems in the SESs of grasslands are identified as: (1)

grassland system (e.g. grazing area, water systems, and animal husbandry); (2) grassland resource units (e.g. forage, water, and animals); (3) users (e.g. herders, farmers and others); and (4) governance system (e.g. government and policy interventions). These four subsystems are relatively separable but interact to produce outcomes at the SES level, which in turn affect the subsystems (Ostrom, 2009). This thesis aims to reveal the impacts of governance systems upon the grassland systems, resource units and users. One of the broad questions this SES framework is used to solve is what impacts are likely to result from using particular rules for the governance, ownership, and use of a resource system and resource units in a specific technological, socioeconomic, and political environment (Ostrom, 2007). Long-term sustainability depends on rules matching the attributes of the resource system, resource units, and users (Ostrom, 2009). This study is devoted to examining whether the policy interventions of China are beneficial to the grassland system, grassland resource units and users in the long term.

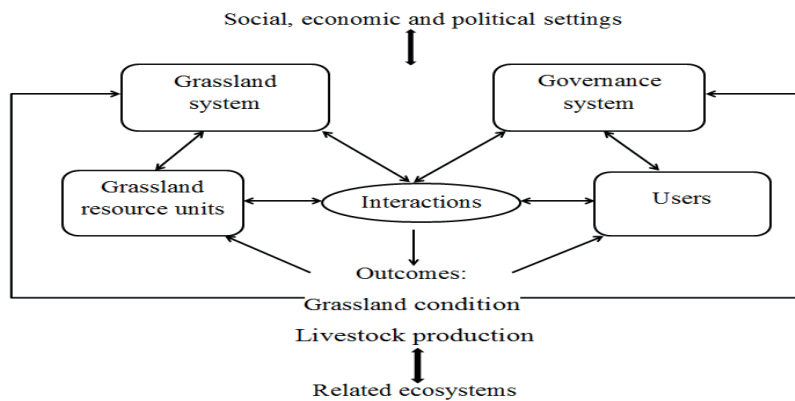


Figure 1.3 Conceptual framework for analysing social-ecological systems of grasslands (Adapted from Ostrom’s framework for analysing social-ecological systems (Ostrom, 2009))

More specifically, this study focuses on evaluating the ecological and economic impacts of China’s policy interventions on grasslands, which essentially reflects the outcomes of governance systems on grassland systems under the social, economic and political settings in which said systems are embedded. The outcomes for grassland systems are specifically measured by grassland condition and livestock production. The social, economic and political settings considered refer to market forces, technological development and climate changes among other factors. The governance system relates to the specific rules for the use and management of China’s grasslands, including the land tenure reform studied in Chapters 3 and 4 and the Ecological Construction Programs studied in Chapter 5. In addition, the conflicts of different users over resource units, specifically the grassland area, are investigated in Chapter 2.

1.4.2 Sampling and data collection

In Chapter 2, the analysis of optimal land allocation is based on a case study. The sample county, Tai Pusi County⁵, is a semi-pastoral county located in the eco-fragile region of Inner Mongolia, which has been characterised by economic backwardness and ecological degradation. The county covers different types of rural land and involves a variety of different stakeholders with divergent interests in rural land use. It is a typical region to investigate the conflicts of rural land allocation in China, reflected in the different preferences for grasslands by different stakeholders. For the data collection, interviews with the representatives of different types of stakeholders were conducted face to face to reveal their preference on land use, involving cultivated land, grassland, forest land and other rural land. The secondary data on socio-economic indicators was collected from statistics and reports from local governments.

Chapters 3, 4 and 5 focus on policy evaluation. Panel data at county-level is used to answer the research questions. Policy interventions have occurred over different years and there are differences in the process of implementation among counties. These differences within and between counties are presented by the panel data. For the analysis of land tenure reform in Chapters 3 and 4, all of the counties of Inner Mongolia were considered as possible sample counties. After excluding the counties with invalid survey feedback, and after dropping the urbanised counties where changes in the grassland area can be attributed mainly to changes in land use types 60 counties are retained as the research sample, including 29 pastoral counties, 18 semi-pastoral counties and 13 crop farming counties⁶. The research period covers 1985-2008, spanning the main period for implementing land tenure reform on the grasslands of Inner Mongolia. For the analysis of Ecological Construction Programs in Chapter 5, all of the counties in the pastoral areas, including 33 pastoral counties and 21 semi-pastoral counties, were used in the research sample. It should be noted that the crop-farming counties and urban districts are excluded from the research sample because they are not the typical regions included for the Ecological Construction Programs. This research period spans 14 years from 2001 to 2014, involving the years before and after the implementation of the specific Ecological Construction Program studied (i.e. the Subsidy and Incentive System for Grassland Conservation).

In terms of the data collection for the policy evaluation, the required dataset includes four categories of information: the policy interventions, the grassland condition, the livestock production, and the socio-economic and climate conditions. The information about policy interventions, such as the progress of land tenure reform and Ecological Construction Programs, was gathered based on the statistics and

⁵ Also named as Tai Pusi Qi in Chinese.

⁶ The crop farming counties considered also have grasslands and experienced the same land tenure reform as the pastoral area.

reports from the provincial government as well as through questionnaires. The questionnaires were sent to the Animal Husbandry Bureau of each county and answered by local experts. In the second phase, interviews were conducted by telephone to confirm the answers that were received. Apart from the questionnaires, interviews were conducted with the officers working in the provincial institutes of Animal Husbandry of Inner Mongolia and with local herders to verify the information thoroughly. Remote sensing and GIS (geographic information system) were employed to obtain data about the grassland condition across extensive grasslands over a long period. The data on livestock production and socio-economic and climate factors was collected based on existing statistical data gathered by governments.

1.4.3 Model selection

In Chapter 2, the model for investigating the optimal land allocation is based on welfare economics, specifically utilising the utility function and social welfare function. The utility function presents the utilities of different stakeholders for land allocation based on their preference. The Analytic Hierarchy Process (AHP) is used to quantify the preference of stakeholders for different types of land. The social welfare function is derived based on the utility functions and the weights of stakeholders in land allocation, in which three scenarios are employed to estimate the weights. Ultimately, the divergences among individual optima derived from the utility functions, the social optimum derived from the social welfare function, and actual rural land allocation, illustrate the conflicts in preferences of different stakeholders regarding rural land use.

Chapters 3, 4 and 5 employ panel data to dissect the effectiveness of policy interventions. The fixed effects model, which is used to study the causes of changes within an entity over time, is employed to conduct an empirical estimation. The fixed effects model has been widely used in economic research (e.g., Fergusson et al., 2002; Huan et al., 2006). Compared with the multiple regression model, the advantage of the fixed effects model is the control over unobserved fixed factors that could affect the estimation (Liu et al., 2010; Verbeek, 2012). Theoretically, the fixed effects model employs within transformation to remove all time-invariant (fixed) explanatory variables, i.e. the model is performed in deviations from individual means (Verbeek, 2012). As such, it provides a method that takes observable explanatory variables as well as unobservable time-invariant variables into account, but the estimation does not depend on the value of time-invariant (fixed) variables, such as soil nutrient levels, soil depth, slope steepness, geographical positions of grasslands, in the case of this study (Verbeek, 2012). In addition, the feasibility of the fixed effects model has been tested using statistical tests based on the specific dataset used in each chapter.

1.5 Contribution of the dissertation

In general, given the challenge of grassland degradation and increasing demand on livestock production, this research makes a contribution to providing references for improving the policy interventions toward more sustainable use and management of grasslands. Moreover, this study provides an empirical analysis of SESs of grasslands in China based on the common framework introduced by Ostrom. The research findings can be used in other studies that employ the same framework regarding different countries and different disciplines. This research hopes to increase the knowledge needed to enhance the sustainability of complex SESs of grasslands, as well as the understanding of the transformation of traditional pastoralism.

This research systematically discusses the major policy interventions for the use and management of grasslands in China, and further evaluates their effectiveness for the grassland condition and livestock production. The policy evaluation essentially presents the outcomes of government-driven transformation of traditional pastoralism in China. Although the transformation of pastoralism and the specific effects of the policy interventions on grassland systems have been investigated by governments and scholars (e.g. Li and Zhang, 2009; Li et al., 2014; Hua and Squires, 2015), there is a lack of studies providing quantitative and empirical analysis based on large-scale areas and long-term observation. As such, remote sensing and county-level statistics are combined for this research, the dataset of which involves the extensive grasslands of Inner Mongolia and spans decades. Economic models are employed to evaluate changes in grassland systems, targeting the specific policy interventions. This makes it possible to disentangle the policy effectiveness and control for the potential impacts of climate, socio-economic factors and time trend on the grassland system. This provides an example of the combined use of remote sensing and economics for academic research. In addition, the different preferences of different stakeholders for grassland use under the existing policy intervention is qualified based on utility functions. The specific relevance of each chapter is explained next.

Chapter 2 presents the preference of different stakeholders for land allocation, and thus mirrors the different perceptions on grassland use. The novel contribution is the methodology, which combines welfare economics, the method Analytic Hierarchy Process (AHP) and scenario analysis. Although the outcomes are based on a specific case, the methodology is generic and likely to be valid for measuring conflicts between stakeholders over land use in other regions. The results provide interesting insights that can help to steer future land-use policies considering the different economic and ecological interests of different stakeholders.

Chapter 3 and 4 illustrate the progress of land tenure reform in the pastoral areas of China and investigate the results of privatisation of grasslands on grassland condition and livestock production when other

factors are controlled for. These chapters provide some references to academic debates over the transformation of pastoralism, or grassland privatisation instead of communal tenure. The policy evaluation is conducted based on panel data which considers the spatial and temporal differences of policy implementation. Moreover, the existing literature about grassland privatisation concentrates on impacts on the grassland ecosystem, but there is a lack of research referring to the impact on local livestock production, examined here in Chapter 4. In this regard, despite the unreliability of official statistics on livestock, these research findings are useful for detecting general trends, if not specific numbers.

Chapter 5 summarises the Ecological Construction Programs in pastoral areas of China, and explains why the actual grassland degradation has not been prevented as effectively as expected. Based on the research results, this chapter points out the potential for improving grassland conservation policies. Moreover, it provides the empirical evidence for the scientific debate about the equilibrium model and the non-equilibrium model regarding deterministic or stochastic stocking rates.

1.6 Organization of the dissertation

This dissertation has six chapters, including a general introduction (Chapter 1), four research chapters that focus on the research objective of this dissertation (Chapter 2, 3, 4 and 5), and a general discussion (Chapter 6). The research scheme of the four research chapters is presented in Figure 1.4.

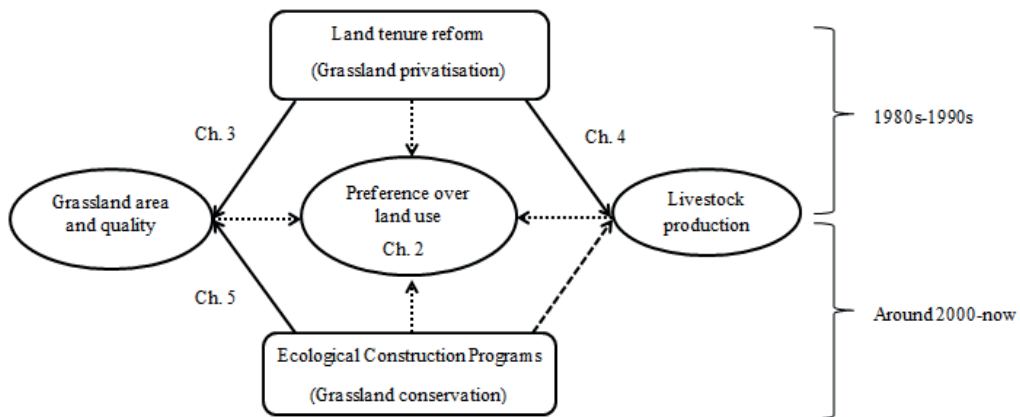


Figure 1.4 The research scheme of the four research chapters

Under the conceptual framework for analysing SESs of grasslands adapted from Ostrom’s framework, the four research chapters elaborate on the outcomes of the governance system over the grassland system and its users. Specifically, Chapters 3 and 4 explore the impacts of land tenure reform (grassland privatisation) on grassland degradation and livestock production, which presents the outcomes of policy

interventions for the grassland system. As previously defined, the grassland system in this research includes grassland conditions as well as livestock production. Grassland degradation is presented by grassland area and quality. Livestock production is indicated by livestock population and meat output. Chapter 5 examines the effectiveness of Ecological Construction Programs on grassland conservation, which presents the outcome of particular rules for the use and management of grassland resource units. In addition, Chapter 2 focuses on investigating the preference over land use by different users under existing social, economic, and political settings. The preference over grassland use is influenced by policies as well as by the outcomes of policy interventions over grassland systems, which is indicated by dashed lines in Figure 1.4. Another significant question marked by the long dashed lines is the effect of Ecological Construction Programs on livestock production, which will not be covered by this research but which is an appropriate candidate for future research. It should be noted that the main period for the implementation of land tenure reform in China is between the 1980s and 1990s, which predates the wide implementation of the Ecological Construction Programs since around 2000. It is assumed that this indication allows us to distinguish these two major policies as well as their main impacts by focusing on different research periods.

Chapter 2

Rural land allocation in eco-fragile regions of northern China

2. RURAL LAND ALLOCATION IN ECO-FRAGILE REGIONS OF NORTHERN CHINA⁷

ABSTRACT: This paper takes a welfare economics perspective to study rural land allocation in eco-fragile regions of northern China and explores the trade-off between stakeholder preferences over the economic and ecological benefits of rural land use. The preference of different stakeholder groups for various types of land is quantified by the Analytic Hierarchy Process, and the individual and social optimum of the allocation of land are investigated using stakeholder utility functions and the social welfare function. Our results show the conflicting views of different stakeholder groups and the potential equilibrium in the allocation of rural land under different scenarios.

KEYWORDS: rural land allocation, individual optima, social optima

⁷ Paper by Min Liu, Wim Heijman, Xueqin Zhu, Liesbeth Dries, and Jikun Huang. *Rangeland Journal*. doi: 10.1071/rj15069

2.1 Introduction

There is increased pressure on rural land use in many parts of the world (Petit, 2009; Williams and Schirmer, 2012). As a result of economic development, technological progress, environmental change and policy and market forces, the total area of rural land is decreasing worldwide (Liu et al., 2010; Verburg et al., 2008). This raises the question of how to efficiently allocate the decreasing amount of rural land, particularly for a populous country such as China. As a basic production factor, rural land is used for production and subsistence to satisfy immediate human needs for food, fuel, and ecosystem services (DeFries et al., 2007; Millennium Ecosystem Assessment, 2003). On the other hand, its multifunctional role has also triggered land use divergence and conflicts because of differing land use targets among stakeholders (Barker and Selman, 1990; Rambonilaza and Dachary-Bernard, 2007; Yang et al., 2012; Pacione, 2013). This paper investigates the optimal allocation of rural land taking into account the ecological and economic benefits for different stakeholders in society.

Changes in rural land use are not only attributed to micro-level household decisions on land use, but are also affected by national or macro-level land use strategies. For example, in China, several national programs have been introduced since the 1990s that attempt to steer rural land allocation in order to satisfy not only human needs for economic development but also the provision of ecosystem services. One of the main programs is the Sloping Land Conversion Program (SLCP, also known as the ‘Grain for Green’ program), initiated by the Chinese central government in 2000 in order to convert sloping cropland into forest or grassland (Liu et al., 2010). Such national programs that are directed at ecological conservation, may constrain the economic activities of local residents, while the majority of rural households in China rely on rural land for their livelihoods and economic benefits. As such, rural land use change associated with economic development and ecological conservation inevitably results in land use conflicts among different stakeholders.

These problems are particularly severe in eco-fragile regions where ecological issues and collective impoverishment are causing serious land use conflicts in the process of rural land use change. This is reflected in overgrazing, poor implementation of eco-environmental policies, deforestation, soil erosion and ecosystem deterioration. One fifth of the territory of China is identified as eco-fragile regions facing such problems (Yuan et al., 2011). In this paper, we therefore focus on land use change and conflicts in the farming-grazing transitional zone in the eco-fragile regions in northern China.

Optimal rural land allocation can be studied either from the macro perspective referring to the global or regional social-economic situation, or from the perspective of land use decisions at the rural household level. The scientific research on land use and land cover change has received increased attention since the 1990s because of prominent problems due to population growth, overuse of natural resources and

environmental deterioration (Turner Li et al., 1993; Qasim et al., 2013). Most research has focused on analysing the land cover dynamics using remote sensing data (Turner Li et al., 1994; Zhan et al., 2007) and the driving forces of land use change based on qualitative analysis, econometrics or game theory (Angelsen, 2001). Others have tried to determine an optimal landscape (Heijman and Mouche, 2013). The current paper will add to the existing literature by using a welfare economics perspective to investigate the equilibrium of rural land allocation, taking into account the interplay between ecological and economic demand. In other words, we will study the optimal rural land allocation based on the trade-offs between economic development and ecological conservation as seen by different stakeholders.

The remainder of this paper is structured as follows. First, we present a conceptual framework of stakeholder interaction in rural land use decisions. Different stakeholder groups are identified based on various land use targets. Next, we derive the theoretical model starting with the utility function and preferences over desired rural land allocation for each stakeholder group. The optimisation of rural land allocation in terms of the benefits to society is explored by deriving the welfare function. This approach is in line with other studies that have modelled environmental problems through welfare analysis (Gerlagh and Keyzer, 2004; Gerlagha and Keyzer, 2003; Zhu, 2004). In the empirical part of the paper, we first describe the data collection methodology. The Analytic Hierarchy Process (AHP) is used to quantify the preference of each stakeholder group for different types of land. The optimal land allocation is derived based on these preferences and different scenarios concerning the weights that are assigned to each stakeholder group in the social welfare function. The paper concludes with a discussion on the differences between existing rural land use policies and the socially optimal allocation as studied in this paper.

2.2 Conceptual framework

To understand the complex relationships in the process of rural land use allocation, we use the concept of social-ecological systems (SEs). SEs represent complex adaptive systems in which social and biophysical agents are interacting at multiple temporal and spatial scales (Levin et al., 2013). Ostrom (2007; 2010) introduced a diagnostic framework for the study of complex social-ecological systems, considering the resource units, resource system, governance system and users as four core subsystems to link social, economic and regulatory settings and related ecosystems (Ostrom, 2009). This has enabled researchers to develop a common language that crosses social and ecological disciplines to analyse how interactions among a variety of factors affect outcomes. On the basis of Ostrom's framework, this paper focuses attention on the influence of the relationship between the governance system and users on the outcome of rural land use. Figure 2.1 demonstrates the analytical framework in this context.

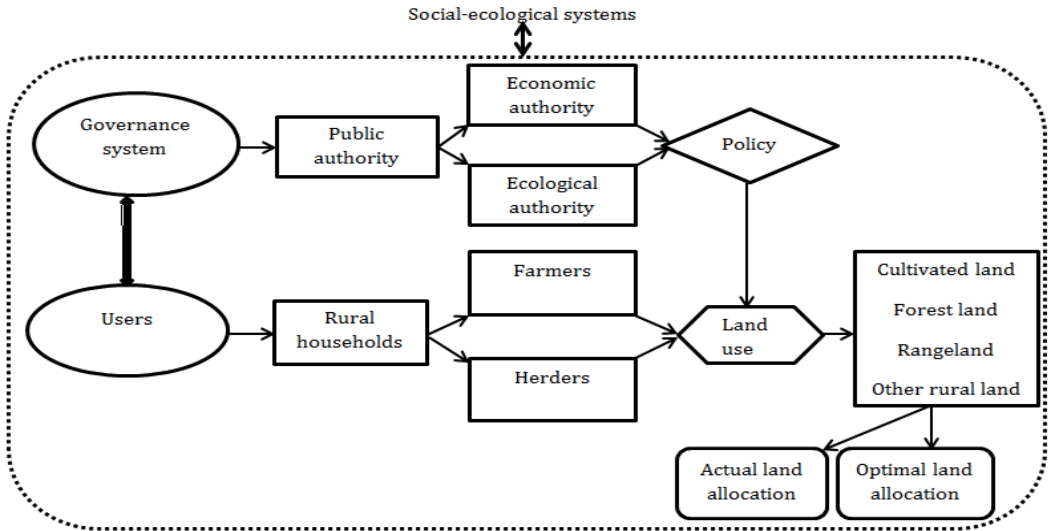


Figure 2.1 The framework for analysing the interaction between different stakeholders⁸

In the process of rural land allocation, the governance system is represented by the rules and regulations set at the level of the government to regulate rural land management for ecological conservation and economic development. Public authorities implement the specific rules related to rural land use. Rural households, as the users of the rural land, are affected directly by these rules in terms of land use decisions for subsistence, production or recreation. In this paper, we assume that public authorities and rural households are the social agents in the social-ecological systems of rural land use. Public authorities are divided into public actors with an economic objective (focusing on economic development) and public actors with an ecological objective (focusing on the conservation of natural resources for current and future generations). This is a simplified representation of reality which assumes that these public objectives can be separated. Among the private stakeholders we distinguish between farmers, who engage in both crop production and animal husbandry, and herders, who are solely dependent on the use of grasslands for livestock production. As such, we divide the social agents into four stakeholder groups, namely herders, farmers, ecological public authority and economic public authority.

In the next sections, we investigate the optimal land allocation based on the trade-offs between ecological conservation and economic development among different stakeholders. Specifically, we analyse the individual and social optimum of rural land allocation. The optimum is based on the different

⁸ We follow the rural land classification by AQSIQ (General Administration of Quality Supervision, Inspection and Quarantine of the People's Republic of China) whereby rural land is divided into cultivated land, forest land, rangeland and other rural land (such as land used for raising animals, agricultural facilities, agricultural roads, pit-ponds, fishponds, irrigation, drying grains and forming ridges among croplands) according to the functions of land use in 2002.

preferences of each stakeholder group as to the rural land value with a combination of diverse land functions, such as various degrees of economic function, ecological function and social function from the cultivated land, rangeland, forest land and other rural land. We describe the role of each stakeholder group in the allocation of rural land as follows: the local ecological public authority aims at protecting the regional ecosystem on behalf of the national government and social ecological demand, the economic public authority achieves the local government's targets for local economic development, and farmers and herders rely on the land for their agricultural production. The herders graze their animals on the rangeland, while the farmers rely primarily on the cultivated land. Section 3 will present the theoretical model which formalizes the conceptual framework.

2.3 Model specification

According to welfare economics, different agents have different demands on the consumption of various goods/services. This has led to the study of individual utility and social welfare optimisation for combinations of different goods characteristics (Heijman and Mouche, 2013; Perman et al., 2011). In our research context, we consider a hypothetical society consisting of four individuals and four goods. More specifically, the four individuals represent the four stakeholder groups (herders, farmers, ecological public authority and economic public authority), and the four sets of goods are cultivated land, rangeland, forest land and other rural land. The four stakeholder groups derive utility from consumption of the four types of land. A Cobb–Douglas form of utility function is used to investigate individual utility and social welfare. The Cobb–Douglas utility function has been applied in previous research to study the optimal consumption, leisure, investment and voluntary retirement problem for an agent (e.g. Koo et al., 2013), the trade-off between goods and leisure (e.g. Train and McFadden, 1978), the ecological discount rate (e.g. Gollier, 2010) and a typical low-rent housing lessees' welfare model (e.g. Wu et al., 2012). It is an appropriate method to investigate the individual and social optimum in utilitarianism, as studied here.

2.3.1 The utility function of each stakeholder group

In this section, the individual optimum is derived through the utility function of each stakeholder group. As with any resource, the allocation of land may be considered optimal when the aggregate social returns from its various uses are maximised (Lopez et al., 1994). Here, we will consider the economic and ecological uses of rural land by each stakeholder group in order to investigate their utility. Considering that the target of stakeholder group i is to maximise its utility (U_i) subject to land constraints, the utility function in the Cobb-Douglas form is,

$$U_i = l_{cul(i)}^{a_i} l_{ran(i)}^{b_i} l_{for(i)}^{c_i} l_{oth(i)}^{d_i}, \quad (1)$$

where $i = 1, 2, 3$ and 4 represent the four stakeholder groups: herders, farmers, ecological public authority and economic public authority, respectively. $l_{cul(i)}$ is the area of the cultivated land which is allocated by stakeholder group i for crop production. $l_{gra(i)}$ is the area of the rangeland allocated by stakeholder group i for grazing animals and fulfilling an ecological protection function. $l_{for(i)}$ is the area of the forest land demanded by stakeholder group i . $l_{oth(i)}$ is the area of the rest of the rural land arranged by stakeholder group i . a_i, b_i, c_i and d_i are the utility elasticities of stakeholder group i on land allocation among the cultivated land, rangeland, forest land and other rural land. The land constraint is as follows:

$$l_{cul(i)} + l_{ran(i)} + l_{for(i)} + l_{oth(i)} = L, \quad (2)$$

$$\text{with } l_{cul(i)}, l_{ran(i)}, l_{for(i)}, l_{oth(i)} \geq 0; 0 \leq a_i, b_i, c_i, d_i \leq 1; a_i + b_i + c_i + d_i = 1.$$

where L is the total area of the rural land in the research region. Equation (2) illustrates that the sum of the cultivated land, rangeland, forest land and other rural land equals the total area of rural land. To find the individual optimum of land allocation, we maximise the utility functions (see Appendix 2-A for the detailed derivation). With the help of the Lagrange optimisation procedure this gives:

$$l_{cul(i)}^0 = a_i \times L, \quad (3)$$

$$l_{ran(i)}^0 = b_i \times L, \quad (4)$$

$$l_{for(i)}^0 = c_i \times L, \quad (5)$$

$$l_{oth(i)}^0 = d_i \times L. \quad (6)$$

$l_{cul(i)}^0, l_{ran(i)}^0, l_{for(i)}^0$ and $l_{oth(i)}^0$ are the resulting individual optima of stakeholder group i , indicating the individually rational optimum depending on the preference of stakeholder group i for rural land use and taking into account land constraints.

2.3.2 The social welfare function

Bergson and Samuelson introduced the social welfare function, which sums up the utility functions of all the individuals in the society (Pollak, 1979). We assume that the four stakeholder groups represent all of the social agents of land use in the eco-fragile regions, therefore our social welfare function is presented by the configuration of the four stakeholder groups' utilities, subject to land constraints. Based on the utility functions of each stakeholder group, the Cobb-Douglas form of the social welfare function is:

$$W = \prod_{i=1}^4 U_i^{\beta_i}, \quad (7)$$

subject to:

$$l_{cul(s)} + l_{ran(s)} + l_{for(s)} + l_{oth(s)} = L, \quad (8)$$

$$\text{with } \sum_{i=1}^4 \beta_i = 1 \text{ and } \beta_i \geq 0, \quad l_{cul(s)}, l_{ran(s)}, l_{for(s)}, l_{oth(s)} \geq 0,$$

where W is the social welfare, $l_{cul(s)}$, $l_{ran(s)}$, $l_{for(s)}$, $l_{oth(s)}$ are the areas of cultivated land, rangeland, forestland and other rural land in the society. U_1, U_2, U_3 and U_4 are the utilities of herders, farmers, ecological public authority and economic public authority respectively based on the social land allocation. β_i is the welfare weight of stakeholder group i in the society.

To find the socially optimal land allocation, we maximise the social welfare function (see Appendix 2-B for the detailed derivation). This gives:

$$l_{cul(s)}^* = L \times \sum_{i=1}^4 \beta_i a_i, \quad (9)$$

$$l_{ran(s)}^* = L \times \sum_{i=1}^4 \beta_i b_i, \quad (10)$$

$$l_{for(s)}^* = L \times \sum_{i=1}^4 \beta_i c_i, \quad (11)$$

$$l_{oth(s)}^* = L \times \sum_{i=1}^4 \beta_i d_i. \quad (12)$$

$l_{cul(s)}^*$, $l_{ran(s)}^*$, $l_{for(s)}^*$ and $l_{oth(s)}^*$ are the resulting socially optimal rural land allocations, which illustrates the optimum for social welfare based on the preferences on land use and welfare weights of each stakeholder group in the society.

2.4 Research region and data collection

An empirical application of the above theoretical model was conducted in Tai Pusi County, located in the eco-fragile region in the mixed farming zone in northern China. Tai Pusi faces both economic backwardness and ecological degradation (Chen, 2007) and is on the southern edge of Otindag Sandy Land, the nearest crucial sand source of sandstorms in Beijing. Its ecosystem plays a significant role in preventing sandstorms from reaching Beijing and various eco-environmental policies with restrictive land use are implemented here. Tai Pusi is one of the poverty-stricken counties of China and two thirds

Figure 2.3 shows the change of the cultivated land, forestland, rangeland and other rural land in Tai Pusi County from 1995 to 2009.

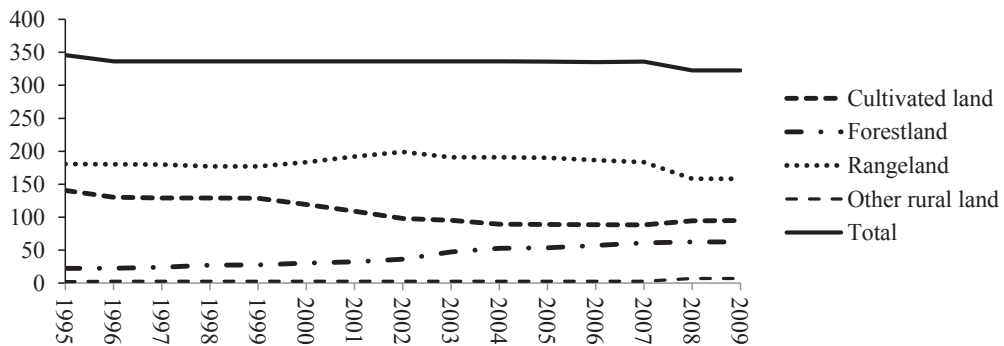


Figure 2.3 The dynamic change of rural land use in Tai Pusi County⁹
Source: Land Resources Bureau, Tai Pusi County.

The largest proportion of rural land has always been represented by rangeland, with cultivated land in second position and other rural land in the last position. In general, rural land use change has occurred, but has had little effect on the order of the dominant land use type. The total area of rural land is decreasing slightly and transitions of land use between cultivated land, rangeland, forest land and other rural land are clearly observable. The area of rangeland decreased rapidly during 2007 and 2008 after a smooth decrease between 2002 and 2006. The area of cultivated land decreased from 1999 until 2008, while conversely, the area of forest land area started increasing from 1999 with a sharp growth in 2002 and 2003.

