Land Tenure Security and Rangeland Management in Inner Mongolia, China

Luci Lu
Department of Geography
McGill University, Montreal

A thesis submitted to McGill University in partial fulfillment of the requirement of the degree of Doctor of Philosophy

Abstract

Recent transformations in political and economic environments around the globe have resulted in broad changes in pastoral property rights and tenure over land, as well as the way that herders manage their lands. The rangelands of China are one of the largest pastoral regions that have experienced significant and somewhat consistent reform of their tenure systems. Grassland areas that had long been held as communal and collectively managed were privatized between the late 1980s to the early 2000s. As a by-product of this land privatization, land rental markets have emerged and grown in Inner Mongolia in the past decade. However, little is known about how rental markets, and namely their land tenure security, affect rangeland and herder outcomes. The objective of this dissertation is to use the emerging rental market in Inner Mongolia as a case study to explore the relationships among grassland change, tenure insecurity, and pastoral resource management. This dissertation is based on the field-collected data from Inner Mongolia, China.

It is commonly assumed that after rangeland privatization in Inner Mongolia, individual herders are confined to grazing on registered land parcels where exclusive ownership is granted. As a result, environmental policies often use the amount of the legally recognized land as a proxy for the land used by a herding household. But how well does this assumption hold? In the first manuscript, I use a detailed household survey on grassland use and management to examine the discrepancies between herders' legally-held (contracted) lands and other land used but not formally registered. The results show that herders' informally-managed lands are significant and have dramatic implications for calculating stocking rates around which numerous policies are based.

In the second manuscript, I use a discrete choice experiment to reveal herders' preferences and trade-offs among a range of tenure-security influencing attributes. It is increasingly agreed that "tenure (in)security" is a composite idea that reflects a number of underlying conditions that lead to a herder feeling confident in their land rights. Yet, there is little evidence regarding how these are traded off. Using a land rental transaction as a scenario, I offered herders hypothetical choices of land contracts with different attributes to evaluate their influence on land users'

perception of tenure security. I find strong evidence that herders prefer formal kinship contracts with additional land rights (i.e., the ability to receive eco-compensation) over other informal or contractual options. I further explain the preference heterogeneity in other attributes with qualitative evidence collected along with the choice experiment data.

Land rental markets facilitate efficient utilization of land, however, the short duration of occupancy and limited property rights mean that rental contracts may discourage longer-term sustainable land management. Direct investigation into the relationship between rangeland rights and ecological outcomes has been hampered by scale-appropriate data on land tenure, resource management, and rangeland outcomes. In the third manuscript, I address these issues with a study design that combines participatory mapping, household surveys, and remote sensing. I analyzed these combined data in a multilevel statistical model, controlling for environmental and land management influences. My results show that rented parcels are associated with worse rangeland outcomes compared to privately-held parcels. This study contributes to the literature by documenting important empirical effects of rental markets and presenting a replicable workflow for integrating earth observations and micro-level survey data. Pragmatically, these results have important implications for incentive and compensatory-based rangeland policy.

Résumé

Les transformations récentes des environnements politiques et économiques dans le monde entier ont entraîné de vastes changements dans les droits de propriété pastoraux et les régimes fonciers, ainsi que dans la manière dont les éleveurs gèrent leurs terres. Les pâturages de Chine sont l'une des plus grandes régions pastorales à avoir connu une réforme importante et relativement cohérente de leurs systèmes fonciers. Les zones de pâturage, qui ont longtemps été détenues par la communauté et gérées collectivement, ont été privatisées entre la fin des années 1980 et le début des années 2000. En tant que sous-produit de cette privatisation, des marchés de location de terres ont émergé et se sont développés en Mongolie intérieure au cours de la dernière décennie. Cependant, on sait peu de choses sur la façon dont les marchés de location, et notamment leur sécurité foncière, affectent les résultats des parcours et des bergers. L'objectif de cette thèse est d'utiliser le marché locatif émergent en Mongolie intérieure comme étude de cas pour explorer les relations entre le changement des prairies, l'insécurité foncière et la gestion des ressources pastorales. Cette thèse s'appuie sur des données recueillies sur le terrain en Mongolie intérieure, en Chine.

Il est communément admis qu'après la privatisation des terres de parcours en Mongolie intérieure, les éleveurs individuels sont confinés au pâturage sur des parcelles de terre enregistrées dont la propriété exclusive leur est accordée. Par conséquent, les politiques environnementales utilisent souvent la superficie des terres légalement reconnues comme une approximation des terres utilisées par un ménage d'éleveurs. Mais dans quelle mesure cette hypothèse se vérifie-t-elle ? Dans le premier manuscrit, j'utilise une enquête détaillée auprès des ménages sur l'utilisation et la gestion des pâturages pour examiner les écarts entre les terres détenues légalement par les éleveurs (sous contrat) et les autres terres utilisées non enregistrées. Les résultats montrent que les terres gérées de manière informelle par les éleveurs sont importantes et ont des implications dramatiques pour le calcul des taux de charge sur lesquels de nombreuses politiques sont basées.

Dans le second manuscrit, j'utilise une expérience de choix discret pour révéler les préférences et les compromis des éleveurs quant à certains attributs influençant leur sécurité foncière. Il est de

plus en plus admis que la (l’) "(in)sécurité foncière" est une notion composite qui reflète un certain nombre de conditions sous-jacentes qui font qu'un éleveur se sent confiant dans ses droits fonciers. Cependant, il existe peu de preuves concernant la manière dont ces conditions sont échangées. En utilisant une transaction de location de terres comme scénario, j'ai proposé aux éleveurs des choix hypothétiques de contrats fonciers avec différents attributs afin d'évaluer leur influence sur la perception de la sécurité foncière par les utilisateurs des terres. Je trouve des preuves solides que les éleveurs préfèrent les contrats formels de parenté avec des droits fonciers supplémentaires (c'est-à-dire la possibilité de recevoir une éco-compensation) à d'autres options informelles ou contractuelles. J'explique en outre l'hétérogénéité des préférences pour d'autres attributs à l'aide de données qualitatives recueillies en même temps que les données de l'expérience des choix.

Les marchés de location de terres facilitent l'utilisation efficace des terres, mais la courte durée de l'occupation et les droits de propriété limités signifient que les contrats de location peuvent décourager la gestion durable des terres à plus long terme. L'étude directe de la relation entre les droits sur les parcours et les résultats écologiques a été entravée par l'absence de données appropriées à l'échelle sur le régime foncier, la gestion des ressources et les résultats sur les parcours. Dans le troisième manuscrit, j'aborde ces questions avec un modèle d'étude qui combine la cartographie participative, les enquêtes auprès des ménages et la télédétection. J'ai analysé ces données combinées dans un modèle statistique à plusieurs niveaux, en contrôlant les influences de l'environnement et de la gestion des terres. Mes résultats montrent que les parcelles louées sont associées à de moins bons résultats pour les pâturages que les parcelles privées. Cette étude contribue à la littérature en documentant les effets empiriques importants des marchés locatifs et en présentant un processus reproductible pour l'intégration des observations terrestres et des données d'enquête au niveau micro. D'un point de vue pragmatique, ces résultats ont des implications importantes pour les politiques d'incitation et de compensation des pâturages.

Acknowledgments

I want to thank my supervisor, Brian Robinson, for providing me with such an invaluable opportunity to learn and explore. I discovered many fascinating aspects of research as well as my academic passion and potential during the Ph.D. study, and I am grateful to Brian for his guidance, support, and encouragement in making this journey fun and rewarding. I would also like to thank Li Ping from the Grassland Research Institution in the Chinese Academy of Agricultural Sciences. Li Ping has given this project her full support and is always willing to spend time discussing and offering her insights on my research ideas. I am thankful to my committee members John Galaty for inviting me to join his lab meetings and activities in the Anthropology Department, and Sarah Turner and Margaret Kalácska for their advice and help in making my dissertation better.

I appreciate all the herders who took the time to participate in the survey and interview, and many of them who shared their knowledge, life experiences, thoughts, and humor with me. My gratitude extends to many of the herders and farmers I have interviewed or worked with in pastoral China over the past ten years. Listening to and learning from the land users during each fieldwork continue to motivate me to think critically.

Additionally, I am thankful to our driver Cheng Jianzhong and his family for using their local network to help in various ways throughout this fieldwork. The surveys and interviews were conducted with the effort of the interpreters Hashigaowa, Tana, Muren, Erdmutu, Bubu, Chaokeorigele, and Gerile. My friend Uyunga has carefully translated, modified, and formatted the survey based on cultural etiquette. Zhang Bao from Inner Mongolia Agricultural University has kindly shared his experience and suggestions on choice experiment design and fieldwork with me. Wu Liwen and Zhi Rong from Li Ping's Lab have helped significantly with the logistics and field preparation for this study. Conducting most of my previous fieldwork and community work by myself, I appreciate Wu Liwen for her company during six months of fieldwork. We worked as each other's research assistant and overcame many obstacles together. I am also indebted to Wu Liwen as she took most of the burden of filing expense reports, which undeniably is one of the most time-consuming but underrecognized parts of fieldwork.

It is a great pleasure for me to be one of the McGill Sustainability Systems Initiative (MSSI) Landscape scholars at McGill. The mentors (Elena Bennet and Brian Robinson) and members of the program have truly inspired and empowered me to learn and grow fast as an independent researcher. I am also fortunate to be able to co-organize events for the Landscapes and Livelihoods Collaboratory with Jackie Hamilton – it was so fun bringing people from different disciplines together! Our GIC director Tim Elrick has provided tremendous help during my doctoral study from offering various skills training to solving issues in my code. I have also greatly benefited from the writing course and online writing sessions offered by Mariève Isabel in Graphos. I am glad to be accompanied by the Robinson, Manaugh, and Chmura lab members in offices 312 and 315 and fellow geographers and David Clark in Burnside. Thanks to Annie Lee, Hongyu Zhang, Holly Cronin, Yiyi Zhang, Ana Lucía Araujo Raurau, Arunabha Dey, Patrice Matthews, and Sai Ma for the bubble tea, boardgames, beers, and laughs. Finally, I thank my mom and dad for being proud of every micro progress that I have made and for always supporting my decisions.

Research funding for this study comes from the US National Science Foundation grant (no. DEB-1313693) by Brian Robinson and funding acquired by Li Ping, which includes grant from the Technology Innovation Program of Chinese Academy of Agricultural Sciences (No. CAAS-ASTIP-2021-IGR-05), the Natural Science Foundation of Inner Mongolia (2018MS07007), and the Key Technology Projects of Inner Mongolia Autonomous Region (2019GG012). I am grateful for the additional financial support from the McGill Sustainability Systems Initiative (MSSI) and Rathlyn Foundation (via Rathlyn GIS Fellowship) for my Ph.D. study. The personal expenses of my fieldwork are covered by the Rathlyn fieldwork award, the graduate field research award from the Institute for the Study of International Development (ISID), and a seed grant from the Social & Cultural Data Science Centers (CSCDS).

“Because a doubt is not a denial. Doubt is a powerful tool, and it should be applied to history.”

— Walter M. Miller Jr., A Canticle for Leibowitz, 1959

Contribution to Original Knowledge

This dissertation uses a mixed methods approach to investigate the impact of the increasing rangeland rental market on land rights, incentive-based conservation policies, and rangeland outcomes. **Chapter 2** focuses on informally-managed lands and virtual lands, the two land-based resources neglected in the literature and in policy design. I use household survey data to show that the negligence toward such *de facto* controlled land leads to a miscalculation of incentives and punishments, potentially undermining herder livelihoods. This dissertation provides insights and practical recommendations for the ongoing rangeland eco-compensation programme and similar incentive-based programmes. **Chapter 3** measures herders' tenure insecurity from an individual herder's perspective. A challenge in the literature is that factors that influence tenure insecurity are usually measured on different scales, making comparison of the relative importance of various factors (e.g., kinship-based contract versus duration of a contract) virtually impossible. To my knowledge, this is the first study that overcomes the difficulties and compares them in an "apples-to-apples" manner. I quantify to what extent each factor contributes to herders' perceived tenure security and provide straightforward policy recommendations. In **Chapter 4**, I designed a novel method to (a) delineate rangeland boundaries using open access satellite data and (b) collect land rights and land management data through participatory mapping with individual herders, allowing evaluation of land rights' relationship with rangeland outcomes at a finer scale and with more precision than any previous work. This study contributes to the "people and pixel" research, especially at the household level by addressing the methodological difficulties of linking herders' land tenure and management practices to their properties.

Contribution of Authors

I am the primary and lead author of all the chapters of this dissertation. For Chapters 2, 3, and 4, I lead the development of the research ideas and methods, collecting data in the field, analyzing and interpreting data, and writing the original draft. Brian Robinson has provided iterative advice on conceptualizing research activities, study design, interpreting the data, and editing the manuscript. Li Ping has provided advice on the design of Chapters 2 and 3, helped edited Chapters 2, 3, and 4, and provided logistic and administrative support during fieldwork. Margaret Kalácska has provided suggestions on methodology and reviewed Chapter 4.

List of Abbreviations

GIS	Geographical Information Systems
GPS	Global Positioning System
NPP	Net Primary Productivity
ANPP	Aboveground Net Primary Productivity
MODIS	Moderate Resolution Imaging Spectroradiometer
CHIRPS	Climate Hazards Group InfraRed Precipitation with Station
GsMaP	Global Satellite Mapping of Precipitation
IMERG	Integrated Multi-satellitE Retrievals for Global Precipitation Measurement
GLDAS	Global Land Data Assimilation System
SRTM	Shuttle Radar Topography Mission
NIR	Near Infrared
SWIR	Shortwave Infrared
NDVI	Normalized Difference Vegetation Index
SAVI	Soil Adjusted Vegetation Index
MSAVI	Modified Soil Adjusted Vegetation Index
PES	Payment for Ecosystem Services
fAPAR	fraction of absorbed photosynthetically active radiation

Table of Contents

List of Figures

List of Tables

Chapter 1

Introduction

1.1 Research objectives

In this dissertation, I use emerging rental markets in Inner Mongolia Autonomous Region, China, as a case study to explore how the changes in land rights and land tenure influence herders' land management practices. Herders in Inner Mongolia generally have 30 years of (renewable) land use and contractual rights to their land. After rangeland use rights were contracted to pastoral households in the 1990s, the land-dependent and subsistence-based local economy gradually transitioned to a relatively intensified economy that depends on infrastructure, external forage supply, and hybrid breeds (Robinson, Li, and Hou 2017; Li and Li 2012). The emerging rental market in IMAR facilitates land use efficiency by reallocating land from non-operating herders to herders with low initial land endowments (Tan et al. 2017). Previous agricultural studies suggest that short-term rental agreements may encourage exploitative grazing since there is no incentive for renters to invest in long-term land management. Nevertheless, the literature is sparse on how rental markets affect rangeland outcomes and interact with incentive-based policies. The growth of rental markets has also resulted in more land transactions among herders, many of which are still informal. Informally managed land raises many questions about the effectiveness of and interactions with incentive-based conservation programmes, many of which are still unknown. This dissertation is structured around three questions:

- How do informal land use and reliance on external purchased forage interact with rangeland policies?
- What are the factors that influence herders' perception of tenure insecurity? Among the factors, which has the most influence?
- How do short-term rental agreements affect herders' land-use intensity?

I answer each question in chapters 2, 3, and 4, respectively, and synthesize the findings in chapter 5. In this chapter, I review the past decades' conceptual advances in rangeland ecology and management and relevant literature on land rights and land tenure security.

1.2 Literature review

The past several decades have seen rapid conceptual advances in non-equilibrium theory, changing our understanding of the causes, patterns, and processes of rangeland change. The ecological models and assumptions we adopt determine the research and management plans that we design (Ellis and Swift 1988; Westoby, Walker, and Noy-Meir 1989). Therefore, it is necessary to have an introduction to the ecological and social theories that guided the design of this dissertation and its experiments. I use the social-ecological systems framework (Ostrom 2009; Mcginnis and Ostrom 2014) to conceptualize the interrelations of these theories. Throughout, I consider rangelands "ecological systems supporting native or naturalized vegetation characterized as grasslands, shrub steppe, shrublands, savannas, and deserts that are managed as adaptive social-ecological systems to provision multiple ecosystem services to benefit human well-being" (Briske 2017; 17). This introductory chapter begins with a review of the unique characteristics of rangeland as a resource system (Figure 1), with a focus on equilibrium/non-equilibrium properties and how different rangeland models shape land management schemes and policies (section 1.2.1). In section 1.2.2, I review the literature on rangeland ecosystem services and the rise of payment for ecosystem as a conservation mechanism. In addition, I discuss the distinct features of rangeland property rights and how it has evolved to accommodate the tempo-spatial distribution of rangeland resources and to maximize herders' security and benefit in an uncertain environment (section 1.2.3). Finally, I introduce the ecological and social context of my study area Xilin Gol in section 1.3.

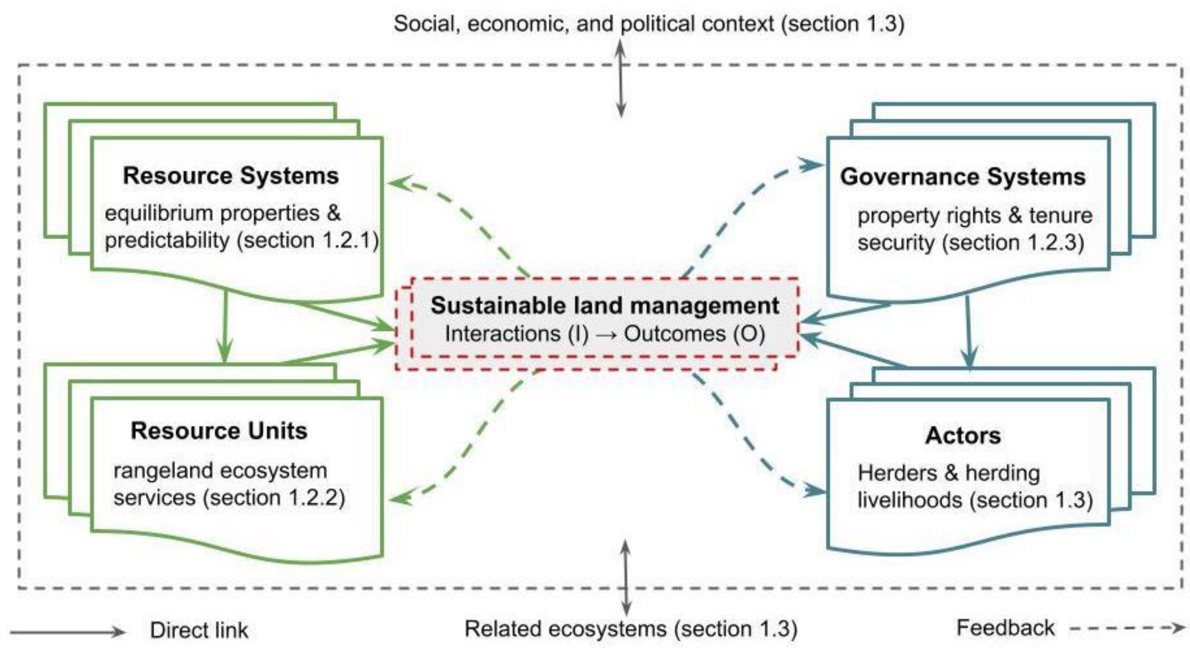


Figure 1. The rangeland social-ecological system framework (modified base on McGinnis and Ostrom 2014).

1.2.1 The abiotic and biotic influences on rangeland systems

To understand rangeland systems and management, we must first examine the abiotic and biotic influences on rangeland systems. Prior to the 1980s, the equilibrium theory guided rangeland science and management around the globe. The dominant equilibrium paradigm at the time assumed that most rangelands are stable systems regulated by internal mechanisms such as the closely coupled plant-livestock interactions (Ellis and Swift 1988). Under equilibrium theory, the condition of rangelands should be able to be determined by livestock density. Equilibrium-based models such as the range succession model predict that plant species composition progresses linearly and continuously from a less desirable stage to a stable "climax" stage (Dyksterhuis 1949). Disturbances such as drought, grazing, or fire cause plant communities to retrogress to an earlier stage. Removing or reducing the intensity of such disturbances will thus stop retrogression and restart succession again. Identifying grazing intensity as one of the critical counter-succession mechanisms, the range succession model justifies the use of destocking – decreasing the density of livestock in grasslands – to restore the ecological and economic values of the rangelands (Westoby, Walker, and Noy-Meir 1989; Sayre 2017).

Alternatively, the non-equilibrium theory proposes that pastoral dynamics are predominately regulated by external abiotic factors such as episodic droughts. The range succession model was challenged in the late 1980s because it could not explain the disturbance-driven vegetation dynamics and the nonlinear herbivore-plant interactions observed, especially in regions with high inter-annual variability of precipitation (Westoby, Walker, and Noy-Meir 1989). According to Ellis and Swift (1988), stochastic multi-year droughts cause massive livestock mortality and prevent the stocking rate from reaching the theoretical carrying capacity. They argue that rangeland change is predominantly governed by external climatic disturbances rather than internal biomass-livestock interactions (Ellis and Swift 1988). As a consequence, grazing has a lesser, if not negligible, impact on biomass in non-equilibrium systems as compared to equilibrium systems. Scientists believe that rather than focusing on carrying capacity, land management in highly variable non-equilibrium range systems should prioritize mobility, fodder supply, and herders' ability to sell livestock during drought years (Ellis and Swift 1988; Scoones 1995). Ellis (1994) suggests non-equilibrium dynamics are more likely to be observed in regions (in Africa) that receive less than 600mm of precipitation per year and have a coefficient of variation of inter-annual precipitation (CVP) greater than 33%.

A critique of Ellis and Swift's non-equilibrium persistent model is that it neglects the spatial-temporal heterogeneity in pastoral resource distribution. According to Illius and O'Connor (1999; 2000), although livestock is uncoupled from forage in the wet season, they are closely coupled, or in equilibrium, with limited (key) dry season resources. Such key resources can be represented by the drainage lines or riverine strips in Zimbabwe that are significant in dry seasons or drought years in sustaining livestock populations (Scoones 1995). Illius and O'Connor (2000) conclude that the existence of episodic droughts does not justify that livestock has no impact on biomass; thus, grazing interventions may still be useful for heterogeneous landscapes.

The fact that livestock Is in equilibrium with key dry season resources has implications for managing both subsistence-based and market-oriented range systems. Areas with a high ratio of key dry season resources are more susceptible to intensive grazing and potential biomass decline (Illius and O'Connor 2000). Theoretically speaking, since supplemental feed (or purchased forage) can be seen as a substitute for key natural resources during the dry season, herders who rely heavily on supplemental feed can buffer against climatic perturbations and maintain high stocking rates and grazing intensity (Behnke and Mortimore 2016). Better access to water supply infrastructure and supplemental feed can reduce or even eliminate drought-induced livestock mortality (Ho 2001; Vetter 2005; Briske, Illius, and Anderies 2017) . However, a heavy reliance on supplemental feed and purchased forage also exposes herders to forage price fluctuations, which are frequently exacerbated by blizzards or drought (Li and Li 2021). It remains understudied how herders balance the risks and potential economic benefits when using supplemental feed.

The equilibrium and non-equilibrium debate has helped show that neither equilibrium nor non-equilibrium theory alone can comprehensively explain rangeland vegetation dynamics (Briske, Fuhlendorf, and Smeins 2003; Vetter 2005). Depending on a system's ecological attributes and the temporal-spatial window assessed, equilibrium *or* non-equilibrium features may be apparent

(Fernandez-Gimenez and Allen-Diaz 1999; Fuhlendorf, Briske, and Smeins 2001). Thus rangeland management schemes can have glaring flaws if designed solely based on equilibrium or non-equilibrium theory. Equilibrium theory justifies the adoption of carrying capacity or fixed stocking rate to restore rangelands (Sayre 2008) and the privatization of communal pastoral lands to enhance livestock productivity (Sayre 2017). However, such land interventions often appear counterproductive on unpredictable and marginal lands, reducing herders' opportunities to access heterogeneous resources and exposing them to greater risks during drought (Ellis and Swift 1988; Nyima 2015; Hobbs et al. 2008).

Overemphasizing disturbance events and climatic variability, on the other hand, absolves livestock owners and range managers of any responsibility, depriving them of the opportunity to learn and gain experience through decision-making (Watson, Burnside, and Holm 1996). Similarly, excessive attention to the coefficient of variation of inter-annual precipitation at 33% as the division line between equilibrium and non-equilibrium dynamics leads to the neglect of other vital drivers (Briske et al. 2020). Research efforts should develop more comprehensive models that acknowledge the various equilibrium and non-equilibrium dynamics at different temporal-spatial scales (Vetter 2005; Briske 2017) and their complex socio-economic and biophysical drivers (Sayre et al. 2012).

1.2.2 Rangeland ecosystem services

Market-based conservation mechanisms such as payment for ecosystem services (PES) have been proposed as an alternative to traditional command and control regulations for managing rangelands. Ecosystem services are the benefits and goods people acquire from the environment (Daily 1997). The services supplied by the ecosystem are divided into four major categories: provisioning services (e.g., goods such as wool and meat), supporting services (often now recognized simply as ecosystem processes or functions, e.g., nutrient cycling and primary productivity), regulating services (e.g., flood control and pollination), and cultural services (e.g., non-material benefits such as recreation) (Millennium Ecosystem Assessment 2005). Payment for ecosystem services is an incentive-based instrument to internalize human-induced externalities (Engel, Pagiola, and Wunder 2008). In such schemes, the beneficiaries of an ecosystem service provide some payment to the providers or managers of the ecosystem service to maintain, restore, or enhance the service.

Rangelands have long been valued for their provisioning services, but only recently have their diverse range of services been more widely acknowledged. In addition to providing livestock, rangelands provide critical regulating services such as water supply and regulation, carbon sequestration (especially belowground), pollination, as well as supporting services such as biomass production and nutrient cycling, and cultural services such as outdoor recreation and cultural heritage preservation (D'Ottavio et al. 2018; Bengtsson et al. 2019). Previously assumed to benefit primarily herders, rangelands are now more commonly considered to provide crucial services to a number of stakeholders. Societal demand for rangeland ecosystem services has gradually shifted from a focus on provisioning services (e.g., meat) to greater emphasis on regulating and cultural services (e.g., carbon sequestration and eco-tourism) (Yahdjian, Sala, and Havstad 2015). Maximizing regulation and supporting services frequently comes at the expense of provisioning services and the economic benefits to land users, and vice versa (Raudsepp-Hearne, Peterson, and Bennett 2010). Such trade-offs among multiple stakeholders must be carefully evaluated before implementing land management plans (Bestelmeyer and Briske 2012; D'Ottavio et al. 2018) in order to maintain ecosystem services while improving human wellbeing (Zhang et al. 2020).

The focus of this dissertation is China's ongoing rangeland eco-compensation program named "Grassland ecological protection reward-subsidy policy (草原生态保护补助奖励政策)," also referred to as "rangeland eco-compensation programme". The eco-compensation programme can be classified as a "PES-like" programme because it focuses on internalizing positive externalities through incentives (Shang et al. 2018), though does not possess all the characteristics of a pure PES scheme (Grieg-Gran, Porras, and Wunder 2005). In the rangeland eco-compensation programme the central government of China provides annual cash payments for pastoral herder households to reduce stocking or cease grazing (Li et al. 2020). The rangeland eco-compensation programme is similar but not identical to PES since the former forces service providers (herders) to participate and change land use practices, which goes against the original voluntary intent of PES (Grieg-Gran, Porras, and Wunder 2005). I should also note that eco-compensation differs significantly from more classical "ecological compensation", which usually refers to a punishment-based scheme paid by the polluter rather than the beneficiaries (Shang et al. 2018). Still, in this case I argue that the rangeland eco-compensation contains additional command-and-control regulatory features since it is delivered universally to all pastoral households regardless of geographical differences in cost and return (Engel, Pagiola, and Wunder 2008).

The reason that China's rangeland eco-compensation has become one of the only large-scale payment-based conservation programmes implemented in the rangeland areas owes to China's unique land tenure systems. Pastoral tenure systems are known for their ambiguous and overlapping access rights and a lack of clear boundaries, which complicate the implementation of payment-based conservation programmes (Dougill et al. 2012). Over the past forty years, China—and Inner Mongolia in particular—has carried out a series of land interventions that registered and clarified private land tenure and land rights. The privatized land systems in Inner Mongolia have created a relatively favorable environment for implementing payment-based conservation programmes. In section 1.2.3, I introduce the key concepts in land tenure research and changes in pastoral tenure and property rights.

1.2.3 Rangeland rights and tenure security

Land tenure is defined by Food and Agriculture Organization (2002; 7) as "the relationship, whether legally or customarily defined, among people, as individuals or groups, with respect to land." USAID (2008; 1) defines land tenure in a similar way:

> [land tenure] is the institutional (political, economic, social, and legal) structure that determines [1] How individuals and groups secure access to land and associated resources, including trees, mineral, pasture, and water; [2] Who can hold and use these resources for how long and under what condition.

Separate from the rights over land (i.e., land tenure) is the security over those rights (Arnot, Luckert, and Boxall 2011; Robinson et al. 2018). The concept of land tenure security has evolved over time and does not have a universally agreed-upon definition by all disciplines. One common approach is to think of tenure security as the composition of the *breadth*, *duration*, and *assurance* of rights (Place, Roth, and Hazell 1994). The *breadth of rights*, or "bundle of rights," includes rights of access, withdrawal, management, exclusion, alienation, and to due process and compensation (Schlager and Ostrom 1992; RRI 2012). Different users may possess different rights. For example, a “claimant” may access and obtain certain resources but is incapable of selling or leasing the rights of access (Schlager and Ostrom 1992). The *duration* of rights determines how long given rights can be held valid. *Assurance of rights* is the “[varying degrees of certainty] that rights and duration are held,” which can be reduced or enhanced by the availability of social, economic, and political resources and the level of legal enforcement (Place, Roth, and Hazell 1994; 20). Sjaastad and Bromley (2000) argue that both the bundle size and duration of rights only indicate the *substance* of rights rather than how certain it can be upheld. They propose that tenure security should only be defined as an individual’s perception of whether his or her rights will be recognized and supported by society (Sjaastad and Bromley 2000).

Taking tenure security to mean how certain an individual is that their rights will be upheld by society, a key question must be answered: does it matter whether rights are recognized by the state or by local communities? State-led formalization can be costly, reinforcing gender and wealth inequality and making land susceptible to elite expropriation (Sjaastad and Cousins 2009). Customary or community-based tenure systems, on the other hand, although proven to stimulate investment in certain regions (Place and Hazell 1993), can result in highly gender-biased systems or reinforce poor dynamics that may exist via local political or ethnic tensions (Stickler and Huntington 2015; Lovo 2016). Van Gelder (2010) argues that security is a composite idea and that both formal and informal institutions affect land users’ perceptions of security, influencing their management practices. Instead of relying on an “appropriate measurement” identified by outsiders, priority should be given to land users’ perception of land tenure security (Sjaastad and Bromley 2000).

Pastoral land tenure is often described as “paradoxical” because herders require exclusive control over essential resources that sustain their livelihoods while also requiring flexible access to resources to address forage shortages caused by unforeseen socio-political and climatic events (Fernandez-Gimenez 2002). The paradoxical nature of rangeland tenure would require it to be an adaptive mechanism that accommodates the tempo-spatial heterogeneous and variable distribution of rangeland resources, but also allow for efficient use of those resources by land users. In response to the tempo-spatial heterogeneity of forage production, herders historically and still frequently migrate to more productive mountain pastures in the summer and return to lowland ranges that are less vulnerable to blizzards in the winter (Huntsinger, Forero, and Sulak 2010; Liao et al. 2014). Herders also remain mobile due to climate variability. Consecutive years of drought and severe winter blizzard causes excess livestock mortality and threatens pastoral livelihoods, which drives herders to expand their geographical extent of herding in order to find forages (Ellis and Swift 1988; Fernandez-Gimenez et al. 2015). Herders still may have emergency access to the forage of other pastoral groups, but this access is often reciprocal and governed by an informal consensus that, in exchange, the forage providers can access the other's resources when the former is affected by a future disaster. Literature shows that, at least in pastoral Asia, free and unrestricted access to rangelands is uncommon in mobile pastoral societies and that exclusion is often achieved by informal rules or kinship connections rather than physical land boundaries (Banks 2001; Fernandez-Gimenez 2002).

There has been an increase in the privatization of communal lands and the fragmentation of dryland landscapes around the globe (Behnke 2008; Hobbs et al. 2008). Behnke (2008) suggests the commercialization of land resources and the centralization of state authority are the primary catalysts of rangeland privatization. Furthermore, the neo-Malthusian discourse and Sahelian drought and famine in the 20^{th} century facilitated the public's acceptance of Hardin's "tragedy of commons", reinforcing the fear of resources under communal control, and justifying land interventions and the adoption of carrying capacity-based western rangeland science (Sayre 2017). The assumption that herders always maximize personal benefit and that "freedom in commons brings ruin to all" (Hardin 1968; 1244) helped rationalize the division of communal grazing land into exclusive properties. In China and elsewhere, policymakers generally assume that the privatization of land use rights incentivizes herders to make long-term sustainable investments in an effort to combat land degradation (Banks 2001). Additionally, it is also believed that land rights privatization enables the state to better implement carrying capacity-based conservation policies at a household level (Williams 1996).

1.3 Study area and methods

1.3.1 Study area

My study area—Xilin Gol prefecture (*league*)—is located in the Inner Mongolia Autonomous Region, a province in Northern China on the border of Mongolia and Russia. Inner Mongolia has a heterogenous landscape shaped by a clear precipitation gradient. A decrease in annual precipitation from the northeast to the southwest leads to a gradual decline of vegetation cover and productivity. The spatial change in precipitation also corresponds with a landscape transition from conifer and broadleaf forest to temperate grassland, and then to desert (Figure 2). In Inner Mongolia, precipitation also determines the temporal variation of vegetation productivity (Zhao et al. 2015). The complex landscape provides habitat for various plant and animal species, many of which are unique to the Mongolian steppe (Wu et al. 2015). In total, 467 bird species, 149 mammal species, and 2447 vascular plant species have been identified in Inner Mongolia (Feng, Yan, and Yang 2019). Inner Mongolia is predominately covered by grassland and barren desert, which supports extensive livestock grazing (Figure 3). Agriculture and agropastoralism are more common in the eastern and southern regions of the province close to dense human settlements, cities, and irrigation sources such as the Yellow river.

Xilin Gol has been an important location for ecological research because it encompasses several steppe types that are the most representative of the temperate Eurasian steppe. Covering over 193,300 km^2, Xilin Gol's rangeland transitions from meadow steppe, typical steppe, to desert steppe from northeast to southwest. The average temperature in Xilin Gol ranges from -15 °C in December to 23 °C in July. In 2019, Xilin Gol received 293.5 mm of precipitation, and 75% of the which (218.7 mm) happened between May and August (Xilin Gol League Statistical Bureau 2020; 27). Four counties (*banners*) were chosen for this study. Sonid Left and Sonid Right, the two counties that represent the desert steppe, had higher temperatures but received less precipitation in 2019 than the other two that represent the typical and meadow steppe (Xilin Hot and East Ujimqin, Table 1) (Xilin Gol League Statistical Bureau 2020; 353).

Table 1. Annual temperature and precipitation of the four counties (*banners*) in Xilin Gol.

	Sonid Left	Sonid Right	Xilin Hot	East Ujimqin
mean temperature (°C)	5.4	6.5	4.4	3.6
mean precipitation (mm)	172.1	191.1	293.5	229.1

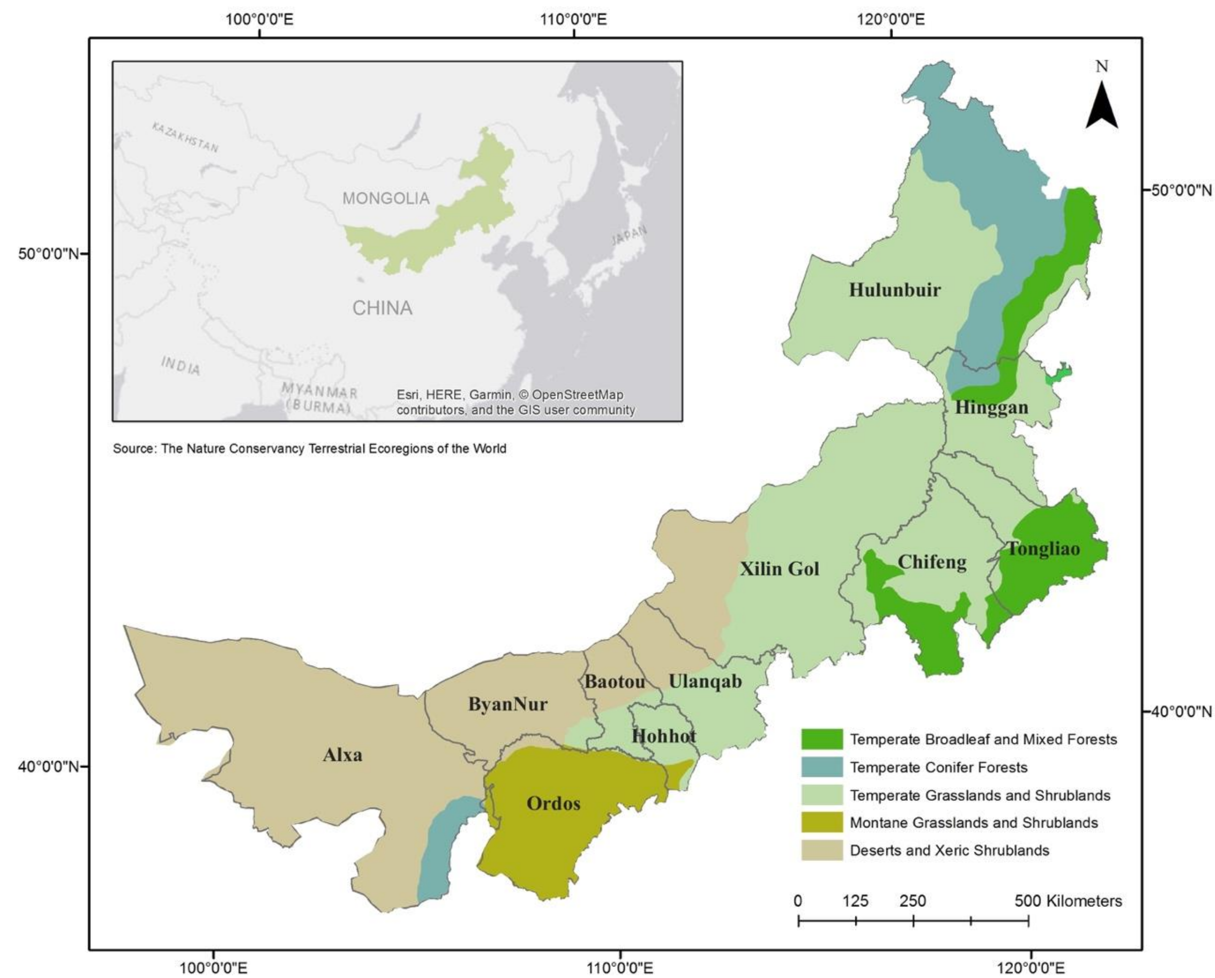


Figure 2. Different types of ecoregions and league/municipality boundaries in Inner Mongolia (The Nature Conservancy 2009).

