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Effects of grazing exclusion on carbon sequestration in China's grassland

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ABSTRACT

Globally, grazing exclusion (GE) is an effective management practice to restore degraded grasslands and improve carbon (C) stock. However, the C dynamics in grasslands ecosystem with GE have not been well characterized. The results of 145 sites published in 118 recent literatures were synthesized to examine the dynamics of plant and soil C sequestrations in grassland ecosystem after GE, and with the recovery age > 27 years under the China's 'Returning Grazing Land to Grassland' Project. Results showed a positive impact of GE on vegetation and soil C stock at most sites. The mean rate of aboveground biomass carbon stock (AGBC) change was 10.64 $g\ m^{-2}\ yr^{-1}$, and the mean rate of belowground biomass (0–30 cm) carbon stock (BGBC) change was 32.14 $g\ m^{-2}\ yr^{-1}$ after GE. The mean rate of soil C stock change was 0.27, 0.23, 0.18, 0.09 $Mg\ ha^{-1}\ yr^{-1}$ in 0–10 cm, 10–20 cm, 20–30 cm, and 30–100 cm (equivalent to 10 cm), respectively. And Grass-dominated grasslands present a higher C sequestration ability than forb-dominated grasslands. Soil C stock rates and vegetation biomass C changes showed an *Exponential Decay* trend since GE, and the AGBC changes reached a steady state (when the rate at the equilibrium point) first, followed by BGBC, and then soil C. The AGBC and BGBC both had opposite views on soil C changes in the top 30 cm soil layers. Soil N is a key factor in the regulation of soil C sequestration since long term GE (> 20 years). The large scale of GE under 'Returning Grazing Land to Grassland' Project significantly increased grassland C stocks. Meanwhile, increased soil N supply to grasslands with GE at the latter recovery stage may enhance ecosystem C sequestration capacity.

1. Introduction

The terrestrial biosphere includes both vegetation and soil, which are sources of goods, services and resources for humankind (Brevik et al., 2015; Keesstra et al., 2016). Soil also acts as a manager of the hydrological, erosional, biological and geochemical cycles that control the Earth system (Keesstra et al., 2012, 2016; Mol and Keesstra, 2012). The terrestrial biosphere can act either as a source or as a sink for atmospheric CO_2 (Novara et al., 2015; García-Díaz et al., 2016), both the vegetation and the soil may play a part in terrestrial ecosystem carbon (C) budget (Deng et al., 2017). It has long been recognized that land use change has a significant effect on the global C cycle through changing C stocks in terrestrial ecosystem (Laganière et al., 2010; Deng et al., 2014a, 2016; Bruun et al., 2015; Muñoz-Rojas et al., 2015; Choudhury et al., 2016; Novara et al., 2016; Deng and Shangguan, 2017). Although the contributions of land use change to anthropogenic CO_2 atmospheric emissions have recently been revised downward (IPCC, 2000), the estimated current annual contribution of 1.2 pg, or about 12–15% of total anthropogenic fluxes, is still significant (Van der Werf et al., 2009; Houghton et al., 2012). Therefore, a new challenge in the context of climate change mitigation is enhancing C sequestration in terrestrial ecosystems to conserve existing C stocks and to remove C from the atmosphere by increasing C pools in the terrestrial ecosystem (Liu et al., 2016; Deng and Shangguan, 2017; Frouz, 2017)

Grasslands are one of the world's most widespread vegetation types, occupy more than a third of the world's land surface, excluding Antarctica and Greenland, and support the livelihoods of approximately one billion people (Kemp et al., 2013). Grasslands can serve as a source of feedback for global climate through their strong potential for C sequestration (Fang et al., 2007). Nearly 100% of uncultivated grasslands are grazed by large mammals, and thus, grazing may be a critical controlling factor affecting ecosystem functioning in grassland ecosystem (Piñeiro et al., 2009; Hafner et al., 2012; Mcsherry and Ritchie, 2013; Deng et al., 2014b; Lin et al., 2015). Globally, overgrazing is one of the most important human disturbances (Mcsherry and Ritchie, 2013; Hu et al., 2016), causing severe degradation of grasslands (Wang et al., 2011; Deng et al., 2014b; Liu et al., 2016). Overgrazing severely reduces grassland productivity, vegetation cover, and the proportion of forage grasses (Schonbach et al., 2011; Deng et al., 2014b; Wang et al., 2016), which increases the risk of soil erosion and desertification

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(Steffens et al., 2008; Zhou et al., 2010). Furthermore, overgrazing may reduce the C sink function of grassland ecosystems (Shrestha and Stahl, 2008; Piñeiro et al., 2009; Schonbach et al., 2011; Zhou et al., 2011; Wang et al., 2014; Liu et al., 2016; Wang et al., 2016). Conversely, grazing exclusion (GE) is considered to be an effective approach to restore degraded grassland ecosystems, as well as to promote C sequestration (Mcsherry and Ritchie, 2013; Deng et al., 2014b; Álvarez-Martínez et al., 2016; Tarhouni et al., 2017).

Vegetation biomass and soil are two major C pools in the grassland ecosystem (Hu et al., 2016). Globally, although the importance of GE in grassland C sequestration and the dynamics of C pools and related controlling factors as a result of GE have been well reported, no general conclusion on the effect of GE on C stocks is promotes or reduces has been drawn yet (Mcsherry and Ritchie, 2013; Hu et al., 2016). For example, GE promotes C sequestration in some case (Steffens et al., 2008; Wang et al., 2016), decreases them in other studies (Shrestha and Stahl, 2008; Hafner et al., 2012), and in some experiments has no effect (Hu et al., 2016). Due to this lack of consensus on the effect of GE on C sequestration of grassland ecosystem, there is little knowledge about how much C is sequestration or loss after grassland with GE (Hu et al., 2016). In addition, most of previous studies merely focus on the comparison of grassland C stock between the GE sites and the grazing sites (Deng et al., 2014b; Chen et al., 2015; Wang et al., 2016; Zhu et al., 2016). Therefore, little is known about the dynamics of C stocks following grassland with GE. Specifically, we know little about how many years the grassland C pools needed to recover to the steady state (equilibrium point), and how the rates of changes in C pools varied across sites in relation to environmental conditions (Mcsherry and Ritchie, 2013; Hu et al., 2016). Moreover, the effects of GE on grassland C sequestration are varied with temperature and precipitation gradients with different climatic conditions, and more systematic analysis is required (Christopher et al., 2009; Luo et al., 2010).

