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Grazing exclusion—An effective approach for naturally restoring degraded grasslands in Northern China

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Abstract

Nearly 90% of the 390 million ha of grasslands in northern China are degraded. ‘Grazing exclusion’ has been implemented as a nature-based solution to rejuvenate degraded grasslands, but the effectiveness of the rejuvenation processes is uncertain. Here, we investigated the effects of grazing exclusion on aboveground plant community traits, soil physiochemical and biological properties, and the mechanisms responsible for enhanced grassland rejuvenation. A meta-analysis across various studies was used to assess the effectiveness. On average, grazing exclusion improved vegetation coverage by 18.5 percentage points and increased aboveground biomass by 1.13 t ha^{-1} and root biomass by 1.27 t ha^{-1}, which represent an increase of 84%, 246%, and 31%, respectively, compared with continuous grazing practices. Grazing exclusion reduced soil bulk density by 13.7% and increased soil water content by 68.9%. Grasslands under grazing exclusion increased soil organic carbon (SOC) in the 0- to 15-cm depth by 3.95 (±0.35 Std err) t ha^{-1} and total soil N, available N, and total soil P in the 0- to 40-cm depth by 2.39 (±0.14), 0.83 (±0.37), and 1.96 (±0.44) t ha^{-1}, respectively, compared with continuous grazing; these values represent an increase of 31%, 25%, 23%, and 14%, respectively. Prolonging the duration (years) of grazing practices enlarged the differences in SOC and soil N content between grazing exclusion and continuous grazing. Grazing exclusion has improved plant community traits and enhanced soil physiochemical and biological properties of degraded grasslands, and thus, this ‘nature-based’ approach can serve as an effective means to rejuvenate degraded grasslands.

KEYWORDS

grassland rejuvenation, ‘nature-based’ solution, plant diversity, SOC, soil biological property, soil physiochemical property

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1 | INTRODUCTION

Grasslands, one of the largest terrestrial ecosystems in the world, are crucial for wildlife habitat (Galli et al., 2012), livestock forage (Odriozola, García-Baquero, Laskurain, & Aldezabal, 2014), and the livelihoods of nearly 800 million people globally (FAOSTAT, 2015). In China, about 390 million ha of grasslands (Wang et al., 2014) cover 41% of the total land area (Anonymous, 2012). Of which, about 84% (or 330 million ha) is suitable for grazing, but a significant proportion has been degraded due to overgrazing (Wu, Zhao, Yu, Luo, & Pan, 2017), reclamation to croplands (Xu, Chen, Luo, & Lin, 2011), and exploitation of by-products or mineral resources (Dong & Yang, 2014). Degraded grasslands are characterized by diminished vegetation coverage and deteriorated soil structure and function and are vulnerable to erosion and desertification (Li, Zhao, Zhang, Zhang, & Shirato, 2004).

To restore degraded grassland ecosystems, China took drastic measures by implementing a series of grassland management policies from 1970s to the 2000s. 'Grazing exclusion' (i.e., total ban on grassland grazing) is one of the highly profiled rejuvenation measures (Conte & Tilt, 2014). This 'leave it alone' approach is considered a "nature-based" solution (Schaubroeck, 2017) as it helps alleviate the conflict between human needs and nature function (Fernandes & Guiomar, 2018). Grazing exclusion promotes living beings by promoting a positive soil–plant–microbiome interaction naturally to fulfill technical tasks that would have required high-energy, artificial means previously (Eggermont et al., 2015). This nature-based solution via natural processes (Nesshöver et al., 2017) requires little financial investment and can potentially bring simultaneous benefits to the nature, economy, and society (Albert, Spangenberg, & Schröter, 2017) and provide a long-term sustainable solution naturally (Schaubroeck, 2017) for grassland management.

With the recent efforts to address food security to eliminate hunger and improve eco-environmental sustainability, many questions have arisen: How effective are the decades of grazing exclusion measures for restoring degraded grasslands? What are the main mechanisms responsible for effective grassland restoration? Can the degraded grasslands be used for food production after years of restoration by grazing exclusion? This study offers a comprehensive assessment of the effectiveness of implementing the decades of grazing exclusion policies. Using a systematic approach, we summarize the major research findings on the effects of grazing intensity and the duration of grazing exclusion on plant community composition and diversity, vegetation characteristics, and soil physiochemical dynamics. We discuss the mechanisms involved in grazing exclusion on soil carbon and N dynamics, soil biological properties, and plant–soil–environment interactions. We offer suggestions on how grasslands in northern China could be managed more efficiently and sustainably using nature-based solutions.

2 | METHODS

We employed a four-step approach in this study. First, we conducted a systematic review of peer-reviewed literature using Agris, PubAg, and Scopus databases and identified studies that met the following criteria: (a) conducted under field conditions (those exclusively conducted in a controlled environment were excluded); (b) in northern or northwest China only (Figure 1a); (c) treatments included grazing exclusion and continuous grazing for three or more years with minimal two replications; (d) at least two or more variables were measured; and (e) papers published in English language journals with an impact factor (many Chinese-language papers on the subject in journals without an impact factor were excluded). Second, relevant data were extracted on a treatment-by-treatment basis from each identified study, entered in Excel spreadsheets, examined visually for accuracy and usefulness, and combined into a Master file. Some of the results presented in graphs in the original papers were converted into values using a graph-to-value conversion program. In some studies, results were aggregated and treatment means in some years or sites were used. Third, variables with sufficient data points to meet the basic criteria of meta-analysis were identified from 46 data-rich publications that cover the main grassland types in northern China. The Professional version (3rd version) of the Comprehensive Meta-Analysis program (Borenstein & Higgins, 2013) was used to analyze the key variables in regard to plant community characteristics and soil physiochemical properties (Table 1). In the analysis, treatment effects were assessed in the same or different subgroups, even if the 'effect size' differed or treatment subgroups were in different study years (Borenstein & Higgins, 2013); the Q-statistic was used to test the null hypothesis that all studies in the analysis share a common effect size; the i^2 statistic was used to quantify the proportion of observed variance that reflects differences in true effect sizes (i.e., heterogeneity); and Tau^2 was used to represent the variance of true effect sizes (Table 2). Further, significant treatment effects were determined using Duncan's multiple-range test at $P \leq 0.05$ using SAS Mixed model (Littell, Milliken, Stroup, & Wolfinger, 2006). Linear and nonlinear regression analyses were performed to determine the relationships between response variables (vegetation coverage, plant biomass, soil organic carbon [SOC], soil water content, etc.) and the number of years in grazing exclusion. Fourth, we summarized key results of 12 other published meta-analyses, including the study on the effects of grazing exclusion on carbon sequestration from 78 studies (Xiong, Shi, Zhang, & Zou, 2016), biomass from 48 studies (Yan, Zhou, & Zhang, 2013), soil microbial communities from 71 studies (Zhao et al., 2017), carbon (C) and nitrogen (N) cycling from 115 studies (Zhou et al., 2017), and grassland management and greenhouse gas (GHG) emissions from 67 studies (Nayak et al., 2015). Additionally, we cited key results from numerous articles that were not included in the meta-analysis.