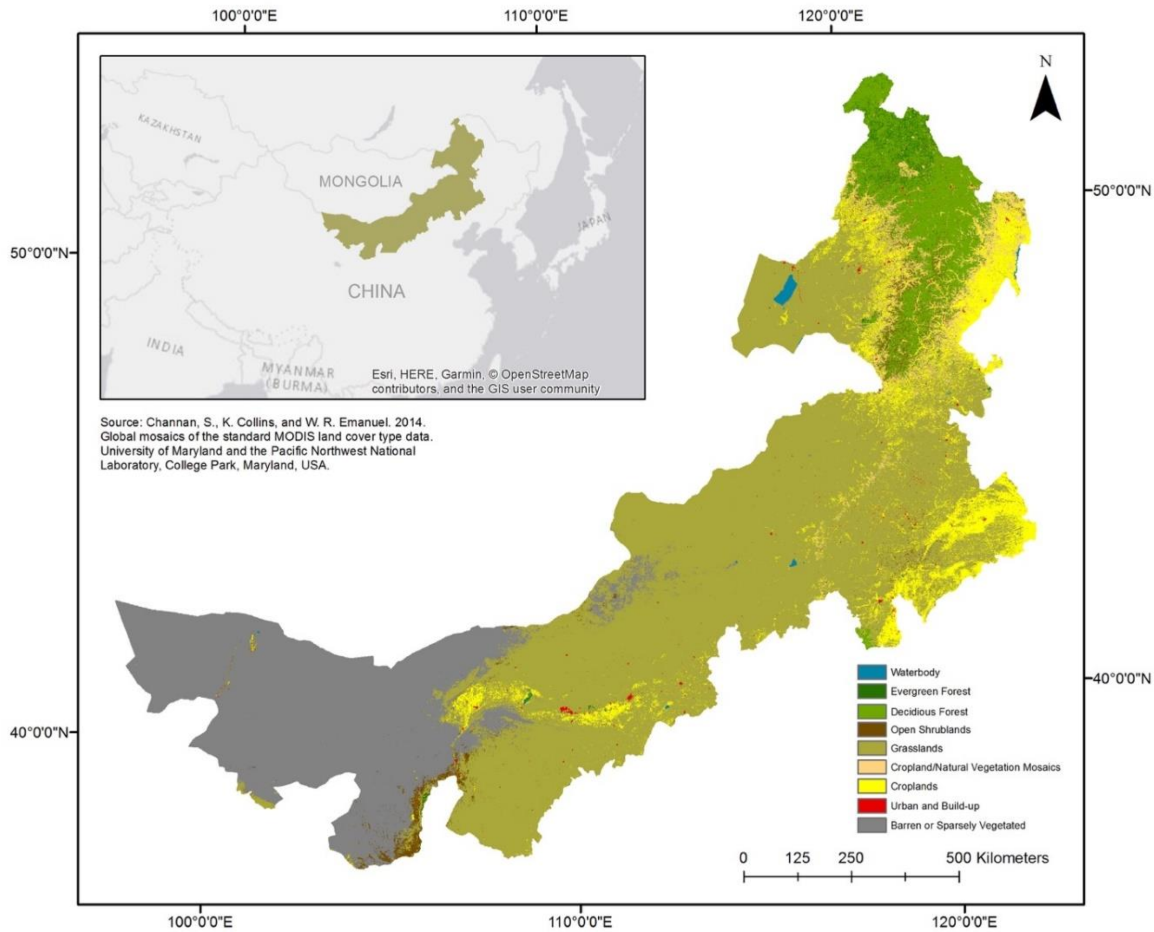


Figure 3. Land use land cover map of Inner Mongolia (Channan, Collins, and Emanuel 2014).

I chose four counties in Xilin Gol (Figure 4) as my study area because a high percentage of rural households in these counties depend on herding for their livelihoods. For East Ujimqin, Sonid Left, Sonid Right, and Xilin Hot county, the proportion of herding output to total gross agricultural output is 92.7%, 82%, 80%, and 78%, respectively (Xilin Gol League Statistical Bureau 2020; 343). As of the end of June of 2019, herders in East Ujimqin and Xilin Hot kept 3 million and 1.2 million head of livestock while herders in Sonid Left and Sonid Right kept 1.1 million and 0.9 million head of livestock (Xilin Gol League Statistical Bureau 2020; 352). Common livestock includes cattle, cows, horses, camels, sheep, and goats. Sheep have a short production cycle which results in a fast return on investment, hence herders often keep large numbers of them. According to a recent household survey (Jimoh et al. 2021), the herd structure of herders usually consists of at least 50% of sheep, followed by goats, and 11% -16% of large ruminants. Selling livestock to slaughterhouses provides herders with the majority of their income, although they also gain a small amount of income from selling wool and cashmere.

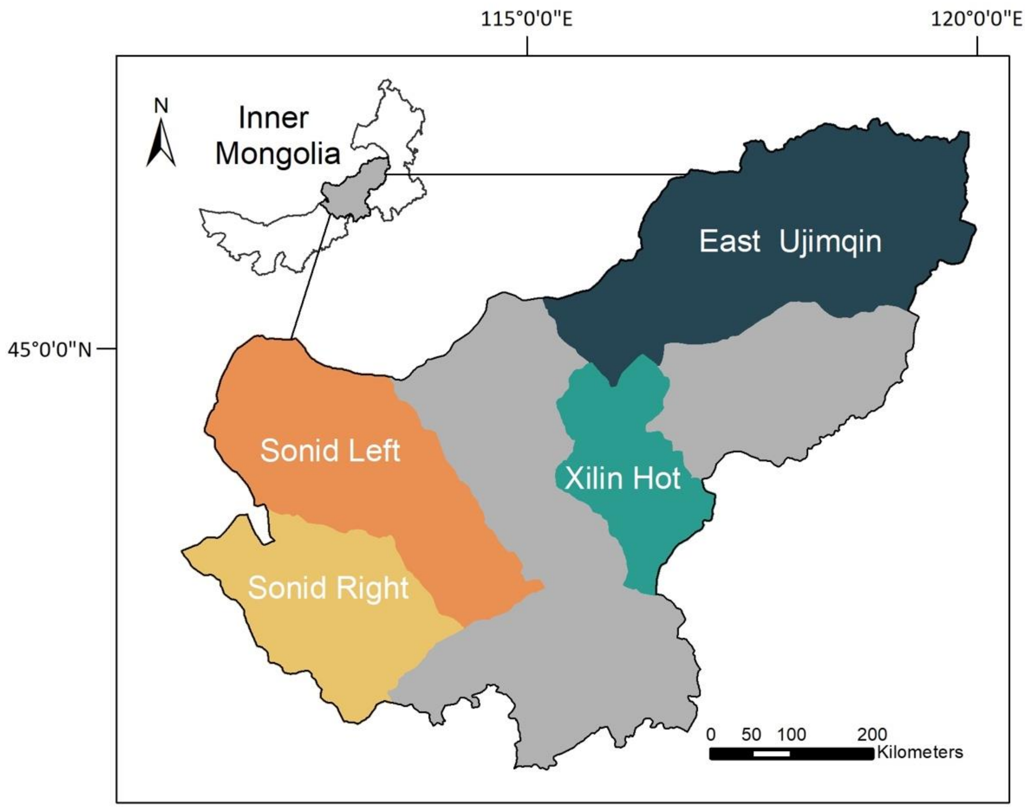


Figure 4. Xilin Gol and the four counties visited in this study.

1.3.2 Payment-based conservation programme in Inner Mongolia

Like other rangeland systems around the globe, the temperate steppes of Inner Mongolia are important carbon sinks. Although rangelands store less carbon per unit compared to forest on average, it still stores up to 34% of the total terrestrial carbon due to the widespread distribution of this ecotype (White, Murray, and Rohweder 2000). Unlike forest systems where the majority of biomass is stored above ground, rangelands retain 90% of the carbon underground as soil organic carbon and below-ground biomass (Bai and Cotrufo 2022). Especially in dry regions, rangelands are also more resilient to today's unpredictable climate compared to forests because rangelands can better withstand extreme events such as drought and heatwaves and recover from them (Dass et al. 2018).

Intensified anthropogenic activities, particularly increased livestock grazing, have resulted in a loss of biodiversity, biomass production, and ecosystem functions and services in Inner Mongolia (Wu et al. 2015; Li et al. 2015). China's rangeland eco-compensation programme policy was developed with the intention of improving rangeland productivity by reducing livestock and thus grazing pressure (Li et al. 2020). It has been in operation since 2011 and is currently starting its third round (2021-2025). This policy provided compensatory payment to 12 million herding households in 8 pastoral provinces every year, and the annual subsidy distribution has increased from 13.6 billion yuan in 2011 to 18.76 billion yuan in 2018 ($2.15 to $2.96 billion USD, respectively) (Li et al. 2020). In 2019, our surveyed households in Inner Mongolia received 45 yuan ($7.1 USD) per hectare per year from the state for limiting their livestock herd to below the state-defined "carrying capacity" of their local natural grassland, or 135 yuan ($21.31 USD) per hectare for retiring from grazing entirely. In other words, herders are provided with payments in place of the revocation of certain land rights—namely limiting their rights of withdrawal of forage by sheep. Payment is made to individual households, but like most compensatory conservation programs, incentive and compensatory pay are offered to the *de jure* instead of the *de facto* landholders. Therefore, despite the policy revoking partial withdrawal rights from the renters, it is the landlords (contract holders who may no longer use the land) who are compensated. In this dissertation, I will examine the effectiveness of an eco-compensation program to influence renters' land use decisions and thus ecosystem services provided in a rental context.

1.3.3 Data collection

Much of the research presented in this dissertation is based on primary data collection in Xilin Gol. I conducted a detailed household survey between June and December of 2019, with support from Dr. Li Ping from the Grassland Research Institute in the Chinese Academy of Agricultural Sciences. I conducted the survey together with Wu Liwen, a graduate student working with Li Ping who was surveying herders on their willingness to adopt new ranching technology. We helped each other on administrative tasks, yet we administered our own surveys. In June, I conducted a detailed pilot survey and semi-structured interview with 20 herders distributed in Sonid Left, Xilin Hot and East Ujimqin. I also consulted local herding cooperative leaders and grassland station officials at this time. Based on the findings of the pilot survey, I spent two weeks revising the survey, interview, and participatory mapping exercises based on the pilot survey results. I then carried out the primary survey with a final sample of 266 households.

The households for this study are selected through stratified random sampling that the Grassland Research Institute has used in previous panel surveys. The four counties were selected because they are ecologically representative of the temperate grasslands and have a high percentage of households that are dependent on herding. Survey data were collected from 18 townships (*sumu*), with households within each township randomly selected. The first chapter is based on basic questions from the survey, drawing from the full sample of 266 households. The second study is based on a discrete choice experiment embedded in the survey, which was given to only a sub-sample of 192 herder households (pilot and pre-test experiments were used to calibrate attributes of the final Bayesian-prior choice experiment). The third study uses digital participatory mapping. As not all land parcel boundaries are clearly recognizable on satellite maps, we identified 214 households from 17 townships to sample, and obtained useful and complete spatial data from 187 of these households.

We drove to each herder's settlement, and I conducted household surveys, choice experiments, and participatory mapping with each participant inside his or her settlement. I conducted the survey in Mandarin. If the respondent only spoke Mongolian, my interpreter—typically the local guide—translated both my questions and the herder's answers. The "Methods" section of each chapter contains detailed descriptions of the research methods used in each study. The research protocol was reviewed and approved by McGill's Research Ethics Board (#37-0619). Informed consent was obtained from each participant of this study.

1.4 Thesis structure

Pastoral Inner Mongolia went through the process of land privatization decades ago. Although, like many rangelands worldwide, land ownership still fundamentally resides with the state (Reid, Fernández-Giménez, and Galvin 2014), Inner Mongolian herders in China have unambiguous private use rights to rangelands. Clear rangeland property rights create suitable conditions for the implementation of incentive-based conservation programmes. As one of the few large-scale incentive-based environmental programmes implemented in rangelands, China's rangeland eco-compensation programme is worth evaluating after a decade of implementation. The result could provide insights and practical lessons for future programmes. Additionally, privatization and formalization of property rights facilitate land transfers and, consequently, the growth of the land rental market. Nevertheless, whether these lands are transferred formally has not been adequately examined. The influence of the rental market on the ongoing incentive policy and herders' resource management practices remains understudied. As both the rental market and incentive-based policy are growing globally (Petrzelka, Ma, and Malin 2013; Salzman et al. 2018), this dissertation aims to understand their interactions and impacts on smallholder households and investigate rangeland management strategies in a dynamic land system.

As a critique of the long-standing eco-centric and production-focused rangeland science, Sayre and others (2012) present a vision of "post-normal" rangeland science in the 21st century, highlighting the variability, heterogeneity, and sometimes irreversibility of rangeland systems. Post-normal rangeland science seeks to accept the complexities of the rangeland socio-ecological systems, analyze the interacting abiotic and biotic factors that drive land change, evaluate various ecosystem services and trade-offs, and incorporate land users' knowledge and management practices in the study design (Sayre et al. 2012). This dissertation builds off this vision and has used key concepts in the development of the three core chapters. In **Chapter 2**, I examine the complexities of land under herders' *de facto* control. More specifically, I quantify herders' use of informal land and purchased forage, two range resources that were often neglected because they are hard to trace and constantly changing. In **Chapter 3**, I measure the different components of land tenure security and the relevant impact of each factor. I elicit herders' preferences through a choice experiment focusing on their perceptions of tenure security. In **Chapter 4**, I delineate individual herder's land boundaries using satellite imagery and acquire the land use and land tenure information through participatory mapping. By integrating the socio-economic data collected in the field and biophysical information retrieved from open-access remote sensing dataset, I build a comprehensive land-use model and quantify the impact of the rental market on rangeland use.

Chapter 2

Complex *de facto* land holding hinders the practicality of payment-based conservation programmes

2.1 Introduction

Market-based conservation mechanisms such as payment for ecosystem services are gaining popularity as an alternative to the traditional top-down, command-and-control environmental policies in incentivizing sustainable land use (Lambin et al. 2014). Ecosystem services are the benefits people obtain from ecosystems, such as water, energy, crop pollination, and nutrient cycling (Millennium Ecosystem Assessment 2005). The service beneficiaries can provide payment for ecosystem services to the service providers to incentivize the latter to maintain, restore or enhance ecosystem services. Increasingly, payment for ecosystem services programmes are implemented globally, focusing on the protection of biodiversity, watershed, and forest (Salzman et al. 2018). The successful implementation of a PES programme is determined by various factors (Pagiola, Arcenas, and Platais 2005). However, one long-recognized prerequisite is secure land tenure, or at the very least, a legally recognized land title so that PES can be paid to individuals or communities (Bremer, Farley, and Lopez-Carr 2014).

Rangelands are increasingly gaining global attention due to the role they play in providing important global ecosystem services (e.g., food provisioning, cultural, and regulating services (Yahdjian, Sala, and Havstad 2015; Bengtsson et al. 2019). Rangelands in Inner Mongolia, China, have undergone dramatic changes in the relationship between formal land title and the implementation of market-based conservation mechanisms over the past several decades. Since 2011, China has implemented a large-scale eco-compensation programme called the "grassland ecological protection reward-subsidy policy" with the goal of halting land degradation and maintaining ecosystem services, mainly through providing incentives to herders to reduce the number of livestock held on a farm – the stocking rate. But how do such policies determine which lands to include and what an appropriate stocking rate should be? Policy-makers and institutions generally only enter into contractual payment or compensation arrangements with households for land over which they have documented management and control rights. Yet, rented out (or rented in) land is invisible to these policies. Further, if a household's herd is distributed between private and rented lands, and sometimes may be fed from purchased fodder grown in distant locations, how can a proper stocking rate be determined?

This chapter seeks to answer the question: are informal land use and reliance on externally purchased forage a problem for rangeland PES policies? To investigate this, we analyze detailed data we gathered from 266 herder households in Xilin Gol, Inner Mongolia on their *de facto* land holdings, livestock, and related rangeland management characteristics. From the survey data we first describe the magnitude of respondents' land held formally and informally and forage purchased from the market – which we translate into 'virtual land' used. To calculate the quantities of virtual land used, we use remote sensing data to approximate a herder's annual parcel-based productivity. Finally, we put these together to review various ways herder stocking rates could be calculated and discuss implications for policy and sustainability more broadly. Our findings suggest current policies that ignore informally-held land miss a large proportion of land used for grazing, and thus produce a narrow and arguably incorrect assessment of stocking rates – which is the primary metric leveraged for monitoring and enforcing grazing policies. Such policies that aim to have large-scale and real on-the-ground impact must resolve these issues.

2.2 Methods

2.2.1 Survey design

A structured questionnaire was administered to collect detailed information on herders' land holdings and land use. Since the initial land allocation to each individual with a village (*gacha*) is consistent, we measured land both in terms of its absolute size and in terms of person. The second measurement, which normalizes land size by the initial land allocation quota, provides an alternative way to contextualize changes in the land size. For instance, the meaning of renting the same 400 hectares of land may be different for a herder with an initial land holding of 100 hectares or 400 hectares. The former herder increased his or her land size by four times (or 400%), whereas the latter herder only increased by 100%.

Our surveys include a variety of information related to the land used by a respondent. Other than the land they contracted from the state, respondents may use land formally contracted by others through rental or other land-sharing arrangements. We asked our respondents about all land being used, including that contracted to them or others. When others' land is being used, we recorded the relationship between the respondent and any other landholders (i.e., whether they are a family member), whether the respondent rents their land, and whether the respondent is permitted unrestricted free use of the rangeland. In addition to land holdings, we recorded each herder's usage of forage that is not produced on their own land, including purchased forage grasses and crops as well as forage consumed by livestock during *otor* (where herders migrate and graze on other people's land). We also conducted semi-structured interviews on selected households and collected demographic and socio-economic information on each household. A total of 266 survey responses were gathered and analyzed in this study.

2.2.2 Estimating forage productivity of individual herder's land

Next, we calculated the amount of land embodied in the herder's external forage inputs. We are interested in knowing how much additional land each household would require if all of the forage they purchased were produced from their own land. Using MODIS data (Figure 5), we estimated the forage productivity from herders' rangeland parcels. Since rangeland productivity varies depending on abiotic conditions and land use intensities, we calculate the productivity of each herder's rangeland instead of using average village productivity.

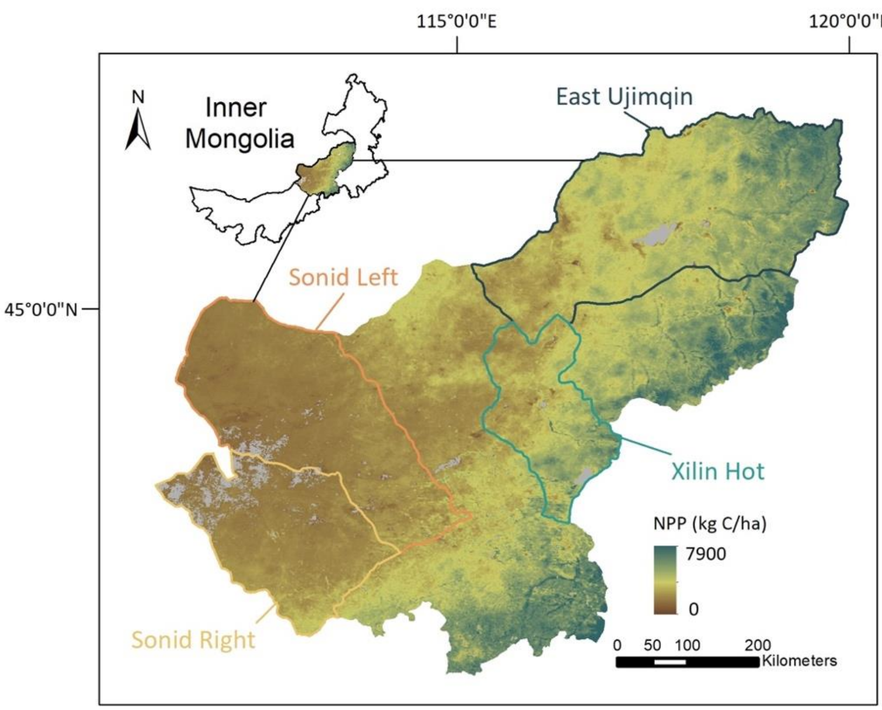


Figure 5. Study area of Xilin Gol. Annual Net primary productivity (NPP) were derived from NASA's MODIS product MOD17A3HGF.

For each surveyed household, we took a GPS point within the household's land using ArcGIS Collector. We used the GPS point's corresponding satellite pixel i to estimate the primary productivity of herders' land. Since free-range sheep graze approximately evenly across a parcel, we anticipate that the satellite-observed primary productivity within one grazing parcel is relatively constant. Therefore, a random point taken within can reasonably represent the overall rangeland quality of that particular land parcel. We calculated the $ANPP$ for pixel i that represents each herder's land as:

$$ANPP_i = (NPP_i * fANPP_i) / carbon\ conversion\ factor$$

where NPP_i is each pixel's Net Primary Production and $fANPP_i$ is the fraction of NPP that is above-ground.

Satellite-based measures allow us to calculate NPP_i, but we must use models or local measurements to estimate the fraction of NPP attributable to ANPP. We derived a location's NPP in 2019 using MODIS/Terra yearly Net Primary Production (MOD17A3HGF) data (Running and Zhao 2015). Unlike forest biomes where up to 80% of NPP can be allocated to ANPP, temperate grasslands allocate most of the biomass below-ground as an evolutionary response to the constant abiotic and biotic disturbances above-ground such as fire or grazing (Ottaviani et al. 2020). The fraction of above-ground NPP to total NPP varies across different grassland types, so parameterization of this ratio with local estimates is critical (Hui and Jackson 2006). Some global scale studies use temperature gradients to differentiate this relationship (Hui and Jackson 2006). However, for our landscape-scale analysis estimating this fraction through local field studies is more accurate. Fan and others (2009) conducted a transect vegetation survey of 48 sample sites in Inner Mongolia and estimated that 8.2%, 13.5%, and 10.2% as of NPP is ANPP for meadow, typical, and desert steppes, respectively. Since MODIS NPP is measured in the unit of carbon content, we use a carbon conversion factor (0.475) to convert ANPP from kilogram carbon to kilogram biomass/dry matter (Raich et al. 1991; Schlesinger and Bernhardt 2020).

ANPP is not forage productivity because only a portion of ANPP can be grazed by herbivores. Therefore, in order to calculate the forage productivity (grazable $ANPP_i$) of each herder's rangeland parcel, we multiplied $ANPP_i$ by a "proper use factor" that varies across different steppe types *j*:

$$Forage\ productivity_i = ANPP_i * Proper\ use\ factor_j$$

A proper use factor is often included in rangeland studies to estimate the amount of grazable biomass (Petz et al. 2014; Millward et al. 2020; Hudson et al. 2021). Other ungrazed parts include trampled plants, invasive or unpalatable species, and biomass that, if grazed, will change plant community composition (Raymond and Miyake 2015). Similar to the fraction of ANPP, the proper use factor differs geographically. We applied a proper use factor of 60%, 50%, and 45% for ANPP estimated on meadow steppe, typical steppe, and desert steppe, respectively (Ministry of Agricultural and Rural Affairs, 2015). Similar studies suggest that the accessibility of biomass is further constrained when the slope exceeds 10 degrees (de Leeuw et al. 2019; Piipponen et al. 2022). Only one of the 266 households is located on a slope greater than 10° (17°) and its proper use factor is further multiplied by 70%, as suggested by de Leeuw and others (2019).

2.2.3 Forage purchased and consumed

We estimated the forage productivity (kg/ ha) of each herder's rangeland parcel using MODIS NPP data. Using this information, we calculated land area needed if the herders' external forage inputs were produced on their land. We followed the global agricultural trade literature (Würtenberger, Koellner, and Binder 2006) and referred to the land area embodied in external forage as the "virtual land" used by a herder. Although it is important to note that, unlike trade literature which often focused on virtual land used in the imported region, we measured the virtual land embodied as if the forage were produced from the herder's own land. Gas and machinery costs used to transport forage are not included in this study.

We further divided virtual land into three categories: forage grasses, forage crops, and forage consumed during the migratory practice "*otor*". This can also be written as:

$$Virtual\ land_{all} = Virtual\ land_{purchased\ forage\ grasses} + Virtual\ land_{purchased\ forage\ crops} + Virtual\ land_{otor}$$

Using MODIS NPP data, we estimated each household's forage grass productivity ($Forage\ Productivity_i$, measured in kilogram per hectare) in 2019 as detailed in section 2.2

above. We then use the amount of forage purchased by each household ($Purchased\ Forage_i$, measured in kilogram and collected in our household survey) to calculate virtual land embodied in purchased forage grasses as:

$$Virtual\ Land_{purchased\ forage\ grasses} =$$
$$Purchased\ Forage_{grasses}/Forage\ production_{i(grasses)}$$

Other than forage grasses, herders also purchased forage crops, including corn stalk silage, dried corn stalks, and corn kernels. We converted the forage crops to forage grasses based on the metabolizable energy each type of forage provides. Calculated in megajoules (MJ) per kilogram, metabolizable energy is a standard measurement of the energy content of forage. Forage grasses in Xilin Gol are mainly composed of needle grass (*Stipa krylovii* and *Stipa grandis*) and Chinese rye grass (*Leymus chinensis*) (Du et al. 2019). The metabolizable energy provided by each kilogram of needle grass and rye grass is 6.07 and 6.02 MJ, respectively, from which we use the mean of 6.045 MJ/kg as the metabolizable energy for forage grasses (Ministry of Agriculture and Rural Affairs 2022). The metabolizable energy for corn stalk silage, corn stalks, and corn kernels are 6.65, 5.72, and 13.04 MJ/kg, respectively (Ministry of Agriculture and Rural Affairs 2022). We converted each type of forage crop into forage grasses by comparing its energy content to that of forage grasses and calculated the virtual land embodied in purchased forage crops as:

$$Virtual\ land_{purchased\ forage\ crops} = (Metabolized\ energy_{forage\ crop}/$$
$$(Metabolized\ energy_{forage\ grasses})*\ amount\ of\ forage\ crop)/$$
$$Forage\ production_{i(grasses)}$$

Other than purchasing forage, Inner Mongolian herders have kept a traditional practice *otor* where during drought, they migrate and graze on another person's land. *Otor* differs from renting land because its duration is much shorter and does not guarantee exclusive land use rights. *Otor* is paid by the amount of forage consumed by different types of livestock. Therefore, knowing that the standard forage consumption for a sheep unit (an adult ewe) is 1.8 kg/day, we calculated the theoretical amount of forage consumed by livestock (thus virtual land used) during each *otor* as:

$$Virtual\ land_{otor} = (Sheep\ unit * 1.8kg\ per\ day * Otor\ days)/$$
$$Forage\ production_{i(grasses)}$$

2.3 Results

We divided all *de facto* land used by a herder household into three categories based on its functionality and on how the contract holder uses the land (Figure 6). The first category is land that belongs to operating landholders. We use "operating landholders" to indicate herders who contracted rangeland from the state in the 1990s and are still actively grazing and managing their contracted land. The second category is land that belongs to "non-operating landholders," which refers to the herders who have contracted land from the state but no longer work the land. "Virtual land" refers to the land size virtually embodied in the external forage that herders used.

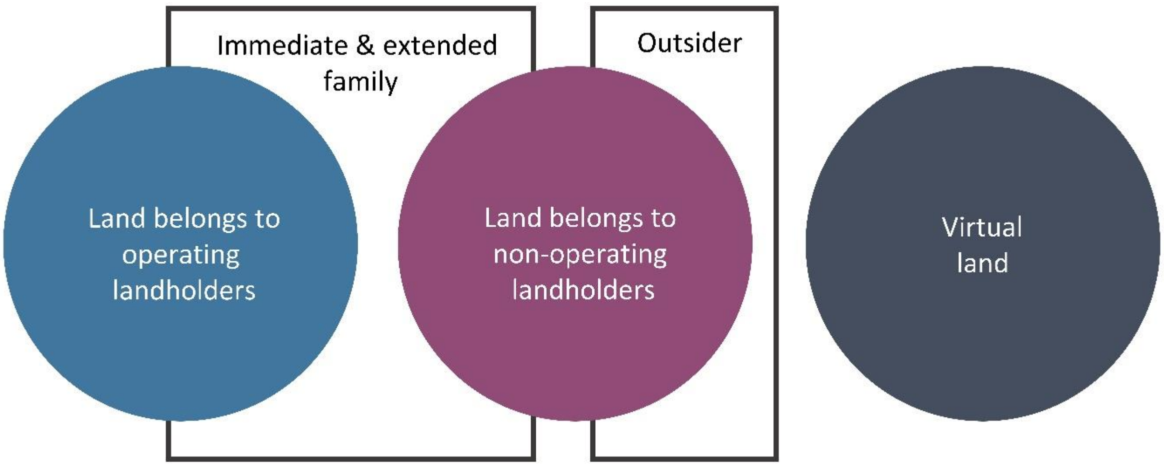


Figure 6. Three different types of land used by an individual household.

2.3.1 Land belongs to operating and non-operating landholders

Operating landholders

In our survey, operating landholders often include our respondent, his or her spouse (if married), and the respondent's (or the couple's) parents. Among the 266 respondents surveyed, 76% (N=202) are using their own contracted land, and 41% (N=110) are using their partner's land. Among the 146 respondents (55% of all households) that use parents' land, 104 of them use land from two parents.

If the land used by our respondent belongs to his or her parents, we categorized it as land from operating landholders since single, young herders often manage their land together with their parents. When a married couple uses their parents' land, the couple often supports their parents financially in return (especially when the respondent is from a multi-sibling family). As a result, the parents form a joint family with our respondent, and they often reserve certain land use and management rights.

The situation is more complicated when it comes to household land pooling. Although legally speaking, each household was given exclusive land use rights and encouraged to manage the land privately, some households still chose to pool their land and graze together into an effective "commons", which has been documented widely in Tibetan communities (Cao et al. 2011; Bai et al. 2021). Interestingly, we also find evidence of common land pooling in Inner Mongolia. Forty-seven households are pooling land with at least one sibling or children that have been married. Additionally, two households are pooling land with a close friend and one with a relative.

According to herders, one of the most common reasons for land pooling is that the amount of land allocated to each household is insufficient. Pooling land together enables livestock to move over a larger area, preventing heavy trampling on one parcel. Some land-pooling households use seasonal rotational grazing so that each parcel is only grazed for a few months every year, which they believe encourages grass recovery. Because the households that pool land also add a proportional amount of livestock, pooling land does not necessarily decrease the stocking rate. A respondent thus appears to have more land due to land pooling, but the stocking rate remains unchanged. Therefore, we exclude pooled land (and the associated livestock) for all respondents who share the land with other households (N=50).

Non-operating landholders

The second category of land belongs to individuals who no longer operate the land. From the perspective of our data, this includes land contracted by outsiders or the respondent's family members and then rented or temporarily occupied by our respondent. There are various reasons why a family member may decide to hand over the land use rights to our respondent. Interviews indicate that the most common reason is that the initial amount of land allocated per person is insufficient for each sibling to support their own family. Some herders then choose to pursue off-farm opportunities and leave the land for the use of one or two siblings. It is also common for a respondent to use the land that belongs to a sister or an aunt who is married to a resident of another village, or the land that belongs to his or her children who live or work in the city.

Non-operating landholders were not part of our sample, but we collected some anecdotal information from the land operator (i.e., our respondent). Many live in nearby cities, primarily due to better access to health care and educational opportunities, and to some extent due to the opportunities for off-farm income, which has also been documented in the literature (Li et al. 2018). Herders have also reported that some of their landlords stop herding as a result of aging, illness, indebtedness, or mismanagement (also noted by Li et al. 2018), and as a result, rental revenue becomes an important income source for the landlords.

A comparison of land size

The initial land allocation quota for the 266 herders ranges from 36.7 to 500 hectares. The median of initial land allocation quota is 133 hectares, and the mean is 152 hectares (Figure 7A). The average amount of land allocated to a qualified desert steppe herder (N=126) is 177 hectares, which is 47 hectares more than the average amount of land (130 hectares) allocated to a typical steppe herder (N =140). It is worth mentioning that the land allocation quota was decided by each village in the 1990s and was used as criteria to contract land to individual households soon after. Eleven herder households in our survey did not receive a land quota in the village where they are currently living, either because they moved in later after the land reform or because they were registered as non-rural residents during land reform which prevented them from contracting land.

The median land size that herders used from operating landholders and non-operating landholders are 272 hectares (μ = 331 ha) and 254 hectares (μ = 318 ha), respectively (Figure 7A). Since the initial land quota varies across villages, we also show land by standardizing the land by the initial land quota (Figure 7B). Herders use a median number of 2 operating landholders' land (μ = 2.6 persons, including the respondents themselves if they received any), and 2 non-operating landholders' land (μ = 2.2 persons). The violin is oak-leaf shaped because the count of the head is usually in integers, with only a few exceptions (e.g., one person's land is allocated to three children, leaving each child with 33% of a single person's land allotment).

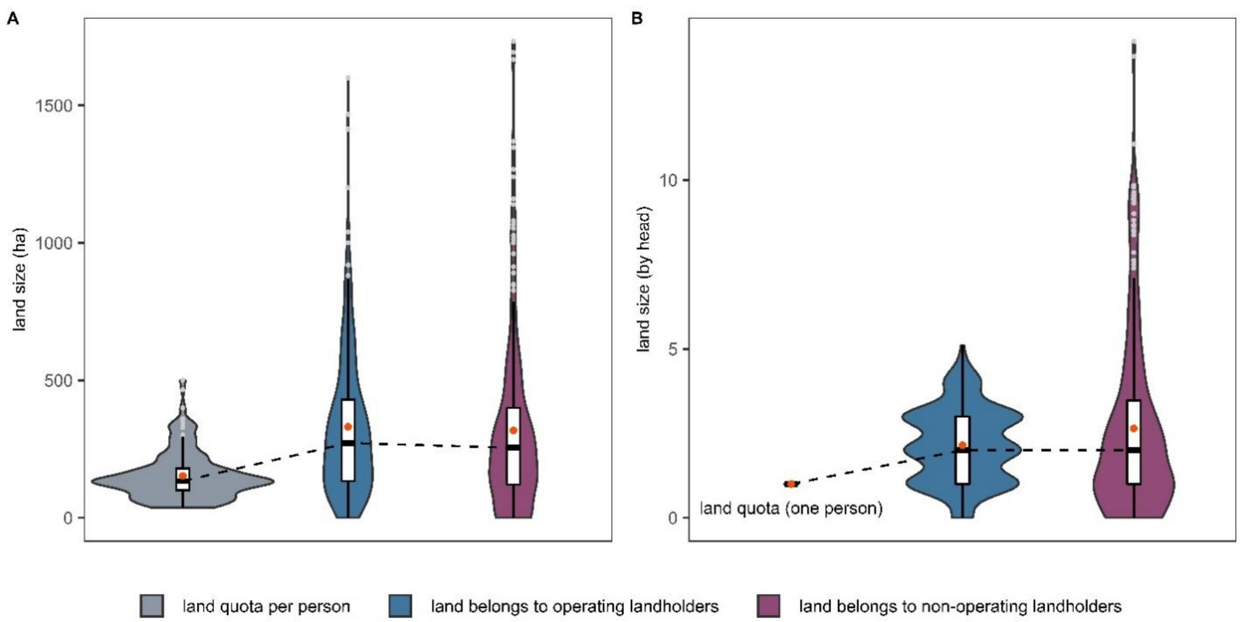


Figure 7. The comparison of land belonging to operating landholders vs. non-operating landholders. (A) shows the distribution of land size in hectares, and (B) shows the distribution of land size by person (i.e., land size normalized by initial land allocation quota). The bold black bar indicates the median, and the red dot indicates the mean of the boxplot.

2.3.2 Virtual land

Household forage production estimated using MODIS NPP data

Corresponding to a decrease in annual precipitation, biomass productivity in Xilin Gol declines from the northeast to the southwest. East Ujimqin and Xilin Hot, two counties in the typical and meadow steppe, produced on average three times as much forage as compared to Sonid Right and Sonid Left, two counties in the desert steppe (Figure 8). Our surveyed households in East Ujimqin and Xilin Hot produced an average of 346 and 323 kg of forage per hectare in 2019. Forage production in these two counties tended to be uniformly distributed, ranging from ~200 kg/ha to ~480kg/ha and dispersed relatively evenly from the means. Surveyed households in Sonid Right and Sonid Left produced an average of 107 and 105 kg of forage per hectare. The range of production is narrower (from ~70 kg/ha to ~140kg/ha) and more observations are centered on the means.

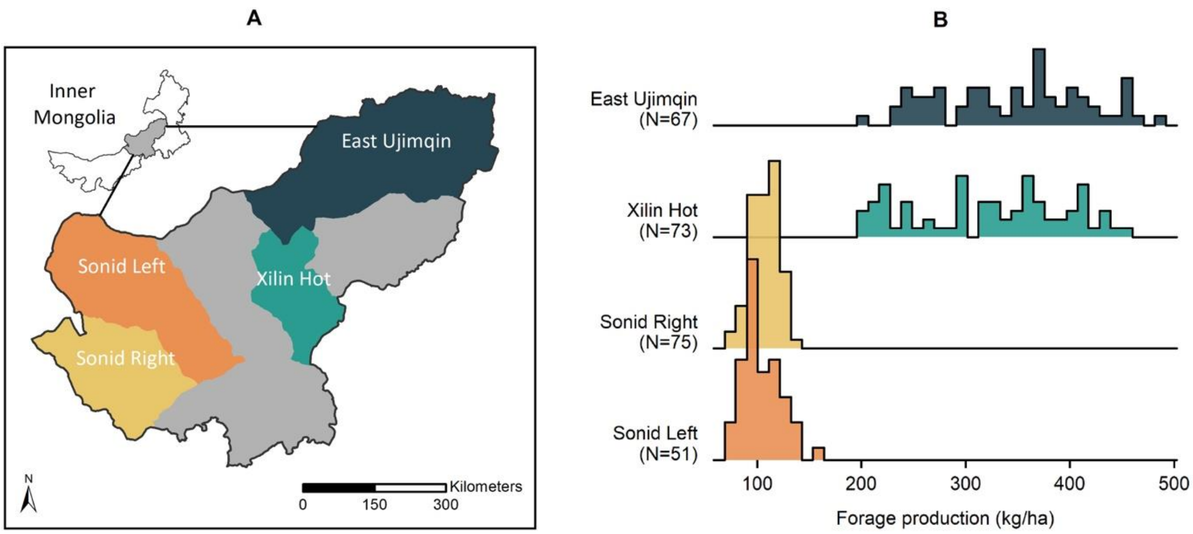


Figure 8. (A) The locations of the four counties in Xilin Gol. (B) Forage production of all surveyed households in each county. Y-axis indicates the counts of the data (i.e., the amount of household within each bin).

The ANPP and forage production estimated from our study are lower than those estimated from ecological surveys conducted inside enclosures. The reason is that lots of ecological studies were conducted in research-based plots where the impact of grazing has been reduced or eliminated. In contrast, all the land parcels we sampled were subject to intensive grazing. Our estimated result is consistent with a previous controlled experiment in Xilin Gol, which demonstrates that ANPP on grazed parcels is approximately 1/3 of the amount measured on ungrazed parcels (Wang et al., 2020).

Virtual land embodied in external forage

The 266 surveyed households use a median of 216 hectares of virtual land (μ =301 ha), which is equivalent to 1.7 person's land (μ =2.5 persons) if normalized by the initial land quota per village (Figure 9). Forage grasses, forage crops, and forage consumed during *otor* account for 48%, 40%, and 12% of the overall virtual land sizes. Among all the households, 83% of them (N=221) purchased forage grasses, and 87% of the households (N=230) purchased forage crops. 15% of the herders (N=39) produced forage on their own land. On average, each household purchased 26.4 tons of forage grasses, 5.5 tons of corn stalk silage, 0.05 tons of corn stalks, and 6.5 tons of corn kernels.

