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RESEARCH ARTICLE

Effects of precipitation on grassland ecosystem restoration under grazing exclusion in Inner Mongolia, China

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Abstract China launched the "Returning Grazing Lands to Grasslands" project about a decade ago to restore severely degraded grasslands. Grassland grazing exclusion was one of the experimental approaches for achieving the grand goal. Here, we evaluate the long-term regional ecological effects of grassland grazing exclusion in the Xilingol region of Inner Mongolia, China. The dynamics of grassland communities over 8 years (2004–2011) were continuously monitored at 11 research sites dominated by temperate steppe ecosystems. These sites represent the diverse landscapes of the Mongolian Plateau in the Arid, Semi-Arid, and Humid Climatic Zones that have varying precipitation levels. The community structure of degraded grasslands was found to recover quickly toward a benign state after grazing exclusion. The exclusion promoted an increase in mean plant community height, coverage, aboveground fresh biomass, and quality. The grasslands recovered fastest and most favorably in the Humid Zone followed by the Semi-Arid Zone and the Arid Zone. The increase in the aboveground biomass and vegetation height correlated significantly with the amount of total growing season precipitation. Precipitation therefore amplified the grazing exclusion effects on grassland restoration. Grazing exclusion was most effective in the relatively moist part of the study region. However, other factors such as global climate change and variability might have interacted with grazing management practices, thereby influencing the outcomes of grassland restoration efforts in Inner Mongolia. Future implementations of grassland ecosystem management should consider the regional climatic heterogeneity to maximize costs/benefits for achieving long-term ecosystem sustainability.

Keywords Climatic change · Community characteristics · Grassland grazing · Livestock exclusion · Temperate steppe

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Introduction

As the largest ecosystem type in China, grassland covers 393 million ha, or 42 %, of the national territory. China ranks second in the world in its amount of grassland, only behind Australia. During the past three decades, the grassland ecosystems in China have been seriously deteriorating under the combined effects of climate change, land use change, and socioeconomic transformation (Qi et al. 2012; Liu et al. 2013). In the arid and semi-arid regions in the Inner Mongolian Plateau, in addition to climate change, land use change and grassland overgrazing have been recognized as the key causes for the declines of grassland coverage and quality, loss of biological diversity, and degradation of ecosystem functions (Zhen et al. 2010; Cease et al. 2012; Li et al. 2012; Liu et al. 2013). Understanding the effects of the both human and natural driving forces behind grassland degradation has important ecosystem management implications in response to global change for the region (Li et al. 2012; Chen et al. 2013).

In 2003, in response to regional environmental concerns, the Chinese government launched an ambitious project called "Returning Grazing Lands to Grasslands" in northern, northwestern, and southwestern China, including the provinces and autonomous regions of Inner Mongolia, Xinjiang, Qinghai, Gansu, Sichuan, Tibet, Ningxia, and Yunnan (Liu et al. 2014). These grassland-dominated regions are all water-limited and vulnerable to both human and natural disturbances. The ultimate goal of the large project was to slow down overgrazing and thus reverse the severe grassland degradation trend. In this project, various degraded grasslands were fenced using a similar approach without considering climatic or ecosystem specifications. Some pastures were closed for several months each year for rotational grazing, while other pastures were fenced for livestock exclusion for 5–10 years and grazing was permanently prohibited. As of 2010, China has invested over $2 billion in total capital for grassland livestock exclusion with an area of 52 million ha. Meanwhile, grain subsidies were provided to local herdsmen for project implementation (China Ministry of Agriculture 2012).

Ecological restoration theories suggest that ecosystems can be potentially restored to their healthy states naturally with their own resilience under certain environmental and ecological conditions (Golodets et al. 2010; Cao et al. 2011). It is often too costly to artificially restore severely damaged ecosystems, if it is doable at all. Therefore, livestock exclusion, a relatively inexpensive approach for ecological restoration, has been widely used in grassland management as a primary approach for curbing grassland degradation and restoring damaged ecosystems toward a healthy state in Inner Mongolia (Yeo 2005; Zhang et al. 2005a; Li et al. 2013).

Field studies in the past have suggested that grazing exclusion plays a positive role in vegetation restoration, which can directly affect plant aboveground biomass, litter production, and root and soil development of grassland ecosystems (Liu et al. 2006; Teague et al. 2011; Vega and Montaña 2011; Heather et al. 2012). Compared to overgrazed grasslands, those under grazing exclusion generally show an increase in plant coverage and enhanced biomass production (Valone et al. 2002; Floyd et al. 2003; Yeo 2005). Several studies in China found that livestock exclusion had positive impacts on the grassland vegetation restoration in the arid and semi-arid regions of China (Zhang et al. 2005b; Xiong et al. 2011). However, these studies noted that the grazing effects varied in scale and magnitude. In general, three ecosystem restoration patterns, as quantified by plant community height, coverage, and biomass, have been observed. That is, increasing over time; initially increasing and then decreasing; and reducing initially, then increasing, but significantly reducing in the end (Li et al. 2013). Other studies suggest that the grazing exclusion practice alone is not likely to reverse the grassland degradation trend in the arid and semi-arid regions. Controversies remain on the positive effects of livestock exclusion on degraded grasslands and some argue that climate change also plays a dominating role and may have masked the human intervention to control grassland degradation (Zheng et al. 2006; Yeh 2010; Zhang et al. 2011; Li et al. 2012; Zhou et al. 2014).

Evaluating the effects of ecological restoration on a large scale is often challenging. Remote sensing (RS) technology has been widely used for regional evaluations of grassland restoration (Zhou et al. 2009). The conditions of grassland degradation have been classified through RS-based monitoring for changes in biomass, coverage, and dominant species of steppe communities (Davidson and Csillag 2003; Tong et al. 2004). Detections of species changes require high

spatial resolution or hyper-spectral RS technology (Pickup et al. 1994). Plant biomass and coverage have been used as two major indicators in the evaluation of large-area grassland vegetation change (Nicholson and Farrar 1994; Wessels et al. 2006). Both RS and ground monitoring studies suggest that vegetation dynamics closely correspond to climatic variability (Cao et al. 2013). However, few studies have examined how grazing exclusion and climate interact in the vegetation restoration processes on a broad scale in the arid and semi-arid regions.

Grazing exclusion has been widely implemented in the Xilingol grassland-the northern frontier of China. Both water and heat distribute unevenly in this region, resulting in large variations in primary productivity (Shao et al. 2013). The short growing season of grassland and the large variability of inter-and intra-annual precipitation explained the low and unstable grassland productivity (Zhou and Wang 2002). In the meadow steppe, the annual fluctuation of grassland productivity is generally less than 50 %, while a two- to four-fold change in ecosystem productivity in a typical steppe or desert steppe is not uncommon (Hao et al. 2003).