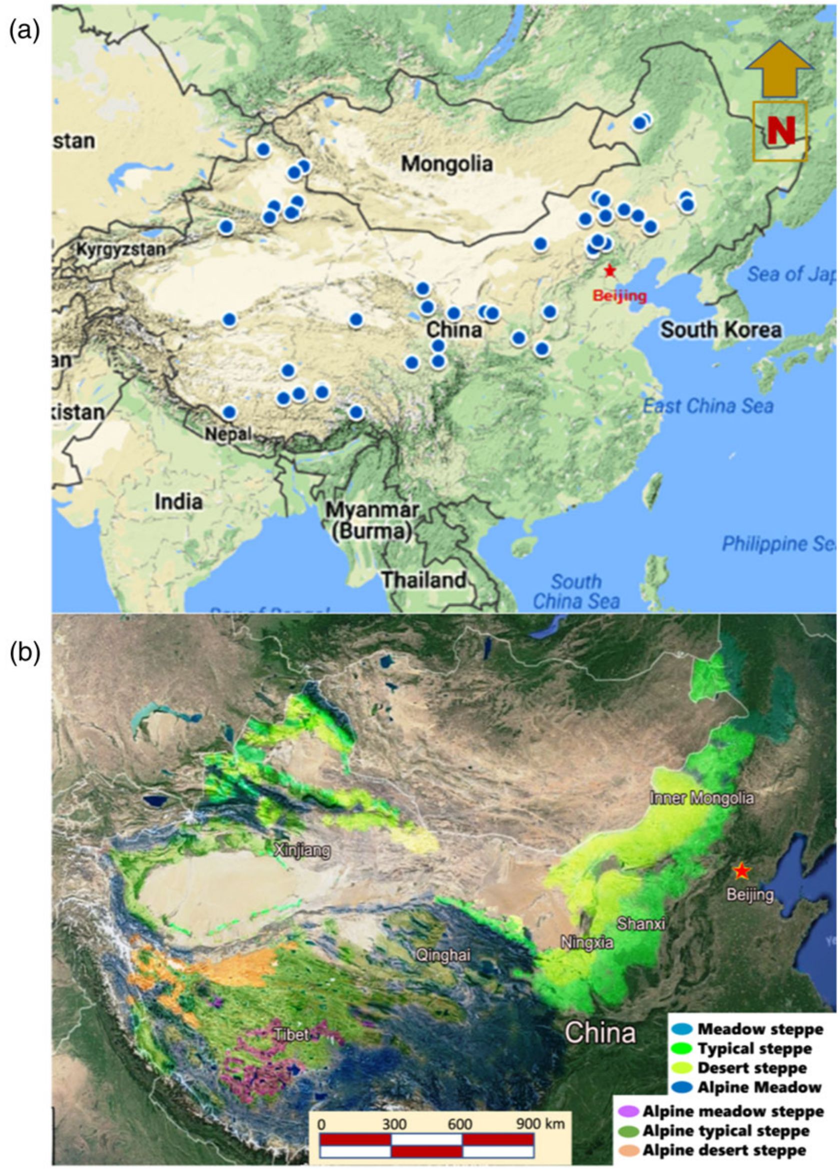

FIGURE 1 A meta-analysis to synthesize the key results of field studies conducted in (a) the major grassland zones in northern China (the blue dots denote the sites of the studies), where (b) about 390 million ha of grasslands mainly consist of meadow steppe, alpine steppe, typical steppe, and desert steppe [Colour figure can be viewed at wileyonlinelibrary.com]

3 | BACKGROUND OF NORTHERN CHINA GRASSLANDS

About 40% of Chinese natural grasslands are concentrated in the temperate arid to semiarid northern part of the country. Annual precipitation ranges from 500 mm in the northeast to as little as 50 mm in the northwest, with 70% occurring from July to September (Chai et al., 2014), whereas annual evaporation ranges from 1,500 to 2,600 mm (Deng, Shan, Zhang, & Turner, 2006). Shortage of water is the dominant biophysical factor affecting the degree of grassland degradation and the success of restoration in northern China. Using Inner Mongolia, where the typical grassland ecosystems are located in northern China, as an example, the long-term annual mean temperature is 3.4°C, with highest monthly mean temperature in July (22.6°C) and lowest in January (−18.3°C; Anonymous, 2018), with windy and dry springs (March–May) and comparatively rainy and warm summers (June–August). Soils in the northern region range from light-colored Chernozems in the northwest to dark-brown- and black-colored Phaeozems and Kastanozems in the northeast (Shi et al., 2006). Elevation in the region ranges from 900 to 1,850 m above sea level, and the topography is high plain and diluvial land near mountains with rolling slopes of about 1 to 5° in cultivated land (R. Zhang, Wang, et al., 2018).

Northern China grasslands are one of the largest agro-pastoral ecotones in the world. The grasslands mainly include meadow steppe, alpine steppe, typical steppe, and desert steppe (Figure 1b). Alpine typical steppe is dominated by *Stipa purpurea* or *Stipa capillata*, alpine desert steppe is dominated by *Stipa breviflora* or *Stipa orientalis*, and subalpine meadow is normally located above the forest zone in high altitude areas.

Historically, herd farmers adopted a nomadic mode of grazing (Figure 2a), a practice that exerted little pressure on grasslands. From the 1950s to 1980s, large areas of grasslands were converted to cropland (Xu et al., 2011), which decreased the availability of grasslands for nomadic grazing (Su et al., 2018). Heavy grazing with high stocking rates severely degraded the remaining grasslands (Figure 2b) and accelerated the degradation (Figure 2c). To curb further degradation and rejuvenate degraded grasslands, China implemented a series of policies. ‘Grazing exclusion’ was established in 1979 (Chen & Tang, 2016) with the goal of total elimination of grazing. A follow-up programme ‘Grain-for-Green’ was established in the late 1990s to encourage food production without affecting grassland ecosystems.

TABLE 1 Data source, basic characteristics, and main variables in the published studies used in our meta-analysis