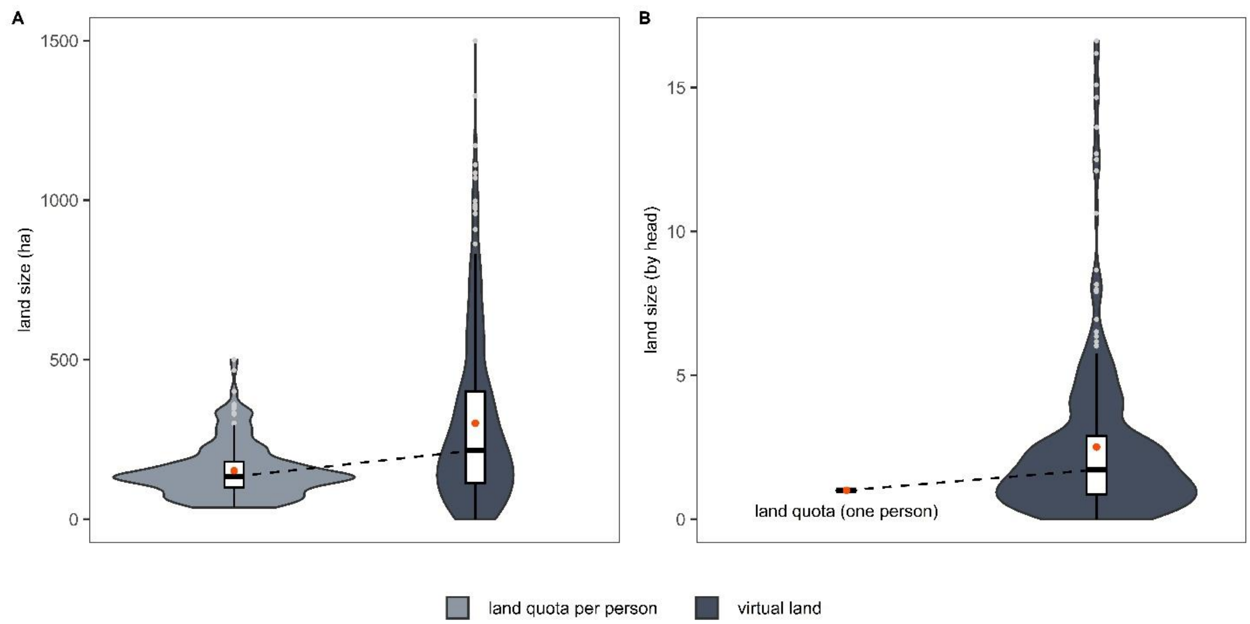


Figure 9. The amount of virtual land used by surveyed herders. (A) shows the distribution of land size in hectares, and (B) shows the distribution of land size by person (i.e., land size normalized by initial land allocation quota). The bold black bar indicates the median, and the red dot indicates the mean of the boxplot.

Contrary to the fact that most herders buy forage from the market, only a small number of herders practice *otor*. In 2019, thirty herders practiced *otor* 34 times. The average *otor* lasts 2.7 months (σ=1.4 months), and 79% of all *otor* (N= 27) started in the summer between May and October. Herders and their livestock travel an average of 57 kilometers (σ= 84 kilometers). Herders usually drive the herd along the highway for a couple of days when the *otor* destination is close. If the destination is further, herders load all livestock onto trucks and transport them there.

The landlords of 16 *otor* are the respondents' relatives (e.g., sister's husband, father-in-law) or close friends. Another 13 *otor* landlords are either villagers or acquaintances. Only the landlords of 5 *otor*, whom the respondents knew through their friends or contacted via WeChat, are total strangers. It is important to emphasize that during *otor*, land use rights is not exclusive. Our respondents sometimes herd together with the landlords on the same parcel of land. Occasionally, respondents may herd the landlord's livestock in exchange for an *otor* fee. A lack of exclusive land use rights is also the reason why *otor* is charged by forage consumption per livestock rather than by the amount of land. On average, a sheep costs 10 yuan (1.5 USD) per month, a cattle costs 140 yuan (20.9 USD), and a horse costs 118 yuan (17.6 USD) per month.

2.3.3 All land

Figure 7 illustrates that the initial land quota's median land size is 133 hectares, and the number increases to 272 hectares, or by 51% when all land belonging to operating landholders is included. If the land belongs to non-operating landholders and virtual land is taken into consideration, the median land size will increase by an additional 67% (from 272 to 815 ha) (Figure 10A). When the size of all land is normalized by the initial land allocation quota for each household, herders use a median of 6 people's land (μ = 7.3) (Figure 10B).

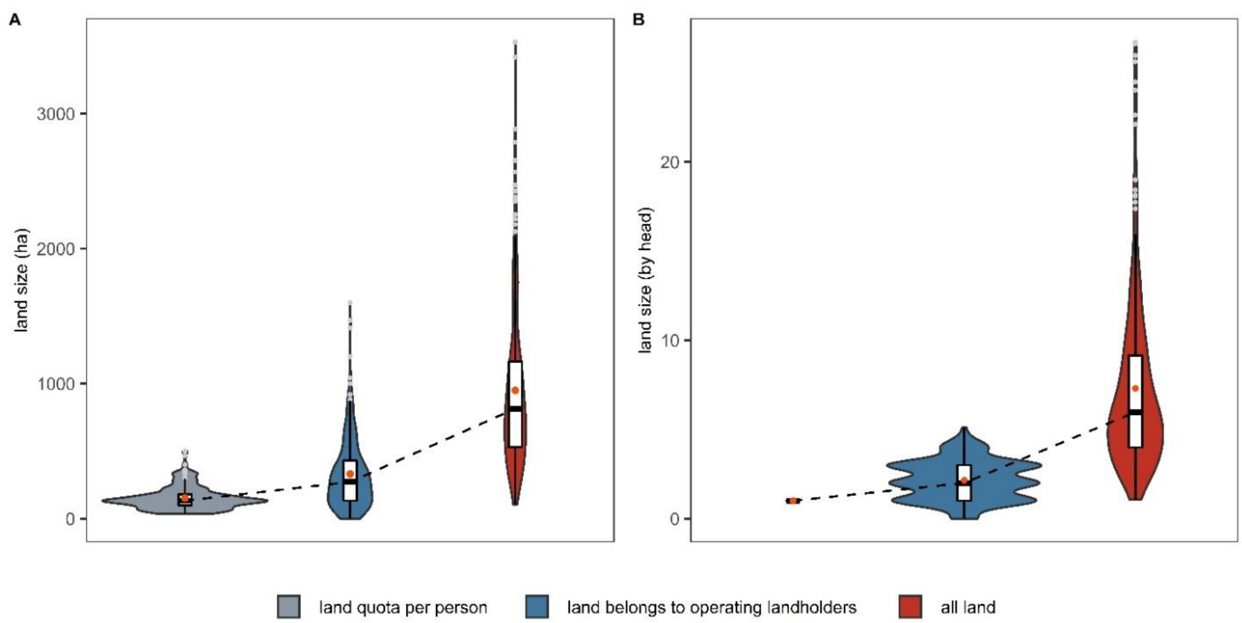


Figure 10. The comparison of land belonging to operating landholders and all land. (A) shows the distribution of land size in hectares, and (B) shows the distribution of land size by person (i.e., land size normalized by initial land allocation quota). The bold black bar indicates the median, and the red dot indicates the mean of the boxplot.

2.3.3 Stocking rate

Depending on how we define "land," the corresponding stocking rate changes significantly. The stocking rate is calculated as the total livestock unit owned by a household divided by the household's land holding. In this study, we compare four different types of stocking rates, which are stocking rates that are 1) estimated and imposed by the government, 2) calculated when the denominator (land) only includes the land that belongs to operating landholders, 3) calculated when the denominator includes the land that belongs to both operating and non-operating landholders, as well as virtual land embodied in external forage, and 4) expected by individual herders (Figure 11).

The official stocking rate (reported by individual herders) ranges from 0.3 to 1.5 sheep/ha, with a median of 0.62 sheep/ha (μ = 0.85 sheep/ha). The official stocking rate is a theoretical metric calculated based on remote sensing data and plot-based biomass measurements (See Discussion section for the detailed process of calculating the official stocking rate). The bimodal data distribution and one-way ANOVA show there is a noticeable difference in the official stocking rate between the two grassland types ($F(1,264) = 623.03$, $p <0.001$). As shown in Figure 8, the desert steppe has lower grass productivity and sustains less livestock per unit of land. The official stocking rate of desert steppe concentrates around the lower peak of the bimodal distribution (range: 0.3- 0.77 sheep/ha). The stocking rate of typical steppe ranges from 0.58 to 1.5 sheep/ha. Although some typical steppe herders have higher land productivity, they are required to have a low stocking rate similar to that of the desert steppe herders, which places certain typical steppe herders in the lower peak of the bimodal distribution. Generally speaking, typical steppe herders are allowed to keep more livestock per hectare.

By contrast, the stocking rate triples if we only define “land” as land that belongs to operating landholders. The median stocking rate increases from 0.62 to 1.84, and the mean stocking rate rises from 0.83 to 2.83. The median stocking rate in this category even exceeds the maximum official stocking rate. The maximum stocking rate in this category has reached an unrealistic 19.51 sheep per ha, with 20 households appearing as outliers. Most outlier households finish their livestock using purchased forage grasses or crops, which is not recognized as “land” in this category.

However, the stocking rate drops again when the land that belongs to non-operating landholders and the virtual lands are included. The median stocking rate returns to 0.66 while the mean restores to 0.74. The positively-skewed, downward-leaning density distribution indicates that a greater percentage of households kept a low stocking rate. Both the range of the distribution

(0.11– 2.22 sheep/ha) and the number of outliers (n= 3) decrease considerably from the previous category.

Finally, we asked individual herders regarding their expected stocking rate. We framed the question as “based on the condition of your rangeland this year, what do you think is a proper stocking rate that can satisfy the needs of your households’ demands and have no adverse impact on the future productivity of your rangeland?” The stocking rate is a sensitive question as it is associated with several environmental policies, fines, and compensations. We placed stocking rate-related questions at the end of the survey in the hopes that respondents would reveal their genuine opinions after our one-hour-long survey and interview. According to the 224 herders who responded to this question, the median expected stocking rate is 1.24 (μ =1.4). Interestingly, the median expected stocking rate is exactly twice as high as the median of the official stocking rate, indicating herders believe they can graze more livestock than the government allows. The influence of land productivity on the expected stocking rate is strong since both typical steppe and desert steppe herders still have a significant difference in their expectations ($F(1, 222) = 47.43$, $p<0.001$).

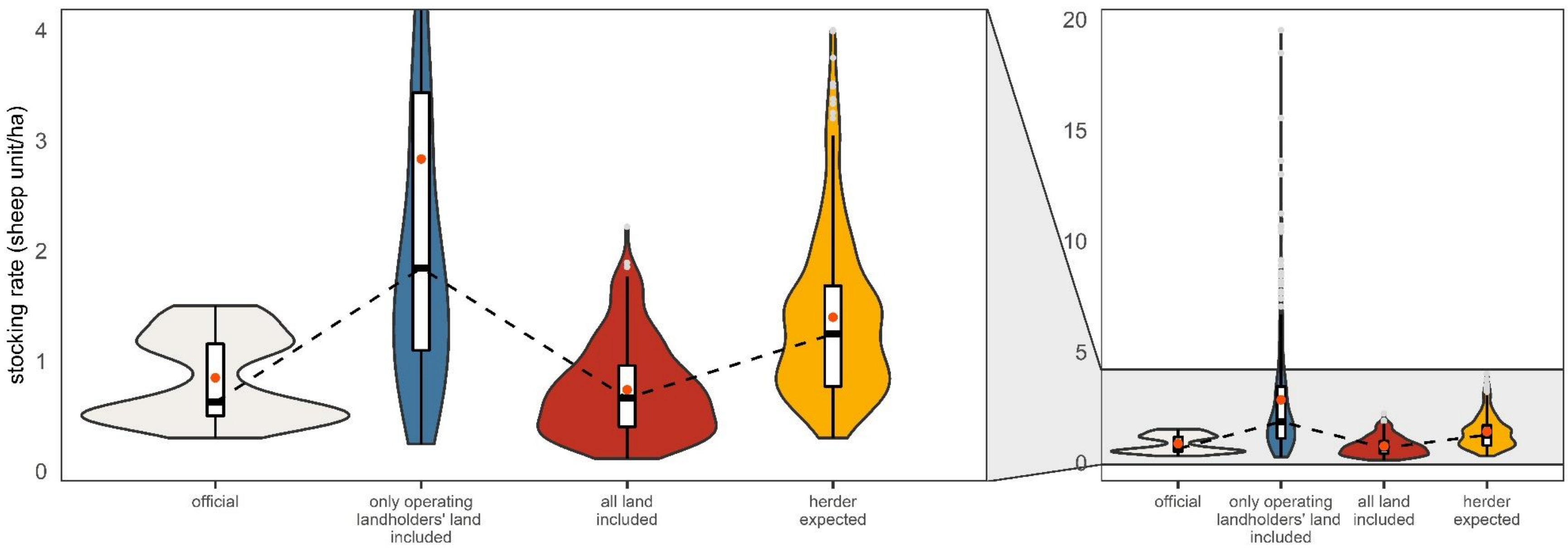


Figure 11. The comparison of four different types of stocking rates. The four violin plots show: 1) the official stocking rate measured and imposed by the government, 2) the stocking rate calculated using only land belonging to operating landholders, 3) the stocking rate calculated using both lands belonging to operating and non-operating landholders as well as virtual land, and 4) stocking rate expected by herders.

2.4 Discussion and conclusion

In this study, we develop a comprehensive survey to estimate all *de facto* land used by a herder household. We found that the surveyed households use a similar amount of land from operating and non-operating landholders Herders also rely on forage that grows elsewhere, including forage purchased and, to a lesser extent, grazed by their livestock during *otor*. Herders' overall land size increases if the amount of external forage used is converted into the size of the land. Additionally, the stocking rate changes considerably depending on who measures it and how "land" is defined. Herders' expected stocking rate is much higher than the official stocking rate. The stocking rate also changes substantially when non-operating landholders' land and virtual land are taken into account. Our study has the following key implications.

2.4.1 A non-negligible amount of rangeland cannot be covered by eco-compensation

Our study shows that a herder household uses a considerable amount of land legally registered under the name of other non-operating landholders. On average, a herder uses 331 hectares (or 2.6 person equivalent) of land from operating landholders and 318 hectares (or 2.2 person equivalent) from non-operating landholders. Using land from non-operating herders or farmers has become a common phenomenon worldwide (Petrzelka, Ma, and Malin 2013). Approximately 40% of the farmlands in the continental United States is rented, and 80% is rented from non-operating landowners (Bigelow, Borchers, and Hubbs 2016; Ranjan et al. 2019). Recent studies are trying to determine the obstacles and potential opportunities to motivate non-operating landowners and their tenants to adopt sustainable land practices (Ranjan et al. 2019; Dell 2019).

The biggest problem with Inner Mongolia's growing number of non-operating landholders is the resulting outflow of eco-compensation. The state is implementing the third round of eco-compensation which has distributed approximately 18.76 billion yuan (2.96 billion USD) to individual herder households each year of the past decade (Li et al. 2020). The privatization of land use rights in Inner Mongolia has facilitated the effective distribution of eco-compensation. To prevent misappropriation from the local government, eco-compensation in Inner Mongolia is deposited directly into the individual herder's bank account based on the amount of *de jure* land they have contracted from the state (Li et al. 2018). As a result, non-operating landholders receive eco-compensation even though they do not work the land. Contrarily, land operators can only be compensated for the land they have *de jure* control over, which may only be a portion of the total amount of land they have *de facto* control over. Therefore, the growth of the land rental market will unavoidably accelerate the outflow of eco-compensation, as long as the *de jure* landholders retain the rights to receive it.

2.4.2 Land-based environmental policy ignores external forage inputs

Our study shows that purchasing forage has become a common practice. On average, a herder uses 38.4 tons of external forage grasses and forage crops, which can be converted to 301 hectares (or 2.6 person equivalent) of land, assuming all were produced from the herder's own land. Recent studies in Inner Mongolia have also shown that the purchase of forage is an important source of household expenditure and one of the primary reasons for the herders to obtain loans (Wang, Brown, and Agrawal 2013; Lu, Huntsinger, and Li 2022).

Herders use external forage to compensate for biomass deficiency caused by severe drought or blizzards (Briske et al. 2015). Episodic weather events have a critical influence on rangeland dynamics and herder livelihoods. Consecutive years of drought or harsh winter blizzards significantly reduce biomass availability and accessibility, cause weight loss and mortality of livestock, and threaten herder livelihoods (Ellis and Swift 1988). Herders on the Mongolian steppe usually cope with such natural hazards by migrating to a relatively resource-abundant area or by growing, reserving, or purchasing forage ahead of time (Fernández-Giménez et al. 2015). For this reason, pastoral land tenure is often described as "paradoxical" as herders need secure access to critical resources, while flexible access to emergency resources to deal with unpredictable social and climatic disturbances (Fernandez-Gimenez 2002). However, mobility is considerably restrained in Inner Mongolia due to the privatization of land use rights. Our surveys show that traditional migratory practices such as *otor* still exist, but the scale (in terms of frequency) is smaller than forage purchases. Our analysis shows that *otor* is usually practiced among acquaintances, friends, and relatives, sometimes for a lower "friend price" and occasionally without pay. Such characteristics suggest that *otor* may still have its reciprocal components (Xie and Li 2008), which raises the entry barriers for practicing *otor* compared to purchasing forage. Improved transportation and information exchange have given herders convenient access to forage, which were often delivered to their homes. Therefore, the storage of purchased forage has replaced mobility and has become the primary coping mechanism in response to the unpredictable environment (Wang, Brown, and Agrawal 2013). From another perspective, land privatization and infrastructure improvement have resulted in a novel type of mobility: fewer sedentarized herders migrate to find forage; instead, forage travels in response to herders' needs.

To our knowledge, no previous literature or policies have considered forage as a form of land in their calculations. Such negligence in smallholder intensification and market integration is not uncommon. Zimmerer, Lambin, and Vanek (2018) point out that smallholders worldwide were frequently stereotyped as a subsistence-based and autarkic population that is either isolated from or extremely vulnerable to the external economic environment. Herders' willingness, capacity, and potential to integrate into the larger market and adopt sustainable land management are frequently underestimated. In practice, our study demonstrates that such oversight drives unfair land policies. We found that external forage inputs contribute to the herders' overall resource budget but are not reflected in the rigid, land-based stocking rate calculation. The omission of purchased forage unavoidably inflated the stocking rate and left a handful of herders who finish their livestock with grain or grasses (therefore relying more heavily on external forage than other herders) with a baffling unrealistic stocking rate (see outliers in Figure 11).

2.4.3 Land, a critical multiplier for ecological metrics, is oversimplified

Our result shows stocking rate changes significantly depending on how we define land. Since a non-negligible amount of herders' land comes from non-operating landholders and external forage supply, we argue that the stocking rate calculated only based on operating landholders' land is excessively stringent for herders. The stocking rate is expressed as the proportion of livestock to land (i.e., ${}^{livestock}/_{land}$). If herders' land (the denominator of the fraction) is consistently measured less than its actual size, then the stocking rate (the value of the fraction) will be inevitably overstated. In order to estimate an actual stocking rate that is fair, it appears that all *de facto* (informally managed and virtually used) land must be included and recorded as we did in this study. But the main question is: is it really possible to document all *de facto* land?

We argue that the administrative complexities involved in calculating all *de facto* land limit the applicability of the stocking rate. Scientists may quickly estimate all *de facto* land used by a household through a household survey (assuming the right questions are asked and herders choose to answer them) because scientists, including the authors of this study, do not need to validate the numbers reported by the respondent. Local supervisory officials face greater difficulties. Because the stocking rate is closely associated with government-issued penalties and eco-compensation, herders must provide proof of the amount of land used in order to estimate an equitable stocking rate. However, this is practically difficult because only 29% of the rental contracts currently made by herders are formal contracts (see Chapter 3 for details). Additionally, lots of land transfers within a family are made informally and without payment. To estimate all *de facto* land accurately, officials would have to verify all informal land transactions made by herders and document their forage purchases, which is both fiscally and practically unrealistic.

The stocking rate, or the number of livestock per unit of land, appears to be an easily measurable metric. However, as the critical multiplier that gives this metric meaning at the property level, *de facto* land is challenging to quantify. Paradoxically, without validating the exact amount of land used by a herder, the official stocking rate as a metric remains conceptually appealing but empirically irrelevant.

2.4.4 Conclusion

Our study shows that the *de facto* land holding of herders is complex. Depending on how it is defined and whether it is calculated correctly, the *de facto* land holding significantly impacts the calculation of the stocking rate. In order to estimate the overall grazing pressure of the individual household, the official or theoretical stocking rate must be multiplied by a number that accurately reflects the size of all land under the household’s *de facto* control. Our study demonstrates that herders are using a large number of lands from non-operating landholders and managing many of them informally. Due to the informal characteristics of local land transfers, we argue that the fiscal and administrative burden of recording and validating all informally managed land may outweigh the benefit.

Paradoxically, without validating all *de facto* land used by the household, the theoretical stocking rate at a property level remains impractical. Moreover, the theoretical stocking rate is counterproductive as operating herders can only receive partial compensation for the total amount of land they are using. At the same time, their carrying capacity is calculated based on a stringent standard that only includes their formally-managed land. Our study highlights the importance of *de facto* land as a multiplier of (any) policy-evaluation metrics. We hope that by evaluating the current challenges, our study will spark a discussion about the development of new household (property)-level applicable metrics.

Connecting statement

Through analyzing household survey data, Chapter 2 shows that herders use a non-negligible amount of land they only have *de facto* control over. As Inner Mongolia's rental market continues to grow, we anticipate that there will be more non-operating landholders and land under rental contracts. It remains understudied, though, if rental tenure is perceived as insecure by herders. How policy can help make Inner Mongolia herders feel more secure in the current setting is also not fully understood. In chapter 3, I use a choice experiment to elicit herders' preferences. I identify the underlying factors that threaten tenure insecurity and evaluate the relative impact of each factor.

Chapter 3

Land tenure insecurity determinants and trade-offs: A choice experiment

3.1 Introduction

Land is the fundamental resource from which most livelihoods are supported. Secure land tenure and land rights are crucial to ensure food security, sustain rural livelihoods, and enhance sustainable resource governance. Land tenure security is a composite idea, with various determinants and components that may influence land users' perception of tenure security and their land management decisions (van Gelder 2010). Tenure security is often defined as the security of the content and the assurance of rights (Sjaastad and Bromley 2000; Arnot, Luckert, and Boxall 2011). The *content* refers to the bundle of rights included and their duration. The *assurance* of rights refers to how certain an individual is that their land rights will be protected. Whether land users feel secure may be impacted by how complete the bundle of rights is and how likely it is that such rights will be protected. Furthermore, the presence of formal and informal institutions at the local level, as well as changes in the broader-level political and economic environment, all influence whether land users feel secure (Robinson, Li, and Hou 2017). Measuring the various factors that influence tenure insecurity is thus critical for developing better land tenure policies. Previous literature focuses on investigating one or several variables (e.g., *de jure* tenure recognition, conflict) and their impacts (Linkow 2016; Lawry et al. 2017), yet there is a lack of quantitative comparison on the relative impact of each factor.

"Apples-to-apples" comparisons of factors that impact tenure insecurity are hampered by the fact that components are often measured on different scales. Depending on the nature of the question, the measurement scales of tenure security can fall into three groups (Figure 12A): categorical, continuous, or binary variables (Tseng et al. 2021). Categorical variables such as a Likert-scale measurement can be used to capture levels of perception (PRIndex 2018). Categorical variables are most appropriate to classify various tenure forms, land rights, or property ownership (Kabubo-Mariara 2007; Leonhardt, Penker, and Salhofer 2019). Continuous variables can compare the occurrences or amounts of certain activities that undermine or strengthen tenure security, such as incidents of land reallocation (Deininger and Jin 2009). Binary variables can be used to indicate the presence/absence of activities like land disputes, expropriations, conflicts, or tenure intervention programmes that offer formal certificates (Holden and Ghebru 2016). Due to the incompatibility of the measurement scales, existing tenure security studies only compare the absolute impacts of one (or a set of) variable(s) between several population groups (Figure 12A), rather than comparing the relative importance of each variable within a population (Figure 12B). This also means that tenure security measurements are often variable-based rather than household-oriented. For example, by comparing two villages' ratings on the "likelihood of eviction" variable, we can estimate whether households in village A perceive themselves to be more at risk than those in village B in aggregate. However, the lack of a standardized measurement scale makes it impossible to compare the importance of the "likelihood of eviction" to another variable (e.g., the lack of land certificate) or to identify from a set of variables the ones that threaten villagers the most.

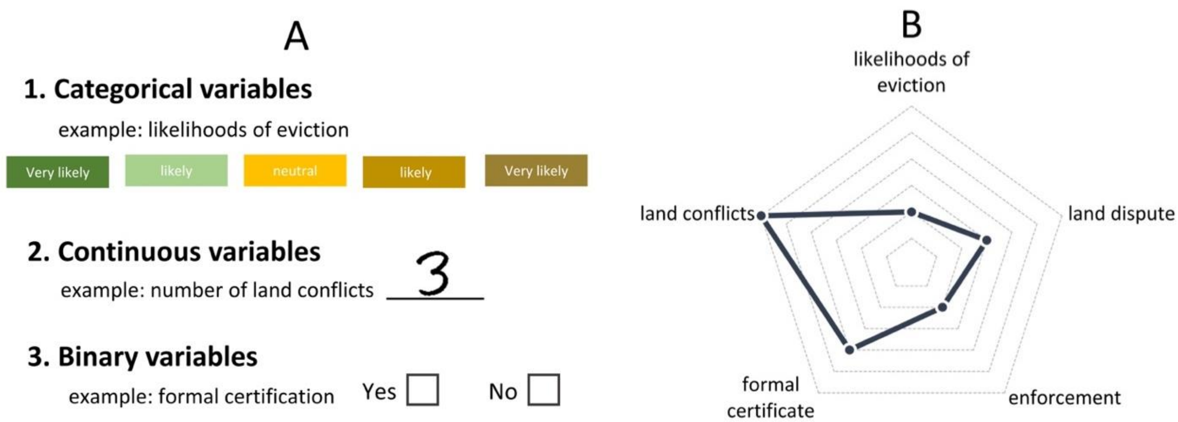


Figure 12. (A) Common measurement scales: categorical, continuous, and binary. Such measurement is variable-based and allows for the direct comparison of one or several variables among various groups. (B) The ideal measurement that cross-examinates the relative importance of each variable within a population. Such measurement focuses on households and can identify the most influential factors.

Since there are many factors that affect land tenure security, it is crucial to compare each factor's relative importance rather than assuming all factors have potentially equal significance. Whether land users perceive their tenure as secure can be influenced by their socio-economic status, history, political unrest, economic crisis, and other issues, which vary according to the local context. Because of these contextual differences, households in one village may try to avoid factor A, while those in the neighbouring village may attempt to avoid factor B. Therefore, not all factors influencing tenure security should be given identical weight. As van Gelder (2010) argues, the future research direction is not on making a comprehensive list of variables that influence tenure security, but rather on clarifying "how much specific factors actually contribute to [tenure security] and which [are] most prominent".

To address this research gap, we used a choice experiment to weigh the different factors that impact tenure insecurity as perceived by herders in Inner Mongolia, China. A choice experiment is a survey-based technique to elicit respondents' preferences. We presented 192 respondents with pairs of hypothetical alternatives comprised of a set of features (characteristics) at various levels (range/degrees). By analyzing the respondents' decisions over these sets of hypothetical alternatives, we assessed the relative importance of each factor that influences tenure security. Since choice experiment can measure participants' trade-offs among hypothetical attributes, it has been used as an *ex-ante* tool to evaluate the feasibility of potential environmental programs (Cranford and Mourato 2014; Rakotonarivo et al. 2017). Similarly, our study provided policy recommendations for land contract registration and payment-based conservation programs.

3.2 Methods

3.2.1 Rental context

Our study area is representative of numerous pastoral areas with emerging rental markets. Historically, pastoralism has been associated with informal institutions, moral economies, and common property regimes. However, privatization and commodification of pastoral resources and lands have become more common due to the changes in global pastoral land tenure, economy, and demography (Yeh and Gaerrang 2011; Korf, Hagmann, and Emmenegger 2015). Previous studies in Inner Mongolia have found that the increase in livestock holding, the intensified production methods, and a new generation of herders that have little to no initial land endowment all contribute to the rising demand for rangeland (Tan et al. 2017; Qiao et al. 2018; Jimoh et al. 2021). The demand for rangeland is reflected by rising land values in our study area. For example, the rangeland price in East Ujimqin has increased by a stunning 1100% from a symbolic compensation of \$150/km^2 in 2000 to an average of \$1800/km^2 in 2014 (Sarigai and Zhang 2015). Little academic attention has been paid to how the rental market functions in non-agroecosystems. It remains unclear what form of contract agreement -formal or informal- is perceived by herders as more secure, and what attributes of land contracts concern herders most. This study explores these questions.

3.2.2 Attributes and levels

Each herder in our study received five choice sets – a hypothetical situation in which they choose among two land rental contracts that have various combinations of attributes, and one "opt-out" option signifying they do not like either of the configured options. The two rental contracts were composed of five attributes at different levels. To determine attributes and attribute 'levels' used in the choice experiment, we drew on existing literature and the authors' (Luci Lu and Li Ping's) years of fieldwork experience in Inner Mongolia. We piloted our choice experiments with 24 herders.

We also aimed to conduct semi-structured interviews alongside the choice experiments about potential tenure insecurity factors. To develop the semi-structured questionnaire, we piloted a questionnaire with approximately 20 herders across three banners in Xilin Gol and consulted local grassland bureau officials and herders' cooperatives leaders, from which we modified the semi-structured interview questions.

We used these 24 pilot choice experiments and semi-structured interviews to ensure herders would find questions and attribute levels realistic. After all attributes and parameters were calibrated, a randomized pilot choice experiment was given to 12 herders, and the resulting priors were used to develop a Bayesian-prior experiment that was distributed to 192 herders in Xilin Gol (see section 3.2.3 choice card design for details).

During the pilot and experiment stages, the herders were interviewed individually instead of through a focus group interview since it is often hard to discuss sensitive issues in focus groups and to elicit individuals to share detailed personal experiences (Secor 2010). Some land-tenure-related topics, such as land appropriation by cadres or neighbours, intra-household land disputes, or herders' attitudes towards eco-compensation and grassland law enforcement may be too sensitive or inappropriate for focus group discussions. A face-to-face survey conducted in a safe environment significantly improved respondents' comfort and anonymity, therefore enhancing their willingness to discuss nuanced land-related issues.

The five key attributes we used for the choice experiment are 1) *contract duration*, 2) whether it is a *kinship* contract and a *formal contract*, 3) whether there is a *10% probability of early contract termination*, 4) whether the herder has *the rights to receive eco-compensation*, and 5) the *price* a herder is willing to pay for the land (Table 2). This set of attributes also distinguishes the uncertainties associated with the *content* of rights (i.e., duration, formalization, additional rights to receive eco-compensation) and the *assurance* of rights (i.e., probability of contract termination).

Table 2. Attributes and levels.

<table>
<tr><th>Attributes</th><th>Description</th><th>Levels</th></tr>
<tr><td>Duration</td><td>duration of the land contract</td><td>1, 3, 5, 10 years (continuous)</td></tr>
<tr><td rowspan="2">Formal contract
&
Kinship contract</td><td>1. personal (informal) contract
2. formal contract</td><td rowspan="2">1a. non-kin villager + informal contract
1b. kin + informal contract
2a. kin + formal contract
2b. non-kin villager + formal contact (categorical)</td></tr>
<tr><td>a. non-kin contract (with fellow villager)
b. kinship contract</td></tr>
<tr><td>Risk of early termination</td><td>10% probability of early termination</td><td>No; Yes (binary)</td></tr>
<tr><td>Additional land rights</td><td>rights to receive eco-compensation</td><td>No; Yes (binary)</td></tr>
<tr><td>Rental price</td><td>payment for the contract (yuan/mu*)</td><td>-10%, -2%, 0% (status quo), +2%, and +10% of the herders’ expected land price (continuous)</td></tr>
</table>

* 1 mu = 1/15 ha

Contract duration: This attribute indicates how long the herders can use the land and examines whether they prefer a long-term contract versus a short-term one. The levels are set as 1, 3, 5, and 10 years. Longer land rights are often believed to have a positive impact on farmers’ long-term farm investments (Bandiera 2007; Lovo 2016). It is reasonable to hypothesize that with the introduction of hybrid livestock breeds and the improved access to the infrastructure, livestock market, and external forage supply (Li and Li 2012), the herders may opt for a long contract aiming to gradually intensify production. Dryland herders, however, are also vulnerable to unpredictable drought, and when it happens, the herders often need immediate access to resources (Ellis and Swift 1988; Fernandez-Gimenez 2002). As a result, it is also possible that the herders may try to avoid risks associated with severe droughts and winter blizzards by renting-in lands opportunistically and, therefore, may favour flexible, short-term contracts.

The herders informed us during the pilot survey that the types of payment (i.e., a lump-sum or yearly payment) have a direct influence on how long they will rent. Land rental, in recent years often paid in lump-sum, represents a considerable investment and expense for herding households. Herders argue that a lump-sum payment over five or ten years would make the levels unrealistic. To eliminate the extra burden of the lump-sum payment, we highlighted in the choice scenario that "all rents are paid yearly".

Formal contract & kinship contract: This composite attribute is made up of two components representing formal and informal institutions. The first component, *formal contract*, examines whether and to what extent the herders value formal land agreements (provided by the grassland supervision bureau) over informal personal agreements. The second component, *kinship contract*, examines whether the herders believe it to be safer to rent from people with whom they have kinship ties. Kinship contracts are common both in pastoral (Qiao et al. 2018) and farmland rental transactions (Ma et al. 2015; Gebru, Holden, and Tilahun 2019). Previous literature shows that kinship trust may reduce transaction costs and possibly bargaining and monitoring costs, which would consequently enhance tenure security (Holden and Ghebru 2006; Ma et al. 2015). The interaction of these two attributes creates the following four hypothetical levels, giving the respondent the choice to make: 1) an informal contract with a non-kin villager, 2) an informal contract with kin, 3) a formal contract with a non-kin villager, and 4) a formal contract with kin. Since the term "informal" often has a negative connotation, in the local context can imply illegality and a contract that should be avoided; we use "personal contract" in the choice cards.

10% probability of early termination: This attribute indicates whether the land contract will be terminated prematurely. We used two levels for this attribute: 1) the landlord will not terminate the contract early, or 2) the landlord has a 10% chance of ending the contract early.

Rights to receive eco-compensation: Current eco-compensation pays directly to *de jure* landholders instead of the *de facto* land using renters. This attribute measures how important it is for the herders to have the rights to receive eco-compensation. This attribute has two levels: for the rented land, 1) the herder is not eligible for eco-compensation, or 2) the herder has the rights to receive *some* eco-compensation.

Rental price: The price attribute evaluates the herders' willingness to pay for a contract. When setting the levels of the price attribute, it is common to use absolute values that deviate from the status quo (e.g., if the status quo is $300 per hectare, the levels can be $250 and $350). Using absolute value, however, causes problems for our study since the price for rangeland varies across counties and within each county. As a result, the price range that is deemed to be acceptable for one herder may be unsuitable (too high or too low) for another. Previous literature suggests that rental price varies based on the type of grassland and is higher on productive steppes than on unproductive steppes (Qiao et al. 2018). This difference in land productivity may explain the price variation across the counties. The price variation within a county, however, is often explained by a herders' personal needs and management plans. Herders who are market-oriented, rear cattle, or aim to enlarge production will pay more for highly productive land, while those who are subsistence-oriented, raise camels, or have extra reserved forage will be satisfied with sparsely vegetated land at a lower cost.

Due to the variation of land price across counties and within a county, we forwent using the absolute change in land price as levels. Instead, we used the relative change from each herder's expected price as levels. We asked the respondents to picture a scenario in which they needed to rent a parcel of land nearby and to list the price in yuan per mu (1 ha equals 15 mu) that they would be willing to pay to rent that specific land parcel (and its rangeland quality). The herders' expected price was used as a benchmark and we chose the deviations from the herders' expectations as levels. For example, if herder A's expected price is 10 yuan/mu, then 10% less than the expectation is 9 yuan/mu. The five levels were set as 1) 10% below expectation, 2) 2% below expectation, 3) equal to expectation, 4) 2% above expectation, and 5) 10% above expectation. In order to make the choice cards easier for herders to comprehend, we converted all relative levels back to absolute prices based on each herder's expected land price reported during the survey. We instantly calculated the absolute price and filled out the choice card before handing it to each herder. Since hay harvesting land is managed and priced differently, we limited our scenario to renting grazing land only.

3.2.3 Choice card design

We expected a small sample size in this study since herder settlements are dispersed, so we used a D-efficiency design to generate the choice sets since it requires a smaller sample size compared to an orthogonal design (Scarpa and Rose 2008). All choice sets were generated from Ngene, a stated choice experiment design software (Choice Metrics 2018). Based on the semi-structured interviews and the 24 "testing" choice experiments, we calibrated the parameters of attributes and levels to make sure they are realistic to herders. Additionally, we further estimated the sign of each parameter for each attribute and level in order to design a D_{prior}- efficient pilot survey. For example, since the herders reported that they would avoid contracts with a 10% chance of early termination, the attribute was assigned a -0.00001 prior parameter since it is anticipated to have a negative impact on utility (Choice Metrics 2018, see Appendix I for details). Previous studies also determine whether a parameter sign is positive or negative through focus group discussions and expert knowledge (Geussens et al. 2019; Rai, Bhattarai, and Neupane 2019). We designed a D_{prior}- efficient pilot experiment based on the signs of parameters and distributed it to 12 herders. Based on the result of this pilot experiment, we estimated the Bayesian prior parameter and generated a $D_{Bayesian}$- efficient choice experiment, which we then gave to 192 herders (see Appendix I for code). A $D_{Bayesian}$- efficient design uses the distribution of the priors instead of a fixed parameter. We generated 20 different choice sets, which were divided into four blocks (five choice sets per block) for the pilot D_{prior}- efficient experiment and the final $D_{Bayesian}$- efficient experiment. We gave each herder five choice sets to balance sampling for variation within individuals and to reduce survey fatigue.

Each herder was surveyed individually in their settlement and was randomly assigned with one of the four blocks of choice sets. The order of the five choice sets in each block was randomized. Each choice set contained three alternatives representing land contracts A and B, along with an opt-out option (Figure 13). The herders were asked to choose the preferred contract option or to decide whether both alternatives A and B were unsuitable (i.e., the opt-out option). The opt-out option was designed to avoid forcing the herders to make decisions between A and B.

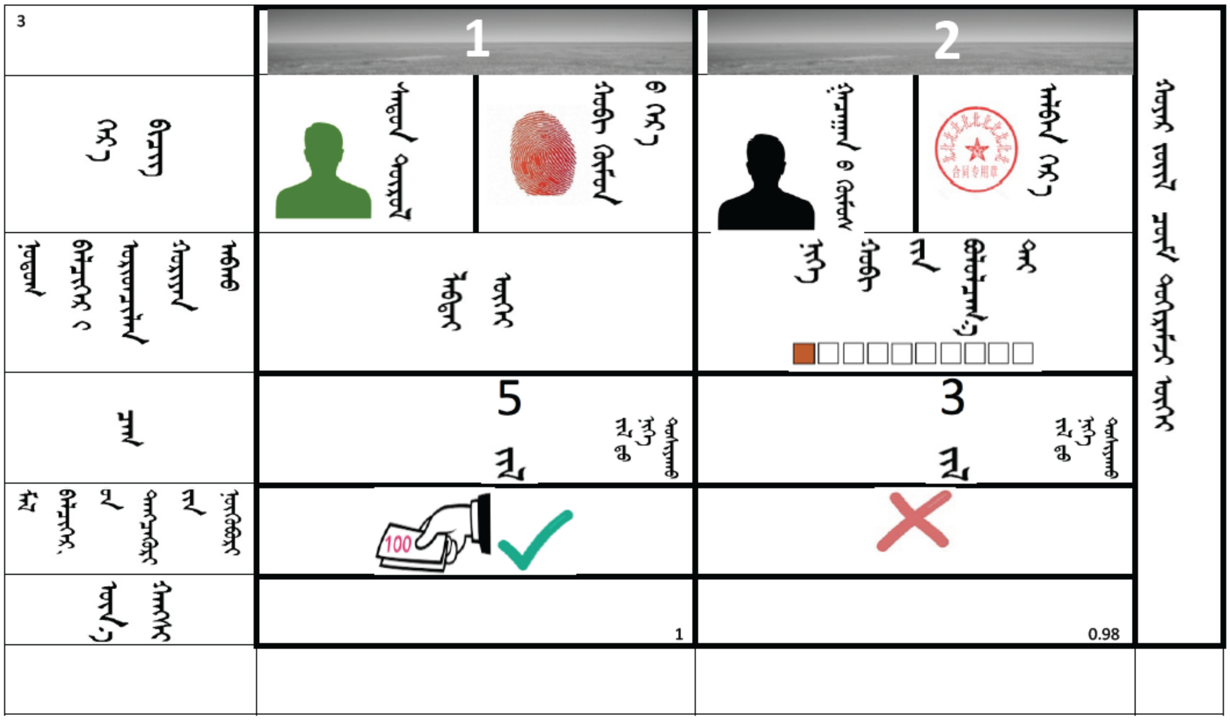


Figure 13. An example of a choice card with alternatives A, B, and an "opt-out" option ("I find both of A and B alternatives unsuitable"). Attributes are 1) formal and kinship contracts, 2) 10% chance of early contract termination, 3) duration, 4) rights to receive eco-compensation, and 5) contract price. Contract price levels are listed in the right corner as percentages and are multiplied by the herders' expected land price. English translation of the choice card can be found in https://github.com/lucixlu/dissertation_translation.