2.5 Estimation of the model parameters

2.5.1 The parameters a_i , b_i , c_i , and d_i

Parameters a_i , b_i , c_i and d_i are the utility elasticities of stakeholder group i on cultivated land, rangeland, forest land and other rural land. Interviews with key informants and stakeholder representatives were done to assess their preferences. Interviewees included 15 herders, 15 farmers, 6 representatives for the economic public authority and 6 for the ecological public authority¹⁰. From 30 public authorities in charge of local affairs in Tai Pusi County, the Local Finance Bureau and the Farming and Grazing bureau were selected as the public authorities for economic affairs, since they pay more attention to local economic development than other departments. Similarly, the Environmental

⁹ Note that garden plots are included in the category of cultivated land as their area is relatively small in the research region.

¹⁰ We believe that a limited number of interviewees per stakeholder group is sufficient to determine the parameter estimates because the variance in interviewee responses was very small (see Appendix 2-C for details).

Protection Agency and the Forestry Bureau represent the ecological public authorities. Based on the interviews, the Analytic Hierarchy Process (AHP) is employed to evaluate the weight values of a_i , b_i , c_i and d_i (see Appendix 2-C for the different steps in the AHP process). AHP, introduced by Thomas Saaty (1980), is one of the most effective tools for dealing with complex decision-making. A large number of studies in different fields have been published based on AHP, such as planning, resource allocation, conflict resolution and optimisation (Vaidya and Kumar, 2006). The values of parameters a_i , b_i , c_i and d_i that we derived using AHP are shown in Table 2.1.

Table 2.1 Values of parameters a_i , b_i , c_i and d_i

	Cultivated land	Rangeland	Forest land	Other rural land	Total
Herders	0.2084	0.5286	0.1606	0.1024	1
Farmers	0.4317	0.2695	0.2202	0.0786	1
Ecological public authority	0.2121	0.3173	0.3746	0.0960	1
Economic public authority	0.4304	0.3134	0.1154	0.1409	1

2.5.2 The welfare weight (β_i)

The social welfare weight β_i will be calculated for three scenarios.

Scenario 1: β_i is considered equal for all four stakeholder groups. The prevailing practice in choosing social welfare weights is to pick the utilitarian weights (Yang and Nordhaus, 2006). Following this approach we assume that each stakeholder group has the same weight in the social welfare function, i.e. $\beta_1 = \beta_2 = \beta_3 = \beta_4 = 1/4$. This scenario is considered the scenario of social fairness in land allocation.

Scenario 2: β_i is considered as the Negishi weight, introduced by Negishi (1972). In this scenario, the social welfare function is maximized using a weighted sum of individual utilities with constant weights that add up to one across the population. In other words, everyone in the equilibrium of Negishi format will be at the maximum level of utility subject to their initial endowments (Stanton, 2011). Furthermore, Manne and Rutherford (1994) interpreted the equilibrium values of the Negishi weights as the shares in the global endowment of economic resources. Zhu and van Ierland (2006) showed that the Negishi weight is the income share of each region in the total economy if the Cobb-Douglas utility function is used. Here, we use the income share of each stakeholder group in the total income as its Negishi weight β_i . Specifically, the total income of the farmers/herders is the population of farmers/herders multiplied by the average net income of farmers/herders in 2012. Similarly, β_3 and β_4 depend on the income of the economic public authority and ecological public authority, respectively. In terms of the income of the public authorities, a proxy income is employed. That is, assuming that the local government balances

its real income and expenditure, the eco-environmental expenditures on rural land are equivalent to the income of the ecological public authority from the rural land, and the economic expenditures on local rural residents are equivalent to the income of the economic public authority from the rural land.

Scenario 3: β_i is considered as the working population share of stakeholder group i , which is to investigate the optimal land allocation from the perspective of the employment structure in the society.

Values of β_i in the three scenarios are shown in Table 2.2 (see Appendix 2-D for more details).

Table 2.2 Values of β_i in three scenarios

	Scenario 1 (β_i equal weights)	Scenario 2 (β_i income share)	Scenario 3 (β_i population share)
Herders	0.25	0.01	0.03
Farmers	0.25	0.53	0.86
Ecological public authority	0.25	0.04	0.00
Economic public authority	0.25	0.42	0.10
Total	1	1	1

2.6 Results

2.6.1 Land use conflicts among stakeholder groups

We find the local optima of each stakeholder group on land allocation according to the individual preferences and stakeholder utility functions. The actual total area of rural land in 2009 was used as the maximum rural land availability in the study region (322,261 hectares).

Table 2.3 Optimal rural land allocation according to each stakeholder group and actual allocation in 2009 (10^3 hectares)

	Cultivated land	Rangeland	Forest land	Other rural land	Total
Herders	67.2	170.3	51.8	33	322.3
Farmers	139.1	86.8	71.1	25.3	322.3
Ecological public authority	68.4	102.3	120.7	30.9	322.3
Economic public authority	138.7	101	37.2	45.4	322.3
Actual allocation in 2009	94.7	158.1	62.4	7.1	322.3

The results in Table 2.3 present the conflicts among stakeholder groups on rural land use. The herders have a strong preference for rangeland, while farmers have more of an interest in cultivated land and a lesser interest in rangeland. The ecological public authority pays more attention to the rangeland and forest land as they have more of an ecological function. Cultivated land and rangeland are also favoured by the economic public authority. Compared to the actual allocation in 2009 we observe large differences to the local optima. This indicates the differences between the actual allocation and the

aspirations of stakeholders. In the following section, we explore the equilibrium in land allocation based on social welfare optimisation under three scenarios.

2.6.2 The optimal social allocation

According to the three scenarios on welfare weight β_i , we identified the optimal rural land allocation in eco-fragile regions from the social welfare perspective. The results are shown in Table 2.4.

Table 2.4 Socially optimal land allocation under three scenarios and actual allocation in 2009 (10^3 hectares)

Land allocation	Cultivated land	Rangeland	Forest land	Other rural land	Total
Scenario 1 (β_i is equal weight)	103.3	115.1	70.2	33.7	322.3
Scenario 2 (β_i is income share)	135.4	94.3	58.5	34.1	322.3
Scenario 3 (β_i is population share)	136.7	90.9	67.1	27.6	322.3
Optimal range	103.3--136.7	90.9--115.1	58.5--70.2	27.6--34.1	
Actual allocation in 2009	94.7	158.1	62.4	7.1	322.3

Considering the welfare weight as the income share of each stakeholder group (scenario 2), cultivated land has much more weight than the other three types of land, a result which is similar to the one in scenario 3. However, the demand for rangeland is higher than for cultivated land if we give equal weight to every stakeholder group (scenario 1), and also the demand for forest land is larger than in the other scenarios. The optimal range is based on the minimum and maximum in the scenarios as the lower and upper bound, in which the highest demand by the society is on the cultivated land and rangeland, followed by forest land, and finally other rural land. Comparing the actual allocation in 2009 with the optimal range, all types of rural land are out of the optimal range except the forest land. In addition, the mean of the optimal range is highest for cultivated land while the actual allocation tends most towards rangeland.

2.6.3 Comparison of individual optimum, social optimum and actual land allocation

Table 2.5 presents an overview of the above results. It shows the individual optimum for the four stakeholder groups, the social optimum and the actual situation in rural land allocation, all as the share of each rural land type in total rural land. Comparing the results within each type of land, we see that the individual optimal shares of cultivated land are outside of the socially optimal range. Herders and the ecological public authority wish to allocate less land for cultivation, while farmers and the economic public authority want to allocate more land for cultivation. Preferences for rangeland are largely within the socially optimal range except for farmers who would like to see less rangeland and herders who

prefer a much larger share of rangeland than what is socially optimal. Unsurprisingly, the ecological public authority would want to allocate much more land for forestry. The actual land allocation in the research region is outside of the socially optimal range for each of the land types except for forest land. Land allocation based on social optimisation would increase the area of cultivated land and decrease the area of rangeland.

Table 2.5 Shares of rural land allocated to cultivated land, rangeland, forest land and other rural land in different situations

	Cultivated land	Rangeland	Forest land	Other rural land	Total
Herders optimal allocation	20.9%	52.8%	16.1%	10.2%	100%
Farmers optimal allocation	43.2%	26.9%	22.0%	7.8%	100%
Ecological public authority optimal allocation	21.2%	31.7%	37.4%	9.6%	100%
Economic public authority optimal allocation	43.0%	31.3%	11.5%	14.1%	100%
The socially optimal range	32.1%--42.4%	28.2%--35.7%	18.2%--21.8%	8.6%--10.6%	
Actual share in 2009	29.4%	49.1%	19.4%	2.2%	100%

2.7 Conclusion

This paper takes a welfare economics perspective to investigate current land use conflicts and socially optimal land allocation. Optimisation based on individual utility functions and AHP reveals land use conflicts among stakeholders due to heterogeneous preferences for different types of land. By optimising the social welfare function, using different welfare weights, we derive the socially optimal range of land allocation. Comparing our empirical results with current land allocation over different land types, we find that a socially optimal land allocation would require a shift towards more cultivated land and less rangeland.

These results should be put in the Chinese policy context. Since the 1990s, the Chinese government has shown increased interest in extending the area of forest land and rangeland for ecosystem protection purposes. Programs such as the SLCPP that aim to convert sloping cropland into forestland or grassland are witness to this policy direction (Liu et al., 2010). However, our results show that the social optimum land allocation requires conversion in the other direction: from grasslands towards cultivated land, while the area of forest land is already within the socially optimal range. The preference for cultivated land over rangeland may be driven by the policy restrictions on rangeland. Farmers can use cultivated land more freely than rangeland. While this insight might seem controversial in view of the current policy

context, our results could also be driven by the overall pressure on rural land, and cultivated land in particular, in China. This pressure is likely to be especially strong in the eco-fragile regions where most of the local residents depend on farming, but where agricultural productivity falls far behind the national average level. This promotes the demand for more cultivated land and land for agricultural infrastructure.

Furthermore, our results may also reflect the misalignment between preferences of the national government – as included in the national policy programs that focus on environmental protection – and preferences of local governments – whose performance evaluation is based primarily on economic growth indicators (Liu and Diamond, 2005). Moreover, it is worth mentioning that a decrease in the area of grasslands in favour of cultivated land does not necessarily conflict with environmental objectives. First, research on the effectiveness of current grassland protection policies has not yet provided conclusive evidence (Zheng et al., 2006; Cao et al., 2013). Second, a conversion to more cultivated land is likely to lead to a more intensified use of the rural land. As a result, this may reduce the pressure on the grasslands as rural residents can sustain their livelihoods through cultivation rather than by increasing the number of animals on the grasslands.

Finally, we would like to point out a number of limitations of our research. While the research method is generic and likely to be valid for other regions facing land use conflicts, the specific outcomes presented in this paper are based on stakeholder interviews in one Chinese county and are therefore not generalizable. Furthermore, the application of the model includes only 4 different categories of land and 4 stakeholder groups which is a simplification of reality. Finally, the analysis only assesses the distribution of land resources and hence abstracts from the allocation of other relevant resources of rural households.

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Chapter 3

Land tenure reform and grassland degradation in Inner Mongolia, China

3. LAND TENURE REFORM AND GRASSLAND DEGRADATION IN INNER MONGOLIA, CHINA¹¹

ABSTRACT: Since the start of the land tenure reform in the pastoral areas of China in the 1980s, grassland use rights have increasingly been assigned to individual households and subsequently more grasslands have been in private use. However, in the same period, most of the grasslands in China have experienced degradation. The question that this paper tries to address is whether the land tenure reform plays a significant role in grassland degradation. It is answered by an empirical analysis of the impact of land tenure reform on the changes in grassland condition, using data from 60 counties in Inner Mongolia between 1985 and 2008. Grassland condition is presented by grassland quantity and quality using spatial information based on remote sensing. The timing of the assignment of grassland use rights and the timing of the actual adoption of private use by households differ among counties. These timing differences and differences in grassland condition among counties allow disentangling the impact of the land tenure reform. A fixed effects model is used to control for climate, agricultural activity and the time-invariant heterogeneity among counties. The model results show that the private use of grasslands following the land tenure reform has had significantly negative effects on grassland quality and quantity in Inner Mongolia. Moreover, the negative effects did not disappear even after several years of experience with private use. In conclusion, our analysis reveals that the land tenure reform, namely privatisation of grassland use rights, is a significant driver of grassland degradation in Inner Mongolia in a long term, which presents “a tragedy of privatisation”, as opposed to the well-known “tragedy of the commons”.

KEYWORDS: privatisation, formal use rights, private use, grassland degradation, fixed effects model

¹¹ Paper by Min Liu, Liesbeth Dries, Wim Heijman, Jikun Huang, Xueqin Zhu, and Xiangzheng Deng, submitted to a peer-reviewed journal.

3.1 Introduction

The tragedy of the commons, a concept introduced by Hardin (1968), has long been part of the conventional wisdom in ecology, economics and political science (McEvoy, 1987). Theoretical research contended that resource degradation was inevitable unless the commons were either privatised or maintained as common land with clear rights of entry and use (Hardin, 1968). Similarly, in the theorem of property rights (Coase, 1960), a clear assignment of property rights is proposed as a precondition for economically efficient resource allocation and environmental sustainability. As a consequence, the privatisation of property rights or use rights over resources has been implemented widely by governments around the world (Little, 1992; Lesorogol, 2003; Squires et al., 2009; Ybarra, 2009). The same privatisation trend has been observed in Inner Asia and Africa regarding the reform of grassland tenure (Sneath, 1998; Lesorogol, 2008).

However, Hardin's theory of the tragedy of the commons and Coase's theorem of property rights have been rejected by pastoral specialists, who found that these theoretical findings provided a poor guide to understanding pastoralism (Feeny et al., 1990; Behnke, 1994; Sneath, 1998). They concluded that exclusionary land tenure is counterproductive to protect the grassland condition in arid and semi-arid areas (Turner, 1999; Galvin, 2009). And some scholars (e.g. Ostrom, 1990; Li and Zhang, 2009) found that traditional common use systems of grasslands often met the criteria for sustainable Social-Ecological Systems (Ostrom, 2009), namely the exclusion of outsiders and the self-organization of resource users (Ostrom, 1990). Therefore, doubts exist about the likelihood that traditional common use systems would induce a tragedy of the commons. Moreover, a growing number of academic studies question the effects of the prevailing privatisation of grassland property rights or use rights on the grassland condition. For instance, some studies assert that grassland privatisation puts an end to mobile pastoralism by introducing inflexible boundaries and identify this as a main driver for grassland degradation (e.g. Fernandez-Gimenez and Le Febre, 2006; Li and Zhang, 2009). Mobile pastoralism is embraced because mobility maintains ecosystem stability and biological diversity and conforms with the sustainable use of natural resources that are subject to strong spatial and temporal heterogeneity, as is the case in arid and semi-arid areas (Fernandez-Gimenez, 2002; Fernandez-Gimenez and Le Febre, 2006; Tessema et al., 2014; Fan et al., 2015). Empirical evidence was provided by Sneath (1998) who used satellite images to show that grassland degradation was much less severe in Mongolia – which kept the common use system for grasslands – than in the Russian and Chinese parts of Inner Asia where the privatisation of grassland property rights or use rights was implemented. Furthermore, Guelke (2003) showed that grassland degradation in South Africa was primarily due to the introduction of privatisation.

In China, land tenure reform started with the Household Production Responsibility System in the cropland areas in the late 1970s and spread to the pastoral areas by the early 1980s. The land tenure

reform in the pastoral areas aims at privatising use rights of grasslands to individual households and displacing traditional common use (Banks et al., 2003). Some differences have been observed in the tenure reform of grasslands among the provinces. For instance, the grasslands tenure reform started in 1984 in Inner Mongolia and Heilongjiang while it had started already in 1983 in Ningxia (Ho, 2000). Furthermore, the privatisation of grassland use rights in some provinces was ahead of the national decision about the tenure reform of grasslands, as the first Grassland Law of China was introduced only in 1985. Moreover, it is reported that most of the grasslands in China have experienced various degrees of degradation since the 1980s (Li et al., 2014). The manifestation of the grassland degradation involves a reduction in the extent of grass cover, a reduction in the density of grass cover, a reduction in the output of forage material, an increase in unpalatable grass species, an increase in soil compaction, changes in plant functional group or a combination of all of them (Feng et al., 2009; Li et al., 2013; Lin et al., 2015). According to Chinese governmental reports, 10% of the total area of grasslands was degraded in the 1970s, increasing to 30% in the 1980s and 50% in the middle of the 1990s (Meng and Gao, 2002). By the 2000s, about 90% of the grasslands were degraded to various extents, with significant regional variation (Unkovich and Nan, 2008; Waldron et al., 2010). It is widely agreed that grassland degradation is one of the most important environmental issues confronting China presently (Han et al., 2008; Feng et al., 2009). As efficient policy interventions derive from correct understandings about the causes of grassland degradation (Brown et al., 2008), and good management practices are capable of improving the grassland condition significantly (Parras-Alcántara et al. 2015), a relevant question is what role the land tenure reform has played in the process of grassland degradation.

Although some studies exist that discuss the impact of the land tenure reform on the grassland condition in China (e.g. Ho, 2001; Li and Zhang, 2009), there is a lack of empirical analysis based on long-term data on a large-scale (Lu et al., 2015). Therefore, this paper uses information on the changes in grassland condition and progress of the land tenure reform in a typical pastoral area in China over a relatively long period (1985-2008) to estimate the effect of land tenure reform on grassland condition. In the following sections, we first provide some background information about the grassland tenure reform in China, including the progress in the assignment of grassland use rights and the actual use of grasslands. Next, the materials of our empirical research are described and the fixed effects model in our case is presented. We conclude with some remarks about the effects of the land tenure reform on the grassland condition in the research region and implications of our findings for future research.

3.2 Materials

3.2.1 Study area

To explore the effect of land tenure reform on the grassland condition in the pastoral areas of China, we selected Inner Mongolia, a province located in northern China, to conduct our empirical study. Inner Mongolia belongs to the arid and semi-arid areas of China that is vulnerable to degradation, desertification and salinization (Feng et al., 2009). It contains 21.7% of the area of China's natural grasslands. Approximately 67% of the total land in Inner Mongolia is classified as grassland, the majority of which is temperate grassland (Angerer et al., 2008). As a traditionally pastoral region populated by Mongolians, the vast majority of local people maintain their livelihoods upon the permanent grasslands and the region has a relatively low population density in spite of rapid economic growth and a soaring population growth elsewhere in China (Angerer et al., 2008). The grassland condition in Inner Mongolia has been a major concern because the region is not only one of the main production regions for animal products of China (Li et al., 2016), it also functions as an ecological barrier for northern China. For instance, the colossal dust storms which rumbled through hundreds of cities and villages of northern China and blanketed the sky of Beijing between 1998 and 2001, are said to have originated from dryland areas and degraded grasslands mainly in Inner Mongolia (Wu et al., 2015). In terms of its status as main region for livestock production in China, the mutton output of Inner Mongolia accounted for 22% of the total output of China in 2013, while 8% of all Chinese beef was produced here (China Animal Industry Yearbook, 2014). Therefore, continued grassland degradation in Inner Mongolia is supposed to harm economic development as well as cause ecosystem instability in China (Meyer, 2006). By the end of the twentieth century, 90% of the grassland condition of Inner Mongolia had been degraded to various extents (Mei et al., 2013), which has stimulated growing attention to the protection of grasslands in recent decades.

Inner Mongolia was the first region for implementation of the land tenure reform in the grassland area (Li and Huntsinger, 2011). According to provincial government reports for Inner Mongolia, the privatisation of grassland use rights began officially through the Double Contracts System in 1982. Because of its limited uptake, the implementation of grassland privatisation was strengthened by the Two Rights and One System policy (Shuang Quan Yi Zhi) in 1996 (Bureau of Animal husbandry of Inner Mongolia, 2000; Li and Zhang, 2009). This is regarded as the second round of privatisation of grassland use rights. To this date, local governments continue to work on clarifying the boundaries of grassland use rights for each household and on issuing certificates (Bureau of Animal husbandry of Inner Mongolia, 2015). In practice, the specific timing and extent of conducting the land tenure reform differed among counties in Inner Mongolia. This paper is therefore based on county-level data with the

aim of disentangling the effects of privatisation on the grassland condition. Figure 3.1 presents the region of Inner Mongolia and marks the counties that are included in our empirical analysis.

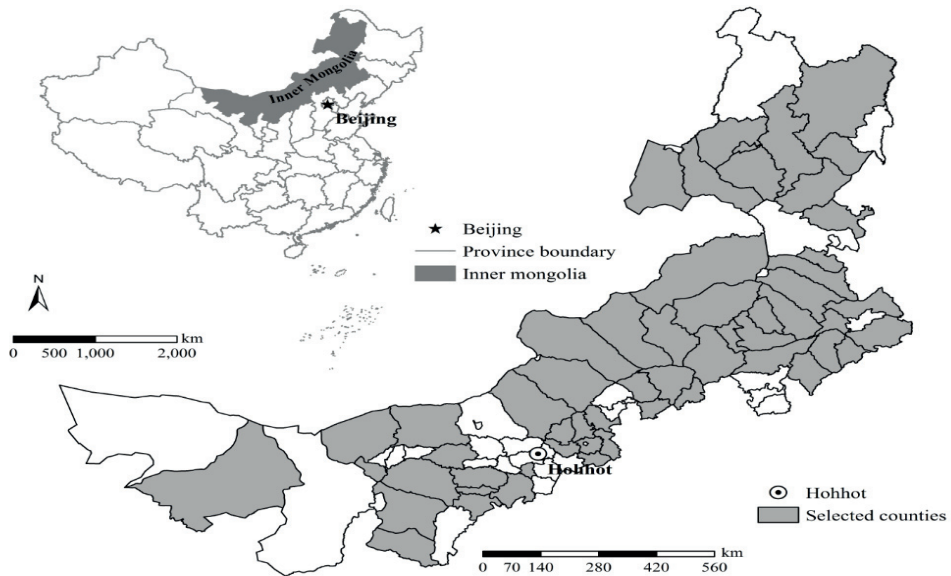


Figure 3.1 Inner Mongolia and the selected counties in this study

3.2.2 Data sources

County-level data were collected for the period 1985-2008, the main period during which the land tenure reform was implemented across Inner Mongolia. Information about the land tenure reform was collected through questionnaires. The questionnaires were sent to the Animal Husbandry Bureau of each county and answered by local experts on the grassland tenure reform. In a second phase, interviews were conducted by telephone to confirm the answers that were received. A total of 74 valid questionnaires were obtained out of 102 counties of Inner Mongolia. Besides the questionnaires, we had interviews with the officers who are working in the provincial institutes of Animal Husbandry of Inner Mongolia and with local herders to verify the progress of land tenure reform on local grasslands.

In the existing literature, grassland condition is indicated by grass cover, height, density, biomass production or density of perennial vegetation (Gu and Li, 2013; Yu and Farrell, 2013). Commonly used methods to measure these indicators include small-scale sampling tests in field studies and data analysis with remote sensing and GIS (Geographic Information System) over large areas. The former focuses on measuring the specific indices of grass production and vegetation diversity directly, while the latter estimates the general grassland condition based on satellite images (Gu and Li, 2013). In this study, the method with remote sensing and GIS is more appropriate because we aim to quantify the spatial and

temporal differences in grassland condition across extensive grasslands during a long period. In this regard, our information on grasslands and climate conditions relies on a land use database developed by the Chinese Academy of Sciences with original data from Landsat Thematic Mapper / Enhanced Thematic Mapper (Plus) (TM/ETM+) images which have a spatial resolution of 30*30m (Deng et al. 2011). Visual interpretation and digitization of Landsat TM/ETM+ images were conducted to generate land use maps for the years 1985, 1995, 2000, 2005 and 2008. Land cover classification system was applied to distinguish grasslands from cultivated land, forestry area, water area, built-up area, and unused land. According to the grass cover, grasslands were further classified into three subclasses which were adopted from the research by Feng et al. (2009) and Deng et al. (2011), including dense grassland, moderate grassland and sparse grassland. In addition, socio-economic information was derived from statistical data collected by local governments. After dropping the urbanised counties - where changes in the grassland area can be attributed mainly to changes in land use types - and those counties that experienced changes in administrative regions, 60 counties out of 74 counties with valid questionnaires are retained as the research sample for this study, as marked in Figure 3.1.

3.2.3 Land tenure reform for the grasslands

3.2.3.1 A review of the legislation on grassland tenure of China

In the collectivist period of China, grasslands were assigned to production teams or People's communes and the grasslands were managed and used communally by all households in the same production team (Hua and Squires, 2015; Taylor, 2006). After that period, given the successful experience of privatising land use rights to individual households in the cropland regions in the late 1970s, the Chinese central government introduced the household-based privatisation of land use rights into the pastoral areas in the early 1980s. Since then, the management of grasslands has increasingly become the responsibility of individual households (Banks et al., 2003).

The first Grassland Law of China was initiated in 1985 and stipulated that the property rights over grasslands are owned either by the state or the collective but that use rights to the grasslands can be contracted out to either households or collectives (Ho, 2000). According to the current Law on the Contracting of Rural Land, the grassland use rights can be contracted to individual households for a duration of 30 to 50 years. In practice, local governments started to assign grassland use rights to households through a Double Contracting System ("cao xu shuang cheng bao") in the 1980s. In 2002, the revised Grassland Law reaffirmed the devolution of land use rights from the state and collectives to individual households (National People's Congress of the People's Republic of China, 2002). The State Council emphasised the specific allocation of the contracted items (e.g. grassland plots, areas, issuing user-right certificates etc.) to households to ensure the long-term stability of contract relations in a

government report in 2011 (Li, 2012). With these legislative efforts, the privatisation of the grasslands has been conducted gradually in the extensive pastoral areas of China, and increasingly private use by households has replaced traditional open access to land (Hua and Squires, 2015; Yang et al., 2004). By 2014, the total area of grasslands that have been assigned to individual households reached 223 million hectares, out of 400 million hectares of grasslands in China (Ministry of Agriculture of China, 2015).

3.2.3.2 The implementation of the land tenure reform on grasslands

Table 3.1 provides a typology of the assignment of use rights and the actual ways of grassland use in the pastoral areas of China since the start of the land tenure reform. This typology is in line with the investigation of land tenure reform on the Tibetan plateau by Banks (2003). Three types of assignment of grassland use rights existed during the last three decades, namely, formal use rights assigned to individual households, to a group of households or to a collective. Correspondingly, there were three ways of grassland use, including private use, joint use and common use. Private use indicates that a certain area of grasslands is managed and used by an individual household, while other users are excluded. Joint use means that a group of households manage and use an area of grasslands together, in which group members are often neighbours or relatives that have small parcels of grasslands and wish to make their land larger and contiguous. Finally, common use means that the grasslands within a collective or administrative village are used by all of the members of the collective or the villagers.

Table 3.1 The assignment of formal use rights of grasslands and the actual ways of grassland use

Formal use rights assigned to	Actual use of grasslands		
	Private use	Joint use	Common use
Individual households	+	+	+
Group of households	na	+	+
Collective/administrative village	na	na	+

Source: adapted from “formal and de facto grassland management units” (Banks et al., 2003)
na indicates not available.

When the use rights are assigned to a collective or an administrative village, the actual way of grassland use is always common use. If use rights are assigned to a group of households, actual use can be common use or joint use. Private use, joint use and common use can all occur when the use rights are assigned to individual households. The intention of the land tenure reform in the pastoral areas is, however, to achieve full private use when use rights are assigned to households. This has spurred criticisms and claims of failure of the privatisation process in China's extensive grasslands, compared to the case of the cropland areas where household-based use rights were assigned overnight and private use was established simultaneously (Ho, 2000; Banks et al., 2003). Scholars (e.g. Richard et al, 2006; Li and

Zhang, 2009; Li, 2012) have identified a number of possible barriers to the completion of the land tenure reform in the pastoral areas of China: (1) the inflexible boundaries stemming from the privatisation of grassland use rights is incompatible with the need for livestock mobility due to the heterogeneity of resources in arid and semi-arid areas (Li and Zhang, 2009); (2) there is a lack of financial resources for protecting private property rights, e.g. to build fences (Ho, 1996; Li et al., 2007); (3) individual household ownership is inconsistent with local or traditional knowledge about property or use rights (Richard et al, 2006); and (4) private use by individual households destroys the traditional institutions of grassland use and management (Li, 2012). In what follows, we will investigate another possible explanation for why the adoption of private use is lagging, namely that privatisation negatively affects the grassland condition.

3.2.3.3 Land tenure reform in sample counties

Table 3.2 gives indications of the land tenure reform progress of grasslands in the selected counties between 1985 and 2008. In 1985, only 6.67% of the 60 counties had assigned grassland use rights to individual households. By 2008, this proportion had increased to 81.67%, and the average number of years since formal use rights had been contracted to individual households was 11.55. Meanwhile, actual changes in the use of the grasslands followed this institutional evolution. In 1985, only 5% of the selected counties had adopted the practice of private use. In 2008, this percentage had reached to 63.33% and the average number of years since private use had been adopted was 8.72. These observations about the land tenure reform in the sample counties demonstrate that the adoption of private use lagged behind the privatisation of formal use rights.

Table 3.2 Land tenure reform progress in the selected counties, 1985-2008

	Unit	1985	1995	2000	2005	2008
The proportion of counties in which grassland use rights have been assigned to individual households	Percent	6.67	23.33	76.67	80.00	81.67
The average number of years since the use rights have been assigned to households	Years	0.18	1.92	5.17	9.13	11.55
The proportion of counties with actual private use of grasslands	Percent	5.00	15.00	58.33	63.33	63.33
The average number of years since actual private use has been adopted	Years	0.13	1.22	3.68	6.82	8.72

3.3 Empirical model

3.3.1 Fixed effects model

The fixed effects model has been widely used in economic research (e.g., Fergusson et al., 2002; Huang et al., 2006). It is used primarily to study the causes of changes within entities over time. The general model is presented as follows:

$$Y_{it} = \beta X_{it} + U_i + \varphi_{it} \quad (1)$$

where i and t present entity i and year t , respectively. Y_{it} is the dependent variable. X_{it} denotes the vector of independent variables. U_i is the specific intercept for each entity and represents all time-invariant (fixed) characteristics of entity i that influence the dependent variable, including both observable and unobservable factors. φ_{it} is the random error term. The difficulty in estimating the parameters of β is that U_i involves unobserved factors. In this regard, a within transformation is used to remove all time-invariant (fixed) explanatory variables, i.e. a model is performed using deviations from individual means (Verbeek, 2012). Therefore, the fixed effects model is a method of estimating the parameter of β that takes the observable as well as unobservable time-invariant explanatory variables (U_i) into account, but the estimation does not depend on the value of U_i (Fergusson et al., 2002).

The fixed effects model is appropriate to solve our research question considering that the results of the land tenure reform within each county can be studied effectively by controlling for the unmeasured heterogeneity among counties. As shown in table 3.2, the progress in land tenure reform changes substantially across our research period and for a considerable proportion of the counties in our dataset, which satisfies the basic requirement of the fixed effects model (Daun-Barnett, 2011). Moreover, our research sample attempts to include all of the relevant counties in Inner Mongolia, rather than random draws, which preliminarily indicates the fixed effects model more appropriate than the random effects model (Verbeek, 2012). Nevertheless, the appropriateness of the fixed effects model will be further examined through statistical testing in section 5.