Reference	Coordinates	Study years	Duration of grazing treat. (year)	Plant properties				Soil properties								
								Physical			Chemical					
				Biomass	Diversity	Veget. cover	Root biomass	Bulk density	Texture	Moisture	SOC	N	Other nutrients	pH	C/N	Biol. activity
Chen et al. (2017a)	37°8′N, 106°49′E	2001–2014	13								X	X	X			X
Chen et al. (2017b)	37°8′N, 106°49′E	2001–2014	13													X
Dong et al. (2012)	33°37′ N, 99°48′ E	1998–2000	2					X			X	X	X		X	
Duan et al. (2012)	30°27′–35°39′N, 83°41′–95°10′E	2006–2009	3	X	X	X										
Duan et al. (2012)	30°27′–35°39′N, 83°41′–95°11′E	2006–2010	N/A	X		X										
Gan, Peng, Peth, and Horn (2012b)	43°38′N, 116°42′E	2007–2009	N/A					X	X		X					
Gong et al. (2014)	42°53′N, 83°42′E	N/A	27	X		X	X	X						X		
Gou, Nan, and Hou (2015)	37°7′N, 106°49′E	2001–2012	11													X
Han et al. (2008)	43°34′N, 119°35′–119°38′E	2003–2005	N/A	X		X					X	X				
Jiao et al. (2016)	37°6′N, 103°31′E	2003–2010	7					X		X		X	X	X		X
Kölbl et al. (2011)	43°33′N, 116°40′E	2004–2006	25					X	X		X	X				
Li, Hao, Zhao, Han, and Willms (2008)	41°46′–41°50′N, 111°2′–111°55′E	2005	8					X			X	X	X	X		
Liu, Nan, and Hou (2011a)	37°8′N, 106°50′E	2001–2009	8							X		X				X
Liu, Nan, and Hou (2011b)	37°8′N, 106°50′E	2001–2010	9	X							X	X		X	X	
Qu et al. (2016)	44°33′N, 123°40′E	2009–2012	3							X	X	X	X	X	X	X
Ren et al. (2012)	43°38′N, 116°42′E	2005–2010	9	X	X											
Rong, Johnson, Wang, and Zhu (2017)	41°44′N, 115°46′E	2010–2014	4	X			X	X			X	X		X		
Rui et al. (2011)	37°37′N, 101°12′E	2006–2009	4							X	X	X				X
Schönbach et al. (2012)	43°38′N, 116°42′E	2005–2008	3	X							X					
Steffens, Kölbl, and Kögel-Knabner (2009)	43°38′N, 116°42′E	1979–2004	25						X		X	X		X		
Sun et al. (2014)	30°57′N, 88°42′E	2012	2					X			X	X	X			
Su et al. (2018)	41°54′N, 116°0′E	2009–2010	1			X		X		X		X	X	X	X	
Wang (2002)	44°45′N, 123°45′E	N/A	5					X		X				X		
Wang, Yan, and Cao (2012)	29°56′–36°41′N, 83°52′–95°1′E	1980–2007	27					X	X		X	X	X	X	X	

(Continues)

TABLE 1 (Continued)

Reference	Coordinates	Study years	Duration of grazing treat. (year)	Plant properties				Soil properties								
								Physical			Chemical					
				Biomass	Diversity	Veget. cover	Root biomass	Bulk density	Texture	Moisture	SOC	N	Other nutrients	pH	C/N	Biol. activity
S. K. Wang, Zuo, et al. (2016)	41°44′N, 115°46′E	2009–2012	3	X	X			X	X	X	X	X		X		
D. Wang, Du, Zhang, Ba, and Hodgkinson (2017)	44°35′N, 123°36′E	2002–2003	17	X						X						
T. Wang, Zhang, et al. (2017)	38°54′–39°11′N, 100°48′–101°12′E	2016	18	X	X	X	X	X			X	X		X	X	
Wiesmeier et al. (2012)	43°38′N, 116°42′E	2005–2008	3					X			X	X	X			
Wu, Li, Cheng, Wei, and Sun (2009)	33°42′N, 102°7′E	1999–2009	10	X	X	X										
Wu et al. (2012)	43°38′N, 116°42′E	N/A	25													X
Xu, Wan, Cheng, and Li (2008)	43°50′N, 116°34′E	1989–2006	15	X				X			X	X				
Xu et al. (2014)	41°46′N, 115°41′E	2001–2012	11	X	X	X	X	X		X	X	X			X	
Xue et al. (2016)	31°38′N, 92°0′E	2013	3							X		X	X	X		X
Yan, Tang, et al. (2016)	49°19′–49°20′N, 119°56′–119°57′E	2009–2014	5	X		X	X									
Yan, Yang, et al. (2016)	49°23′–49°25′N, 120°5′–120°11′E	2009–2014	5					X		X						
L. L. Yang, Gong, et al. (2016)	44°48′–44°50′N, 116°2′–116°30′E	2011–2012	N/A	X		X						X				
Z. Yang, Xiong, et al. (2016)	34°54′N, 102°6′E	2010–2015	5	X	X	X	X	X				X	X	X		
Zhang and Dong (2009)	34°36′–35°53′N, 111°15′–112°37′E	N/A	N/A									X	X	X		
Zhang, Zhao, Li, and Zhou (2004)	42°55′N, 120°42′E	1992–1996	5	X	X	X	X									
Zhang et al. (2015)	31°23′N, 90°2′E	2010–2013	3	X		X										
Zhang et al. (2017)	42°55′N, 120°42′E	2014	25	X	X	X	X	X		X		X		X		
Zhao et al. (2007)	43°50′N, 87°37′E	2004–2005	5	X	X											
Zhou et al. (2012)	42°52′43°57′N, 83°42′–89°45′E	2003	N/A		X											

Note. SOC: soil organic carbon.

TABLE 2 Heterogeneity, i^2, and Tau^2 values for the main variables in the meta-analysis

		Heterogeneity				Tau^2			
Variable	Number of standardized comparison	Q value	df (Q)	P value	i^2	Tau^2	Standard error	Variance	Tau
Plant property									
Vegetative cover	30	929	29	0.000	96.9	498	165	27,158	22.32
Plant biomass	41	1,456	40	0.000	97.3	257,256	122,702	15,055,819,606	507
Plant diversity	25	1,178	24	0.000	98.0	0.261	0.146	0.021	0.51
Root biomass	20	211	19	0.000	91.0	785,767	399,819	159,855,002,842	886
Soil property									
Soil bulk density	37	3,420	36	0.000	98.9	0.010	0.004	0.000	0.10
Soil water content	33	1,107	32	0.000	97.1	0.660	0.354	0.126	0.81
SOC	67	1,188	60	0.000	95.0	6.50	2.19	4.80	2.55
Soil total N	72	91,393	60	0.000	99.9	1.20	0.63	0.40	1.10
Soil available N	39	670	38	0.000	94.3	4.22	2.06	4.25	2.06
Soil C:N ratio	21	1,173	20	0.000	98.3	1.007	0.526	0.276	1.00
Soil avail P	29	384	28	0.000	92.7	4.73	2.05	4.20	2.18
Soil pH	32	1,071	31	0.000	97.1	0.023	0.011	0.000	0.15

Standardized, paired comparison between grazing practices in meta-analysis (Borenstein & Higgins, 2013).

In the early 2000s, a program 'Returning Croplands to Grasses' was established with the aim of banning grazing on severely degraded grassland, alternating seasonal-grazing with fallow in moderately degraded grassland, and rotating grazing on slightly degraded grassland. Fencing practices are commonly used to implement 'grazing exclusion' to promote a natural recovery of the degraded grasslands (Figure 3).

4 | GRAZING EXCLUSION AND PLANT COMMUNITIES

4.1 | Plant productivity

Natural grasslands in northern China are historically rich in plant species with a productive community structure, but overgrazing has altered the vegetation characteristics and severely degraded the grasslands in many areas. Our meta-analysis of data-rich studies revealed that grazing intensity is one of the most influential factors that significantly ($P < 0.05$) affects vegetation coverage, aboveground biomass, and root biomass of grasslands (Table 3). Compared with other grazing practices (normal to heavy grazing), grazing exclusion improved vegetation coverage by an average 18.5 percentage points (range 17.0–19.8 from 228 paired comparisons, Figure S1) and increased aboveground biomass by an average 1.14 t ha^{-1} (range 0.97–1.31 t ha^{-1} from 313 paired comparisons, Figure S2) and root biomass by 1.80 t ha^{-1} (range 1.36–2.23 t ha^{-1} from 213 paired comparisons, Figure S3). Expressed as percentage change, grazing exclusion increased vegetation coverage, aboveground biomass, and root biomass by an average 84%, 246%, and 31%, respectively. Grazing exclusion led a rapid recovery of the plant community structure of degraded grasslands to a benign state (Hao et al., 2014).

