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Grassland tenure reform and grassland quality in China

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Abstract

This paper investigates the impact of land tenure reform on grassland quality in pastoral areas of China. Using nearly 40 years of remote sensing combined with survey data in the pastoral area of China, we find that the privatization of land use rights without physical (i.e., fences) or legal (i.e., certificates) protection has little impact on improving grassland quality measured by the Normalized Difference Vegetation Index (NDVI). The enhanced privatization of grassland use rights with physical or legal security significantly increases grassland quality. We show that after the privatization of land use rights with security protection, grassland quality experienced about a 3% increase. Our results suggest that switching to privatized use rights without security protection from previously cooperatively managed land may undermine the positive environmental effects of land tenure reform.

KEYWORDS

China, grassland quality, land use rights, privatization, property rights, security, two-way fixed effects, weighted average of difference-in-difference estimators

JEL CLASSIFICATION

Q15

1 | INTRODUCTION

There is a long-lasting debate on whether privatized ownership or collective action is more effective in managing common pool resources (CPRs). Economists have long realized the importance of property rights in mitigating environmental degradation caused by the tragedy of the common (Coase, 1960). Researchers who support privately managed natural resources (Banks, 2003;

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Libecap, 2009; Randall, 1975) believe that well-defined property rights incentivize land users to protect their lands and maximize long-term benefits (Smith, 1981; Welch, 1983). The other strand of literature advocates collective action management (e.g., Calvo-Mendieta et al., 2017; Doss & Meinzen-Dick, 2015; Poteete & Ostrom, 2008; Runge, 1986) by arguing that privatization of property rights may fail when facing high transaction costs. In particular, Ostrom (1990) shows that natural resources can be managed effectively through collective action based on a series of empirical studies of groundwater basins and provides a general theory on the institutional arrangement regarding effective governance of common-pool resources (Dietz et al., 2003).

The impact of privatization on environmental outcomes is context specific and depends on various institutional and cultural factors. Existing literature on the environmental impacts of property rights focuses on the forest, fishery, or cropland sectors (e.g., BenYishay et al., 2017; Costello et al., 2008; Isaksen & Richter, 2019), whereas empirical evidence based on long-term measurement data on grassland is lacking. As pointed out by Liscow (2013) and other scholars, the environmental outcomes of property rights are mixed because the improved property rights have both conservation effects and investment effects on the environmental outcomes. Improved property rights could lead landowners to discount the future less and obtain the long-term benefits, therefore they are more likely to protect their lands (i.e., conservation effects) (Farzin, 1984). At the same time, improved property rights could increase investment in land and the returns of land intensification use (i.e., investment effects), which may be also impacted by land use monitor and farmers' access through the land registry (Jung et al., 2022).

Currently, grassland property rights are mixed around the globe, with both privatization and collective management in effect (Augustine et al., 2021; Lesorogol, 2008; Liu et al., 2020; van Etten, 2013). Grassland is one of the largest and most important ecosystems in the world (Havstad et al., 2007; Peciña et al., 2019; Ren et al., 2018; Sala & Paruelo, 1997) and suffers from severe degradation globally (Brondizio et al., 2019). Grassland differs substantially from cropland and forest in natural resource characteristics (Evrendilek et al., 2004; Webb et al., 1978). It is therefore essential to understand the environmental impacts of privatized property rights on grassland. Unlike the crop and forest sectors, very few studies have examined the property rights impacts on grassland quality empirically. Existing studies using case studies concluded that communal forms of pasture tenure and management are advantageous given the socio-economic and ecological context of the Tibetan Plateau (Richard et al., 2006) and Inner Mongolia (Li et al., 2007) in China.

Grassland tenure reform in the pastoral areas of China provides us with a unique opportunity to examine the impacts of property rights on grassland quality using quantitative methods. The land tenure reform in China transfers grassland use rights from communities to private individuals. Following the implementation of the Household Responsibility System in the crop area, privatization of livestock and grassland use rights started in the pastoral area in the 1980s. The livestock, previously owned and managed by peoples' communes collectively, were first priced to ensure a fair allocation and then privatized to individual households. Grassland use rights were then privatized to individual households, whereas herders were informed of the vague location and area of their grassland. No fences were built and no ownership certifications were distributed to protect their grassland use rights at the beginning, which we define as "privatization with use rights only." It is similar to de facto property rights because the property rights are not specified by a government with recognized authority. As land tenure reforms evolved, some villages started building fences to make clear boundaries, and other villages distributed legal certificates to herders with descriptions of the geographical location of their grasslands. Either building fences or distributing legal certificates increase the security protection of privatized grassland use rights, which we call "privatization with security protection" and is similar to the *de jure* property rights that appeared in the literature (Alston et al., 2011; Klümper et al., 2018).

We analyze the impacts of land tenure reform on grassland quality based on a dataset of nearly 40 years of remote sensing data and land tenure reform data in ecologically fragile regions. Before the grassland tenure reform, both grassland and grazing livestock were managed by the communities (or "people's commune") collectively. The tenure reform changes the ownership of grazing livestock and the use rights of the grassland. Herders can make management decisions according to their best interests after the land tenure reform. Initially, only use rights were privatized, followed by the formal ownership privatization of livestock, including the issuing of legal ownership certificates. Therefore, herders have "the right of control" compared to previously "the right of exclusion" under the cooperative management after the tenure reform where income can also be derived from the controlled properties (Klein & Robinson, 2011).