The ultimate goal of the ecological restoration is to restore grassland structure and its associated ecological functions. The structures of plant communities, grassland productivity, and species diversity are the key indicators of ecological restoration success (Christensen et al. 1996; Allen et al. 1997; Bradshaw 1997; Dobson et al. 1997). Numerous researchers have studied the grassland restoration processes in the Xilingol grassland. Previous work focused on community productivity, distribution patterns, species diversity, and soil physical, chemical, and biological properties (Li et al. 1994; Wang et al. 2001; Liu et al. 2002). These field studies suggested that the plant species and diversity of degraded grasslands increased and the community structure and dominance of species changed greatly after grazing exclusion. With the increase of grazing exclusion over time, plant coverage, density, biomass, and height reached a maximum and then declined (Shan et al. 2008). Studies on how external environmental factors such as climate change contribute to the effectiveness of grazing exclusion on a large scale are relatively rare. A climate change study by Lu et al. (2009) suggests that the Inner Mongolia region has been getting warmer and perhaps drier due to an increase in air temperature and variable precipitation over the past four decades. According to this study, air temperature increased at the rates of +0.41 °C per decade in the grassland and +0.39 °C in desert biomes from 1960 to 1990, which are larger than the rate of +0.27 °C in the forest biomes. Previous studies on the effects of livestock exclusion on grassland vegetation restoration in Xilingol were mostly conducted at a single experimental site (Liu et al. 2006; Shan et al. 2008) and mainly focused on certain steppe types such as *Leymus chinensis*, *Stipa baicalensis*, and *S. grandis*. Little is known about the effectiveness of grazing exclusion under different climatic conditions at landscape and regional scales.

In 2004, the China Meteorological Administration established a comparative monitoring study on plant community characteristics in Xilingol that spanned a wide range of precipitation regimes. *In situ* comparative monitoring data have been collected continuously at 11 grazing and fencing sites for more than 8 years. We analyzed this long-term monitoring data to better understand the regional processes and controls of the ecological restoration of degraded grasslands under grazing exclusions across three climatic zones.

Our overall objective was to quantify the role of precipitation in reversing grassland degradation from the perspective of the coupled natural and human system in Xilingol, Inner Mongolia. Specifically, we were to answer the following questions: (1) What is the restoration trend of a temperate steppe under grazing exclusion in different climatic conditions? (2) What is the magnitude of the responses of different parameters of community characteristics to grazing exclusion? (3) What are the differences of degraded grassland restoration among three ecological types under different climatic regimes?

Methods

Location and climate

The Xilingol grassland region (41–47°N, 111–120°E) is located in the central part of the Eurasian Steppe (Fig. 1) with a typical temperate steppe representing the major native grasslands of Inner Mongolia in northern China. Efforts have been made to preserve this typical temperate steppe with various levels of

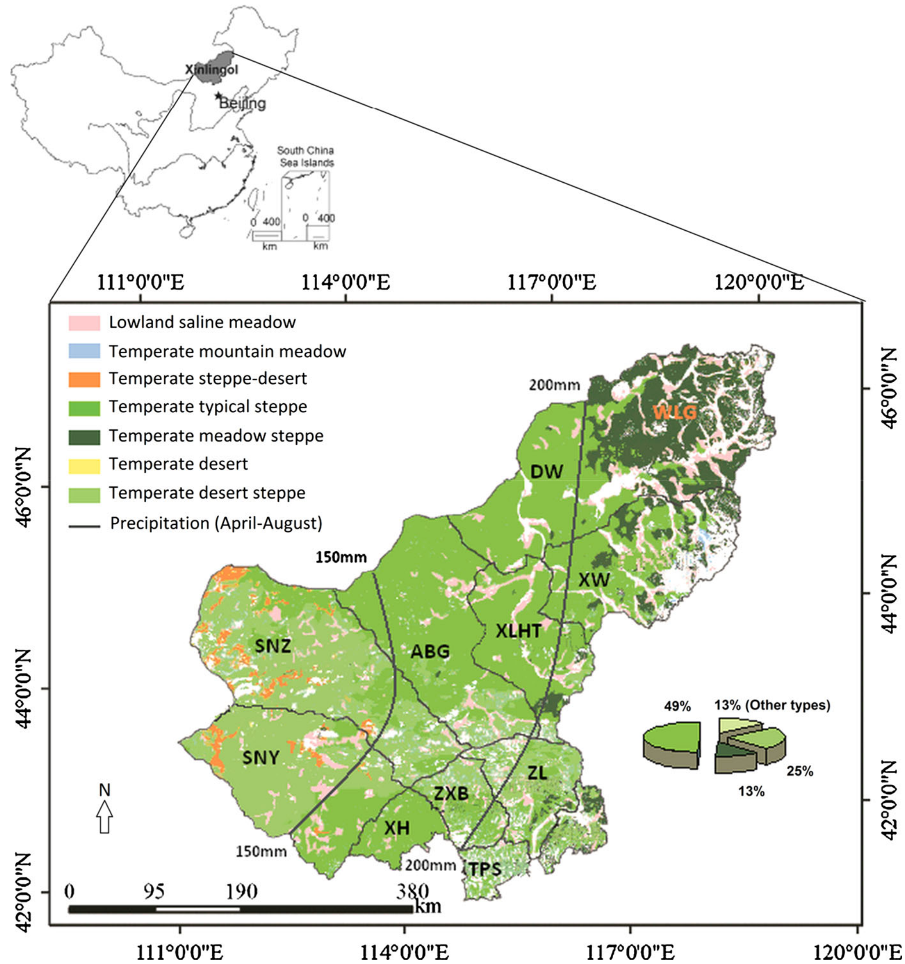


Fig. 1 The study area and observation sites in Xilingol region (modified from Batu et al. 2012). *SNY* Sonid Right Banner, *SNZ* Sonid Left Banner, *XLHT* Xilinhot, *ABG* Abag Banner, *DW* East Ujimqin Banner, *XW* West Ujimqin Banner, *XH* Bordered Yellow Banner, *ZXB* Plain and Bordered White Banner, *ZL* Plain Blue Banner, *TPS* Taibus Banner, and *WLG* Wulagai

success over the past few decades. The terrain is characterized by low hills with moderate slopes. The soils in this area follow an obvious pattern from the southeast to the northwest, ranging from chernozem to dark chestnut soils and light chestnut soils. The vegetation types in the research region include various formations of desert steppes, typical steppes, and meadow steppes (Liu et al. 2002). The region has a continental temperate climate with four distinct seasons characterized by long, cold winters and short frost-free summers. The annual average temperature is 2.4 °C with large annual and daily temperature fluctuations (Fig. 2a). Precipitation falls mainly in June, July, and August with large intra-annual variability (Fig. 2a). The annual precipitation gradually decreases from about 400 mm in the eastern part of the Xilingol region to about 200 mm in the western part, while the corresponding annual pan evaporation increases from 1,600 to 2,400 mm. The climate records at the Xilinhot Station indicated an increased trend in air temperature and decreased trend in precipitation during 1956–2011 (Fig. 2b). The study period of 2004–2011, when grassland restoration data were collected for this study, was considered as a dry period (Fig. 2c).