Grasslands in China cover approximately 40% of the total national land, accounting for approximately 6–8% of the total world grassland area and contain 9–16% of the world's total grassland C stocks (Ni, 2002; Fan et al., 2008), of which most is associated with grazing (Zhao et al., 2005). Due to the heavier grazing pressure, China has a markedly higher percentage of degraded grasslands than other countries at the same latitude. Grazing exclusion is regarded as the most effective method for restoring degraded grasslands and reversing grassland desertification. For a long period, to promote degraded pasture recovery and to balance the livestock rate with forage productivity, China has implemented the policy of 'Returning Grazing Land to Grassland' Project. As a consequence of this policy, grasslands have improved in China (China Ministry of Agriculture, 2008). Therefore, grazed and non-grazing grasslands across regional grassland types provide us with a natural comparative experiment to test the effects of GE on the grasslands C sequestration dynamics along environmental gradients across China grassland. Despite Hu et al. (2016) have studied the effect of GE on grassland C changes in China used a synthesis analysis, which only reported the changes in plant biomass and soil organic C (SOC) concentrations not C stocks reported. Although plant biomass and SOC concentrations can reflect C changes in the ecosystems to some extent, C stocks can more reflect how much C that an ecosystem own, especially in the soil, because grazing effects on SOC concentration may be confounded by grazer effects on soil bulk density (BD) (Mcsherry and Ritchie, 2013). Thus, while much informative research has been done using SOC concentrations (Hu et al., 2016), it is important to consider only C stock in the synthesis, as we were interested mainly in soil's potential to sequester CO_2 (Smith et al., 2014).

To explore the effect of grazing exclusion on C stock in China grassland, this study gathered 118 existing studies from the literature in which GE effect on grassland biomass and SOC to conduct a synthesis. A synthesis offer an important advantage over traditional narrative reviews in that they provide a quantitative approach to comparing results between studies (Wang et al., 2011; Deng et al., 2014a, 2014b; Hu et al., 2016). Thus our study represents a relatively novel approach to address the following questions: (1) What are the temporal pattern of the rates of C stock changes along the years of GE? And how many years does the grassland ecosystem recover to a steady state? (2) what are the critical factors (e.g. age, soil depth, temperature, precipitation, soil N, grassland type) to effect on the rate of C change. And (3) How much C can be sequestrated per year in C pools for grasslands in China with GE? We hypothesize that GE improves both vegetation biomass C stock and soil C stock, and that the rates of C change for all C pools decline linearly with age.

2. Materials and methods

2.1. Data compilation

All of the available peer-reviewed publications and concerning changes in grazing exclusion grassland were collected in our synthesis. And one database was compiled by searching the Web of Science and China National Knowledge Infrastructure for studies that were published before May 2016. The following key words were used to select the studies: Grazing exclusion, fencing, soil carbon/nitrogen (C/N), biomass, grassland, and China. In our study, the following criteria were used to select publications for analysis:

(1) Soil C stocks were provided or could be calculated based on SOC or SOM concentration, bulk density and soil depth;
(2) There were data on both the grazing exclusion grassland (GE) and the grazing grassland (CK);
(3) Only studies using paired-site chronosequence, with similar soil and climatic conditions for both the grazing and grazing exclusion sites, were selected for the database;
(4) The number of years since land use conversion were either clearly given or could be directly derived;
(5) In the studies, only the first rotation of land use conversion was considered and data for 0–100 cm soil layers were extracted;
(6) Location, mean annual temperature (MAT, °C), and mean annual precipitation (MAP, mm) clearly given;
(7) Adequate replications and uniform soils (studies were excluded if the experiments were not adequately replicated or if the paired sites or sites in chronological sequence were confounded by different soil types);
(8) Sampling depths for belowground biomass (BGB) varied in different studies. Considering that most root biomass is distributed in the first 30 cm and most studies sampled root to this depth, data of root biomass in 0–30 cm were used to investigate the rate of C stock change in BGB.

In total, the final dataset comprised 118 studies (Appendix Dataset S1) most of them published between 2005 and 2016, including 145 sites in ten provinces of China (Fig. 1), which distributed most area of the China's 'Returning Grazing Land to Grassland' Project.

The raw data were either obtained from tables or extracted by digitizing graphs using the GetData Graph Digitizer (version 2.24, Russian Federation). For each paper, the following information was compiled: sources, location (longitude and latitude), climatic data (mean annual temperature and precipitation), land use conversion types (including both grazing exclusion sites and grazing sites, dominant species, age (years since grazing exclusion), above- and below-ground biomasses (AGB and BGB), soil depth from soil surface, soil bulk density, and amount of SOC and TN in each layer of 0–100 cm soil depths (Appendix Dataset S1). To depict more apparent trends of the C pools, the ages of GE were divided into ten groups: 1–3, 4–6, 7–9, 10–12, 13–15, 16–18, 19–21, 22–24, 25–27, and > 27 years. This age groups also used by the Hu et al. (2016)'s study. In addition, we divided the dominant species into two functional groups: grass (plant species of the Poaceae) and forb (any herbaceous, dicotyledonous broad-leaved