We investigate the impacts of privatization of grassland use rights, as well as the impact of additional physical (i.e., fences) or legal (i.e., certification) security after the privatization. We find that only privatizing grassland use rights to individual households without physical or legal security protection has little influence on grassland quality in the short term. However, as grassland tenure reforms evolved, enhancing the security of privatized grassland use rights either by building fences or issuing legal certificates significantly improved grassland quality. Specifically, we find that after the privatization of grassland use rights with fences or certificates, grassland quality improved by about 3% based on the weighted difference-in-differences estimator proposed by de Chaisemartin and d'Haultfoeuille (2020). The positive significant results are robust to a set of different specifications. Our empirical results are also consistent with theoretical literature on the strength of property rights on the property owners' extractive behaviors where stronger property rights contribute to more economically efficient resource uses (Costello & Grainger, 2018).

This paper makes primary two contributions to the literature. We provide the first empirical evidence on the impacts of property rights on grassland quality based on a comprehensive, long-term dataset in the pastoral area of China. Unlike the case study in a county of Inner Mongolia of China by (Li et al., 2007), our results represent the average treatment effects of privatization of grassland use rights in 27 counties in five provinces in China using robust identification strategies. Second, we add evidence to our understanding of the environmental impacts of both privatization of grassland use rights and the associated security protection by directly measuring grassland quality. Examining the impacts of security protection of grassland property rights provides important policy implications as our results indicate private property rights without security protections (such as physical or legal security) have limited improvement on the grassland quality.

2 | LITERATURE REVIEW

The empirical literature on the impacts of property rights has expanded rapidly in recent years as property rights are directly connected to land use and rural environment (Rodgers, 2009). Most studies focus on economic and social outcomes or conservation behaviors that indirectly reflect environmental outcomes. Privatized property rights can reshape economic incentives and generate a series of positive economic and social outcomes relative to common-pool resources, such as optimizing resource allocation (Zhao, 2020), incentivizing investment (Abdulai et al., 2011; Bambio & Agha, 2018; Besley, 1995), speeding up economic development (Hornbeck, 2010), and improving long-term health (Xu, 2021). Other studies investigate the impact of property right security on economic and social outcomes, such as investment in cropland (Huntington & Shenoy, 2021), labor allocation and migration (de Janvry et al., 2015), agricultural productivity (Linkow, 2016), and social tensions and disputes (Alston et al., 2000; Deininger & Castagnini, 2006; di Falco et al., 2020). Related to our study, Bühler (2022) finds that grazing lands with well-defined property rights are over 10% more productive than lands without, based on a spatial discontinuities model. Chari et al. (2021) study land property reform in rural China and find an increase in land rental activities among rural households and aggregate productivity. In addition, environmental markets may generate substantial net benefits compared to open access management based on a case study on groundwater rights in southern California (Ayres et al., 2021).

A growing body of literature on the environmental impacts of property rights mainly focuses on forest and fishery sectors and presents mixed results. Of the 48 studies on environmental outcomes, 73% showed positive effects, 15% had negative outcomes, and 29% had cases in which no effect was identified (Tseng et al., 2021). Isaksen and Richter (2019) apply quasi-experimental approaches and find that private property rights lower the probability of a fish stock collapsing, but the estimated impact varies with country and species characteristics compared to open access. Using a global database of fisheries institutions and catch data from 1950 to 2003, Costello et al. (2008) showed that implementation of rights-based catch shares can provide individual incentives for sustainable harvests that is less prone to collapse. However, some literature also presents opposing empirical evidence where private and secure property rights have no significant environmental impacts or even accelerate environmental degradation (Cao et al., 2018; Kabubo-Mariara, 2002; Miteva et al., 2019). Liscow (2013) finds that property rights significantly increase deforestation in Nicaragua when property rights increase investment, which leads to increased agricultural productivity and returns to deforestation. BenYishay et al. (2017) find that the formalization of Indigenous communities' land rights has no effect on satellite-based greenness measures of forest cover in Brazil. Lipscomb and Prabakaran (2020) find no overall impact of a large property rights reform on deforestation during the sample period in the Brazilian Amazon though substantial heterogeneity exists across counties. Although land registry programs in Brazil do not significantly affect crop area but influence pasture expansion by replacing natural vegetation cover (Jung et al., 2022).

One challenge to studying the environmental outcomes of property rights is the potential reverse causality as the implementation of the property rights institutions may depend on the ecosystems that have performed in the past (Ayres et al., 2021; Isaksen & Richter, 2019; Levine, 2005). Recent studies use applied econometrics and randomized control trials (RCT) to overcome the endogeneity of property rights changes (Huntington & Shenoy, 2021). For example, Ayres et al. (2021) applied a spatial regression discontinuity design to a major aquifer in water-scarce Southern California and found that a groundwater market generated substantial net benefits, as capitalized in land values. Another challenge is to directly measure the environmental or ecological functions using relatively simple indicators, especially for a long period. Due to the lack of data on direct measurement of environmental outcomes, many researchers focus on farmers' conservation behaviors, such as investment in improving soil quality in the crop sector and reducing harvest rates in forest or fishery sectors. These conservation behaviors cannot fully reflect the magnitude of environmental changes. For example, farmers may adopt one technology that benefits the environment but may also adopt other technologies for more economic profit that degrade the environment. Unlike other sectors, grassland ecosystems have relatively simple ecological structures and satellite-based greenness data have been used widely to measure grassland quality (e.g., Hou et al., 2021; Liu et al., 2019; Piao et al., 2006). The Normalized Difference Vegetation Index (NDVI) well predicts ecological functions, such as habitat quality (Weber et al., 2018), annual net primary productivity (NPP) (Rasmussen, 1998), ecosystem functional types (Paruelo et al., 2001), and biozones (Soriano & Paruelo, 1992).