Zoning systems for regional analysis

The 11 research sites are Sonid Right Banner (SNY), Sonid Left Banner (SNZ), Xilinhot (XLHT), Abag Banner (ABG), East Ujimqin Banner (DW), West Ujimqin Banner (XW), Bordered Yellow Banner (XH), Plain and Bordered White Banner (ZXB), Plain Blue Banner (ZL), Taibus Banner (TPS), and Wulagai (WLG) (Fig. 1). In order to examine the coupled effects of grazing exclusion and climate, we divided

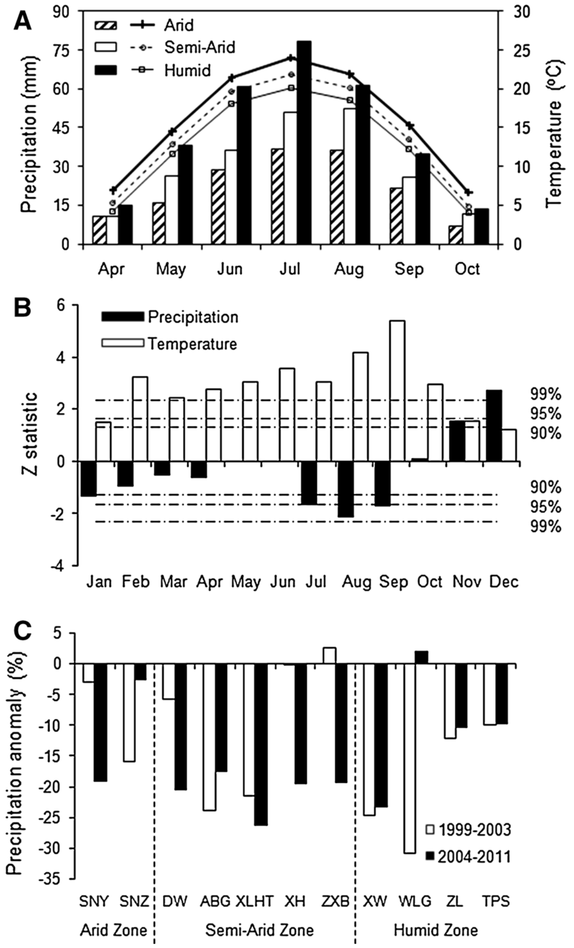


Fig. 2 The climate characteristics for **a** long-term (2004–2011) mean monthly precipitation and air temperature for three zones: Arid Zone, Semi-Arid Zone, and Humid Zone (*bars* for precipitation and *lines* for air temperature); **b** M–K trends for seasonal precipitation and temperature (1956–2011) at the XLHT meteorological observation station (*horizontal dashed lines* representing significance level of 90, 95, and 99 %); and **c** anomalies of grass growing season precipitation for two periods of 2004–2011 and 1999–2003 compared with the period of 1960–2000 at 11 research sites

the Xilingol grassland into three climatic zones according to their total growing seasonal (April–August) precipitation, P_t, during 2004–2011: Arid Zone ($P_t \leq 150$ mm) (SNZ and SNY); Semi-Arid Zone (150 mm $< P_t \leq 200$ mm) (DW, ABG, XLHT, XH, and ZXB); and Humid Zone ($P_t > 200$ mm) (XW, WLG, ZL, and TPS) (Figs. 1, 2a). As an alternative, we also divided the study region into three ecological types: desert steppe (SNZ and SNY), typical steppe (ABG, XLHT, ZL, ZXB, XH, and TPS), and meadow steppe (DW, XW, and WLG). We explored the different effects of the two classification approaches on interpreting the ecosystem responses to grazing exclusion practices on a regional scale.

Experiment design and monitoring

We conducted the comparative analysis at the 11 sites (Fig. 1) by contrasting the observed vegetation properties of the grassland in fenced (i.e., treatment) and grazing zones (i.e., control). Each research site (Fig. 1), about 5 km × 5 km in size, was located in a relatively remote area, away from roads and water bodies with no human interference but with easy access for management. The entire monitoring area at each site was divided into a grazing zone and a fenced zone to detect the restoration effects of the enclosure on grass height, aboveground biomass, and grass quality. The monitored grazing pastures represented the main pasture type, growing condition, and grazing intensity. The number of grazing animals in the research area is constant during 2004–2011. The four corners of each monitoring area were clearly marked with a precision GPS unit. Within the monitoring area, metal fences were used to form a 50 × 50 m non-grazing plot separated from the rest of the area subject to normal grazing by sheep. The fenced plot had four 1 × 22 m subplots to monitor grass development and height and four 1 × 1 m subplots to monitor coverage and aboveground biomass. We used 2 m buffers to separate two adjacent subplots and the subplots from the metal fences.

The field data were collected between the 25th and 30th day of each month during 2004–2011. Here, we used the data from the month of August because August represented the peak of grass growth in the region. The aboveground vegetation characteristics were monitored, including the community and species compositions, community species heights, community coverage, community aboveground fresh biomass, and proportion of high-quality herbage. In this paper, high-quality herbage was referred to as the grass that had a feeding evaluation ranking of good to excellent (Chen 1979). Before the plant biomass was harvested and weighed, a visual inspection was first conducted within the quadrat and then the coverage of the mixed grass and the community coverage was measured (Qin et al. 2006). We acquired daily air temperature and precipitation during the grass growing season for all

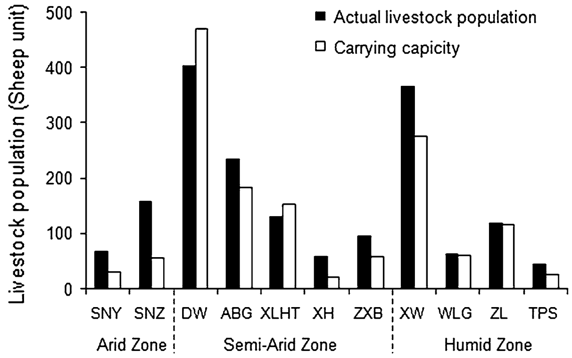


Fig. 3 Actual livestock population and carrying capacity at the end of 2002 in Xilingol. The carrying capacity was calculated based on Li et al. (1994) and the actual livestock population was obtained from statistics of animal husbandry in Inner Mongolia

11 sites in Xilingol during 2004–2011 from the China Meteorological Administration.

Statistical analysis

Both the absolute and relative values of the monitored vegetation properties were used to evaluate the effects of exclusion. The relative restoration rate of each variable was defined as the differences of the measurements between the fenced and grazed conditions divided by the values under the grazing condition (i.e., control). For example, the annual change rate for the height of the mixed grassland (H_{RD}) was calculated as:

$$H_{RD} = \frac{H_f - H_g}{H_g} \times 100\,\%, \quad (1)$$

where H_f and H_g are the community heights measured in the fenced area and grazing area, respectively.

We estimated grassland carrying capacity (Fig. 3) using a classification system for 18 types of steppes developed based on a study of community composition and population biomass of 120 plots of steppe in the central-eastern part of the Mongolian Plateau (Li et al. 1994). The overloading index, I_o, was used to quantify the severity of overgrazing by livestock:

$$I_0 = \frac{P_a - P_c}{A_g}, \quad (2)$$

where P_a is the actual livestock population (Sheep unit), P_c is the carrying capacity (Sheep unit), and A_g is the area of available grassland (ha).

The paired t test was used to determine whether vegetation was changed with and without grazing. A one-way analysis of variance (ANOVA) was also used for the single-factor analysis of variance, Duncan analysis, and least significant difference (LSD) to compare the differences of multiple samples of three climatic zones or three ecological types. The Pearson Correlation procedure was applied to assess the relationships among different factors. We used Mann–Kendall statistics to detect temporal trends of variables.