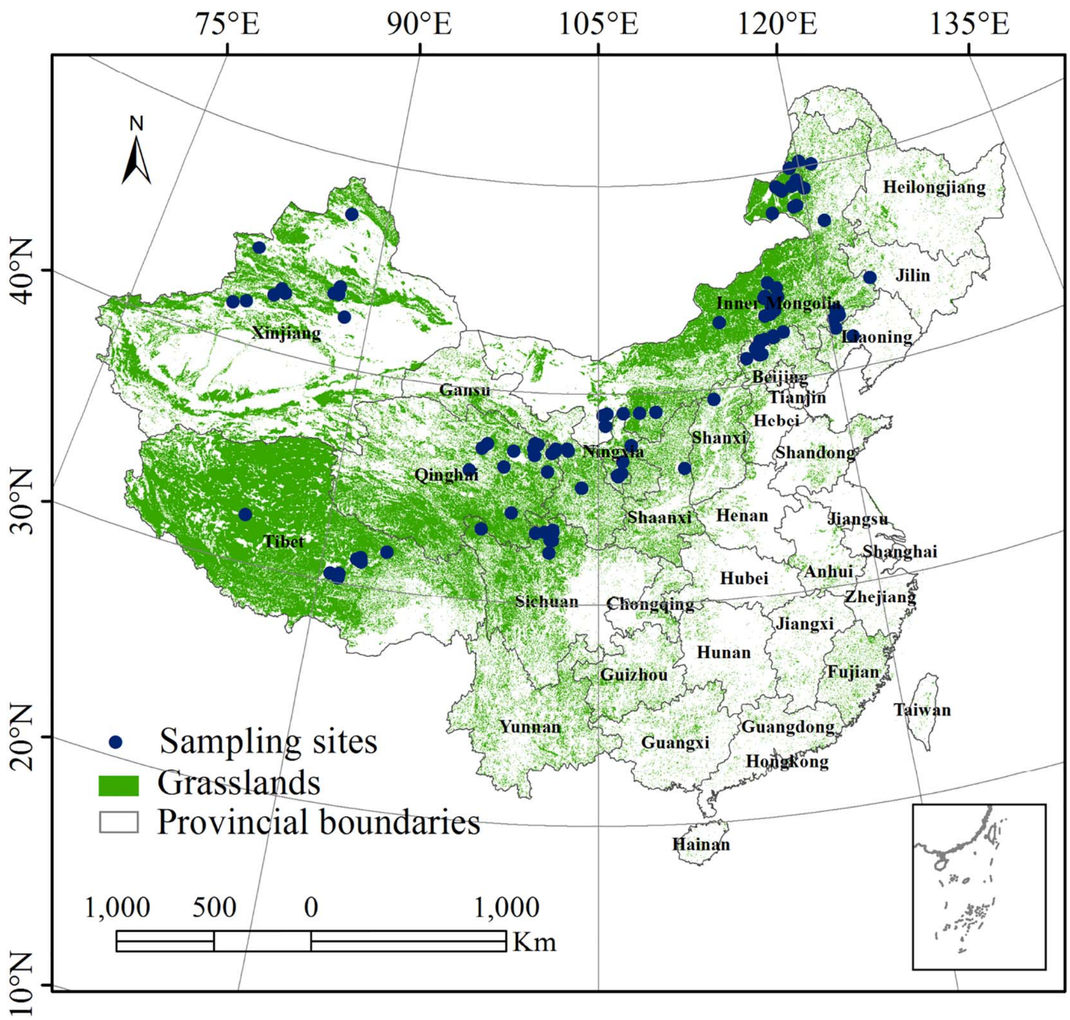

Fig. 1. Sampling sites distribution of the individual studies collected in this synthesis.

plant), to explore the effect of grassland type on the rate of C changes.

2.2. Data calculation

2.2.1. Vegetation C stock

The study used the following equation to calculate the vegetation C stock (Fang et al., 2007):

$$C_B = B \times C_f \tag{1}$$

In which, C_B is the vegetation C stock (g m^{-2}), B is the vegetation biomass (g m^{-2}), and C_f is the plant biomass C coefficient. The study set 0.45 as the plant biomass C coefficient for estimating the herbaceous C stock.

2.2.2. Soil C stock

If the samples reported only had SOM, their SOC were calculated by the relation between SOM and SOC. The formula for the calculation is as follows:

$$\mathrm{SOC} = \mathrm{SOM} \times 0.58 \tag{2}$$

where SOC is the soil organic C (g kg^{-1}) and SOM is the soil organic matter (g kg^{-1}).

The SOC stocks was calculated using the following equation

$$C_s = \frac{\mathrm{SOC} \times \mathrm{BD} \times \mathrm{D}}{10} \tag{3}$$

in which, C_s is soil organic C stocks (Mg ha^{-1}); SOC is soil organic C concentration (g kg^{-1}); BD is soil bulk density (g cm^{-3}); and D is soil thickness (cm).

Soil BD estimates are critical for calculations of C_s, but many studies did not measure this attribute. We established an empirical relationship between SOC concentration and soil BD with the reported values for grazing (CK) sites and grazing exclusion (GE) sites from the Appendix Dataset S1 (Fig. 2). Then, the missing values of soil BD were interpolated using the predicted values from the empirical functions (*Exponential Decay*, *Double*, 4 *Parameter*) in Fig. 2. The formula for the calculation is as follows (Eqs. (3) and (4)):

$$\mathrm{BD_{CK}} = 0.20e^{-1.48\mathrm{SOC}} + 1.49e^{-0.01\mathrm{SOC}},\ r^2 = 0.789,\ p < 0.0001, \tag{4}$$

$$\mathrm{BD_{GE}} = 0.22e^{-0.68\mathrm{SOC}} + 1.44e^{-0.01\mathrm{SOC}},\ r^2 = 0.824,\ p < 0.0001, \tag{5}$$

To increase the comparability of data derived from different studies, the original soil C data were converted to soil C stocks in the top 100 cm using the depth functions developed by Jobbágy and Jackson (2000) according to the following equations:

$$\mathrm{Y} = 1 - \beta^d \tag{6}$$

$$\mathrm{X}_{100} = \frac{1 - \beta^{100}}{1 - \beta^{d0}} \times \mathrm{X}_{d0} \tag{7}$$

For observations that only had 0–100 cm soil C stocks, using Eq. (6) we can derive:

$$\mathrm{X}_{d0} = \frac{1 - \beta^{d0}}{1 - \beta^{100}} \times \mathrm{X}_{100} \tag{8}$$

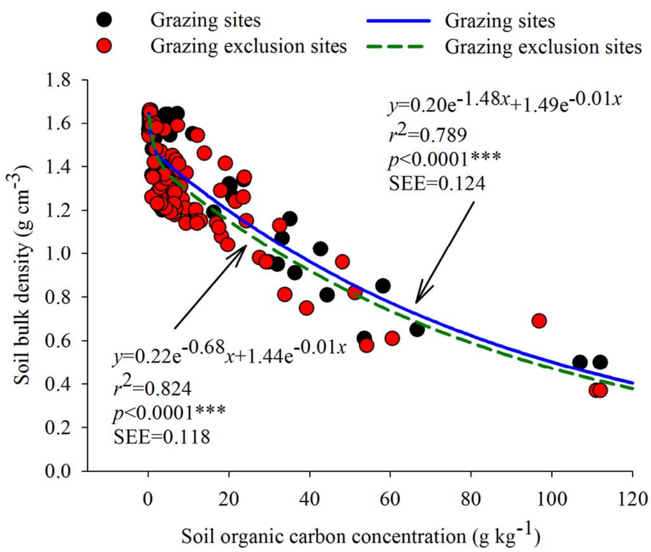

Fig. 2. Empirical functions (*Exponential Decay*, *Double*, 4 *Parameter*) for estimating the missing soil bulk density based on data from studies reporting soil organic carbon concentration and soil bulk density in the two land use types of grazing and grazing exclusion sites. r^2 is coefficient of determination, SEE is standard error of estimate. n = 104.

where Y represents the cumulative proportion of the soil C stock from the soil surface to depth d (cm); β is the relative rate of decrease in the soil C stock with soil depth; X_{100} denotes the soil C stock in the upper 100 cm; *d*0 denotes the original soil depth available in individual studies (cm); and X_{d0} is the original soil C stock. Although Jobbágy and Jackson (2000) provided the depth distribution of soil C for 11 biome types globally, there was no significant difference in the depth distribution among biome types or between individual biomes and the global average. Therefore, in the present study, the global average depth distributions for C were adopted to calculate β (i.e., 0.9786) in the equations.

It should be noted that potential uncertainties may be introduced by this dataset standardization, mainly due to the difference in C distribution through the soil profile between grazing sites and grazing exclusion sites, and among the different stages following grazing exclusion. However, as has been stated, there was no significant difference among the 11 biome types included in Jobbágy and Jackson (2000) or between individual biomes and the global average in terms of soil C distribution with depth. The same method was used by Yang et al. (2011) and Li et al. (2012), both of whom concluded that depth correction did not alter the overall pattern of soil C stock dynamics during vegetation development.