The remainder of the paper is organized as follows. Section 2 introduces the background and our data collection process. Section 3 presents empirical models and identification strategies. Section 4 presents results with additional analyses. Section 5 concludes the paper.

3 | BACKGROUND AND DATA COLLECTION

In the 1970s, the collectivization and centralization of agricultural production created a severe shortage of agricultural products across China. The Chinese central government, therefore, started to implement the Household Responsibility System around 1980, where households become responsible for the profits and losses similar to an enterprise. In the rural areas, farmers became independent economic entities and are responsible for the profits and losses of agricultural production. This market-based solution incentivized farmers to increase productivity. Following the success of the Household Responsibility System under land tenure reform on cropland, the central government started grassland tenure reform in the pastoral area in the early 1980s to incentivize herders' production and promote economic growth (Chen & Davis, 1998; Ding, 2003; Lin, 1988). Before grassland tenure reform, both grassland and grazing livestock were managed by the communities (or "people's commune") collectively.[1] During this collectivist period, the communities served governmental, political, and economic functions, and allowed workers to share local welfare from collective actions. In our context, all villagers worked for their communities and were paid stipends (Li, 2008).

The main objective of grassland tenure reform was to privatize both livestock property rights and grassland use rights to individual households, which were previously owned and managed by the communities. The grassland tenure reform unfolded in two stages. In the first stage, livestock was priced and assigned to individual households according to household population. During this period, privately owned and managed livestock was grazed on publicly owned and managed grassland. In the second stage, the grassland use rights were assigned to individual households. Although the gap years between the two stages differed across regions, all regions followed a similar process.

Although the assignment of livestock property rights was completed relatively quickly, the assignment of grassland use rights lasted for a long period. Our field data[2] show that only 57% of the villages had their grassland use rights assigned to individual households by 1995, more than 10 years after the reform started (Figure 1). During this period, grassland borders between households were not well defined. Only 43% of villages reported that their villagers had fences to separate grassland between households physically by 1995. Issuing the certificate for privatization of grassland use rights was also incomplete. Only 34% of villages reported that their villagers received official certification of grassland use rights by 1995. During this period, no provinces have issued a definite guideline on the length of grassland use rights assigned to individuals.

Given the incompleteness and slow process of privatization of grassland use rights, the central government accelerated grassland tenure reform in the mid-1990s. Following the call from the

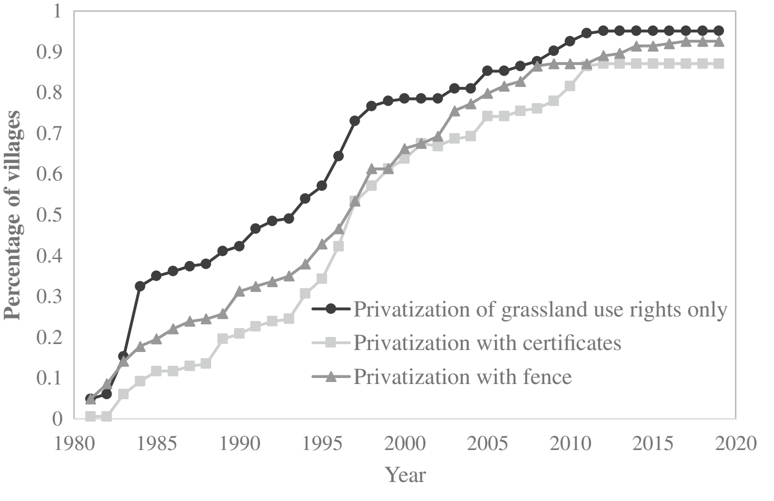


FIGURE 1 Evolution of grassland tenure reform

[1]Before grassland tenure reform, both grassland and grazing livestock were managed by the communities (or "people's commune") collectively.
[2]The details of field data collection will be described later in this section.

central government, the provinces started to speed up and formalize the reform process with specific provisions included, such as distributing a legal privatization certificate to an individual household with the length of grassland use rights (usually 30–50 years) and clear grassland border between neighboring households. One of the most important border clarifying measures was building fences. The central government invested 15.6 billion RMB (about US $2.4 billion) to build fences in eight provinces between 2003 and 2011, which covered 56 million ha (0.21 million square miles) area (Miao & Zhang, 2012). By the end of 2018, a total of 287.2 million ha of grassland use rights were privatized to individual households, accounting for 88.2% of the usable grassland area in China (National Forestry and Grassland Administration, 2018). Our survey data also show that 87% of villages had been designated with specific locations of each household's pasture and issued contract certificates in 2018, and 93% of villages had fences (Figure 1).

To evaluate the impacts of privatization of land use rights on grassland quality, we build a comprehensive dataset that includes detailed measurements of the privatization of land use rights, grassland quality measured by NDVI, and socioeconomic characteristics at the village level. The dataset covers 162 villages in five major pastoral provinces (i.e., Xinjiang, Tibet, Qinghai, Gansu, Ningxia, and Inner Mongolia) in China from 1981 to 2019. The grassland quality is commonly measured by NDVI, which was constructed based on infrared and near-infrared channel remote sensing images and has been widely used as an indicator of vegetation coverage (e.g., Peters et al., 2002; Zhumanova et al., 2018). As grassland ecosystems have relatively simple ecological structures, NDVI is commonly used to measure grassland quality (e.g., Hou et al., 2021; Liu et al., 2019; Piao et al., 2006). Existing literature also shows that NDVI could be used to predict ecological functions, such as habitat quality (Weber et al., 2018), annual net primary productivity (NPP) (Rasmussen, 1998), ecosystem functional types (Paruelo et al., 2001), and biozones (Soriano & Paruelo, 1992).