3.3.2 Variable definitions and expected results

The dependent variable in the empirical analysis is the grassland condition. The grassland condition includes the situations of both quantity and quality of grasslands, which are quantified considering the extent and density of grass cover that are suggested as the crucial manifestation of grassland degradation (Feng et al., 2009; Li et al., 2013). Specifically, the quantity of grasslands is measured by the total area of grasslands in hectares, while the areas with less than 5 percent of grass cover are not counted as grasslands. And the quality of grasslands is presented through the areas of grasslands of different subclasses that are classified according to the different density of grass cover, including dense grassland,

moderate grassland and sparse grassland. According to the research by Feng et al. (2009) and Deng et al. (2011), dense grassland has the highest quality and its grass cover is more than 50 percent. Moderate grassland has grass covering 20-50 percent of the land, and sparse grassland with 5-20 percent. In our case, it is supposed that the grassland degradation is manifested by the decrease of total, dense and moderate grassland areas, as well as the increase of sparse grassland area when total grassland area with decrease or no changes. The latter considers that the dense and moderate grasslands may degrade into sparse grassland, which was observed by the research about the moving direction of grassland centre of gravity in China during 1976-1996 by Feng et al. (2009). Hence, four dependent variables are included in the empirical analysis to indicate the grassland condition: total grassland area, dense grassland area, moderate grassland area, and sparse grassland area. The following equations for estimation are derived from the general fixed effects model (see equation (1)):

$$G_{it} = g_i + a_1L_{it} + a_2L_{it}^2 + a_3T_{it} + a_4T_{it}^2 + a_5F_{it} + a_6F_{it}^2 + a_7TF_{it} + a_8C_{it} + a_9C_{it}^2 + \varepsilon_{it} \quad (2)$$

$$D_{it} = d_i + b_1L_{it} + b_2L_{it}^2 + b_3T_{it} + b_4T_{it}^2 + b_5F_{it} + b_6F_{it}^2 + b_7TF_{it} + b_8C_{it} + b_9C_{it}^2 + \mu_{it} \quad (3)$$

$$M_{it} = m_i + c_1L_{it} + c_2L_{it}^2 + c_3T_{it} + c_4T_{it}^2 + c_5F_{it} + c_6F_{it}^2 + c_7TF_{it} + c_8C_{it} + c_9C_{it}^2 + \gamma_{it} \quad (4)$$

$$S_{it} = s_i + d_1L_{it} + d_2L_{it}^2 + d_3T_{it} + d_4T_{it}^2 + d_5F_{it} + d_6F_{it}^2 + d_7TF_{it} + d_8C_{it} + d_9C_{it}^2 + \sigma_{it} \quad (5)$$

where i and t present the i^{th} county and year t . Total grassland area (G_{it}), dense grassland area (D_{it}), moderate grassland area (M_{it}) and sparse grassland area (S_{it}) are the dependent variables. Equations (2), (3), (4) and (5) will be referred to as the total, the dense, the moderate and the sparse grassland model, respectively.

L_{it} indicates our main variable of interest, the progress of the land tenure reform in county i until year t . It is measured as the number of years until year t that private use has been adopted in county i . We focus on the years of actual private use of the grasslands rather than the years of the assignment of formal use rights to households because it is considered that the actual use will affect the grassland condition. As was shown in table 3.2, the adoption of actual private use lags behind the assignment of formal use rights, and the correlation value between the years of assigning formal use rights and actually adopting private use is 0.825. In this regard, including both the years of formal use rights assignment and the years of actual private use as variables in the model would cause multicollinearity. The quadratic term of L_{it} , L_{it}^2 , is included in the model because a non-linear relationship is expected between the land tenure reform and the grassland condition.

To correctly estimate the effect of the land tenure reform, we control for other factors that may affect the condition of the grasslands, namely, the factors related to climate (Gao et al., 2010; Li et al., 2012;

Lu et al., 2015), agricultural activities (He et al., 2014; Hua and Squires, 2015) and heterogeneity among counties (Deng et al. 2011; He et al., 2014). Climate factors are presented by temperature (T_{it}) and precipitation (F_{it}) (Gong et al., 2015). Their quadratic terms (T_{it}^2 and F_{it}^2) and cross term (TF_{it}) are considered (Li et al., 2013). C_{it} is the percentage change in cultivated land in county i in the five years preceding year t and C_{it}^2 is its quadratic term. C_{it} and C_{it}^2 are considered as controls for the switch of land between grassland and arable land which was shown to have a significant and direct effect on grassland quantity in China (Feng et al., 2009; Li et al., 2013). Finally, the factors about the heterogeneity among counties such as elevation, slope, soil type and distance to the provincial capital do not change significantly over time and are therefore treated as time-invariant (fixed) factors in the model, denoted by g_i , d_i , m_i and s_i in equations (2) to (5). Variable definitions and the expected impact of independent variables on the grassland condition are listed in table 3.3.

Table 3.3 Variable definitions and their expected results

Variables	Variable definition	Unit	Expected results			
			Total grassland model	Dense grassland model	Moderate grassland model	Sparse grassland model
G_{it}	Total grassland area of county i in year t	Hectares	Dependent variable			
D_{it}	Dense grassland area of county i in year t	Hectares		Dependent variable		
M_{it}	Moderate grassland area of county i in year t	Hectares			Dependent variable	
S_{it}	Sparse grassland area of county i in year t	Hectares				Dependent variable
L_{it}	The number of years until year t that actual private use of grasslands has been adopted in county i	Years	+/-	+/-	+/-	+/-
L_{it}^2	Quadratic term of L_{it}	Years ²	-/+	-/+	-/+	-/+
T_{it}	The annual average temperature in county i in year t	Celsius degree	+/-	+/-	+/-	+/-
T_{it}^2	Quadratic term of T_{it}	Celsius degree ²	-/+	-/+	-/+	-/+
F_{it}	The annual precipitation in county i in year t	Millimeter	+	+	+	-
F_{it}^2	Quadratic term of F_{it}	Millimeter ²	-	-	-	+
TF_{it}	Interaction term of T_{it} and F_{it}	Celsius degree* Millimeter	+/-	+/-	+/-	+/-
C_{it}	The average percentage change in the area of cultivated land in county i in the five years preceding year t	Percentage	-	-	-	+
C_{it}^2	Quadratic term of C_{it}	Percentage ²	+	+	+	-

Given the contradicting views in the literature about the effect of privatisation on grassland quality, the expected effect of L_{it} in our estimations is ambiguous (e.g. Hardin, 1968; Guelke, 2003; Galvin, 2009). However, we expect that the effects of the land tenure reform will be strong at the beginning and reduce in intensity over the years. Hence an opposite sign to that of L_{it} is expected for the coefficients of L_{it}^2 in the four models. It is noted that the expected effects of all explanatory variables in the sparse grassland model are the opposite of the expectations in the other three models because grassland degradation involves a decrease in total, dense and moderate grassland areas, but an increase in sparse grassland area. Additionally, the expected sign of temperature (T_{it}) is ambiguous because higher temperatures facilitate plant growth, but also cause fast evaporation from ground surfaces and stimulate the development of pests (Deng et al., 2013; Li et al., 2013). Precipitation (F_{it}) improves the grassland condition in the arid and semi-arid areas (Li and Zhang, 2009). Both temperature and precipitation are expected to have nonlinear relationships with the grassland condition (Li et al., 2012; Zhou et al., 2002), and therefore the opposite signs to those of T_{it} and F_{it} are expected for the coefficients of T_{it}^2 and F_{it}^2 in the four models. The interaction term of T_{it} and F_{it} is also included because the effect of temperature depends on the amount of precipitation. Finally, the intensive agricultural activities were suggested to cause grassland degradation (Zhang et al., 2007; Feng et al., 2009). We therefore expect negative coefficients for C_{it} in the total, dense and moderate grassland models, and a positive coefficient in the sparse grassland model. Again a nonlinear relationship is expected for this variable.

3.3.3 Descriptive statistics on the variables

Table 3.4 shows the mean value and standard deviation of our main variables for the research sample. In general, the areas of total grassland (G_{it}), dense grassland (D_{it}) and moderate grassland (M_{it}) decreased, and the area of sparse grassland (S_{it}) presented fluctuant increases between 1985 and 2008. The areas of total, dense and moderate grasslands experienced their highest points in 1985, and in 2000 for sparse grassland area. Specifically comparing 1985 and 2008, the total grassland area decreased which indicates the degradation of grassland quantity. Furthermore, given the decrease in total grassland area, the decrease in dense and moderate grassland areas and increase in sparse grassland area demonstrate the degradation of grasslands with higher grass cover into lower quality grasslands. Moreover, the average number of years with actual private use in the 60 sample counties was increasing from 0.13 in 1985 to 8.72 in 2008. And the standard deviations of L_{it} indicate that the differences in adopting private use among the 60 sample counties were increasing from 1985 to 2008. The average temperature reached its highest point in 2008, while annual precipitation was the highest in 1985 and the lowest in 2000. The average percentage change in cultivated land increased from 1985 to 2000 then decreased to become even negative in 2005.

Table 3.4 Descriptive statistics of main variables

Variables	Obs	Unit	1985			2000			2005			2008		
			Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.	Mean	Std. Dev.		
G_{it}	60	Hectares	778,779	810,035	772,780	815,044	762,376	808,624	759,026	806,050	761,399	810,785		
D_{it}	60	Hectares	380,860	517,236	377,815	520,029	364,280	514,489	365,300	513,908	365,027	513,817		
M_{it}	60	Hectares	278,432	378,658	277,557	380,816	275,346	374,282	272,747	368,618	272,815	368,679		
S_{it}	60	Hectares	120,627	160,957	118,894	160,332	123,152	162,608	121,057	158,800	122,126	160,195		
L_{it}	60	Years	0.13	0.60	1.22	3.24	3.68	4.69	6.82	6.47	8.72	7.69		
T_{it}	60	Celsius degree	3.31	2.86	4.45	2.55	4.52	2.98	4.74	2.80	5.20	2.58		
F_{it}	60	Millimeter	373.87	113.74	335.78	84.35	260.55	80.70	273.40	109.74	346.65	82.89		
C_{it}	60	Percent	2.06	8.86	2.99	5.65	5.52	6.61	-2.06	7.54	0.65	5.63		

Source: based on our survey and Statistical Yearbooks of Inner Mongolia

3.4 Results

In this section, we first examine whether our data are appropriate for using the fixed effects model through statistical tests, involving the choice of the fixed effects or random effects model and a test for cross-sectional dependence. Moreover, we examine whether the variable of private use (L_{it}) and its quadratic term (L_{it}^2) are endogenous variables. This may be the case because the moment at which actual private use was adopted in a county could be decided by local households rather than by the (exogenous) formal government policy.

Specifically, the Hausman test is employed to determine whether a fixed or a random effects model is more appropriate. The test results for all four models reject the null hypothesis at $p < 0.05$, indicating that the fixed effects model is more appropriate. Cross-sectional dependence in panel-data models (also called contemporaneous correlation) is then tested using the Pasaran CD test considering that our panel data includes many more cross-sectional units (60) than time periods (5). Test results show that residuals are not correlated across entities and hence no cross-sectional dependence occurs in any of the four models. Ultimately, the Durbin-Wu-Hausman test is employed to examine endogeneity of the variables of L_{it} and L_{it}^2 . The test results reject the null hypothesis of exogeneity of L_{it} in the total and dense grassland models. The null hypothesis of exogeneity of L_{it}^2 is rejected in the total, dense and moderate grassland models. It indicates that endogeneity is found in the total, dense and moderate grassland models, which therefore require the instrumental variables approach. In this regard, the number of years for which formal use rights have been assigned to households is strongly correlated with the number of years for which actual private use has been adopted (the correlation coefficient is 0.825). Moreover, the implementation of this formal policy was not decided by the local government or households, but it followed the policy guidelines from higher level governments, which means the variable about formal use rights can be supposed to be exogenous. We therefore consider the number of years with formal use rights and its quadratic term as the instrumental variables of L_{it} and L_{it}^2 . The instrumental variables tests further prove that they are valid and strong instrumental variables for L_{it} and L_{it}^2 . After these tests, our four models are run using fixed effects and based on instrumental variables in the models with endogeneity. The results are shown in table 3.5.

Table 3.5 Estimated results based on the fixed effects model

Variables	Total grassland model (G_{it})		Dense grassland model (D_{it})		Moderate grassland model (M_{it})		Sparse grassland model (S_{it})	
	coefficient	z	coefficient	z	coefficient	z	coefficient	t
L_{it}	-3,252.50***	-3.28	-1,273.95*	-1.78	-2,137.97**	-2.12	-569.16*	-1.67
L_{it}^2	116.66**	2.24	-6.85	-0.18	84.24	1.46	61.62***	3.45
T_{it}	6,530.85	1.51	4,097.36	1.31	116.18	0.03	1,370.21	0.70
T_{it}^2	-879.51***	-2.56	-356.23	-1.44	-227.41	-0.81	-181.93	-1.18
F_{it}	113.83	0.94	207.22**	2.37	-31.48	-0.32	-110.58**	-2.02
F_{it}^2	-0.06	-0.42	-0.17	-1.62	0.01	0.06	0.16**	2.37
TF_{it}	-7.05	-0.82	-8.60	-1.38	4.87	0.69	-2.42	-0.62
C_{it}	-65,265.84***	-2.99	-33,047.87**	-2.10	-50,368.11***	-2.86	20,782.77**	2.12
C_{it}^2	195,365.30*	1.81	4,989.18	0.06	230,107.50**	2.59	-81,822.8*	-1.68
<i>Constant</i>	749,546.10***	29.27	333,232.90***	18.07	287,707.00***	13.68	140,156.3***	12.15
R^2	0.1668		0.1548		0.0365		0.1255	
<i>Obs</i>	300		300		300		300	

*significant at 10%; **significant at 5%; ***significant at 1%.

As listed in table 3.5, the coefficient of L_{it} is significant and negative and the coefficient of L_{it}^2 is significant and positive in the total grassland model. It indicates that a possible turning point exists at which the decrease in total grassland area may be reversed as the increase in the number of years with private use. Similarly in the sparse grassland model, the coefficient of L_{it} is significant and negative, and the coefficient of L_{it}^2 is significant and positive as well. Again the possibility of a turning point exists at which the decrease in sparse grassland area will reverse. It is noted that the coefficients of L_{it} are significant and negative in the dense and the moderate grassland models, but the coefficients of L_{it}^2 are not significant. It indicates that when other variables remain constant, the dense (moderate) grassland area decreases by 1274 (2138) hectares if the period with private use increases by one year. In addition, the significant and negative coefficient of T_{it}^2 indicates that temperature has a nonlinear relationship with the total grassland area. The significant coefficients of F_{it} show that precipitation is able to increase the dense grassland area significantly, but conversely reduce the sparse grassland area. And sparse grassland area has a nonlinear relationship with precipitation as presented by its significant coefficient of F_{it}^2 . Ultimately, the growth of cultivated land (C_{it}) has significant and negative effects on the total, dense and moderate grassland areas, and a significant and positive effect on the sparse grassland area. Nonlinear relationships are present in the total, moderate and sparse grassland models as shown by their significant coefficients of C_{it}^2 .

Comparing the above estimated model results with our expected results in table 3.3, the coefficients of significant variables in the models are almost consistent with our expectations, but the coefficients of L_{it} and L_{it}^2 in the sparse grassland model did not present opposite signs with that in the other three models. This difference indicates that the private use results in the decrease of sparse grassland area firstly and then increase, rather than increase from the beginning as we expected. A possible explanation could be that some sparse grassland degrades to such a degree that the grass cover drops from 5-20% to less than 5%. Such severe levels of degradation would however not be picked up by the variable of sparse grassland because land areas with less than 5% grass cover are no longer counted as grassland areas. And then the decrease of sparse grassland area per se is offset by its increase caused by the degradation of grassland quality that the dense and moderate grasslands degrade into sparse grassland after several years of private use.

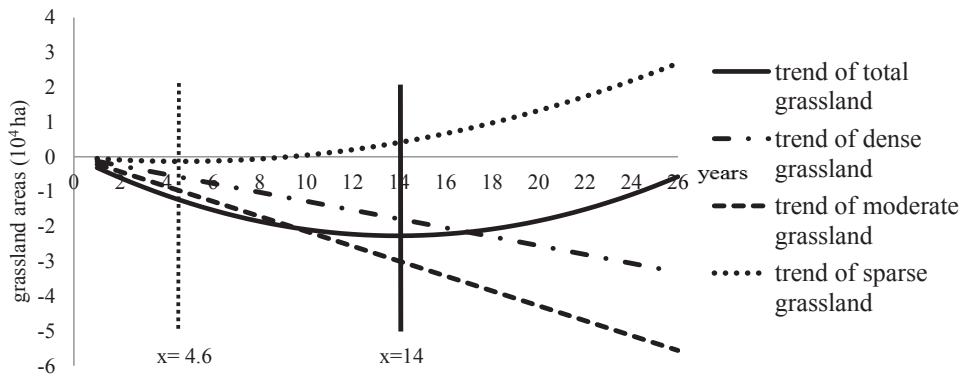


Figure 3.2 Estimated changes in the areas of different grasslands as the increase in the number of years with private use

In figure 3.2, we employ the significant regression coefficients of L_{it} and L_{it}^2 to particularly depict the changes in the areas of different grasslands as the number of years with private use increases. It is interpreted as the estimated effects of the land tenure reform on grassland condition while all of the other factors are controlled for. Specifically, the x-axis indicates the number of years with private use and the y-axis presents the areas of different grasslands. Considering the longest duration with private use till 2008 was 26 years in our original dataset, we drawn the maximum year of the x-axis at 26. The figure shows that the dense and moderate grassland areas present continuous decreases without reversion as their coefficients for L_{it}^2 are not significant in the model. And the moderate grassland area decreases more quickly than dense grassland. However, both sparse and total grassland areas demonstrate the predicted upward trends after the initial decreases as their coefficients for L_{it} and L_{it}^2 are significant. That is, sparse and total grassland areas decrease at first but then start to increase after a certain number of years with private use. The turning point is 4.6 years for the sparse grassland area and 14 years for the total grassland area in our case.

As such, the figure indicates that the total grassland area decreases and is not able to recover to the initial level until several decades after the start of the land tenure reform. Moreover, the continuing decrease in dense and moderate grassland areas indicates that the grasslands with higher grass cover will sustain degradation over time under privatisation. This trend is confirmed by the increase in the sparse grassland area while the reduction in the total grassland area. Subsequently, the increase in the total grassland area may stem from the increase in the sparse grassland area, rather than the improvement of grassland quantity. In short, the empirical

analysis confirms the negative effects of privatisation on grassland quantity and quality in the long term.

3.5 Discussion

In line with the results of Sneath (1998), Li et al. (2007) and Li and Huntsinger (2011), we find a negative impact of privatisation on the grassland condition. This may have discouraged herders to adopt private use, which could be an inherent explanation for the slow progress in the implementation of the land tenure reform on the grasslands of China in recent decades. But conversely, some scholars (see e.g. Li et al., 2007) have argued that the observed negative impacts of the privatisation of grassland use rights result from constraints in the supporting conditions for a successful privatisation process, rather than from the privatisation per se. Specifically, they point to a lack of resources for pastoral herders for instance to build fences and to effectively protect privatised grasslands from overuse by outsiders. An interesting avenue for further research would be to explore in more detail whether the so-called tragedy of privatisation has its roots in privatised use rights per se or in its failed implementation in practice.

Another challenge for future empirical research is to take into account the specific effects of overgrazing and the impacts of ecological reconstruction projects that have been initiated to protect the grassland condition in China since the 2000s (Li et al., 2014). The existing evidence on the relationship between livestock production and grassland degradation is vague and controversial (Harris, 2010; Cao et al., 2013b). Moreover, Liu et al. (2016) has shown that the impact of ecological projects on grassland condition was offset to some extent by socio-economic and climate factors. While these elements were not directly investigated in this paper because of limitations in the data, they were approximated by including the variables of agricultural activities in the model.

Concerning the governance of China's grasslands, instead of the private use by households, the community-based governance is fiercely advocated by pastoral specialists (e.g. Fernandez-Gimenez et al., 2012). They claim that community-based governance can facilitate the reciprocity of resource use among households, including capital investment and labour force use (e.g. Vetter, 2005; Li and Zhang, 2009; Cao et al., 2013b). Some empirical evidence exists to support community-based governance as well. For instance, Cao et al. (2013a) observed that the biomass, vegetation cover and species richness of grasslands are significantly higher under multi-household management than under single-household management in Maqu county, Gansu province. Ho (2001) showed that grasslands can be successfully managed by a village

community on the basis of a community-based arrangement at low transaction costs. However, more empirical evidence is needed on a large-scale to allow assessing the benefits of community-based governance over private use, taking into account the conditions of current social, economic and policy incentives in the pastoral areas.

3.6 Conclusions

This paper used quantitative analysis to investigate the relationship between grassland privatisation and grassland degradation. First, a review was made of the controversial views on the impact of privatisation on the grassland condition. Next, we summarized the progress in the land tenure reform in the pastoral areas of China. The evolution of grassland degradation was demonstrated in a typical pastoral area, using data for 60 counties in Inner Mongolia between 1985 and 2008. A fixed effects model was employed to disentangle the effect of the land tenure reform on the grassland condition while controlling for factors related to climate, agricultural activity and heterogeneity among counties. The model results show that private use causes the reduction of the total grassland area and the degradation of dense and moderate grasslands into sparse grassland in the long run. In conclusion, our analysis suggests that the land tenure reform, namely the privatisation of grassland use rights, plays a significant role in the degradation of grassland quantity and quality in Inner Mongolia. In other words, we observe “a tragedy of privatisation” (Guelke, 2003; Li and Huntsinger, 2011), as opposed to the well-known “tragedy of the commons”.

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Chapter 4

*How does land tenure reform impact upon
pastoral livestock production? An empirical
study for Inner Mongolia, China*

4. HOW DOES LAND TENURE REFORM IMPACT UPON PASTORAL LIVESTOCK PRODUCTION? AN EMPIRICAL STUDY FOR INNER MONGOLIA, CHINA¹²

ABSTRACT: How has land tenure reform affected livestock production in pastoral areas of China? This question is explored by estimating what impact assigning grassland use rights has on livestock production based on county-level data for Inner Mongolia between 1985 and 2008. The timing of the introduction of the household-based assignment of grassland use rights differed between counties, enabling a comparison of the effects of the land tenure reform. The changes in livestock production over time are examined by analysing data on changes in livestock population and meat output. The descriptive analysis shows that livestock production increased at a higher speed in the crop farming areas, but that the development of livestock productivity was faster in the pastoral areas. In the empirical analysis, we employed a fixed effects model to disentangle the effects of land tenure reform on livestock production from factors related to market forces, grassland condition, technological development and environmental heterogeneity. The model results reveal that the implementation of land tenure reform had significant and negative effects on the increase in livestock production, although total livestock production actually increased. It therefore appears that land reform is in itself unable to offset the impact of other factors that accelerate the increase in livestock production. Moreover, the constraining effect of land tenure reform on the increase in livestock production decreases with the number of years for which land tenure reform has been implemented, and ultimately disappears. Remarkably, the constraining effect of land tenure reform is stronger on the increase of livestock population than on that of meat output. This indicates that land tenure reform is beneficial in that it improves livestock productivity. In conclusion, land tenure reform, namely the privatisation of grassland use rights, puts a ceiling on livestock production, which could be a possible reason as to why it has been difficult to implement the reform on grasslands. However, the reform does prove to be beneficial in improving the livestock productivity of pastoral areas.

KEYWORDS: assignment of property rights, use rights, pastoral area, livestock population, meat output, fixed effects model

¹² Paper by Min Liu, Jikun Huang, Liesbeth Dries, Wim Heijman, and Xueqin Zhu, submitted to a *peer-reviewed journal*.

4.1 Introduction

Land tenure reform in China is characterised by assigning long-term land use rights to individual households, involving cropland, grassland, and forest land (see e.g. Hu, 1997; Banks, 2003; Zhang et al., 2012a). According to the current Law of the People's Republic of China on the Contracting of Rural Land, use rights of grassland are contracted to individual households for a duration of 30-50 years. At the end of the 1970s, land tenure reform was first implemented through the Household Production Responsibility System in the crop farming areas of China and was completed rapidly and successfully. Research showed that the assignment of individual cropland use rights gave farmer the incentive of improved agricultural production and spurred the marketing of agricultural goods (see e.g. Ho, 1996; Hu, 1997; Krusekopf, 2002; Banks, 2003). Based on this outcome, the central government of China continued land tenure reform in pastoral areas in the early 1980s and assigned grassland use rights as well as livestock property rights that had been owned by the communes to individual households. However, the assignment of grassland use rights has not been completed as successfully as that of cropland in China (Liu et al., 2015). For instance, compared with the cropland in China, where household-based use rights were assigned overnight, the assignment of grassland use rights is still incomplete despite efforts for 30 years. By 2014, around 84% of grasslands had been allocated to individual households in China, and the central government continues to emphasise the need for clarifying grassland use rights for individual households (Ministry of Agriculture of China, 2015). In practice, in some areas of Inner Mongolia, only the grasslands for mowing forages have been allocated to individual households while the grazing grasslands are still owned and used by all of the local households (Li et al., 2007). Scholars (e.g. Richard et al, 2006; Li and Zhang, 2009; Li, 2012) have identified a number of possible barriers to the completion of the land tenure reform in the pastoral areas of China. This paper will explore possible reasons by focussing on the aspect of pastoral livestock production.

Land tenure has been the concern of academia for a long time. The privatisation of grassland property rights or use rights has been regarded as a panacea to avoid overgrazing and protect the grassland ecosystem, in order to avoid the situation termed 'the tragedy of the commons' (Hardin, 1968; McEvoy, 1987; Ybarra, 2009). Coase's (1960) theorem of property rights also argued that a clear assignment of property rights is a precondition for economically efficient resource allocation and environmental sustainability. The followers of the 'tragedy of the commons' and property rights theories advocate that the grassland resource should be either privatised or maintained as common land whilst clearly assigning rights of entry and use to

promote the sustainable use of grassland through exclusiveness. Following this conventional wisdom, most governments in the world have assigned grassland property rights or use rights in an attempt to ensure the sustainable development of grasslands (Li et al., 2014). In line with observations about the effects of privatisation of cropland, the central government of China believes that the assignment of grassland use rights provides land users with incentives to graze animals within the carrying capacity of the land, as well as to increase land investment in order to conserve their own grasslands (Banks, 2003).

However, the findings on the effects of privatising grassland resources are complex (e.g. see Li and Zhang, 2009). During recent years, some scholars have expressed concern that the traditional common-use system and nomadism have been replaced by private use and settlement due to grassland privatisation (Banks et al., 2003). This transformation of traditional pastoralism has impeded mobility and flexible grazing (Fernandez-Gimenez, 2000; Wang et al., 2013) and limited herders' access to emergency pastures and other key productive resources (Niamir-Fuller and Turner, 1999; Fernandez-Gimenez, 2002). This, in turn, has increased feeding costs and reduced herders' ability to withstand natural risks (e.g. Li et al., 2007), causing constraints on the development of animal husbandry in pastoral areas. Some voices have also stated that grassland privatisation reduces the amount of land available for livestock grazing, further leading to a reduction in the number of livestock that an individual can potentially own, ultimately resulting in poverty (Mwangi, 2007). In addition, research has shown that the carrying capacity of grassland is reduced due to decreased access to heterogeneous landscapes (Boone and Hobbs, 2004; Hobbs et al., 2008; Boone et al., 2005). As such, the total population of livestock that could be supported by a grassland ecosystem is predicted to decline as a result of the spatial and social boundaries stemming from privatisation.

In fact, much of the growth in animal supply has been coming from rapidly expanding intensive breeding systems rather than through traditional grazing systems in pastoral areas (FAO, 2015). Similarly, the main production areas for livestock products in China have experienced a geographic shift from being pastoral areas to being crop farming areas (Li et al., 2008). The market share of livestock products from grazing systems is decreasing compared to that of crop-livestock mixed or industrialised systems (Squires et al., 2009). This trend is suggested to be attributed to the reduction in grass yields due to grassland degradation, and raises environmental concern about the ecosystem of permanent grassland (Li et al., 2008; Squires et al., 2009). Besides these possible factors, we wish to ascertain whether the privatisation of grasslands has played a role in the changes in livestock production of pastoral areas. Despite an abundance of

academic arguments and government reports on the results of the privatisation of grassland, there is a lack of quantitative studies, especially based on large-scale areas and long-term observations (e.g. Li and Huntsinger, 2011; Yu and Farrell, 2013; Conte, 2015). Moreover, the existing literature is short of empirical analysis targeting the impact of grassland privatisation on livestock production, although some studies are concerned with ecological effectiveness. Nevertheless, animal husbandry provides livelihoods to millions of people in pastoral areas and has the potential capacity to meet the rapidly increasing global demand for livestock products which is stimulated by growing populations, urbanisation and rising disposable incomes (FAO, 2015). In this regard, livestock production in pastoral areas is a very significant area and one which deserves being paid close attention to.

In the following section, we first describe the land tenure reform and the current livestock production in the research region. Next, in section 3, the data collection method is illustrated and a descriptive analysis based on the collected data is used to elaborate on the changes in livestock production in the pastoral areas compared to crop farming areas. In sections 4 and 5, we present the empirical model used in the study and the model results on how land tenure reform and other potential factors affect livestock production in the pastoral areas. In section 6, we discuss the model results and their underlying reasons. We conclude this paper with remarks on the effects of grassland privatisation on livestock production in pastoral areas.

4.2 Research region

China has around 400 million hectares of grassland, accounting for nearly 40% of its total territory, this being the second largest area of grassland in the world after Australia (Hua and Squires, 2015). Inner Mongolia, a province located in the arid and semi-arid areas of northern China, has 118.3 million hectares of land; in 2014, its permanent population was 25 million. It accounts for 21.7% of China's permanent area of grasslands. Approximately 67% of the total land area in Inner Mongolia is classified as grassland, the majority of which can be sub-classified as temperate grassland (Angerer et al., 2008). Inner Mongolia plays an important role in the supply of animal products as well as in the ecosystem of China due to its extensive grasslands. In the pastoral areas populated by Mongolians, the vast majority of local people maintain their livelihoods through grazing their livestock on the grasslands (Angerer et al., 2008). Inner Mongolia was one of the first regions in which land tenure reform was implemented on grasslands in China because of its crucial position in China's grassland resource and livestock production (Li and Huntsinger, 2011).