2.2.3. C sequestration rate

The C sequestration rate is estimated depending on changes to C stocks in different time sequences. The study set the C stocks of grazing sites as the baseline for calculating the rate of C stock change since grazing exclusion. We first calculated the C sequestration value (ΔC) for each grazing exclusion site since grazing exclusion used the following equation:

$$\Delta C = C_{LUn} - C_{LU0} \tag{9}$$

in which, C_{LUn} represents C stocks [Biomass C stocks (g m^{-2}) or soil C stocks (Mg ha^{-1})] at grazing exclusion site, and C_{LU0} is C stocks at the paired grazing site.

We used mean annual absolute rate of change in C stock to indicate C sequestration rate following grazing exclusion (R_s, Mg ha^{-1} yr^{-1}). The calculated equation is as follows:

$$R_s = \frac{\Delta C_s}{\Delta Age} \tag{10}$$

In order to reflect the dynamics of C stocks, C sequestration were summed for each category. In this case, a methodology reported previously (Luo et al., 2006; Deng et al., 2016) was used to calculate 95% CI of means for C sequestration, as shown in Eqs. (11) and (12):

$$SE_{total} = \sqrt{\frac{V_S}{n}} \tag{11}$$

$$95\%CI = 1.96 \times SE_{total} \tag{12}$$

where SE_{total} denotes the standard error of the relative change in C stock. V_S and n are the variance of relative C stock change and the number of observations, respectively. In this study, the 95% confidence interval (CI) was calculated for each category. And the observed effect sizes are considered statistically different from zero if the 95% CI does not include zero.

In addition, in order to explore soil carbon–nitrogen coupling relationship after grazing exclusion, we have done a regression analysis between rates of soil C sequestration and rates of soil N sequestration since grazing exclusion. The method of estimating the rates of soil N sequestration was similar to the method of soil C sequestration rate in our study.

2.3. Data analysis

ANOVA was conducted to evaluate whether the rates of changes in biomass C stock and soil C stock changes were significantly different in different recovery ages and grassland types. Differences were evaluated at the 0.05 significance level ($p < 0.05$). When testing for the homogeneity of variance was passed and significance was observed at the $p < 0.05$ level, a least significant difference (LSD) test was used for multiple comparisons. Pearson correlation analysis was conducted to analyze the correlations between the rates of C stock change and climatic factors (MAP, MAT) and recovery age across the grassland site since grazing exclusion. Regression analysis was conducted to analyze the relationships between the rates of soil C change and the rates of soil N change. Meanwhile, *t*-tests were conducted to evaluate whether GE significantly increased C/N ratios in different soil layers. In addition, a multivariable linear regression analysis was conducted to quantify the contributions of relevant factors to the variations in the rates of soil C change. All statistical analyses were performed using the software program SPSS, ver. 17.0 (SPSS Inc., Chicago, IL, USA).

3. Results

3.1. Changes in plant biomasses C

Overall, grazing exclusion had significantly increase biomass C stock ($p < 0.05$) (Figs. 3A and B, 4). The mean rate of aboveground biomass C stock (AGBC) change was 10.64 g m^{-2} yr^{-1} (95% CI = 2.67), and the mean rate of belowground biomass C stock (BGBC) change was 32.14 g m^{-2} yr^{-1} (95% CI = 7.89) after grazing exclusion (Fig. 3A). The rate of AGBC change was significantly declined along with the years of grazing exclusion increase, which showed an *Exponential Decay* trend since grazing exclusion (Fig. 4A). The rate was higher in the early stage (< 3 yr), with the rate of 27.75 g m^{-2} yr^{-1}, and after (> 3 yr), the rate showed a non-significant difference among each recovery age since grazing exclusion, with the mean rate of 5.47 g m^{-2} yr^{-1} (Fig. 4A). Similar to the rate of AGBC change, the rate of BGBC change also showed an *Exponential Decay* trend since grazing exclusion (Fig. 4B). However, the duration of large increase in the early stage of BGBC change was longer than that for AGBC change. An obvious increase of BGBC (60.55 g m^{-2} yr^{-1}) was found in the first 6 years, followed by mild increase in years > 6, with the mean rate of 12.11 g m^{-2} yr^{-1} (Fig. 4B).

Regressing the rates of changes in biomass C pools with the year of grazing exclusion, showed an exponential decrease depicting the