The NDVI data for the period 1981–1999 were recorded at a spatial resolution of 8×8 km^2, which was acquired from the GIMMS (Global Inventory Modeling and Mapping Studies) product from NASA (National Aeronautics and Space Administration). For the period 2000–2019, the NDVI data were recorded at a spatial resolution of 1×1 km^2 and were acquired from the MOD13A3 product from NASA Earth data. More detailed information about the GIMMS and MOD13A3 dataset products can be found in Tucker et al. (2005) and Didan (2010), respectively. We measure the grassland quality by calculating the NDVI for each village using the spatial NDVI data as well as the recorded GPS coordinates of the village. The time trend of NDVI in each province is presented in Figure S1 in Appendix A, which shows that there are substantial spatial and temporal variations in the NDVI measurements during our study window in the region. Note that each province has a different NDVI baseline, although there is no obvious divergence in the grassland quality trends over the years.

The privatization of land use rights and socioeconomic characteristics are obtained from a large-scale, multiyear field survey. To investigate the influences of land tenure reform on grassland quality, the research team surveyed Qinghai and Gansu provinces in 2017, Xinjiang and Inner Mongolia in 2018, and Tibet in 2019. The five provinces represent the major pastoral region in China, accounting for 70% of China's total grassland according to the National Bureau of Statistics of China in 2019. To identify and choose the sample from the provinces, the research team adopted a stratified random sampling strategy to generate a sample of villages. In each province, we first identified the three most important grassland types according to their land areas and assign the counties to each grassland type. Then we divided all the counties with the same grassland type into two groups according to the grassland area per capita in the survey year. The grassland area per capita was calculated by dividing the total grassland area in a county by its rural population. As grazing on grassland is the major production activity in the pastoral area, grassland area per capita is highly related to herder income and other economic conditions. We, therefore, use this indicator to select the sample to ensure the sample is representative. One county is randomly selected from each group. As a result, we selected six counties in each province except Gansu and Inner Mongolia. We selected four counties in Gansu and five counties in Inner Mongolia. In total, we sampled 27 counties in the five provinces.

Three townships were selected from each county according to the per-capita grassland area. We divided all townships in each of these selected counties into three terciles according to the per-capita grassland area. One township was randomly selected from each tercile, which yields a total of 81 townships. One village was then randomly selected from the higher per-capita grassland area tercile and the other from the lower tercile of each selected township, which yielded a sample of 162 villages.

Structured survey questionnaires were designed to elicit information on the process of land use rights reform by interviewing village leaders. Appendix B provides detailed information on conducting the survey. Village leaders were also asked for information such as which years their village started to have access to the national electricity grid, internet, road, satellite TV, and package delivery. Figure S2 in Appendix A shows the percentages of villages that have access to these amenities over the years, reflecting a rapid improvement in the standard of living in these rural regions. All the villages in our sample have access to satellite TV, and about 80% of the villages have access to the national grid, internet, and rural roads. However, the access to package delivery is below 20% in the sampled area in 2018. These trends will be controlled in our regression analyses to exclude the potential influence of new information and technology or reduced transportation costs on grassland quality.

4 | EMPIRICAL MODEL

Recent literature shows the commonly used two-way fixed effects model may provide biased treatment effects. We calculate the weighted average of DID estimator following Athey and Imbens (2021) and de Chaisemartin and d'Haultfoeuille (2020). To provide a reference for understanding the weighted average treatment effects, we first set up the two-way fixed effects model as the base model.

4.1 | Two-way fixed effects model

We estimate a two-way fixed effects model to provide benchmark results. In the two-way fixed effects model, identification of the impact of grassland tenure reform comes from the cross-year variations in grassland quality in the treated villages, compared to the change in the control villages that have not received the treatment in a given year. The model is set up as follows:

$$y_{i,t} = \beta P_{i,t} + \tau S_{i,t} + \theta X_{i,t} + \eta_i + \eta_{ct} + \epsilon_{i,t}, \tag{1}$$

where $y_{i,t}$ is the dependent variable representing grassland quality in village i in year t, measured by the NDVI index in log form. $P_{i,t}$ is a dummy variable to indicate whether a village has started the process of privatization of grassland use rights but without security protection. Specifically, $P_{i,t}$ equals 1 when village i in year t has privatized grassland use rights but did not receive legal certificates or build fences (i.e., privatization of grassland use right only), and 0 otherwise. $S_{i,t}$ is also a dummy variable, indicating that grassland use rights are provided with additional protection and assurance through fences or legal certificates. Specifically, $S_{i,t}$ equals 1 if village i in year t has received certificates or built fences after the start of the privatization process (i.e., privatization with security protection), and 0 otherwise.

The vector $X_{i,t}$ controls for the time-varying characteristics at village i in year t, including whether the village has access to the national electricity grid, internet, road, satellite TV, package delivery, as well as the size of the rural labor force. The village-level fixed effects are captured by

η_i. We also control for county-by-year fixed effects η_{ct} to capture the common time trends for villages within the same county in a year. The idiosyncratic error term is denoted as $\epsilon_{i,t}$. The coefficient β represents the effect of privatization of grassland use rights without security protection, whereas the coefficient τ is of primary interest and represents the treatment effect after the completion of grassland use rights privatization (i.e., privatization of grassland use rights with security protection).