In lieu of experiment replications for each site, we compared the biomass of each of the 8 years with the baseline year (2004) for each non-grazed/grazed pair at each site. The statistical power for aboveground biomass of the designed experiment was also calculated by G*Power 3.1.9.2 (Faul et al. 2007, 2009). Conventionally, a test with a power greater than 0.8 (or $\beta \leq 0.2$) is considered statistically powerful (Mazen et al. 1985). We found that the calculation power values were higher than this threshold value at 7 of the 11 sites, indicating a sufficient sample size and high experiment precision.

Results

Biomass change

Grass biomass for both the grazed and fenced plots dramatically increased at 5 of the 11 sites and decreased significantly at three sites (TPS, ZL, ZXB) when compared to the baseline of 2004 (Fig. 4). Biomass responded differently to climatic variability between the grazing types and between sites. For example, biomass increased in the fenced area but decreased under grazing at the SNY site. However, opposite changes occurred at the TPS site. No difference was found at the ZXB site. Both grazing types showed a decreasing trend from 2004 to 2010 and showed an abrupt increase in the wet year of 2011. The amount of grass biomass in the non-grazed area was higher than the grazed area at 6 of the 11 sites (Fig. 4).

Vegetation restoration

After 8 years of exclusion, the grassland community generally moved toward a benign state at most sites, as gauged by both absolute (Fig. 5a) and relative (Fig. 5b) ecosystem structure indictors including

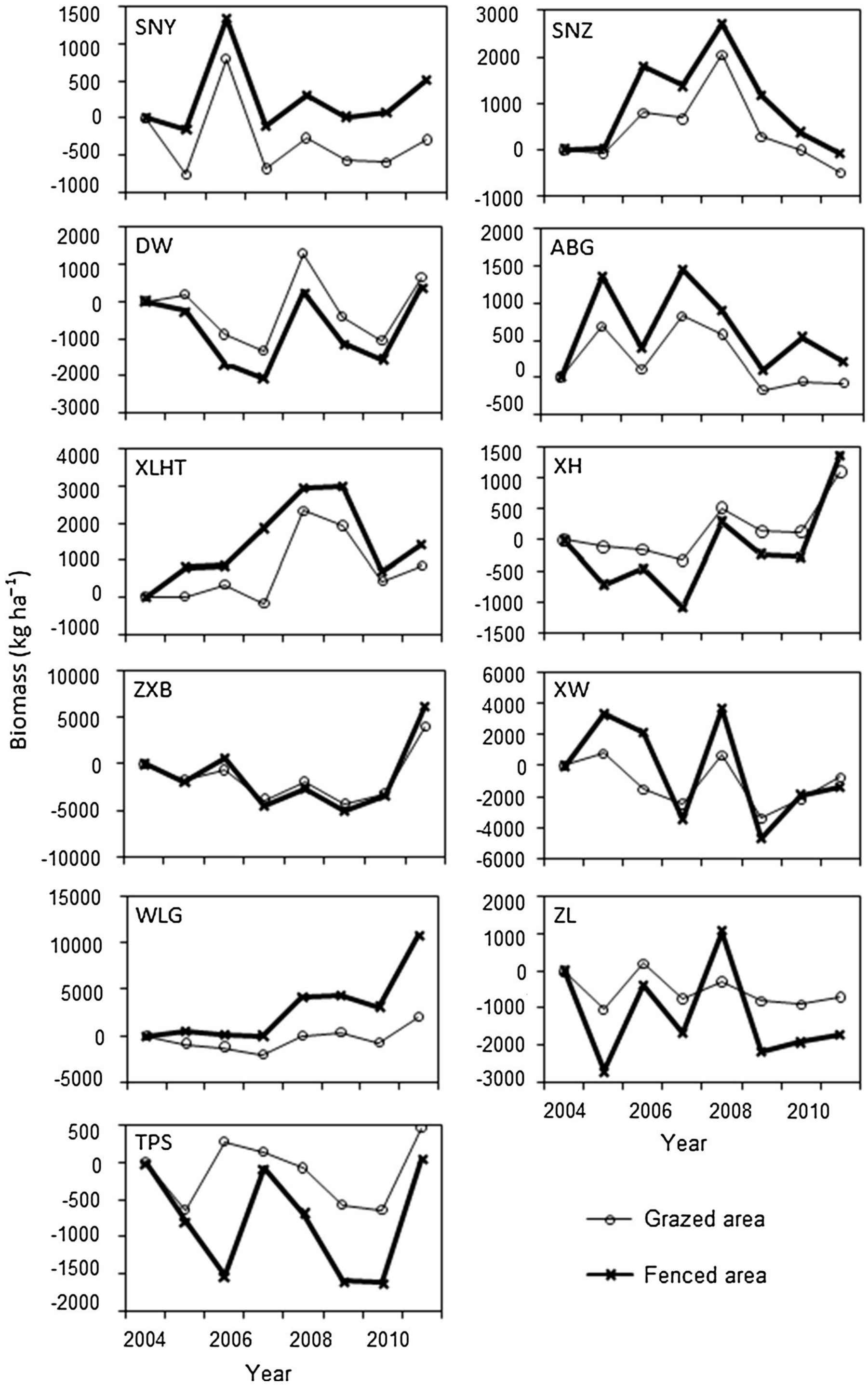


Fig. 4 Changes of aboveground biomass over time (2004–2011) with year 2004 as the baseline for each pair (grazing vs. non-grazing) at 11 monitoring sites. Negatives indicate a decrease in biomass and positives indicate an increase

height, coverage, aboveground biomass, and high-quality herbage. The absolute change values (or relative restoration rates) increased significantly ($p < 0.05$) by 200–2,500 kg ha^{-1} (5–270 %) for community biomass, about 1–11 cm (10–150 %) for the height, and 1–30 % (10–100 %) for the coverage