The herders in this study were identified through a stratified random sampling design. In total, 192 herders were surveyed in the final $D_{Bayesian}$ choice experiment, who came from three counties (Xilin Hot, East Ujimqin, Sonid Right) in the Xilin Gol prefecture, averaging approximately 64 responses per county. Bilingual choice sets were provided to all the herders so that they could use either the Mongol or Mandarin choice sets based on their preference. The investigator described the experiment scenario in Mandarin and an interpreter translated if the herder only spoke Mongol. We used visual aids in the choice sets to facilitate better comprehension. Other than the experiment, we surveyed the person in the household who made the *de facto* land use decisions such as renting land or selling livestock. In addition to the semi-structured interview, we also asked questions about the household's demographic and social-economic context. A GPS point was recorded inside each herder's land parcel. The point was used as a proxy for the herder's land and was used to derive satellite-based precipitation and productivity data from Google Earth Engine. For each point (herder's land), we estimated the annual net primary productivity (NPP) using NASA's MODIS MOD17A3 dataset (Running and Zhao 2015). Annual precipitation was calculated using the Climate Hazards Group InfraRed Precipitation with Station (CHIRPS) data (Funk et al. 2015). The JavaScript Code for extracting biophysical data extraction is available in Appendix II.

3.2.4 Econometric model

A choice experiment uses random utility theory and assumes that when individuals are offered alternatives composed of different attributes, they will choose the one with the highest utility (i.e., benefit, satisfaction) (Adamowicz et al. 1998). In this study, we assume that herders will choose the alternatives that maximize utility via secure land tenure. Assuming a herder *n* chooses among *j* alternatives, the actual utility (also known as latent utility, a construct in the herder's mind) can be specified as U_{nj}. Since researchers cannot examine the exhaustive lists of factors that influence the individual's actual utility U_{nj}, we use V_{nj} to indicate the part of utility that can be observed by our model and ε_{nj} to indicate the factors that influence utility yet beyond the measurement of our model, also known as the unobservable "error" or "random" component of the utility model (Adamowicz et al. 1998). Hence, the utility model can be written as:

$$U_{nj} = V_{nj} + \varepsilon_{nj}$$

The probability that the herder *n* chooses alternative *i* within a choice set can be expressed as:

$$\begin{aligned} \text{Prob}_{ni} &= \text{Prob}\ (U_{ni} > U_{nj}\ \forall i \neq j) \\ &= \text{Prob}\ (V_{ni} + \varepsilon_{ni} > V_{nj} + \varepsilon_{nj}\ \forall i \neq j) \end{aligned}$$

Latent utility can be modeled through a conditional logit model as $U_{nj} = X_{nj}\beta + \varepsilon_{njt}$, X_{nj} is the vector of observed attributes and variables that corresponds to individual *n* and alternative j, and β represents the vector of coefficient, or marginal utility for all variables (McFadden 1973). However, conditional logit is known for its restrictive independence of irrelevant alternatives (IIA) property, which assumes the herders' preference, or the odds-ratio of alternative A and B, does not change based on the presence of alternative C. As is standard in the literature, we use a random parameter logit to overcome this limitation. In addition to relaxing this restrictive substitution pattern, random parameter logit also allows the variation of individual preference and the correlation of unobserved factors over repeated observations of one respondent (Train 2009).

A random parameter logit specifies latent utility as $U_{nj}= X_{nj}\beta_n + \varepsilon_{nj}$, allowing β_n to vary over individuals instead of remaining fixed for the entire population. Instead of calculating the coefficient β_n, we estimate the parameter θ that defines the distribution of β_n, which can also be written as $f(\beta \mid \theta)$. The mixed logit probability of choosing alternative i can be denoted as (Train 2009):

$$P_{ni} = \int \frac{e^{X_{nj}\beta_n}}{\sum_{j=1}^{J} e^{X_{nj}\beta_n}} f(\beta \mid \theta)\, d\beta$$

In this study, we specify that θ has a normal distribution and estimate its mean and covariance. We allow all attributes to be random parameters but leave price to be a fixed coefficient considering that the increase of contract price decreases marginal utility for all respondents (Hynes et al. 2021). Stata 15 was used to estimate all models. The random parameter logit was estimated through the stimulation of 500 Halton draws; thus when showing results for these models, we report mean point estimates and standard errors and their respective significance.

3.3 Descriptive results

3.3.1 Household characteristics

Table 3 shows the demographic and socio-economic characteristics of all 192 surveyed herder households. On average, herders surveyed are 47 years old, with 25 years of herding experience. The household size remains relatively small ($\mu = 3.7$) as the younger generation often chooses to work in the city or start their own farms. Among the respondents, 72% are ethnic Mongols and 28% are Han. Male respondents make up 80% of the sample. Herders on average completed middle school education (8.6 years of education) and most of them are literate in Mongol or Mandarin. Only eight herders (4%) receive incomes from off-farm activities. All livestock (e.g., cattle, camels, goats) are converted into standard sheep units based on their daily consumption of forage as compared to that of a ewe.

Table 3. Descriptive statistics of the surveyed households.

Variable	Unit	Mean	Std. Dev.	Min	Max
education	years (of schooling)	8.6	3.2	0	16
livestock holding	standard sheep unit	580.6	400.4	58	2825
age	years	46.5	9.6	25	69
years of herding	years	25.1	11.6	1	60
household size	numbers of people	3.7	1.3	1	9
herders in the household	numbers of people	2.1	0.8	1	4
distance to city	kilometer	63.2	45.4	4	210
loan	yuan (0.15 dollar)	94612.5	94279.6	0	500000

Variable	Category	Count
ethnicity	Mongol	139
	Han	53
gender	male	154
	female	38
off farm income	no	184
	yes	8

3.3.2 Households' land contract contexts

Among the 192 households, 109 of them made 201 rental transactions during 2013-2019. Eighty-three households did not rent-in any land in the past seven years and only grazed on their contracted lands. Among these 201 land rental contracts, 120 (59.7%) are informal and 58 (28.9%) are formal (Figure 14A). An informal contract means our respondent only has a personal (oral or written) contract with the landlord. In the context of Inner Mongolia, a formal contract is one that has been formulated and approved by the county's Ecological Protection Bureau. Before being submitted to the Ecological Protection Bureau, a rental contract must first be approved by the township (*sumu*)'s ecological protection law enforcement team, which is responsible for enforcing grassland laws and monitoring land use. Additional approval from the herders' village (*gacha*) collective is also required. We classify the remaining 23 (11.4%) as "semi-formal" contracts because they are only approved at the village level. It is worthwhile highlighting that a 83% of the total formal contracts (N=48) are made in East Ujimqin.

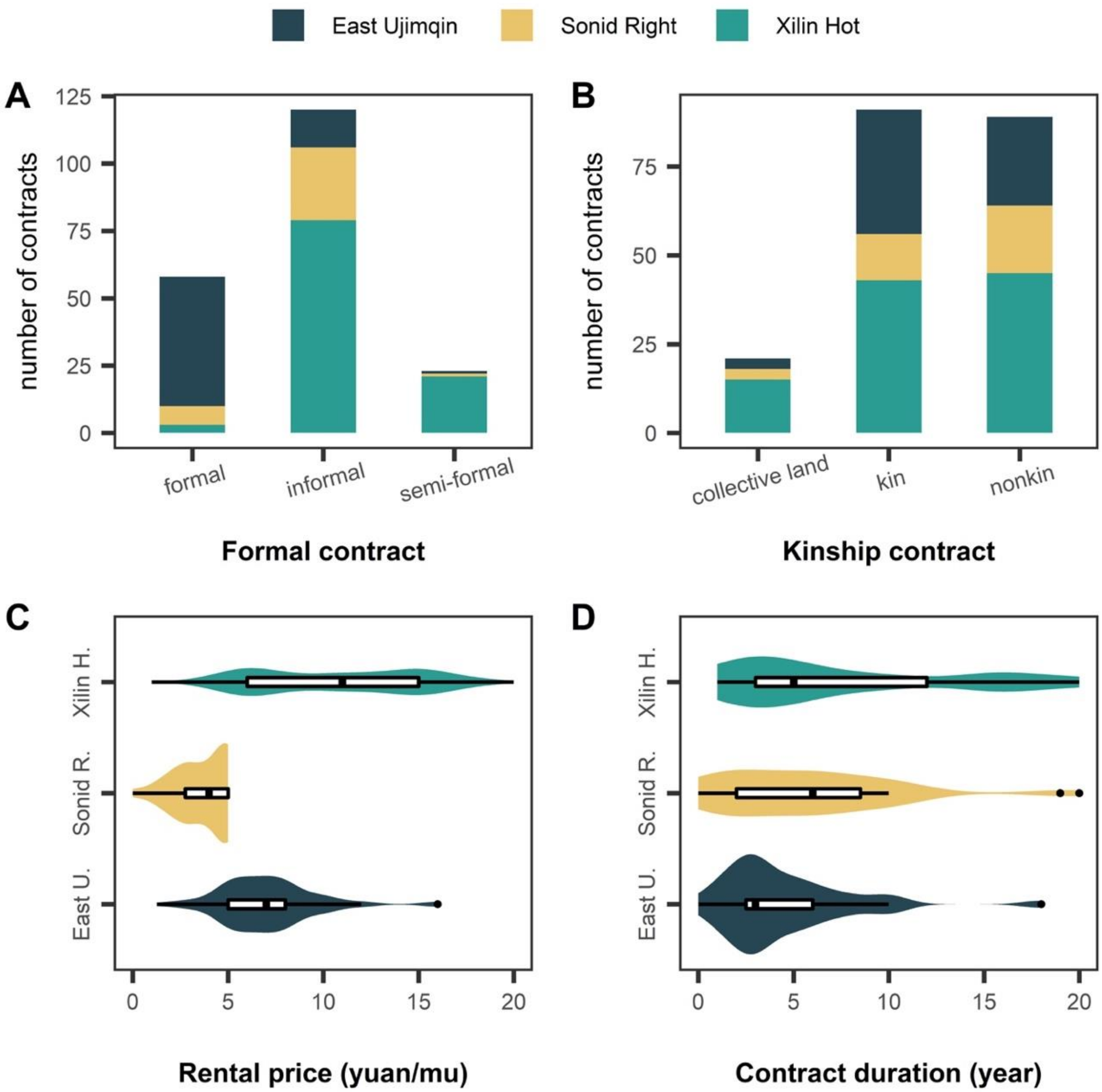


Figure 14. Descriptive statistics of the 201 land contracts made by 109 households during 2013-2019. (A) shows whether the contracts are formal, informal, or semi-formal (i.e., approved by administrative units at the village level but less formal than those approved by the Ecological Protection Bureau). (B) shows whether the landlords are related to our respondents. The distribution of rental price (C) and duration (D) for all the contracts are displayed as violin plots. The width of the violin plots indicates the frequencies of values. The boxplot inside the violin plots shows the median, interquartile range, and outliers of the data.

Kinship contracts are still common. The number of kinship contracts (N=91) is slightly larger than that of non-kinship contracts (N=89) (Figure 14B). Additionally, 21 contracts (10.4%) are made between the herders and the village collectives, which we label as "collective lands" because they do not fall under the "kinship contract or not" category. Collective grazing lands are often divided into parcels and the herders draw lots each year to decide who have the use rights. All rents are paid directly to the village collectives.

The price and duration of contracts vary across counties. The median price of all land contracts is 7 yuan/mu per year (one mu equals to 1/15 hectare). Due to its low rangeland productivity, Sonid Right county has a lower median land price (4 yuan/mu) compared to East Ujimqin (7 yuan/mu) and Xilin Hot (11 yuan/mu) (Figure 14C). The median duration of the 201 land contracts is 5 years. The East Ujimqin herders' median contract duration (3 years) is shorter compared to Xilin Hot (5 years) and Sonid Right herders (6 years) (Figure 14D). The shorter duration of East Ujimqin also corresponds with its higher percentage of formal contracts, given that only contracts shorter than three years are approved by the Ecological Protection Bureau. The median size of land transferred is 2000 mu. Land contracts made in East Ujimqin and Sonid Right have a median land size of 3000 mu, which is approximately twice the land rented in Xilin Hot (1545 mu).

3.3.3 Biophysical influence on land price

Our results show a strong positive correlation between estimated net primary productivity, precipitation, and the herders' expected land price at the household level (Figure 15). Among the three counties and within each county, there is a positive correlation between precipitation and productivity, indicating that rangeland productivity increases as yearly precipitation increases. In two of the three counties, the price of land in 2019 positively correlates with the primary productivity and the amount of precipitation each household received that year. Additionally, Figure 15 shows that the expected land price varies within each county, justifying our decision to use the relative differences from the individual's expected price as levels instead of using absolute differences.

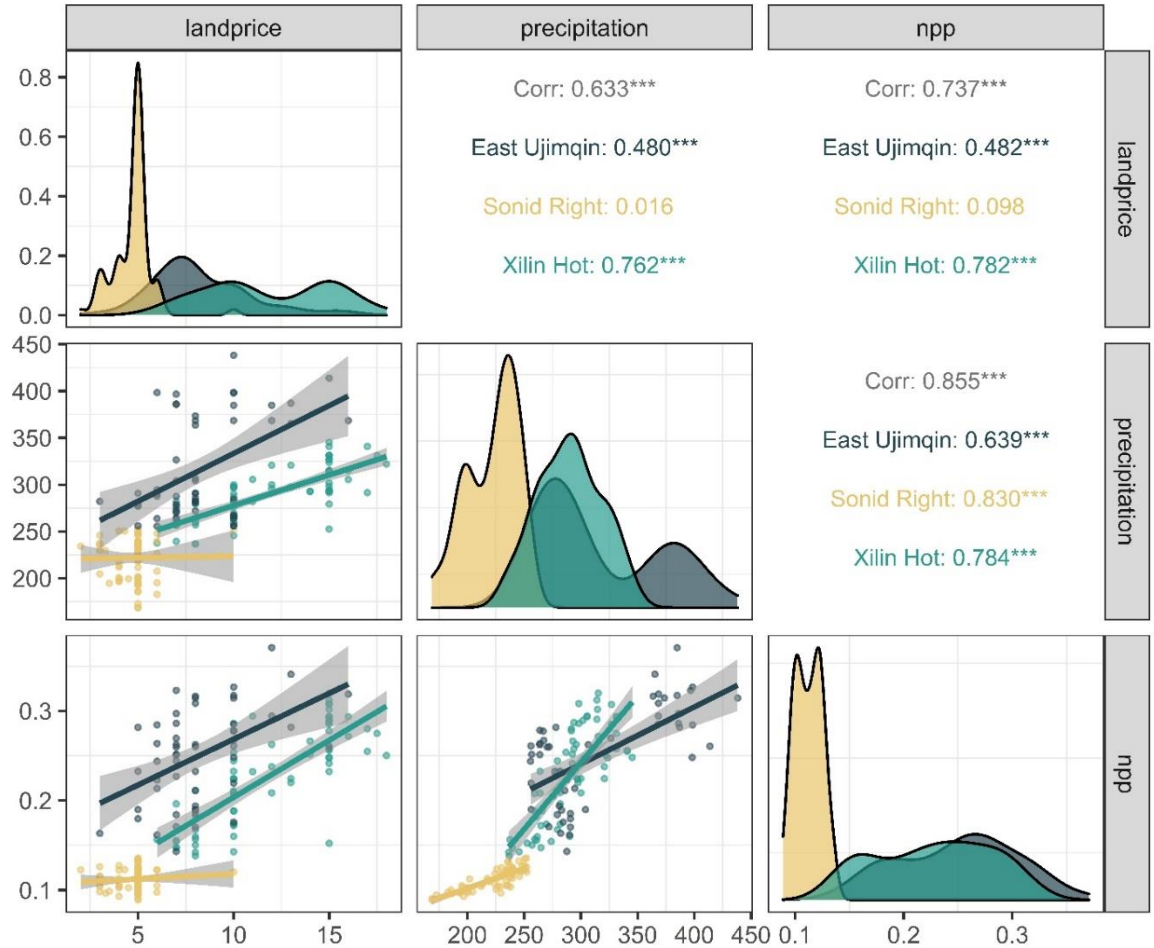


Figure 15. The correlation between herders' expected land price, precipitation, and net primary productivity (NPP). *** indicates p<0.001. Precipitation is the annual sum of precipitation per millimeters received by each herder's land, while NPP is an estimation of the mass of carbon in kilogram per square meter per year. Land price is measured in yuan per mu.

3.4 Choice model results and discussion

Table 4 shows the models used in this study. Each of the 192 herders answered 5 choice sets, generating a total of 2880 observations. Only four herders chose the "opt-out" option once in one of the five choice sets that were provided to them. Model (I) is a conditional logit model which assumes all herders have a homogenous preference for each attribute. Having a positive coefficient means that the attribute increases herders' marginal utility, while having a negative coefficient indicates the opposite. Model (II) displays a random parameter logit or the so-called mixed logit. A random parameter logit assumes that respondents' preference varies from one to another and estimates the mean and standard deviation of the distribution of each coefficient over a number of random draws from the distribution; thus, statistical significance can be estimated for both point estimates and standard deviations. Model (II) shows four out of the six random parameters have significant standard deviations at the 99% confidence interval, which indicates substantial heterogeneity in herders' preference, justifying the use of random parameter logit and is our preferred model. The conditional logit is displayed for comparative purposes (model I) and is qualitatively consistent with model II. For robustness, we interact select household socio-economic and demographic variables with the alternative specific constant and include the result in model (III). We further include an interaction term between the amount of precipitation and contract duration in model (IV) to account for the influence of precipitation on contract duration.

Table 4. Model results.

	(I)	(II)		(III)		(IV)	
Variables	Coefficient	Mean	SD	Mean	SD	Mean	SD
Kin + Informal	0.247	-0.0947	0.944*	-0.0908	0.931*	-0.0625	0.778+
	(0.204)	(0.327)	(0.420)	(0.327)	(0.422)	(0.319)	(0.447)
Kin + Formal	0.886***	1.115***	0.0362	1.116***	0.0388	1.088***	0.0668
	(0.144)	(0.210)	(0.797)	(0.210)	(0.802)	(0.206)	(0.782)
Non-kin + Formal	1.031***	1.310***	1.255***	1.308***	1.249***	1.253***	1.123**
	(0.141)	(0.244)	(0.336)	(0.244)	(0.337)	(0.234)	(0.346)
Additional land rights	0.868***	1.043***	1.109***	1.039***	1.111***	1.005***	1.041***
	(0.147)	(0.240)	(0.252)	(0.240)	(0.253)	(0.233)	(0.246)
Early termination	-0.161***	-0.174***	0.0875***	-0.173***	0.0878***	-0.168***	0.0889***
	(0.0147)	(0.0268)	(0.0246)	(0.0268)	(0.0246)	(0.0261)	(0.0238)
Duration	0.0341*	0.0620*	0.224***	0.0618*	0.224***	-0.244+	0.211***
	(0.0160)	(0.0284)	(0.0425)	(0.0284)	(0.0426)	(0.125)	(0.0410)
Contract price	-0.0236*	-0.00957		-0.00971		-0.0103	
	(0.0105)	(0.0152)		(0.0152)		(0.0149)	
alternative specific constant (asc)	-4.600***	-4.057***		-5.265		-5.308	
	(0.427)	(0.442)		(3.418)		(3.415)	
livestock*asc				-0.00180		-0.00174	
				(0.00192)		(0.00191)	
Ethnicity*asc				0.706		0.713	
				(1.182)		(1.176)	
Age*asc				0.00170		0.00187	
				(0.0463)		(0.0462)	
Education*asc				0.108		0.109	
				(0.149)		(0.149)	
Household size*asc				0.137		0.141	
				(0.367)		(0.363)	
Precipitation*duration						0.00112*	
						(0.000449)	
Log Likelihood	-936.2305	-568.1771		-567.2629		-564.1147	

AIC	1888.461	1164.354	1172.526	1168.229
BIC	1936.185	1247.872	1285.871	1287.54
# Households	192	192	192	192
# Observations	2880	2880	2880	2880

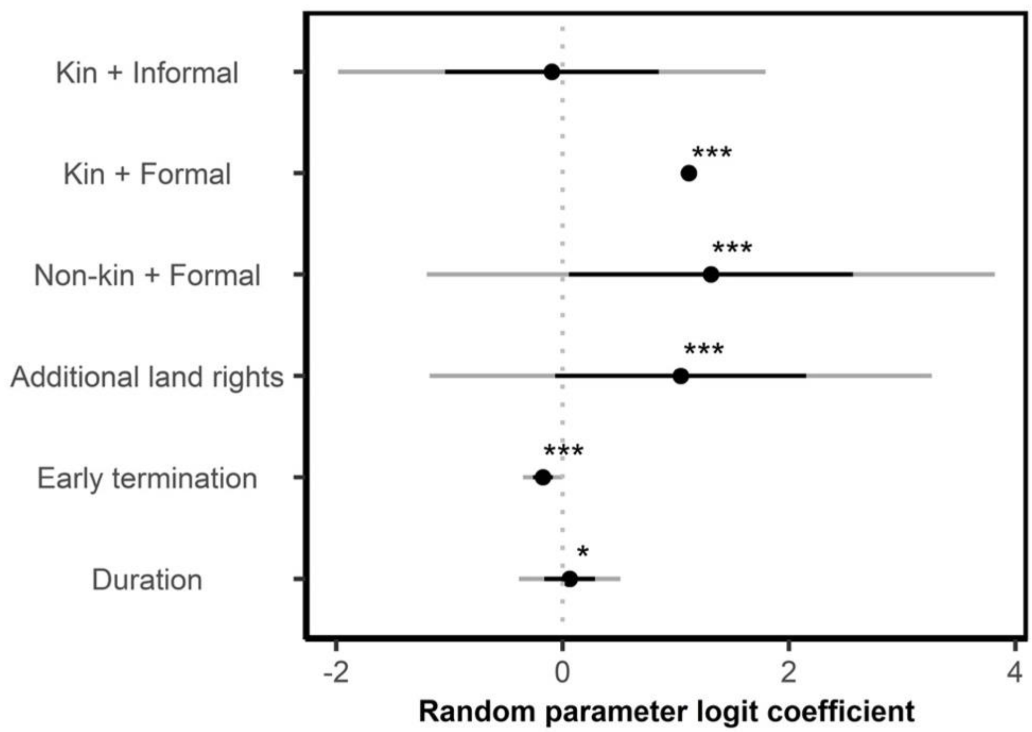
Standard errors in parentheses. *** p<0.001, ** p<0.01, * p<0.05, + p<0.1

3.4.1 Formal contract

All models show formal contracts are strongly preferred over informal ones. Kin and non-kin formal contracts are statistically indistinguishable. A mean of 1.31 and a standard deviation of 1.13 indicate that formal contracts with non-kin villagers are preferred (Figure 16). The coefficient of kinship formal contract is 1.12, but an insignificant and very small standard deviation (0.04) means that there is less variation in the preference of this parameter. A lack of variance indicates that there is much more consensus on the positive marginal utility of kinship formal contract.

In general, our results suggest herders prefer formal contracts regardless of whether the lessor is related, implying herders place great trust in formal institutions and state authorities that formulate and back these contracts. The way Abbott and others (2000) conceptualize legalization may help in explaining why herders prefer legally recognized contracts. According to Abbott and others (2000), the characteristics of legalized institutions can be summarized as *obligation*, *precision*, and *delegation*, which refers to the compliance of all parties to the rules, a refinement of the rules, and the introduction of additional authoritative parties who can implement the rules and resolve conflicts. Such characteristics may make formal contract more appealing to herders. In the case of Inner Mongolia, a formal contract clarifies the potentially ambiguous or conflicting land rights and *obligations*. The rights and responsibilities of both renters and lessors are described *precise*ly in the contract (Appendix III provides a digital copy of a formal contract provided by an East Ujimqin respondent). For example, the contract specifies that only the lessors are entitled to compensation when rented land is expropriated by the state. A renter has the right to sub-lease, but written approval from the lessor is required. Land use norms such as no hunting, crop cultivation, or exploitative hay harvesting were also emphasized as contract items. Finally, authoritative *delegates* were introduced. A standard contract is formulated by the county-level Ecological Protection Bureau. It also needs to be signed by the representatives of the bureau, the township-level ecological protection law enforcement team, and the village collective committee.

Model results



Key findings and explanations

- Herders prefer formal contract regardless of whether the lessor is related, indicating that they trust the formal institutions and authorities that formulate and issue the contracts.
- Herders favour kinship contracts only when they are formal. A formal contract may allow herders to get the benefits of renting from kin while avoiding the inconveniences of resolving disputes with them out of court.
- Herders prefer having the additional rights to receive eco-compensation and avoid risks of early contract termination.

Figure 16. The mean and standard deviation (θ of $f(\beta \mid \theta)$) for random parameters from model II. Symbols ***, **, and * represent statistical significance at 0.1%, 1% and 5% levels. The black line indicates the distribution of individual coefficient a standard deviation from each side of the mean (i.e., 68.3% of the preference distribution) and the grey line shows an additional standard deviation (covering 95.4% of the preference distribution). The base level for the kinship and formal contract attribute is “non-kin + informal contract”; the base level of the duration attribute is “3 years”.

Although only about one-third of land contracts are registered formally, our descriptive data and choice experiment analysis show that herders express a preference for formal contracts. It should be emphasized, however, that herders' preference for a formal contract does not necessarily mean that they are likely to obtain one in the future. In part this is due to 45% of the contracts being kinship contracts, and kin could interpret the proposal of signing a formal contract as an indication of mistrust. In addition, whether herders will acquire a formal contract is also influenced by various factors such as the land property value, their legal awareness, and the accessibility and efficiency of the contract-issuing department, which could be of interest in future studies. So far, our survey shows that only East Ujimqin herders have been actively obtaining legal contracts.

3.4.2 Kinship contract

Our results show that the kinship contract alone, as well as the trust and norms it symbolizes, is insufficient in constituting a secure contract. As shown in Figure 16, there is little distinction on average between kinship contracts and non-kin contracts when both contracts are informal. Our findings do not show that kinship contracts are safer relative to non-kin contracts. Interview data indicate herders seem to also have opposite preferences too. Some favours kinship contracts, indicating that "kin are easy to talk to, and we can always negotiate if there are changes that need to be made" (male, 42). At the same time, others express that they "prefer to deal with outsiders rather than kin" (male, 38).

Admittedly, kinship contracts have certain inherent advantages, including trust (Ma et al. 2015), reduction of the tenant's transaction costs on finding, selecting, and bargaining with the landlord, and potentially decreasing the likelihood of moral hazard (e.g., the landlord rescinds a contract unexpectedly) (Sadoulet, Janvry, and Fukui 1997; Holden and Ghebru 2006). The fact that 45% of the contracts recorded in our study are kinship contracts also indicates that such contracts are not actively avoided by herders.

However, if moral hazards indeed occur, renters often bear higher social costs when in a kinship contract. During the interviews, herders mention that it is not rare for a lessor to raise rent or evict a tenant, especially if the contract is informal. In that case, kinship norms and long-established relationships often make it difficult for the renters to negotiate and claim their rights privately. When such unexpected circumstances occur among kin, formal contracts are able to offer additional legal protection to kinship contracts (Woolthuis, Hillebrand, and Nooteboom 2005). Herders thus enjoy the benefits of renting from kin while avoiding the potential dilemma of settling land disputes out-of-court with them. This may explain that although on average non-kin formal contracts increase marginal utility the most, the herders appear to have a much more homogenous agreement that formal kinship contracts improve tenure security.

3.4.3 Risk of early contract termination

Unsurprisingly, the “10% chance of early contract termination” attribute is avoided by the herders. A mean of -0.174 and a small standard deviation of 0.0875 show that a 10% increase in the chances of early contract termination decreases marginal utility and is avoided by 98% of the herders. The most impacted stakeholders appear to be the land-scare herders, who usually depend on the rented land to sustain the herds. Some land-scare herders argue “I have nowhere to go (male, 37)” and “there is no [or not enough] place[s] to store my sheep (male, 58)” if a contract terminates earlier. Early contract termination can take different forms, including eviction, state appropriation, or the withdrawal of certain land rights. Nevertheless, it refers to the uncertainties associated with certain contracts, making tenants’ efforts and investments futile (Sjaastad and Bromley 2000). Our result is supported by earlier findings by Qin, Carlsson, and Xu (2011), which demonstrate that the risk of early contract termination is a major influencing factor for tenure security as perceived by Chinese land users.

3.4.4 Additional land rights

The findings from the choice experiment indicate that most herders favour having the additional rights to receive eco-compensation. Rights to compensation is a component of the *bundle of rights*, referring to whether landholders can request equitable compensation when the state withdraws or expropriates land and land rights (RRI 2012). The ongoing eco-compensation program in Inner Mongolia currently compensates herders for the partial revocation of their land rights. More specifically, the eco-compensation program revoked herders’ rights to withdraw rangeland resources (i.e., graze) freely by compulsively requiring all herders to reduce their herd size. However, because eco-compensation was typically paid to the *de jure* contract holders, the lessors (the non-operating landholders) are the only recipients of eco-compensation, despite the renters (the land users) bearing the actual cost of the withdrawal/management restrictions. The deprivation of land rights and compensation may disincentivize renters from managing the land sustainably. Conversely, our findings suggest entitling renters to compensation would likely increase their marginal utility and may indeed help incentivize better management.

Herders on average prefer receiving eco-compensation as an additional land right for the land they rent. Nevertheless, the size of the standard deviation around the mean shows that individual herders' preferences vary greatly. Concerns of future conflicts caused by shifting land rights may explain why some herders believe this attribute decreases marginal utility. Currently, only the *de jure* landholders have the right to compensation regardless of whether they are using the lands or not, therefore eco-compensation itself has turned into social welfare for the lessors (Li et al. 2018). During interviews, some herders stated their concern that depriving the rights to compensation (welfare) of the landlords might stir discontentment and potential conflicts. Therefore, despite our study revealing that additional rights to renters increase marginal utility, it remains a question how these existing rights can be shifted from the lessors to the renters without causing conflicts.

3.4.5 Contract duration

The coefficient estimate for contract duration is relatively weak across all models, suggesting that herders are generally content with the 3-year contract. The large and significant standard deviation shows herders still have various preferences for contracts' duration. This contradicts most studies in agricultural regions, which often conclude that longer duration is more advantageous (e.g., Bandiera 2007; Lovo 2016). The biophysical rangeland context of Inner Mongolia, in contrast to the mechanized and intensified agricultural systems, is governed by inter-annual fluctuations of precipitation and forage production (Sloat et al. 2018). Herders often meticulously calculate the risks associated with the temporal-spatial heterogeneous distribution of precipitation. Most renters indicate they might be able to withstand a year of drought, but consecutive years of drought and lower than expected rangeland productivity may result in significant financial strain. The influence of unpredictable inter-annual rainfall on herders' choices is noted by one herder:

> A ten-year contract is too long. What if it becomes dry after the first one or two years that I started the contract? I cannot even change to another place. If the [duration of the] contract is short, it will be much more convenient for me to switch to better quality rangeland during a drought (male, 57).

Therefore, a fixed, long-duration contract may be associated with higher risks when compared to short and flexible contracts. However, herders do not have a strong preference for contracts that are too short. Since the primary goal of renting land is to enlarge livestock production rather than to avoid risks, annual contracts are also not preferred by herders due to the potential associated transaction cost.

In addition to the influence of the *variability* of precipitation, we tested the relationship between the annual sum of precipitation (*aridity*) and herder's choice in the extended Model (IV). Model (IV) included the interaction terms of precipitation and the duration attribute. After controlling for the influence of precipitation, our results show that the mean coefficient for duration becomes negative at the 1% significance level. This may indicate that herders in a wetter environment (typical and meadow steppe) prefer longer contracts, while herders in a drier environment prefer shorter contracts. More research is needed to disclose the detailed relationship between precipitation patterns and tenure security.

Compared to other attributes, our findings on contract duration appear nuanced. Our results show that duration matters, but unlike farmers, herders may not necessarily prioritize a long-duration contract. Herders' preference for the duration is heterogeneous and may be related to the amount of annual precipitation received. However, there is no clear evidence that another contract length is better than the baseline length of a contract (i.e., three years).

3.5 Conclusion

In this study, we used a discrete choice experiment to compare and quantify the relative importance of the factors that influence tenure security. Through a choice experiment, we evaluated to what extent certain factors are favored or avoided by the land users. Additionally, a random parameter logit allows us to better understand the preference heterogeneities within the herder population.

A limitation of a choice experiment is that it is constrained to a selection of attributes and a specific study population. Although all attributes were chosen guided by a detailed literature review and qualitative pilot interviews, we inevitably filter out other variables that are comparatively less important or impact only part of the population. In Inner Mongolia, other factors that potentially threaten herders' tenure security include illegal land appropriation, intra-household conflict, inheritance dispute, the loss of land rights after a woman married and moved to another village, lack of enforcement, volatile environmental policies, and the environmental impact of coal mining. Additionally, by using a choice experiment, we are constrained to understand only overall and generalized behavioral patterns; we are unable to fully unfold the contextual reasons that drive each individual's land management decisions (Robinson et al. 2007). Detailed qualitative methods such as participatory observation, in-depth interviews, or oral history may help to better explore the reasons behind herders' preferences. Another limitation of the study design is that our price range was likely set too conservatively. Although the coefficient of the price variable is negative as expected, it also appears imprecisely estimated. This means that compared to other factors, 2% to 10% price changes do not appreciably affect herders' decisions. Further studies could expand the price range.

Our findings have two policy recommendations. Firstly, our data shows that although less than 30% of the current contracts are formal, having the contract protected by legal institutions increases herders' marginal utility (and tenure security). Therefore, the next step in improving tenure security will be to improve the public understanding of formal contracts and to remove administrative barriers to obtaining one. To overcome the difficulties in information transmission caused by the remoteness and dispersion of herders' settlements, legal knowledge may be disseminated to herders using both traditional and new media platforms, such as radio talk shows and social media. Moreover, education campaigns should normalize formal contracts among kin, making herders feel less uneasy or guilty about "breaking the norm" if they decide to obtain a formal contract.

Secondly, our study demonstrates that renters' marginal utility increases when they are eligible for rights to compensation. So far, *de jure* landowners who have rights to due process and compensation are frequently the (only) recipients of incentive-based environmental policies, even though they may not be actively managing the land. As the land rental market among smallholders expands globally, the current policy may be in danger of losing its ability to incentivize sustainable land use practices on a larger scale. We suggest that eco-compensation or payment for ecosystem services programs designers carefully evaluate the recipients' land-use status and their land rights.

Appendix I: Ngene code for choice experiment design

Small sign Design [used in pilot survey]

```
Design
;alts=option1*, option2*, opt-out
;rows=20
;eff=(mnl,d)
;alg=swap(stop=total(500000iterations))
;block=4
;model:
U(option1)=b2[0.00001]*Duration[1,3,5,10]+b3.dummy[0.00001|0.00002|0.00003]*Formalizati
on[1,2,3,0]+b4.dummy[0.00001]*PES[1,0]+b5[-0.00001]*Eviction[0,10]+b6[-
0.00001]*Price[92,98,100,102,108]/
U(option2)=b2*Duration+b3*Formalization     +b4*PES     +b5*Eviction+b6*Price/
U(opt-out)=asc
$
```

Bayesian Design [(normal distribution, coefficient, std.err)]

```
Design
;alts=option1*, option2*, opt-out
;rows=20
;eff=(mnl,d)
;alg=swap(stop=total(500000iterations))
;bdraws=halton(500)
;block=4
;model:
U(option1)=b2[(n,0.0812, 0.0519)]*Duration[1,3,5,10]+b3.dummy[(n,-2.1828,0.7771)|
(n,0.0503,0.5446)|(n,0.1311,0.6058)]*Formalization[1,2,3,0]+
b4.dummy[(n,1.0143,0.4147)]*PES[1,0]+b5[(n,-0.0832,0.03920)]*Eviction[0,10]+
b6[(n,0.0657,0.0393)]*Price[92,98,100,102,108]/
U(option2)=b2*Duration+b3*Formalization     +b4*PES     +b5*Eviction+b6*Price/
U(opt-out)=asc
$
```

Appendix II: JavaScript code to retrieve biophysical data from Google Earth Engine

```
// MODIS NPP data extraction
var xg = ee.FeatureCollection("users/lux3/XilinGol/Xilingol");
var hh = ee.FeatureCollection("users/lux3/XilinGol/192hh_location");

var visualization = {
  bands: ['Npp'],
  min: 800.0,
  max: 2000.0,
  palette: ['bbe029', '0a9501', '074b03']
};

var dataset = ee.ImageCollection('MODIS/006/MOD17A3HGF')
              .map(function(image){return image.clip(xg)});
var NPP2019 = dataset.filter(ee.Filter.calendarRange(2019, 2019, 'year'));

Map.setCenter(114.372, 42.482, 6);
Map.addLayer(dataset.mean(), visualization, 'NPP');
print(dataset);

var triplets=NPP2019.map(function(image) {
  return image.select('Npp').reduceRegions({
    collection:hh.select(['HHid']),
    reducer:ee.Reducer.mean(),
    scale:500
  }).filter(ee.Filter.neq('mean',null))
  .map(function(f) {
    return f.set('imageID', image.id());
  });
}).flatten

Export.table.toDrive({
  collection: triplets,
  fileFormat: 'CSV',
  folder:"GEE"
});

// CHIRPS precipitation data extraction
var chirps = ee.ImageCollection('UCSB-CHG/CHIRPS/PENTAD');
var xilingol =ee.FeatureCollection("users/lux3/XilinGol/Xilingol");
var hh = ee.FeatureCollection("users/lux3/XilinGol/192hh_location");

Map.setCenter(115.07, 44.518, 6);
```

```
var years = ee.List.sequence(1990, 2019);

var PrecbyYear = ee.ImageCollection.fromImages(
    years.map(function(y) {
      return chirps
        .filter(ee.Filter.calendarRange(y, y, 'year'))
        .reduce(ee.Reducer.sum())
        .set('year', y);
    })
  );

print(PrecbyYear, "PrecbyYear");

var palette = ['#ffffcc','#a1dab4','#41b6c4','#2c7fb8','#253494']
var visParams = {
  min:0,
  max: 300,
  palette: palette
}

Map.addLayer(PrecbyYear.mean().clip(xilingol), visParams, "30 year mean");
Map.addLayer(PrecbyYear, visParams, 'Total Precipitation')
Map.addLayer(hh, {}, "hh")

var triplets=PrecbyYear.map(function(image) {
  return image.select('precipitation_sum').reduceRegions({
    collection:hh.select(['HHid']),
    reducer:ee.Reducer.mean(),
    scale:5000
  }).filter(ee.Filter.neq('mean',null))
  .map(function(f) {
    return f.set('imageID', image.id());
  });
}).flatten();

Export.table.toDrive({
  collection: triplets,
  fileFormat: 'CSV',
  folder:"GEE"
});
```

Appendix III: A copy of the formal contract issued by the East Ujimqin Banner

English translation can be found in https://github.com/lucixlu/dissertation_translation.