4.2 | Weighted difference-in-differences model

Recent literature shows that the treatment effect estimated using the two-way fixed effects model is not easily interpretable and cannot be regarded as the average treatment effect directly (e.g., Athey & Imbens, 2021; de Chaisemartin & d'Haultfoeuille, 2020). Specifically, when the estimated treatment effect is considered as the weighted sums of average treatment effects, the weights may be negative and may alter the sign of the estimated treatment effect. Let the average treatment effect τ be:

$$\tau = E\left(\frac{1}{N_s}\sum\nolimits_i \sum\nolimits_t I(D_{i,t} \neq D_{i,t-1})\left(y_{i,t}(1) - y_{i,t}(0)\right)\right), t \geq 2, \tag{2}$$

where $I(\cdot)$ is the indicator function, equal to 1 if $D_{i,t} \neq D_{i,t-1}$ and 0 otherwise; $y_{i,t}$ denotes the potential outcome of village i in year t with the treatment variable $D_{i,t} = 1$ (i.e., privatization with security protection), and $y_{i,t}(0)$ denotes the potential outcome of the village i in year t with the treatment variable $D_{i,t} = 0$ (i.e., before land tenure reform), and $N_S = \sum_i \sum_t I(D_{i,t} \neq D_{i,t-1})$. The treatment estimator τ represents the average treatment effects when a village has completed the process of privatization of grassland use rights with either physical security (i.e., fence) or legal security (i.e., certificate).

To account for potential bias generated by negative weights, we use the weighted average of DID estimators τ_w proposed in de Chaisemartin and d'Haultfoeuille (2020) to obtain interpretable estimates. In our context, once households in the village receive privatization of grassland use rights, households maintain the use rights thereafter, which effectively constitutes a staggered adoption design (Athey & Imbens, 2021; Athey & Stern, 1998). Thus, $D_{i,t} \geq D_{i,t-1}, \forall i,t,t \geq 2$. Following de Chaisemartin and d'Haultfoeuille (2020), we define

$$N_{d,',t} = \sum\nolimits_i I(D_{i,t} = d,\ D_{i,t-1} = d') \tag{3}$$

where $I(\cdot)$ is indicator function equals 1 if $D_{i,t} = d$ and $D_{i,t-1} = d'$, and 0 otherwise, d or d' is a binary variable indicating the treatment status and equals 1 if treated and 0 otherwise, $N_{d,d',t}$ is the number of observations with treatment d' at period $t-1$ and treatment d at period t added across all groups. The weighted average of DID estimator τ_w can be written as

$$\tau_w = \sum_{t=2}^{T} \frac{N_{1,0,t}}{N_S}\left(\sum\nolimits_i \frac{I(D_{i,t} = 1,\ D_{i,t-1} = 0)}{N_{1,0,t}}\left(y_{i,t} - y_{i,t-1}\right) - \sum\nolimits_i \frac{I(D_{i,t} = D_{i,t-1} = 0)}{N_{0,0,t}}\left(y_{i,t} - y_{i,t-1}\right)\right) \tag{4}$$

where $N_S = \sum_i \sum_t I(D_{i,t} \neq D_{i,t-1})$. The weighted average of DID estimator τ_w provides an unbiased estimation of the treatment effect τ under the stable groups and common trend assumptions. We provide more justifications on the assumptions in the result section.

4.3 | Dynamic and Heterogenous Impacts

As noted above, the privatization of grassland use rights in China is a complicated process and includes a combination of three policies (i.e., privatization of grassland use rights only, privatization of grassland use rights with fence, and privatization of grassland use rights with certificates). To test the robustness of our results, we analyze the dynamic impact of each policy separately by conducting an event study analysis. The event study is specified as below:

$$y_{i,t} = \sum\nolimits_{k=-3}^{-2} \tau_k I(k=t) + \sum\nolimits_{m=0}^{3} \tau_m I(m=t) + \tau D_{i,t} + \theta X_{i,t} + \eta_i + \eta_{ct} + \epsilon_{i,t}, \tag{5}$$

where τ_k are coefficients on the dummy variables for the years before the start of a policy and τ_m are the coefficients on the dummy variables for years after the policy. Due to the long study period, we collapse the years into several "bins" as some periods have only a few observations. Specifically, we use the years of 1–4 before the starting of a policy ($k = -1$) as the baseline and divide the period before the starting of the contract into three groups (i.e., $k = -1$ for 1–4 years before the starting of the policy; $k = -2$ for 5–9 years before the starting of the policy; $k = -3$ for 10 years before the starting of the policy). The post-policy period is also divided into three periods (i.e., $m = 0$ for the year when the policy started; $m = 1$ for 1–5 years after the starting of the policy; $m = 2$ for those of 6–15 years after the policy, and $m = 3$ for those over 15 years after the policy). The indicator variable $I(k=t)$ is equal to 1 when the observed period belongs to one of the six periods stated above and 0 otherwise. When conducting event study analysis for one policy, we control for the other two policy variables, denoted by $D_{i,t}$. All other specifications are the same as Equation (1).

We also analyze the heterogeneity effects by including additional interaction terms with the two explanatory variables $P_{i,t}$ and $S_{i,t}$. We check the heterogeneous effects from three perspectives. Note that some characteristics, such as the grassland land area in a village and ethnic groups, are highly stable across years in our sample size. Therefore, we no longer control for village fixed effects when conducting heterogeneous analyses. First, we use the proportion of ethnic minority groups in a village to denote the ethnic difference. Minor groups are often localized, whereas the Han group (the dominant ethnic group in China) generally migrated to the pastoral area. The migration group is expected to care less about their grassland quality than the locals. Second, we use grassland area to indicate the size of a village. A larger village may be difficult to form informal governance to assist the formal institutions (Li et al., 2021). We expect the positive impacts in improving grassland quality by the privatization of grassland use rights is larger in small villages as they can monitor each other to obey informal rules, if any. Last, we use grassland area per capita in a given year to indicate the resource endowment of the villages. Grassland area per capita is the average grassland area per capita during our data window and is calculated by dividing the average grassland area by the population at the village level. We expected that the privatization of grassland uses rights has a larger impact on improving grassland quality for the herders with larger grassland area per capita because herders with a larger grassland area may use grassland less intensively after privation compared to the herders with a smaller grassland area per capita.