4.2 | Plant diversity and richness

Diversity is regarded as an important indicator of plant community functioning. Enhancing plant diversity typically increases plant productivity (X. Zhang, Liu, et al., 2018), improves the community's stability to tolerate abiotic extremes (Bloor & Bardgett, 2012), and promotes plant–soil interactions for the provision of positive feedback to the plant community (Loranger-Merciris, Barthes, Gastine, & Leadley, 2006). Our meta-analysis of 268 paired comparisons revealed that plant diversity values spread widely across different experiments (Figure S4). Grazing exclusion enhanced overall plant diversity by 11.4% across studies with an i^2 value of 98.0 and Tau^2 value of 0.261, which were significant at $P = 0.011$. In a 12-year field experiment on the desert steppe of Inner Mongolia, X. Zhang, Liu, et al. (2018) found that increased grazing intensity (from nongrazing to heavy grazing) significantly decreased species richness ($R^2 = 0.94$), Margalef index ($R^2 = 0.95$), Shannon–Wiener index ($R^2 = 0.99$), and Pielou's index ($R^2 = 0.94$). However, a published meta-analysis with data from 78 articles showed that grazing exclusion provided little or no benefit to plant diversity (Xiong et al., 2016).

Plant species richness is one of the key properties of grasslands. Grazing exclusion tends to increase the richness of a plant community (Zhu, Deng, Zhang, & Shangguan, 2016) through improving the seed bank of annual and perennial species (Zuo et al., 2012) or increasing seed germination rates (Zhao, Su, Wu, & Gillet, 2011). In some cases, grazing exclusion promotes rapid accumulation of some palatable genera, such as *Stipa* (Zhao, Li, & Qi, 2007), bunchgrass (e.g., *Festuca idahoensis*), and rhizomatous grasses (Huang, Wang, & Wu, 2007). Grazing practices also affect the size and composition of bud banks, which affect population regeneration and community dynamics (Qian, Wang, Liu, & Busso, 2017). However, the richness response varies with other factors (Qian et al., 2017) such as seed morphology and germination characteristics (Chen et al., 2015) and the resistance to biotic stresses (Chen, Christensen, Nan, & Hou, 2017b). Also, the positive effect on community richness may diminish with the length of grazing exclusion.

4.3 | Heterogeneity

Northern China grasslands include various steppes (desert, typical, alpine, and meadow) composed of different plant communities (Kang,

FIGURE 2 Traditional nomadic feeding practices adopted in northern China for thousands of years have a rich history of using (a) 'temporal rotation' grazing systems as herders move from one place to the next according to the growing season. Since the 1950s, most grasslands have been (b) heavily grazed with a stocking rate higher than the grasslands can hold, which has caused (c) severe degradation with low productivity and poor quality [Colour figure can be viewed at wileyonlinelibrary.com]

Han, Zhang, & Sun, 2007), which are located in different regions with diverse topographical and climatic conditions (Figure 1). These spatiotemporal variabilities lead to an inconsistent response of plant diversity to grazing practices. Plant diversity is usually more sensitive in alpine steppes in the humid and semihumid northeastern than in the arid and semiarid steppes of the northwest (Ren, Lü, & Fu, 2016). Grazing has a highly selective pressure toward certain species, and grazing exclusion promotes the rapid recovery of dominant plant species with other species providing niche complementarity to the dominant species (Figure 4). Diversity heterogeneity can be related to the distribution of grazing animals (Ruifrok, Postma, Olff, & Smit, 2014), the different groups of plant species (Qu et al., 2016), and the critical threshold of N-induced vegetation recovery capacity (Tang, Deng, An, Yan, & Shangguan, 2017). In the scientific literature, it is not always clear whether grazing exclusion promotes plant species enrichment or perhaps the loss of plant diversity and their consequences for grassland ecosystem functioning.

5 | GRAZING EXCLUSION AND SOIL PROPERTIES

5.1 | Soil physical properties

Our meta-analysis revealed significant effects of grazing practices on soil physiochemical properties (Table 2), and the magnitude of the effects varied between variables (Table 3). On average, grazing exclusion led to a 13.7% reduction in bulk density (or 0.047–0.054 units from 569 paired comparisons; Figure S5) and 68.9% increase in soil water content (from 285 paired comparisons; Figure S6). The mechanisms responsible for reducing soil bulk density and increasing soil water content with grazing exclusion can be complex, but it is largely attributable to (a) decreased precompression stress and increased saturated hydraulic conductivity and anisotropy (vertical vs. horizontal conductivity; Reszkowska et al., 2011); (b) reduced soil evaporation (Krümmelbein, Peth, Zhao, & Horn, 2009); (c) decreased tensile strength of aggregates due to improved dynamics of soil macroporosity and vertical continuity of macropores (Kölbl et al., 2011); (d) a pronounced recovery of soil strength in steppes with high precipitation (Reszkowska et al., 2011); and (e) increased soil organic matter inputs (detailed in Section 5.2) that enhances soil carbon-induced wetness (Kölbl et al., 2011).

The magnitude of the effect of grazing exclusion on soil physical properties varies with various factors. In the arid and semidesert grasslands of Inner Mongolia, a 6-year grazing exclusion lowered soil bulk density through increased aboveground and belowground biomasses (Wu et al., 2014). In comparison, an 8-year grazing exclusion in alpine grasslands did not improve soil bulk density or particle size distribution (Lu et al., 2015). Heavy grazing typically reduces the fine mineral fraction (silt and clay) and increases the sand fraction, resulting in coarser textured soil (Huang et al., 2007). Heavy grazing significantly increases the tensile strength of aggregates (Reszkowska et al., 2011), which decreases the shear resistance of soil to scouring (Gan, Peng, Peth, & Horn, 2013) and increases soil erodibility (Zhou, Gan, Shangguan, & Dong, 2010). Additionally, reduced soil aggregation with heavy grazing has a negative effect on soil crusting (Zhang et al., 2013).