东乌珠穆沁旗草原承包经营权租赁合同

甲方（出租方）：____
身份证号码：____
居住住址：____
发包方地址：_______________苏木（镇）____________________嘎查
联系电话：____

乙方（承租方）：____
身份证号码：____
居住住址：____
联系电话：____

为规范草原承包经营权流转行为，维护流转双方当事人合法 权益，根据《中华人民共和国草原法》、《中华人民共和国合同法》、《内蒙古自治区草原管理条例》、《内蒙古自治区草原管理条例实施细则》、《东乌珠穆沁旗草原承包经营权流转管理办法》等有关法律法规规定，本着自愿、平等、协商、有偿的原则，甲方愿将以下草原的承包经营权租赁给乙方，经双方协商一致，特订立此合同，共同遵照执行。

一、租赁草场基本信息

租赁草场位于东乌珠穆沁旗___________________________苏木（镇）_______________嘎查， 草场证承包登记人_______________, 草场证登记面积___________亩，本次租赁面积______________________亩，适宜载畜量为____________只羊单位。

具体位置坐标：____

二、租赁期限、租赁面积、租金及交纳方式

租赁期限______， 自___年__月___日起至____年__月____日止。

租赁面积为_________亩；其中放牧场________亩、打草场________亩，人工草地__________亩。 租赁草场租金为每年________元／亩，租金金额_________元（大写：___________）。租金交纳方式：__________________________________

三、甲乙双方权利义务

（一）甲方权利义务

1、依法享有草原承包经营权流转的权利。

2、按时向乙方收取约定的租赁价款。

3、监督乙方合理利用、保护流转草原，制止乙方损坏流转草原和其他草原生态资源的行为。依照草原相关法律、法规，监督乙方按照生态保护部门核定的适宜载畜量饲养牲畜，

发现超载过牧及禁牧、休牧期间有偷牧行为时，要及时予以制止， 并及时报告相关部门。

4、维护乙方的合法畜牧业生产经营自主权，不得干涉乙方的依法正常的畜牧业生产经营活动。不得非法变更、解除流转合同。

5、享受流转草原被依法征占用时所得相应补偿。

6、应保存草原承包经营权证和草牧场承包合同。

7、法律、法规规定的其他权力义务。

（二）乙方权利义务

1、依法享有租赁草场的经营畜牧业生产权及所得利益，有非法权自主组织、生产、经营和处置产品。

2、按时向甲方交付约定的租赁价款。

3、依法保护和合理利用草原，不得擅自改变草牧场用途，

4、乙方在租赁草原上饲养牲畜，必须遵守草原有关法律、法规，要严格按照草蓄平衡饲养牲畜，严格按照打草场管理办法相关规程打草刈割作业，不得超载过牧、开垦草原、乱采滥挖野生植物、狩猎、兴建畜牧业以外的永久性建筑物。

5、草原租赁期间，乙方将承租草原进行再流转，必须取得甲方书面同意。

6、草原租赁到期时，及时向甲方交还租赁草原或者协商继续租赁。乙方在同等条件下享有优先承租权。

7、租赁草原被整体或部分被依法征用或占用而无法继续履行合同，乙方应得到不能继续使用的整体或部分草原的租赁费。

8、法律、法规规定的其他权利与义务。

四、双方违约责任

1、合同执行期间任何一方不得擅自变更或解除合同。如有一方违反 合同应按本合同和有关法律、法规规定，承担违约合同，责任向对方支付违约金，由违约方向对方造成的损失应支付赔偿金，并继续履行合同。

2、乙方在租赁草牧场期间，对租赁草场因管理不善、超载过牧等原因造成草牧场生产力下降、草场等级降低的，应采取积极措施恢复补救并对造成的损失进行补偿；如掠夺性经营、开垦、承租的草原再次进行擅自流转，从而造成租赁草场严重破坏的，除应依法处理外，应对甲方进行相应的赔偿，同时终止履行草牧场租赁合同。 不得造成流转草牧场等级下降、造成永久性损害。

五、免责条件

因不可抗力因素导致合同无法履行时，双方相互承担责任，造成的损失由双方自行承担。

六、争议解决方式

如在租赁草牧场内发生草牧场界线纠纷，应由甲方负责协调解决。合同执行过程中发生的其他纠纷应平等协商解决，协商不成的可申请相关部门协调解决，也可申请仲裁或向人民法院提起诉讼。

七、其他事宜

1、甲乙双方当事人协商解决流转草场原有基础设施和新建基础设施的利用和管理。

2、本合同流转草场的四至界线、面积、等级和地块必须与草原承包经营权证及草原承包合同的所列数据和内容相一致。

3、本合同甲乙双方法定代表人签字盖章，所属嘎查委员会（发包方）、苏木（镇）草原生态保护执法队审核盖章，旗生态保护局备案后生效。

八、附款

1、本合同一式三份，双方各执一份，旗生态保护局备案一份。

2、本合同未尽事宜由甲乙双方协商后，可签订补充合同。

3、草原权属证复印件、甲乙方双方身份证复印件为本合同附件，附合同之后。

甲乙双方协商的补充条款：____

____

____

甲方（出租方）（签章）__________________

乙方（承租方）（签章）__________________

嘎查委员会意见（盖章）____

苏木（镇）生态保护执法队审核意见（签章）__________________________

东乌珠穆沁旗生态保护局备案意见（盖章）___________________

Connecting statement

Chapters 2 and 3 discussed the impacts of rental markets from different angles. Analyzing each household's land holding and whether the land is under *de facto* and *de jure* control, chapter 2 examined the rental market's direct impacts on land holding and land rights. Chapter 3 examined the insecurity of rental contracts and identified the key factors that herders perceive as threatening (or enhancing) tenure security. Rental agreements are often considered insecure in agricultural economics literature because of their short duration and limited property rights, which discourage land investment. However, quantitative evidence is lacking due to the difficulties of linking individual land users to their land and monitoring the long-term impact of property rights on land change. In this chapter, I combine remote sensing, participatory mapping, and household surveys at the parcel level to measure the impact of the rental on rangeland resources.

Chapter 4

Rights and rangelands: integrating household survey and satellite data at parcel level

4.1 Introduction

Property rights have long been recognized as fundamental to the sustainable management of land resources and are recognized as foundational to global development agendas such as the Sustainable Development Goals and the Paris Climate Agreement. Secure rights allow landholders to be more forward-looking and have greater incentives to invest in longer-term outcomes, often promoting greater care and stewardship of land into the future (RRI 2012). While there are number of ways to develop secure right regimes (Ostrom, Janssen, and Anderies 2007; Sze et al. 2021), privatization of property is still a dominant policy tool used to combat land degradation (Tseng et al. 2021).

A consequence of land privatization is that it gives landowners the right to allow others to use their land under rental or sub-lease agreements. Rental land as a “byproduct” of privatization has been largely ignored in the study of social-ecological land system dynamics, despite the robustness and growth of rental markets worldwide in various sectors. For example, the efficiency and equity of agricultural rented markets have been widely documented (Holden, Otsuka, and Place 2010), but how rental markets affect ecosystem and land outcomes has only recently gained attention .

Rangeland systems cover 30% to 40% of global land surface and are a highly important biome for water, biodiversity, livestock farming (Sayre et al. 2013), and rental markets for rangelands in particular are experiencing considerable growth (Neudert 2015; Su, Tang, and Qiu 2021). Nevertheless, parcel-level comparisons of outcomes in rented rangelands to those with more complete property rights are exceedingly rare. This is partly explained by the challenges in integrating data on land tenure, rangeland management, and ecological outcomes at a scale appropriate for linking management decisions – i.e., at a parcel or household level (Rindfuss et al. 2004; Liverman and Cuesta 2008). Linking herders to parcels of their land at a micro-scale is further complicated by the unique characteristics of rangeland socio-ecological systems (Lambin et al. 2009), such as the influence of highly variable inter-and intra-annual precipitation, the lack of clearly defined property boundaries, widespread informal land agreements, and potentially complex land-use practices.

In this study, we address these methodological issues with a combination of participatory mapping, household surveys, and remote sensing to understand the location and intensity of households' livestock production activities. A combination of the empirical context and innovative field methods allows us to identify differences between rented and privately-held rangelands by linking data on household management, land rights, land use, and remotely sensed rangeland productivity outcomes for 400 parcels that belong to 187 households in Inner Mongolia, China. We statistically test for the influence of a parcel being rented versus privately-held in a multilevel modeling framework. This combination of methods allows us to identify how the relationship between parcel-level land tenure and rangeland health and productivity.

4.2 Methods

4.2.1 Methodological approach

Tying herder behavior to rangeland outcomes is difficult due to a lack of scale-appropriate (parcel-level) data on land tenure, resource management, and rangeland outcomes over time. To overcome this, we streamlined an analytic process to allow herders to identify individual parcels, describe their herding rights and practices, and then tie these to high-resolution remotely sensed data over time. First, we pre-processed and tiled Sentinal-2 images for use on tablet computers, highlighting fences (that are hard to discern on standard "real color" basemaps) and other landscape features that allow herders to better identify their parcel boundaries on satellite images (Figure 17A). Individual herders then identified their parcel boundaries in the field, at which time we additionally asked each herder about land rights and any dedicated use for each parcel (i.e., all-season, winter, and summer grazing, lambing, or hay harvesting) (Figure 17B). After compiling all herder parcels into a spatial database, we used these parcels as "cookie-cutter" polygons to extract historical Landsat 8 images from which we calculated metrics of rangeland productivity and abiotic and topographic information. Finally, we used statistical models to assess the relative influence of rental tenure on rangeland outcomes, while controlling for well-documented biotic and abiotic factors (Figure S1, S2, and S3), household characteristics, and township-level unobservable factors (Figure 17C).

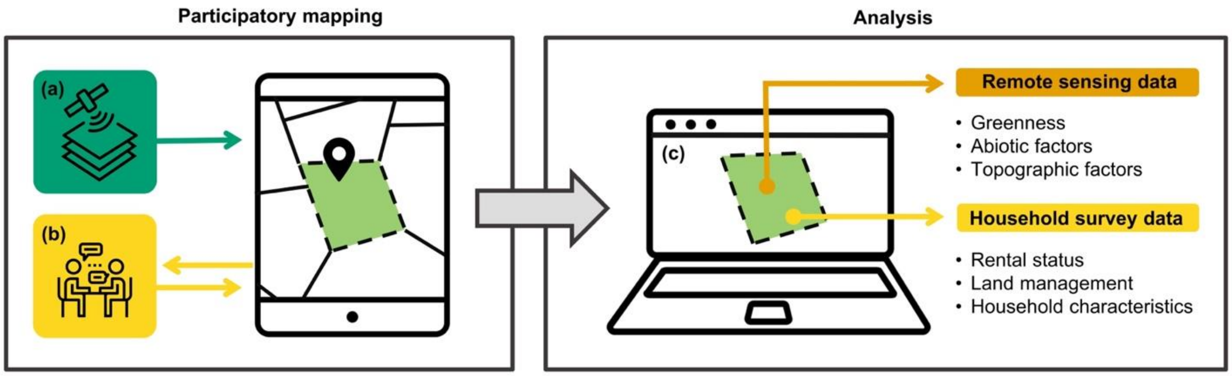


Figure 17. Methodological flowchart. We (A) created Sentinel-2 basemaps and stored them in a tablet computer, (B) conducted tablet-based surveys and participatory mapping with individual herders, and (C) extracted remote sensing data from land parcels identified by the herders, integrated remote sensing and household survey data, and analyzed data.

4.2.2 Study area

Pastoral land in China is often defined as "quasi-privatized": rangelands are technically collectively owned, but individual households have 30 to 50-year private contracts over land parcels (here referred to as "privately held parcels") (Banks 2003; Reid, Fernández-Giménez, and Galvin 2014). In the 1980s, rangelands were contracted first to small groups of herders and later to individual households in the 1990s (Li and Huntsinger 2011), which were often then fenced. Herders born after the 1990s share their parents' contracted land with siblings. Rental markets thus provide opportunities for younger herders, especially those who lack initial land endowments or other off-farm opportunities, to acquire land from an aging cohort of landholders (Jimoh et al. 2021). China's "Grassland ecological protection reward-subsidy" policy provides compensatory payments to land contractors to retire from herding or reduce livestock holdings (Robinson, Li, and Hou 2017), though renters cannot receive the payments from this program.

4.3 Participatory mapping

4.3.1 Preparing basemaps

To facilitate participatory mapping, basemaps were derived from the Europe Space Agency's Sentinel-2 images, focusing on Level 2A "Bottom-Of-Atmosphere" reflectance product. We identified thirteen 100 km^2 tiles that cover the sampled households located within the 49T and 50T Universal Transverse Mercator zone (Figure 18). For each tile, we downloaded an image with the least amount of cloud cover in December 2018 and March, June, and September 2019. We displayed the growing season images in near-infrared band combination (near-infrared, red, green) and winter images in shortwave infrared band combination (shortwave infrared, near-infrared, red).

These band combinations allow for visualization of wavelength ranges not usually visible to the human eye, which helps us identify fences and property boundaries via spectral differences in lightly and heavily grazed land parcels. Rangeland parcels are often utilized for distinct purposes, durations, and under various intensities, leaving different amounts of standing biomass among parcels. During the growing season, ungrazed or lightly grazed land will have taller and denser grass compared to heavily grazed land, all else being equal (Figure 19D). A near-infrared band composite accentuates biomass contrasts and helps highlight fences that separate parcels (Figure 19E). Dry grasses and snow also accumulate along fence edges, thickening the trace of the fence which, in shortwave infrared band composite appears clearly as bright cyan lines (Figure 19F).

We converted all GeoTIFF satellite images into map tiles using ArcGIS Pro. We displayed these map tiles as offline base maps using ArcGIS Collector and collected spatial data in the field. Collector displays our real-time movement and location on top of the map tiles and allows herders to interact with the tiles by panning and zooming.

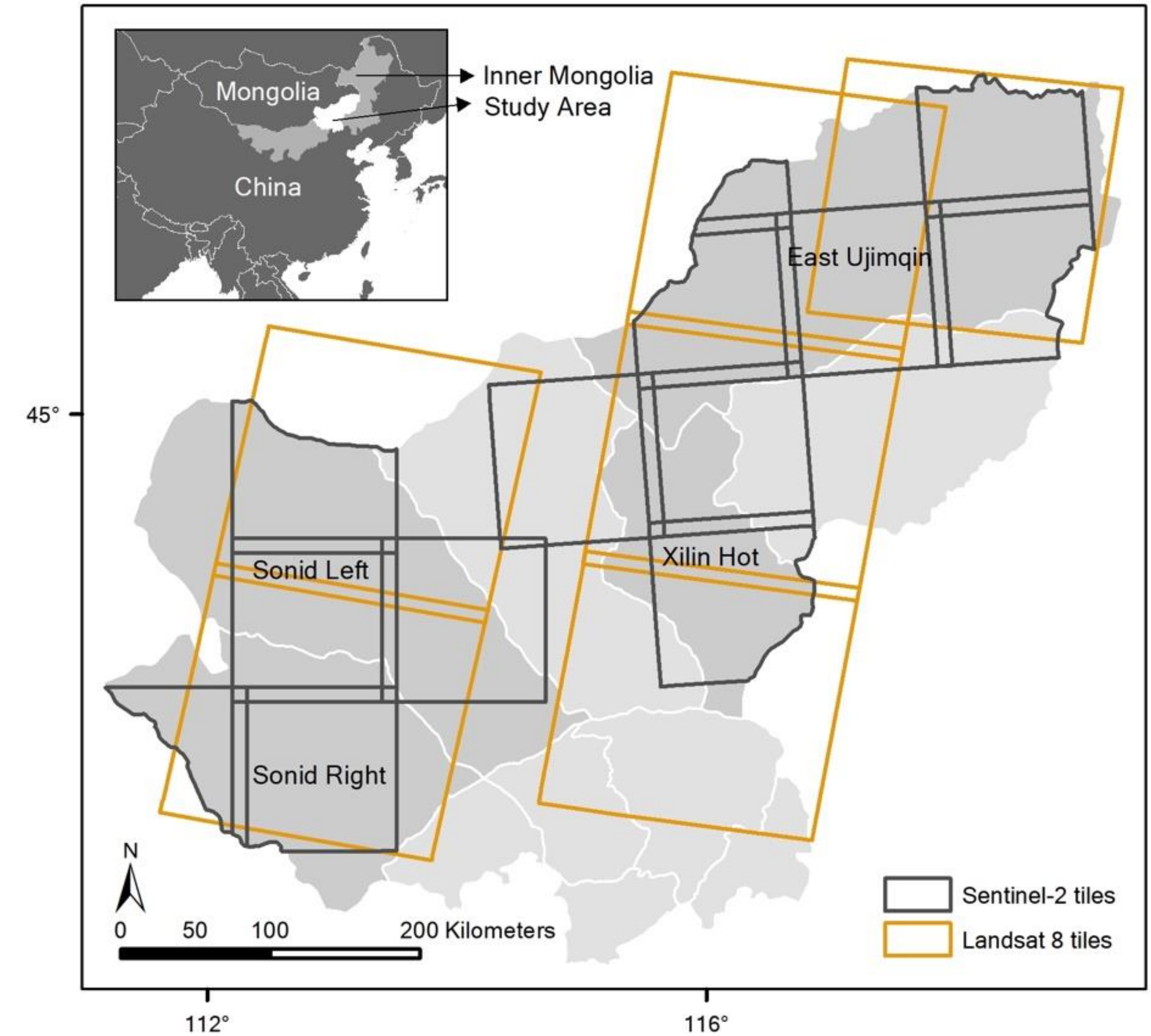


Figure 18. Map of the study area. Boxes represent tiles from satellite imagery.

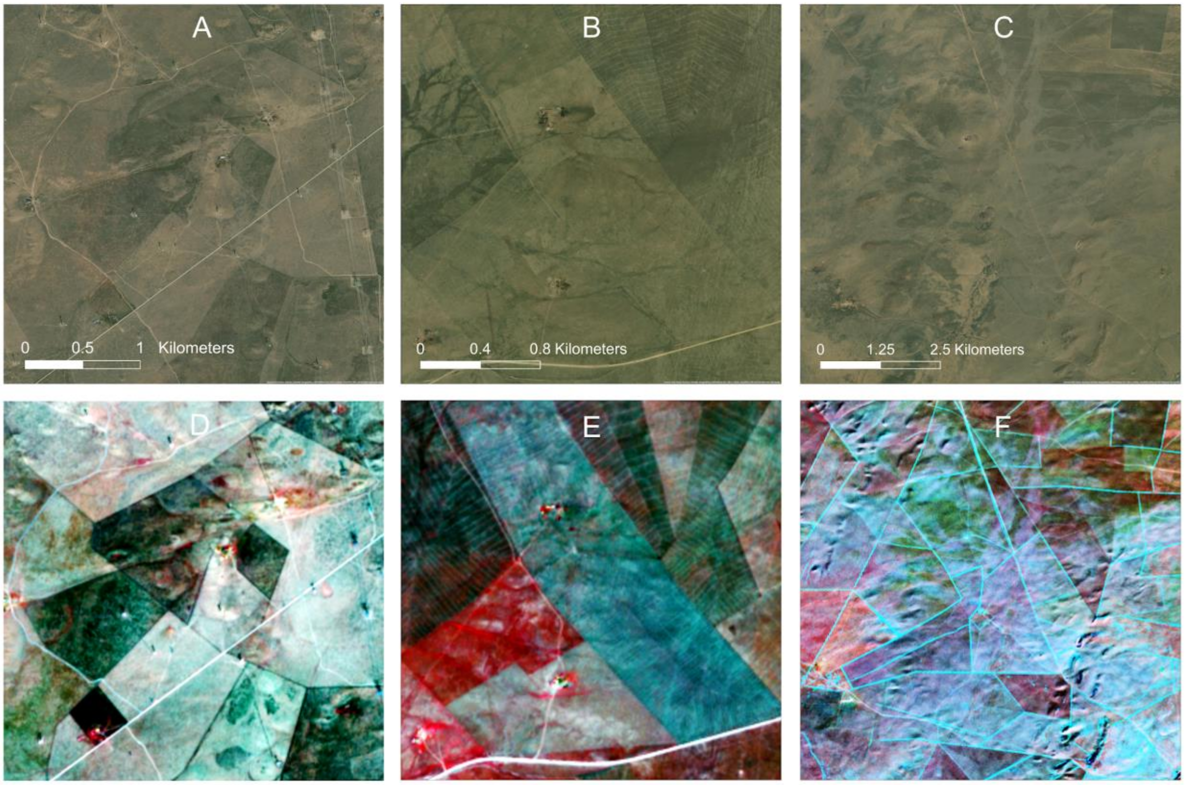


Figure 19. The comparison of three ArcMap's default high-resolution "World Imagery" basemap images (A, B, and C), and the same images in near-infrared (D and E) and shortwave infrared band combination (F). The near-infrared composite of the satellite image (D) captured in September 2019 shows a strong contrast between the brighter parcels that have already been used for summer grazing and the darker parcels reserved for winter. The darkest square parcel located at the bottom left of image (D) is an enclosure kept ungrazed for years, often reserved for extreme drought. Image (E) shows the contrast between different intensities of hay harvesting (dark and light green parcels with stripes) and grazing (white and red parcel) among two households. The shortwave infrared band combination (F) shows how land parcel boundaries that are difficult to discern in "natural color" maps become more visible.

4.3.2 Participatory mapping process

To further verify whether a land parcel is used and inquire about parcel-based land management information, we conducted participatory mapping with herders. To bridge herders' *first-person* spatial knowledge with a 2-dimensional *bird's eye* satellite image (Taylor and Tversky 1992), we identified visible land features while viewing maps with herders. To verify understanding, we quantified the herder's reported parcel geometry (Kwinta and Gniadek 2017) and spatial relationship among features (Tversky 2003) with the ones we measured using the ArcGIS Collector. We followed a 3-step process to identify land parcels with herders:

1) We oriented herders to cardinal directions on the landscape. We arranged the tablet towards the north and invited the herder to orient themselves with the satellite imagery. We continually reinforced the cardinal directions during the mapping process.
2) We presented herders with three options for identifying their land parcels based on their comfort and digital literacy: through satellite imagery on the tablet, a sketch map drawn by the investigator with the basic land features (e.g., communal wells, motorcycle paths, concrete roads, gates, etc.), or blank sketching paper with cardinal directions indicated.
3) To verify each land parcel, we asked herders questions about the parcel's geometry (e.g., size, shape, and edge length) and the parcel or herder's spatial relationships (e.g., direction and distance) with key land features. We then compare herders' responses with our measurements derived from ArcGIS Collector (see detailed steps in Figure S4 and S5). We also intentionally asked about land features that did not belong to the herder to make sure the respondent disagreed.

For each parcel identified, we asked the herder detailed land management and property rights questions including the rental situation, land use purpose, and duration. Participatory mapping was accompanied by a standard household survey that recorded demographic and livelihood information.

4.3.3 Rangeland parcel sample

Of the 512 parcels initially identified, 415 passed all three steps in the verification process (above). Of parcels that did not pass, only a partial fence was recognizable in the basemap or we could not reconcile a parcel's satellite-observed geometry and the herder's description. Among the fully recognized parcels, we excluded nine because they were primarily covered by saline-alkali marshes or gravel pits and another six that were reserved for times of severe drought (Figure 19D). In the end, 400 fully recognizable parcels from 187 households in 17 townships (Table S1) are included in our analysis, giving us 2800 parcel-year observations are available over the 7-year period. Among these, 306 parcel-years were covered by clouds (Table S2) and an additional 215 were classified as having an “unknown” rental status. We labeled a parcel’s tenure status as “unknown” when, e.g., a parcel was rented by our respondent from 2015-2019 but before 2015 it was not tied to our respondent. In the end, our sample contains 2279 parcel-year observations.

4.4 Multilevel models

To assess the relationship between land tenure and rangeland outcomes, we build several statistical models that take into account known abiotic and household management factors that influence rangeland outcomes. Multilevel analysis is chosen because we have repeated measures for each parcel over time, and there is clustering in parcels’ greenness that is possibly correlated with other nearby parcels. Therefore, observations on a given parcel may share similar characteristics with other geographically or temporally proximate observations. Our multilevel framework follows the form:

$$VI_{ijkt} = \beta_0 + \beta_1 A_{ijkt} + \beta_2 T_{ijk} + \beta_3 M_{jk} + \beta_4 Z_{jk} + \beta_5 R_{ijkt} + \mu_k + \varepsilon_{ijkt}$$

where VI_{ijkt} is the satellite-image derived vegetation index for land parcel *i* belonging to household *j* in township *k* at time *t*. For each land parcel, we measured its vegetation index, abiotic influence, and rental situation (rented or privately held) at time period *t* (a specific day in September of each year from 2013 to 2017). A_{ijkt} represents time-variant abiotic factors that constrain VI_{ijkt} at the parcel level, and T_{ijk} are time-constant topographic factors. M_{jk} represents rangeland management factors stocking rate and fodder expenditure, and Z_{jk} represents household demographic or socioeconomic characteristics. The impact of interest is R_{ijkt}, the impact of land being rented (vs privately held) on rangeland outcomes. The random intercept μ_k captures time-invariant unobservable at the township level (level 1), and ε_{ijkt} are the residuals at the observation (parcel) level (level 0). To help control for time and differences in vegetation productivity by banner, we de-meaned (centered) the log-transformed VI by banner and year. All analyses were performed using Stata 15 (StataCorp 2017).

4.5 Data

4.5.1 Vegetation Index

We used soil-adjusted vegetation index (VI) as a proxy of relative vegetation greenness and vigor. Although Vis are dimensionless and do not measure biomass quality or quantity directly, previous literature has established robust relationships between VIs and rangeland biophysical parameters such as the fraction of absorbed photosynthetically active radiation (fAPAR) and leaf area index (Weiser et al. 1986; Fensholt, Sandholt, and Rasmussen 2004). The most widely used VI, Normalized Difference Vegetation Index (NDVI) utilizes vegetation's unique spectral characteristic on near-infrared and red wavelength regions and measures relative photosynthesis activities as: $NDVI = (\rho NIR - \rho Red)/(\rho NIR + \rho Red)$.

One shortcoming of NDVI is that it can be influenced by soil background reflectance when the vegetation canopy does not fully cover a pixel (Huete 1988). Consequently, soil-adjusted VIs such as the Soil Adjusted Vegetation Index (SAVI) and the Modified Soil Adjusted Vegetation Index (MSAVI) have been developed to minimize the impact of soil on vegetation reflectance properties. Because SAVI requires prior knowledge of a fixed soil brightness adjusting factor, we chose MSAVI for this study because MSAVI uses an inductive method to reduce soil influence and is suitable for our landscape scale that has heterogeneous soil properties (Qi et al. 1994), and has been used in past work in the Xilin Gol prefecture in Inner Mongolia (Liu et al. 2007). MSAVI is calculated as:

$$MSAVI = (2 * \rho NIR + 1 - \sqrt{(2 * \rho NIR + 1)^2 - 8 * (\rho NIR - \rho Red)})/2.$$

We used the atmospherically corrected Landsat 8 surface reflectance dataset in Google Earth Engine to calculate MSAVI (Gorelick et al. 2017). Landsat 8 was chosen over other satellite data because it offers moderate spatial resolution at 30 meters and, importantly, provides a consistent annual source from 2013. We queried the full Landsat 8 Image Collection from 2013 to 2019 to only select the image tiles covering our area and date of interest. All households surveyed were located within six Landsat tiles on three swaths, namely two tiles in row 29 and 30 of path 126 (which covers our targeted households in Sonid Left and Sonid Right county), three tiles in row 28, 29, and 30 of path 124, and one tile in row 28 of path 123 (which covers our targeted households of Xilin Hot and East Ujimqin county) (Figure 18). These tiles on paths 126, 124 and 123 also roughly encompass desert steppe, typical steppe, and meadow steppe, respectively. Within each path, all parcels were visited on the same date. Observations on path 126 were always visited two days later than those on path 124, and five days later than those on path 123 due to Landsat's north-south orbital pattern. Since the abiotic-VI relationship may be different within the primary growing season, we selected September (the end of productive grassland growth for the year) from which to sample (see Figure S6 for the specific date chosen for this study). September was chosen over the other months of the growing season partly because it offers more cloud-free observations (89.07% of the total 2800 observations remain cloudless), but also because it provides a better estimate of a final outcome that relates to the season's land use. Before any indices were calculated, we applied a function to mask clouds and shadows to the image collection's pixel quality band.

4.5.2 Abiotic and topographic variables

There is a lag between abiotic factors' occurrence and when they influence vegetation greenness (Wang, Rich, and Price 2003), so we measured the average value of three abiotic variables (land surface temperature, precipitation, and incoming shortwave radiation) from Google Earth Engine during the 14 days in advance of the vegetation index retrieval date each year. For example, the vegetation index measured on September 17th, 2017 of path 126 is paired with the average value of our abiotic variables between September 3rd to 16th, 2017.

Daily land surface temperature data come from the Moderate Resolution Imaging Spectroradiometer (MODIS) MOD11A1 dataset (Wan, Hook, and Hulley 2015). Daily precipitation was calculated using the Global Satellite Mapping of Precipitation (GSMaP) hourly gauge-adjusted data (Kubota et al. 2020), which has been shown to offer the most accurate daily precipitation estimation for China (Tang et al. 2020). Downward shortwave radiation comes from the Global Land Data Assimilation System dataset (GLDAS-2.1) (Rodell et al. 2004), from which we chose the maximum radiation value every day at 14:00 local time.

Topographic variables for each parcel include average elevation (meters), slope (degrees), and aspect northing ("northness"), all derived from the 30-meter resolution Shuttle Radar Topography Mission (SRTM) digital elevation dataset (Farr et al. 2007). We converted degrees to radians and calculated aspect northing as the cosine of radians, which ranges from -1 (due south) to 1 (due north), with 0 indicating east or west.

4.5.3 Land tenure and management variables

Our main interest is whether a parcel is rented or privately held each year affects rangeland outcomes. A "rented" parcel is defined as a parcel for which the tenant has paid a landlord for use rights over the rangeland and has some form of contractual agreement (informal or formal). We also differentiated renting from "otor", a traditional Mongol drought-coping practice where land may be borrowed or rented in a time of need, but is generally much shorter in duration (Xie and Li 2008) (See Table S3 for detailed comparisons).

We used a household's stocking rate (# sheep/hectare) and annual fodder expenditure (RMB/sheep) to measure herders' land management intensity. The stocking rate proxies herders' dependence on the natural grassland while fodder indicates herders' acquisition of external food resources that would offset the reliance on natural grassland for livestock consumption. Stocking rate is the most commonly used indicator for grazing intensity, generally measured in "standard sheep units" (SSU) per hectare. SSU allows for other livestock to be converted to a common unit (sheep) according to their daily consumption needs relative to a representative ewe. Following the Chinese Agricultural Industry Standard issued by the Ministry of Agriculture and Rural Affairs (Ministry of Agriculture and Rural Affairs 2015), we used the following conversion factors to calculate SSU: 0.8 for goats, 5 for local breeds of cattle, 8 for crossbred cattle (e.g., Holstein Friesian), 5.5 for local horses, and 9 for camels. All lambs and calves are converted to 0.5 of their corresponding adult livestock. In this study, we use a household's *de facto* landholding to calculate stocking rate (instead of the *de jure* land that herders often reported to standard census surveys) to accurately capture intensity of grassland use. *De facto* land can also include land formally held by (absentee) siblings, parents, children, or relatives who live and work in urban areas. All land was included, regardless of being formal or informally held. Fodder expenditure is normalized by SSU, which includes expenditure on hay, hay pellets, grain (corn and oats), corn silage, and straw. We also controlled for respondent and household characteristics including years of herding, years of education, ethnicity, gender, household size, herding labor, and distance to city.

4.6 Results

4.6.1 Household and parcel characteristics

Our data were collected from herding households across four counties in Xilin Gol prefecture of Inner Mongolia, China. Of the 187 herder households for whom we successfully identified at least one of their land parcels, 74 households grazed only on their own privately held parcels between 2013-2019, 108 used a combination of rented and privately held parcels, and 5 households are only renters (Table 5). Among the 400 parcels identified, 90 were rented and 310 were privately held for at least one year between 2013 and 2019 (Table 6). Although at the household level, 60.5% of the surveyed households (n = 113) grazed on a rented parcel at some time, some of these could be tens of kilometers from the respondent's homes and therefore were not able to be captured through participatory mapping.

Table 5. Descriptive statistics of household-level data.

		Typical and meadow steppe (N=109[a])						desert steppe(N=78)					
		East Ujimqin (*N*= 50)			**Xilin Hot** (*N*= 59)			**Sonid Left** (*N*= 15)			**Sonid Right** (*N*= 63)		
variable	unit	Privately held[b] (*N*= 13)	rent (*N*= 37)		Privately held (*N*= 17)	rent (*N*= 42)		Privately held (*N*=6)	rent (*N*=9)		Privately held (*N*=38)	rent (*N*=25)	
Land management													
stocking rate (avg)	sheep/ha	1.47(0.54)	1.22(0.53)		1.77(0.81)	1.49(0.65)		0.72(0.34)	0.77(0.38)		0.9(0.43)	0.74(0.33)	
fodder expenditure (avg)	100 yuan /sheep	0.44(0.34)	0.42(0.36)		1.09(0.56)	1.65(1.2)		0.91(0.58)	1.68(1.1)		1.3(1.14)	1.93(1.31)*	
Household characteristics													
distance to city (avg)	10 km	8.92(6.95)	7.41(5.61)		7.49(2.17)	6.1(2.67)		4.75(2.36)	5.33(2.15)		3.83(2.64)	2.8(2.18)	
decades herding (avg)	10 years	2.08(1.46)	2.22(1.06)		2.19(1.25)	2.71(1.02)		1.63(1.5)	2.36(0.65)		2.73(1.24)	2.51(1.32)	
Education (avg)	years	9.38(3.43)	8.76(3.74)		9.65(3.44)	8.98(3.1)		10.83(5.95)	10(1.5)		7.74(2.93)	7.92(2.43)	
gender (N)[c]													
Female		2	6	$X^2(1)$	4	8	$X^2(1)$	4	1	$X^2(1)$	6	5	$X^2(1)$
Male		11	31	=0.01	13	34	=0.15	2	8	= 5*	32	20	=0.19
Ethnicity(N)													
Han		0	7	$X^2(1)$	2	18	$X^2(1)$	1	5	$X^2(1)$	9	15	$X^2(1)$
Mongol		13	30	=2.86	15	24	=5.22*	5	4	=2.27	29	10	=8.43**
household size (avg)	# of people	3.92(0.86)	3.95(1.25)		3.29(0.92)	3.74(1.15)		4.33(1.03)	3.56(1.13)		3.71(1.33)	3.32(0.85)	
herding labor (avg)	# of people	2.54(0.66)	2.08(0.68)		2(0.79)	1.93(0.71)		2.5(0.84)	1.78(0.67)		2.13(0.84)	2.08(0.57)	

Standard deviations in parenthesis. *, **, *** indicates significance at 0.05, 0.01, 0.001 using mean comparison tests.

[a] N indicates number of households.

[b] "Contract" means the household grazed only on contracted land, while "rent" means the household used at least one rented land parcel between 2013-2019.

[c] Gender of the respondent (who is also the household decision-maker).

Table 6. Descriptive statistics of parcel-level data.

		Typical and meadow steppe(*n*=258[a])				desert steppe(*n*=142)			
		East Ujimqin (*n*=115[a])		**Xilin Hot** (*n*=143)		**Sonid Left** (*n*=29)		**Sonid Right** (*n*=113)	
variable	unit	Privately held (*n*=91)	rent (*n*=24)	Privately held (*n*=102)	rent (*n*=41)	Privately held (*n*=23)	rent (*n*=6)	Privately held (*n*=94)	rent (*n*=19)
Abiotic[b] (*14 days average*)									
land surface temp	Celsius	27.34(1.01)	27.51(0.92)	29.4 (1.05)	28.63(1.16)***	31.95 (0.64)	32.07 (0.66)	31.6(0.56)	31.78(0.46)
precipitation	mm	1.91(0.15)	1.96(0.15)	1.45(0.06)	1.46(0.05)	0.74(0.03)	0.76(0.03)	1.26 (0.19)	1.33(0.14)
solar radiation	W/m^2	568.72(10.72)	565.84(11.01)	576.78 (9.15)	584.03 (9.11)***	611.92(5.43)	610.65(5.18)	623.09(5.55)	621.92(4.44)
Abiotic (*14 days maximum*)									
land surface temp	Celsius	34.06(1.08)	34.16(0.94)	36.19(0.71)	36.02(0.69)	38.18(0.7)	38.26(0.83)	37.84(0.49)	37.98(0.61)
precipitation	mm	13.89(1.98)	14.49(1.82)	9.8(0.46)	9.9(0.42)	5.01(0.07)	5.04(0.04)	8.99(1.33)	9.52(0.98)
solar radiation	W/m^2	688.89(4.53)	689.29(4.03)	708.43(5.18)	712.86(5.65)***	730.61(3.71)	729.48 (3.23)	745.37 (1.28)	745.34(1)
Topographic									
elevation	meter	908.48 (54.24)	920.08(50.84)	1082. 3(157.2)	1173.67(154.56)**	967.92(35.64)	971.57 (30.49)	1107.02 (39.88)	1116.36 (19.96)
slope	degree	3.03(0.99)	3.04(0.78)	2.71(1.36)	3.1(1.46)	2.6(0.28)	2.6(0.23)	2.41(0.36)	2.32(0.24)
aspect northing[c]	[-1, 1]	-0.09(0.29)	-0.12(0.29)	0.11(0.22)	0.21(0.22)*	0.03(0.07)	0.03(0.12)	0.09(0.11)	0.08(0.15)

Standard deviations in parenthesis. *, **, *** indicates significance at 0.05, 0.01, 0.001 using mean comparison tests.

[a] n indicates number of parcels.

[b] Abiotic variables are calculated as the mean of 7 years (2013-2019).

[c] Aspect northing shows whether a parcel is "north facing" or "south facing". It ranges from -1 (due south) to 1 (due north).

4.6.2 Rental impact

We estimated a number of models to assess predictors of grassland greenness (MSAVI). Figure 20 shows results for our preferred multilevel model that estimates rangeland greenness as a function of abiotic, topographic, land tenure, land management, and socioeconomic influences (see table S5 for numeric results). The model suggests rented land is significantly associated with lower greenness. MSAVI-greenness is measured on a -1 to 1 scale with no real-world analog, limiting the interpretability of effect sizes. Still, the model suggest that a rented parcel has, on average, about a 2% lower value relative to a privately held parcel in the same year and county, all other factors being equal.