5 | RESULTS

We first present the results from the two-way fixed effects model as a reference (Table 1). The dependent variable is NDVI in log form. We choose the log of NDVI as the dependent variable to reduce the impact of outliers and for the ease of coefficient interpretation. We also use the NDVI in absolute value as the dependent variable as a robustness check. Results are included in Table S1 in Appendix A and are consistent with results using NDVI in log form.

TABLE 1 Regression results from the two-way fixed effects model

	(1) log(NDVI)	(2) log(NDVI)	(3) log(NDVI)	(4) log(NDVI)
Privatization with use right only	0.0113 (0.0308)	0.00947 (0.0354)**		
Privatization with security protection	0.0410 (0.0241)*	0.0603 (0.0278)	0.0362 (0.0213)*	0.0508 (0.0285)*
Control variables	No	Yes	No	Yes
Village fixed effect	Yes	Yes	Yes	Yes
County-year fixed effect	Yes	Yes	Yes	Yes
N	4992	4143	4992	3965
Adj R-squared	0.947	0.950	0.947	0.949

Note: The ** and * denote significance at the 5% and 10% levels, respectively. Standard errors are clustered by village. Control variables include whether the village has access to national electricity, internet, road, satellite TV, package delivery, and the size of the rural labor force.

In Table 1, Column (1) presents the results when controlling for village-level fixed effects and county-by-year fixed effects. Column (2) adds time-varying controls in addition to the fixed effects controls in Column (1). To enable a clean comparison before the start (i.e., open access) and after the completion of the privatization process (i.e., privatization with security protection), we dropped the observations when a village in a specific year is in the privatization process but without certificates or fences (i.e., privatization with use rights only). The corresponding results without and with time-varying controls are presented in Columns (3) and (4), respectively. Results show that when grassland use rights were initially privatized to individual households without any physical (i.e., fence) or legal security (i.e., certificate), there are no detectable changes to grassland quality. Specifically, the coefficients of p are positive but insignificant in the first two columns. In contrast, the coefficients of S are positive and statistically significant in all four models, suggesting that when the privatized grassland use rights were enhanced by either physical or legal security, the NDVI was increased by 3.6%–6.0%.

As pointed out by Athey and Imbens (2021) and de Chaisemartin and d'Haultfoeuille (2020), the bias from the above two-way fixed effects model is more pronounced when there is a higher percentage of negative weights associated with the fixed effect. Our results show that τ_w is a weighted sum of 3709 average treatment effects on the treated, of which 2210 receive positive weights and 1499 receive negative weights. The negative weights sum up to -0.7230, indicating that the average treatment effects from the above two-way fixed effects model are biased upward. Therefore, the above results using the traditional two-way fixed effects are only suggestive, and the treatment effect estimated from the weighted DID estimator is more reliable after correcting for potential negative weights.

Based on the weighted DID estimator applied in a two-way fixed effects framework, our results show that $\tau_w = 0.0279$ with a standard error of 0.0152 (p-value $= 0.065$), suggesting a significant positive impact at a 10% level. Compared to the 3.6%–6% increase in grassland quality from the traditional two-way fixed effects model, this number is smaller but still significant from both statistical and economical perspectives. Our weighted DID estimate shows that implementing privatization of grassland use rights with security protection increases grassland quality by 2.79%.

To verify the results from the weighted DID estimator, we first test the stable group assumption. The stable group assumption requires that for each pair of consecutive years, there are groups whose treatment does not change. In our context, this assumption requires that there must be villages where households have not experienced changes in grassland use rights in years *t–1* and *t*, if the households in a village experience changes in grassland use rights from year *t–1* to *t*. This assumption is satisfied in our dataset. There are eight villages where households never started the process of privatization of grassland use rights during the data window. At the beginning of our data, there are

153 villages classified in the stable group from the year 1981–1982. However, the size of the stable group gradually decreases to 8 after the year 2012, suggesting most privatization of grassland use rights had been mostly completed by 2012.

Another important assumption for the weighted DID estimator is the common trend assumption. To test the plausibility of the common trend assumption, we conduct placebo tests by calculating a placebo estimator. The placebo estimator compares the outcome in year *t–1* in two sets of groups, including the observations that are untreated in year *t–1* but treated at year *t* and those untreated at *t–1* and *t*. The placebo estimator, τ_p, should not be significantly different from 0 if the common trend assumption holds. Figure 2 presents the estimation results as well as 95% confidence intervals for the placebo test and the weighted DID estimator. We find that τ_p=0.0010 with a standard error of 0.0179, thus highly insignificant. Our placebo results show that when we move the privatization of grassland use rights 1 year ahead, we no longer observe significant positive impacts. The counterfactual treatment effects are estimated tightly at zero, suggesting the plausibility of common trend assumptions on the potential outcomes.

We also conduct an event study for robustness check (Figure 3). We find that the change in NDVI was insignificant in either the period of 5–9 years before or 10 years before the privatization of grassland use rights without any security protection (Figure 3a). This indicates that the parallel trends assumption is likely to hold. Results are similar for the event study analysis based on the timing of fence and certificate treatment, respectively (Figure 3b,c). The event study analysis also suggests the dynamic effects of each policy. Figure 3a shows that privatization of grassland use rights without security protection has no significant effect on improving grassland quality at any period. Figure 3b,c show that fences and certificates have significant and positive effects on grassland quality. The positive effects persist for as long as 15 years after the land tenure reform policy was implemented, as the magnitudes of the coefficients are relatively stable. After 15 years, the impact of tenure reform combined with additional protections decreases although the effect remains positive (Figure 3b,c). Because the number of observations decreases as we expand the post-intervention time horizon, the estimations are less precisely estimated due to a smaller sample size. To address additional concerns on potential selection bias, we conducted a pretreatment parallel trend test in Appendix C. Results show that there is little difference between the villages in the control and treated groups before the privatization, mitigating the concerns about the potential selection bias.