5.2 | Soil carbon

The grassland carbon cycle, a significant global carbon cycle component (G. Hu, Liu, Yin, & Song, 2016), typically includes three carbon pools: plant carbon, litter-fall carbon, and soil carbon. The size of these carbon pools is closely related to natural variables and human activities. Grazing management is a main driver for the change in SOC that contributes to the sustainability of grasslands. Our meta-analysis of 1,321 paired comparisons from 43 studies showed that grazing exclusion practices increased SOC in the 0- to 15-cm soil layer significantly compared with other grazing practices with a standardized mean

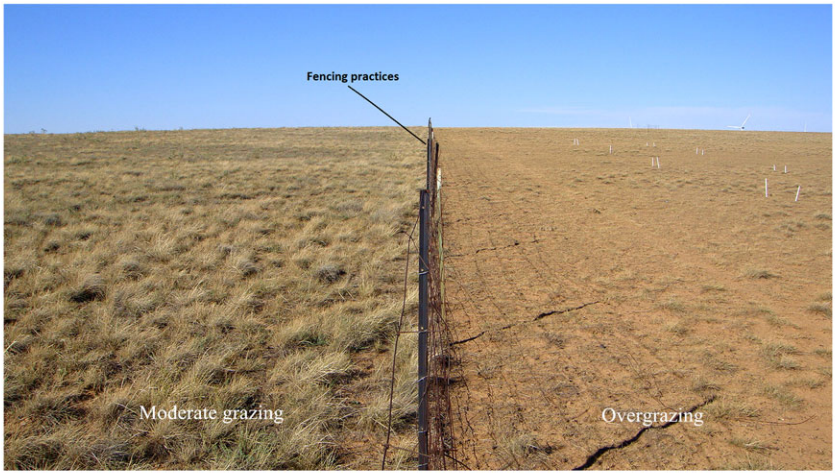

FIGURE 3 Fencing practices have been commonly used to implement "grazing exclusion" policies in northern China [Colour figure can be viewed at wileyonlinelibrary.com]

TABLE 3 Mean differences between grazing exclusion and other grazing practices in various plant and soil traits identified in the meta-analysis

Effect	Variable	Number of standardized comparison[a]	Effect size and 95% confidence interval					Test of null (2-tail)	
			Mean difference	Std err	Variance	Lower limit	Upper limit	*Z* value	*P* value
Plant property									
Vegetation coverage, %									
Fixed		30	18.5	0.7	0.5	17.0	19.9	25.5	0.000
Random		30	25.2	4.2	17.7	17.0	33.5	6.0	0.000
Aboveground biomass, kg ha^{-1}									
Fixed		41	309	13	162	284	334	24	0.000
Random		41	1,142	86	7,454	973	1,311	13	0.000
Root biomass (kg ha^{-1})									
Fixed		20	1,718	60	3,618	1,600	1,836	29	0.000
Random		20	1,795	224	50,085	1,356	2,233	8	0.000
Plant diversity									
Fixed		25	0.03	0.01	0.00	0.01	0.06	2.53	0.011
Random		25	0.24	0.11	0.01	0.01	0.46	2.07	0.038
Soil property									
Soil bulk density (g cm^{-3})									
Fixed		37	−0.05	0.00	0.00	−0.05	−0.05	−30.12	0.000
Random		37	−0.10	0.02	0.00	−0.13	−0.06	−5.50	0.000
Soil water content (%)									
Fixed		33	0.19	0.02	0.00	0.15	0.24	8.36	0.000
Random		33	1.58	0.17	0.03	1.24	1.91	9.17	0.000
SOC (t ha^{-1})									
Fixed		61	3.64	0.07	0.01	3.50	3.79	49.49	0.000
Random		61	3.95	0.35	0.12	3.27	4.63	11.37	0.000
Total soil N (t ha^{-1})									
Fixed		61	0.35	0.00	0.00	0.35	0.36	104.28	0.000
Random		61	2.39	0.14	0.02	2.11	2.67	16.90	0.000
Available soil N (t ha^{-1})									
Fixed		39	0.90	0.08	0.01	0.75	1.05	11.73	0.000
Random		39	0.83	0.37	0.13	0.11	1.54	2.26	0.024
Soil C:N ratio									
Fixed		21	0.82	0.03	0.00	0.77	0.87	29.69	0.000
Random		21	0.29	0.24	0.06	−0.18	0.76	1.20	0.231

(Continues)

TABLE 3 (Continued)

Effect	Variable	Number of standardized comparison[a]	Effect size and 95% confidence interval					Test of null (2-tail)	
			Mean difference	Std err	Variance	Lower limit	Upper limit	Z value	P value
Soil pH									
Fixed		32	0.02	0.00	0.00	0.01	0.03	4.91	0.000
Random		32	−0.10	0.03	0.00	−0.16	−0.04	−3.29	0.001
Total soil P (t ha^{-1})									
Fixed		29	3.07	0.11	0.01	2.86	3.29	27.84	0.000
Random		29	1.96	0.44	0.20	1.09	2.83	4.40	0.000

Note. SOC: soil organic carbon.

[a]Standardized, paired comparison between grazing practices in meta-analysis (Borenstein & Higgins, 2013).

FIGURE 4 With grazing exclusion, the dominant plant species may increase growth and productivity whereas other species provide some niche complementarity to the community (photo taken in a moderately grazed grassland in Inner Mongolia where shrubs and scattered trees provide some niche complementarity to the grass-dominated plant community) [Colour figure can be viewed at wileyonlinelibrary.com]

difference of 3.95 t ha^{-1} (range 3.27–4.63 t ha^{-1}, or 31.4%; Figure S7). The duration (year) of grazing exclusion had a marginal impact on the absolute value of SOC (Figure 5a), but the percent difference was significant between continuous grazing and grazing exclusion; the longer the exclusion duration, the greater the difference in SOC between the two grazing practices (Figure 5b). A meta-analysis by other researchers revealed that grazing exclusion increased the amount of carbon by 112% as litter fall, belowground biomass by 26%, and soil carbon by 14% compared with other grazing practices (Xiong et al., 2016). Similarly, a study in the Xilin River basin of northern China with *Leymus chinensis* L. and *Carex tristachya* showed that as the exclusion duration increased from 0 to 11 years, SOC stocks increased by 14.3% (or 0.26 g kg^{-1} of soil; Chen & Tang, 2016). However, a further increase of grazing exclusion duration from 11 to 31 years did not increase SOC stocks.

The mechanisms involved in SOC changes in response to grazing exclusion during grassland restoration are not clear. There are some indications in the literature that the SOC changes are most likely attributable to the following:

1. Grazing exclusion boosts net primary production of the grassland due to reduced grazing pressure (Huang, Brümmer, & Huntsinger, 2016; Wang, Johnson, Rong, & Wang, 2016), leading to increased plant biomass accumulation on the soil surface (Li et al., 2013). The reduction in grazing pressure also leads to more belowground root biomass than aboveground biomass, and the increased root: Shoot ratio accelerates SOC accumulation (Liu, Liu, Wu, Wang, & Chen, 2014).
2. With grazing exclusion, the cessation of animal trampling enhances soil aggregation that fosters physical protection of SOC, and the increased input of organic matter acts as binding agents for aggregation. Long-term grazing exclusion can result in a higher proportion of SOC occluded in soil aggregates (Wu, Zhang, Qian, & Huang, 2013).
3. Grazing exclusion stimulates the development of a heterogeneous structure of carbon-rich spots in highly productive patches (Kölbl et al., 2011; Wiesmeier et al., 2009). These patches can be regarded as "islands of fertility" (Hibbard, Archer, Schimel, & Valentine, 2001) where rainwater and organic materials are redistributed from bare soil areas (runoff zones with low infiltration) to patches (run-on zone with higher infiltration). This process favors plant biomass accumulation and thus more input of organic matter into the soil.