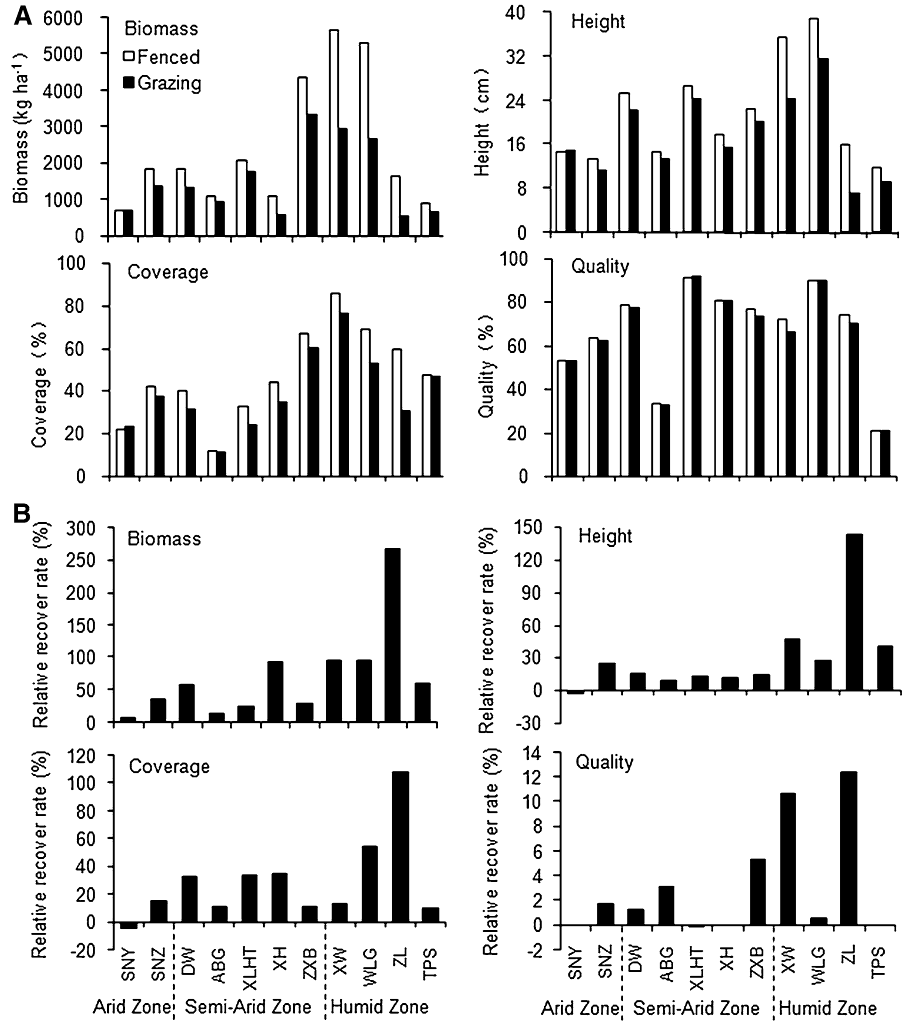


Fig. 5 Eight-year (2004–2011) means of (**a**) absolute restoration value and (**b**) relative restoration rates of community height, coverage, aboveground fresh biomass, and grass quality between grazing and fenced areas in all 11 sites

Table 1 The *p* value determined by the matched-pair t-test to examine differences in ecosystem structure between grazing and non-grazing areas for the period of 2004–2011

Vegetation parameter	Arid Zone		Semi-Arid Zone					Humid Zone			
	SNY	SNZ	DW	ABG	XLHT	XH	ZXB	XW	WLG	ZL	TPS
Biomass	>0.1	0.008	0.004	>0.1	>0.1	0.001	0.001	0.000	0.005	0.000	>0.1
Height	>0.1	0.030	0.004	>0.1	0.077	>0.1	0.025	0.001	0.015	0.000	0.002
Coverage	>0.1	0.029	0.001	>0.1	>0.1	0.013	0.011	0.001	0.034	0.000	>0.1
Quality	>0.1	>0.1	>0.1	>0.1	>0.1	>0.1	>0.1	>0.1	>0.1	>0.1	>0.1

at seven sites (Table 1). No significant differences were found for any ecosystem indicators at the SNY site. The proportion of high-quality herbage increased by 0.5–4 % (1–15 %) at seven sites, but decreased at SNY, XLHT, XH, and TPS.

Both the maximum absolute and relative restoration of biomass and height occurred in the Humid Zone (XW, ZL, and WLG). The maximum absolute increase of coverage was found at two sites in the Humid Zone (ZL and WLG) and one site in the Semi-Arid Zone

Table 2 Pearson Correlation statistics for correlations between the eight-year mean relative restoration rate (R_f) and precipitation during the grass growing season (P_t) as well as R_f and the overloading index in 2002 (I_o) of 11 sites ($n = 11$)

Tested variables	Test	Biomass	Height	Coverage	High-quality
R_f and P_t	Correlation	0.60	0.62	0.52	0.40
	Significance (p value)	0.05	0.04	0.10	0.22
R_f and I_o	Correlation	−0.12	−0.06	−0.39	−0.09
	Significance (p value)	0.37	0.86	0.23	0.80

(XH). The maximum relative restoration of height occurred at three sites in the Humid Zone (ZL, XW, and TPS) (Fig. 5a, b). Recoveries at ZL, XW, and WLG were the largest, with improvements by about 100–270 % for biomass, 30–150 % for height, and 15–100 % for coverage. The improvements were relatively small at SNY and ABG and not obvious at SNY. This result suggested that the relative restoration was similar to the absolute restoration process, but some differences existed due to the differences in baseline conditions among the 11 sites.

Potential factors controlling vegetation restoration

Both the mean plant biomass and height responses were positively and ($p < 0.05$) correlated with precipitation ($R = 0.60$, $p = 0.049$ and $R = 0.62$, $p = 0.042$, respectively) (Table 2). Changes in coverage and grass quality were not strongly influenced by precipitation ($R = 0.52$, $p = 0.10$ and $R = 0.40$, $p = 0.224$, respectively). However, the negative correlations between the previous overloading index and relative restoration rates for all four vegetation characteristic parameters were weak ($R = -0.12$, $p = 0.37$; $R = -0.06$, $p = 0.86$; $R = -0.39$, $p = 0.231$; and $R = -0.09$, $p = 0.80$, respectively).

Restoration among the climatic zones

Different climatic zones responded distinctly to grazing exclusion experiments. In contrast to the Semi-Arid Zone and the Humid Zone, sites in the Arid Zone showed no obvious improvements for height, coverage, aboveground biomass, or high-quality herbage, especially at the western site of this zone (SNY). The aboveground biomass in two of the three climatic zones responded significantly ($p < 0.01$) to exclusion in terms of both absolute (Fig. 6a) and relative restoration (Fig. 6b). In the fenced areas, the eight-year annual means of aboveground biomass in the Humid Zone, the Semi-Arid Zone, and the Arid Zone were 3,369, 2,087, and 1,274 kg ha^{-1}, respectively, representing a significant increase in comparison with the control (grazing) areas. On a relative term, the biomass growth in the Semi-Arid Zone and the Humid Zone increased significantly by 52 and 141 % ($p = 0.0$). The biomass increase in the Arid Zone, however, was only 21 %, which was not statistically significant ($p = 0.1$).

The community coverage and height also noticeably improved in all three climate zones after exclusion (Fig. 6). The vegetation coverage of the Humid Zone increased significantly by 57.2 % ($p = 0.00$). Similarly, vegetation coverage increased significantly in the Semi-Arid Zone by 22.4 % ($p = 0.00$). However, the responses in the Arid Zone were small (5.7 %) and were not statistically significant. The community height in the Humid Zone also improved significantly ($p = 0.005$) by 71 %. The restoration effects on community height in the Semi-Arid Zone and the Arid Zone were not statistically significant (18 and 11 %).

Restoration trend by climatic zones

In the Semi-arid Zone, the peak values of the annual relative restoration rate for community height, coverage, and aboveground biomass all occurred in the first 4 years (2004–2007), reaching a relatively high value around the fourth year, followed by a period with a stable or decreasing trend (Fig. 7). For example, the annual mean growth rate of aboveground biomass at XLHT increased after exclusion, reaching a relatively high value in the fourth year (2007) and then decreased over the following 4 years. The vegetation coverage and height also had synchronous changes. In 2010, compared with the grazing area, community height in the fenced area of XLHT and XH as well as the