According to the current administrative divisions of Inner Mongolia, there are 102 counties, including 33 pastoral counties and 21 semi-pastoral counties. The remaining 48 counties are dominated by crop farming or urban districts. Pastoral counties are characterised by traditional grazing systems and permanent grassland is the dominant land type. In semi-pastoral counties, both permanent grassland and cropland are the dominant land types. Mixed crop-livestock systems also exist, where intensive animal husbandry and cropping both occur (Waldron et al., 2010). The majority of the pastoral areas of Inner Mongolia are found in the pastoral counties as well as semi-pastoral counties. Figure 4.1 presents the location of Inner Mongolia in China and its three types of counties.

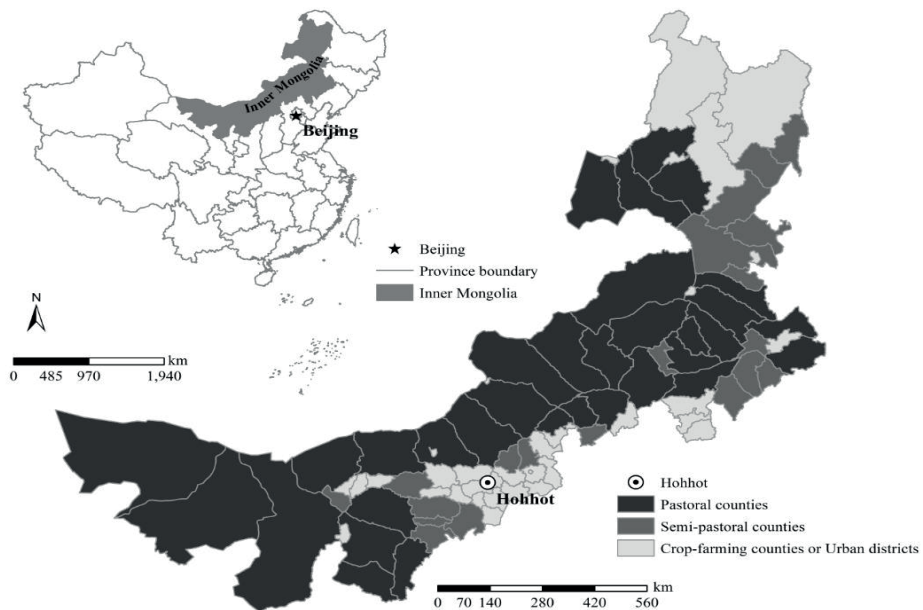


Figure 4.1 Inner Mongolia and its 102 counties

4.2.1 Land tenure reform in the pastoral areas of Inner Mongolia

Land tenure reform started in Inner Mongolia with the Double Contracts System (Cao Xu Shuang Cheng Bao) in 1982. This system aimed at assigning to individual households the livestock and grasslands that were managed by the communes during the collectivist period of China. Meanwhile, the users (individual households) of the grasslands are regulated in order to control their livestock numbers according to a determinant stocking rate with the aim of avoiding overuse of grasslands (Ho, 2000). The assignment of property rights over livestock was completed promptly but grasslands were not strictly assigned to individual households in

the majority of the pastoral areas in the 1980s. Particularly, the use of grasslands merely indicated a rough direction and position to herders, but local herders and their animals still roamed wherever they preferred. Hinton (1990) called this phenomenon privately controlled stock of animals on publicly owned lands in Inner Mongolia. The result was an uncontrolled scramble for wherever forage existed, which amounted to a general attack on a range of vegetation (Hu, 1997). Subsequently, the assignment of grassland use rights to individual households was strengthened by the Two Rights and One System policy (Shuang Quan Yi Zhi) in 1996 (Bureau of Animal husbandry of Inner Mongolia, 2000; Li and Zhang, 2009). This stimulated the progress of grassland privatisation greatly and is regarded as the second round of land tenure reform in the pastoral areas of Inner Mongolia (Yang, 2007). In recent years, growing attention has been paid to ensure the long-term stability of grassland use rights to individual households, such as confirming grassland plots, areas and contracts (Li, 2012). In 2015, local governments in Inner Mongolia are still working on clarifying the boundaries of grassland use rights for each household and issuing certificates (Bureau of Animal husbandry of Inner Mongolia, 2015).

Compared with crop farming areas, the implementation of the land tenure reform on grasslands is relatively complex and has taken a long time. More specifically, the grasslands have been subject to several different types of ownership (Yu and Farrell, 2013) and correspondingly, various types of land use have arisen since the start of the land tenure reform in Inner Mongolia. Table 4.1 provides a typology of the land tenure reform on the changes in formal use rights and actual types of grassland use, which is in line with the investigation of the Tibetan plateau by Banks et al. (2003).

Table 4.1 Typology of land tenure reform on grasslands of Inner Mongolia

Formal use rights owned by	Actual grassland use		
	Private use	Joint use	Common use
Individual households	+	+	+
A group of households	na	+	+
Collective/administrative village	na	na	+

Source: adapted from “formal and de facto grassland management units” (Banks et al., 2003)
na indicates not available

As presented in Table 4.1, with the implementation of land tenure reform, three types of ownership of formal grassland use rights have arisen, namely individual household ownership, ownership by a group of households and collective ownership. There are also three types of actual grassland use, these being private use by individual households, joint use by a group of households and common use by all of the villagers. When formal use rights are owned by an administrative or natural village, the only type of grassland use is common use. This arose primarily before or at the beginning of the implementation of the land tenure reform when the grassland rights had not yet been assigned. If the formal use rights are owned by a group of households, the land use is either common or joint. This situation mostly occurs in areas where the grassland resource is too scarce to be assigned to individual households, or where grassland use rights have not been assigned. Private use, joint use and common use exist simultaneously when the formal use rights are owned by individual households. The intention of the land tenure reform in the pastoral areas, however, is to achieve full private use when use rights are assigned to households. This reflects the gap between the household-based assignment of formal use rights and the actual adoption of private use. In practice, the specific timing and extent of conducting the land tenure reform¹³, including the changes in formal use rights and use patterns, differed among counties in Inner Mongolia.

4.2.2 Livestock production in Inner Mongolia

Inner Mongolia is one of the main production regions for animal products in China. In 2013, it accounted for 18% of the population of sheep¹⁴ of China, ranking first out of all Chinese provinces. The mutton output accounted for 22% of the total meat output in China. Six per cent

¹³In the remainder of the paper, the land tenure reform means the grassland use rights are assigned to individual households.

¹⁴The term 'sheep' in this paper includes both sheep and goats, in accordance with the China Statistical Yearbook.

of cattle were raised in Inner Mongolia, and 8% of China's total beef output was produced there, in both cases ranking second in China. Figures 4.2 and 4.3 present the livestock production of Inner Mongolia from 1979 to 2013, based on data from the China Statistical Yearbook (Zhong Guo Tong Ji Nian Jian). Livestock production is interpreted by the livestock population and meat output. Sheep and cattle are the dominant animals being raised in Inner Mongolia (Zhang et al., 2012b). We therefore employ the population of sheep and cattle to represent the livestock population, and the outputs of mutton and beef for meat output.

Figure 4.2 shows that the population of sheep decreased in 1982 and then generally presented an increasing trend until 1999, apart from a decrease during 1991-1993. It experienced a sharp increase from 2001 to 2004 after a decrease from 1999 to 2001, and remained relatively steady between 2005 and 2013. The population of cattle saw slight fluctuations but remained almost unchanged between 1979 and 2000. It reached the lowest point in 2001 after a slight decrease from 1999 and then experienced an increase between 2002 and 2010, followed by a slight decrease until 2013. It is evident that the number of sheep exceeded cattle numbers by a wide margin during the whole period.

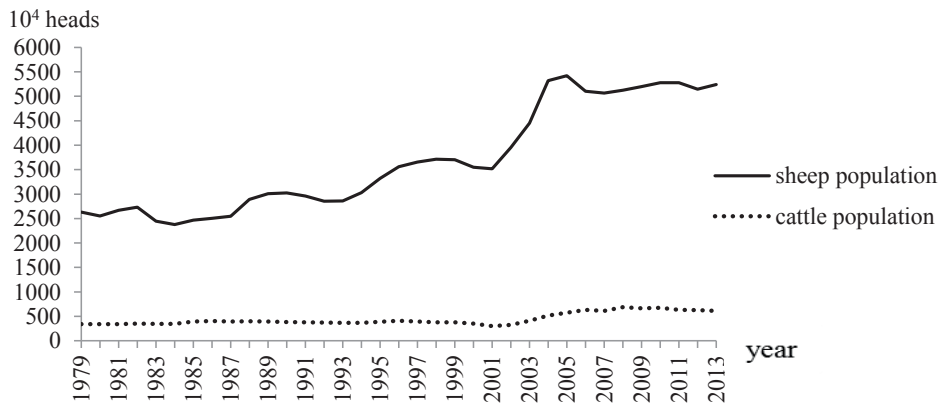


Figure 4.2 Livestock population of Inner Mongolian from 1979 to 2013

Figure 4.3 depicts Inner Mongolia's mutton and beef outputs from 1979 to 2013. Until 2002, the outputs of mutton and beef experienced fluctuating and slow increases. The output of mutton and beef increased sharply between 2002 and 2005, after which the output of mutton remained relatively steady, while beef continued to increase slightly between 2006 and 2013. In general, mutton output exceeded beef output during the whole period. Mutton output in 2013

was around 14 times higher than in 1979, while beef output was 19 times higher in 2013 than in 1979.

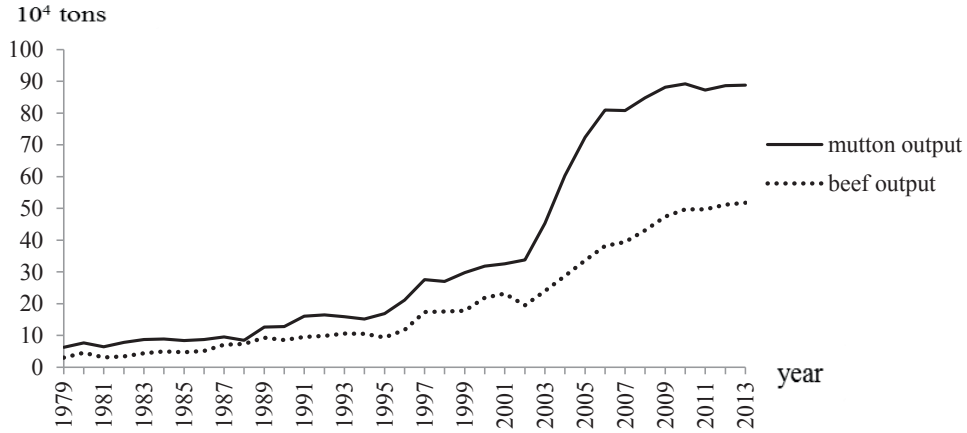


Figure 4.3 Mutton and beef outputs of Inner Mongolia from 1979 to 2013

Figure 4.4 illustrates the development of livestock productivity in Inner Mongolia from 1979 to 2013. The livestock productivity for both sheep and cattle has increased. The increase of productivity for cattle experienced obvious fluctuations, while the productivity for sheep saw a slight and steady increase. It is evident that the output per cattle increased faster than the output per sheep.

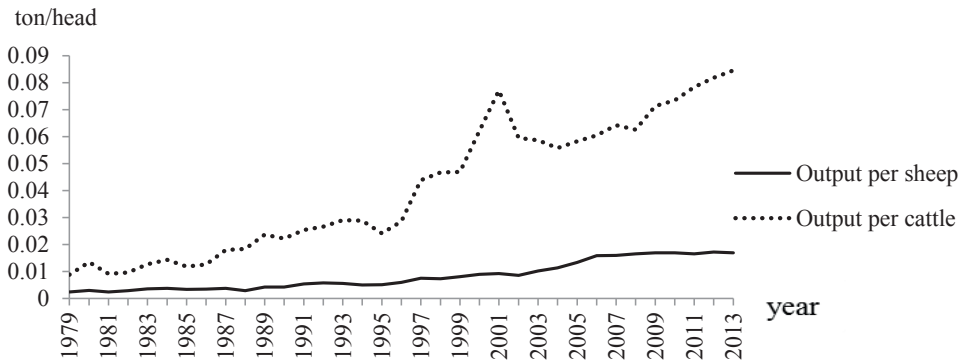


Figure 4.4 Livestock productivity of Inner Mongolia from 1979 to 2013

Although livestock population, meat output and livestock productivity have generally increased in Inner Mongolia over time, the importance and competitiveness of its livestock production

within China have been threatened by Henan and Shandong provinces, which are the dominant crop farming areas of China where intensive animal husbandry systems have developed in recent years (Su, 2010). Furthermore, a number of ecological projects for grassland conservation and the promotion of non-farming industries have been introduced in Inner Mongolia and have impacted upon the development of animal husbandry in the region (Squires et al., 2009). As a result, the share represented by animal husbandry in the gross domestic product of Inner Mongolia has decreased. Moreover, since 2001, the provincial government of Inner Mongolia has laid emphasis on the development of livestock production in its crop farming areas instead of in the pastoral areas. In 2007, 70% of livestock production in Inner Mongolia was located in crop farming areas (Su, 2010). This geographic shift from pastoral areas to the crop farming areas is discussed in the following section.

4.3 Data description

4.3.1 Data collection

We aim to estimate the impact of land tenure reform on pastoral livestock production. The pastoral areas are therefore distinguished from the crop farming areas for the purposes of data collection. Given that the timing and extent of the implementation of land tenure reform differed between counties, our empirical study of the effects of the reform on livestock production is conducted based on county-level data. The other factors that are controlled for their potential impacts on the pastoral livestock production are grassland condition, market forces, technological development and environmental heterogeneity (Li et al., 2008; Tessema et al., 2014). We therefore collected three types of data at the county-level in Inner Mongolia: survey data on land tenure reform of each county, observational data on grassland conditions and statistical data on socio-economic indicators.

First, the data on land tenure reform was collected via questionnaires in each county in Inner Mongolia. Based on the typology shown in Table 4.1, the questionnaire focused on when formal use rights were owned by collectives, groups and individual households respectively, and when grassland was actually in common use, joint use or private use. Questionnaires were sent to the Animal Husbandry Bureau of each county and were answered by key informants on local land tenure reform. Interviews were conducted by telephone to confirm the answers after receiving feedback from each county. We ultimately obtained valid feedback from 74 out of 102 counties. In addition, we conducted interviews with the officers who are working in the provincial institutes of Animal Husbandry of Inner Mongolia as well as with local herders to verify the

progress of land tenure reform on local grasslands. Second, information about grassland conditions was obtained based on remote sensing and an analysis of Geographic Information Systems. This relies on a database developed by the Chinese Academy of Sciences with original data from Landsat Thematic Mapper / Enhanced Thematic Mapper (Plus) (TM/ETM+) images (Deng et al., 2011). GIS satellite images were only collected in 1985, 1995, 2000, 2005 and 2008. We therefore use the average growth rate of grassland areas between the years of observation to estimate the grassland condition for each year. Third, the data on socio-economic indicators is based on existing statistical data collected by local governments. Specifically, the data about livestock production was gathered from the Statistical Yearbooks of Inner Mongolia. Market forces are represented by the real producer price of mutton and beef. This information was collected based on the Annual Compilation of Cost-benefit Data of Chinese Agricultural products (Zhong Guo Nong Chan Pin Cheng Ben Shou Yi Hui Bian) and deflated by a producer price index. Specific data on technological development is lacking, and is thus proxied by a time variable. The factor of environmental heterogeneity among counties is removed by the fixed effects model as a time-invariant variable.

The research period covers 1985-2008, which includes the main period when land tenure reform was implemented on the grasslands of Inner Mongolia. After excluding the urbanised counties and counties that underwent changes in administrative regions during the research period, 60 counties are retained in the research sample. They include 27 pastoral counties, 18 semi-pastoral counties and 15 crop farming counties. In fact, the crop farming counties only have a few permanent grasslands and their livestock production relies mainly on cropping rather than grazing. Furthermore, household-based assignment of grassland use rights was barely implemented because the areas of grasslands in most crop farming counties are too small to be subdivided. Therefore, the information from crop farming counties is only used in the descriptive analysis to compare the relative changes in livestock production between crop farming areas (crop farming counties) and pastoral areas (pastoral and semi-pastoral counties).

4.3.2 Descriptive analysis

Based on the collected data, we first illustrate the progress of the land tenure reform¹⁵ in the pastoral areas of Inner Mongolia from 1985 to 2008. In Figure 4.5, the proportion of counties

¹⁵ Considering that some counties did not complete the land tenure reform fully, we denote a county as having implemented the land tenure reform if at least 50% of the grassland area of this county has been assigned to individual households.

in which formal use rights were assigned to households increased from 9% in 1985 to 91% in 2008; the proportion of counties that adopted actual private use rose from 7% in 1985 to 71% in 2008. A sharp increase in formal and actual use rights adoption is observed between 1995 and 1998. This is consistent with the second round of land tenure reform implemented in this period. It can be seen that the adoption of actual private use lagged behind the assignment of formal use rights; neither were fully completed until 2008.

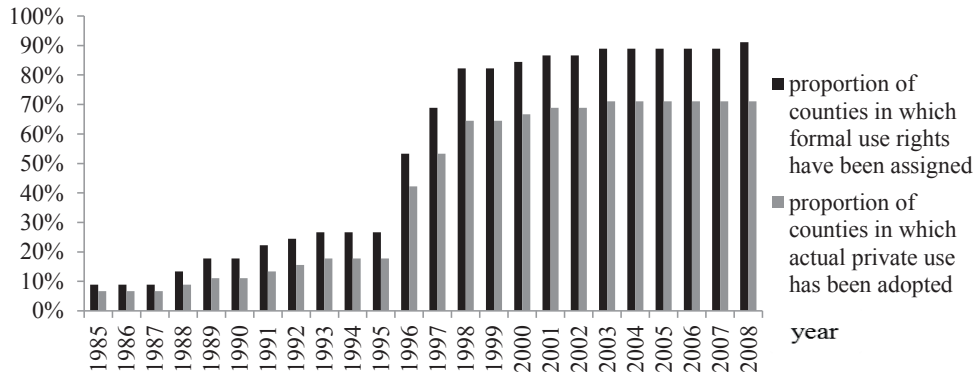


Figure 4.5 Progress of the land tenure reform in the pastoral areas of Inner Mongolia from 1985 to 2008

Figures 4.6, 4.7 and 4.8 demonstrate the changes in livestock production and productivity of the pastoral areas relative to the crop farming areas from 1985 to 2008. The changes are represented as an index where the level in 1985 is set equal to 100. Figure 4.6 indicates that both sheep population and mutton output increased in pastoral areas as well as in crop farming areas from 1985 to 2008, and that the increase of sheep population was slower than that of mutton output, especially after 1995. In addition, after 2001, the growth rate of the sheep population in crop farming areas significantly exceeded that in pastoral areas. And the growth rate of mutton output of crop farming areas exceeded that of pastoral areas from 1999 onwards.

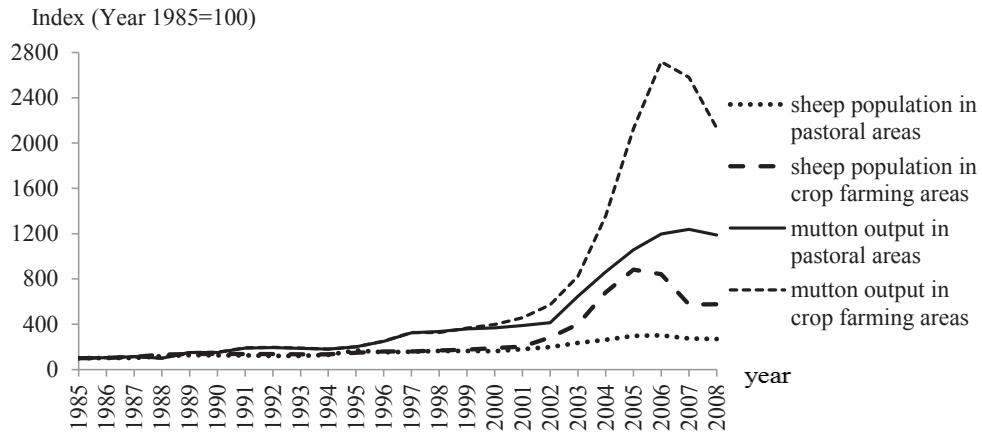


Figure 4.6 Comparison of sheep population and mutton output between pastoral and crop farming areas of Inner Mongolia from 1985 to 2008

Figure 4.7 shows that the cattle population increased slightly compared with 1985, and that the output of beef has increased significantly in both pastoral and crop farming areas over the past 24 years. The increase of beef output was more obvious than that of cattle population. Moreover, the growth rate of the cattle population in pastoral areas was exceeded by that in crop farming areas from 2001 onwards. Also, the growth rate of the beef output in pastoral areas was exceeded by the beef output in crop farming areas after 2000.

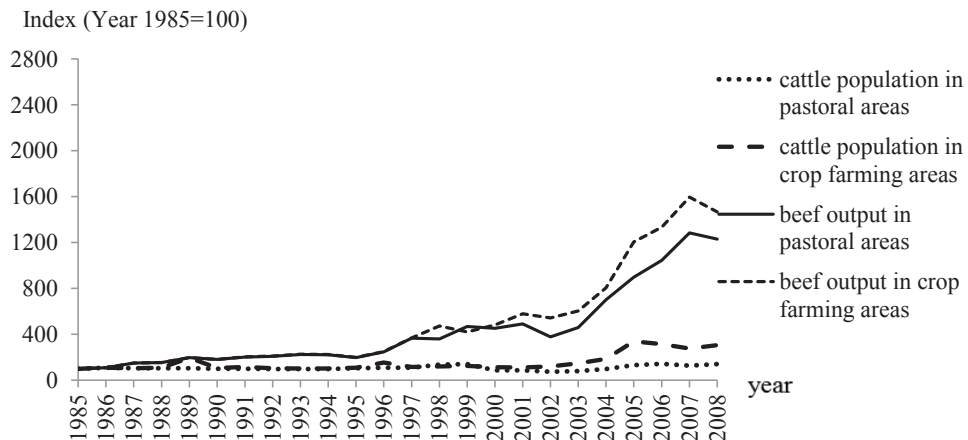


Figure 4.7 Comparison of cattle population and beef output between pastoral and crop farming areas of Inner Mongolia from 1985 to 2008

Figure 4.8 compares the development of livestock productivity between pastoral and crop farming areas from 1985 to 2008. It is evident that the increment of output per sheep in pastoral areas proved to be larger than that of crop farming areas after 2002. The increment of output per cattle in pastoral areas was larger than that of crop farming areas since 1999. The development of output per cattle was faster than per sheep in both areas.

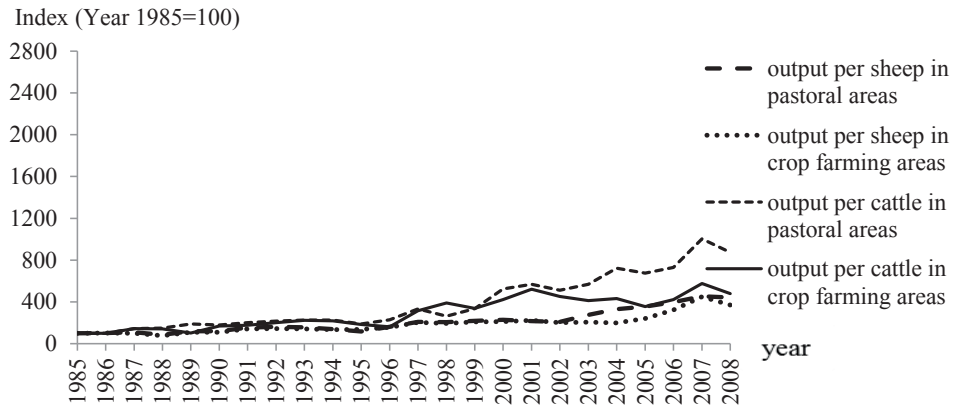


Figure 4.8 Comparison of livestock productivity between pastoral and crop farming areas of Inner Mongolia from 1985 to 2008

In short, land tenure reform was implemented progressively in the pastoral areas in the period 1985-2008. Meanwhile, it appears that the increase of livestock production was faster in crop farming areas than in pastoral areas, but conversely the development of livestock productivity was slower in crop farming areas. It can be seen that the competitiveness of animal husbandry in pastoral areas has been threatened by crop farming areas in recent decades as the livestock population and meat output has grown faster in crop farming areas since around 2000. This is consistent with the fact that in China, the main producing areas of livestock products have experienced an adjustment, moving from a grazing system to a crop-livestock mixed system and from pastoral areas to crop farming areas (Li et al., 2008). Interestingly, the development of livestock productivity was faster in pastoral areas than in crop farming areas, which may be attributed to the transformation of traditional pastoralism in the pastoral areas of Inner Mongolia. As such, we assume that land tenure reform constrained the increase of livestock production, but spurred the development of livestock productivity of pastoral areas. We will assess this assumption with an empirical model based on the data from 45 sample counties of pastoral areas in Inner Mongolia.

4.4 Empirical model

A fixed effects model is widely used in economic research, primarily to study the causes of changes within entities over time (e.g., Fergusson et al., 2002; Huang et al., 2006). The model employs within transformation to remove all time-invariant (fixed) explanatory variables, i.e. the model is performed in deviations from individual means (Verbeek, 2012). As such, the fixed effects model provides a method that takes observable as well as unobservable time-invariant explanatory variables into account, but the estimation is not dependent on the value of time-invariant (fixed) variables (Verbeek, 2012). Such an approach is appropriate in this study considering that the results of land tenure reform within each county over time can be studied effectively by controlling for the unmeasured heterogeneity among counties. On the other hand, the explanatory variables of land tenure reform we are interested in show changes in value across our research period for a substantial proportion of counties in our data (see Figure 4.5), which satisfies the basic requirement of the fixed effects model (Daun-Barnett, 2011). In addition, our research sample attempts to include all counties in the pastoral areas of Inner Mongolia, rather than random draws, which preliminarily indicates that a fixed effects model is more appropriate than a random effects model (Verbeek, 2012).

Based on the theoretical framework of the fixed effects model, we formulate the effects of the land tenure reform and other potential factors on livestock production into the following equations:

$$\begin{aligned} \log(S_{it}) = & s_i + a_1 \log(R_{it-1}) + a_2(1/(R_{it-1})) + a_3 \log(M_{it-1}) + a_4 \log(T_{it-1}) + a_5 \log(G_{it-1}) \\ & + a_6 Y_t + a_7 P_i * Y_t + \varepsilon_{it} \end{aligned} \quad (1)$$

$$\begin{aligned} \log(C_{it}) = & g_i + b_1 \log(R_{it-1}) + b_2(1/(R_{it-1})) + b_3 \log(B_{it-1}) + b_4 \log(T_{it-1}) + b_5 \log(G_{it-1}) \\ & + b_6 Y_t + b_7 P_i * Y_t + \mu_{it} \end{aligned} \quad (2)$$

$$\begin{aligned} \log(U_{it}) = & u_i + c_1 \log(R_{it-1}) + c_2(1/(R_{it-1})) + c_3 \log(M_{it-1}) + c_4 \log(T_{it-1}) + c_5 \log(G_{it-1}) \\ & + c_6 Y_t + c_7 P_i * Y_t + \lambda_{it} \end{aligned} \quad (3)$$

$$\begin{aligned} \log(F_{it}) = & f_i + d_1 \log(R_{it-1}) + d_2(1/(R_{it-1})) + d_3 \log(B_{it-1}) + d_4 \log(T_{it-1}) + d_5 \log(G_{it-1}) \\ & + d_6 Y_t + d_7 P_i * Y_t + \delta_{it} \end{aligned} \quad (4)$$

where i and t present the i^{th} county and year t . Variables are defined in Table 4.2 and summary statistics are provided.

Table 4.2 Definition of variables and summary statistics¹⁶

Variable	Variable definition	Unit	Observations	Mean	Std. Dev	Min	Max
S_{it}	Ratio of sheep population of county i in year t over that in 1985	na	1080	1.73	1.74	0.20 ¹⁷	23.12
C_{it}	Ratio of cattle population of county i in year t over that in 1985	na	1080	1.08	0.89	0.12	18.68
U_{it}	Ratio of mutton output of county i in year t over that in 1985	na	1080	4.29	6.47	0.60	68.90
F_{it}	Ratio of beef output of county i in year t over that in 1985	na	1080	4.30	5.93	0.14	64.58
R_{it-1}	Number of years that formal grassland use rights have been assigned to individual households in county i until year $t-1$	Years	1035	4.28	5.69	0	25
M_{it-1}	Real producer price of mutton in year $t-1$	Yuan/kilo	1035	1.15	0.49	0.59	1.94
B_{it-1}	Real producer price of beef in year $t-1$	Yuan/kilo	1035	1.29	0.71	0.63	3.10
T_{it-1}	Ratio of total grassland area of county i in year t over that in 1985, with one year lag	na	1035	0.98	0.05	0.79	1.11
G_{it-1}	Share of good quality grassland area over total grassland area of county i in year t , with one year lag	na	1035	0.80	0.20	0.09	1.00
Y_t	Year t	na	1080	12.50 ¹⁸	6.93	1	24
P_i	=1 if county i is a pastoral county, =0 otherwise	na	1080	0.60 ¹⁹	na	0	1

¹⁶ The summary statistics based on the variables that are used in the model can be found in appendix 4-A.

¹⁷ It is noted that the minimum (or maximum) values for livestock production may appear to be unusually small (or large). On the one hand, these extreme values can be attributed to the high (or low) values in the base year (1985). On the other hand, livestock structures may have sharply changed due to government interventions. For instance, the implementation of the grazing ban severely decreased the sheep population in several counties. In our analyses, we assume that these government-driven factors are captured by the time variable.

¹⁸ Year 1985 equals 1, 1986 equals 2, etc.

¹⁹ The mean of P_i refers to the share of observations for which the dummy variable equals 1.

Sheep and cattle population and mutton and beef output are used to represent the livestock production. The ratios of these four indicators in year t compared with the base year 1985 (S_{it} , C_{it} , U_{it} and F_{it}), expressed in logarithms, are employed as the four dependent variables in equations (1)-(4). The four equations are denoted as the sheep population, cattle population, mutton output and beef output models, respectively. Variable R_{it} indicates the land tenure reform of county i in year t , which is presented by the number of years in which the formal use rights of grassland have been owned by individual households in county i by year t . We assume that land tenure reform has a non-linear relationship with livestock production. Plenty of scholars have proved the existence of a nonlinear relationship between access to land and agricultural production (e.g. Finan et al., 2005). Ostrom (2007) claimed that the problems linked with social-ecological systems require serious study which take account of complex, multivariable, nonlinear, cross-scale, and changing systems. Our functions attempt to explore the trend of livestock production under the land tenure reform over time based on the assumption of their non-linear relationship, which can be concave, convex, quasi-concave or quasi-convex. The logarithm and reciprocal of the variable of land tenure reform are employed to present the non-linear relationships²⁰. Considering that the value of the land tenure reform variable is zero for some counties as they had not implemented the reform by 2008, we add a small original value (e.g., 0.01)²¹ to R_{it} when R_{it} is specified in $\log(R_{it})$ or $1/(R_{it})$.