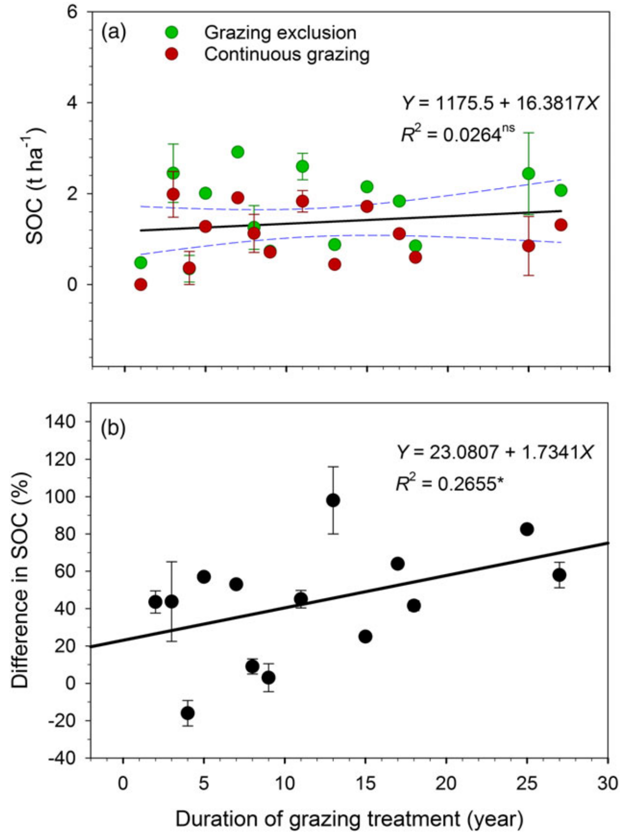

FIGURE 5 Relationship of duration (year) of grazing exclusion with (a) the amount of SOC and (b) percent differences in SOC between grazing exclusion and continuous grazing*. And *ns* represent the regression slope (*b* value) equaling to zero at $P < 0.05$ and $P \geq 0.05$, respectively. SOC: soil organic carbon [Colour figure can be viewed at wileyonlinelibrary.com]

4. The change in grassland SOC is a result of plant–soil–microbiome interactions. Grazing exclusion enhances the carbon accumulation in the soil but at the meantime, it increases soil microbial respiration that depletes soil carbon (Li et al., 2013). Ultimately, the magnitude of the effect of grazing exclusion on the quantity of SOC is a function of the amount of organic matter input and the turnover process.
5. The magnitude of the SOC change with grazing exclusion is highly related to abiotic (soil temperature, soil water content, and soil nutrient content), biotic (plant community structure, litter input, and microbial activity), and climatic and geographical factors (T. Chen, Nan, et al., 2018; Wittmer, Auerswald, Bai, Schäufele, & Schnyder, 2010). For example, in arid and semiarid grasslands of northern China, grazing exclusion generally increased SOC concentration (Gao & Cheng, 2013; S. K. Wang, Zuo, et al., 2016). In the more humid regions of the northeast, grazing exclusion had little effect on SOC (Su et al., 2015).

5.3 | Soil chemical properties

Besides the effects on SOC status, grazing exclusion has a large effect on the dynamics of other chemical elements in grassland soils. With the values averaged across the 0- to 40-cm soil depth, our meta-analysis showed that grazing exclusion increased total soil N by 2.39 t ha^{-1} (range 2.11–2.67 t ha^{-1}) from 992 standardized paired comparisons (Figure S8), soil available N by 0.83 t ha^{-1} (range 0.11–1.54 t ha^{-1}) from 547 standardized paired comparisons (Figure S9), and total soil P by 1.96 t ha^{-1} (range 1.09–2.83 t ha^{-1}) from 419 standardized paired comparisons (Figure S10), compared with continuous grazing practices; these values represent an increase of 25%, 23%, and 14%, respectively. The wide range of the difference in soil N and P was partly relative to the duration (year) of grazing practices (data not shown). The differences in soil N and P status between grazing exclusion and continuous grazing increased with the duration of exclusion; the longer the exclusion duration, the greater the differences in soil N and P between the two grazing systems.

Overall, the enhancement of soil carbon sequestration with grazing exclusion is greater than the increase in soil N, leading to an increase in the soil C:N ratio of 0.82 units across studies (Figure S11). The increase in soil C:N ratio plays a key role in the restoration of degraded grasslands (Chen, Christensen, Nan, & Hou, 2017a). Also, there is a close relationship between grazing practices and some micronutrients such as Cu, Mn, and Zn (Jiao, Nie, Zhao, & Cao, 2016). Grazing exclusion may stimulate a heterogeneous distribution of certain soil particles in a soil niche, leading to an increased capacity to restore nutrient stocks in degraded soils (Ma, Ding, & Li, 2016).

In summary, our meta-analysis across studies shows that grazing exclusion has improved aboveground plant properties, including an average increase of grassland vegetation coverage by 83.6% (Figure 6a), plant biomass by 246.0% (Figure 6b), root biomass by 31.2% (Figure 6c), and plant diversity by 11.4% (Figure 6d). Also, grazing exclusion practices have increased SOC by an average of 31.4% (Figure 6e) and total soil N by 25.4% (Figure 6f) and decreased soil bulk density by 13.7% (Figure 6g). Grazing practices did not affect overall soil C:N ratio in the grassland soil (Figure 6 h). However, there is a tendency of increasing available soil N (by 23.0%), total soil P (14.2%), and soil water content (by 68.9%) in grasslands with grazing exclusion compared with continuous grazing (data not shown). These percent changes are averages across various studies evaluated in our meta-analysis, and it was impossible to determine an annualized percentage value accurately. Also, the magnitude of the effects varied with other factors such as climate (S. Chen, Nan, et al., 2018; Wittmer et al., 2010), geographic location (Xu, Xie, & Wang, 2014), and steppe type (Liu, Zhao, Zhao, & Zhu, 2013), among others.

5.4 | Soil biological properties

Soil microbes play a critical role in maintaining soil functionality as they control SOC dynamics (Hooker & Stark, 2012; Yu, Li, Jin, Liu, & Wang, 2017) and nutrient supply (Jia et al., 2016) and provide feedback to plant growth (Borrell et al., 2017; Niu, Bainard, Bandara, Hamel, & Gan, 2017). Thus, soil microbes are important components of grassland ecosystems (Rasche & Cadisch, 2013). Other published meta-analyses revealed that light grazing practices had no effect on