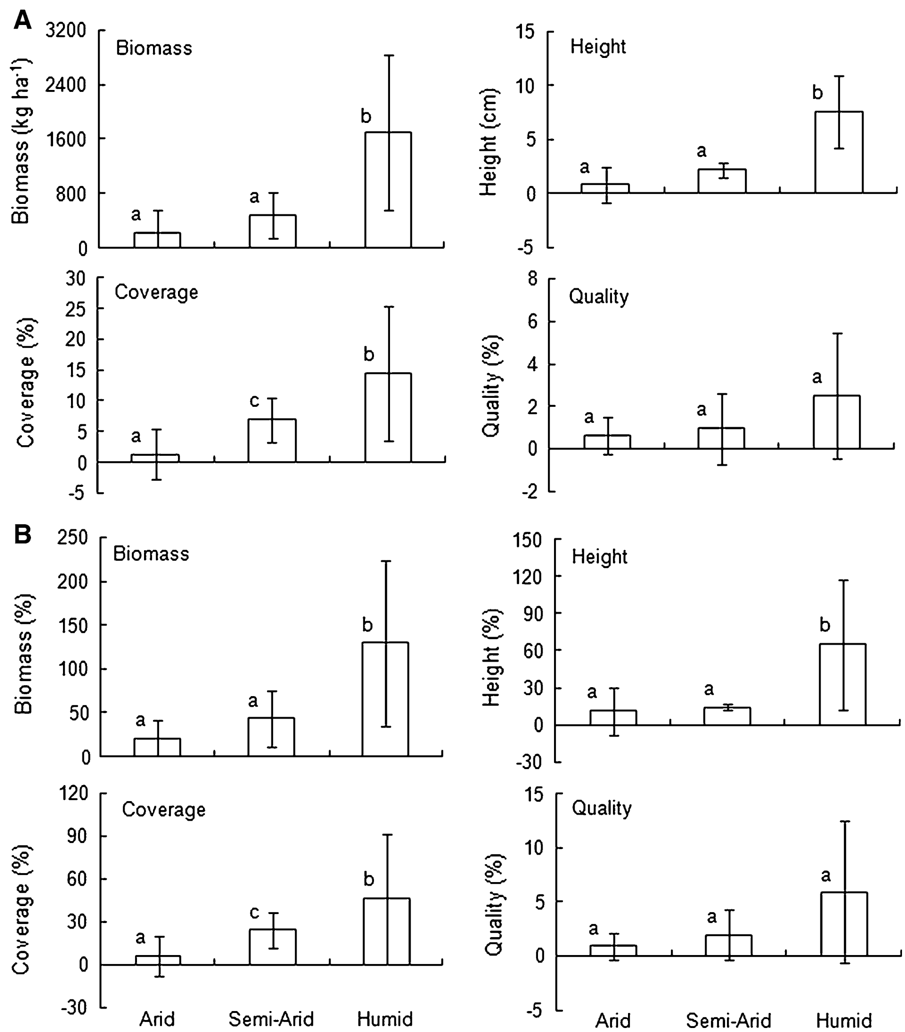


Fig. 6 Eight-year (2004–2011) means of (**a**) absolute restoration value and (**b**) relative restoration rates of community height, coverage, aboveground fresh biomass, and grass quality due to exclusion in three climatic zones: Arid Zone, Semi-Arid Zone, and Humid Zone. Different letters indicate whether the difference among the three different climatic zones is significantly different at $p < 0.001$ (*a* and *b*) or $p < 0.05$ (*a* and *c*, *b* and *c*). The vertical lines are for standard deviation

aboveground biomass of XLHT reduced instead. The harvest for residual biomass from prior years represented the accumulation of biomass and was likely the causal mechanism for this temporal trend pattern.

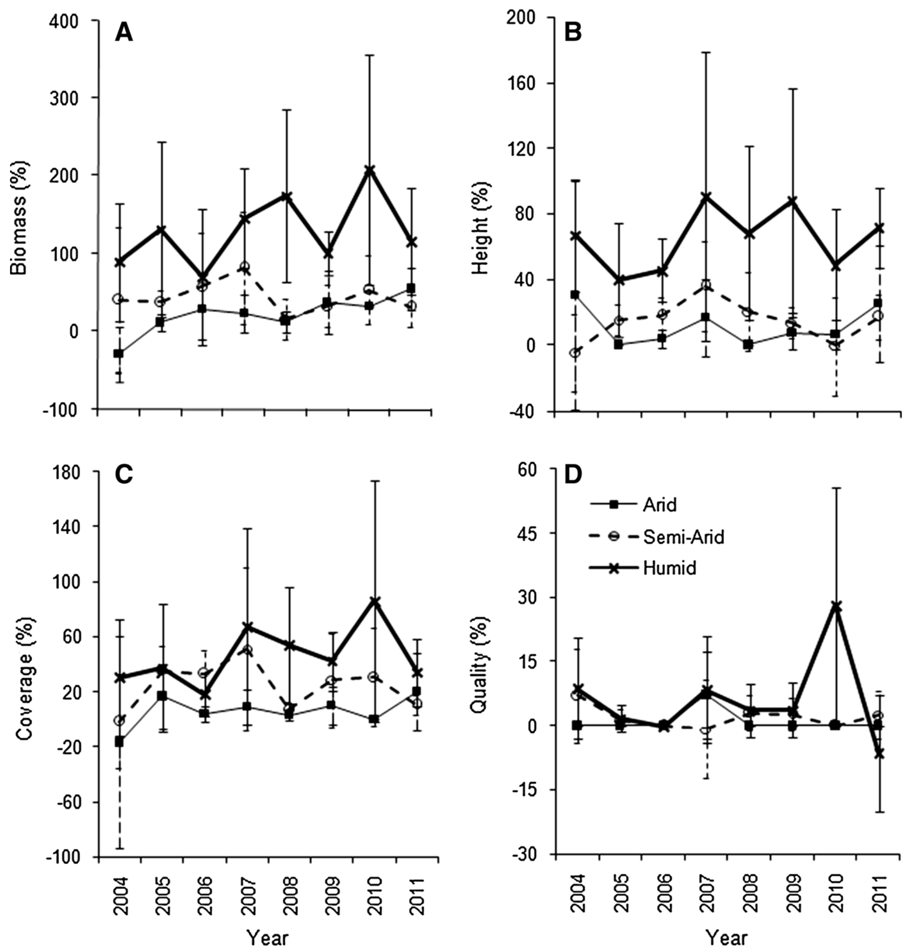
The restoration trend in the Humid Zone was different from that in the Semi-Arid Zone. Biomass, coverage, and grass quality peaked later in the Humid Zone than in the Semi-Arid Zone. In the Arid Zone, there were no obvious overall changes for grass height, coverage, or aboveground biomass.

Although the proportion of high-quality herbage continuously increased over time under exclusion, the changes of annual relative response rates diminished over time at all sites. Over the period of exclusion, the proportion of sedge (*Carex liparocarpos*) and *Allium polyrrhizum* decreased while the dominant positions of *Stipa* (*Stipa capillata*) and guinea grass (*Leymus chinensis*) constantly strengthened. For example, the absolute height of sedge at XW was higher over time than that of guinea grass, but it later became similar and eventually much lower than that of guinea grass (Fig. 8a). However, with the increase in the fenced time, the absolute height of *Allium polyrrhizum* at SNZ was much larger than that of *Stipa* in 2004, but almost equal in 2008, and lower thereafter (Fig. 8b).