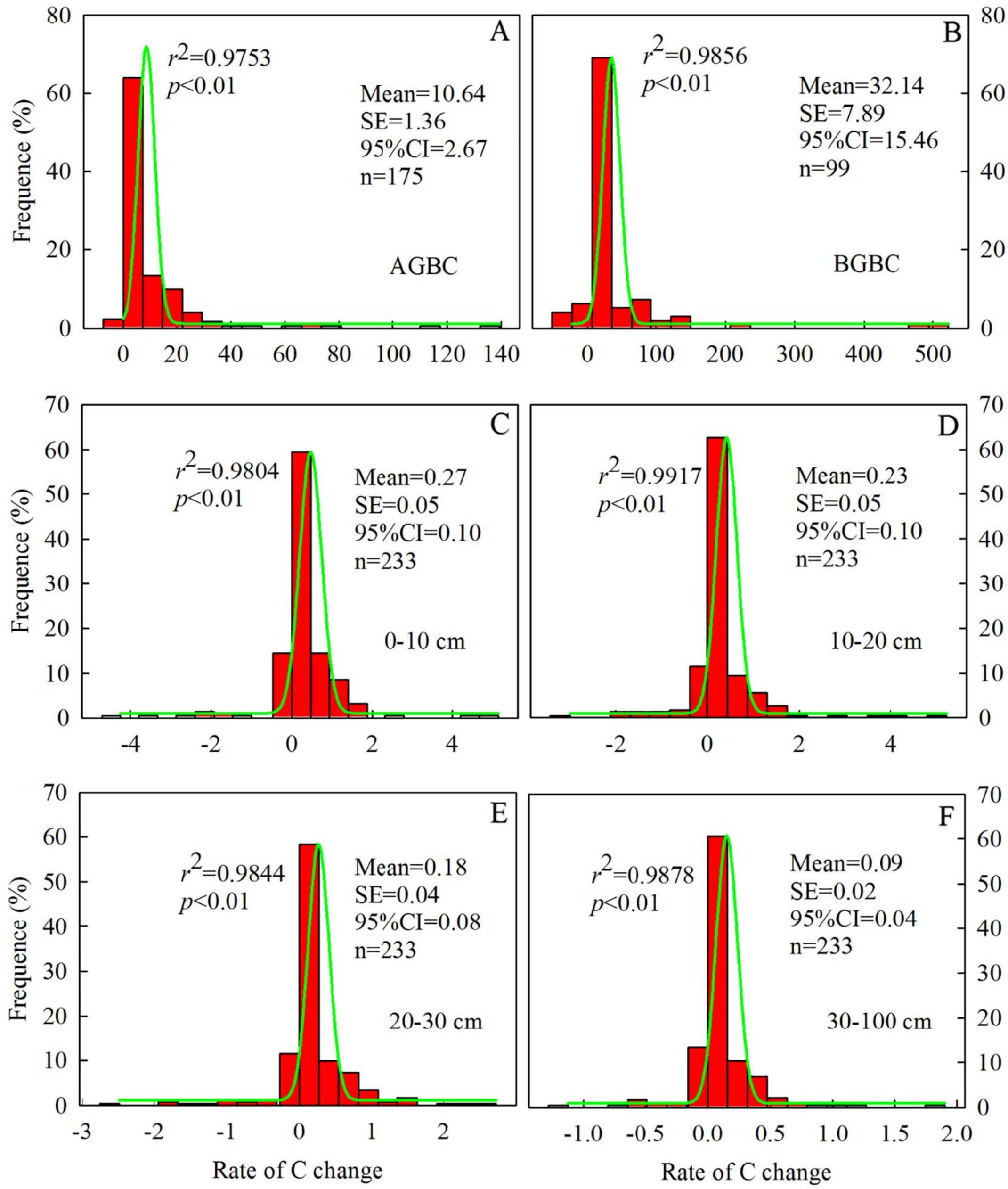

Fig. 3. Frequency distribution of the rate of (A) AGBC ($g\ m^{-2}\ yr^{-1}$), (B) BGBC ($g\ m^{-2}\ yr^{-1}$), soil C ($Mg\ ha^{-1}\ yr^{-1}$) of 0–10 cm (C), 10–20 cm (D), 20–30 cm (E) and 30–100 cm (equivalent to 10 cm) (F) changes with grazing exclusion. The curve was fitted by a Gaussian function (4 *Parameter*). Note: AGBC, aboveground biomass carbon stock; BGBC, belowground biomass carbon stock. r^2 is coefficient of determination, SE is standard error.

dynamics of the rates of changes in AGBC and BGBC (Appendix Fig. S1). In comparison, AGBC changes reached a steady state (when the rate at the equilibrium point) first, followed by BGBC, indicating that the increase of BGBC lags behind the accumulation of AGBC (Appendix Fig. S1).

3.2. Changes in soil C stocks

The result showed that soil C stock increased with grazing exclusion at most sites in grasslands in China (Fig. 3). The rates of soil C stock change varied greatly among observations, exhibiting a skewed distribution (Fig. 3C–F). Briefly, > 82–87% of the total observations illustrated increases of soil C stock with grazing exclusion, and on the contrary, some observations (i.g. 13–18%) was decreased with grazing exclusion. The mean rate of soil C stock change was 0.27 (95% CI = 0.10), 0.23 (95% CI = 0.10), 0.18 (95% CI = 0.08), 0.09 (95% CI = 0.04) $Mg\ ha^{-1}\ yr^{-1}$ in 0–10 cm, 10–20 cm, 20–30 cm, and 30–100 cm (equivalent to 10 cm), respectively.

Soil C stock changes had similar temporal patterns in either the topsoils (< 30 cm) or the subsoil layer (30–100 cm) (Figs. 5 and 6). When used the whole dataset collected to synthesis, the results showed that the rates of soil C stock change had non-significant difference among different recovery ages since grazing exclusion in the 0–10, 10–20, 20–30 and 30–100 cm soil layers ($p > 0.05$) (Fig. 5). Moreover, the rates in the four soil layers all showed the rates had non-significant difference to zero in the early stage (< 3 yr), indicating that soil C stock had non-significant changes ($p > 0.05$) (Fig. 5). However, the rate of soil C stock change all larger than zero in the later (> 3 yr), indicating that soil C stock had a significant changes in the years > 3 since grazing exclusion ($p > 0.05$) (Fig. 5). Due to > 82%–87% of the total observations illustrated increases of soil C stock with grazing exclusion, and only 13%–18% was of the total observations decreased with

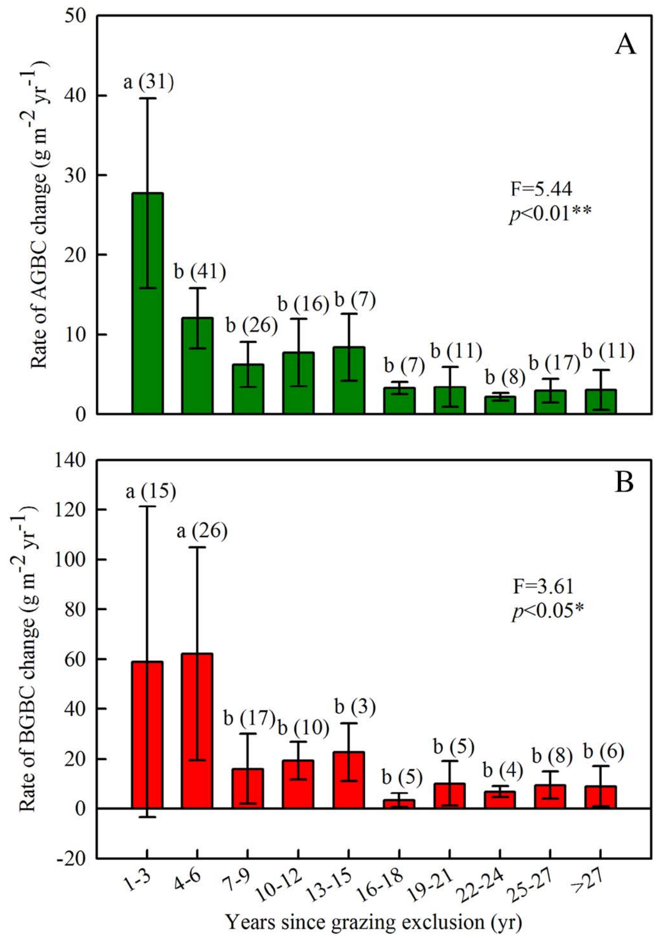

Fig. 4. Variations of the rates of changes in aboveground biomass carbon stock, AGBC (A), and belowground biomass carbon stock, BGBC (B), with the age of grazing exclusion. Note: The error bar indicates mean ± CI (95%). The different letters above the error bars indicate significant difference among the different restoration stages at 0.05 level ($p < 0.05$). Values in parenthesis are the number of observations.