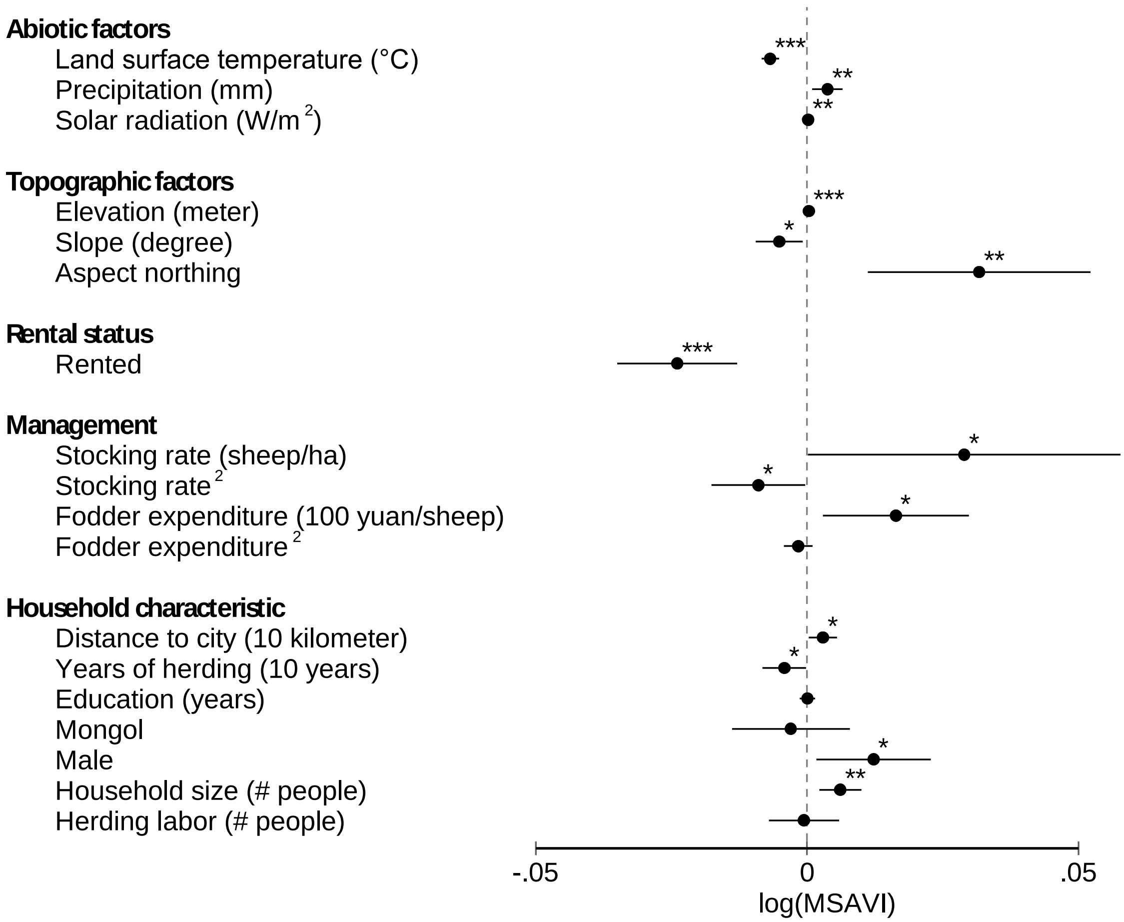


Figure 20. The effect of rented tenure on rangeland outcomes in a multilevel framework. *** p<0.001, ** p<0.01, * p<0.05.

The abiotic (temperature, precipitation, and solar radiation) and topographic (elevation, slope, and aspect) variables used in the model influence primary productivity, in agreement with previous literature (Schloss et al. 1999; Zhou et al. 2021). In terms of land management factors, our study shows stocking rate and fodder expenditure are associated with greenness in a nonlinear way. The initial positive livestock-vegetation relationship weakens and reverses as livestock per hectare increases (Figure 21). Thus, in our sample low stocking rates are positively associated with grassland health, but greater numbers of sheep per hectare relate to worse grassland outcomes, as we would expect. Herders' expenditure on fodder (reliance on external resources) generally has an increasing positive effect on greenness that stabilizes as fodder expenditure increases—suggesting that greater amounts of feed coming from external markets puts less pressure on local grassland resources.

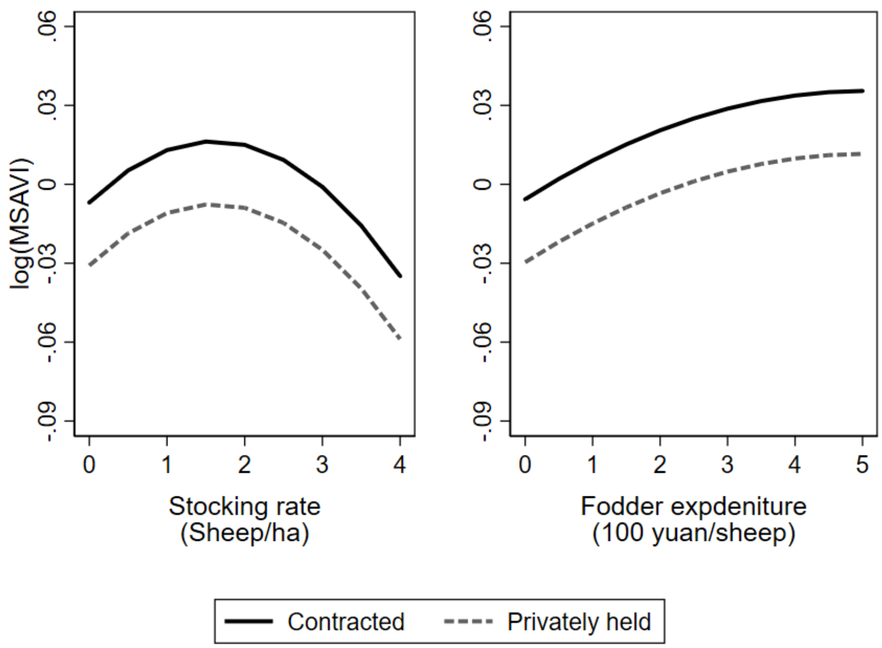


Figure 21. Predicted percentage change in log MSAVI corresponding to 1 unit increase in stocking rate and fodder expenditure for rented versus privately held land. Greenness is measured by the percentage change in Modified Soil-Adjusted Vegetation Index (MSAVI) centered at a cluster (year and county) mean.

We conducted a number of robustness checks on the model above, including a model that uses a 7-day window to measure abiotic influences (table S6), one that uses NDVI to measure greenness (table S7), and a model that specifies each parcel's primary use as designated by the herder (i.e., all-season, winter, and summer grazing, lambing, or hay harvesting) (Table S4 and S8). We also estimate a model that excludes the banner of Xilin Gol (Table S9), which is the only county in our sample where rental and privately held parcel characteristics appear to have some differences (Table 6). All results from these checks are qualitatively consistent with those presented in Figure 20.

4.6.3 Discussion

With a multilevel statistical model that controls for abiotic, topographic, land management, and household-level socioeconomic influences, our result shows that rented parcels are associated with parcels that are less green compared to long-term privately held parcels. Given the growth in rental markets worldwide, our results suggest an increasing need for policy attention on the active management of rented land.

Renter-operated land is common across the globe. Approximately 40% of all farmlands in the United States are managed by renters (Bigelow, Borchers, and Hubbs 2016) while in the Czech Republic and the European Union 83% (Sklenicka et al. 2014) and 53% (Swinnen et al. 2013) of the agricultural lands are under leasehold, respectively. Rental of agricultural land is increasing in Africa and Asia as well, especially in countries such as Ethiopia and China, where permanent land transfer is infeasible (Holden, Otsuka, and Place 2010; Gao, Huang, and Rozelle 2012). Rental allows for flexible distribution of land utilization based on comparative returns to labor. Yet, our study suggests there is a conflict between sustainable land management and the potential efficiency gains that can come through land rental markets.

Rental tenure can also hamper the effectiveness of market-based conservation instruments such as payment for ecosystem services. Policies that aim to incentivize land management lack clear mechanisms to connect to land operators in rental markets. Since most current incentive payments are offered to the *de jure* instead of the *de facto* landholders, such incentive loses salience in the rental context because it has little influence on the renters' land use decisions and thus the ecosystem services provided. For example, Inner Mongolia's "Grassland ecological protection reward-subsidy policy" has provided payments to 12 million herding households over the past decade to limit livestock and consequent grazing pressure (Li et al. 2020). However, as only the *de jure* landholders have the rights to receive compensation, compensation flows to absentee landlords instead of feeding back into renters' land management practices (Li et al. 2018). Incentivizing the renters directly may be better, but still transaction costs for identifying, monitoring, and enforcing payment for ecosystem contracts with short-term renters are often too high to implement proper incentives. As payment for ecosystem services and similar programs gain popularity (Salzman et al. 2018), creativity is needed to deal with this rental paradox in incentive-based land management policies.

Despite developing a unique approach to temporal-spatial vegetation analysis at the parcel level, our study is still constrained by the limitations of open-access satellite data and cross-sectional surveys. Multispectral satellite-derived vegetation indices handle rangeland compositional change poorly (e.g., transitions from native to invasive plants) (Karnieli et al. 2013; Hopping et al. 2018), thus our results assume that greener pastures are indeed "better." Hyperspectral data may offer ways to identify and map the compositional change based on plant species' unique spectra (Sankey et al. 2021), though there are technical difficulties of conducting a landscape-scale hyperspectral analysis for all surveyed households. Finally, our socio-economic data are constructed from a cross-sectional survey, and therefore may not reflect, for example, unforeseen shocks (e.g., loss of a family member, financial crisis) that could influence land use decisions (Hopping et al. 2018; Crook, Robinson, and Li 2020). Future studies can benefit from pairing biophysical time series with longitudinal survey data.

4.7 Conclusion

In this study we measured detailed land tenure information over rangeland parcels and parcel-level vegetation change from 2013 to 2019. Remote sensing allows us to reconstruct and analyze longer-term vegetation dynamics and their abiotic influences, while household survey and participatory mapping enable us to explore the underlying drivers of change in land management and land-use intensity. Our study shows that more comprehensive land-use models can be constructed by linking remote sensing and household survey data at the parcel level. We found that rented land parcels have large and consistent negative effects on rangeland greenness relative to private parcels. Renters' lack of the right to receive compensation or incentives in market-based programs, should be recognized and incorporated into policies that aim to incentivize sustainable rangeland management.

Appendix IV: Supplementary figures and tables

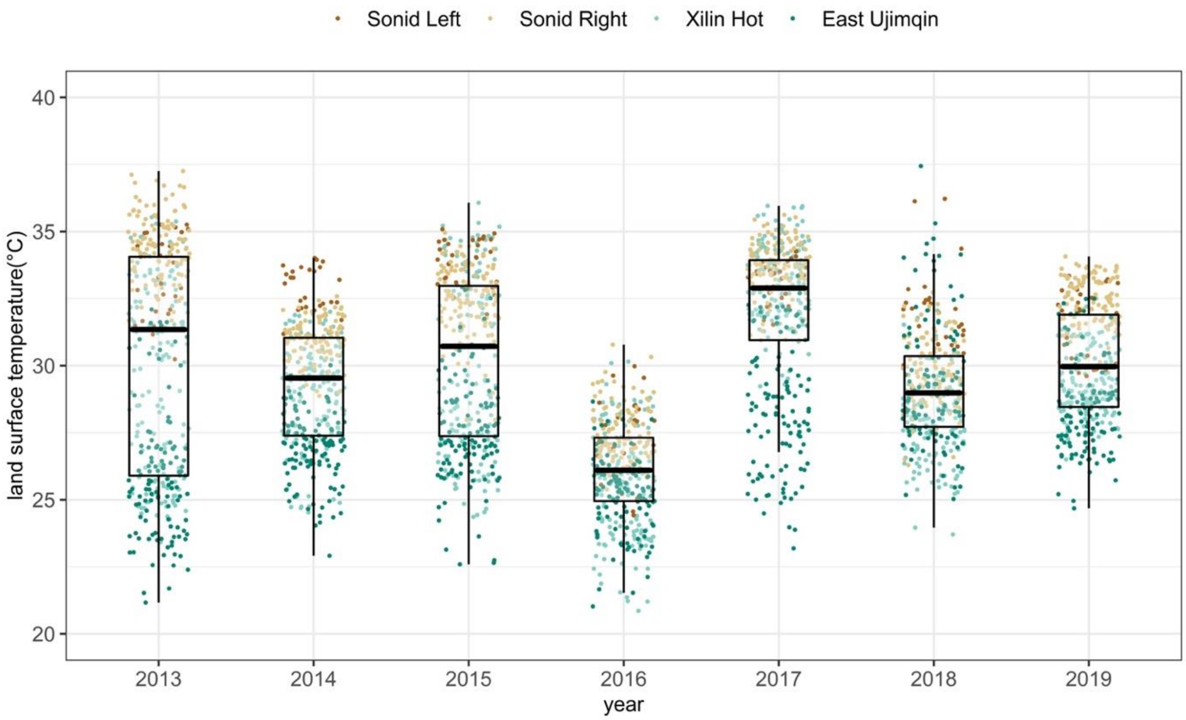


Figure S1. Land surface temperature of the 2800 observations over 400 parcels. Each point represents the average daily land surface temperature measured during the 14 days in advance of the specific vegetation index retrieval date between 2013 and 2019.

Figure S2. Precipitation of the 2800 observations over 400 parcels. Each point represents the average daily precipitation measured during the 14 days in advance of the specific vegetation index retrieval date between 2013 and 2019.

Figure S3. Downward shortwave radiation of the 2800 observations over 400 parcels. Each point represents the average daily shortwave radiation measured during the 14 days in advance of the specific vegetation index retrieval date between 2013 and 2019.

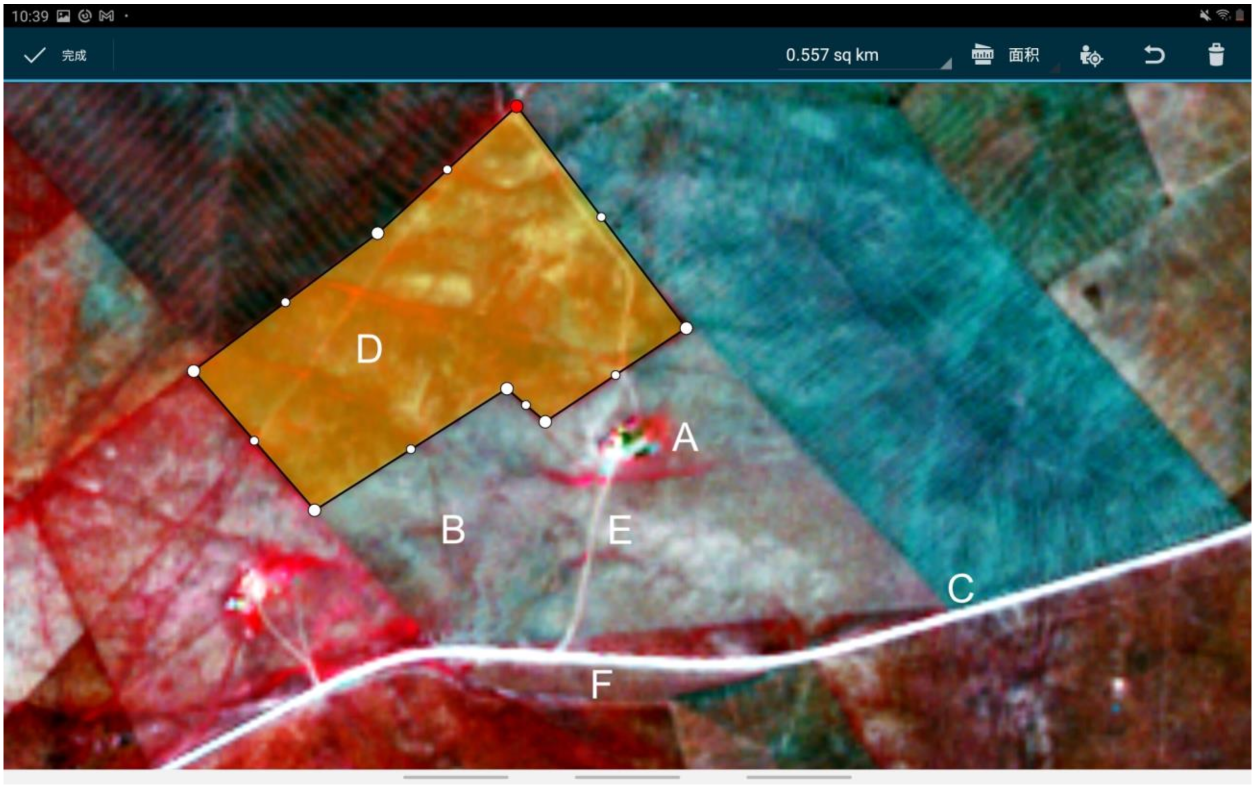


Figure S4. ArcGIS Collector interface and participatory mapping data. During the survey, a blue dot would appear and indicate the real-time location of the survey investigator. All surveys were conducted inside each herder's settlement. Collector's "measurement" tool can be used to measure the geometry of the herder's land (e.g., parcel size and edge length) and the land or herder's spatial relationship to other key features (e.g., distance to road or neighbor's settlement) for instant validation. In this figure, the herders' settlement (A) is located within a summer grazing parcel (B). The settlement is approximately 500 meters north of a concrete road (C), and 150 meters south of the herders' fenced parcel reserved for hay harvesting (D). The motorcycle path (E) goes to the herders' settlement and then passes through both the grazing and hay harvesting parcels. The path parallels the fence and connects to the neighbor's land at the northern corner of the hay harvesting parcel. The size of parcel B is ~60 acres (900 mu), while the size of parcel D is ~70 acres (1050 mu). Both land parcels have irregular shapes but they share the same length of 1000 meters (from northeast to southwest). The herder also recognized a long and narrow fenced parcel (F) that is divided by the road, with a maximum width (from north to south) of approximately 170 meters.

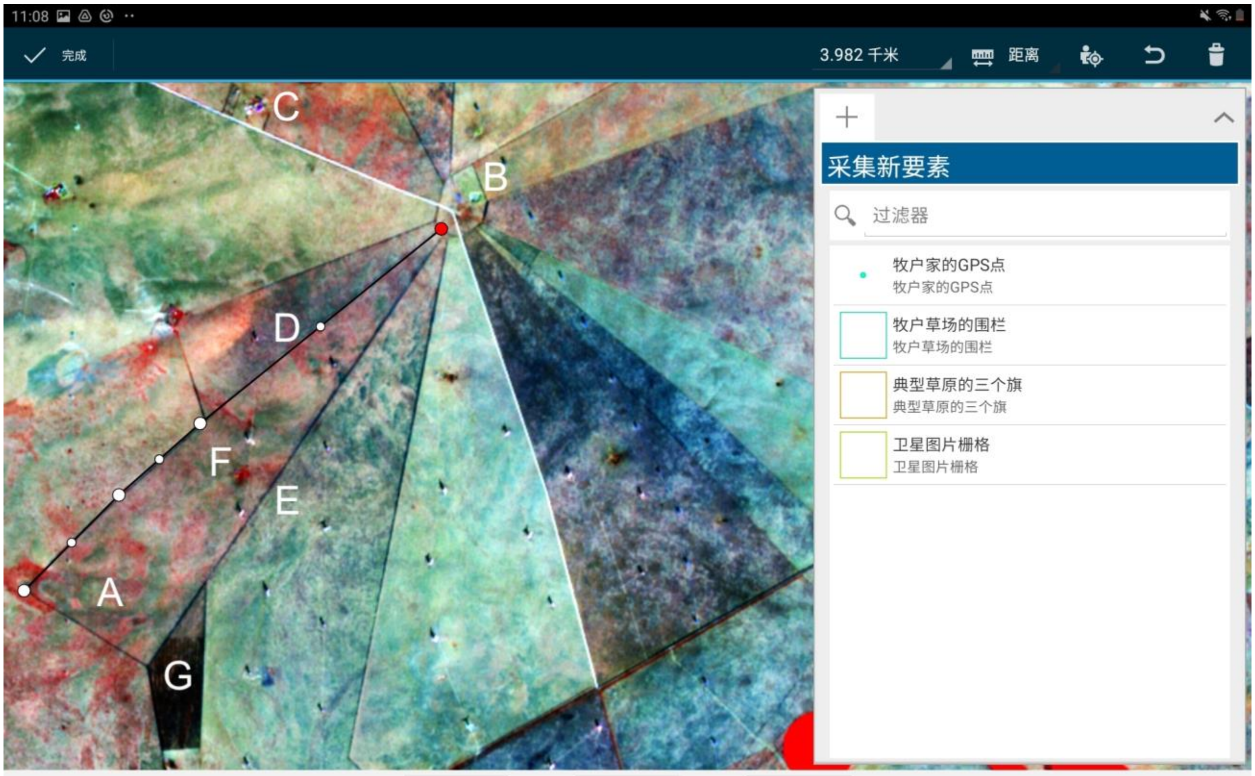


Figure S5. In this figure, our respondent has a wedge-shaped parcel (A) that radiates from the southwest side of the old communal well (B), which is 1.5 km southeast from the herder's settlement (C) where this participatory mapping is conducted. Our ArcGIS Collector measurement shows that the size of this parcel is 546 acres (3314 mu). Our measurement shows that the parcel's upper edge (D) is approximately 4km, and the lower edge (E) is 3.8km. Additionally, three electric towers (F) are located inside this parcel, and one dark-color parcel (G, often is ungrazed and reserved land) is located at the southern end of the parcel, which can all be used as key land features to validate with herders. All validation questions are designed to elicit spatial information and further discussions, instead of merely asking herder to judge whether our measurement is correct. For example, questions were focused on "How big is your land parcel? Can you describe the shape? If counting from the communal well, how long is the fence edge on the north side? Is there any recognizable object in your land? What are they? How are these objects arranged spatially?". We avoid questions with pre-assumption such as "Is your land wedge-shaped? Are there three electric towers on your land?".

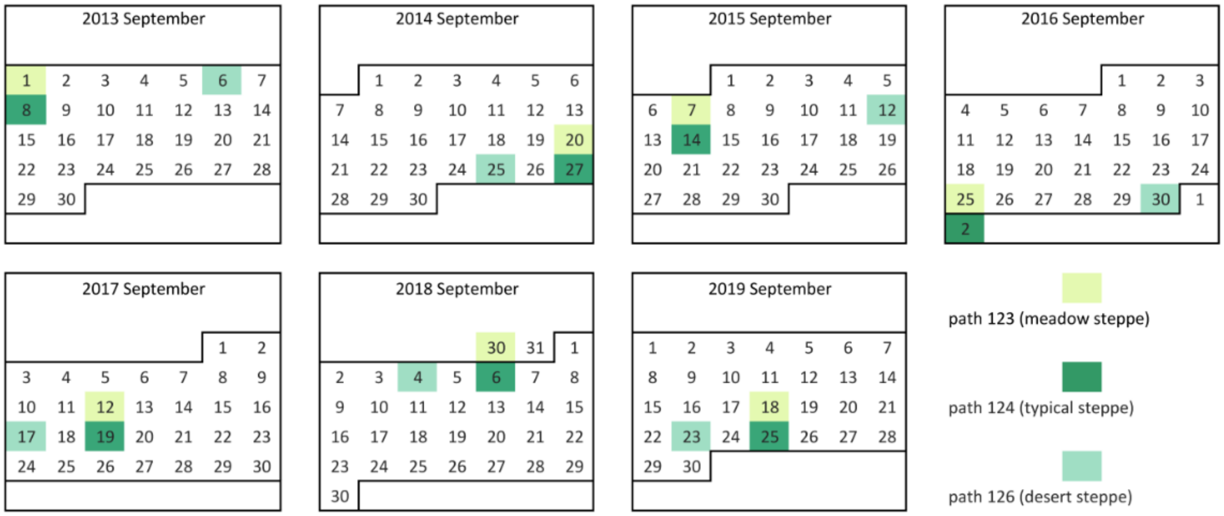


Figure S6. The acquisition date of Landsat-derived vegetation index (2013-2019). Additionally, we extracted the 14-day average value of three abiotic variables (land surface temperature, precipitation, and downward shortwave radiation) of each parcel in advance of the vegetation index retrieval date each year.

Table S1. Township or equivalent administrative units visited in this study.

Administrative unit	
Township 乡 (n=0)	An administrative unit in China, subordinate to county (*banner*)
Sumu 苏木(n=15)	Township equivalent administrative unit specifically used in Inner Mongolia
State ranch 国营牧场 (n=2)	Township equivalent administrative unit established during the collective era (1958-1984).

Table S2. Parcels rented vs. privately held each year. A total of 432 observations are rented and 2130 are privately held (i.e., contracted from the state). Additionally, 238 observations were labeled as having "unknown" rental status and were excluded from the analysis. For example, if a surveyed parcel was rented by a respondent only from 2015-2019 but was not tied to the respondent before 2015, then a parcel's rental status is defined as "unknown" in 2013 and 2014.

Rental Status	2013	2014	2015	2016	2017	2018	2019	Total
Rented	43	47	55	58	70	76	83	432
Contracted	299	301	302	301	307	310	310	2130
Unknown	58	52	43	41	23	14	7	238
Total	400	400	400	400	400	400	400	2800

Table S3. Difference between rental and otor.

	Rental	otor
Duration	Usually 1 to 3 years (in very few cases 10 years or more if the landlord has already migrated to the city and has an agreement with the tenant on how to adjust price*).	Usually 1 to 3 months, can be up to 6 months but rarely over than a year.
Primary purpose	Sustain or enlarge production	Deal with drought-induced forage shortage
Contract formality	Contracts can be formal (approved by the Grassland Supervision Bureau) or informal (personal or oral).	Informal
Payment method	Based on the land size (e.g., 5 yuan per mu)	Based on monthly livestock consumption (e.g., 100 yuan per cattle per month)
Distance traveled	Varies among households, mostly within banner/county	Varies among households
Land use rights	Renters have exclusive use rights	Exclusive land use rights are not guaranteed (respondents sometimes share the land with the land contractor).
History	A new form of land transaction developed in the last decade	Traditional practice that has a much longer history
* Currently, the grassland supervision bureau only approves formal contracts equal to or less than 3 years. Longer contracts (in fixed price) are discouraged because these contracts are subjected to more unpredictable climatic and regional land price changes and therefore induce greater chances of conflicts. Such conflicts may arise when there is a significant drought-induced biomass decline and tenants may request a refund. Similarly, the landlord can also be unsatisfied with the lump-sum payment received in the first year because of the increase in general land market price in the later years.		

Table S4. Land use purposes of all 400 parcels over the 7 years (2800 observations).

Land use purpose	# of observations
All season grazing	969
Hay harvesting	156
Lambing	518
Summer grazing	527
Winter grazing	630
Total	2800

Table S5. Log MSAVI as a function of land tenure, land management, and household-level socioeconomic characteristics in addition to abiotic and topographic influences.

Variables	
Abiotic factors	
Land surface temperature (°C)	-0.00680***
	(0.000826)
Precipitation (mm)	0.00376**
	(0.00143)
Solar radiation (W/m^2)	0.000183**
	(5.59e-05)
Topographic factors	
Elevation (meter)	0.000327***
	(5.11e-05)
Slope (degree)	-0.00514*
	(0.00220)
Aspect northing	0.0317**
	(0.0105)
Land tenure & management	
Rental Status (ref: Privately held)	-0.0239***
Rented	(0.00563)
Stocking rate (sheep/ha)	0.0289*
	(0.0147)
Stocking rate^2	-0.00898*
	(0.00440)
Fodder expenditure (100 yuan/sheep)	0.0164*
	(0.00685)
Fodder expenditure^2	-0.00162
	(0.00135)
Distance to city (10 kilometer)	0.00292*
	(0.00133)
Years of herding (10 years)	-0.00418*
	(0.00206)
Education(years)	6.70e-05
	(0.000725)
Ethnicity (ref: Han)	-0.00301
Mongol	(0.00554)
Gender (ref: Female)	0.0123*
Male	(0.00536)
Household size (# people)	0.00611**
	(0.00198)
Herding labor (# people)	-0.000604
	(0.00330)
Observations	2,279
AIC	-4282.709
BIC	-4162.347
Log likelihood	2162.354

Standard errors in parentheses

*** p<0.001, ** p<0.01, * p<0.05

Table S6. Alternative model specification, substituting 14-day for 7-day abiotic variables.

Variables	
Abiotic factors	
Land surface temperature (°C)	-0.00541***
	(0.000718)
Precipitation (mm)	-0.000412
	(0.00134)
Solar radiation (W/m^2)	5.72e-05
	(3.21e-05)
Topographic factors	
Elevation (meter)	0.000328***
	(5.12e-05)
Slope (degree)	-0.00499*
	(0.00221)
Aspect northing	0.0327**
	(0.0105)
Land tenure & management	
Rental Status (ref: Privately-held)	-0.0234***
Rented	(0.00563)
Stocking rate (sheep/ha)	0.0292*
	(0.0147)
Stocking rate^2	-0.00897*
	(0.00441)
Fodder expenditure (100 yuan/sheep)	0.0159*
	(0.00686)
Fodder expenditure^2	-0.00155
	(0.00135)
Household characteristics	
Distance to city (10 kilometer)	0.00294*
	(0.00134)
Years of herding (10 years)	-0.00439*
	(0.00206)
Education (years)	6.78e-05
	(0.000726)
Ethnicity (ref: Han)	-0.00354
Mongol	(0.00554)
Gender (ref: Female)	0.0127*
Male	(0.00537)
Household size (# people)	0.00587**
	(0.00198)
Herding labor (# people)	-0.000340
	(0.00331)
Observations	2,279
AIC	-4276.753
BIC	-4156.392
Log likelihood	2159.376

Standard errors in parentheses

*** p<0.001, ** p<0.01, * p<0.05

Table S7. Alternative model specification, substituting MSAVI for NDVI.

Variables	
Abiotic factors	
Land surface temperature (°C)	-0.00914***
	(0.00106)
Precipitation (mm)	0.00504**
	(0.00184)
Solar radiation (W/m^2)	0.000254***
	(7.18e-05)
Topographic factors	
Elevation (meter)	0.000398***
	(6.59e-05)
Slope (degree)	-0.00660*
	(0.00283)
Aspect northing	0.0433**
	(0.0135)
Land tenure & management	
Rental Status (ref: Privately held)	-0.0320***
Rented	(0.00723)
Stocking rate (sheep/ha)	0.0388*
	(0.0189)
Stocking rate^2	-0.0121*
	(0.00566)
Fodder expenditure (100 yuan/sheep)	0.0223*
	(0.00880)
Fodder expenditure^2	-0.00234
	(0.00173)
Household characteristics	
Distance to city (10 kilometer)	0.00363*
	(0.00172)
Years of herding (10 years)	-0.00555*
	(0.00265)
Education (years)	-9.46e-05
	(0.000932)
Ethnicity (ref: Han)	-0.00396
Mongol	(0.00712)
Gender (ref: Female)	0.0157*
Male	(0.00689)
Household size (# people)	0.00751**
	(0.00255)
Herding labor (# people)	-0.000143
	(0.00424)
Observations	2,279
AIC	-3139.687
BIC	-3019.326
Log likelihood	1590.844

Standard errors in parentheses
*** p<0.001, ** p<0.01, * p<0.05

Table S8. Alternative model specification, including parcel-based land-use purpose and durations as designated by the herder (i.e., portfolio effect). Land parcels were classified as 1) all-season grazing parcel, 2) hay harvesting field (only used during haying season in late August), 3) lambing parcel (only used 1.5-2 months in the lambing season), 4) summer grazing parcel, and 5) winter grazing parcel. Our result indicates parcels that are used for lambing and winter grazing (which are used in much shorter duration) are greener overall than parcels that are used all year round. This indicates land use purposes and durations have a strong impact on vegetation greenness.

Variables	
Abiotic factors	
Land surface temperature (°C)	-0.00696***
	(0.000812)
Precipitation (mm)	0.00373**
	(0.00141)
Solar radiation (W/m^2)	0.000183***
	(5.50e-05)
Topographic factors	
Elevation (meter)	0.000374***
	(5.11e-05)
Slope (degree)	-0.00513*
	(0.00217)
Aspect northing	0.0329**
	(0.0104)
Land tenure & management	
Rental Status (ref: Privately held)	-0.0118*
Rented	(0.00573)
Stocking rate (sheep/ha)	0.0240
	(0.0145)
Stocking rate^2	-0.00693
	(0.00434)
Fodder expenditure (100 yuan/sheep)	0.0182**
	(0.00676)
Fodder expenditure^2	-0.00246
	(0.00133)
Household characteristics	
Distance to city (10 kilometer)	0.00349**
	(0.00133)
Years of herding (10 years)	-0.00288
	(0.00203)
Education (years)	0.000585
	(0.000721)
Ethnicity (ref: Han)	-0.000211
Mongol	(0.00549)
Gender (ref: Female)	0.0121*
Male	(0.00528)
Household size (# people)	0.00505*
	(0.00196)
Herding labor (# people)	-0.00123
	(0.00325)
Land use (ref: all-season grazing)	
Haying	-0.000126

	(0.0103)
Lambing	0.0436***
	(0.00586)
Summer grazing	-0.00718
	(0.00595)
Winter grazing	0.0209***
	(0.00569)
Observations	2,279
AIC	-4352.772
BIC	-4209.484
Log likelihood	2201.386

Standard errors in parentheses
*** p<0.001, ** p<0.01, * p<0.05

Table S9. Alternative model specification, without all observations in Xilin Hot.

Variables	
Abiotic factors	
Land surface temperature (°C)	-0.00503***
	(0.00121)
Precipitation (mm)	0.00364*
	(0.00167)
Solar radiation (W/m^2)	0.000168*
	(7.34e-05)
Topographic factors	
Elevation (meter)	0.000293***
	(8.00e-05)
Slope (degree)	-0.00705
	(0.00484)
Aspect northing	0.0386**
	(0.0148)
Land tenure & management	
Rental Status (ref: Privately held)	-0.0279***
Rented	(0.00797)
Stocking rate (sheep/hectare)	0.0150
	(0.0272)
Stocking rate^2	-0.00402
	(0.0103)
Fodder expenditure (100 yuan/sheep)	0.00125
	(0.00882)
Fodder expenditure^2	0.00118
	(0.00165)
Household characteristics	
Distance to city (10 kilometer)	0.00247
	(0.00169)
Years of herding (10 years)	-0.00199
	(0.00261)
Education (years)	0.00122
	(0.000972)
Ethnicity (ref: Han)	-0.00838
Mongol	(0.00693)
Gender (ref: Female)	0.0121
Male	(0.00736)
Household size (# people)	0.00723**
	(0.00267)
Herding labor (# people)	-0.00472
	(0.00422)
Observations	1,451
AIC	-2585.243
BIC	-2474.363
Log likelihood	1313.622

Standard errors in parentheses
*** p<0.001, ** p<0.01, * p<0.05

Chapter 5

Discussion and conclusion

5.1 Overview

Using the land rental market in Inner Mongolia as a case study, my dissertation examines the complex relationships between land rights, tenure security, and rangeland resource management. To understand the varying degrees of impacts that land users, institutions, and biophysical factors have on rangeland change, I integrate knowledge from land economics, development geography, and rangeland ecology. Additionally, I adopt methods from geographic information-remote sensing science to analyze the patterns of rangeland biomass change over time and methods from behavioral economics to quantify herders' perceived tenure insecurity. Using detailed data collected through a household survey, I quantified the magnitude of herders' informally managed land and their external forage inputs and discussed the consequences when such land resources were overlooked (**Chapter 2**). I then used a discrete choice experiment to estimate herders' preferences on rental contracts and examined the relative threat of each tenure insecurity factor (**Chapter 3**). Additionally, I designed a novel workflow that integrated micro-level survey data with open-access satellite data, comparing the rangeland outcome of rented versus privately held rangelands (**Chapter 4**). This chapter uses the following three sections (5.1, 5.2, and 5.3) to synthesize each research topic and discuss its original contribution. In the final section (5.4), I will outline the policy recommendations and the priorities for future research.

5.2 Chapter 2: The importance of *de facto* land holding in policy design

Herders' complex *de facto* land holdings – namely via informally managed land and external forage inputs – make payment-based conservation programmes impractical. In pastoral China, the stocking rate is the key metric used to implement and monitor eco-compensation programmes. A household's maximum carrying capacity ($Stocking\ rate_{property}$) is calculated by multiplying the official stocking rate ($Stocking\ rate_{official}$, which is expressed in the unit of sheep/ha and is constant within a village) by the size of the household's land, which can also be written as:

$$Stocking\ rate_{property} = Stocking\ rate_{official} * Land\ size$$

Since the land system of Inner Mongolia is highly formalized, privatized, and well-monitored, it is commonly assumed that we can accurately estimate the amount of land that each household has and, consequently, the carrying capacity of individual households' land. However, my dissertation shows that herders use a considerable amount of land from non-operating land users and manage many of them informally. Herders also use forage produced distantly, which contributes to the number of livestock they can support for a given size of physical land resources. The focus of current eco-compensation programmes is on *de jure* land; and informally managed land and external forage input are not considered as land resources. Chapter 2 shows that the magnitude of this issue is not trivial. As a consequence, herders were compensated less while subject to harsher livestock holding regulations. In the following section, I will discuss another problem with the equation mentioned above and reflect on stocking rate-based policy and scientific research.

5.2.1 *de facto* land and the additional problem of scale

Even if we are able to estimate all *de facto* land accurately, it is still problematic to linearly upscale the official stocking rate to a property level. My dissertation shows that the stocking rate varies depending on whether it is determined by local herders or policymakers. Herders' median expected stocking rate is twice as much as the official stocking rate, suggesting that smallholders think that the official stocking rate is overly restrictive. I argue that the fundamental (yet frequently overlooked) reason for this discrepancy is that the two stocking rates are calculated using vastly different measurements and scales. Despite the fact that the two stocking rates are expressed in the same measurement unit (i.e., livestock per hectare of land), they are inherently incompatible with one another.

The official stocking rate is a measurement of ideal or theoretical grazing pressure. The official stocking rate is often anticipated to be precise, scientific, and objective because it is calculated using plot-based vegetation surveys and pixel-based remote sensing data. Every year, the Inner Mongolia Grassland Survey and Planning Institute conducts a plot-level ecological survey during the month of peak grassland productivity (Ministry of Agriculture and Rural Affairs 2006), see standard *NY/T 1233-2006* for detailed instruction). A one-square-meter plot is sampled for above-ground biomass for each type of grassland, and plot-level NPP is calculated. Using field data, surveyors establish a statistical relationship between plot-based NPP and MODIS NDVI and extrapolate landscape-scale NPP using this relationship (Liu et al. 2003). Similar to what we did in this study, surveyors then use steppe-specific parameters provided by standard *NY/T 635-*

2015 to estimate forage production (and subsequently the theoretical stocking rate) for each village (Ministry of Agriculture and Rural Affairs 2005).

As a proxy to measure rangeland ecological conditions, the official stocking rate is easy for scientists to apprehend and use in their studies. As an administrative unit, the official stocking rate is a convenient standard for bureaucrats to regulate herders' land management behavior and evaluate policy outcomes. However, the official stocking rate has two underlying assumptions that scientists frequently ignore. Firstly, the official stocking rate represents the ideal carrying capacity that policymakers expect. It is important to note that the "proper use factor" used in the calculation (and in this dissertation) only takes the policymakers' definition of "proper use" into account. Secondly, the theoretical stocking rate measurement is conducted in environmental settings free of actual human intervention. The measurement standard *NY/T 635-2015* does not require an ecological survey to be conducted in a grazed plot. Therefore, the calculation of the official stocking rate does not take into account real herders and their actual land management capacity, knowledge, or history. Instead, the official stocking rate is designed based on a fictitious herder, a "stock character, the Everyman cultivator (Scott 1998)" who altruistically prioritizes using land properly and sustainably over all other purposes.

The official stocking rate is a theoretical and ecologically oriented metric. Contrarily, the stocking rate expected by local herders reflects herders' priorities in meeting their actual needs. In contrast to policymakers who prioritize regulating services, herders value provisioning services and the associated economic benefits (Fang and Wu 2022). Both the numerator (livestock) and the denominator (land) in the fraction of herders' expected stocking rate are subject to change due to the herders' actual land management decisions. Herders' land management plans and actions are governed by the complex interactions of social and biophysical factors at multiple levels (Brunson 2012). Herders adjust their livestock, land, and forage in response to the unexpected social and biophysical changes at a local level, such as increasing stocking to pay unexpected medical bills or increasing forage purchases after a severe drought. Broader-level economic opportunities or crises also influence herders' land-use decisions. The fluctuation of mutton prices often influences their decisions to expand or reduce the herds. However, the land users' cultural and socio-economic demands, as well as their management objectives and competencies, are overlooked in the official stocking rate (Harris 2010; Bestelmeyer and Briske 2012).

The measurement of the official and expected stocking rates on their own is not problematic. The problem is that they are frequently compared without taking into account the incompatibility of their scales and measurement methods. It is often assumed that the official stocking rate, which is estimated at the plot- and pixel-level, is scale-independent and can be linearly generalized to the household, property, or even a higher level (Sayre 2017). However, by linearly upscaling the official stocking rate to the level of a household, we magnify the normalized assumptions embedded in this ecology-based metric, creating a sharp contrast when compared to herder's expected stocking rate that reflects the real household concerns. Despite the state's effort in the use and enforcement of eco-compensation policies to help sustainably manage rangelands, eco-compensation has been widely reported to have failed both in terms of implementation and the reduction of stocking rate (Kolås 2014; Yin et al. 2019; Zhang et al. 2019; Fang and Wu 2022). Based on the findings of this dissertation, I argue that the main reason why eco-compensation remains unenforced is that it overlooks the social and market complexities and is overly focused on conservation goals.