Vegetation restoration among ecological types

The restoration effects of grazing exclusion did not show a clear pattern in terms of their dependence on the ecological types of meadow steppe, typical steppe, and desert steppe. Community height, coverage,

Fig. 7 Trends of the annual relative restoration rate of community aboveground fresh biomass (**a**), height (**b**), coverage (**c**), and grass quality (**d**) due to exclusion (2004–2011). The *vertical lines* are for standard deviation



aboveground biomass, and high-quality herbage all have shown high dependence on climate zones. The differences in relative restoration rates for height among the three ecological types were significant only at $p = 0.013$, which is much lower than that for climatic zones ($p = 0.0$) (Fig. 9). Thus, using climate as a zoning method appeared to be a better approach than the traditional ecological type method in detecting the effects of grazing exclusion practices.

Discussion

The Xilingol region experienced grassland degradation and desertification over the past two decades. The causes were rooted in overgrazing driven by the one-sided pursuit of economic interests coupled with climate warming and extreme weather events such as drought. During 1996–2000, with the dramatic increase in livestock population, the aboveground biomass decreased sharply (Table 3). During 2002–2006, with the implementation of the ecological environment protection policy, the livestock population was somewhat controlled. However, due to periodic droughts, the grass biomass did not recover better than that of 1996–2000. This illustrated the coupled and interactive impacts on community aboveground biomass (Table 3).

The eight-year field monitoring data for the large region provided a rare opportunity to examine the combined effects of grazing exclusion and spatial climate variability on ecosystem dynamics. The data analysis described above suggested that the degraded grassland community structure could generally quickly develop toward a benign state across the study region. The main reason was that the degraded grassland had experienced severe overgrazing before the exclusion treatments and the growth and

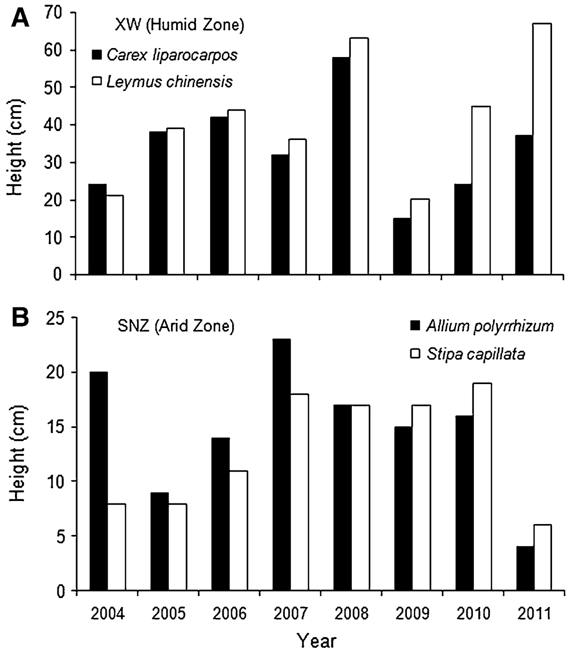


Fig. 8 Differences of the responses of herbage height to exclusion at **a** XW and **b** SNZ with different climates during 2004–2011

development of vegetation was inhibited due to long-term extensive damage by livestock (Table 3). Livestock exclusion eliminated the effects of grazing disturbances and extended the time for the grass to recover to improve vegetation community structure and increase aboveground biomass. Our findings agreed with the results from previous studies that enclosures in the degraded grassland would increase community coverage, height, aboveground biomass, and the proportion of high-quality herbage, reduce poisonous plants, and remarkably improve grass yields (Wang et al. 2001; Liu et al. 2002; Shan et al. 2008).

The new knowledge learned from this study was that there were regional differences in response to grazing exclusions and the restoration rate was influenced by the precipitation regime and previous livestock grazing history. Precipitation during the growing season had a significant positive impact on the restoration of the community aboveground biomass and height. The Humid Zone, with more precipitation, recovered most, followed by the Semi-Arid Zone, and then the Arid Zone. Thus, the effectiveness of grazing exclusion was impacted by precipitation during the grass growing season; the more the precipitation, the faster the restoration.

It is perhaps not surprising that water availability (i.e., growing season precipitation) was a major driver for the restoration of temperate grasslands on the Mongolian Plateau. The dry habitat had high growing season temperatures and high water loss by evapotranspiration and thus the distribution and growth of plants was predominantly determined by precipitation (Chen and Wang 2000; Shao et al. 2013).

The effects of water availability were especially pronounced for the Arid Zone, where the variability of

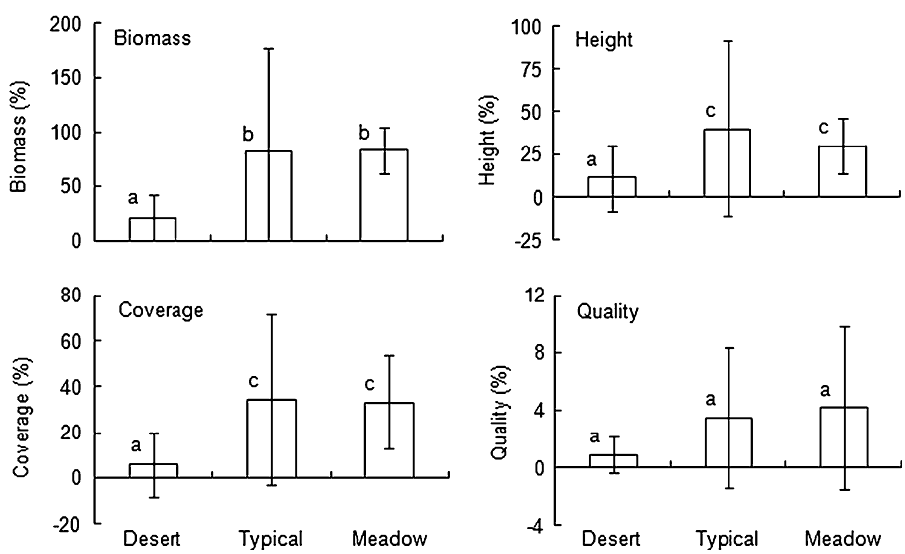


Fig. 9 Eight-year (2004–2011) means of the relative restoration rates of community height, coverage, aboveground fresh biomass, and grass quality due to exclusion in three ecological types: Desert Zone, Typical Zone, and Meadow Zone. Different letters indicate whether the difference among three ecological types is significantly different at $p < 0.001$ (*a*, *b*) or $p < 0.05$ (*a* and *c*, *b* and *c*). The *vertical lines* are for standard deviation

Table 3 Dominant plants, elevation (E), livestock population per area of available grassland (LP, calculated based on the statistics of animal husbandry in Banners of Xilingol), aboveground biomass (BIO, remote sensing data obtained from the Inner Mongolia Grassland Survey and Design Institute), mean precipitation for grass growing season (P_t, Inner Mongolia Meteorological Observation Station), and standard error for P_t (SE) during the periods of 1984–1992, 1996–2000, and 2002–2006 at 11 sites