grazing exclusion. So we have done the other temporal dynamics analysis of soil C stock changes excluding the observations with the rates were less than zero. The re-analysis results showed that the rates of soil C stock change were significantly different among the different recovery ages ($p < 0.05$) (Fig. 6). The rate of soil C stock change in the four soil layers was significantly declined along with the years of grazing exclusion increase, which showed an *Exponential Decay* trend since grazing exclusion (Fig. 6). Overall, according to the differences in magnitude of the increase of soil C stock since grazing exclusion, three recovery periods can be determined, but the duration of the second period were different in the four soil layers. For example, for the surface (0–10 cm) soil layer, the second recovery period is about 4–15 years, with the mean rate of 0.52 Mg ha^{-1} yr^{-1}; and for the deeper (30–100 cm) soil layer, the second recovery period is about 4–6 years, with the mean rate of 0.22 Mg ha^{-1} yr^{-1} (equivalent to 10 cm) (Fig. 6A and D). Moreover, the rate was highest in the early stage (< 3 yr), with the rate of 1.24, 1.52, 0.66 and 0.48 Mg ha^{-1} yr^{-1} in the 0–10, 10–20, 20–30 and 30–100 cm (equivalent to 10 cm) soil layers, respectively (Fig. 6). From the above two analysis, we can know that there had great spatial variability in the first 3 years since grazing exclusion, due to more observations with reductions of soil C stock being observed at some sites in the early recovery age than the later, leading to the rate in the first 3 years had significant difference between the two analysis. Because the negative effect of soil C stock at some sites, leading to the rate of soil C stock change in the early stage in grazing exclusion site had non-significant difference with grazing site.

ANOVA analyses indicated that grazing exclusion had significantly increased soil C stock ($p < 0.05$) in the 0–100 cm soil profile (Fig. 7). The rates of soil C stock change were significant decreased from surface soil to deeper soil (Fig. 7). The average rate of increase was 0.27 Mg ha^{-1} yr^{-1} in 0–10 cm, followed by 0.23 Mg ha^{-1} yr^{-1} in 10–20 cm, 0.18 Mg ha^{-1} yr^{-1} in 20–30 cm, and 0.09 Mg ha^{-1} yr^{-1} in 30–100 cm. Moreover, the rates of soil C stock change in the topsoil (0–30 cm) were significant higher than that in the deeper soil (30–100 cm) ($p < 0.05$), and the mean rate of soil C stock change in the topsoil (0–30 cm) was 0.23 Mg ha^{-1} yr^{-1} (Fig. 7). In addition, similar to the trend of biomass C stock changes, the rates of changes in soil C stocks with the year of grazing exclusion also showed an exponential decrease (Appendix Fig. S1). In comparison, the rates of soil C stocks changes reached a steady state (when the rate at the equilibrium point) followed by AGBC and BGBC (Appendix Fig. S1), indicating that the increase of soil C stock lags behind the accumulation of biomass C stock after grazing exclusion.

3.3. Factors effect on C sequestration

Pearson correlations analysis showed that the rate of C changes in AGBC and BGBC both had significant positive correlations with the MAP ($p < 0.01$), and negative correlations with Age ($p < 0.01$), but they hadn't significant correlations with the MAT ($p > 0.05$) (Table 1). The rate of C changes in the 0–10, 10–20, 20–30 cm soil layers had significant positive correlations with the MAT ($p < 0.05$, $p < 0.01$, $p < 0.01$), and they all hadn't significant correlations with the MAP and Age ($p > 0.05$) (Table 1). In the deeper soil of 30–100 cm, the rate of C changes had no significant correlation with MAP, MAT and age ($p > 0.05$) (Table 1).

The multivariable linear regression model analysis had similar results with the Pearson correlation analysis. The MAP and age had significant effect on rate of C changes in AGBC and BGBC and the MAT had significant effect on the rate of C changes in the 0–10, 10–20 and 20–30 cm soil layers (Table 2). Moreover, the results showed that MAP, MAT and Age played more important roles in affecting the rate of C changes in AGB and BGB than that affect the rate of soil C changes since grazing exclusion (Table 2).

A significant linearly positive correlation between the rates of soil C change and the rates of soil N change was found in grasslands with GE, indicating that soil C change is strongly coupled with N change ($p < 0.01$, Fig. 8A). The C/N ratios at grazing exclusion sites was higher ($p < 0.05$) in the late stage (> 20 years) compared to the early recover stage (< 20 years). The ratios in the 0–10, 11–20, > 20 years were 10.8, 10.5 and 12.2, respectively. And the ratios at grazing sites (range from 10.1 to 10.7) had no significant difference among the three recovery stages (Fig. 8B). However, soil C/N ratios at grazing exclusion sites were significantly higher compared with grazing sites in the late recovery stage (i.e., > 20 years) (Fig. 8b).

4. Discussions

4.1. Change in biomass C pool following with GE

GE can alter plant allocation pattern at the community level, for example, plant cover, density height, and biomass increased significantly following GE (Liang et al., 2009; Wang et al., 2014). In our study, we found the mean rate of aboveground biomass C stock (AGBC) change was 10.64 g m^{-2} yr^{-1} and the mean rate of belowground biomass C stock (BGBC) change was 32.14 g m^{-2} yr^{-1} after GE in China's grassland (Fig. 3). Many previous have found that the AGBC and BGBC is significant increased after GE in the arid and semi-arid environments. A > 200% increase in AGBC within the enclosure was also reported from 5 to 15 year enclosures in northeast Africa rangelands (Yayneshet et al., 2009). Bagchi and Ritchie (2010) reported a 32–33% increase in AGBC and a 21–63% increase in BGBC in GE compared to