5.3 Chapter 3: Land tenure insecurity determinants and trade-offs

In chapter 3, I use a choice experiment to compare the relative importance of the factors that influence tenure security. Various factors affect land users' tenure security, including whether they have a complete bundle of rights and whether those rights are upheld by society. However, the relative importance of each factor has rarely been studied due to the incompatibility of the measurement scales. By offering herders sets of hypothetical land contracts and analyzing their choices, I examine how each factor impacts herders' overall utility (i.e., maximum tenure security). This study shows that formal contracts increase herders' utility, highlighting the importance of having state institutions uphold the contracts. I also find that kinship contracts are preferred but only when they are legally recognized. Finally, when the state restricts land use, herders' utility increases by having the rights to receive compensation. This study acknowledges the local government's effort on formalizing and standardizing land agreements. Additionally, I highlight the significance of and challenges in formalizing kinship contracts and providing renters with rights to compensation. This chapter demonstrates that choice experiments can weigh the significance of each factor that impacts tenure security. By identifying the factors that pose the greatest threat to herders, we can allocate policy and research resources more efficiently.

5.4 Chapter 4: The impact of the rental market on rangeland resource management

How does land tenure change influence land users' management practices? I approach this classic land economics question in this chapter with novel methods. I leverage the recent advancement in remote sensing and geographic information science to obtain temporal-spatial explicit environmental data at the parcel level, which I then combine with socio-economic data obtained from participatory mapping and household surveys. I study the rangeland changes of 400 parcels belonging to 187 households in Xilin Gol from 2013 to 2019, controlling for all abiotic and managerial influences. This study shows that privately held lands have better rangeland outcomes as compared to rented land parcels, suggesting that contracted lands are used less intensively than rented land. I think renters' lack of rights to compensation may explain their intensive land use practices. The following two sections will discuss this study's theoretical and methodological contributions to participatory mapping and "people and pixels" literature.

5.4.1 Participatory mapping: from static and fuzzy to interactive and precise

This study shows that we can leverage the new technologies in geographic information system and remote sensing to obtain micro-level land data and conduct "precise participatory mapping." Precision refers to "the level of spatial resolution, or, to the accuracy and reliability of the geo-referencing" (McCall 2006). The research question determines the degree of spatial precision needed for each study. Studies that locate community conflicts, risks, and resources through local knowledge often have lower requirements on spatial precision (both in terms of map details and accuracies of geo-referencing), while studies that identify land and resource titles have higher requirements (McCall 2006).

Some participatory methods, such as geo-referenced sketch map or participatory three-dimensional modeling (Cadag and Gaillard 2012; Shrestha and Medley 2016) have been widely used to acquire abstract perceptions or knowledge. However, such methods appear insufficient when our research attention shifted to explicit land features such as parcel boundaries. The primary barriers to detecting fine land features using conventional approaches are the static temporal and spectral resolution and the fixed map extent provided by most aerial base maps. Although the free base maps provided by Google Maps and Bing Maps have a high spatial resolution for much of the Earth's surface, most of these easy-access images are static in time. Although images are updated more frequently in urban areas, images taken in rural areas may have been captured several years ago, and land entitlement and land use may have changed since then. Since map services often collage pixels from multiple dates or years to create cloud-free images, such images even lose their temporal information completely.

Moreover, base maps used in participatory mapping are often preset to the RGB band combination to mimic natural color seen by humans. Such "real color" satellite imagery may help respondents differentiate simple land cover types such as forest or bare land, but often fail in other circumstances and more sophisticated applications (e.g., identifying features that are smaller than a pixel, such as the wire fences in our case). In addition to the static temporal and spectral resolution, map extent is un-adjustable too. Typical base maps that are presented to the participants are often printed out at a fixed extent (usually at a village level). Village-level maps allow more villages to participate because they cover larger areas at the expense of household-level details. Due to the unscalable characteristic of printed maps, rural participatory mapping is often constrained in the form of focus groups and is limited by investigating village-level questions.

In this study, I developed a novel method to conduct "precise participatory mapping" using customized, interactive, and portable satellite base maps in a tablet computer that supports real-time GPS tracking. Firstly, the open-access Sentinel-2 satellite offers a 2-5 days temporal revisit, allowing us to reveal land-cover changes that happened in recent weeks or even days in the base map. I differentiated parcel boundaries by taking advantage of the non-RGB bands (i.e., near-infrared and shortwave infrared) that the multi-spectral sensor provided. This study shows that a satellite product that combines the right *temporal* resolution and *spectral* range is more useful in identifying specific features as compared to "crisp" satellite images which only have high spatial resolution. Secondly, by using zoomable and interactive maps, I provide herders opportunities to explore and identify spatial relationships at multiple scales. Herders can easily examine their (parcels') distance to the village center or the nearest mountain (in several kilometers), to their neighbor's property (in hundred meters), and to a well or path on their own property (in meters). Other than distance, interactive maps allow us to observe and measure the land features' geometry and spatial relationships and validate with herders. The verification and triangulation of multiple spatial relationships have allowed us to geo-referenced herders' 3-dimensional mental map onto a 2-dimensional satellite image, thus making "precise participatory mapping" possible. Thirdly, by making the base maps compatible with a 10-inch portable mobile device, we can conduct land mapping with each herder inside their settlement, improving herders' confidentiality and comfortability.

5.4.2 Social-ecological data integration at parcel-level

Using these "cookie-cutting" parcel polygons as basic units, I developed a replicable workflow to integrate remote sensing data with micro-level household survey data. From a social science perspective, such an integrated approach "pixelized the social" - our remote sensing time series quantify the *consequences* of individual herders' land decisions (Geoghegan et al. 1998). Although such consequences can also be measured through the conventional field-based, plot-size ecological analysis, remote sensing has broadened the spatial and temporal extent of the measurements (Turner 2003). This study has shown that by using cloud-based geospatial datasets, we are able to extend the observation window from plot to parcel and over the past seven years. Other than the increased extent, the availability of high temporal-spatial resolution satellite imagery also refines the grains of the assessment. This research benefits significantly from the frequent temporal revisit of the sensors, which allows us to monitor nuance land change within each parcel by pairing vegetation indices with the abiotic changes that happened in the previous week. Such temporal-spatially explicit data is exceptionally important in rangeland studies because rangeland ecosystems are innately heterogeneous and constantly changing in response to abiotic influences and disturbances (Fuhlendorf et al. 2017). In sum, using Wiens' analogy of a sieve to explain scale (Wiens 1989), continuous and finer-resolution satellite data allows us to enlarge the diameter of our observation sieve (the measurement extent) while simultaneously reducing the size of the mesh (the temporal-spatial grain). Additionally, such data integration allows us to examine the multiple *causes* of land change comprehensively. Conventional rangeland experiments are conducted in controlled plots, where all human interventions or management practices are simplified as one variable, grazing intensity. Such ecology-centric experiments frequently ignore the complex underlying factors that drive the intensified land-use practices (Harris 2010; Sayre et al. 2012) and are often used as evidence for enclosure-oriented policies that label herders as irrational and exploitative land managers. Interdisciplinary social-ecological studies stress that the underlying drivers of land-use change are often complex and context-dependent (Lambin et al. 2001; Turner, Lambin, and Reenberg 2007). Land users' socio-economic situations and resource institutions significantly impact how they respond to external economic opportunities and make land management decisions (Lambin et al. 2001; Brunson 2012; Hopping et al. 2018). Land economics studies can capture the complex socio-economic, management, and institutional variables through detailed household surveys; however, household surveys cannot quantify the environmental outcomes and often have to rely on "poor and binary" proxy (e.g., fallow and building terrace) on herder's management activities (Fenske 2011; Carter and Yao 2002). Using individual parcels as a medium, I demonstrate that the complex social variables that determine land use behavior can be associated with its environmental outcomes, allowing us to investigate the proximate and underlying causes through more comprehensive models.

The integration of social and environmental data at the household level is needed to understand human-environment dynamics since most land-use decisions are made at the household level (Rindfuss et al. 2004). At the household level, we can observe how land decisions directly influence ecosystem functions and processes and how land users react and adapt to the changing environment. For example, when a severe drought occurs and reduces forage production, herders often respond immediately by purchasing feed and fodder or migrating their livestock temporarily, which is faster and more efficient than broader-scale social organizations. Additionally, data integration at the household level allows us to investigate the complexities and heterogeneities among villages and households, which is often masked in aggregated level analysis (Lambin et al. 2009). Such a disaggregated study can better tailor environmental policy suggestions to the specific local context. The integration of remote sensing and social data at the household level was long considered challenging in practice due to issues such as the mismatch of social and satellite data and the complex "one-to-many" or "many-to-one" household-land relationships (Rindfuss et al. 2004). It has been (and is still) challenging to accurately identify the boundaries of land parcels and logically link these parcels back to individual households within a reasonable time and research budget (Rindfuss et al. 2004; Turner and Geoghegan 2004). This study thus contributes to knowledge by designing a novel workflow that delineates parcel boundaries and integrates social and satellite data at the parcel level.

It has been over 20 years since the concept of social-ecological data integration was first comprehensively discussed in the book "People and pixels: linking remote sensing and social science" (National Research Council 1998). There have been rapid advances in remote sensing and GIS technologies during the past 20 years. The major advances include the increased accessibility of satellite data at finer temporal-spatial resolution; the advancement of machine learning classification methods; and the availability of cloud platforms that allow easy query and analysis of big data (Dong et al. 2019; Zhu et al. 2019). The increasing open access to geospatial data and analysis tools has also made our study possible.

This study shows that as remote sensing technologies become more accessible, land users have a greater potential to participate in "people and pixels" research. Before conducting participatory mapping, I anticipated some herders might be concerned or feel insecure because their property was being monitored. However, I did not see any herders expressing their displeasure. Instead, most herders were interested in engaging in mapping activities. Some herders expressed surprise when seeing their land boundaries, and some investigated their neighbours' and nearby lands and resources. Some herders were curious about how such maps were made and took cellphone photos of their land. One important observation from this study is that herders now have greater digital and map literacy due to improved internet access. And as this study and a plethora of literature on participatory mapping have demonstrated, land users are curious about maps and willing to share their spatial and environmental knowledge. Therefore, future research can focus on facilitating land users to share micro-level and spatial-explicit information (i.e., "linking people *with* pixels").

5.5 Future research

The previous sections synthesize each chapter and highlight its contribution to knowledge. Based the findings of this dissertation, I present a set of recommendations and potential research topics for practitioners and researchers.

5.5.1 Design socially meaningful metrics

This study demonstrates that assumptions that simplify and romanticize smallholders and their land management practices lead to reductionist research and impractical policies, which, if implemented coercively, might undermine smallholder livelihoods. Future research can further investigate and understand herders' complex land tenure, land rights, and land use practices. Future experiments and socio-ecological models should take into account herders' management practices and the underlying social drivers (Sayre et al. 2012), as well as herders' local environmental knowledge (Reynolds et al. 2007). Scientists should think critically about the official stocking rate and be cautious of accepting it unconditionally as a baseline of how much livestock a household should keep. Scientists should also be aware if their study reinforces the pessimistic degradation narrative, which disempowers local herders and rationalizes top-down coercive policies.

5.5.2 *de facto* land and rights to compensation are important

Household-based eco-compensation programmes should assess *ex-ante* the size of the land rental market and the cost (affordability) of tracking and registering all land transactions. So far, PES only incentivizes *de jure* landholders. The expansion of the rental market inevitably results in an increase of non-operating *de jure* landholders and the outflow of eco-compensation, which poses a critical challenge for resource management. Even if payment-based programmes figure out a way to incentivize *de facto* landholders, this study shows that rental transactions' complexity and informality may hamper the implementation of the programmes. The case study of Inner Mongolia demonstrates that although the state has sufficient expenditure to register rangelands (twice), the flexible and largely informal land transactions still lie beyond the purview of its regulation. Household-targeted, land-based eco-compensation or PES programmes should only be implemented when all land transactions are formalized and herders' external forage inputs can be reasonably estimated. Suppose such conditions cannot be met (which is a common scenario for states with limited fiscal resources); in that case, policymakers should explore the feasibility of informal or semi-formal alternatives such as co-management of local communities and authorities. Nevertheless, elite manipulation, nepotism, and corruption can still be common issues for communities with weak governance capability.

5.5.3 Recognize herders as important stakeholders

The carrying capacity-based eco-compensation is a conservation-oriented policy that prioritizes enhancing and "restoring" rangelands' regulating services. The improvement of regulating services is unavoidably at the cost of rangelands' provisioning services and herders' economic benefits and wellbeing. Therefore, it is necessary to assess if such trade-offs are acceptable to herders and whether the compensation is sufficient. Can herders actually make a living if they strictly comply with the official stocking rate? What is a feasible "proper use factor" from the herders' perspective? How do herders adjust their herds in response to the fluctuating price of mutton and forage, as well as the unpredictable change in precipitation? How can informal and formal institutions improve to help herders buffer the market and biophysical risks? All these policy-related questions can only be answered when herders are included as stakeholders.

5.5.4 Link people *with* pixels

As discussed earlier, participatory mappings often have a low requirement for spatial precision. Quantitative social research improves spatial precision by using GPS coordinates to represent land users and their properties but often overlooks land users' engagement. Our study demonstrates that "precise participatory mapping," or allowing herders to share spatially explicit land information (e.g., land rights and land use of a given parcel), is possible. However, the maps must be scalable and customized to their individual properties. One exciting direction for future micro-level "people and pixels" research, in my opinion, is to design (land) user-friendly maps using existing data and technologies to acquire precise spatial information on land use and land rights. Such design will encourage herders' active participation and knowledge sharing, allowing us to improve socio-ecological models and better understand the roles of human in environmental change.

References

Abbott, Kenneth W., Robert O. Keohane, Andrew Moravcsik, Anne-Marie Slaughter, and Duncan Snidal. 2000. “The Concept of Legalization.” *International Organization* 54 (3): 401–19. https://doi.org/10.1162/002081800551271.

Adamowicz, W., P. Boxall, M. Williams, and J. Louviere. 1998. “Stated Preference Approaches for Measuring Passive Use Values: Choice Experiments and Contingent Valuation.” *American Journal of Agricultural Economics* 80 (1): 64–75. https://doi.org/10.2307/3180269.

Arnot, Chris D, Martin K Luckert, and Peter C Boxall. 2011. “What Is Tenure Security? Conceptual Implications for Empirical Analysis.” *Land Economics* 87 (2): 297–311. https://doi.org/10.3368/le.87.2.297.

Bai, Yongfei, and M. Francesca Cotrufo. 2022. “Grassland Soil Carbon Sequestration: Current Understanding, Challenges, and Solutions.” *Science* 377 (6606): 603–8. https://doi.org/10.1126/science.abo2380.

Bandiera, Oriana. 2007. “Land Tenure, Investment Incentives, and the Choice of Techniques: Evidence from Nicaragua.” *World Bank Economic Review* 21 (3): 487–508. https://doi.org/10.1093/wber/lhm005.

Banks, Tony. 2001. “Property Rights and the Environment in Pastoral China: Evidence from the Field.” *Development and Change* 32 (4): 717–40.

———. 2003. “Property Rights Reform in Rangeland China: Dilemmas on the Road to the Household Ranch.” *World Development* 31 (12): 2129–42. https://doi.org/10.1016/j.worlddev.2003.06.010.

Behnke, Roy H. 2008. “The Drivers of Fragmentation in Arid and Semi-Arid Landscapes.” In *Fragmentation in Semi-Arid and Arid Landscapes: Consequences for Human and Natural Systems*, edited by Kathleen A. Galvin, Robin S. Reid, Roy H Behnke, and N. Thompson Hobbs, 305–40. Dordrecht, The Netherlands: Springer.

Behnke, Roy, and Michael Mortimore. 2016. “Introduction: The End of Desertification?” In *The End of Desertification? Disputing Environmental Change in the Drylands*, edited by Roy H Behnke and Michael Mortimore, 1–34. Heidelberg New York: Springer.

Bengtsson, J., J. M. Bullock, B. Egoh, C. Everson, T. Everson, T. O’Connor, P. J. O’Farrell, H. G. Smith, and R. Lindborg. 2019. “Grasslands-More Important for Ecosystem Services than You Might Think.” *Ecosphere* 10 (2): e02582. https://doi.org/10.1002/ecs2.2582.

Bestelmeyer, Brandon T, and David D Briske. 2012. “Grand Challenges for Resilience-Based Management of Rangelands.” *Rangeland Ecology and Management* 65 (6): 654–63. https://doi.org/10.2111/REM-D-12-00072.1.

Bigelow, Daniel, Allison Borchers, and Todd Hubbs. 2016. “U.S. Farmland Ownership, Tenure, and Transfer.” August. *United States Department of Agriculture (USDA)*. http://www.ers.usda.gov/publications/eib-economic-information-bulletin/eib-161.aspx.

Bremer, Leah L, Kathleen A Farley, and David Lopez-Carr. 2014. “What Factors Influence Participation in Payment for Ecosystem Services Programs? An Evaluation of Ecuador’s SocioPáramo Program.” *Land Use Policy* 36: 122–33. https://doi.org/10.1016/j.landusepol.2013.08.002.

Briske, D. D., S. D. Fuhlendorf, and F. E. Smeins. 2003. "Vegetation Dynamics on Rangelands: A Critique of the Current Paradigms." *Journal of Applied Ecology* 40 (4): 601–14. https://doi.org/10.1046/j.1365-2664.2003.00837.x.

Briske, David D. 2017. "Rangeland Systems: Foundation for a Conceptual Framework." In *Rangeland Systems: Processes, Management and Challenges.*, 1–24. Springer Series on Environmental Management. Switzerland: Springer Nature.

Briske, David D., D. Layne Coppock, Andrew W. Illius, and Samuel D. Fuhlendorf. 2020. "Strategies for Global Rangeland Stewardship: Assessment through the Lens of the Equilibrium–Non-Equilibrium Debate." *Journal of Applied Ecology* 57 (6): 1056–67. https://doi.org/10.1111/1365-2664.13610.

Briske, David D, Andrew W Illius, and J Marty Anderies. 2017. "Nonequilibrium Ecology and Resilience Theory." In *Rangeland Systems: Processes, Management and Challenges*, edited by David D. Briske, 197–227. Springer Series on Environmental Management. Switzerland: Springer Nature.

Briske, David D., Linda A. Joyce, H. Wayne Polley, Joel R. Brown, Klaus Wolter, Jack A. Morgan, Bruce A. McCarl, and Derek W. Bailey. 2015. "Climate-Change Adaptation on Rangelands: Linking Regional Exposure with Diverse Adaptive Capacity." *Frontiers in Ecology and the Environment* 13 (5): 249–56. https://doi.org/10.1890/140266.

Brunson, Mark W. 2012. "The Elusive Promise of Social-Ecological Approaches to Rangeland Management." *Rangeland Ecology and Management* 65 (6): 632–37. https://doi.org/10.2111/REM-D-11-00117.1.

Cadag, Jake Rom D., and J. C. Gaillard. 2012. "Integrating Knowledge and Actions in Disaster Risk Reduction: The Contribution of Participatory Mapping." *Area* 44 (1): 100–109. https://doi.org/10.1111/j.1475-4762.2011.01065.x.

Carter, Michael R, and Yang Yao. 2002. "Local versus Global Separability in Agricultural Household Models: The Factor Price Equalization Effect of Land Transfer Rights." *American Journal of Agricultural Economics* 84 (3): 702–15.

Channan, S, K Collins, and W.R Emanuel. 2014. "Global Mosaics of the Standard MODIS Land Cover Type Data." University of Maryland and the Pacific Northwest National Laboratory, College Park, Maryland, USA.

Choice Metrics. 2018. "Ngene 1.2 User Manual & Reference Guide." Australia.

Cranford, Matthew, and Susana Mourato. 2014. "Credit-Based Payments for Ecosystem Services: Evidence from a Choice Experiment in Ecuador." *World Development* 64: 503–20. https://doi.org/10.1016/j.worlddev.2014.06.019.

Crook, David R., Brian E. Robinson, and Ping Li. 2020. "The Impact of Snowstorms, Droughts and Locust Outbreaks on Livestock Production in Inner Mongolia: Anticipation and Adaptation to Environmental Shocks." *Ecological Economics* 177. https://doi.org/10.1016/j.ecolecon.2020.106761.

Daily, Gretchen C. 1997. "Introduction: What Are Ecosystem Services?" In *Nature's Services: Societal Dependence on Natural Ecosystems,* edited by Gretchen C Daily, 1–10. Washington DC: Island Press.

Dass, Pawlok, Benjamin Z. Houlton, Yingping Wang, and David Warlind. 2018. "Grasslands May Be More Reliable Carbon Sinks than Forests in California." *Environmental Research Letters* 13 (7): 074027. https://doi.org/10.1088/1748-9326/aacb39.

Deininger, Klaus, and Songqing Jin. 2009. “Securing Property Rights in Transition: Lessons from Implementation of China’s Rural Land Contracting Law.” *Journal of Economic Behavior and Organization* 70 (1–2): 22–38. https://doi.org/10.1016/j.jebo.2009.01.001.

Dell, Randy. 2019. “Non-Operating Landowners and Conservation on Rented Farmland.” The Nature Conservancy. https://www.nature.org/content/dam/tnc/nature/en/documents/NOLS-non-operating-landowners-final.pdf.

Dong, Jinwei, Graciela Metternicht, Patrick Hostert, Rasmus Fensholt, and Rinku Roy Chowdhury. 2019. “Remote Sensing and Geospatial Technologies in Support of a Normative Land System Science: Status and Prospects.” *Current Opinion in Environmental Sustainability* 38: 44–52. https://doi.org/10.1016/j.cosust.2019.05.003.

D’Ottavio, P., M. Francioni, L. Trozzo, E. Sedić, K. Budimir, P. Avanzolini, M. F. Trombetta, C. Porqueddu, R. Santilocchi, and M. Toderi. 2018. “Trends and Approaches in the Analysis of Ecosystem Services Provided by Grazing Systems: A Review.” *Grass and Forage Science* 73 (1): 15–25. https://doi.org/10.1111/gfs.12299.

Dougill, Andrew J., Lindsay C. Stringer, Julia Leventon, Mike Riddell, Henri Rueff, Dominick V. Spracklen, and Edward Butt. 2012. “Lessons from Community-Based Payment for Ecosystem Service Schemes: From Forests to Rangelands.” *Philosophical Transactions of the Royal Society B: Biological Sciences* 367 (1606): 3178–90. https://doi.org/10.1098/rstb.2011.0418.

Du, Shuai, Hao Liu, Sihan You, Gentu Ge, and Yushan Jia. 2019. “Effects of Baling Density and Stacking Method on Chemical Composition and Mycotoxin Levels of Native Hay.” *Acta Prataculturae Sinica* 28 (7): 143–50.

Dyksterhuis, E. J. 1949. “Condition and Management of Range Land Based on Quantitative Ecology.” *Journal of Range Management* 2 (3): 104. https://doi.org/10.2307/3893680.

Ellis, James E. 1994. “Climate Variability and Complex Ecosystem Dynamics: Implications for Pastoral Development.” In *Living with Uncertainty: New Directions in Pastoral Development in Africa*, edited by Ian Scoones, 37–46. Northern Yorkshire, UK: Intermediate Technology Publications.

Ellis, James E., and David A. Swift. 1988. “Stability of African Pastoral Ecosystems: Alternate Paradigms and Implications for Development.” *Rangeland Ecology & Management* 41 (6): 450–59.

Engel, Stefanie, Stefano Pagiola, and Sven Wunder. 2008. “Designing Payments for Environmental Services in Theory and Practice: An Overview of the Issues.” *Ecological Economics* 65 (4): 663–74. https://doi.org/10.1016/j.ecolecon.2008.03.011.

Fan, J. W., K. Wang, W. Harris, H. P. Zhong, Z. M. Hu, B. Han, W. Y. Zhang, and J. B. Wang. 2009. “Allocation of Vegetation Biomass across a Climate-Related Gradient in the Grasslands of Inner Mongolia.” *Journal of Arid Environments* 73 (4–5): 521–28. https://doi.org/10.1016/j.jaridenv.2008.12.004.

Fang, Xuening, and Jianguo Wu. 2022. “Causes of Overgrazing in Inner Mongolian Grasslands: Searching for Deep Leverage Points of Intervention.” *Ecology and Society* 27 (1). https://doi.org/10.5751/ES-12878-270108.

Farr, Tom G., Paul A. Rosen, Edward Caro, Robert Crippen, Riley Duren, Scott Hensley, Michael Kobrick, et al. 2007. “The Shuttle Radar Topography Mission.” *Reviews of Geophysics* 45 (2). https://doi.org/10.1029/2005RG000183.

Feng, Gang, Hui Yan, and Xueting Yang. 2019. "Climate and Food Diversity as Drivers of Mammal Diversity in Inner Mongolia." *Ecology and Evolution* 9 (4): 2142–48. https://doi.org/10.1002/ece3.4908.

Fensholt, Rasmus, Inge Sandholt, and Michael Schultz Rasmussen. 2004. "Evaluation of MODIS LAI, FAPAR and the Relation between FAPAR and NDVI in a Semi-Arid Environment Using in Situ Measurements." *Remote Sensing of Environment* 91 (3–4): 490–507. https://doi.org/10.1016/j.rse.2004.04.009.

Fenske, James. 2011. "Land Tenure and Investment Incentives: Evidence from West Africa." *Journal of Development Economics* 95 (2): 137–56. https://doi.org/10.1016/j.jdeveco.2010.05.001.

Fernandez-Gimenez, Maria E. 2002. "Spatial and Social Boundaries and the Paradox of Pastoral Land Tenure: A Case Study from Postsocialist Mongolia." *Human Ecology* 30 (1): 49–78. https://doi.org/10.1023/A:1014562913014.

Fernandez-Gimenez, Maria E, and Barbara Allen-Diaz. 1999. "Testing a Non-Equilibrium Model of Rangeland Vegetation Dynamics in Mongolia." *Journal of Applied Ecology* 36 (6): 871–85. https://doi.org/10.1046/j.1365-2664.1999.00447.x.

Fernández-Giménez, María E., Baival Batkhishig, Batjav Batbuyan, and Tungalag Ulambayar. 2015. "Lessons from the Dzud: Community-Based Rangeland Management Increases the Adaptive Capacity of Mongolian Herders to Winter Disasters." *World Development* 68 (1): 48–65. https://doi.org/10.1016/j.worlddev.2014.11.015.

Food and Agriculture Organization. 2002. *Land Tenure and Rural Development*. Rome. https://www.fao.org/3/Y4307E/y4307e00.htm.

Fuhlendorf, Samuel D., David D. Briske, and Fred E. Smeins. 2001. "Herbaceous Vegetation Change in Variable Rangeland Environments: The Relative Contribution of Grazing and Climatic Variability." *Applied Vegetation Science* 4 (2): 177–88. https://doi.org/10.1111/j.1654-109X.2001.tb00486.x.

Fuhlendorf, Samuel D., Richard WS Fynn, Devan Allen McGranahan, and Dirac Twidwell. 2017. "Heterogeneity as the Basis for Rangeland Management." In *Rangeland Systems:Processes, Management and Challenges*, edited by David D. Briske, 169–96. Springer Series on Environmental Management. Switzerland: Springer Nature.

Funk, Chris, Pete Peterson, Martin Landsfeld, Diego Pedreros, James Verdin, Shraddhanand Shukla, Gregory Husak, et al. 2015. "The Climate Hazards Infrared Precipitation with Stations - A New Environmental Record for Monitoring Extremes." *Scientific Data* 2: 1–21. https://doi.org/10.1038/sdata.2015.66.

Gao, Liangliang, Jikun Huang, and Scott Rozelle. 2012. "Rental Markets for Cultivated Land and Agricultural Investments in China." *Agricultural Economics* 43 (4): 391–403. https://doi.org/10.1111/j.1574-0862.2012.00591.x.

Gebru, Menasbo, Stein T. Holden, and Mesfin Tilahun. 2019. "Tenants' Land Access in the Rental Market: Evidence from Northern Ethiopia." *Agricultural Economics* 50 (3): 291–302. https://doi.org/10.1111/agec.12484.

Geoghegan, Jacqueline, L Pritchard, Yelena Ogneva-Himmelberger, R.Roy Chowdhury, Steven Sanderson, and B.L.I Turner. 1998. "'Socializing the Pixel' and 'Pixelizing the Social' in Land-Use and Land-Cover Change." In *People and Pixels: Linking Remote Sensing and Social Science*, edited by Diana Liverman, Emilio F. Moran, Ronald R. Rindfuss, and Paul C. Stern. Washington, DC: National Academy Press.

Geussens, K, Goedele Van den Broeck, K Vanderhaegen, B Verbist, and M Maertens. 2019. “Land Use Policy Farmers’s Perspectives on Payments for Ecosystem Services in Uganda.” *Land Use Policy* 84: 316–27. https://doi.org/10.1016/j.landusepol.2019.03.020.
Gorelick, Noel, Matt Hancher, Mike Dixon, Simon Ilyushchenko, David Thau, and Rebecca Moore. 2017. “Google Earth Engine: Planetary-Scale Geospatial Analysis for Everyone.” *Remote Sensing of Environment* 202: 18–27. https://doi.org/10.1016/j.rse.2017.06.031.
Grieg-Gran, Maryanne, Ina Porras, and Sven Wunder. 2005. “How Can Market Mechanisms for Forest Environmental Services Help the Poor? Preliminary Lessons from Latin America.” *World Development* 33 (9): 1511–27. https://doi.org/10.1016/j.worlddev.2005.05.002.
Hardin, Garrett. 1968. “The Tragedy of the Commons.” *Science* 162 (3859): 1243–48. https://doi.org/10.1126/science.162.3859.1243.
Harris, R.B. 2010. “Rangeland Degradation on the Qinghai-Tibetan Plateau: A Review of the Evidence of Its Magnitude and Causes.” *Journal of Arid Environments* 74 (1): 1–12. https://doi.org/10.1016/j.jaridenv.2009.06.014.
Ho, Peter. 2001. “Rangeland Degradation in North China Revisited? A Preliminary Statistical Analysis to Validate Non-Equilibrium Range Ecology.” *Journal of Development Studies* 37 (3): 99–133. https://doi.org/10.1080/00220380412331321991.
Hobbs, N. Thompson, Kathleen A. Galvin, Chris J. Stokes, Jill M. Lackett, Andrew J. Ash, Randall B. Boone, Robin S. Reid, and Philip K. Thornton. 2008. “Fragmentation of Rangelands: Implications for Humans, Animals, and Landscapes.” *Global Environmental Change* 18 (4): 776–85. https://doi.org/10.1016/j.gloenvcha.2008.07.011.
Holden, Stein, and Hosa’ena Ghebru. 2006. “Kinship,Transaction Costs and Land Rental Market Participation.” Working Paper, Norwegian University of Life Sciences, Ås.
Holden, Stein T, and Hosaena Ghebru. 2016. “Land Rental Market Legal Restrictions in Northern Ethiopia.” *Land Use Policy* 55: 212–21. https://doi.org/10.1016/j.landusepol.2016.04.006.
Holden, Stein T, Keijiro Otsuka, and Frank M Place. 2010. *The Emergence of Land Markets in Africa*. *The Emergence of Land Markets in Africa*. Washington DC: Resources for the Future.
Hopping, Kelly A., Emily T. Yeh, Gaerrang, and Richard B. Harris. 2018. “Linking People, Pixels, and Pastures: A Multi-Method, Interdisciplinary Investigation of How Rangeland Management Affects Vegetation on the Tibetan Plateau.” *Applied Geography* 94: 147–62. https://doi.org/10.1016/j.apgeog.2018.03.013.
Hudson, Tipton D., Matthew C. Reeves, Sonia A. Hall, Georgine G. Yorgey, and J. Shannon Neibergs. 2021. “Big Landscapes Meet Big Data: Informing Grazing Management in a Variable and Changing World.” *Rangelands* 43 (1): 17–28. https://doi.org/10.1016/j.rala.2020.10.006.
Huete, A. R. 1988. “A Soil-Adjusted Vegetation Index (SAVI).” *Remote Sensing of Environment* 25 (3): 295–309. https://doi.org/10.1016/0034-4257(88)90106-X.
Hui, Dafeng, and Robert B. Jackson. 2006. “Geographical and Interannual Variability in Biomass Partitioning in Grassland Ecosystems: A Synthesis of Field Data.” *New Phytologist* 169 (1): 85–93. https://doi.org/10.1111/j.1469-8137.2005.01569.x.
Huntsinger, Lynn, Larry C. Forero, and Adriana Sulak. 2010. “Transhumance and Pastoralist Resilience in the Western United States.” *Pastoralism: Research, Policy and Practice* 1 (1): 9–36. https://doi.org/10.3362/2041-7136.2010.002.

Hynes, Stephen, Wenting Chen, Kofi Vondolia, Claire Armstrong, and Eamonn O'Connor. 2021. "Valuing the Ecosystem Service Benefits from Kelp Forest Restoration: A Choice Experiment from Norway." *Ecological Economics* 179 (2019): 106833. https://doi.org/10.1016/j.ecolecon.2020.106833.

Illius, A. W., and T. G. O'Connor. 1999. "On the Relevance of Nonequilibrium Concepts to Arid and Semiarid Grazing Systems." *Ecological Applications* 9 (3): 798–813. https://doi.org/10.1890/1051-0761(1999)009[0798:OTRONC]2.0.CO;2.

Illius, A W, and T.G. O'Connor. 2000. "Resource Heterogeneity and Ungulate Population Dynamics." *Oikos* 89 (2): 283–94. https://doi.org/10.1034/j.1600-0706.2000.890209.x.

Jimoh, Saheed Olaide, Ping Li, Wenqiang Ding, and Xiangyang Hou. 2021. "Socio-Ecological Factors and Risk Perception of Herders Impact Grassland Rent in Inner Mongolia, China." *Rangeland Ecology and Management* 75: 68–80. https://doi.org/10.1016/j.rama.2020.12.001.

Kabubo-Mariara, Jane. 2007. "Land Conservation and Tenure Security in Kenya: Boserup's Hypothesis Revisited." *Ecological Economics* 64 (1): 25–35. https://doi.org/10.1016/j.ecolecon.2007.06.007.

Karnieli, A, Y Bayarjargal, M Bayasgalan, B Mandakh, Ch Dugarjav, J Burgheimer, S Khudulmur, S N Bazha, and P D Gunin. 2013. "Do Vegetation Indices Provide a Reliable Indication of Vegetation Degradation? A Case Study in the Mongolian Pastures." *International Journal of Remote Sensing* 34 (17): 6243–62. https://doi.org/10.1080/01431161.2013.793865.

Kolås, Åshild. 2014. "Degradation Discourse and Green Governmentality in the Xilinguole Grasslands of Inner Mongolia." *Development and Change* 45 (2): 308–28. https://doi.org/10.1111/dech.12077.

Korf, Benedikt, Tobias Hagmann, and Rony Emmenegger. 2015. "Re-Spacing African Drylands: Territorialization, Sedentarization and Indigenous Commodification in the Ethiopian Pastoral Frontier." *Journal of Peasant Studies* 42 (5): 881–901. https://doi.org/10.1080/03066150.2015.1006628.

Kubota, Takuji, Kazumasa Aonashi, Tomoo Ushio, Shoichi Shige, Yukari N Takayabu, Misako Kachi, Yoriko Arai, Tomoko Tashima, Takeshi Masaki, and Nozomi Kawamoto. 2020. "Global Satellite Mapping of Precipitation (GSMaP) Products in the GPM Era." *Satellite Precipitation Measurement* 1: 355–73. https://doi.org/10.1007/978-3-030-24568-9_20.

Kwinta, Andrzej, and Jacek Gniadek. 2017. "The Description of Parcel Geometry and Its Application in Terms of Land Consolidation Planning." *Computers and Electronics in Agriculture* 136: 117–24. https://doi.org/10.1016/j.compag.2017.03.006.

Lambin, E.F., H Geist, J.F. Reynolds, and D. Stafford-Smith, Mark. 2009. "Coupled Human-Environment System Approaches to Desertification." In *Recent Advances in Remote Sensing and Geoinformation Processing for Land Degradation Assessment*, edited by Achim; Roder and Joachim Hill, 1–14. Leiden: CRC Press.

Lambin, Eric F., Patrick Meyfroidt, Ximena Rueda, Allen Blackman, Jan Börner, Paolo Omar Cerutti, Thomas Dietsch, et al. 2014. "Effectiveness and Synergies of Policy Instruments for Land Use Governance in Tropical Regions." *Global Environmental Change* 28 (1): 129–40. https://doi.org/10.1016/j.gloenvcha.2014.06.007.

Lambin, Eric F, B.L. Turner, Helmut J Geist, Samuel B Agbola, Arild Angelsen, John W Bruce, Oliver T Coomes, et al. 2001. "The Causes of Land-Use and Land-Cover Change:

Moving beyond the Myths." *Global Environmental Change* 11 (4): 261–69. https://doi.org/10.1016/S0959-3780(01)00007-3.
Lawry, Steven, Cyrus Samii, Ruth Hall, Aaron Leopold, Donna Hornby, Steven Lawry, Cyrus Samii, Ruth Hall, Aaron Leopold, and Donna Hornby. 2017. "The Impact of Land Property Rights Interventions on Investment and Agricultural Productivity in Developing Countries: A Systematic Review." *Journal of Development Effectiveness* 9 (1): 61–81. https://doi.org/10.1080/19439342.2016.1160947.
Leeuw, Jan de, Afag Rizayeva, Elmaddin Namazov, Emil Bayramov, Michael T. Marshall, Jonathan Etzold, and Regina Neudert. 2019. "Application of the MODIS MOD 17 Net Primary Production Product in Grassland Carrying Capacity Assessment." *International Journal of Applied Earth Observation and Geoinformation* 78: 66–76. https://doi.org/10.1016/j.jag.2018.09.014.
Leonhardt, Heidi, Marianne Penker, and Klaus Salhofer. 2019. "Do Farmers Care about Rented Land? A Multi-Method Study on Land Tenure and Soil Conservation." *Land Use Policy* 82: 228–39. https://doi.org/10.1016/j.landusepol.2018.12.006.
Li, Ang, Jianguo Wu, Xueyao Zhang, Jianguo Xue, Zhifeng Liu, Xingguo Han, and Jianhui Huang. 2018. "China's New Rural 'Separating Three Property Rights' Land Reform Results in Grassland Degradation: Evidence from Inner Mongolia." *Land Use Policy* 71: 170–82. https://doi.org/10.1016/j.landusepol.2017.11.052.
Li, Wenhuai, Shuxia Zhan, Zhichun Lan, X. Ben Wu, and Yongfei Bai. 2015. "Scale-Dependent Patterns and Mechanisms of Grazing-Induced Biodiversity Loss: Evidence from a Field Manipulation Experiment in Semiarid Steppe." *Landscape Ecology* 30 (9): 1751–65. https://doi.org/10.1007/s10980-014-0146-4.
Li, Wenjun, and Lynn Huntsinger. 2011. "China's Grassland Contract Policy and Its Impacts on Herder Ability to Benefit in Inner Mongolia: Tragic Feedbacks." *Ecology and Society* 16 (2): 1. https://doi.org/10.5751/ES-03969-160201.
Li, Wenjun, and Yanbo Li. 2012. "Managing Rangeland as a Complex System: How Government Interventions Decouple Social Systems from Ecological Systems." *Ecology and Society* 17 (1). https://doi.org/10.5751/ES-04531-170109.
Li, Xin-yi, Xiao-fei Yin, Xiao-li Zhou, and Ping Li. 2020. "Background and Achievements of China's Subsidy and Award Policy for Farmers and Herdsmen." *Acta Prataculturae Sinica*. https://doi.org/10.11686/cyxb2019453.
Li, Yanbo, and Wenjun Li. 2021. "Do Fodder Import and Credit Loans Lead to Climate Resiliency in the Pastoral Social-Ecological System of Inner Mongolia?" *Ecology and Society* 26 (1). https://doi.org/10.5751/ES-12245-260127.
Liao, Chuan, Stephen J Morreale, Karim Aly S. Kassam, Patrick J Sullivan, and Ding Fei. 2014. "Following the Green: Coupled Pastoral Migration and Vegetation Dynamics in the Altay and Tianshan Mountains of Xinjiang, China." *Applied Geography* 46: 61–70. https://doi.org/10.1016/j.apgeog.2013.10.010.
Linkow, Benjamin. 2016. "Causes and Consequences of Perceived Land Tenure Insecurity: Survey Evidence from Burkina Faso." *Land Economics* 92 (2): 308–27. https://doi.org/10.3368/le.92.2.308.
Liu, Zhan Yu, Jing Feng Huang, Xin Hong Wu, and Yong Ping Dong. 2007. "Comparison of Vegetation Indices and Red-Edge Parameters for Estimating Grassland Cover from Canopy Reflectance Data." *Journal of Integrative Plant Biology* 49 (3): 299–306. https://doi.org/10.1111/j.1744-7909.2007.00401.x.