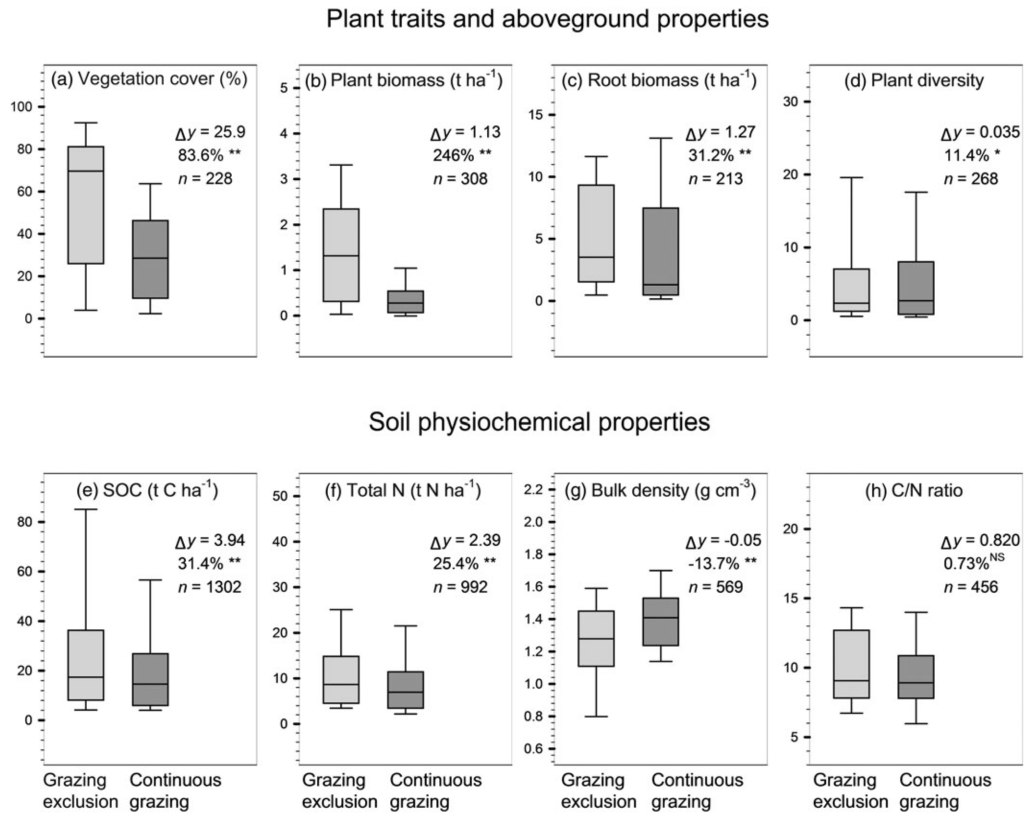

FIGURE 6 Summary on the effects of grazing practices (grazing exclusion vs. continuous grazing) on main plant traits and soil physiochemical properties. Each boxplot shows the median, 25 and 75 quartiles, and range of values. Δy and % values represent the differences between the two grazing systems in absolute value and percent differences, respectively; *n* is number of replicates from the studies included in our meta-analysis; **, *, and *ns* denote significance in mean between the two grazing systems at $P < 0.01$, $P < 0.05$, and $P \geq 0.05$, respectively. The quantity of SOC was estimated for the 0- to 15-cm soil depth, the amounts of total soil N, available soil N, and total soil P were for the 0- to 40-cm soil depths, and the C:N ratio (C/N) was the values reported in the articles included in our meta-analysis. SOC: soil organic carbon

soil microbial community, but heavy grazing significantly reduced the size of microbial community (Zhao et al., 2017). Grazing has a significant effect on the composition of soil microbial biodiversity (Chávez, Escobar, Anghinoni, de Faccio Carvalho, & Meurer, 2011; Eldridge et al., 2017), which is particularly significant in arid northwestern China. For example, in the dry steppe of the Loess Plateau, grazing exclusion for 20 years significantly increased microbial biomass in the 0- to 10-cm layer compared with continuous grazing (Cheng et al., 2011). Grazing exclusion improved the abundance of soil bacteria in the dominant taxonomic groups including *Actinobacteria*, *Proteobacteria*, *Acidobacteria*, *Firmicutes*, *Planctomycetes*, *Chloroflexi*, *Gemmatimonadetes*, and *Bacteroidetes* (Cheng, Jing, Wei, & Jing, 2016), as well as soil macroinvertebrate abundance (Liu et al., 2013). Those effects on plant and microbial community traits often interact with the status of SOC and soil nutrients (Cheng et al., 2016).

Due to the complex nature of the characteristics of soil microbiomes in response to grazing practices, a quantitative assessment of the effect is difficult. We briefly highlight some of the possible mechanisms responsible for enhanced soil biological properties with grazing practices:

1. Different grazing intensities alter the diversity of soil fungi and bacteria under the principle of 'competitive exclusion', meaning that some microbial species are more efficient at exploiting available resources than others when competing for the same resources. Grazing exclusion allows the dominant species to have a competitive advantage for greater fitness than the subordinate species. Thus, the relative abundance of specific taxa in the soil is resource-selective, and the availability of specific resources drives the alteration of relative abundance and biodiversity (Bainard, Hamel, & Gan, 2016).
2. Microbial competitors in grassland soil are vast, and their pool size varies with steppe type and human activities. Substantial changes in the soil microbial community can occur as a result of soil disturbance (Bainard, Bainard, Hamel, & Gan, 2014). Grazing exclusion enables the soil to maintain a silent status with little disturbance, allowing a natural recovery of the beneficial microbial community. Of course, the status of soil microbial composition and community function can vary with abiotic conditions such as soil pH (T. Wang, Zhang, Li, & Li, 2017; Zhang et al., 2017), soil water content and temperature (Dorji, Moe, Klein, & Totland, 2014; Gan, Peng, Peth, & Horn, 2012a), or fertility status (Yan, Yang, et al., 2016).
3. Specific macrofaunal groups may favor different living conditions in grassland soils to adapt to their specific habitats (Liu et al., 2013). Grazing exclusion with no soil disturbance allows a favorable soil environment where subordinate microbial taxa are

released from competitive exclusion by altering the relative abundance of dominant microbial taxa; this process ultimately helps to increase the biodiversity.

These possible mechanisms are largely based on how soil microbial communities respond to soil environment and grazing practices. A need to understand the mechanisms of soil biological properties in association with grassland management practices is pressing in northwestern China where climatic conditions are highly variable, and drought is highly frequent, which may affect grassland soil biological properties.

6 | INTERACTIONS OF GRASS–CLIMATE–SOIL–HUMAN ACTIVITY

Grasslands in northern China are spread across diverse landscapes in the arid, semiarid, and humid climatic zones with varying weather conditions. Thus, the outcome of grassland restoration efforts from severe degradation is a consequence of the complex interaction between meteorological, topographic, and soil environments with plant community structure and management practices (Figure 7). Climate has a significant impact on many aspects of aboveground properties, such as species composition (Luo et al., 2013), herb abundance (Li et al., 2017), shrub encroachment (J. Chen, Li, et al., 2015), and forb patch densities (L. Chen, Li, et al., 2015). In arid and semiarid areas, C_3 species such as *Stipa grandis* are highly competitive when March–June temperatures are low and precipitation is high, whereas C_4 species such as *Cleistogenes squarrosa* often benefit from higher March–June temperatures and lower precipitation (Ren, Schönbach, Wan, Gierus, & Taube, 2012). Degraded grasslands may recover faster with grazing exclusion in the more humid northeast areas than in the arid and semiarid northwest areas, as precipitation amplifies the restoration process (Hao et al., 2014). However, the magnitude of climatic impact on grassland restoration interacts with management practices (Gao et al., 2013; X. Yang, Liu, et al., 2016). For example, the percent change in aboveground biomass had a quadratic relationship with precipitation under light grazing, but a linear relationship under heavy grazing (Yan et al., 2013).

Grassland restoration is a vital process that includes the restructuring of belowground properties (J. Wang, Li, & Bian, 2016) as root traits are often altered with grazing practices (Chen et al., 2017a; Zhang et al., 2017). The root:shoot ratio, a key variable for assessing the effectiveness of grassland restoration, often differs with grazing practices (J. Wang, Li, & Bian, 2016), grassland type (Wang, Niu, Yang, & Zhou, 2010), and the characteristics of biomass partitioning between aboveground parts and roots at the community level (Yang, Fang, Ma, Guo, & Mohammat, 2010).