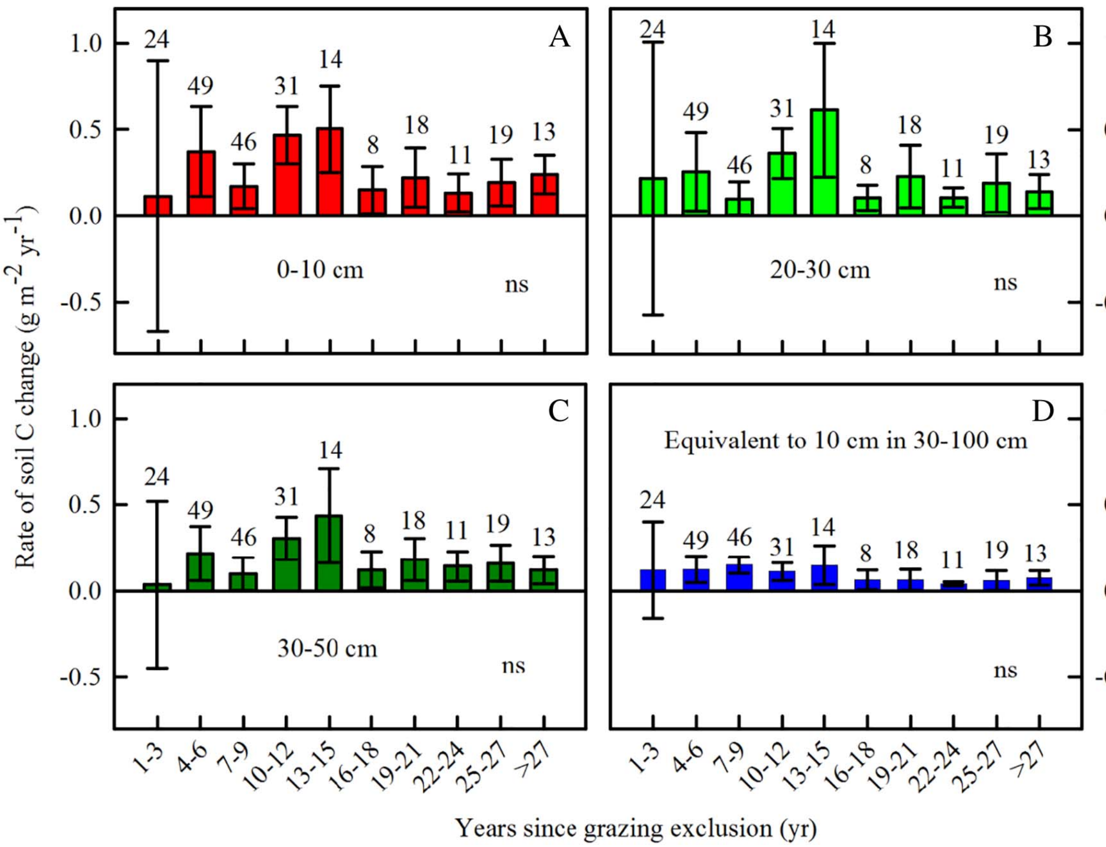

Fig. 5. Variations of the rate of soil C change with the ages of grazing exclusion in different soil layers (The dataset is the whole data collected). Note: The error bar indicates mean ± CI (95%). ns, indicate non-significant difference at 0.05 level among the different restoration stages ($p > 0.05$). Values above the bars were the number of observations.

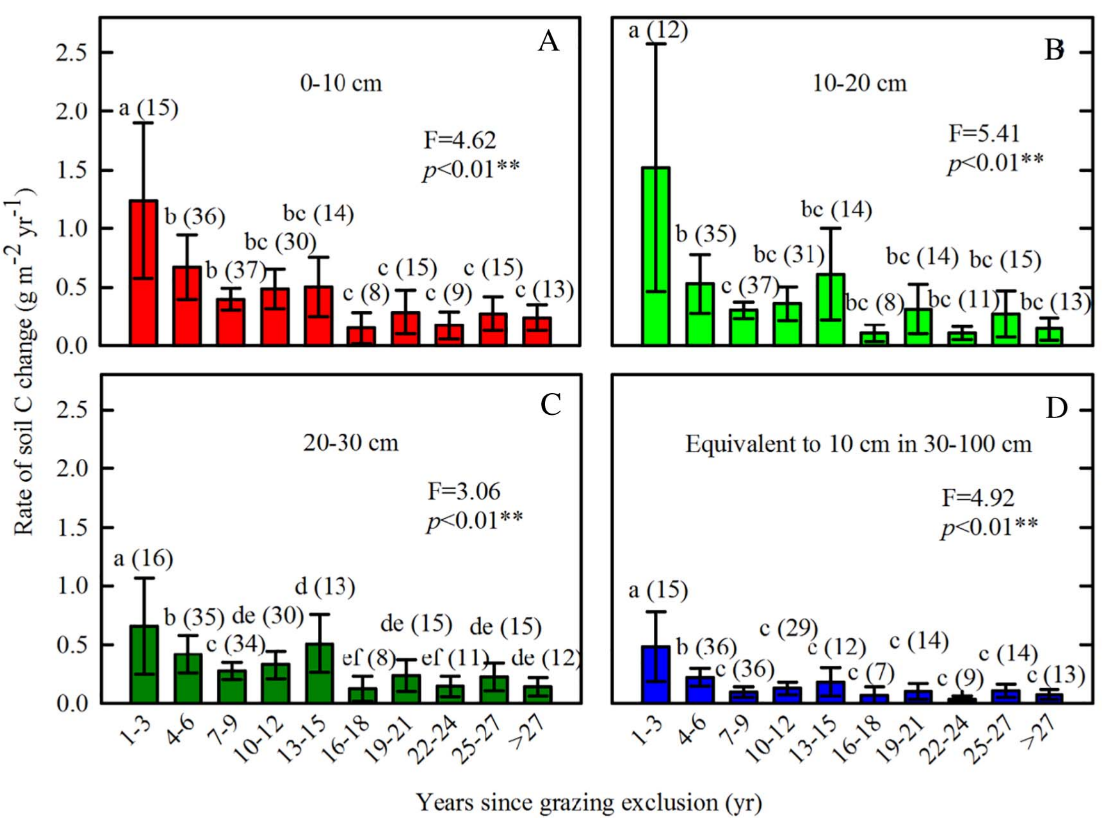

Fig. 6. Variations of the rate of soil C change with the ages of grazing exclusion in different soil layers (The dataset excluded the observations with the rates were less than zero). Note: The error bar indicates mean ± CI (95%). ns, indicate non-significant difference at 0.05 level among the different restoration stages ($p > 0.05$). Values above the bars were the number of observations.

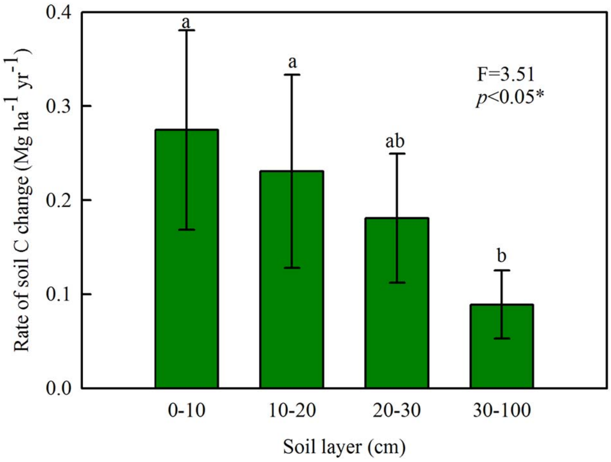

Fig. 7. Variations of the rate of soil C change at different soil layers. Note: the error bar indicates mean ± CI (95%). The different letters above the error bars indicate significant difference among the different soil layers at 0.05 level ($p < 0.05$). n = 233.

Table 1
Pearson correlation coefficients between the rate of C changes in AGB, BGB, and 0–100 cm soil layers and E, N, MAP, MAT and age across the grassland site since grazing exclusion.