Liverman, Diana M., and Rosa Maria Roman Cuesta. 2008. "Human Interactions with the Earth System: People and Pixels Revisited." *Earth Surface Processes and Landforms* 33: 1458–71. https://doi.org/10.1002/esp.

Lovo, Stefania. 2016. "Tenure Insecurity and Investment in Soil Conservation. Evidence from Malawi." *World Development* 78: 219–29. https://doi.org/10.1016/j.worlddev.2015.10.023.

Lu, Yu, Lynn Huntsinger, and Wen Jun Li. 2022. "Microcredit Programs May Increase Risk to Pastoralist Livelihoods in Inner Mongolia." *Ambio* 51 (4): 1063–77. https://doi.org/10.1007/s13280-021-01586-y.

Ma, Xianlei, Nico Heerink, Ekko van Ierland, Hairu Lang, and Xiaoping Shi. 2015. "Land Rental Market Development in Rural China - Impact of Tenure Security and Trust." In *ICAE 2015, Agriculture in an Interconnected World*, 1–36.

McCall, Michael K. 2006. "Precision for Whom? Mapping Ambiguity and Certainty in (Participatory) GIS." *Participatory Learning and Action* 54 (1): 114–19.

McFadden, Daniel. 1973. "Conditional Logit Analysis of Qualitative Choice Behavior." *Frontiers in Econometrics*, 105–42. https://doi.org/10.1108/eb028592.

Mcginnis, Michael D, and Elinor Ostrom. 2014. "Ecology and Society: Social-Ecological System Framework: Initial Changes and Continuing Challenges." *Ecology and Society* 19 (2): 30.

Millennium Ecosystem Assessment. 2005. *Millennium Ecosystem Assessment.* Washington, D.C: Island Press.

Millward, Michael F., Derek W. Bailey, Andres F. Cibils, and Jerry L. Holechek. 2020. "A GPS-Based Evaluation of Factors Commonly Used to Adjust Cattle Stocking Rates on Both Extensive and Mountainous Rangelands." *Rangelands* 42 (3): 63–71. https://doi.org/10.1016/j.rala.2020.04.001.

Ministry of Agriculture and Rural Affairs. 2006. "Technical Rules for Monitoring of Rangleand Resources and Ecology, NY/T 1233-2006."

———. 2015. "Calculation of Rangeland Carrying Capacity. Agricultural Industry Standard of the People's Republic of China, NY/T 635-2015." Ministry of Agriculture and Rural Affairs of the People's Republic of China.

———. 2022. "Nutrient Requirements of Meat-Type Sheep and Goat."

National Research Council, Division of Behavioural and Social Eduction, Committee on the Human Dimensions of Global Change, Board on Environmental Change and Society. 1998. *People and Pixels: Linking Remote Sensing and Social Science*. Edited by Diana Liverman, Emilio F. Moran, Ronald R. Rindfuss, and Paul C. Stern. Washington DC: National Academies Press.

Neudert, Regina. 2015. "Is Individualized Rangeland Lease Institutionally Incompatible with Mobile Pastoralism? - A Case Study from Post-Socialist Azerbaijan." *Human Ecology* 43 (6): 785–98. https://doi.org/10.1007/s10745-015-9792-7.

Nyima, Yonten. 2015. "What Factors Determine Carrying Capacity? A Case Study from Pastoral Tibet." *Area* 47 (1): 73–80. https://doi.org/10.1111/area.12137.

Ostrom, Elinor. 2009. "A General Framework for Analyzing Sustainability of Social-Ecological Systems." *Science* 325 (5939): 419–23.

Ostrom, Elinor, Marco A. Janssen, and John M. Anderies. 2007. "Going beyond Panaceas." *Proceedings of the National Academy of Sciences of the United States of America* 104 (39): 15176–78. https://doi.org/10.1073/pnas.0701886104.

Ottaviani, Gianluigi, Rafael Molina-Venegas, Tristan Charles-Dominique, Stefano Chelli, Giandiego Campetella, Roberto Canullo, and Jitka Klimešová. 2020. "The Neglected Belowground Dimension of Plant Dominance." *Trends in Ecology and Evolution* 35 (9): 763–66. https://doi.org/10.1016/j.tree.2020.06.006.

Pagiola, Stefano, Agustin Arcenas, and Gunars Platais. 2005. "Can Payments for Environmental Services Help Reduce Poverty? An Exploration of the Issues and the Evidence to Date from Latin America." *World Development* 33 (2): 237–53. https://doi.org/10.1016/j.worlddev.2004.07.011.

Petrzelka, Peggy, Zhao Ma, and Stephanie Malin. 2013. "The Elephant in the Room: Absentee Landowner Issues in Conservation and Land Management." *Land Use Policy* 30 (1): 157–66. https://doi.org/10.1016/j.landusepol.2012.03.015.

Petz, Katalin, Rob Alkemade, Michel Bakkenes, Catharina J.E. Schulp, Marijn van der Velde, and Rik Leemans. 2014. "Mapping and Modelling Trade-Offs and Synergies between Grazing Intensity and Ecosystem Services in Rangelands Using Global-Scale Datasets and Models." *Global Environmental Change* 29: 223–34. https://doi.org/10.1016/j.gloenvcha.2014.08.007.

Piipponen, Johannes, Mika Jalava, Jan Leeuw, Afag Rizayeva, Cecile Godde, Gabriel Cramer, Mario Herrero, and Matti Kummu. 2022. "Global Trends in Grassland Carrying Capacity and Relative Stocking Density of Livestock." *Global Change Biology* 28 (12): 3902–19. https://doi.org/10.1111/gcb.16174.

Place, Frank, and Peter Hazell. 1993. "Productivity Effects of Indigenous Land Tenure Systems in Sub-Saharan Africa." *American Journal of Agricultural Economics* 75 (1): 10–19. https://doi.org/10.2307/1242949.

Place, Frank, Michael Roth, and Peter Hazell. 1994. "Land Tenure Security and Agricultural Performance in Africa: Overview of Research Methodology." In *Searching for Land Tenure Security in Africa*, edited by John W. Bruce and S.E. Migot-Adholla, 15–39. Dubuque, Iowa: Kendall/Hunt.

PRIndex. 2018. "PRIndex Analytical Report 2017: Findings from 3-Country Test." https://www.prindex.net/documents/27/Prindex_Analytical_Report_180311.pdf.

Qi, J, A Chehbouni, A R Huete, Y H Kerr, and S Sorooshian. 1994. "A Modify Soil Adjust Vegetation Index." *Remote Sensing of Environment* 48: 119–26.

Qiao, Guanghua, Bao Zhang, Jing Zhang, and Colin Brown. 2018. "Land Rental, Prices and the Management of China's Grasslands: The Case of Inner Mongolia Autonomous Region." *The Rangeland Journal* 40 (3): 231–40.

Qin, Ping, Fredrik Carlsson, and Jintao Xu. 2011. "Forest Tenure Reform in China: A Choice Experiment on Farmers' Property Rights Preferences." *Land Economics* 87 (3): 473–87. https://doi.org/10.3368/le.87.3.473.

Rai, Rajesh Kumar, Dipendra Bhattarai, and Sajan Neupane. 2019. "Designing Solid Waste Collection Strategy in Small Municipalities of Developing Countries Using Choice Experiment." *Journal of Urban Management* 8 (3): 386–95. https://doi.org/10.1016/j.jum.2018.12.008.

Raich, Author J.W., E.B. Rastetter, J.M. Melillo, D.W. Kicklighter, P A Steudler, J. Peterson, A.L. Grace, B. Moore Iii, and C.J. Vörösmarty. 1991. "Potential Net Primary Productivity in South America: Application of a Global Model." *Ecological Applications* 1 (4): 399–429. https://doi.org/10.2307/1941899.

Rakotonarivo, O. Sarobidy, Jette B Jacobsen, Helle O. Larsen, Julia P G Jones, Martin R Nielsen, Bruno S. Ramamonjisoa, Rina H Mandimbiniaina, and Neal Hockley. 2017. "Qualitative and Quantitative Evidence on the True Local Welfare Costs of Forest Conservation in Madagascar: Are Discrete Choice Experiments a Valid Ex Ante Tool ?" *World Development* 94: 478–91. https://doi.org/10.1016/j.worlddev.2017.02.009.

Ranjan, Pranay, Chloe B. Wardropper, Francis R. Eanes, Sheila M.W. Reddy, Seth C. Harden, Yuta J. Masuda, and Linda S. Prokopy. 2019. "Understanding Barriers and Opportunities for Adoption of Conservation Practices on Rented Farmland in the US." *Land Use Policy* 80: 214–23. https://doi.org/10.1016/j.landusepol.2018.09.039.

Raudsepp-Hearne, C., G. D. Peterson, and E. M. Bennett. 2010. "Ecosystem Service Bundles for Analyzing Tradeoffs in Diverse Landscapes." *Proceedings of the National Academy of Sciences* 107 (11): 5242–47. https://doi.org/10.1073/pnas.0907284107.

Raymond, Hunt E, and Brian A Miyake. 2015. "Comparison of Stocking Rates From Remote Sensing and Geospatial Data." *Rangeland Ecology & Management* 59 (1): 11–18.

Reid, Robin S, María E Fernández-Giménez, and Kathleen A Galvin. 2014. "Dynamics and Resilience of Rangelands and Pastoral Peoples around the Globe." *Annual Review of Environment and Resources*. https://doi.org/10.1146/annurev-environ-020713-163329.

Reynolds, James F., D. Mark Stafford Smith, Eric F. Lambin, B. L. Turner, Michael Mortimore, Simon P. J. Batterbury, Thomas E Downing, et al. 2007. "Global Desertification: Building a Science for Dryland Development." *Science* 316 (5826): 847–51. https://doi.org/10.1126/science.1131634.

Rindfuss, Ronald R, Pramote Prasartkul, Stephen J Walsh, Barbara Entwisle, Yothin Sawangdee, and John B Vogler. 2004. "Household-Parcel Linkages in Nang Rong, Thailand." In *People and the Environment: Approaches for Linking Household and Community Surveys to Remote Sensing and GIS*, edited by Jefferson Fox, Ronald R. Rindfuss, Stephen J. Walsh, and Vinod Mishra, 131–72. Boston, MA: Springer.

Rindfuss, Ronald R, Stephen J Walsh, B. L. Turner, Jefferson Fox, and Vinod Mishra. 2004. "Developing a Science of Land Change: Challenges and Methodological Issues." *Proceedings of the National Academy of Sciences of the United States of America* 101 (39): 13976–81. https://doi.org/10.1073/pnas.0401545101.

Robinson, Brian E, Ping Li, and Xiangyang Hou. 2017. "Institutional Change in Social-Ecological Systems: The Evolution of Grassland Management in Inner Mongolia." *Global Environmental Change* 47: 64–75. https://doi.org/10.1016/j.gloenvcha.2017.08.012.

Robinson, Brian E., Yuta J. Masuda, Allison Kelly, Margaret B. Holland, Charles Bedford, Malcolm Childress, Diana Fletschner, et al. 2018. "Incorporating Land Tenure Security into Conservation." *Conservation Letters* 11 (2): e12383. https://doi.org/10.1111/conl.12383.

Rodell, M., P. R. Houser, U. Jambor, J. Gottschalck, K. Mitchell, C.-J. Meng, K. Arsenault, et al. 2004. "The Global Land Data Assimilation System." *Bulletin of the American Meteorological Society* 85 (3): 381–94. https://doi.org/10.1175/BAMS-85-3-381.

RRI. 2012. "What Rights? A Comparative Analysis of Developing Countries' National Legislation on Community and Indigenous Peoples' Forest Tenure Rights." Washington DC: Rights and Resources Initiative. https://rightsandresources.org/wp-content/exported-pdf/whatrightsnovember13final.pdf.

Running, Steven W, and Maosheng Zhao. 2015. “Daily GPP and Annual NPP (MOD17A2/A3) Products NASA Earth Observing System MODIS Land Algorithm.” *MOD17 User’s Guide* 2015: 1–28.

Sadoulet, Elisabeth, Alain Janvry, and Seiichi Fukui. 1997. “The Meaning of Kinship in Sharecropping Contracts.” *American Journal of Agricultural Economics* 79 (2): 394–406. https://doi.org/10.2307/1244138.

Salzman, James, Genevieve Bennett, Nathaniel Carroll, Allie Goldstein, and Michael Jenkins. 2018. “The Global Status and Trends of Payments for Ecosystem Services.” *Nature Sustainability* 1 (3): 136–44. https://doi.org/10.1038/s41893-018-0033-0.

Sankey, Joel B., Temuulen T. Sankey, Junran Li, Sujith Ravi, Guan Wang, Joshua Caster, and Alan Kasprak. 2021. “Quantifying Plant-Soil-Nutrient Dynamics in Rangelands: Fusion of UAV Hyperspectral-LiDAR, UAV Multispectral-Photogrammetry, and Ground-Based LiDAR-Digital Photography in a Shrub-Encroached Desert Grassland.” *Remote Sensing of Environment* 253: 112223. https://doi.org/10.1016/j.rse.2020.112223.

Sarigai, and Yufeng Zhang. 2015. “The Investigation and Study on Grassland Circulation Price at Dongwuzhumuqin Region in Xilinguole League.” *China Land Sciences* 29 (11): 89–94.

Sayre, Nathan F. 2008. “The Genesis, History, and Limits of Carrying Capacity.” *Annals of the Association of American Geographers* 98 (1): 120–34. https://doi.org/10.1080/00045600701734356.

Sayre, Nathan F. 2017. *The Politics of Scale*. *The Politics of Scale*. Chicago: University of Chicago Press.

Sayre, Nathan F., William DeBuys, Brandon T. Bestelmeyer, and Kris M. Havstad. 2012. “‘The Range Problem’ After a Century of Rangeland Science: New Research Themes for Altered Landscapes.” *Rangeland Ecology & Management* 65 (6): 545–52. https://doi.org/10.2111/REM-D-11-00113.1.

Sayre, Nathan F., Ryan Rj McAllister, Brandon T. Bestelmeyer, Mark Moritz, and Matthew D. Turner. 2013. “Earth Stewardship of Rangelands: Coping with Ecological, Economic, and Political Marginality.” *Frontiers in Ecology and the Environment* 11 (7): 348–54. https://doi.org/10.1890/120333.

Scarpa, Riccardo, and John M. Rose. 2008. “Design Efficiency for Non-Market Valuation with Choice Modelling: How to Measure It, What to Report and Why.” *Australian Journal of Agricultural and Resource Economics* 52 (3): 253–82. https://doi.org/10.1111/j.1467-8489.2007.00436.x.

Schlager, Edella, and Elinor Ostrom. 1992. “Property-Rights Regimes and Natural Resources: A Conceptual Analysis.” *Land Economics* 68 (3): 249–62. https://doi.org/10.2307/3146375.

Schlesinger, William H., and Emily S. Bernhardt. 2020. “The Carbon Cycle of Terrestrial Ecosystems.” In *Biogeochemistry (Fourth Edition)*, edited by William H. Schlesinger and Emily S. Bernhardt, 141–82. Academic Press. https://doi.org/10.1016/B978-0-12-814608-8.00005-0.

Schloss, A. L., D. W. Kicklighter, J. Kaduk, and U. Wittenberg. 1999. “Comparing Global Models of Terrestrial Net Primary Productivity (NPP): Comparison of NPP to Climate and the Normalized Difference Vegetation Index (NDVI).” *Global Change Biology* 5: 25–34. https://doi.org/10.1046/j.1365-2486.1999.00004.x.

Scoones, Ian. 1995. “Exploiting Heterogeneity: Habitat Use by Cattle in Dryland Zimbabwe.” *Journal of Arid Environments* 29 (2): 221–37. https://doi.org/10.1016/S0140-1963(05)80092-8.

Scott, James C. 1998. *Seeing like a State: How Certain Schemes to Improve the Human Condition Have Failed*. New Haven: Yale university Press.

Secor, Anna. 2010. “Social Surveys, Interviews, and Focus Groups.” In *Research Methods in Geography: A Critical Introduction*, edited by Basil Gomez and John Paul Jones III, 194–205. United Kingdom: John Wiley & Sons.

Shang, Wenxiu, Yicheng Gong, Zhongjing Wang, and Michael J. Stewardson. 2018. “Eco-Compensation in China: Theory, Practices and Suggestions for the Future.” *Journal of Environmental Management* 210: 162–70. https://doi.org/10.1016/j.jenvman.2017.12.077.

Shrestha, Sushma, and Kimberly E. Medley. 2016. “Landscape Mapping: Gaining ‘Sense of Place’ for Conservation in the Manaslu Conservation Area, Nepal.” *Journal of Ethnobiology* 36 (2): 326. https://doi.org/10.2993/0278-0771-36.2.326.

Sjaastad, Espen, and Daniel W Bromley. 2000. “The Prejudices of Property Rights: On Individualism, Specificity, and Security in Property Regimes.” *Development Policy Review* 18 (4): 365–89. https://doi.org/10.1111/1467-7679.00117.

Sjaastad, Espen, and Ben Cousins. 2009. “Formalisation of Land Rights in the South: An Overview.” *Land Use Policy* 26 (1): 1–9. https://doi.org/10.1016/j.landusepol.2008.05.004.

Sklenicka, Petr, Vratislava Janovska, Miroslav Salek, Josef Vlasak, and Kristina Molnarova. 2014. “The Farmland Rental Paradox: Extreme Land Ownership Fragmentation as a New Form of Land Degradation.” *Land Use Policy* 38: 587–93. https://doi.org/10.1016/j.landusepol.2014.01.006.

Sloat, Lindsey L., James S. Gerber, Leah H. Samberg, William K. Smith, Mario Herrero, Laerte G. Ferreira, Cécile M. Godde, and Paul C. West. 2018. “Increasing Importance of Precipitation Variability on Global Livestock Grazing Lands.” *Nature Climate Change* 8 (3): 214–18. https://doi.org/10.1038/s41558-018-0081-5.

StataCorp. 2017. “Stata Statistical Software: Release 15.” College Station, TX: StataCorp LLC.

Stickler, M Mercedes, and Heather Huntington. 2015. “Perceptions of Tenure Security: An Exploratory Analysis of Pre-Treatment Data in Rural Communities across Ethiopia, Guinea, Liberia, and Zambia.” In *World Bank Conference on Land and Poverty*, 1–50. Washington, D.C. https://www.land-links.org/wp-content/uploads/2016/09/Perceptions_of_Tenure_Security-Ethiopia_Guinea_Liberia_Zambia.pdf.

Su, Liufang, Jianjun Tang, and Huanguang Qiu. 2021. “Intended and Unintended Environmental Consequences of Grassland Rental in Pastoral China.” *Journal of Environmental Management* 285: 112126. https://doi.org/10.1016/j.jenvman.2021.112126.

Swinnen, Jo, Pavel Ciaian, d’Artis Kancs, Kristine Van Herck, and Liesbet Vranken. 2013. “Possible Effects on EU Land Markets of New CAP Direct Payments.” European Parliament’s Committee on Agriculture and Rural Development. https://doi.org/10.2861/19298.

Sze, Jocelyne S., L. Roman Carrasco, Dylan Childs, and David P. Edwards. 2021. “Reduced Deforestation and Degradation in Indigenous Lands Pan-Tropically.” *Nature Sustainability* 2: 1–8. https://doi.org/10.1038/s41893-021-00815-2.

Tan, Shu hao, Bo Liu, Qiao yun Zhang, Yong Zhu, Jia hui Yang, and Xin jie Fang. 2017. "Understanding Grassland Rental Markets and Their Determinants in Eastern Inner Mongolia, PR China." *Land Use Policy* 67: 733–41. https://doi.org/10.1016/j.landusepol.2017.07.006.

Tang, Guoqiang, Martyn P. Clark, Simon Michael Papalexiou, Ziqiang Ma, and Yang Hong. 2020. "Have Satellite Precipitation Products Improved over Last Two Decades? A Comprehensive Comparison of GPM IMERG with Nine Satellite and Reanalysis Datasets." *Remote Sensing of Environment* 240: 111697. https://doi.org/10.1016/j.rse.2020.111697.

Taylor, Holly A., and Barbara Tversky. 1992. "Spatial Mental Models Derived from Survey and Route Descriptions." *Journal of Memory and Language* 31 (2): 261–92. https://doi.org/10.1016/0749-596X(92)90014-O.

The Nature Conservancy. 2009. "Global Ecoregions, Major Habitat Types, Biogeographical Realms and The Nature Conservancy Terrestrial Assessment Units." The Nature Conservancy. https://tnc.maps.arcgis.com/home/item.html?id=7b7fb9d945544d41b3e7a91494c42930.

Train, Kenneth E. 2009. *Discrete Choice Methods with Simulation*. Cambridge university press.

Tseng, Tzu-Wei Joy, Brian E. Robinson, Marc F. Bellemare, Ariel BenYishay, Allen Blackman, Timothy Boucher, Malcolm Childress, et al. 2021. "Influence of Land Tenure Interventions on Human Well-Being and Environmental Outcomes." *Nature Sustainability* 4 (3): 242–51. https://doi.org/10.1038/s41893-020-00648-5.

Turner, Billie L, and Jacqueline Geoghegan. 2004. "Land-Cover and Land-Use Change (LCLUC) in the Southern Yucatán Peninsular Region (SYPR)." In *People and the Environment: Approaches for Linking Household and Community Surveys to Remote Sensing and GIS*, edited by Jefferson Fox, Ronald R. Rindfuss, Stephen Walsh, and Vinod Mishra, 31–60. Boston, MA: Springer.

Turner, B.L., Eric F. Lambin, and Annette Reenberg. 2007. "The Emergence of Land Change Science for Global Environmental Change and Sustainability." *Proceedings of the National Academy of Sciences* 104: 20666–71. https://doi.org/10.1017/CBO9781107415324.004.

Turner, Matthew D. 2003. "Methodological Reflections on the Use of Remote Sensing and Geographic Information Science in Human Ecological Research." *Human Ecology* 31 (2): 255–79. https://doi.org/10.1023/A:1023984813957.

Tversky, Barbara. 2003. "Structures of Mental Spaces: How People Think about Space." *Environment and Behavior* 35 (1): 66–80. https://doi.org/10.1177/0013916502238865.

USAID. 2008. "Land Tenure and Property Rights: Tools for Transformational Development," 1–8.

van Gelder, Jean-Louis. 2010. "What Tenure Security? The Case for a Tripartite View." *Land Use Policy* 27 (2): 449–56. https://doi.org/10.1016/j.landusepol.2009.06.008.

Vetter, S. 2005. "Rangelands at Equilibrium and Non-Equilibrium: Recent Developments in the Debate." *Journal of Arid Environments* 62 (2): 321–41. https://doi.org/10.1016/j.jaridenv.2004.11.015.

Wan, Z., S. Hook, and G. Hulley. 2015. "MOD11A1 MODIS/Terra Land Surface Temperature/Emissivity Daily L3 Global 1km SIN Grid V006." NASA EOSDIS Land Processes DAAC. https://doi.org/10.5067/MODIS/MOD11A1.006.

Wang, J., P. M. Rich, and K. P. Price. 2003. "Temporal Responses of NDVI to Precipitation and Temperature in the Central Great Plains, USA." *International Journal of Remote Sensing* 24 (11): 2345–64. https://doi.org/10.1080/01431160210154812.

Wang, Jun, Daniel G Brown, and Arun Agrawal. 2013. "Climate Adaptation, Local Institutions, and Rural Livelihoods: A Comparative Study of Herder Communities in Mongolia and Inner Mongolia, China." *Global Environmental Change* 23 (6): 1673–83. https://doi.org/10.1016/j.gloenvcha.2013.08.014.

Watson, L.W., D.G. Burnside, and A.McR. Holm. 1996. "Event-Driven or Continuous; Which Is the Better Model for Managers?" *Rangeland Journal* 18 (2): 351–69.

Weiser, R. L., G. Asrar, G. P. Miller, and E. T. Kanemasu. 1986. "Assessing Grassland Biophysical Characteristics from Spectral Measurements." *Remote Sensing of Environment* 20 (2): 141–52. https://doi.org/10.1016/0034-4257(86)90019-2.

Westoby, Mark, Brian Walker, and Imanuel Noy-Meir. 1989. "Opportunistic Management for Rangelands Not at Equilibrium." *Journal of Range Management* 42 (4): 266–74. https://doi.org/10.2307/3899492.

White, Robin, Siobhan Murray, and Mark Rohweder. 2000. *Pilot Analysis of Global Ecosystems: Grassland Ecosystems*. Washington, DC: World Resources Institute. https://www.wri.org/research/pilot-analysis-global-ecosystems-grassland-ecosystems.

Wiens, J A. 1989. "Spatial Scaling in Ecology." *Functional Ecology* 3 (4): 385. https://doi.org/10.2307/2389612.

Williams, Dee Mack. 1996. "Grassland Enclosures: Catalyst of Land Degradation in Inner Mongolia." *Human Organization* 55 (3): 307–13. https://doi.org/10.17730/humo.55.3.u46ht013r361668t.

Woolthuis, Rosalinde Klein, Bas Hillebrand, and Bart Nooteboom. 2005. "Trust, Contract and Relationship Development." *Organization Studies* 26 (6): 813–40. https://doi.org/10.1177/0170840605054594.

Wu, Jianguo, Qing Zhang, Ang Li, and Cunzhu Liang. 2015. "Historical Landscape Dynamics of Inner Mongolia: Patterns, Drivers, and Impacts." *Landscape Ecology* 30 (9): 1579–98. https://doi.org/10.1007/s10980-015-0209-1.

Würtenberger, Laura, Thomas Koellner, and Claudia R. Binder. 2006. "Virtual Land Use and Agricultural Trade: Estimating Environmental and Socio-Economic Impacts." *Ecological Economics* 57 (4): 679–97. https://doi.org/10.1016/j.ecolecon.2005.06.004.

Xie, Yina, and Wenjun Li. 2008. "Why Do Herders Insist on Otor? Maintaining Mobility in Inner Mongolia." *Nomadic Peoples* 12 (2): 35–52. https://doi.org/10.3167/np.2008.120203.

Xilin Gol League Statistical Bureau. 2020. "Xilin Gol Satistical Year Book." http://tjj.xlgl.gov.cn/ywlm/tjsj/lnsj/sczz/202103/P020210322343429592912.pdf.

Yahdjian, Laura, Osvaldo E. Sala, and Kris M. Havstad. 2015. "Rangeland Ecosystem Services: Shifting Focus from Supply to Reconciling Supply and Demand." *Frontiers in Ecology and the Environment* 13 (1): 44–51. https://doi.org/10.1890/140156.

Yeh, Emily T. and Gaerrang. 2011. "Tibetan Pastoralism in Neoliberalising China: Continuity and Change in Gouli." *Area* 43 (2): 165–72. https://doi.org/10.1111/J.1475-4762.2010.00976.X.

Yin, Yantin, Yulu Hou, Colin Langford, Haihua Bai, and Xianyang Hou. 2019. "Herder Stocking Rate and Household Income under the Grassland Ecological Protection Award Policy in

Northern China." *Land Use Policy* 82: 120–29. https://doi.org/10.1016/j.landusepol.2018.11.037.

Zhang, Jing, Colin Brown, Guanghua Qiao, and Bao Zhang. 2019. "Effect of Eco-Compensation Schemes on Household Income Structures and Herder Satisfaction: Lessons From the Grassland Ecosystem Subsidy and Award Scheme in Inner Mongolia." *Ecological Economics* 159 (2018): 46–53. https://doi.org/10.1016/j.ecolecon.2019.01.006.

Zhang, Qing, Alexander Buyantuev, Xuening Fang, Peng Han, Ang Li, Frank Yonghong Li, Cunzhu Liang, et al. 2020. "Ecology and Sustainability of the Inner Mongolian Grassland: Looking Back and Moving Forward." *Landscape Ecology* 35 (11): 2413–32. https://doi.org/10.1007/s10980-020-01083-9.

Zhao, Xia, Huifeng Hu, Haihua Shen, Daojing Zhou, Liming Zhou, Ranga B. Myneni, and Jingyun Fang. 2015. "Satellite-Indicated Long-Term Vegetation Changes and Their Drivers on the Mongolian Plateau." *Landscape Ecology* 30 (9): 1599–1611. https://doi.org/10.1007/s10980-014-0095-y.

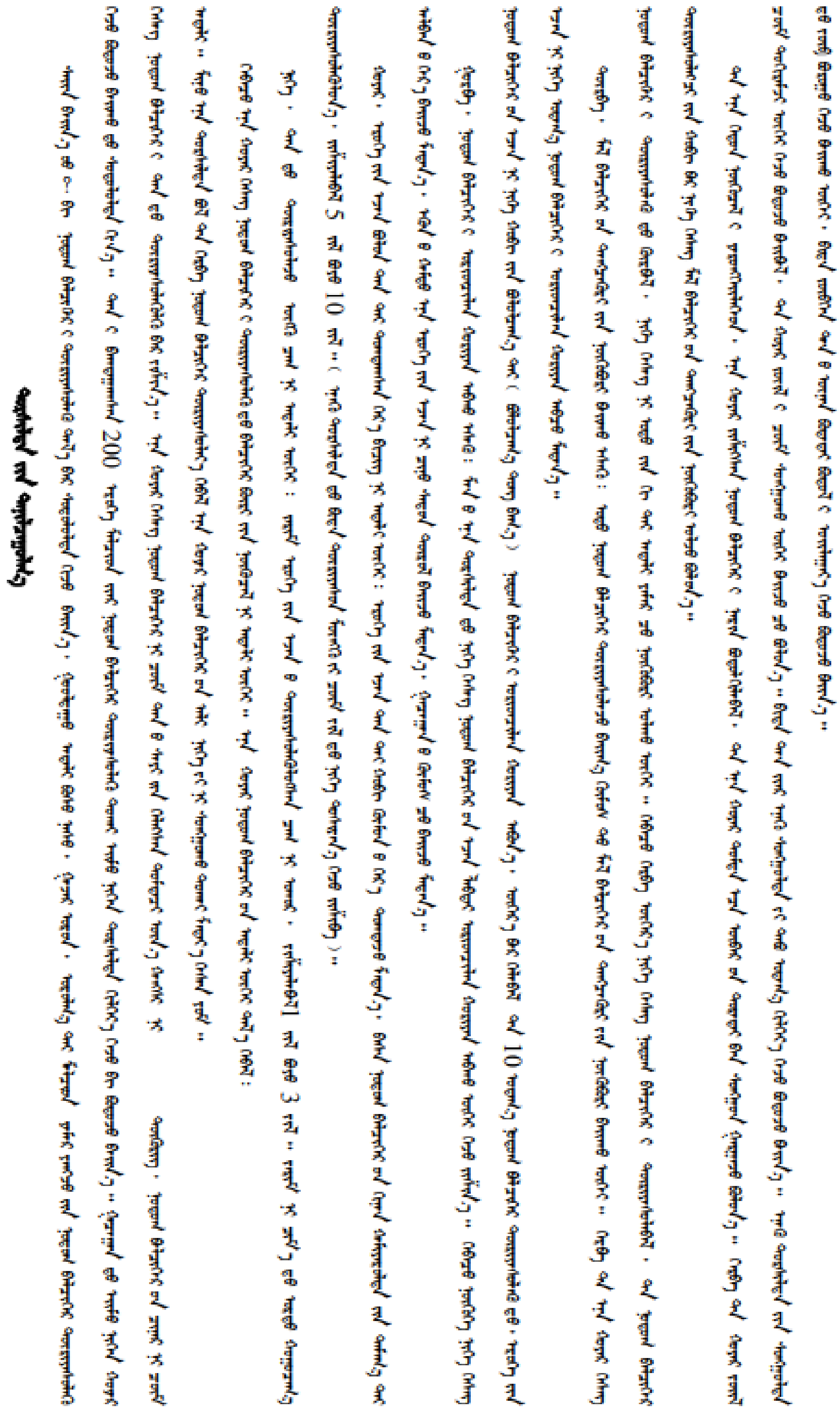
Zhou, Yanyan, Dongxia Yue, Chen Li, Xinliang Mu, and Jianjun Guo. 2021. "Identifying the Spatial Drivers of Net Primary Productivity: A Case Study in the Bailong River Basin, China." *Global Ecology and Conservation* 28: e01685. https://doi.org/10.1016/j.gecco.2021.e01685.

Zhu, Zhe, Michael A. Wulder, David P. Roy, Curtis E. Woodcock, Matthew C. Hansen, Volker C. Radeloff, Sean P. Healey, et al. 2019. "Benefits of the Free and Open Landsat Data Policy." *Remote Sensing of Environment* 224: 382–85. https://doi.org/10.1016/j.rse.2019.02.016.

Zimmerer, Karl S., Eric F. Lambin, and Steven J. Vanek. 2018. "Smallholder Telecoupling and Potential Sustainability." *Ecology and Society* 23 (1). https://doi.org/10.5751/ES-09935-230130.

Appendix V: Choice experiment scenario description

English translation can be found in https://github.com/lucixlu/dissertation_translation.



Appendix VI: Household survey (on *de facto* land use)

English translation can be found in https://github.com/lucixlu/dissertation_translation.

苏木:　　嘎查:　　日期:　　当天的第___户　　户主:___________　　对应问卷号码：1525________

自己用的草场						
用的草场面积	分下来的草场面积	租的面积	租的价格	时间	分草场多少亩/人	哪一年分草场

实际用的　　个人的草场						
家庭成员						
1. 媳妇有草场吗	1.有　2.没有		谁在使用/费用:			
2. 是否分家：	1.分了　2.没有	一起放牧的人有:		牛羊:		
3. 是否分冬夏/接羔草场	1.分了　2.没有	冬季：	夏季：	接羔：		
4. 户主的父母出生于本苏木吗？	1.是　2.不是					
5. 确权了吗？	1.确了　2.没有		确权的变动:			

草场	面积	经过别人草场	距离	时间 20xx-xx-xx	价格	从谁手上租	合同	用草场的月份	牲畜用水
1 租		1.是 2.否	1. 挨着自家草场 2. _____里地 _____方向		元 1.一年一付 2.一次性付 ____年	1.亲戚/好朋友 2.邻居 3.大队里的人 4.大队的地 5. 其他	1.口头合同 2.个人合同 3.嘎查盖章 4.草原监理		1.一直有井 2.新打了井 3.拉水 ____天一次 一次____吨
2 租		1.是 2.否	1. 挨着自家草场 2. _____里地 _____方向		元 1.一年一付 2.一次性付 ____年	1.亲戚/好朋友 2.邻居 3.大队上的人 4. 大队的地 5. 其他	1.口头 2.个人 3.嘎查盖章 4.草原监理		1.一直有井 2.新打了井 3.拉水 ____天一次 一次____吨

租回来的草场	
1. 现在租的几户人的地?	
2. 最早从哪一年开始租?	

草场	距离	哪年	价格	从谁手上租	信息来源	用草场的时间
1 走 场	1. _____里地 _____方向 2. 赶过去要多久？		羊： 牛： 马： 总共：	1.亲戚/好朋友 2.邻居 3.大队里的其他人 4.大队的地 5. 其他		______月 到 ______月
2 走 场	1. _____里地 _____方向 2. 赶过去要多久？		羊： 牛： 马： 总共：	1.亲戚/好朋友 2.邻居 3.大队里的其他人 4.大队的地 5. 其他		______月 到 ______月
2 走 场	1. _____里地 _____方向 2. 赶过去要多久？		羊： 牛： 马： 总共：	1.亲戚/好朋友 2.邻居 3.大队里的其他人 4.大队的地 5. 其他		______月 到 ______月

Appendix VII: Semi-structured interview questions

English translation can be found in https://github.com/lucixlu/dissertation_translation.

1. 基础问题

   - 您现在是租的有地吗?租的有多少亩? 一共有几块地？
   - 是从一户人手上租的吗? 你和户主的关系是什么？是从亲戚手上租的吗?还是从熟人/邻居手上租的?
   - 大队有让你租的地吗? 如果有的话，有什么规则吗？你们是怎么抽签的？

2. 距离

   - 有租在附近的吗?有多远?（有 10 里地吗?）
   - 如果远的话，你是怎么把牲畜赶过去呢?
   - 你认为离远了对你来说有影响吗? 你认为怎么样一个距离最合适?
   - 你认为要去另外一个牧场，其中经过别人家的草场，这个对你有影响吗?

3. 自己的草场分冬夏营盘吗?

   - 放牧的话，一年有几个月再上面放牧?
   - 自己的草场上放牧几个月?主要是干什么?
   - （冬夏营盘的牧民）你为什么选择分两个营盘？这会给你带来什么好处吗？
   - （不分冬夏营盘的的牧民）你觉得从从几十年前那种四季草场，变成现在不分季节，对你来说有影响吗？

4. 时长

   - 你现在租地租的多长时间?从哪一年到哪一年(季度)？
   - 你觉得自己租的这个时间对你来说够用吗?

5. 续约

   - 你现在是租______年，如果到期了你希望从现在租你的这个人手里继续租下去吗?为什么?你觉得再续约对你来来说容不容易?
   - 如果不能续的话怎么办?
   - 你觉得续约对你来说重不重要?

6. 价格

   - 你租的是个什么价格?
   - 这个一般是这个地区的均价吗?这个价格变化大吗?
   - 一年一付吗?要一次性付清吗?要一次性付清这个是以前一直有的，还是最近的?

7. 原因

   - 当时为什么开始租地?
   - 租地了，这一只羊的成本会不会下降一点?

8. 能不能租到草场

   - 你认为过去五年里，租草场的竞争大不大?如果是的，你觉得是为什么?
   - 在你租草场的这些年里，你总是能租到自己想要的草场吗?前几年也可以吗?如果不能的话，是哪一年以及原因是什么? 如果你没有办法租到足够的草场，你会做什么?
   - 有没有嘎查外的人进来放牧?你觉得嘎查外的人进来放牧，对你有影响吗?如果不让嘎查外的人进来租草场了 ，觉得这个好不好?

9. 协议

- 你用的是口头协议还是个人协议?
- (假设) 没有纸面协议，对你有影响吗?

10. 租给别人的可能性

- 你身边有没有发生过那种一块地租给几个人?
- 有没有出现过人在合同到期之前把地收回去的?
- 有没有出现过别人的牲畜进你家租的草场啃的情况?

11. 提前终结的可能性:

- 你觉得户主有没有可能可能这个合同提前收回?

12. 户主对租地的限制

- 租的地可以打草吗?(有人专门租打草场吗)?价格是多少?
- 你这个租来的地，和打草的地有什么区别?
- 户主对你的怎么用这个草场有要求吗?
- 合同上列的有什么要求吗？
- 比如说你可以在上面打井吗?建棚圈吗?
- 比如说你转租给别人这样行不行?

13. 你选择租地的标准

- 在租一块地的时候，除了草以外，你最看重的一个条件什么?

14. 村集体

- 有人提出来可以像在农区一样，进城的人把土地租给嘎查，然后由嘎查租给牧民。你觉得你会愿意从村集体手上租吗?为什么?

15. 安全性

- 你觉得附近有煤矿对你有影响吗?
- 你家附近有关于草场产生过冲突吗?如果有得话，他们最终是怎么解决的?
- 走场：您如何定义走场？你认为你租的地算走场吗？