The degree of restoration of degraded grasslands in northern China largely depends on various factors, including different grazing intensities (nongrazing, mild to moderate, and heavy grazing), grazing systems (seasonal vs. continuous grazing), and duration of grazing exclusion (3 to 31 years). Also, anthropogenic activities can have some significant impacts on human-induced net primary productivity (Ren & Zhou, 2018). Inconsistent outcomes of the effect of grazing exclusion on grassland restoration may occur across Chinese grassland zones. Below, we summarize the mechanisms responsible for the various outcomes:effects of grazing practices:

1. *Complex nature of plant community*. Chinese grasslands have a wide range of ecotypes ranging from semidesert, arid, and

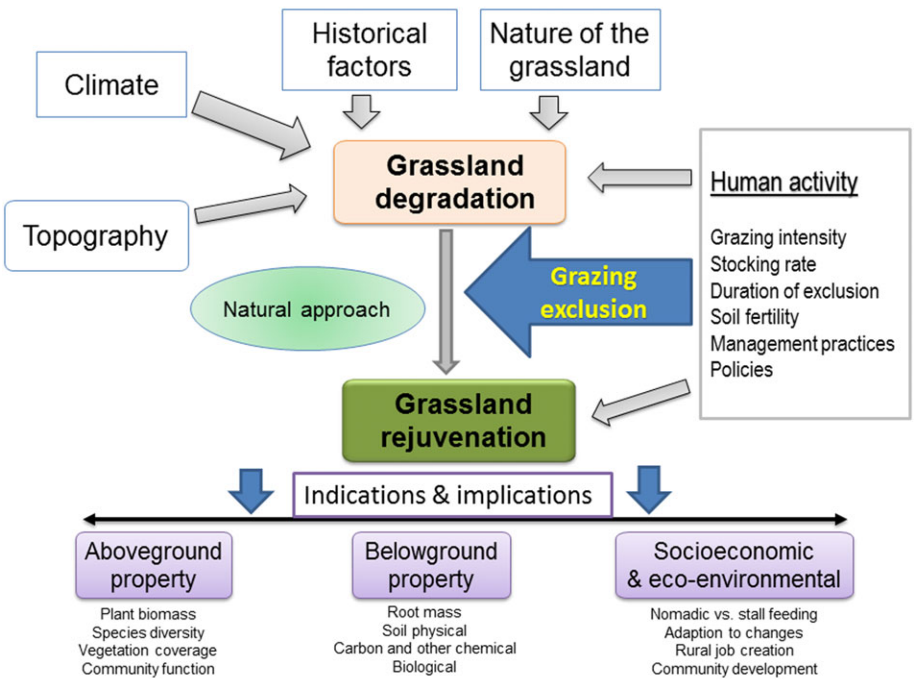

FIGURE 7 Grassland degradation in northern China is due to many factor interactions including topography, climate, grassland type, historical reasons, and human activity. Rejuvenation efforts through grazing exclusion practices affected the aboveground and belowground properties of grasslands and may have significant socioeconomic, ecological, and environmental implications [Colour figure can be viewed at wileyonlinelibrary.com]

semiarid steppes in the Great West to the high, alpine pastures in the Qinghai-Tibetan plateau and the meadows and forest steppes in the northeast (Figure 1b). Grazing exclusion can drastically increase whole-ecosystem C storage in mid- and tallgrass communities but is minimal in shortgrass communities. The ratio of root:soil carbon often differs between tallgrass and shortgrass communities, leading to the differences in SOC stocks between communities. The presence of legumes in grasslands can increase species richness in some degraded steppes (Z. Hu, Li, et al., 2016). Fine roots of legume plants in the upper soil profile can act as a principal driver in mediating the effect of plant community composition on soil biogeochemistry (Liu, Gan, Bueckert, Van Rees, & Warkentin, 2010). Legume species may promote plant community succession and accelerate the recovery of degraded grasslands with coarse soil.

2. *Duration of grazing exclusion and stocking rates* have a profound impact on the physical, chemical, and biological properties of grassland soils. In general, the longer the period of grazing exclusion, the greater the improvement in soil properties, as a new equilibrium can be reached after a few decades of grazing exclusion (Li, Zhao, Chen, Luo, & Wang, 2012). *Stocking rates* have a significant impact on C and N cycling in the plant–soil system of grasslands (Jiao et al., 2016). In general, the higher the stocking rate, the more negative the impact on soil properties. Stocking rates may have direct or indirect impacts on soil properties by influencing plant stands (Wang et al., 2014), soil and root respiration (Chen, Hou, Chen, Wan, & Millner, 2015), and animal behavior while grazing (Lin et al., 2011). An optimal stocking rate is difficult to define as it varies with various factors, such as grazing practices (Dong, Zhao, Wu, & Chang, 2014) and animal type.
3. *Geographical and climatic conditions* have a fundamental effect on the soil biogeochemistry of grasslands in relation to grazing practices. Across the major grasslands in northern China, annual evaporation is higher in the northwest than in the northeast, whereas annual precipitation has the reverse pattern. The variable climates not only affect the degree of degradation but also the progress of rejuvenation of degraded grasslands (Ma et al., 2017). Grazing exclusion increases plant community height, coverage, and aboveground biomass, but the magnitude of this effect varies between geographic locations (Hao et al., 2014). Thus, the effectiveness of grazing exclusion on grassland rejuvenation alters with climatic heterogeneity.
4. *Historical degree of degradation* affects the outcomes of the restoration effort for degraded grasslands (Jiang, Han, & Wu, 2006). Some grasslands in the northwestern desert steppes are degraded substantially due to historical reasons, such that a short period (2–6 years) of grazing exclusion is unlikely to restore the grassland properties to a productive level (Pei, Fu, & Wan, 2008). In contrast, a shorter period of grazing exclusion (3–5 years) will likely have an ameliorating effect on lightly degraded grasslands (Steffens, Kölbl, Totsche, & Kögel-Knabner, 2008).

7 | CONCLUSIONS

Grassland ecosystems in northern China face significant challenges due to the increased demand for food by the ever-growing human population, rapid urbanization that competes for available resources, and the evidenced global climate change. Most of the grasslands in arid and semiarid northern China were degraded mainly due to overgrazing, which reduced grassland productivity, lost soil fertility, and increased risks for erosion and desertification. Grassland management policies have been established for decades to help rejuvenate degraded grasslands, and numerous studies have been published individually. In this study, we reviewed the relevant studies, analyzed the results collectively using a meta-analysis approach, and summarized the key findings. We conclude that grazing exclusion practices can be employed as an effective approach to rejuvenate degraded grasslands naturally, as the practice has improved plant community traits, and enhanced soil physiochemical and biological properties of degraded grasslands. For the long-term sustainability of grasslands, we suggest multidisciplinary research across different grassland ecozones is conducted to determine how the sedentarization and privatization of grasslands associated with the grazing exclusion policies may impact the effectiveness of the rejuvenation processes. Additional, steps may be taken to explore the potential of developing 'grassland ecotourism' that gives urban citizens opportunity to nurture eco-culture and appreciate eco-beauty of grasslands.

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SUPPORTING INFORMATION

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

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