Based on the existing academic research, the factors of market forces, grassland condition, technology development and environmental heterogeneity are widely suggested to impact the pastoral livestock production (Li et al., 2008; Squires et al., 2009; Tessema et al., 2014). As such, they are considered as control variables in our functions for disentangling the impact of land tenure reform. Market forces are proxied by the local mutton price (M_{it}) and the beef price (B_{it}) which are expected to affect supply decisions of livestock producers (Komarek et al., 2012). The grassland condition refers to the quantity and quality of grasslands. It indicates the carrying capacity for livestock, reflecting grassland degradation and climate change factors such as temperature and water availability (Henry et al., 2012). Specifically in our model, the ratio of total grassland area in year t over that in 1985 is used to present the changes in quantity

²⁰We have also tested a model with a quadratic term instead of the reciprocal to specify the non-linear relationship. The results of both model specifications were comparable, which indicates the model specification is reliable and the model results are robust (see appendix 4-B for details of the specification with the quadratic form).

²¹0.01 is small enough to be regarded as the original value because we tried smaller values (such as 0.001, and 0.0001) and obtained consistent model results.

of grassland (T_{it}). The grassland quality is presented by the share of the area of good quality grassland²² in the total grassland area in year t (G_{it}).

Technological development is presented by the time variable Y_t . It should be noted that the development of technology in pastoral areas is mostly government-driven, mainly as a result of the growing interest of the central government in the ecological benefits of grasslands in recent years. For instance, a series of Ecological Construction Programs for grassland conservation have been implemented since 2000 to develop intensive animal husbandry through improving the technology of animal husbandry (Li et al., 2014). These have stimulated the improvement of forage production, feeding and fattening techniques and the use of improved breeds (Liao, 2009). The time variable Y_t is therefore used to interpret the impact of these ecological programs as well. P_i is a dummy variable that is one for pastoral counties and zero otherwise. The interaction term of Y_t and P_i is included in the equations to recognise the differences between pastoral counties and semi-pastoral counties in terms of technological development and ecological policies. Apart from the time and dummy variables, we treat all independent variables with a one year lag because market forces, grassland condition and land tenure reform are all considered to affect livestock production with a time delay. Moreover, market forces and grassland condition are used in logarithmic form.

Finally, other factors such as elevation, terrain slope and distance to the provincial capital, which are not expected to change significantly over time, are treated as time-invariant (fixed) factors. They are represented by the terms s_i , g_i , u_i and f_i in the models. The coefficients of the independent variables are a_n , b_n , c_n and d_n ($n=1, 2, \dots, 7$) and the random error terms are ϵ_{it} , μ_{it} , λ_{it} and δ_{it} . There are 1035 observations used in the fixed effects model, covering 45 counties and 23 years.

4.5 Model results

Before the estimation, we first conduct a Hausman test to choose between the fixed effects or random effects regression as the estimation technique. The test results for the models of sheep population, mutton output and beef output reject the null hypothesis at $p < 0.01$, indicating that the fixed effects regression is more appropriate. Only the test result for the model of cattle

²² 'Good quality grassland' denotes grassland where the canopy cover of grass is more than 20%. Grasslands are divided into three categories according to the canopy cover, namely dense grassland, moderate grassland and sparse grassland (Deng et al., 2011). In this paper, good quality grassland includes dense grassland and moderate grassland.

population does not reject the null hypothesis at $p < 0.01$. We conducted the random effects and fixed effects regression for the model of cattle population. The results are comparable, with the exception of the coefficient of the constant term. To compare the results of the four models consistently, we use the fixed effects regression for all of them.

Furthermore, we discuss the exogeneity of the variable of land tenure reform (R_{it}). R_{it} indicates the number of years for which formal grassland use rights have been assigned. In practice, the assignment of formal grassland use rights was implemented by the county-level government based on top-down executive orders. According to our survey, the differences in the assignment of formal grassland use rights among counties resulted mainly from the implementation efficiency of each county government and the county's geographical location. For instance, a county that is located in a remote area was less motivated to assign formal use rights. In this regard, the speed of implementation of the land tenure reform by local governments was not related to local livestock production, grassland condition, or other factors that we include in the model. Hence, we treat R_{it} as an exogenous variable. In contrast, the decision by local households to actually adopt private use is likely to have taken into account the grassland condition, living customs, production pattern etc., and would therefore have created a potential problem of endogeneity. The exogeneity of policy implementation in China was also supported by Liu et al. (2010) based on their research about the implementation of the Sloping Land Conversion Program (SLCP).

The fixed effects model is used to estimate our four models based on the panel data set. The model results about how the factors impacted upon livestock production in the pastoral areas are listed in Table 4.3.

Table 4.3 Model results of the factors that impact upon livestock production in the pastoral areas

Variables	Sheep population model		Cattle population model		Mutton output model		Beef output model	
	$\log(S_{it})$	t	$\log(C_{it})$	t	$\log(U_{it})$	t	$\log(F_{it})$	t
	coefficient		coefficient		coefficient		coefficient	
$\log(R_{it-1})$	-0.18***	-5.68	-0.10***	-3.36	-0.09***	-2.63	-0.08*	-1.83
$1/(R_{it-1})$	-0.01***	-4.93	-0.01***	-3.52	-0.004**	-2.33	-0.005*	-1.94
$\log(M_{it-1})$	0.23***	3.28	na	na	0.06	0.87	na	na
$\log(B_{it-1})$	na	na	0.18***	3.83	na	na	0.19***	2.95
$\log(T_{it-1})$	-0.08	-0.21	1.54***	4.10	-0.64	-1.53	0.57	1.09
$\log(G_{it-1})$	1.17**	2.21	-0.25	-0.52	1.67***	3.03	0.83	1.21
Y_t	0.04***	8.18	0.01***	2.58	0.12***	22.08	0.10***	17.11
$P_t * Y_t$	-0.003	-0.91	-0.03***	-8.51	-0.02***	-4.76	-0.05***	-10.65
Constant	0.38**	2.36	0.15	1.00	0.24	1.44	0.50**	2.46
R^2	0.30		0.12 ²³		0.77		0.60	
Observations	103,5		103,5		103,5		103,5	

*significant at 10%, **significant at 5%; ***significant at 1%.

²³The low R^2 for the cattle population model may be explained by the fact that cattle production is much more prominent in crop farming counties and major changes in cattle population overtime also occurred mainly in these counties. Moreover, some factors that affect cattle population are not specifically captured by the explanatory variables of our model. For instance, changes in the cattle population in some counties may have resulted from government-driven adjustment policies. This factor is not explicitly included in our model because of data limitations, and we have merely used the time variable to capture the potential effects of technological and policy developments. We have stated this point in the discussion section as our research limitation. Despite the low value of R^2 , the effect of land tenure reform on the cattle population is shown to be significant.

First, we focus on the coefficients of $\log(R_{it-1})$ and $1/(R_{it-1})$, indicating the effects of the land tenure reform on livestock production over time. All of these coefficients are significant and negative in the four models. Based on the coefficients of $\log(R_{it-1})$ and $1/(R_{it-1})$, quasi-concave curves are obtained as depicted in Fig. 9. The horizontal axis presents the number of years that the land tenure reform has been implemented and the vertical axis shows the livestock production index (livestock production in the first year of land tenure reform equals 100). Considering the longest duration for formal use rights privatised until 2008 was 26 years in our research sample, we determined the maximum year of the horizontal-axis as 26.

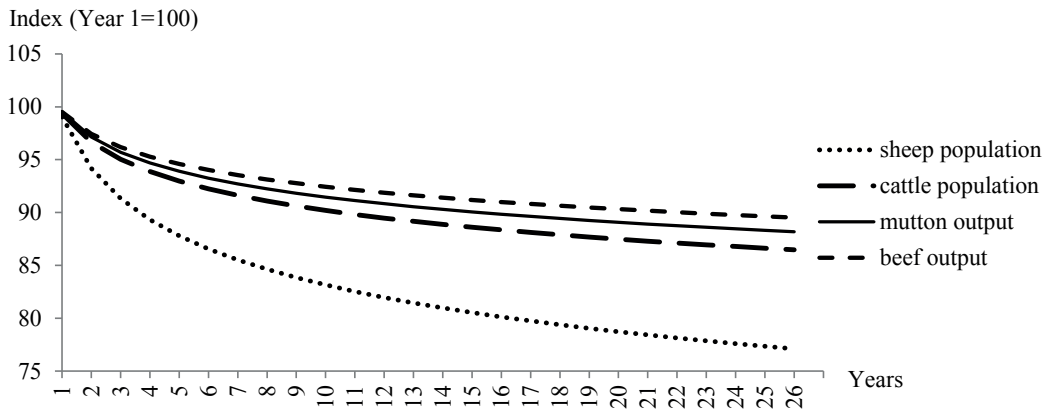


Figure 4.9 Relationship between land tenure reform and livestock production

In Figure 4.9, the indexes of sheep population, cattle population, mutton output and beef output are all decreasing with the number of years that the land tenure reform is implemented, and the reduction is fastest in the first years of the reform while it slows down in later years. Moreover, the decrease in sheep population is faster than that of mutton output, indicating that land tenure reform had a more negative impact on the sheep population than on mutton output. Similarly, land tenure reform has affected the cattle population more than beef output as the decrease in cattle population is faster than that of beef output. These results indicate that the implementation of land tenure reform promotes the development of livestock productivity in pastoral areas although it constrains the increase in livestock population. Moreover, the mutton output (sheep population) decreases faster than beef output (cattle population).

It is interesting to compare Figure 4.9 with Figures 4.6 and 4.7 – which present the increasing trends in the sheep population, mutton output and beef output after 1985. The seemingly contradicting trends of livestock production in these figures can be explained by the effects of the other factors considered in our models. For instance, the producer price has a significant and positive effect on livestock production. The mutton price elasticity indicates that the sheep population in year t will increase by 0.23% when

the mutton price in year $t-1$ increases by 1%. Similarly, cattle population (beef output) in year t will increase by 0.18% (0.19%) when the beef price in year $t-1$ increases by 1%. The better grassland condition also has a significant and positive effect on livestock production. Cattle population in year t will increase by 1.54% if the total grassland area in year $t-1$ increases by 1%. Sheep population (mutton output) in year t will increase by 1.17% (1.67%) if the good quality grassland area in year $t-1$ increases by 1%. In an attempt to improve grassland condition, China has implemented various ecological projects. However, in reality, 90% of the grasslands are still degraded to various degrees (Mei et al., 2013). For example, between 1991 and 2002, 3.1% of the available natural grassland was lost, either through its being degraded into unused land, or being transformed into other purposes. During the same period, 1.68 million hectares of land was rehabilitated into natural grassland through eco-environmental projects. This resulted in a net decline of 6.38million hectares (around 2.4% of the available natural grassland) (Qu et al., 2011). In this regard, the deteriorating grasslands could have impeded the livestock production.

The coefficients of the time variable are significant and positive in four models. It indicates that the technology development affects livestock production positively, and it can be seen that the positive effects are stronger with respect to meat output than for the livestock population. The interaction term of the time variable and the dummy variable of pastoral counties is also significant, but negative, in the cattle population, mutton output and beef output models. It indicates that cattle population, mutton output and beef output increase less in the pastoral counties than in the semi-pastoral counties with technology development. Sheep population increases significantly without a distinction between pastoral or semi-pastoral counties. In general, it appears that the development of livestock production is slower in the pastoral counties than in the semi-pastoral counties over time, with the exception of the development of the sheep population which does not differ between county types. Notably, Figures 4.6 and 4.7 showed that livestock production experienced a geographic shift from pastoral areas to the crop farming areas. Our results extended this finding to the specific comparison between pastoral counties and semi-pastoral counties. That is, livestock production, with exception of the sheep population, has experienced a geographic shift from semi-pastoral counties to pastoral counties in the pastoral areas.

4.6 Discussion

The model results demonstrate that the implementation of land tenure reform has significantly constrained the increase of livestock population and meat output in the pastoral areas of Inner Mongolia. This finding can be explained in various ways. On the one hand, the implementation of land tenure reform may have effectively controlled the increase in livestock population, as was the aim of land tenure reform, which expected to internalise the social costs of overgrazing by privatisation and thereby avoid the tragedy of the commons. The increase in meat output is correspondingly restricted due to the

constraints on livestock population. On the other hand, the spatial and social boundaries stemming from land tenure reform have impeded livestock production because fragmentation and loss in mobility increase the costs of feeding and reduce herders' ability to withstand natural risks (e.g. Li et al., 2007). These results are in line with findings from Mongolia where livestock production declined and poverty rose sharply after grassland privatisation in the 1990s (Griffin, 1995). The model results also illustrate that the decrease in livestock numbers and meat output occurred especially in the early years of the land reform and that this effect weakened over time, which may be attributed to the gradual adaptation of production patterns to the privatised grasslands. Moreover, the constraining effects on the increase of livestock production could be a potential reason to explain why land tenure reform was so slow to be implemented and is still in progress after 30 years. This raises doubts about the adaptability of extending policy measures designed for crop farming areas to the pastoral areas. Although the land tenure reform promoted agricultural production successfully in the crop farming areas of China, the pastoral areas may have produced different results.

Compared with the constraining effects of the land tenure reform, market forces have a strong positive effect on livestock production. The passing of time implies technological development, and also promotes the increase in livestock production, although more slowly in pastoral counties than in semi-pastoral counties. This points towards potential approaches for stimulating livestock production in pastoral areas. In addition, improved grassland conditions also have positive effects on livestock production, which suggests that grassland conservation is necessary not only to improve the ecosystem, but also to develop sustainable livestock production.

Given the controlling effects of land tenure reform on the increase in livestock population and the facilitating effects on improving livestock productivity, it is recommended that the government supervises the implementation of assigning grassland use rights to individual households and clarifies fuzzy grassland boundaries between households. However, pastoralism that embraces common use may also have benefits for the sustainable development of animal husbandry in pastoral areas due to the advantages of the mobility, flexibility and reciprocity (e.g. Fernandez-Gimenez, 2002). In this regard, the privatisation of grassland property rights or use rights may be a precondition to ensuring cooperative use based on stable property rights or use rights in practice. In fact, recent institutional innovations in pastoral areas of China has followed two distinct pathways: the strict implementation of clarifying grassland use rights to individual households on the one hand, and the encouragement of co-management and cooperative use of grasslands based on substantive community participation on the other hand (Banks, 2003).

As a final note, we would like to mention a number of limitations of our analysis that present challenges for future research. Our use of the time variable is a simplified way to capture the effect of technological

development and ecological policy. One challenge for future work is therefore to ascertain the specific effects of technological improvements and ecological policies on livestock production in pastoral areas. Moreover, the data on the grassland quality only represents the canopy cover, but does not indicate its edibility by animals, which is a serious limitation related to the measurement of the grassland conditions. Furthermore, the accuracy of Chinese livestock statistics is questionable (Waldron et al., 2007). The statistical collection of data on the livestock population is extremely difficult in China because small rural households raise livestock on a small scale and supply chains are dominated by countless small traders and processors, unlike the centralised slaughter and auction systems that facilitate statistical collection in developed countries (Waldron et al., 2007). In this regard, our research findings should be useful for detecting general trends if not specific numbers.

4.7 Conclusions

This paper explores the period during which the household-based assignment of grassland use rights has been implemented in order to describe the progress of land tenure reform and its impact on the grasslands of China. We focus on comparing the development of livestock production between pastoral areas and crop farming areas under the land tenure reform and evaluating the effect of land tenure reform on livestock production. Livestock production was presented by sheep population, cattle population, mutton output and beef output. The descriptive analysis shows that land tenure reform was implemented progressively in the pastoral areas, rather than the case of the cropland areas where household-based use rights were completed overnight (Banks et al., 2003; Ho, 2000). It appears that the increase of livestock production was faster in crop farming areas than in pastoral areas, while conversely the development of livestock productivity was slower in crop farming areas. A fixed effects model is employed to estimate the impacts of the land tenure reform on livestock production based on a dataset of 45 counties in the pastoral areas of Inner Mongolia from 1985 to 2008. The model controls for factors that obscure the relationship between livestock production and land tenure reform, such as market forces, grassland condition, technological development and environmental heterogeneity among counties. The model results provide quantitative evidence that land tenure reform has put a ceiling on livestock production, but this constraining effect is unable to offset the impact of other factors that accelerate the increase in livestock production, which explains the actual increase in livestock production. Moreover, the constraining effect of land tenure reform on the increase in livestock production decreases with the number of years since the implementation of land tenure reform and ultimately disappears. Remarkably, the constraining effect is stronger on the increase in livestock population than on that of meat output, which indicates that land tenure reform stimulates the development of livestock productivity. With respect to the other factors, it appears that the cattle population tends to be affected by the quantity of grassland, while the sheep population and mutton output are more affected by the quality of grassland.

The factor of technological development impacts on the increase in meat output more than that on the increase in livestock population, and the influence is more evident in the semi-pastoral counties than in the pastoral counties. It can be seen that technological development also prompts the development of livestock productivity.

The development of livestock productivity that is caused by the implementation of land tenure reform is different from that due to technological development. More specifically, land tenure reform in pastoral areas has transformed the production patterns of pastoralists through the privatisation of grasslands (Ho, 2000). For instance, the indigenous people in Inner Mongolia employed nomadic management and seasonal transhumance to graze animals in areas where pastures and water were available (Wang et al., 2013). Since the spread of the land tenure reform on the grasslands, a growing number of households have built physical boundaries (such as fences) in order to prevent others from using their grasslands, which implies that mobile grazing is no longer possible and most areas of grassland are in small-scale private use (Hua and Squires, 2015). The traditional nomadic production pattern has therefore gradually been displaced by a sedentary pattern. Our research results essentially indicate that this transformation, resulting from land tenure reform, has prompted livestock productivity in the pastoral areas of Inner Mongolia. In other words, land tenure reform entailing grassland privatisation probably stimulates herders to put more effort into improving the output per animal, instead of increasing the amount of animals, which is beneficial to the development of livestock production systems in pastoral areas. However, the constraining effects of grassland privatisation on the increase in livestock production could also be a possible reason to explain why land tenure reform was implemented with difficulty on grasslands, especially in terms of the lag between the reform and the actual adoption of private use.

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Chapter 5

The impact of Ecological Construction Programs on grassland conservation in the pastoral areas of China

5. THE IMPACT OF ECOLOGICAL CONSTRUCTION PROGRAMS ON GRASSLAND CONSERVATION IN THE PASTORAL AREAS OF CHINA²⁴

ABSTRACT: A series of Ecological Construction Programs have been initiated to protect the condition of grasslands in China during recent decades. However, grassland degradation is still severe and conditions have not been restored as intended. This paper aims to empirically examine the effectiveness of Ecological Construction Programs for protecting the grassland condition in the extensive pastoral areas of China. We focus on one Ecological Construction Program that has been implemented widely on the grasslands, the Subsidy and Incentive System for Grassland Conservation (SISGC). The Normalized Difference Vegetation Index (NDVI), measured with remote sensing technology, is used to quantify the grassland condition between 2001 and 2014. With data from 54 counties in the pastoral areas of Inner Mongolia, we estimate the impact of SISGC on the grassland condition. A fixed effects model is employed to control for livestock production, climate, time trends and time-invariant heterogeneity between counties. The model results provide quantitative evidence that the condition of the grasslands has improved significantly due to SISGC, but that the effectiveness of SISGC was offset to some extent by other socio-economic and climate factors, such as increased producer prices and high temperature. This may explain why the actual grassland degradation has not been prevented as effectively as was expected. In addition, the impact of SISGC was stronger in counties with a worse initial grassland condition. Furthermore, the effects of producer prices and climate changes were also more pronounced in these counties.

KEYWORDS: Ecological Construction Programs, grassland degradation, NDVI, fixed effects model, Inner Mongolia

²⁴ Paper by Min Liu, Liesbeth Dries, Wim Heijman, Jikun Huang, Xueqin Zhu, Yuanning Hu, and Haibin Chen, has been accepted by *Land Degradation and Development*.

5.1 Introduction

Grasslands are important as a feed source for livestock, as a habitat for wildlife, for environmental protection and for the in situ conservation of plant genetic resources (FAO, 2008). However, rapid increases in livestock populations have increased pressures on the world's grasslands, leading to widespread deterioration, particularly in arid and semi-arid environments (Suttie et al., 2005). China has around 392 million hectares of grasslands, accounting for 12% of the world's grasslands and 41.7% of the national land area (Fan et al., 2008). Nearly 80% of these grasslands are in arid and semi-arid regions (National Bureau of Statistics of China, 2009). Approximately 17 million herders and agro-herders maintain their livelihoods on the grasslands in China (Li et al., 2014). According to Chinese governmental reports, 10% of the total area of grasslands was degraded in the 1970s, increasing to 30% in the 1980s and 50% in the middle of the 1990s. By the 2000s, about 90% of the grasslands were degraded to various extent, with significant regional variation (Unkovich and Nan, 2008; Waldron et al., 2010). The manifestation of grassland degradation includes initial lowering of grassland productivity, fragmentation of grass cover, reduction in soil fertility, soil compaction, an increase in unpalatable grass species or a combination of all of them (Feng et al., 2009; Li et al., 2013). Grassland degradation, especially in northern China, is considered to be the cause of serious environmental and ecological problems, such as Yangtze River floods, Yellow River droughts and sand storms of Beijing (Harris, 2010). Grassland degradation therefore threatens not only animal husbandry on the grasslands but also the livelihoods of millions of people and the ecological security of China (Huang et al., 2013). To address these problems, national and local governments have initiated an ambitious series of Ecological Construction Programs for grassland protection in recent decades.

In 1985, China passed the first national Grassland Law which explicitly stipulated the protection and improvement of grasslands (Ho, 2000). The Law devolved grassland use rights from the state and collectives to individual households through long-term contracts, and introduced maximum stocking rates for livestock on the grassland areas. Since the serious drought in China in 1997 and the massive floods of 1998, additional Ecological Construction Programs have been introduced, for example, the Conversion of Cropland to Forest and Grassland Program (also known as the Grain for Green Program), the Returning Grazing to Grassland Program, and the Program to Combat Desertification around Beijing and Tianjin, which aimed at preventing grasslands from degradation and were actively implemented in the 2000s (Liu et al., 2010; Hua and Squires, 2015). Since 2011, another major Ecological Construction Program, the Subsidy and Incentive System for Grassland Conservation (SISGC), has been initiated in eight pastoral provinces of China, including Inner Mongolia, Xinjiang, Tibet, Qinghai, Sichuan, Gansu, Ningxia and Yunnan (State Council of the People's Republic of China, 2010). Compared to the Ecological Construction Programs for grassland conservation before 2011, the SISGC includes much

wider areas that involve most of the available natural grasslands of China and provides herders with more ecological compensations. Moreover, its monitoring system for policy implementation has been improved with the use of more targeted monitoring staffs (Guan Hu Yuan) that are employed by local governments.

In general, China's Ecological Construction Programs include two categories of measures for grassland conservation, namely sowing grass or planting trees in the eco-fragile areas to restore the degraded grasslands and reducing livestock grazing to prevent grassland degradation (Ho and Azadi, 2010; Kou et al., 2016). This coincides with the measures for preventing desertification in China: controlling desertification by plantation and combating desertification by natural recovery through isolating the degraded area from external human influences (Miao et al., 2015). In practice, these two measures cover a vast array of regulations (Waldron et al., 2010). Particularly, the reduction in grazing is targeted by regulations determining stocking rate and grazing bans (either permanently, temporarily or seasonally) (Qu et al., 2011). The stocking rate is based on the theoretical maximum of the available biomass production from grasslands that can be consumed by grazing herds without impairing the capacity of the pasture to regrow the following year (Fernández-Giménez et al., 2011).

Despite the fact that Ecological Construction Programs have been widely introduced in the pastoral areas of China for more than ten years, grassland degradation is still severe (Waldron et al., 2010; Li and Huntsinger, 2011; Ministry of Agriculture of China, 2012; Li et al., 2014). Although the average vegetation coverage of China's grasslands increased by 0.4% between 2014 and 2015, and there has been an increase of 3% since 2011, more than one third of the grasslands are still suffering moderate and serious degradation (Ministry of Agriculture of China, 2014; 2016). Some academic studies have therefore criticized the effectiveness of the Ecological Construction Programs on grassland conservation, and attributed the causes of this ineffectiveness to inappropriate policy measures and failed implementation (Li and Zhang, 2009). For example, some authors argue that the determination of a maximum stocking rate and various grazing bans are inappropriate management measures for the conservation of grasslands in arid and semi-arid areas (Higgins et al., 2007). In particular, deterministic stocking rates are questioned because they largely ignore the grasslands' spatial heterogeneity and climate variability (Li and Huntsinger, 2011). Moreover, it is claimed that appropriate grazing is beneficial to plant succession processes (Zhang et al., 2015). Deterministic stocking rates have also led to resistance from local herders that are unwilling to limit their own livestock numbers based on a top-down government measure (Li and Zhang, 2009). High supervision costs of the implementation of the policy make that deterministic stocking rates and grazing bans are difficult to sustain effectively for a long period, especially in extensive grassland areas with a scattered population distribution (Li and Zhang, 2009; Waldron et al., 2010). Considering these criticisms and the fact that the grassland

condition is still deteriorating, despite the current policy measures, this paper will examine the effectiveness of Ecological Construction Programs on grassland protection and discuss the potential factors that impact the policy effectiveness.

Some academic studies and governmental reports discuss the impacts of Ecological Construction Programs. A number of them focus on limited regions based on household-level or experimental field data (e.g. Gu and Li, 2013; Zhang et al., 2015). Others conduct spatial and temporal comparisons but without controlling for the possible impacts of other socio-economic and climate factors on the grassland condition. Some studies also have systematically discussed the impacts of particular aspects of eco-environmental policy interventions, including grassland laws (Nelson, 2006), grazing bans (Yeh, 2005), and herder resettlement policies (Dickinson and Webber, 2007), but they have mainly used qualitative analysis. In short, the existing literature lacks estimations on the effectiveness of Ecological Construction Programs for grassland conservation that are based on large-scale and long-term samples and control for the impacts of other potential factors through a quantitative analysis. This paper employs an econometric model based on data from 54 counties in the extensive pastoral areas of Inner Mongolia from 2001 to 2014 to disentangle the impact of one specific Ecological Construction Programs, namely the SISGC, on the grassland condition. In the following, the research region, data collection and methods are described. Next, the empirical results about the relationship between SISGC and grassland condition are presented and interpreted. Ultimately, we conclude with an assessment of the effectiveness of SISGC for grassland conservation based on the comparison between the estimated results from our model and the actual condition of the grassland.

5.2 Material and methods

5.2.1 Research region

We conduct our empirical study based on Inner Mongolia, which is located in northern China and in which animal husbandry is the traditional and dominant agricultural industry. Most of the population are indigenous Mongolians and are reliant on grazing animals on the natural grasslands in order to maintain a livelihood. Inner Mongolia belongs to the arid and semi-arid areas of China and contains 21.7% of the area of China's natural grasslands. Approximately 67% of the total land in Inner Mongolia is classified as grassland, the majority of which can be sub-classified as temperate grassland (Angerer et al., 2008). The current administrative divisions of Inner Mongolia include 102 counties, consisting of 33 pastoral counties and 21 semi-pastoral counties. The remaining 48 counties are dominated by crop farming or urban districts. The agricultural industry of pastoral counties is characterized by its extensive grazing; natural grassland is the dominant land type. In semi-pastoral counties, both natural grassland and cropland are the dominant land types, in which there is more intensive animal husbandry along with cropping (Waldron et al., 2010). The pastoral and semi-pastoral counties include almost all of the natural

grasslands of Inner Mongolia. Figure 5.1 illustrates the location of Inner Mongolia in China and the distribution of its pastoral and semi-pastoral counties.

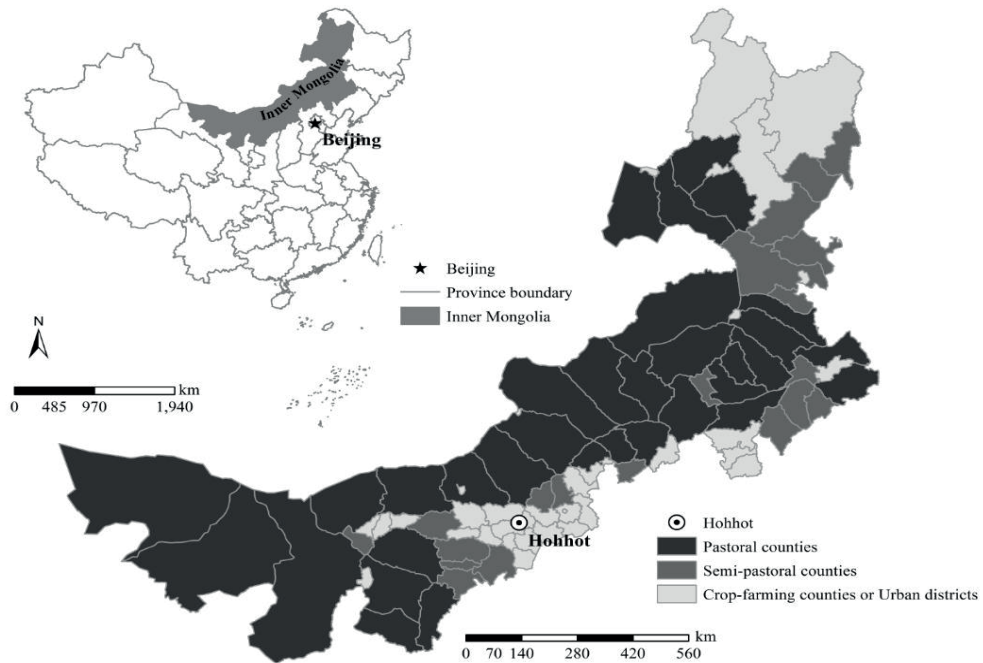


Figure 5.1 Inner Mongolia and its pastoral and semi-pastoral counties

According to government reports, by the year 2000, 90% of the natural grasslands of Inner Mongolia had been degraded to some extent, reflected by the degraded composition of plant species, declining biodiversity, accelerated soil erosion etc. (Waldron et al., 2010; Briske et al., 2015). Moreover, the results of large-scale field ecological surveys highlighted that the average grassland biomass productivity in Inner Mongolia has reduced from 1871 kg/ha in 1961 to 900 kg/ha in 2010 (Wang et al., 2013). The extensive grasslands of Inner Mongolia play an important role as ecological barriers for northern China. For instance, the colossal dust storms which rumbled through hundreds of cities and villages of northern China and blanketed the sky of Beijing between 1998 and 2001 were said to have originated from dryland areas and degraded grasslands mainly in Inner Mongolia (Wu et al., 2015). Subsequently, the pastoral areas of Inner Mongolia have become the typical regions for implementing various Ecological Construction Programs for grassland conservation in China since the early 2000s.