We further conduct heterogenous analyses based on the two-way fixed effects model. Results are presented in Table 2. Our results show that the villages with different sizes of grassland areas and villages with different levels of grassland per capita do not experience differential grassland quality changes after the privatization based on the results in Columns (2) and (3), suggesting that the positive environmental outcome generated by grassland use rights are similar across large and small landowners. However, we find that the percentage of ethnic groups negatively moderates the estimated

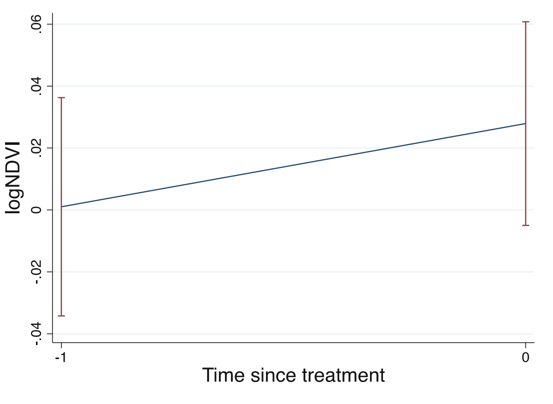
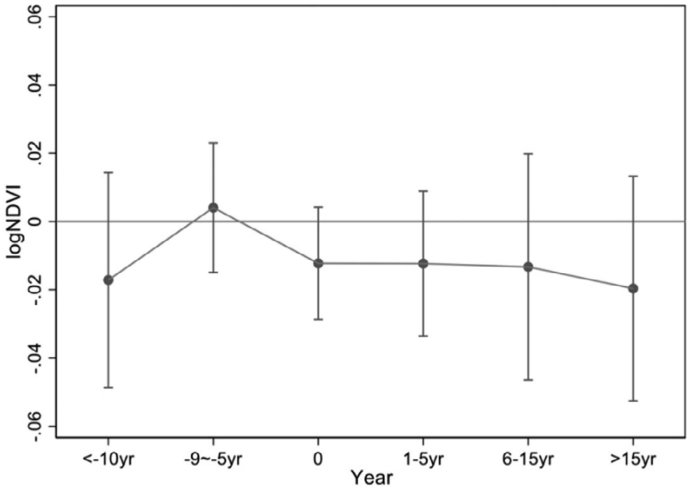
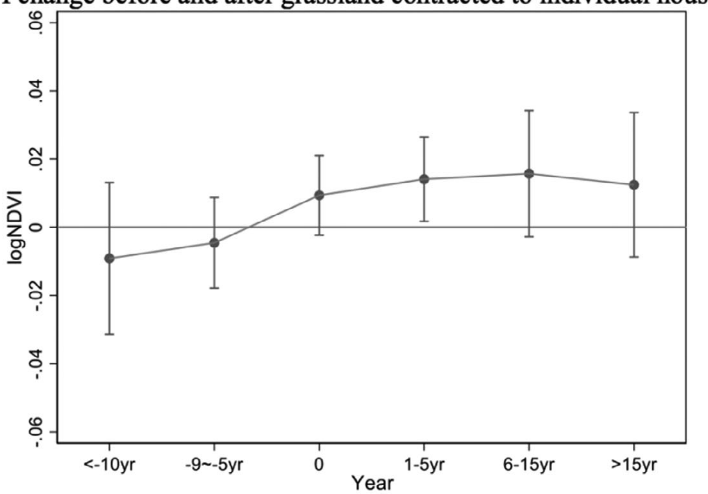


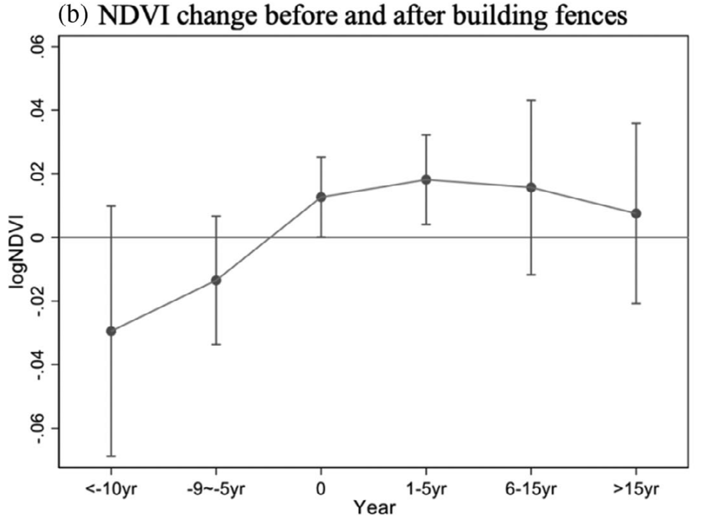
FIGURE 2 The impact of privatization on the NDVI change was estimated by the weighted DID and placebo test



(a) **NDVI change before and after grassland contracted to individual households**