Zone	Sites	Dominant plants	E (m)	Year	P_t (SE) (mm)	LP (head ha^{-1})	BIO (kg ha^{-1})
Arid Zone	SNY	*Stipa capillata, Ephedra sinica*	1,102	1984–1992	139 (13.7)	0.28	544
				1996–2000	188 (27.2)	0.43	317
				2002–2006	143 (29.2)	0.25	309
	SNZ	*Stipa capillata, Convolvulus ammannii*	1,037	1984–1992	136 (15.9)	0.21	357
				1996–2000	141 (18.9)	0.37	264
				2002–2006	140 (23.7)	0.23	258
Semi-Arid Zone	DW	*Leymus chinensis, Stipagrandis P. Smirn.*	839	1984–1992	223 (24.1)	0.32	1,333
				1996–2000	231 (49.6)	0.57	771
				2002–2006	179 (29.3)	0.56	699
	ABG	*Leymus chinensis, Stipa capillata*	1,126	1984–1992	195 (16.6)	0.29	607
				1996–2000	227 (55.4)	0.57	379
				2002–2006	175 (19.4)	0.43	367
	XLHT	*Leymus chinensis, Stipa krylovii*	1003	1984–1992	210 (15.6)	0.44	1,115
				1996–2000	238 (45.1)	0.78	478
				2002–2006	187 (42.8)	0.60	456
	XH	*Leymus chinensis, Stipa breviflora*	1,322	1984–1992	220 (20.0)	0.79	750
				1996–2000	220 (20.1)	1.00	483
				2002–2006	202 (28.0)	0.81	457
	ZXB	*Leymus chinensis, Stipa capillata*	1,346	1984–1992	276 (15.4)	1.12	1,436
				1996–2000	258 (43.0)	1.08	432
				2002–2006	289 (36.6)	0.86	409
Humid Zone	XW	*Carex liparocarpos, Leymus chinensis*	1,001	1984–1992	263 (18.9)	0.51	1,753
				1996–2000	298 (63.2)	0.84	787
				2002–2006	205 (13.4)	0.76	733
	WLG	*Leymus chinensis, Stipa capillata*	860	1984–1992	303 (25.9)	–	–
				1996–2000	230 (60.9)	–	–
				2002–2006	221 (22.7)	–	–
	ZL	*Leymus chinensis, Agropyron cristatum*	1,300	1984–1992	298 (32.8)	0.93	1,251
				1996–2000	288 (23.4)	0.76	618
				2002–2006	294 (42.6)	0.47	618
	TPS	*Leymus chinensis, Stipa capillata*	1,469	1984–1992	303 (26.6)	1.80	1,304
				1996–2000	291 (26.0)	1.34	404
				2002–2006	303 (26.0)	1.85	397

annual precipitation was high and droughts were common. Therefore, some disturbance factors such as overgrazing would likely result in much more severe ecosystem degradation. Our study period of 2004–2011 represented a period drier than normal (Fig. 2c) and the Arid Zone received much less precipitation than the long-term average. In addition, grasslands in this zone degraded severely before the exclusion. Ecosystems in this zone would have difficulty recovering or have a hard time being restored under coupled stressors of human disturbances and natural factors. For the Semi-Arid Zone

and the Humid Zone, with relatively favorable soil moisture conditions, it was relatively easier to restore the degraded ecosystems.

For sites that experienced successive droughts (e.g., ABG, XLHT) before or during exclusion (Fig. 2c), the ecosystems indeed recovered slower. During 1999–2001, XLHT and DW suffered successive droughts and locust plagues (Zhang et al. 2006), causing serious grassland degradation. Perennial grass almost disappeared and vegetation became sparse. Both of these sites had slow recoveries with only little improvement of pasture vegetation after exclusion (Fig. 5). The actual livestock population in 2002 for the XLHT site did not exceed its carrying capacity (Fig. 3), but ecological restoration was limited. Compared with XW, ZL, and WLG, TPS also recovered slowly, mainly because it was the most serious overloading area in the entire Xilingol region (Fig. 3). All of these findings were good examples that coupled impacts of the previous disturbances and prolonged droughts might have contributed to the slow restoration at some sites.

The strong correlations between the growing season precipitation and relative restoration rate of biomass confirmed our hypothesis that climate was a dominant factor in influencing the role of grazing exclusion in grassland restoration. Our study suggests that traditional grassland restoration pathway planning only by ecological types may not be as appropriate as using precipitation as a key additional indicator. In this study, the climatic zones were divided by the total precipitation during the grass growing season. During the study period, the climate fluctuated dramatically. There was an increasing trend in air temperature and a decreasing trend in precipitation in these zones (Fig. 2b). The observed climatic variability might partially explain why the responses of degraded grasslands to grazing exclusions could be better explained by climatic zones than ecological types.

Our results also showed that the relative restoration rates for height, coverage, and aboveground biomass peaked around the fourth year in the Semi-Arid Zone. However, in the Humid Zone, which received more precipitation than the other two zones, biomass, coverage, and grass quality peaked later than in the fourth year. The Arid Zone had no obvious trend for any of the four characteristic parameters. The contrasts among these zones indicated that there was large variability of recovering potentials among the three zones and grazing management practices should be designed accordingly. The mechanisms for the occurrence of peaks around the fourth year have yet to be understood.

We found that precipitation might have amplified the effects of exclusion for both magnitude and trend in the Xilingol region where moisture was a dominant environmental factor. There is sufficient evidence that the Inner Mongolia region is getting warmer and perhaps drier, either due to increases of water loss potential or water resource exhaustion due to human activities (e.g., mining and groundwater withdrawal). Water shortages will bring more challenges to future grassland management. More stringent and bold grazing management policies may be necessary. The implementation of livestock exclusion should be better targeted from the perspective of the coupled natural and human system to prevent a one-size-fits-all management mode, either in time or space.

Besides climate-induced changes in water availability, our data suggested that the effectiveness of grazing exclusion was also influenced by grassland conditions prior to exclusion. We illustrated that the magnitudes of previous overgrazing activities might have some negative impacts on the restorations in some areas. Overall, our data suggested that as long as water and vegetation conditions were reasonably good, the grasslands in these regions could recover quickly with grazing exclusion practices. Further testing regarding the effects of livestock sizes in different zones should be carried out.

Conclusions

Our long-term regional monitoring data indicated that grazing exclusion was an effective measure for curbing the natural grassland degradation in Inner Mongolia. However, grazing exclusion may not be a permanent solution and this management practice should not be generalized across the Inner Mongolia landscapes because different responses were observed under different climatic regimes. The differential responses were reflected in different vegetation characteristics examined and their changes. The coupled impacts of past livestock grazing disturbances and prolonged droughts might have contributed to the current slow restoration in the study region. Our results illustrated that while livestock exclusion was

undoubtedly a key driver of vegetation change in some areas, other factors such as climate and its interactions with other factors also played an important role.

Our results suggested that effective grassland management must be site-specific and must be developed based on local environmental conditions. For those areas with rapid responses to exclusion, such as the Humid Zone and Semi-Arid Zone, short-term exclusion combined with rotational grazing should be considered in order to achieve sustainable use of grasslands. The seasonal grazing method should also be adopted when the grassland shave recovered to a certain extent so that a balance between vegetation and livestock grazing intensity can be gradually established. For degraded grassland in the Arid Zone, more active artificial and semi-artificial grassland establishment is perhaps an efficient method.

Future studies should focus on longer-term in situ observations to better understand the ecological restoration processes. In order to develop better science-based grassland restoration strategies, the research on the effects of exclusion on grassland plant diversity, soil seed banks, and other ecological processes should be further strengthened. Future studies should also consider the coupled effects of grazing, climate, soil nutrients, insects (Li et al. 2012; Cease et al. 2012) and diseases, and other human activities. Research is also needed for considering the timing of grazing exclusion under different degrees of degradation and grassland adaptability to exclusion and climate change, including extreme weather and climate events (Li et al. 2012; Liu et al. 2013).

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