Since 2011, the SISGC has been initiated in Inner Mongolia. Considering that the SISGC is currently the most important large-scale Ecological Construction Program for grassland conservation and inherits the main management measures from earlier programs, namely the deterministic stocking rate and permanent grazing ban, we take it as a typical example to explore the effectiveness of Ecological

Construction Programs in the extensive pastoral areas of China. It is reported by the local government that 67.3 million hectares of available natural grasslands are covered by SISGC, which includes almost all of the available natural grassland areas of Inner Mongolia. 30 million hectares of them belong to the area with the permanent grazing ban and the remaining 37.3 million hectares belong to the forage-livestock balance area. In practice, every county in the pastoral areas contains different areas of grasslands which are subject to the permanent grazing ban, deterministic stocking rate or both of them. Almost all rural households in the pastoral areas have been covered by SISGC, and they have been given subsidies (or incentives) as well as the obligation to comply with the permanent grazing ban (or the deterministic stocking rate).

5.2.2 Data collection

The empirical model is conducted using county-level data to estimate the impact of the SISGC on the condition of grasslands in the pastoral areas of Inner Mongolia. The dataset includes 54 counties consisting of the 33 pastoral counties and 21 semi-pastoral counties, and spans 14 years from 2001 to 2014 including the period before and after the introduction of the SISGC. The data collected include indicators of the grassland condition, the status of SISGC, and climate and socio-economic factors. In the existing literature, the grassland condition has been indicated by vegetation coverage, height, density, biomass production and density of perennial vegetation (Gu and Li, 2013; Yu and Farrell, 2013). These indicators are commonly measured using two methods: direct sampling tests undertaken during small-scale fieldwork; and estimation over large areas with remote sensing technology (Gu and Li, 2013). The former focuses on measuring the micro-indices of grass, while the latter estimates the macro grassland condition on the basis of satellite images. For this study, the remote sensing technology appeared more feasible and appropriate considering that we aim to quantify the grassland condition of the 54 counties in Inner Mongolia over 14 years. The Normalised Difference Vegetation Index (NDVI) is a widely-used indicator to quantify the grassland condition on the basis of the remote sensing technology (e.g. Yang et al., 1998; Liu et al., 2013; Gong et al., 2015). Theoretically, the measurement of NDVI is based on the differential absorption, transmittance and reflectance of energy by the vegetation in the red and near-infrared portions of the electromagnetic spectrum (Senay and Elliott, 2000). NDVI is calculated on a per-pixel basis as the ratio of the difference of the near infrared and red bands over their sum from a remotely-sensed image. It provides a clear description of land surface features and is regarded as a proxy for terrestrial vegetation condition because it is strongly associated with percentage of vegetation coverage, leaf area index, potential photosynthesis, aboveground net primary productivity and biomass availability (Tan and Li, 2015). Based on the data set of MOD13A1 from NASA's Earth Science Data Systems Program, which is the Moderate Resolution Imaging Spectroradiometer (MODIS) imagery with 500m spatial resolution and 16 days temporal resolution, we employed the Maximum Value

Composite (MVC) method to get the NDVI of each year during 2001-2014 (Holben, 1986). Ultimately, the image layer of administrative divisions of Inner Mongolia is used to determine the NDVI for each county.

Information about SISGC was provided by the Agriculture and Animal Husbandry Bureau of Inner Mongolia. Moreover, interviews were conducted with local herders and public officers who are key informants on the implementation of Ecological Construction Programs in Inner Mongolia to improve our understanding of these programs and their possible impacts. The data on socio-economic indicators was collected based on existing statistical data gathered by national and local Statistical bureaus. For instance, the producer price of live sheep was collected based on the Annual Compilation of Cost-benefit Data of Chinese Agricultural Products and deflated with the Producer Price Index of Agricultural Products of Inner Mongolia. The data on climate indicators were gathered from the Statistical Yearbooks of Inner Mongolia. It should be noted that we used the city-level climate data to represent the temperature and precipitation of each county that affiliates to its corresponding city.

5.2.3 Statistical analysis on grassland condition

Based on the data of NDVI over the 54 counties of Inner Mongolia from 2001 to 2014, table 5.1 outlines the changes in grassland condition at different time intervals.

Table 5.1 Grassland condition (NDVI) at different time intervals of SISGC

Periods	Status of SISGC	Obs.	Mean	Std. Dev.	Min	Max	Welch's t-test	
Period 1 (2001-2010)	Before SISGC	540	0.467	0.199	0.074	0.853	na	
Period 2 (2011-2014)	After SISGC	216	0.502	0.201	0.083	0.851	-0.035**	
Period 2	2011	The first year of SISGC	54	0.482	0.208	0.084	0.851	-0.015
	2012	The second year of SISGC	54	0.528	0.186	0.085	0.850	-0.061**
	2013	The third year of SISGC	54	0.520	0.203	0.084	0.840	-0.053*
	2014	The fourth year of SISGC	54	0.480	0.208	0.083	0.850	-0.013

Sources: Statistics are derived from the data set of NASA's Earth Science Data Systems Program
 Note: *significant at 10%; **significant at 5%; ***significant at 1%.

According to the Welch's t-test on NDVI between periods 1 and 2 (Lee, 1992), the overall mean of NDVI is significantly larger in the period after SISGC than before, which implies an improvement in overall grassland condition. Moreover, all of the mean NDVI values are larger in the years with SISGC than the overall mean of NDVI in the period before SISGC. However, the Welch's t-test on NDVI

between period 1 and each year after SISGC indicates that the difference is only significant in the years 2012 and 2013.

5.2.4 Fixed effects model

A fixed effects model has been widely used in economic research, primarily to study the causes of changes within entities over time (e.g. Fergusson et al., 2002; Huang et al., 2006). The model employs within transformation to remove all time-invariant (fixed) explanatory variables, i.e. the model is performed in deviations from individual means (Verbeek, 2012). As such, the fixed effects model provides a method that takes observable explanatory variables as well as unobservable time-invariant variables into account, but the estimation does not depend on the value of time-invariant (fixed) variables (Verbeek, 2012). Such an approach is appropriate for this study because the causes of grassland condition changes within each county over time can be studied by controlling for the measured as well as unmeasured time-invariant heterogeneity between counties. In addition, our research samples include all of the counties in the pastoral areas of Inner Mongolia, rather than random draws, which preliminarily indicates that the fixed effects model is more appropriate than the random effects model (Verbeek, 2012). Nevertheless, the appropriateness of the fixed effects model will be further examined through statistical testing. Based on the theoretical framework of the fixed effects model, we formulate the impacts of SISGC and other potential factors on grassland condition in the following equation:

$$G_{it} = h_i + \sum_{y=2011}^{2014} a_y P_{iy} + \sum_{y=2011}^{2014} b_y (P_{iy} * D_i) + c_1 S_{it-2} + \sum_{m=4}^9 d_m T_{itm} + d_1 TV_{it} + \sum_{m=4}^9 e_m R_{itm} + e_1 RV_{it} + f_1 Y_t + g_1 (Y_t * D_i) + \varepsilon_{it} \quad (1)$$

where i and t indicate the i^{th} county and year t , from 2001 to 2014. Grassland condition (G_{it}), presented by NDVI, is employed as the dependent variable of equation (1). P_{iy} is a dummy variable to manifest whether county i implemented SISGC in the specific year y which involves the first year with SISGC ($y=2011$), the second year ($y=2012$), the third year ($y=2013$) and the fourth year ($y=2014$). The interaction term of SISGC (P_{iy}) and the pastoral or semi-pastoral attribute of county i (D_i) is considered.

Besides the policy variables, other potential factors are controlled. For example, the livestock production and climate factor are prevalently discussed as the drivers that impact the grassland condition in the arid and semi-arid areas (e.g. Li et al., 2012; He et al., 2014). In this study, the real producer price of live sheep (S_{it}) is used as the proxy variable of livestock production in the pastoral areas because of the limitations of the data. More specifically, the livestock population in the pastoral areas was under-reported by herders due to the stocking rate regulation (Brown et al., 2008). Furthermore, in the

statistical data on the meat output of the pastoral areas of Inner Mongolia, no distinction is made between outputs from extensive grazing, and outputs from industrial raising and intensive animal husbandry. Since sheep is the animal which is most often raised by local herders (Zhang et al., 2012), the real producer price of live sheep with a two-year lag (S_{it-2}) is used to indicate the situation of livestock production in the empirical model. In this regard, the price is exogenously decided by the global market of China, and the two-year lag is considered as the timespan during which local herders adjust their sheep population to respond to the changes in sheep price. According to information from our household survey in Inner Mongolia, the timespan is around two years because it covers the breeding period of breeders; herders then reserve more new lambs in order to expand their sheep population during the next production stage. The climate factor is elaborated by the monthly average temperature (T_{itm}) and precipitation (R_{itm}) during the growing season of grass, including the six months from April to September with $m=4$ to 9 (Zhao and Guo, 2009), and the variance of annual average temperature (TV_{it}) and precipitation (RV_{it}). The annual variance is presented by the standard deviations of monthly average temperature and precipitation in all twelve months of each year. In addition, time trend (Y_t) is included considering the impact of factors that change with time, such as the development of intensive animal husbandry and the increasing area of artificial grass for grassland restoration. The interaction term of time trend and the pastoral or semi-pastoral attribute of county i ($Y_t * D_i$) is also considered in order to explore whether there are differences in the grassland condition with time change between pastoral and semi-pastoral counties. Finally, the factors, such as elevation, slope, soil type, distance to the provincial capital etc., that reflect the heterogeneity of grasslands and do not change over time are treated as time-invariant (fixed) factors in the model, and are denoted by h_i in equation (1). $a_y, b_y, c_1, d_m, d_1, e_m, e_1, f_1$ and g_1 are the coefficients of the independent variables and ε_{it} is the random error term. The specific definition of these variables is stated in table 5.2.



Table 5.2 Variable definitions and their expected signs in the model

Variables	Variable definition	Unit	Expected signs
G_{it}	NDVI of county i in year t	na	Dependent variable
P_{iy}	=1 if county i implemented SISGC in year y (y=2011, 2012, 2013 and 2014), =0 otherwise	na	+
D_i	=1 if county i is a pastoral county, =0 otherwise	na	na†
S_{it}	The ratio of real producer price of live sheep in county i between year t and year 2001	na	-
T_{itm}	The average temperature of county i in month m of year t (m=4, 5, 6, 7, 8 and 9)	Degrees Celsius	+/-
TV_{it}	The temperature variation of county i in year t	Degrees Celsius	+/-
R_{itm}	The total precipitation of county i in month m of year t (m=4, 5, 6, 7, 8 and 9)	Millimeters	+
RV_{it}	The precipitation variation of county i in year t	Millimeters	+/-
Y_t	Year t	na	+

Note: † indicates that D_i is included in the interaction terms of $(P_{iy} * D_i)$ and $(Y_t * D_i)$. The text presents the expected signs of $(P_{iy} * D_i)$ and $(Y_t * D_i)$

According to the existing literature, we propose the expected signs of the explanatory variables on grassland condition in table 5.2. Given that the government reports and most of academic findings present that SISGC has improved the grassland condition of Inner Mongolia (Li et al., 2014; Ministry of Agriculture of China, 2016), we expect positive coefficients for P_{iy} in each year when SISGC was implemented. The expected signs of the interaction terms $(D_i * P_{iy})$ in our estimations are ambiguous because it is uncertain whether the effectiveness of SISGC is better in the pastoral counties than in the semi-pastoral counties or vice versa. Additionally, the real producer price of live sheep (S_{it}) is expected to have negative effects on the grassland condition because higher prices stimulate more livestock production that may result in excessive grazing pressure and further lead to grassland degradation (Zhang et al., 2014). The expected signs of monthly average temperature (T_{itm}) in the growing season are ambiguous because the high temperature facilitates plant growth as well as the evaporation of the ground surfaces and the evapotranspiration from the vegetation (Deng et al., 2013), and fast evaporation and evapotranspiration have negative effects on plant growth in the arid and semi-arid areas. Given that precipitation is considered as the crucial factor in improving grassland condition in the arid and semi-arid areas (Li and Zhang, 2009), we expect positive coefficients for R_{itm} in the model. The expected signs of the annual variation in temperature (TV_{it}) and precipitation (RV_{it}) are ambiguous as the uneven distribution of climate has uncertain effects on grass growth. Time variable (Y_t) is expected to have positive effects on grassland condition. For instance, the development of intensive animal husbandry is supposed to replace extensive grazing and further reduce the use and damage of natural grasslands.

Furthermore, the interaction term of time variable and the attribute of county i ($Y_t * D_i$) is expected to have a negative coefficient because the pastoral counties experienced less intensive animal husbandry than the semi-pastoral counties (Waldron et al., 2010).

In addition, we expect that these explanatory variables may have different effects in different types of counties, especially for counties that have a different initial grassland condition. It is suggested that the effectiveness of large-scale ecological conservation programs has been heterogeneous in different regions even under the same policy context with incentive measures (Lv et al., 2015). The 54 counties in the sample are therefore divided over two groups based on the NDVI of each county in 2001. This results in 27 counties in the low NDVI group and 27 counties in the high NDVI group. NDVI of the low NDVI group in 2001 ranges from 0.074 to 0.427, and the mean is 0.254. The range of NDVI in the high NDVI group in 2001 is between 0.453 and 0.836, and the mean is 0.614. The fixed effects model (equation (1)) will be conducted separately for three groups of counties: all counties; low NDVI counties; high NDVI counties.

5.2.5 Descriptive statistics for the variables

Table 5.3 shows the characteristics of the variables included in the fixed effects model. The overall mean of NDVI (G_{it}) is 0.477 for all 54 counties from 2001 to 2014, 0.314 for the 27 low NDVI counties, and 0.641 for the 27 high NDVI counties. Because all counties implemented SISGC in 2011, therefore the mean value of P_{i2011} over 14 years is 0.071. The mean value of $D_i * P_{i2011}$ (0.044) indicates that the percentage of samples from the pastoral counties in 2011 (33) is 4.4% of the total observations (756).

Table 5.3 Descriptive statistics of the variables in the fixed effects model

Variables	All counties			Low NDVI counties			High NDVI counties		
	Mean	Std. Dev.	Obs.	Mean	Std. Dev.	Obs.	Mean	Std. Dev.	Obs.
G_{it}	0.477	0.200	756	0.314	0.125	378	0.641	0.104	378
P_{i2011}	0.071	na	756	0.071	na	378	0.071	na	378
P_{i2012}	0.071	na	756	0.071	na	378	0.071	na	378
P_{i2013}	0.071	na	756	0.071	na	378	0.071	na	378
P_{i2014}	0.071	na	756	0.071	na	378	0.071	na	378
$D_i * P_{i2011}$	0.044	na	756	0.048	na	378	0.040	na	378
$D_i * P_{i2012}$	0.044	na	756	0.048	na	378	0.040	na	378
$D_i * P_{i2013}$	0.044	na	756	0.048	na	378	0.040	na	378
$D_i * P_{i2014}$	0.044	na	756	0.048	na	378	0.040	na	378
S_{it-2}	1.530	0.598	648	1.530	0.599	324	1.530	0.599	324
T_{it4}	7.967	3.318	756	7.333	3.850	378	8.602	2.535	378
T_{it5}	15.529	2.362	756	15.066	2.706	378	15.992	1.849	378
T_{it6}	20.813	2.055	756	20.750	2.227	378	20.877	1.868	378
T_{it7}	22.833	1.667	756	22.720	1.853	378	22.946	1.451	378
T_{it8}	21.035	1.862	756	20.905	2.032	378	21.164	1.668	378
T_{it9}	15.000	2.067	756	14.593	2.404	378	15.407	1.564	378
TV_{it}	13.076	na	756	13.683	na	378	12.470	na	378
R_{it4}	14.965	15.912	756	13.287	14.345	378	16.644	17.193	378
R_{it5}	29.526	22.371	756	26.623	20.128	378	32.429	24.088	378
R_{it6}	57.486	38.445	756	50.960	36.319	378	64.013	39.439	378
R_{it7}	81.173	51.808	756	72.722	49.640	378	89.624	52.611	378
R_{it8}	57.617	39.026	756	54.424	40.651	378	60.810	37.110	378
R_{it9}	34.058	26.697	756	29.750	21.998	378	38.366	30.103	378
RV_{it}	31.864	na	756	29.134	na	378	34.594	na	378
Y_t	7.500	4.034	756	7.500	4.036	378	7.500	4.036	378
$Y_t * D_i$	4.583	na	756	5.000	na	378	4.167	na	378

Note: The mean of dummy variables refers to the share of observations for which the variable equals 1.

Sources: statistics are derived from Statistical Yearbooks and our survey.

5.3 Results

5.3.1 Model results

The Hausman test is employed to determine whether the fixed effects model is more appropriate than the random effects model. The Hausman test rejected the null hypothesis at $p < 0.05$, indicating that the fixed effects model is more appropriate for our estimation.

Table 5.4 Estimated results of the fixed effects model

Variable	All counties		Low NDVI counties		High NDVI counties	
	Coefficient	t	Coefficient	t	Coefficient	t
P_{i2011}	0.042***	2.83	0.025	0.81	0.056**	2.60
P_{i2012}	0.130***	7.36	0.153***	5.46	0.110***	3.74
P_{i2013}	0.189***	7.33	0.210***	3.96	0.185***	4.07
P_{i2014}	0.150***	4.91	0.163***	3.14	0.141***	3.24
$D_i * P_{i2011}$	0.016	1.20	0.035	1.23	0.008	0.64
$D_i * P_{i2012}$	0.018	1.24	0.016	0.81	0.022	0.88
$D_i * P_{i2013}$	0.012	0.78	0.014	0.43	0.016	0.61
$D_i * P_{i2014}$	0.010	0.62	0.026	1.29	0.014	0.64
S_{it-2}	-0.110***	-5.11	-0.161***	-3.94	-0.065*	-2.04
T_{it4}	-0.001	-1.02	0.001	0.34	-0.001	-0.71
T_{it5}	-0.008***	-3.17	-0.001	-0.46	-0.015***	-4.91
T_{it6}	-0.007***	-3.66	-0.003	-0.72	-0.010***	-3.78
T_{it7}	-0.005	-1.52	-0.011**	-2.64	0.003	0.57
T_{it8}	-0.006***	-2.83	-0.008***	-3.14	-0.001	-0.17
T_{it9}	0.0003	0.09	-0.001	-0.27	-0.0003	-0.05
TV_{it}	-0.001	-0.59	-0.004**	-2.69	0.009***	2.85
R_{it4}	-0.0002	-1.21	-0.0001	-0.68	-0.0001	-0.88
R_{it5}	0.0003***	3.07	0.001***	3.04	0.0001	0.96
R_{it6}	-0.00001	-0.09	0.0001	0.73	-0.0001	-0.84
R_{it7}	-0.0001	-0.94	-0.000005	-0.03	-0.0001	-0.60
R_{it8}	0.00005	0.57	0.0002**	2.11	0.0001	0.72
R_{it9}	0.0001	1.23	0.0004***	2.79	-0.00003	-0.26
RV_{it}	-0.0001	-0.30	-0.001*	-1.74	0.0002	0.34
Y_t	0.005***	2.61	0.012***	3.96	-0.003	-1.24
$Y_t * D_i$	-0.003*	-1.69	-0.004	-1.10	-0.003	-0.90
Constant	1.085	10.02	0.985***	6.36	1.039***	5.51
R^2	0.3913		0.4483		0.5156	
Observations	648		324		324	

Note: *significant at 10%; **significant at 5%; ***significant at 1%.

Table 5.4 presents the estimated results of the fixed effects model for the three groups of counties. In the group of all counties, the coefficients of the variables about SISGC in each year are significant and positive. Specifically, the significant coefficient of P_{i2011} indicates that

NDVI increases by 0.042 in the first year of SISGC when the effects of other potential factors on NDVI are controlled. Similarly, NDVI increases by 0.13 until the second year of SISGC, 0.189 until the third year, and 0.15 until the fourth year. The effectiveness of SISGC is accumulated year by year. Therefore, the effectiveness of SISGC on NDVI in the first year is increased by 0.042, 0.088 for the second year and 0.059 for the third year. It is remarkable that the effectiveness of SISGC in the fourth year is -0.039, although the accumulated effectiveness of SISGC on NDVI is a 0.15 increase by then. The interaction terms of SISGC in each year and pastoral attribute of the county are not significant, which indicates that there are no significant differences in the impacts of SISGC on NDVI between pastoral counties and semi-pastoral counties. With respect to the effects of other potential factors, the variable of the real producer price of live sheep presents significant and negative impacts on NDVI. It indicates that a 1% increase in the real producer price in year $t-2$ relative to the price in 2001 causes a decrease of 0.11 in NDVI when the factors of SISGC, climate, time trend and time-invariant heterogeneity between counties remain constant. Moreover, the temperature in May, June and August has significant and negative effects on NDVI. This is explained through the increasing temperature causing faster evaporation from the ground surfaces and evapotranspiration from the vegetation, and allowing more pests and mice to breed in the arid and semi-arid areas, which are significantly detrimental to grassland condition in the growing season (Li, 2009; Li et al., 2013). Precipitation in May, on the other hand, has significant and positive effects. Abundant precipitation is beneficial to the restoration of grassland condition (Li and Zhang, 2009). The grassland condition is improved slowly with the factors that change with time trend, and the effects of time variance on grassland condition are stronger in the semi-pastoral counties than in the pastoral counties.

Comparing the model results between the groups of low NDVI counties and high NDVI counties, the coefficient of the variable of SISGC in the first year is not significant in the group of low NDVI counties. This may be explained by the lagged implementation of SISGC in the low NDVI counties considering that these counties are not first regions for initiating ecological construction programs and are usually located in the relatively remote areas. However, the coefficients of the variables of SISGC in the second, third and fourth years are larger in the group of low NDVI counties than in the group of high NDVI counties. This indicates that SISGC has stronger impacts on the grassland conservation in the low NDVI counties than in the high NDVI counties. This result is consistent with the implementation planning of SISGC, namely, grazing bans that provide higher subsidies and implement more strict constraints for

grazing are conducted in the areas with more severe degradation or inferior grassland condition. Interestingly, the significant and negative coefficient of the real producer price indicates that a 1% increase causes a 0.161 decrease in NDVI in the low NDVI counties, but only a 0.065 decrease in NDVI in the high NDVI counties. Market forces seem to have a stronger impact in the counties with worse initial grassland condition. Similarly, climate changes seem to have more impact in the counties with worse initial grassland condition.

5.3.2 Comparison between the estimated results and actual grassland condition

Figure 5.2 depicts the estimated NDVI based on the significant coefficients of the variables of SISGC in the model including all counties. The mean value of actual NDVI in the period 2001-2010 is used as the initial value. The actual NDVI is drawn from the mean value of NDVI of the 54 counties in the periods 2001-2010, 2011, 2012, 2013 and 2014.

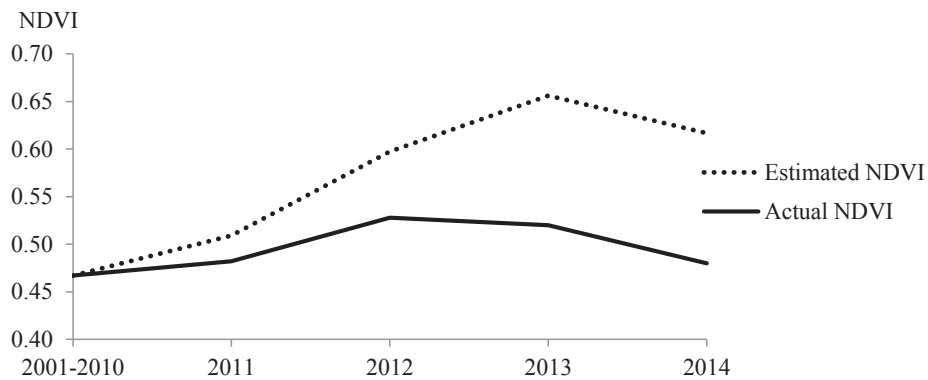


Figure 5.2 Comparison between the estimated NDVI and actual NDVI

Sources: The data of actual NDVI is derived from the data set of NASA’s Earth Science Data Systems Program

The estimated NDVI and actual NDVI in each year of SISGC are larger than the initial value. Comparing the estimated NDVI and actual NDVI in figure 5.2, it shows that the actual NDVI is clearly lower than the estimated NDVI during the years of SISGC. The differences between the actual and estimated grassland condition are attributed to the other potential factors that offset the effectiveness of SISGC, including the producer price and climate factors which showed significant coefficients in the model results. For instance, data showed that the producer price continuously increased in the period 2011 to 2014 from 1.48 times to 2.54 times the producer price in 2001. Although the large increase in the producer price cannot impact upon

grassland condition directly, strong market forces may stimulate herders to raise much more livestock. Increased grazing pressure following the large increase in the producer price induces significantly negative impacts on the grassland condition that reduces or neutralises the positive impacts stemming from SISGC, with the result that the actual grassland condition under the SISGC is not as good as expected.

5.4 Discussion

The effectiveness of SISGC with first an increasing and then a decreasing trend is in line with the results of existing research that questioned the failed implementation and ineffective management measures of Ecological Construction Programs (Bao and Chen, 1997; Waldron et al., 2010). For example, it is suggested that top-down management entails high supervision costs, making the implementation of the Ecological Construction Programs difficult to sustain for a long period effectively (Li and Zhang, 2009; Waldron et al., 2010). In our case, the local governments, who follow the instructions of the national government as planned at the beginning of SISGC, encountered increasing supervision costs when the producer price of live sheep soared especially in the third and fourth years of SISGC which prompted local herders to gain more economic benefits by increasing illegal grazing. It thus may incur the implementation of SISGC failure during the later years of SISGC. With respect to the ineffective management measures, it has been proved that grassland is in a better condition, including plant species diversity and herbage mass, during the early years of the permanent grazing ban, while there is a subsequent decline or level-off during its later years (Yang et al., 2005; Zhang et al., 2015). The ecological explanation for this is that annual and biennial vegetation could invade during the early years of permanent grazing ban, which would increase species richness. However, the subsequent recovery of dominant species increases their prevalence and further leads to a decrease in biodiversity in the next years (Bao and Chen, 1997). Considering that the permanent grazing ban is widely employed in SISGC, the fact that the grassland condition showed a gradual increase at first and then decreased under the impact of SISGC may correspond to this process of ecological succession.

In addition, we would like to point out several limitations of our research. Firstly, the four years of SISGC which we studied may not be long enough to estimate the changing trend of its effectiveness on grassland condition. Secondly, we used dummy variables to present the status of SISGC in our model due to data limitations. Indicators that involve specific differences of SISGC among counties, such as grassland areas under SISGC, amount of subsidies and

enforcement strength of each county, would be more meaningful for estimating the effectiveness of policy intervention than the dummy variables. In our case, because all of the counties in the sample have participated in SISGC since 2011, the dummy variables of SISGC do not distinguish the specific differences of implementation of SISGC between counties and between years, and only the differences between before and after SISGC are presented. Moreover, we considered SISGC as a single policy, rather than investigating the results of its two specific regulations, deterministic stocking rates and permanent grazing bans, separately. Our research thus only presented the general result of SISGC. Although we showed that SISGC has a stronger impact in low NDVI counties, the impacts of specific measures need to be explored more thoroughly. Finally, our empirical model does not allow disentangling the impacts of other Ecological Construction Programs that were mainly implemented before SISGC started, such as the Conversion of Cropland to Forest and Grassland Program, Returning Grazing to Grassland Program, or the Program to Combat Desertification around Beijing and Tianjin.

5.5 Conclusions

This paper began by sketching a review about the policy interventions for grassland protection in China, involving the national Grassland Law and a series of Ecological Construction Programs. And then we took the latest Ecological Construction Program (SISGC) as an example to examine its effectiveness for grassland conservation in the extensive pastoral areas over the long term. The grassland condition was presented by the NDVI, rendered by remote sensing technology. The situation of SISGC was defined by dummy variables, which presented the situation of each year with SISGC. The fixed effects model was used to control for the factors that obscure the relationships between grassland condition and SISGC, including livestock production, climate factor, time trend and time-invariant heterogeneity between counties. Based on the data of 54 sample counties in the pastoral areas of Inner Mongolia from 2001 to 2014, the empirical model results provided quantitative evidence that grassland condition has been generally improved under the auspices of SISGC. The comparison between the actual NDVI and the estimated results show that the actual grassland condition is worse than the estimated results when other factors are assumed constant. One explanation for this is that the effectiveness of SISGC on grassland condition is offset by the impact of other potential factors, such as producer price and climate changes. Moreover, the impact of SISGC on the grassland condition is stronger in the low NDVI counties, while the offset from producer price and climate factors is more pronounced in these counties as well. In conclusion, our research findings are

that the Ecological Construction Programs such as SISGC are effective for grassland conservation, but that the effectiveness on grassland protection is reduced or neutralised by other socio-economic and climate factors. For future policy design, an improvement in the Ecological Construction Programs should take into account the potential offsetting forces from other factors related to market and climate conditions.

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Chapter 6

Discussion and conclusions

6. DISCUSSION AND CONCLUSIONS

6.1 Introduction

China is faced with grassland degradation, rapid increases in demand for livestock products and high levels of poverty among rural households. This dissertation outlines China's grassland policies and studies the outcomes of the policy interventions for grassland systems in Inner Mongolia. More specifically, this thesis focuses on evaluating the effects of the land tenure reform and Ecological Construction Programs. These policy evaluations essentially show the outcomes of the government-driven transformation of traditional pastoralism and differ from other existing research that has been concerned with the internal process of pastoralism transformation (e.g. Mwangi, 2007).

To date, the transformation of pastoralism has widely been employed by governments around the world by developing sedentarisation instead of nomadism, privatisation of grasslands instead of common use, intensification instead of extensive grazing, commercialisation of livestock systems instead of subsistence, determinative stocking rate instead of opportunistic strategies with stochastic stocking rates, to name a few (Homewood, 1995; Li and Zhang, 2009). Although plenty of studies have discussed the pros and cons of these pastoralism transformation programs from the perspective of ecology, sociology, and other fields (e.g. Suttie et al., 2005; Mwangi, 2007; Li and Zhang, 2009; Hua and Squires, 2015), there is a lack of literature based on large-scale areas and long-term observation to provide empirical evidence for scientific debate. This study employs panel data involving extensive pastoral areas and spanning several decades to quantify the ecological and economic outcomes of the pastoralism transformation in Inner Mongolia, China. Specifically, the impact of land tenure reform represents the outcomes of grassland privatisation instead of communal tenure. The effectiveness of Ecological Construction Programs presents the outcomes of current management systems on the grasslands, such as the grazing ban and deterministic stocking rate instead of stochastic stocking rates. Moreover, the preference for grasslands by stakeholders reflects the perception of grassland resource users on the pastoralism transformation. The conceptual framework for our research is based on Ostrom's Social-Ecological Systems (SESs) framework (Ostrom, 2007; 2009). Chapters 2 through 5 address four specific research questions.