(b) **NDVI change before and after building fences**



(c) **NDVI change before and after issuing certificates**

FIGURE 3 NDVI change before and after the initial operation

TABLE 2 Heterogenous analyses based on the two-way fixed effects model

Dependent variable: log(NDVI)	(1) Interaction variable: ethnic group	(2) Interaction variable: village size in terms of grassland area (thousand ha)	(3) Interaction variable: grassland area per capita (thousand ha per ha)
P (Privatization with use right only)	−1.011 (0.692)	0.0385 (0.0934)	0.0856 (0.0794)
S (Privatization with security protection)	0.715 (0.268)***	0.133 (0.0563)**	0.123 (0.0488)**
Interaction variable	0.666 (0.336)**	−0.000807 (0.000859)	−0.0972 (0.0428)**
P * interaction variable	1.083 (0.723)	−0.000718 (0.00165)	−0.627 (0.432)
S * interaction variable	−0.661 (0.276)**	−0.000482 (0.000818)	−0.0209 (0.0350)
Control variables	Yes	Yes	Yes
Village fixed effect	No	No	No
County-year fixed effect	Yes	Yes	Yes
N	4143	4143	4111

Note: The ***, **, * denote significance at the 1%, 5% and 10% levels, respectively. Standard errors are clustered by village. Control variables include whether the village has access to national electricity, internet, road, satellite TV, package delivery, and the size of the rural labor force.

treatment effect. The negatively significant coefficient of *S* interacted ethnic group, measured by the percentage of the minority population, suggesting that the incentives generated by the privatization of grassland use rights are weaker for the minority ethnic groups in Column (1). As indicated by collective action theory and associated empirical studies, the smaller size of the group and the more homogeneous the group, in terms of mutual dependence on and shared interests in the resource, the more likely collective action to succeed in managing common pool resources (Banks, 2003; Ostrom, 1990). This partially explains the smaller positive effects of privatization of land use rights on grassland quality for ethnic groups areas than the non-ethnic group. Note that the ethnic group, grassland area at the village level, and the grassland area per capital are all constructed as time-invariant variables at the village level. As a result, we have only controlled for the county-by-year fixed effects to mitigate the concerns for cross-sectional variations as well as differential time-trend at the county level when estimating the heterogeneous impacts of grassland ownership changes.

6 | DISCUSSION AND CONCLUSION

This paper examines the impact of land tenure reform on grassland quality in pastoral areas of China. Using nearly 40 years of remote sensing data combined with survey data on land tenure reform in the pastoral area of China, we investigated the impact of privatization of grassland use rights as well as the impact of privatization with physical or legal security protection in comparison to open access grassland with weak management. Before the grassland tenure reform, both grassland and livestock were owned and managed by People's commune, whereas production incentives were very low and agricultural products were in large shortage. At the beginning of the grassland tenure reform, livestock was allocated to individual households and grazed on open access grassland. At this stage, grassland was open access within one village but with a clear village boundary, which means that village members could use grassland freely in their own village but could not access grassland in other villages. Some villages might have grassland management measures such as informal institutions (Li et al., 2021). However, the major goal at this stage was still to incentivize production and improve economic development rather than protect the grassland ecosystem.

We find that the privatization of land use rights with low security protection has little impact on improving grassland quality, whereas enhanced privatization of grassland use rights with physical security such as fences or legal security such as use rights certificates increases grassland quality by about 3%. The magnitude of the grassland quality improvement is similar to a national ecological compensation program in China. The Chinese government invested over 25 billion US dollars during 2011–2020 through the ecological compensation program to improve the grassland quality. Results show that this program improved grassland quality measured by NDVI by 3%–5% (Hou et al., 2021), which is close to the magnitude caused by land tenure reform in this paper. On the cost side, our household survey data show that the fence cost about 2700 yuan per ha grassland on average, although the cost ranges from about 2000–6000 yuan per ha grassland depending on the location. The fences can last for about 10 years, suggesting the fence cost (about 200–600 yuan per ha per year) is higher than the average subsidized payment from the national ecological compensation program (i.e., about 120 yuan per ha grassland per year). In addition to improvement in grassland quality, fences have other functions such as reducing border conflicts between herders and villages. However, due to data limitations, it is difficult to provide a comprehensive cost benefit analysis for building fences to enhance the security of property rights

Existing literature points out that the impact of land tenure reform on environmental outcomes is context specific, and results may not be transferable in general. Our results imply that increased security and assurance of private land use rights are more likely to positively affect environmental outcomes. Our results are different from the qualitative analyses based on a case study in Inner Mongolia by Li et al. (2007), which concluded that the privatization of grassland use rights did not mitigate the tragedy of the commons and exacerbated grassland degradation. From a policy perspective, only implementing privatized land use rights without security assurance may undermine the positive environmental effects of land tenure reform. One caveat is that due to the short gap between fence and certification, we cannot separate the effects of fences and certification of title in our context. Future studies may separate these two effects when such data become available.

Note that our results are context specific. Our study focuses on the pastoral area of China, and the impact of privatization may differ in other regions that implement a similar use right change. In addition, grassland improvement may also be achieved from an enhanced CPR management practice where the local community and state government can negotiate to co-manage the resources, and the traditional social organization and system are respected (Li & Li, 2012). Li and Li (2012) also points out that establishing the excludability through the use right privatization may be an oversimplified measure due to the complexity of the grassland property rights and management practices in pastoral area of China. Furthermore, the outcome of CRPs may depend on the size of the grazing land as well as the cooperative. Based on our data, the average size of a privatized grazing land is about 7000 hectares. People's commune varies greatly in size, from 50 to 1000 households, but perhaps averaged about 300 households in Xinjiang (Hudson, 1938). It is possible that a smaller privatized grazing land or cooperative may lead to a better environmental outcome. Although our empirical results show a significant improvement on the grassland improvement through the use right privatizations with security protections, future research can compare our results with "improved," well-functioning CRPs to assist in policy decisions regarding grassland management.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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