Chapter 2 starts with studying the perceptions of users on grassland use under the current policy interventions. The specific research question is, *what are the individual and social optima of*

rural land allocation by stakeholders: among cultivated land, grassland, forest and other rural land? It is answered by quantifying the preference of different stakeholders over rural land allocation, and thereby reflects the different interests of different stakeholders over grassland use. Chapters 3, 4 and 5 focus on estimating the outcomes of policy interventions targeting the grasslands. Chapter 3 addresses the question, *what is the impact of land tenure reform on the grassland condition?* It first outlines the process of land tenure reform in the pastoral areas of China and then investigates the changes in grassland quantity and quality that are caused by the reform. Chapter 4 answers the question, *what is the impact of the land tenure reform on livestock production?* It presents the economic outcome of the land tenure reform by exploring the effects of the reform on the livestock population and meat output. Chapter 5 focuses on the question, *what is the impact of Ecological Construction Programs on grassland conservation?* It estimates the ecological outcome of one typical Ecological Construction Program (the Subsidy and Incentive System for Grassland Conservation).

This final chapter recaps the main findings of each chapter, presents the contribution to the existing scientific debates and draws lessons for policy improvement. It concludes with the research limitation and recommendations for future research.

6.2 Synthesis of research findings

6.2.1 Different preference on grassland use by different stakeholders

The individual optima of rural land allocation reveal the different preferences of different stakeholders for grassland use. The social optima illustrate a possible trade-off among stakeholders over grassland use. The comparisons between individual optima, social optima and actual allocation reflect the conflicts over grassland use among different stakeholders. In our case study, the optima of ecological and economic authorities for desirable grassland areas are within the range of the social optimum, whereas herders prefer more and farmers prefer less grassland than the social optimum. In addition, the actual area of grassland already exceeds the upper range of social optima, implying the further need to reduce the grassland area for the sake of social welfare. This finding appears to contradict the prevailing policy direction of grassland conservation that attempts to expand the grassland area.

6.2.2 Grassland privatisation is a significant reason to cause grassland degradation in a long term

Land tenure reform in the pastoral areas of China engages in privatising use rights of grasslands (Banks et al., 2003). The impact of land tenure reform on the grassland ecosystem reflects the ecological effectiveness of grassland privatisation. The results of this research prove that grassland privatisation has a significant and negative impact on the condition of the grasslands. More specifically, private use causes the reduction of the total grassland area and leads to the degradation of dense and moderate grasslands into sparse grassland in the long run. As such, we observe “a tragedy of privatisation” (Guelke, 2003; Li and Huntsinger, 2011), as opposed to the well-known “tragedy of the commons”.

6.2.3 Grassland privatisation puts a ceiling on the livestock production of pastoral areas, but it promotes the development of livestock productivity

The implementation of land tenure reform is shown to have significant and negative effects on the increase in livestock production. That is, land tenure reform that refers to the privatisation of grassland use rights puts a ceiling on livestock production. This may be a possible reason for why the reform has not been implemented as successfully in the grasslands as in cropland of China. However, land reform in itself is unable to offset the impact of other factors that stimulate the increase in livestock production in practice. Moreover, the constraining effect of land tenure reform on livestock production steadily decreases following the initial implementation of the reform and ultimately disappears. Furthermore, the constraining effect of land tenure reform is stronger on the increase of livestock population than on that of meat output, indicating that land tenure reform improves livestock productivity.

6.2.4 Ecological Construction Programs have significant impacts on preventing grassland degradation

A series of Ecological Construction Programs have been initiated for grassland conservation in the 2000s. This research provides quantitative evidence that the condition of grasslands has improved significantly under the auspices of Ecological Construction Programs, specifically under the program of Subsidy and Incentive System for Grassland Conservation (SISGC). However, the ecological effectiveness of the Ecological Construction Programs is offset to some extent by other socio-economic and climate factors, such as market conditions and climate change. This may explain why the actual condition of the grasslands is not as good as had been



expected. In addition, the case of SISGC shows that the impact of Ecological Construction Programs on grassland conservation is stronger in the counties with worse initial grassland condition, while the effects of market conditions and climate factors were also more pronounced in these counties. Finally, the effectiveness of SISGC is shown to experience an initial increasing but subsequently decreasing trend in the years since SISGC is first implemented.

In summary, the findings of the research presented in chapters 2 to 5 suggest that the transformation of traditional pastoralism through policy intervention impacts the grassland system significantly and complexly. For instance, the outcomes of grassland privatisation (i.e. land tenure reform) have two different sides. On the one side, it causes further grassland degradation, but on the other, it is beneficial to the development of animal husbandry in pastoral areas by controlling excessive growth of the livestock population while prompting livestock productivity. Moreover, Ecological Construction Programs that mainly rely on the control of livestock grazing for grassland conservation has led to an improvement in the grassland conditions, but this effect is offset to some extent by other socio-economic and climate factors.

6.3 Contribution to scientific debates

6.3.1 The tragedy of the commons versus the tragedy of privatisation

The concept of the tragedy of the commons has long been part of the conventional wisdom in ecology, economics and political science (McEvoy 1987). It contends that resource degradation is inevitable unless the commons are either privatised or maintained as common land with clear rights of entry and use (Hardin 1968). Similarly, the theorem of property rights proposes a clear assignment of property rights as a precondition for economically efficient resource allocation and environmental sustainability (Coase 1960). Moreover, privatisation is expected to increase agricultural productivity and the wise use and conservation of resources, because the transferability of rights under private property arrangements is supposed to ensure that resources end up with the most productive users (Mwangi, 2007). As such, some scholars and policy makers have repeatedly identified traditional pastoralism as an irrational, ecologically destructive and economically inefficient system (Homewood, 1995; Kamara et al., 2005; Ho and Azadi, 2010).

However, Hardin's theory of the tragedy of the commons and Coase's theorem of property rights have been rejected by pastoral specialists, who found that these theoretical findings provided a poor guide toward the sustainable use and management of grasslands (Feeny et al.

1990; Behnke 1994; Sneath 1998). Some new arguments have arisen in support of traditional pastoralism, emphasising mobility, flexibility and reciprocity of grassland use, which conforms to the attribute of grassland resources with strong temporal variability and spatial heterogeneity (Fernandez-Gimenez and Le Febre, 2006). It is argued that traditional pastoralism could maintain ecosystem stability and biological diversity in terms of the sustainable use of grasslands (Fernandez-Gimenez and Le Febre, 2006; Fan et al., 2015). In particular, the communal tenure of traditional pastoralism buffers change and uncertainty in grasslands, allowing pastoralists to maintain a relatively reliable flow of resources from a relatively unreliable environment (Roe et al. 1998). Indeed, traditional pastoralism has developed flexible institutions that enable the persistence of production on such patchy and unpredictable low-productivity environments for centuries (Fernandez-Gimenez and Le Febre, 2006; Li and Huntsinger, 2011).

Recently, the literature has increasing interests in the common property regime of grasslands, especially considering the benefits of mobility, flexibility and reciprocity of grassland use, such as mutual aid, pooled labour as well as access to water resources, medicinal herbs and natural shelters, which potentially maintain the ecological and economic resilience of the grassland systems (Cao et al., 2013). In contrast, the exclusionary land tenure and pasture enclosure are suggested to be counterproductive to improve grassland systems because they threaten the integrity of ecological systems of arid and semi-arid areas, particularly impeding mobility and resource access in times of disaster (Mwangi, 2007; Galvin 2009). In this regard, the tragedy of privatisation is put forward (Guelke 2003; Li and Huntsinger 2011), in contrast to the well-known tragedy of the commons. In addition, grassland privatisation reduces the amount of land available for livestock grazing, further leading to a reduction in the number of livestock that an individual can potentially own, ultimately resulting in poverty Mwangi (2007). Some scholars suggested that the highly spatial heterogeneity of grassland resources leads to dramatically inequitable assignments, especially the forage quality and water access, which may cause severe conflicts over grassland use (Yan et al., 2005). The other negative social consequences of the implementation of grassland privatisation include increased labour burdens for each household and the loss of labour economies of scale (Yan et al., 2005; Cao et al., 2013).

Given these arguments over traditional pastoralism, especially regarding property rights and use rights, Chapters 3 and 4 of this research provide the empirical analysis on the ecological and economic impacts of the transformation of grassland use rights in China. Chapter 3 shows

that grassland privatisation causes grassland degradation in pastoral areas. Chapter 4 finds that grassland privatisation places a ceiling on the livestock production of pastoral areas, while it stimulates the development of livestock productivity. Hence, this research supports the idea of a tragedy of privatisation, drawing from the negative ecological outcomes of privatisation. Nevertheless, grassland privatisation is beneficial to the development of local animal husbandry considering the effects of controlling excessive growth of the livestock population and prompting livestock productivity, showing that grassland privatisation can stimulate herders to increase efforts to improve the output per animal, instead of increasing the number of animals.

6.3.2 Deterministic stocking rates versus stochastic stocking rates

The control of the livestock population has been the most widespread approach to halting grassland degradation in China, Africa and other parts of the world since overgrazing became a concern in the 2000s (Vetter, 2005). However, a debate around the validity of the control of the livestock population for preventing grassland degradation in the arid and semi-arid areas has emerged, often referred to as a debate between the equilibrium model and the non-equilibrium model (e.g. Clements, 1916; Ho, 2001; Vetter, 2005). The equilibrium model, or Clementsian succession theory (Clements, 1916), advocates the stability and predictability of grasslands, which stresses the importance of biotic feedbacks such as density-dependent regulation of livestock population and the feedback of livestock density on vegetation composition, cover and productivity (Ho, 2001; Vetter, 2005). In other words, the equilibrium model suggests that the stocking rate is predictable and deterministic, and grassland degradation occurs if the carrying capacity²⁵ is exceeded due to overstocking. The use and management of grasslands based on the equilibrium model therefore focus on deterministic and low stocking rates, i.e. controlling the livestock population for grassland conservation.

In contrast, the non-equilibrium model advocates that the grassland condition in the arid and semi-arid areas are primarily driven by stochastic abiotic factors which result in highly variable and unpredictable primary production, particularly due to the factor of precipitation (Ho, 2001). The argument carried further implies that grassland degradation is part of a natural process of vegetation decline and growth in response to precipitation (Ho, 2001). The grassland use and management based on the non-equilibrium model claims that the livestock population has negligible feedback on the vegetation as the number of animals is kept below equilibrium densities by frequent droughts, and that grassland degradation as a result of overgrazing is thus

²⁵ Carrying capacity indicates that grazing pressure is counterbalanced by the natural regeneration of plants.

unlikely (Sullivan and Rohde, 2002). As such, the non-equilibrium model embraces opportunistic strategies with high and stochastic stocking rates in which the number of livestock is maximised according to the availability of forage, and grassland degradation is not caused by overgrazing.

The debates over grassland management between the equilibrium model and the non-equilibrium model eventually are reflected in the question of whether the livestock population can be allowed to increase without the threaten of grassland degradation (Vetter, 2005). The Chinese policy interventions provide an outstanding example of managing grasslands based on the equilibrium model. It is believed that overgrazing is the main cause of grassland degradation, therefore the control of the livestock population has been employed for grassland conservation. Chapter 5 of this research investigates the ecological outcomes of the policy intervention of deterministic stocking rates and grazing bans in Inner Mongolia, China. It suggests that control of the livestock population improves the grassland condition significantly in the arid and semi-arid areas of China in the short term. However, there are some doubts over the long term, as the effectiveness of grassland conservation measures appears to decrease in later years. As such, this research agrees that grassland management based on the equilibrium model might be effective for grassland conservation in the arid and semi-arid pastoral areas temporarily, but that the non-equilibrium model may be more valid for long-term strategies.

6.3.3 Top-down intervention versus bottom-up intervention

The debate over concrete uses and management strategies for natural resources is still ongoing: in particular, whether a top-down or a bottom-up approach can best achieve conservation goals (Oestreicher et al., 2009; Mauerhofer, 2011). The top-down approach assumes that legislation and policies set explicit aims and objectives, providing a blueprint that is then directly translated into action on the ground (Urwin and Jordan, 2008). Top-down intervention generally follows a decision by government authorities and policy makers. Although such official acts in the public interest are considered capable of designing optimal rules to govern resource use for an entire region, this prescription is not supported by empirical research (Ostrom, 1999). The top-down approach has been criticised as leading to poor compliance with statutory law and limited local community participation in monitoring and enforcing natural resource conservation regulations (Hartter and Ryan, 2010). Hence, top-down resource management strategies have been linked to marginalising poor populations and exacerbating natural resource depletion (Oestreicher et al., 2009).

A bottom-up approach that aims to guide self-organisation towards desirable states (Ball and Helbing, 2012), recognises the decisions and strategies of local resource users in managing resources (Castella et al., 2007). It has been suggested that sustainable natural resource management directives must be initiated from the grassroots level, where the problems and solutions are perceived as local (Hartter and Ryan, 2010). Research in multiple disciplines has found that some top-down policy interventions accelerate resource destruction, whereas self-organisation by some resource users has been successful in resource conservation (Ostrom, 2009). Field studies also have found that local resource users, with or without the assistance of external authorities, have created a wide diversity of institutional arrangements for coping with common-pool resources²⁶ (Ostrom, 1999). Moreover, a series of relatively autonomous, self-organised resource governance systems might do a better job of regulating small common-pool resources than a single central authority. A decentralised, polycentric, bottom-up approach, involving multiple institutions instead of a single global one, may provide better conditions both for cooperation to thrive and for ensuring the maintenance of such institutions (Vasconcelos et al., 2013). Local resource users, who share moral and ethical standards for how to behave in the groups they form, face lower transaction costs in reaching agreements and lower costs of monitoring (Ostrom, 2009). In terms of grassland management, Sneath (1998) showed that the grasslands under central government management in China and Russia are associated with more degradation than those under traditional, self-organised group-property regimes in Mongolia.

Scholars have increasingly embraced the bottom-up, self-organised approach to policy intervention. The research cases by Schoon (2008) found that the bottom-up approach increases cooperation and coordination at the operational level, but that top-down approach can generate successful cooperation at higher levels of governance. Policy interventions in grassland conservation in China have generally been conducted via a top-down approach, starting with explicit aims and quotas that are distributed from the central government to the provinces, followed by subsequent distribution down through counties, townships, and finally to participating villages (Bennett, 2008). This is criticised because of the lack of sufficient consultation with local communities and rural households to identify their needs and learn the local constraints (Bennett, 2008). Chapter 5 focuses on investigating the ecological outcome of top-down interventions into grassland conservation in China. The research results reveal that

²⁶ A common-pool resource, such as an irrigation system, a fishing ground, a forest, or open access grasslands, is a natural or man-made resource, from which it is difficult to exclude or limit users once the resource is provided, where one person's consumption of resource units makes those units unavailable to others (Ostrom et al., 1994).

the policy intervention impacts grassland condition significantly and positively, but the effectiveness on grassland conservation is not as good as expected because of the counterbalancing effect of other factors. In this regard, the policy improvement of grassland conservation in China could consider more bottom-up self-organisation interventions that involve learning about local market forces, climate conditions, and so on.

6.3.4 The one-size-fits-all versus tailor-made policies for resource governance

Scholars have tended to develop simple theoretical models to analyse aspects of resource problems and to prescribe universal solutions for resource use (Ostrom, 2009). Governmental agencies often impose one-size-fits-all policy for resource governance. However, the universal policy solution frequently fails (Pritchett and Woolcock, 2004; Ostrom, 2009). In particular, national government agencies are often unsuccessful in their efforts to design effective and uniform sets of rules to regulate important common-pool resources across a broad domain (Ostrom, 1999).

In the case of China, land tenure reform started with the Household Production Responsibility System in cropland areas in the late 1970s and spread to the pastoral areas by the early 1980s. The reform intended to reallocate communal land to peasant households. In the cropland areas, the household-based use rights system was established rapidly, and 94.2% of rural households had adopted the system by the end of 1983 (Lin, 1987; Banks et al., 2003). Chinese scholars showed that this reform of cropland use rights created millions of smallholders with relative autonomy over land use decisions and crop selection (Tilt, 2008). Moreover, the household-based farming system improved the incentive structure for agricultural production when monitoring was difficult within the commune system (Lin, 1987). This assignment of cropland resulted in the growth of agricultural output and income in the reform period.

However, in the pastoral areas, the implementation of the land tenure reform is not as successful as was the case in the cropland areas. The assignment of grassland use rights was implemented slowly and was still in progress after 30 years of efforts, and a second round of reform to accelerate the implementation in the 1990s. The adoption of actual private use by individual households evidently lagged behind the assignment of formal use rights by governments. The research results in Chapter 3 showed that the land tenure reform was detrimental to the grassland ecosystem, although it is beneficial to the development of the animal husbandry system of pastoral areas according to the research results in Chapter 4. The conclusion can be



made that the land tenure reform that seemed suitable for managing cropland resources may not fit the management of grassland resources. This result is in line with the research finding that individual property may not be suitable for most pastoral settings in which resource productivity is variable due to environmental variability (Mwangi, 2007).

With respect to resource conservation, the Chinese government initiated a series of ecological programs for ecological restoration and resource development in its forest sector in the late 1990s, including the Natural Forest Protection Program, the Sloping Land Conversion Program, the Desertification Combating Program around Beijing and Tianjin, the Shelterbelt Development Program, and the Wildlife Conservation and Nature Reserve Development Program (Liu et al., 2010). Plenty of scholars have shown that these ecological programs for forest conservation have significant and positive impacts on improving resource conditions as well as increasing rural households' income (e.g. Liu et al., 2010; Mullan et al., 2010). Following these ecological programs for forest conservation, a series of programs have been initiated to protect the grasslands of China, especially during the 2000s. In general, sowing grass and reducing grazing (such as with a grazing ban) are imposed as key components for grassland conservation, which coincide with the main measures of planting trees and restricting logging (such as with a logging ban) for forest conservation (Mullan et al., 2010). Chapter 5 of this research shows that the effectiveness of the large-scale ecological programs differs across different regions with the same policy context. In addition, Chapter 2 of this research shows that different stakeholders have divergent preferences in different types of rural land. Given these research results, it is evident that there is no one-size-fits-all policy for the governance of cropland, grassland and forest land, as well as for the interests of different stakeholders. In this case, tailor-made policies that take into account the specific circumstances, such as the resource types and stakeholders, are needed for resource governance.

6.4 Policy implications

With severe grassland degradation, high population pressure and increased production requirements, the sustainable use and management of grasslands, which enables the maintenance of productive capacity without compromising the ecological interests of human communities and the ecosystem itself, are essential. An increasing number of scholars support the model of traditional pastoralism as one approach for sustainable grassland use, due to its advantages of mobility, flexibility and reciprocity that fit grassland use for heterogeneously distributed resources. In this respect, some have suggested abolishing the existing grassland

privatisation, tearing down wire fences between the patches, repealing the deterministic stocking rates and further resuming nomadism (Wu and Du, 2008). However, there has also been criticism of traditional pastoralism as being incapable of coping with the rapidly increasing population pressure and economic demands relying on finite grassland resources, because traditional pastoralism is primarily subsistence-oriented. In other words, traditional pastoralism may only operate well during the phases of low economic demand and minimal environmental degradation. As such, it is important to transform the traditional pastoralism into a model that can adapt to contemporary socio-economic conditions.

The case of Inner Mongolia presents lessons for the improvement of the government-driven transformation of traditional pastoralism in arid and semi-arid pastoral areas. In particular, it shows that the assignment of grassland use rights to individual households has negative impacts on improving the grassland condition. On the other hand, the privatised use rights have put a ceiling on the livestock production of pastoral areas, while it promotes the development of livestock productivity. Some scholars have mentioned that the household is not an efficient unit for meeting the demands of large-scale animal husbandry, and household-based privatisation has provided a rapid increase in livestock production costs in the pastoral areas of China (Li et al., 2014). Admittedly, the cooperative use of grasslands on a large scale offers the opportunity to benefit from the maximum heterogeneity in the landscape. Given the status quo of privatisation implemented in pastoral societies, a possible solution is to stimulate the local community toward cooperative use and management of grassland resources based on the existing privatised property rights or use rights of grasslands. In this regard, the privatisation of grassland property rights or use rights could be viewed as a precondition for guaranteeing stable cooperation. The cooperative households who reach an agreement are encouraged to remove their fences and pool their grassland resource in order to provide a better match for forage and water availability. Besides the grassland resource, other agricultural inputs like labour and capital could be pooled for efficient distribution as well. Economies of scale in production and the exclusion of outsiders are also important (Mwangi, 2007). The proposal of cooperative use is in line with the research findings of Mwangi (2007) that showed how parcel re-aggregation can offer promise for ecosystem sustainability as well as for the economic sustainability of African grasslands.

In addition, a wide diversity of institutional arrangements for grassland use are needed, considering the heterogeneity in grassland resources per se and the different socio-economic

and cultural backgrounds of pastoral societies, instead of extending the same policy measures designed for crop-farming areas to pastoral areas. This study has found that the same land tenure reform had different policy effectiveness on grassland and cropland. Especially some unintended effects were observed on grassland systems, which suggest the need to clarify the different attributes between grassland systems and cropland systems before transforming the property rights or use rights of grasslands. Clearly, the top-down decision by central governments for resource management makes it difficult to consider all of the specific attributes of each resource at local levels, but it is advocated that resource users who understand the local Social-Ecological Systems (SESs) well are able to use the resource more efficiently through self-organisation and self-crafted institutions (e.g. Ostrom, 1999; Baerlein et al., 2015). Users that have full autonomy at the collective-choice level to craft and enforce some of their own rules, also confront lower transaction costs (Ostrom, 2009). As such, it is recommended to create specific institutional agreements for grassland use by local communities. This bottom-up self-organisation management would have the advantage of relatively complete information for developing efficient use of grasslands, which specifically takes into account the highly variable topography of grasslands, strongly spatial and temporal heterogeneity of grassland resources, traditional knowledge about grassland use and production, and other important factors (Yan et al., 2005; Wu and Du, 2008; Fernandez-Gimenez et al., 2012).

Theoretically, cooperative use of grasslands among neighbouring households could conform to the theory of “common pool resources” which involves self-organisation by the local community in efforts to achieve a sustainable Social-Ecological System (Ostrom, 2009). Nevertheless, cooperative use of grasslands and bottom-up self-organisation management should undergo further research concerning internal organisation, transaction costs, etc. Moreover, although the self-organisation by local herders is predominantly suggested as a localised management model, the assistance of external authorities plays an important role in balancing the various needs of the whole society. The long-term sustainability of rules devised at a SES level depends on monitoring and enforcement, without being overruled by government policies (Ostrom, 2009). As such, a combination of top-down and bottom-up approaches is perhaps the most effective approach to be considered. The study by Fernandez-Gimenez et al. (2012) also demonstrated that the strengthening of adaptive capacity and resilience of grassland systems requires joint efforts and improved communication between the government and local communities.

6.5 Limitations and recommendations for future research

To conclude, I would like to point out a number of limitations of this study and challenges for future research. First, I follow Ostrom (2007)'s recommendation that the problems linked to SESs require serious study of complex, multivariable, nonlinear, cross-scale and changing systems, and that more knowledge about specific variables of SESs, the interactions between variables and dynamics of SESs need to be explored more deeply. More specifically, in Chapter 2, we assume four categories of rural land, four stakeholder types and three scenarios to simplify the reality of land allocation in the SES. For further research, the more components and the more complicated interactions between the components of rural land allocation linked SESs need to be considered. Moreover, chapters 3, 4 and 5 present general outcomes of the policy interventions in grassland systems, whereas the interaction between grassland condition and livestock production is a significant aspect to be studied more concretely.

With regard to the complex SESs of grasslands, our research provides preliminary outcomes of governance system to grassland system at county-level. An interesting avenue for future research is to understand the specific impacts of policy interventions at the household-level, such as the costs and benefits that herders faced during the policy interventions. Further research would be to explore this in more detail, for example: what changes herders adopted in order to adapt to the government-driven transformation of traditional pastoralism; how herders adjusted their grassland use and livestock production patterns coping with the increasing population, the growing demand for livestock products and the grassland degradation; what is the resilience of pastoral societies in the changing natural and social environment. Moreover, our research focuses on the pastoral areas of Inner Mongolia, which is a typical region for studying the SESs of grasslands of China. However, the crop-farming areas that also include some permanent grassland have more complicated SESs and more serious grassland degradation than the pastoral areas, and thus even more problems to be addressed. The same research questions that are addressed in this study for the pastoral areas would be also interesting for those crop farming areas, such as the dynamics of their grassland ecosystem, livestock production and household livelihoods under the changing social, economic and political settings. Moreover, the accuracy of Chinese livestock statistics is questionable (Waldron et al., 2007). The standardisation of data collection on livestock population and meat output must be improved, especially regarding the production of rural households. Given more reliable data on livestock production, the scepticism over the existence and extent of

overgrazing would be dissected, which would significantly contribute to the debates between the equilibrium model and non-equilibrium model for arid and semi-arid areas.

Last but not least, cooperative use of grasslands on a large scale and bottom-up self-organisation management are considered possible solutions for policy improvement in this research, but more empirical analysis is required to testify to their validity and investigate the possibility of implementation. There are no simple solutions to complex governance problems (Ostrom, 2007). As such, we call for more research to learn about the SESs of grasslands based on the same framework introduced by Ostrom, in order to develop broader and more integrated insights into SESs. A large number of government-driven transformations of traditional pastoralism are under way in Africa, Asia and other regions. The quest to better understand the approaches and paths of these transformations is an important part of ensuring the sustainability of grassland SESs around the world.

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Appendixes

Appendix 2-A: Derivation of the individually optimal allocation of land

For the individual optimal allocation of land, each stakeholder type i ($i = 1, 2, 3$ and 4 representing four stakeholder types, including the herders, the farmers, the ecological public authorities and economic public authorities) maximizes its utility function by choosing the amounts of land for different purposes. That is:

$$MAX U_i = l_{cul(i)}^{a_i} l_{ran(i)}^{b_i} l_{for(i)}^{c_i} l_{oth(i)}^{d_i}, \quad (2A.1)$$

Subject to:

$$l_{cul(i)} + l_{ran(i)} + l_{for(i)} + l_{oth(i)} = L, \quad (2A.2)$$

where $l_{cul(i)}, l_{ran(i)}, l_{for(i)}, l_{oth(i)} \geq 0$; $0 \leq a_i, b_i, c_i, d_i \leq 1$; $a_i + b_i + c_i + d_i = 1$

The logarithmic transformation of the utility function gives:

$$u_i = \ln U_i = a_i \ln l_{cul(i)} + b_i \ln l_{ran(i)} + c_i \ln l_{for(i)} + d_i \ln l_{oth(i)}. \quad (2A.3)$$

The Lagrange function is then:

$$Lu_i = a_i \ln l_{cul(i)} + b_i \ln l_{ran(i)} + c_i \ln l_{for(i)} + d_i \ln l_{oth(i)} + \lambda (L - l_{cro(i)} - l_{ran(i)} - l_{for(i)} - l_{oth(i)}) \quad (2A.4)$$

Taking the partial derivations of Lu_i with respect to each variable, and setting the first order condition gives:

$$\frac{\partial Lu_i}{\partial l_{cul(i)}} = \frac{a_i}{l_{ar(i)}} - \lambda = 0, \quad (2A.5)$$

$$\frac{\partial Lu_i}{\partial l_{ran(i)}} = \frac{b_i}{l_{ran(i)}} - \lambda = 0, \quad (2A.6)$$

$$\frac{\partial Lu_i}{\partial l_{for(i)}} = \frac{c_i}{l_{for(i)}} - \lambda = 0, \quad (2A.7)$$

$$\frac{\partial Lu_i}{\partial l_{oth(i)}} = \frac{d_i}{l_{oth(i)}} - \lambda = 0. \quad (2A.8)$$

Combining (2A.5) to (2A.8) gives:

$$\frac{a_i}{l_{cul(i)}} = \frac{b_i}{l_{ran(i)}} = \frac{c_i}{l_{for(i)}} = \frac{d_i}{l_{oth(i)}}, \quad (2A.9)$$

Or :

$$a_i l_{ran(i)} = b_i l_{cul(i)}, a_i l_{for(i)} = c_i l_{cul(i)}, a_i l_{oth(i)} = d_i l_{cul(i)}. \quad (2A.10)$$

Combining (A2) with (A9), we have:

$$\frac{a_i}{l_{cul(i)}} = \frac{a_i}{L - l_{ran(i)} - l_{for(i)} - l_{oth(i)}}, \quad (2A.11)$$

Rearrange (A10) and (A11) combined with $a_i + b_i + c_i + d_i = 1$, we obtain:

$$l_{cul(i)}^0 = a_i \times L, \quad (2A.12)$$

$$l_{ran(i)}^0 = b_i \times L, \quad (2A.13)$$

$$l_{for(i)}^0 = c_i \times L, \quad (2A.14)$$

$$l_{oth(i)}^0 = d_i \times L, \quad (2A.15)$$

This completes the proof of equation (3) to (6) in the main text of Chapter 2.

Appendix 2-B: Derivation of the socially optimal allocation of land

For the social optimal allocation of land, maximizing the social welfare is:

$$MAX W = \prod_{i=1}^4 U_i^{\beta_i}, \quad (2B.1)$$

Subject to:

$$l_{cul(s)} + l_{ran(s)} + l_{for(s)} + l_{oth(s)} = L, \quad (2B.2)$$

where $\sum_{i=1}^4 \beta_i = 1, \beta_i \geq 0, 0 \leq a_i, b_i, c_i, d_i \leq 1; l_{culs}, l_{rans}, l_{fors}, l_{oths} \geq 0; \sum_{i=1}^4 \beta_i = 1$ and $\beta_i \geq 0$

Plugging equation (2A.1) into (2B.1), we obtain:

$$\begin{aligned} W = & \left(l_{cul(s)}^{a_1} l_{ran(s)}^{b_1} l_{for(s)}^{c_1} l_{oth(s)}^{d_1} \right)^{\beta_1} * \left(l_{cul(s)}^{a_2} l_{ran(s)}^{b_2} l_{for(s)}^{c_2} l_{oth(s)}^{d_2} \right)^{\beta_2} \\ & * \left(l_{cul(s)}^{a_3} l_{ran(s)}^{b_3} l_{for(s)}^{c_3} l_{oth(s)}^{d_3} \right)^{\beta_3} \\ & * \left(l_{cul(s)}^{a_4} l_{ran(s)}^{b_4} l_{for(s)}^{c_4} l_{oth(s)}^{d_4} \right)^{\beta_4} \end{aligned} \quad (2B.3)$$

The logarithmic transformation of the social welfare function is:

$$\begin{aligned} w = \ln W = & \beta_1 \ln \left(l_{cul(s)}^{a_1} l_{ran(s)}^{b_1} l_{for(s)}^{c_1} l_{oth(s)}^{d_1} \right) + \beta_2 \ln \left(l_{cul(s)}^{a_2} l_{ran(s)}^{b_2} l_{for(s)}^{c_2} \right) \\ & + \beta_3 \ln \left(l_{cul(s)}^{a_3} l_{ran(s)}^{b_3} l_{for(s)}^{c_3} l_{oth(s)}^{d_3} \right) \\ & + \beta_4 \ln \left(l_{cul(s)}^{a_4} l_{ran(s)}^{b_4} l_{for(s)}^{c_4} l_{oth(s)}^{d_4} \right) \end{aligned} \quad (2B.4)$$

The Lagrange function is then:

$$\begin{aligned} Lw = & \beta_1 \ln \left(l_{cul(s)}^{a_1} l_{ran(s)}^{b_1} l_{for(s)}^{c_1} l_{oth(s)}^{d_1} \right) + \beta_2 \ln \left(l_{cul(s)}^{a_2} l_{ran(s)}^{b_2} l_{for(s)}^{c_2} l_{oth(s)}^{d_2} \right) \\ & + \beta_3 \ln \left(l_{cul(s)}^{a_3} l_{ran(s)}^{b_3} l_{for(s)}^{c_3} l_{oth(s)}^{d_3} \right) + \beta_4 \ln \left(l_{cul(s)}^{a_4} l_{ran(s)}^{b_4} l_{for(s)}^{c_4} l_{oth(s)}^{d_4} \right) \\ & + \lambda (L - l_{cul(s)} - l_{ran(s)} - l_{for(s)} - l_{oth(s)}) \end{aligned} \quad (2B.5)$$

Taking the partial derivations of Lw with respect to each variable, and setting the first order condition gives:

$$\frac{\partial LW}{\partial l_{cul(s)}} = \frac{\sum_{i=1}^4 \beta_i a_i}{l_{cul(s)}} - \lambda = 0, \quad (2B.6)$$

$$\frac{\partial LW}{\partial l_{ran(s)}} = \frac{\sum_{i=1}^4 \beta_i b_i}{l_{ran(s)}} - \lambda = 0, \quad (2B.7)$$

$$\frac{\partial LW}{\partial l_{for(s)}} = \frac{\sum_{i=1}^4 \beta_i c_i}{l_{for(s)}} - \lambda = 0, \quad (2B.8)$$

$$\frac{\partial LW}{\partial l_{oth(s)}} = \frac{\sum_{i=1}^4 \beta_i d_i}{l_{oth(s)}} - \lambda = 0, \quad (2B.9)$$

Combining (B6) to (B9) gives:

$$\frac{\sum_{i=1}^4 \beta_i a_i}{l_{cul(s)}} = \frac{\sum_{i=1}^4 \beta_i b_i}{l_{ran(s)}} = \frac{\sum_{i=1}^4 \beta_i c_i}{l_{for(s)}} = \frac{\sum_{i=1}^4 \beta_i d_i}{l_{oth(s)}}, \quad (2B.10)$$

Or:

$$\sum_{i=1}^4 \beta_i a_i l_{ran(s)} = \sum_{i=1}^4 \beta_i b_i l_{for(s)}, \quad (2B.11)$$

$$\sum_{i=1}^4 \beta_i a_i l_{for(s)} = \sum_{i=1}^4 \beta_i c_i l_{for(s)}, \quad (2B.12)$$

$$\sum_{i=1}^4 \beta_i a_i l_{oth(s)} = \sum_{i=1}^4 \beta_i d_i l_{for(s)}. \quad (2B.13)$$

Combining (B2) with (B10), we have:

$$\frac{\sum_{i=1}^4 \beta_i a_i}{l_{cul(s)}} = \frac{\sum_{i=1}^4 \beta_i a_i}{L - l_{ran(s)} - l_{for(s)} - l_{oth(s)}}, \quad (2B.14)$$

Plug (B11)(B12) and (B13) into (B14), we obtain:

$$l_{cul(s)}^* = \frac{\sum_{i=1}^4 \beta_i a_i}{\sum_{i=1}^4 \beta_i a_i + \sum_{i=1}^4 \beta_i b_i + \sum_{i=1}^4 \beta_i c_i + \sum_{i=1}^4 \beta_i d_i} L, \quad (2B.15)$$

$$l_{ran(s)}^* = \frac{\sum_{i=1}^4 \beta_i b_i}{\sum_{i=1}^4 \beta_i a_i + \sum_{i=1}^4 \beta_i b_i + \sum_{i=1}^4 \beta_i c_i + \sum_{i=1}^4 \beta_i d_i} L, \quad (2B.16)$$

$$l_{for(s)}^* = \frac{\sum_{i=1}^4 \beta_i c_i}{\sum_{i=1}^4 \beta_i a_i + \sum_{i=1}^4 \beta_i b_i + \sum_{i=1}^4 \beta_i c_i + \sum_{i=1}^4 \beta_i d_i} L, \quad (2B.17)$$

$$l_{oth(s)}^* = \frac{\sum_{i=1}^4 \beta_i d_i}{\sum_{i=1}^4 \beta_i a_i + \sum_{i=1}^4 \beta_i b_i + \sum_{i=1}^4 \beta_i c_i + \sum_{i=1}^4 \beta_i d_i} L, \quad (2B.18)$$

Since $\sum_{i=1}^4 \beta_i a_i + \sum_{i=1}^4 \beta_i b_i + \sum_{i=1}^4 \beta_i c_i + \sum_{i=1}^4 \beta_i d_i = 1$

We have:

$$l_{cul(s)}^* = L \times \sum_{i=1}^4 \beta_i a_i \quad (2B.19)$$

$$l_{ran(s)}^* = L \times \sum_{i=1}^4 \beta_i b_i \quad (2B.20)$$

$$l_{for(s)}^* = L \times \sum_{i=1}^4 \beta_i c_i \quad (2B.21)$$

$$l_{oth(s)}^* = L \times \sum_{i=1}^4 \beta_i d_i \quad (2B.22)$$

This completes the proof of equation (9) to (12) in the main text of Chapter 2.

Appendix 2-C: Questionnaires and results of Analytic Hierarchy Process

Appendix 2-C1: Questionnaires of Analytic Hierarchy Process

1. Which is your stakeholder type in rural land allocation?
 - a. Herder
 - b. Farmer
 - c. Economic authorities
 - d. Ecological authorities

2. Comparing the economic benefit and ecological benefit of rural land, which one concerning the efficient allocation of rural land is more important for you, and how much important?

Economic benefit										Ecological benefit						
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9

3. Comparing the cultivated land and grassland, which one concerning the economic benefit is more beneficial to you, and how many benefits?

Cultivated land										Grassland						
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9

4. Comparing the cultivated land and forest land, which one concerning the economic benefit is more beneficial to you, and how many benefits?

Cultivated land										Forest land						
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9

5. Comparing the cultivated land and other rural land, which one concerning the economic benefit is more beneficial to you, and how many benefits?

Cultivated land										Other rural land						
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9

6. Comparing the grassland and forest land, which one concerning the economic benefit is more beneficial to you, and how many benefits?

Grassland										Forest land						
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9

7. Comparing the grassland and other rural land, which one concerning the economic benefit is more beneficial to you, and how many benefits?

Grassland										Other rural land						
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9

8. Comparing the forest land and other rural land, which one concerning the economic benefit is more beneficial to you, and how many benefits?

Forest land									Other rural land								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

9. Comparing the cultivated land and grassland, which one concerning the ecological benefit is more beneficial to you, and how many benefits?

Cultivated land									Grassland								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

10. Comparing the cultivated land and forest land, which one concerning the ecological benefit is more beneficial to you, and how many benefits?

Cultivated land									Forest land								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

11. Comparing the cultivated land and other rural land, which one concerning the ecological benefit is more beneficial to you, and how many benefits?

Cultivated land									Other rural land								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

12. Comparing the grassland and forest land, which one concerning the ecological benefit is more beneficial to you, and how many benefits?

Grassland									Forest land								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

13. Comparing the grassland and other rural land, which one concerning the ecological benefit is more beneficial to you, and how many benefits?

Grassland									Other rural land								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

14. Comparing the forest land and other rural land, which one concerning the ecological benefit is more beneficial to you, and how many benefits?

Forest land									Other rural land								
9	8	7	6	5	4	3	2	1	2	3	4	5	6	7	8	9	

Note: other rural land indicates land used for raising animals, agricultural facilities, agricultural roads, pit-ponds, fishponds, irrigation, drying grains and forming ridges among croplands.

For above scoring in paired comparisons, the following Fundamental Scale is used to make judgments: 1= Equal; 2=Between Equal and Moderate; 3=Moderate; 4=Between Moderate and Strong; 5=Strong; 6=Between Strong and Very Strong; 7=Very Strong; 8=Between Very Strong and Extreme; 9=Extreme

Appendix 2-C2: Results of Analytic Hierarchy Process

Based on the data from above questionnaires, we employ Super Decisions software to calculate every stakeholder's preference on four types of rural land. And then we take the average value of all representatives of each stakeholder type to represent each stakeholder type's decision in allocation of rural land.

Table 2-C1 Values of a_1, b_1, c_1 and d_1 (herders)

Weights for interviewees in the herder type				
Alternatives	Cultivated land (a_1)	Grassland (b_1)	Forest land (c_1)	Other rural land (d_1)
1	0.1377	0.6311	0.1032	0.1279
2	0.1207	0.6529	0.1120	0.1145
3	0.1232	0.5577	0.1877	0.1314
4	0.2252	0.5715	0.1381	0.0651
5	0.1730	0.5388	0.1653	0.1230
6	0.2014	0.5049	0.2174	0.0763
7	0.2781	0.4993	0.1336	0.0890
8	0.1836	0.3494	0.2909	0.1760
9	0.2409	0.5294	0.1372	0.0925
10	0.2464	0.4913	0.1615	0.1007
11	0.1907	0.5105	0.1716	0.1271
12	0.2679	0.5079	0.1565	0.0676
13	0.1254	0.6754	0.1069	0.0923
14	0.3548	0.4256	0.1487	0.0708
15	0.2573	0.4838	0.1779	0.0811
Average	0.2084	0.5286	0.1606	0.1024
Variance	0.0046	0.0070	0.0023	0.0009

Table 2-C2 Values of a_2, b_2, c_2 and d_2 (farmers)

Weights for interviewees in the farmer type				
Alternatives	Cultivated land (a_2)	Grassland (b_2)	Forest land (c_2)	Other rural land (d_2)
1	0.4873	0.2432	0.2003	0.0692
2	0.3031	0.4334	0.1906	0.0729
3	0.4909	0.1336	0.2824	0.0932
4	0.4564	0.2182	0.2531	0.0723
5	0.5433	0.1180	0.2527	0.0859
6	0.4681	0.2129	0.2324	0.0866
7	0.3800	0.2071	0.2929	0.1200
8	0.4695	0.2701	0.1813	0.0790
9	0.4768	0.3130	0.1466	0.0636
10	0.5239	0.1900	0.1831	0.1030
11	0.3352	0.4396	0.1739	0.0513
12	0.4853	0.1285	0.2944	0.0918
13	0.3750	0.3639	0.1942	0.0669
14	0.3661	0.3595	0.2215	0.0529
15	0.3146	0.4114	0.2041	0.0699
Average	0.4317	0.2695	0.2202	0.0786
Variance	0.0061	0.0123	0.0021	0.0003

Table 2-C3 Values of a_3, b_3, c_3 and d_3 (ecological authorities)

Weights for interviewees in the ecological authorities type				
Alternatives	Cultivated land (a_3)	Grassland (b_3)	Forest land (c_3)	Other rural land (d_3)
1	0.1867	0.3099	0.3808	0.1227
2	0.1963	0.3318	0.3740	0.0979
3	0.1685	0.3894	0.3647	0.0774
4	0.2507	0.2620	0.4008	0.0865
5	0.1985	0.3178	0.3836	0.1000
6	0.2718	0.2930	0.3438	0.0915
Average	0.2121	0.3173	0.3746	0.0960
Variance	0.0016	0.0018	0.0004	0.0002

Table 2-C4 Values of a_4 , b_4 , c_4 and d_4 (economic authorities)

Weights for interviewees in the ecological authorities type				
Alternatives	Cultivated land (a_4)	Grassland (b_4)	Forest land (c_4)	Other rural land (d_4)
1	0.5768	0.1787	0.1082	0.1364
2	0.4479	0.3403	0.0848	0.1271
3	0.4815	0.2614	0.1194	0.1377
4	0.3849	0.2982	0.1183	0.1986
5	0.3246	0.4247	0.1374	0.1133
6	0.3669	0.3768	0.1242	0.1321
Average	0.4304	0.3133	0.1154	0.1409
Variance	0.0083	0.0076	0.0003	0.0009

Appendix 2-D: Estimation of parameter β_i

Table 2-D1 Data for welfare weights β_i

Item	Unit	Value
Population of farmers	Person	168514
Farmer's net income in 2012	RMB per capita	6730
Farmers' total income	Million RMB	1134.10
Population of herders	Person	2986
Herder's net income in 2012	RMB per capita	7898
Herders' total income	Million RMB	23.58
Ecological expenditure in 2012	Million RMB	118.96
Rural ecological expenditure share	%	70
Ecological expenditure on rural area	Million RMB	83.3
Economic expenditure in 2012	Million RMB	1109.42
Rural population share	%	81.2
Economic expenditure on rural area	Million RMB	900.8

Source: Statistical Bureau, Tai Pusi County in 2012.

Table 2-D2 Value of β_i based on income distribution and labour force of stakeholder types

	Herders (β_1)	Farmers (β_2)	Ecological public authorities (β_3)	Economic public authorities (β_4)	Total
Income in 2012 (million RMB)	23.58	1134.099	83.3	900.8	2158.669
Income distribution(β_i)	0.0110	0.5295	0.0389	0.4206	1
Labour force in each stakeholder type in 2012	1962	55665	200	6601	64428
Labour force distribution(β_i)	0.0304	0.8640	0.0031	0.1025	1

Source: Statistical Bureau, Tai Pusi County in 2012.

Appendix 4-A: Descriptive statistics of the model variables

Table 4-A1 Descriptive statistics of the model variables

Variables	Mean	Std. Dev.	Min	Max	Observations
$\log(S_{it})$	0.36	0.54	-1.59 ²⁷	3.14	1080
$\log(C_{it})$	-0.05	0.49	-2.08	2.93	1080
$\log(U_{it})$	1.02	0.81	-0.50	4.23	1080
$\log(F_{it})$	1.05	0.79	-2.00	4.17	1080
$\log(R_{it-1})$	-1.13	3.30	-4.61	3.26	1080
$1/(R_{it-1})$	46.69	49.80	0.04	100 ²⁸	1080
$\log(M_{it-1})$	0.05	0.41	-0.53	0.66	1035
$\log(B_{it-1})$	0.12	0.50	-0.45	1.13	1035
$\log(T_{it-1})$	-0.03	0.06	-0.23	0.10	1035
$\log(G_{it-1})$	-0.28	0.42	-2.40	0.01	1035
Y_t	12.50	6.93	1 ²⁹	24	1080
$P_i * Y_t$	7.50	8.14	0	24	1080

²⁷ The logarithm of any positive number less than 1 is negative.

²⁸ $1/(R_{it-1})$ has the max value of 100 because we assigned a small original value (0.01) to R_{it-1} . In this case, it indicates R_{it-1} equals to 0.

²⁹ Year 1985 equals 1, 1986 equals 2, etc.

Appendix 4-B: The model specification with quadratic term

With respect to the specification of our models, we use the quadratic term instead of the reciprocal of land tenure reform variable. That is, $1/(R_{it-1})$ is replaced by $[\log(R_{it-1})]^2$, as every equation in below. Comparing table 4.3 with table 4-B1, and figure 4.9 with figure 4-B1, model results for these two types of specification are almost the same, which indicates the model specification is reliable and the model results are robust.

$$\begin{aligned} \log(S_{it}) = & s_i + a_1 \log(R_{it-1}) + a_2 [\log(R_{it-1})]^2 + a_3 \log(M_{it-1}) + a_4 \log(T_{it-1}) \\ & + a_5 \log(G_{it-1}) + a_6 Y_t + a_7 P_i * Y_t + \varepsilon_{it} \end{aligned} \quad (4B.1)$$

$$\begin{aligned} \log(C_{it}) = & g_i + b_1 \log(R_{it-1}) + b_2 [\log(R_{it-1})]^2 + b_3 \log(B_{it-1}) + b_4 \log(T_{it-1}) \\ & + b_5 \log(G_{it-1}) + b_6 Y_t + b_7 P_i * Y_t + \mu_{it} \end{aligned} \quad (4B.2)$$

$$\begin{aligned} \log(U_{it}) = & u_i + c_1 \log(R_{it-1}) + c_2 [\log(R_{it-1})]^2 + c_3 \log(M_{it-1}) + c_4 \log(T_{it-1}) \\ & + c_5 \log(G_{it-1}) + c_6 Y_t + c_7 P_i * Y_t + \lambda_{it} \end{aligned} \quad (4B.3)$$

$$\begin{aligned} \log(F_{it}) = & f_i + d_1 \log(R_{it-1}) + d_2 [\log(R_{it-1})]^2 + d_3 \log(B_{it-1}) + d_4 \log(T_{it-1}) \\ & + d_5 \log(G_{it-1}) + d_6 Y_t + d_7 P_i * Y_t + \delta_{it} \end{aligned} \quad (4B.4)$$

Table 4-B1 Model results based on Eqs. (4B.1)-(4B.4)

Variables	Sheep population model $\log(S_{it})$		Cattle population model $\log(C_{it})$		Mutton output model $\log(U_{it})$		Beef output model $\log(F_{it})$	
	coefficient	t	coefficient	t	coefficient	t	coefficient	t
$\log(R_{it-1})$	-0.13***	-6.86	-0.06***	-3.03	-0.07***	-3.36	-0.04*	-1.67
$[\log(R_{it-1})]^2$	-0.03***	-5.80	-0.02***	-3.38	-0.02***	-2.97	-0.01*	-1.91
$\log(M_{it-1})$	0.25***	3.62	na	na	0.08	1.11	na	na
$\log(B_{it-1})$	na	na	0.18***	3.84	na	na	0.19***	2.96
$\log(T_{it-1})$	-0.08	-0.19	1.54***	4.12	-0.64	-1.52	0.57	1.10
$\log(G_{it-1})$	1.06**	2.02	-0.28	-0.58	1.60***	2.90	0.81	1.17
Y_t	0.04***	8.47	0.01***	2.68	0.12***	22.23	0.10***	17.14
$P_t * Y_t$	-0.001	-0.38	-0.03***	-8.22	-0.02***	-4.38	-0.05***	-10.39
<i>Constant</i>	0.31**	1.98	0.10	0.68	0.21	1.28	0.47**	2.30
R^2	0.30		0.12		0.77		0.60	
<i>Observations</i>	1035		1035		1035		1035	

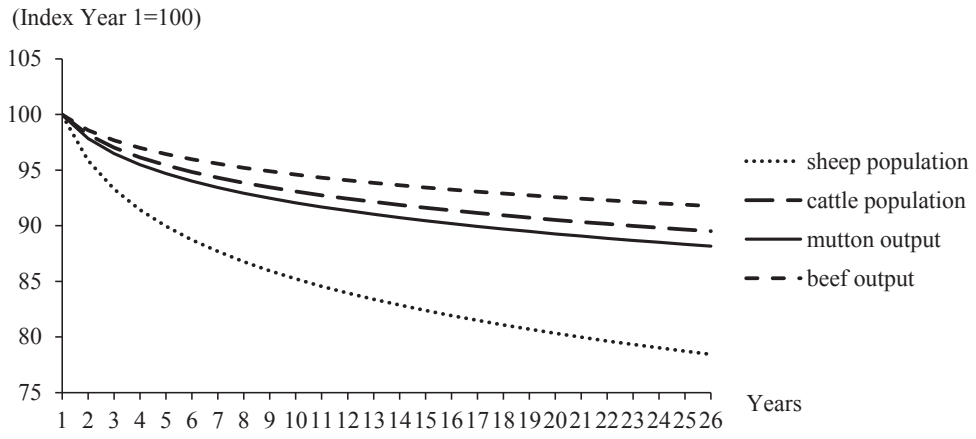


Figure 4-B1 Relationship between land tenure reform and livestock production based on the model results in table 4-B1

Summary

Grasslands play a significant role in the global ecosystem, livestock production and millions of household livelihoods. However, grassland degradation has become a worldwide problem, and rural poverty has become exacerbated in some pastoral areas. The sustainable management and use of grasslands that maintain the productive and adaptive capacity of ecosystems while providing for the well-being of human communities are crucial concerns. In practice, various policy interventions have been initiated by governments all over the world to devote to grassland conservation, livestock production and improving local household livelihoods. This research aims to evaluate the effectiveness of the policy interventions for grassland ecosystems and livestock production in the pastoral areas of China. The conceptual framework for this research is based on Ostrom's Social-Ecological Systems (SESs) framework. Four specific research questions are addressed. It is hoped that the research findings of this thesis can make a contribution to providing some references for policy development in light of the sustainable management and use of grasslands.

China has around 392 million hectares of grasslands, accounting for 12% of the world's grasslands and 41.7% of the national land area. The study about China's grassland system and its grassland policies could provide significant views to understand the grasslands of this world. Inner Mongolia, that includes the typical pastoral areas of China, is employed as the specific research region. The impacts of China's grassland policies on the grassland systems are dissected based on the methodology of econometrics, welfare economics and institutional economics and data from remote sensing technology and socio-economic survey. More specifically, the major policy interventions for the governance of grasslands in China include the land tenure reform during the 1980s-1990s and a series of Ecological Construction Programs occurring mainly after the year 2000. The former aims at privatizing use rights of grasslands to individual households and displacing traditional common use. The latter mainly concerns grassland conservation through employing grazing bans and deterministic stocking rates. The effectiveness of these policy interventions is investigated through estimating their impact on grassland condition and livestock production. The empirical analysis about policy impacts is conducted in chapter 3, 4 and 5, based on panel data that spans several decades. In addition, a case study based on a county in the eco-fragile area is conducted in chapter 2 to study the preference for different types of rural land by different stakeholders, which mirrors the perception of resource users on grassland use under the social, economic and political settings.

More in detail, chapter 2 analyses the divergence in preferences among different stakeholders over the allocation of four types of rural land: cultivated land, grassland, forest and other land. Considering the heterogeneity of stakeholders that are concerned with land-use decisions, four types of stakeholders are

distinguished, namely, ecological authorities, economic authorities, herders and farmers. The diverging preferences of these four stakeholder types over the different types of land use were quantified using the Analytic Hierarchy Process. Weights for each stakeholder type were derived for three scenarios: equal weights, weights based on income distribution and weights based on labour force distribution. Welfare analysis was employed then to determine the individual optimal allocation by maximising the utility function of each stakeholder type. Social optimal allocation was derived by maximising the social welfare function, which is the weighted sum of individual utilities. Tai Pusi County, located in an eco-fragile area of northern China, was taken as a case to present the empirical analysis. Individual optima revealed the degree of divergences among stakeholders, and the social optima revealed the optimal allocation based on social welfare. The research results provide insights on how to achieve an efficient allocation of rural land, balancing the ecological and economic benefits of different stakeholders from different types of land. As such, the desirable area of grasslands for different users as well as for the whole society is revealed, which further reflects the conflicts of different stakeholders on grassland use.

Prior to the policy intervention of assigning the grasslands to individual households, grazing was mobile and flexible in China, which allowed herders and their animals to roam over forage and water resources. Since the start of the land tenure reform in the pastoral areas of China in the 1980s, grassland use rights have increasingly been assigned to individual households and subsequently more grasslands have been in private use. On the other hand, most of the grasslands in China have experienced severe degradation after the reform. Chapter 3 aims at examining whether land tenure reform plays a significant role in grassland degradation, i.e. the impact of grassland privatisation on grassland condition. It first outlines the process of land tenure reform in the pastoral areas of China over time and then investigates the changes in grassland condition that are caused by the reform. Grassland condition is presented by grassland quantity and quality using spatial information based on remote sensing. The timing of the assignment of grassland use rights and the timing of the actual adoption of private use by households differ among counties, which allows to disentangle the impact of the land tenure reform at county-level. A fixed effects model that controls for the factors of climate, agricultural activity and environmental heterogeneity is used to conduct the empirical analysis, based on a dataset involving 60 counties of Inner Mongolia between 1985 and 2008. The model results show that the private use of grasslands following the land tenure reform causes the reduction of the total grassland area and leads to the degradation of dense and moderate grasslands into sparse grasslands in the long run. These results point to the existence of “a tragedy of privatisation”, as opposed to the well-known “tragedy of the commons”.

Chapter 4 aims at understanding whether land tenure reform has played a role in the changes in livestock production of pastoral areas, i.e. the impact of grassland privatisation on local animal husbandry. This objective is investigated based on county-level data for Inner Mongolia between 1985 and 2008. The

changes in livestock production over time are examined by analysing data on changes in livestock population and meat output. The descriptive analysis shows that livestock production increased at a higher speed in the crop farming areas, but the development of livestock productivity was faster in the pastoral areas. In the empirical analysis, we employed a fixed effects model to disentangle the effects of land tenure reform on livestock production from factors related to market forces, grassland condition, technology development and environmental heterogeneity. The model results reveal that the implementation of land tenure reform had significant and negative effects on the increase in livestock production, although the total livestock production actually increased over the research period. This shows that land reform in itself is unable to offset the impact of other factors that accelerate the increase in livestock production. Moreover, the constraining effect of land tenure reform on the increase in livestock production decreases with the years of implementing land tenure reform and ultimately disappears. Remarkably, the constraining effect of land tenure reform is stronger on the increase of livestock population than that of meat output. This indicates that land tenure reform has led to an improvement in livestock productivity. In conclusion, land tenure reform puts a ceiling on livestock production, which could be a possible reason why it has been difficult to implement the reform on grasslands. However, the reform has improved the livestock productivity of pastoral areas.

A series of Ecological Construction Programs have been initiated to protect the condition of grasslands in China during recent decades. However, grassland degradation is still severe and conditions have not been restored as intended. Chapter 5 investigates the effectiveness of Ecological Construction Programs for protecting the grassland condition in the extensive pastoral areas of China. One Ecological Construction Program that has been implemented widely on the grasslands, specifically the Subsidy and Incentive System for Grassland Conservation (SISGC), is taken as the concrete example for policy evaluation. The Normalized Difference Vegetation Index (NDVI), measured with remote sensing technology, is used to quantify the grassland condition between 2001 and 2014. With data from 54 counties in the pastoral areas of Inner Mongolia, we estimate the impact of SISGC on the grassland condition. A fixed effects model is employed to control for livestock production, climate, time trends and environmental heterogeneity. The model results provide quantitative evidence that the condition of the grasslands has improved significantly due to SISGC, but that the effectiveness of SISGC was offset to some extent by other socio-economic and climate factors, such as increased producer prices and high temperatures. This may explain why the actual grassland degradation has not been prevented as effectively as was expected. In addition, the impact of SISGC was stronger in counties with a worse initial grassland condition. Furthermore, the effects of producer prices and climate changes were also more pronounced in these counties.

Chapter 6 concludes this research with the main findings as well as the contribution to the existing scientific debates. Moreover, policy implications are suggested based on the findings of this research. It is finalised with the research limitation and recommendations for future research.

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Biography

Min Liu was born on the 26th of April, 1988 in Hanzhong City, Shaanxi Province, China. She studied Tourism Management at Northwest Agriculture and Forestry University, China from 2006 to 2010 and got her Bachelor degree of management. She then was recommended for Master of Agricultural Economics and Management in Northwest Agriculture and Forestry University, and obtained her Master degree in 2012.

In October 2012, Min started to study at Wageningen University, the Netherlands, funded by Chinese Scholarship Council to pursue her PhD degree under the supervision by Prof. Wim Heijman, Prof. Jikun Huang, Dr. Liesbeth Dries and Dr. Xueqin Zhu at Agricultural Economics and Rural Policy group. During the final year of her PhD, she joined in the Development Research Group of World Bank, Washington D.C. to work with Dr. Klaus Deininger. Her current research interests focus on social-ecological systems, grassland degradation, livestock production, eco-environmental policies, land use and management, land tenure, rural household livelihoods and rural transformation.

Min Liu

Wageningen School of Social Sciences (WASS)

Completed Training and Supervision Plan

*One credit according to ECTS is on average equivalent to 28 hours of study load



Name of the learning activity	Department/Institute	Year	ECTS*
A) Project related competences			
Advanced Microeconomics (ECH-32306)	WUR	2013	6
Theories and Models in Environmental Economics (ENR-30306)	WUR	2013	6
Rural Economic Analysis (AEP-31306)	WUR	2013	6
Advanced Econometrics (AEP-60306)	WUR	2014	
Cost-Benefit Analysis and Environmental Valuation (DEC-31306)	WUR	2014	
Agricultural Economics & Policy	WASS	2016	2
B) General research related competences			
Workshop: interviewing professionals	WASS	2012	0.5
Research Methodology: From topic to proposal	WASS	2013	4
Doing systematic literature review	WASS	2013	3.7
WASS Introduction Course for new PhD candidates	WASS	2013	1.0
Techniques for Writing and Presenting a Scientific Paper	WASS	2015	1.2
<i>'Tragedy of the commons or tragedy of privatization? The impact of land tenure reform on grassland condition in Inner Mongolia, China'</i>	- EAAE PhD Workshop, Roma - WASS PhD day - International Conference "Food in the Bio-based Economy", Wageningen - 29th International conference of agricultural economists (ICAE), Milano	2015	1
<i>'What is the impact of land tenure reform on livestock production in pastoral areas? An empirical study conducted in Inner Mongolia, China'</i>	- Workshop on tenure reforms in different land use types in China, Beijing, China - The Fourth International Conference on Natural Resources and Agricultural Economics), Hohhot, China	2015	1
<i>'Individual and Social Optima of Rural Land Allocation by Stakeholders: A Case Study on Eco-fragile Areas of Northern China'</i>	The 29th International conference of agricultural economists (ICAE), Milano, Italy	2015	1
<i>'The impact of Ecological Construction Programs on grassland conservation in the pastoral areas of China'</i>	- The 2016 China Conference of the Chinese Economists Society, Shenzheng, China - Lunch seminar at the Department of Agriculture, Food and Environment, University of Pisa, Italy	2016	1
C) Career related competences/personal development			
Internship	Development Research Group, World Bank	2016	6
Total			40.4