Rate of C changes	MAP	MAT	Age	N
AGBC	**0.313****	− 0.053	**− 0.360****	**175**
BGBC	**0.195***	− 0.088	**− 0.257****	**99**
0–10 cm soil	0.023	**0.138***	− 0.020	233
10–20 cm soil	0.115	**0.188****	− 0.014	233
20–30 cm soil	0.084	**0.184****	0.013	233
30–100 cm soil	0.023	0.119	− 0.064	233

* Indicate correlation is significant at the 0.05 level (2-tailed) ($p < 0.05$)
** Indicate correlation is significant at the 0.01 level (2-tailed) ($p < 0.01$).

open grazing land in the northern India. Schuman et al. (1999) observed a 20–52% and 7–16% increase of C in AGBC and BGBC (0–60 cm depth), respectively, after 12 years of GE on a native mixed grassland in Wyoming, USA. In a heavily grazed region in Norway, Speed et al. (2014) have observed that the rate of AGBC change was around 31.67 g m^{-2} yr^{-1} following 12 years of GE. These rates are not high, compared to, for example, the 50.0 g m^{-2} y^{-1} reported for the impact of the cessation of livestock grazing in Scottish upland grasslands (Smith et al., 2014). In an East African savanna ecosystem, long-term GE (> 20 years) had increased the AGBC and BGBC from 0.48 to 0.75 Mg ha^{-1}, from 0.66 to 1.56 Mg ha^{-1}, respectively (Yusuf et al., 2015). However, not all studies have reported the positive results, for example, in South America with higher MAP (From 861 mm to 1406 mm) and MAT (from 14.9 °C to 18.9 °C), Piñeiro et al. (2009) found that the BGBC was lower in GE than in grazed stands, such changes represented a loss of 12.8 g m^{-2} y^{-1}.

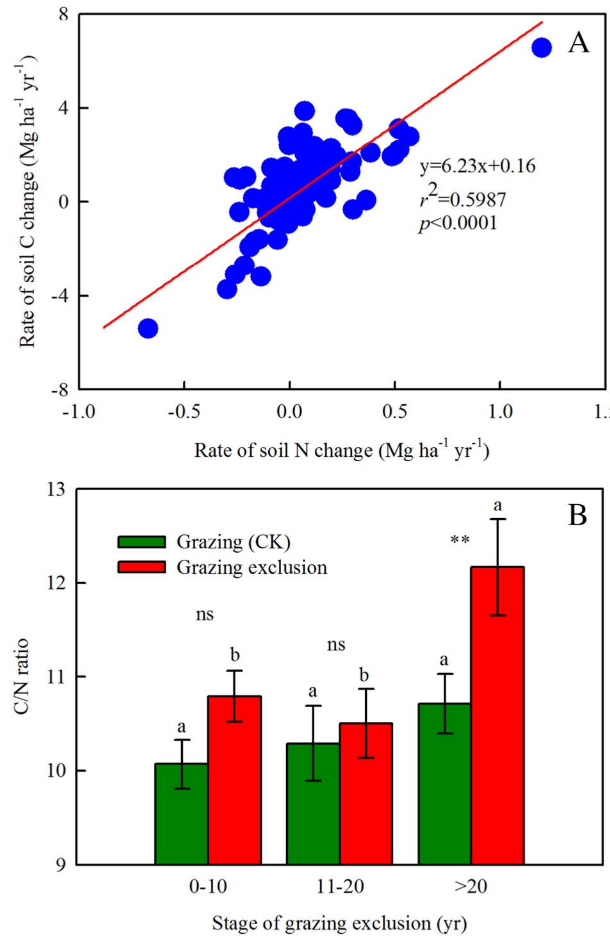

Fig. 8. Effect of grazing exclusion on carbon–nitrogen coupling relationship: (A) the relationship between rates of soil C sequestration and rates of soil N sequestration since grazing exclusion, (B) soil C/N ratio at early (1–10 years), middle (11–25 years), and late (> 20 years) stages at grazing sites (CK) and grazing exclusion sites. Note: ns indicate nonsignificant difference ($p > 0.05$), and ** indicate significant difference between grazing sites and grazing exclusion sites at 0.01 level ($p < 0.01$). The different letters above the error bars indicate significant difference among the different restoration stages at 0.05 level ($p < 0.05$). The C/N ratio values are mean ± SE.

4.2. Change in of soil C pool following with GE

Previous studies have found mixed results of GE effects on soil C accumulation, with studies showing positive (Pei et al., 2008; Golluscio et al., 2009), neutral (Shrestha and Stahl, 2008) or negative effects of GE (Reeder and Schuman, 2002). In our study, we found soil C stock increased with grazing exclusion at most sites in grasslands in China (Fig. 3). This result is consistent with most studies on grassland of the world (Pei et al., 2008; Steffens et al., 2008; Golluscio et al., 2009; Hu et al., 2016). There are three mechanisms support this increase of soil C

Table 2
Multivariable linear regression model analysis between the rate of C changes in AGB, BGB, and 0–100 cm soil layers and MAP, MAT and age across the grassland site since grazing exclusion.

Rate of C changes	Equation	R^2	Sig. (p)	N
AGBC	Y = 0.302MAP* − 0.04MAT − 0.349Age*	0.223	0.000	175
BGBC	Y = 0.212MAP* − 0.086MAT − 0.243Age*	0.111	0.011	99
0–10 cm soil	Y = 0.020MAP + 0.136MAT* − 0.022Age	0.019	0.215	233
10–20 cm soil	Y = 0.111MAP + 0.186MAT* − 0.013Age	0.048	0.011	233
20–30 cm soil	Y = 0.081MAP + 0.181MAT* + 0.013Age	0.040	0.025	233
30–100 cm soil	Y = 0.019MAP + 0.118MAT − 0.065Age	0.018	0.236	233

Note: The equations' regression coefficient is standardized coefficients. $p < 0.05$ indicate significant.
* Indicate the effect was significant among the three variables (MAP, MAT and age).

stock. First, GE reduces the output of C from the ecosystem to livestock and increase of net primary productivity due to the removal of grazing pressure accelerates organic matter (litter, dead roots, mycorrhizae, and exudates) input into the soil (Deng et al., 2014b; Zhu et al., 2016). Second, GE can increase soil moisture through improving the capacity of soil water conservation by reducing bare soil water evaporation due to the increase of vegetation height, canopy cover, and mulch resulting in higher plant productivity and C input (Savadogo et al., 2007; Wu et al., 2010). Third, vegetation recovery reduces C losses from wind erosion due to the denser plant canopy and higher mulch cover (Zhou et al., 2011). For a few sites, nonsignificant change or even decrease of soil C has been observed as a result of GE (Fig. 3; Appendix Dataset S1). Similar results have also been reported for grasslands in other regions of the world (Wienhold et al., 2001; Reeder and Schuman, 2002). Reasons for the decrease may be historical grazing practices and grazing intensities before GE (Shrestha and Stahl, 2008).

Overall, for China's grassland, the mean rate of soil C stock change was 0.27, 0.50, 0.68, 1.30 Mg ha^{-1} yr^{-1} in 0–10 cm, 0–20 cm, 0–30 cm, and 0–100 cm, respectively (Fig. 3, Appendix Fig. S2). The rates were not very high. For example, in some case studies, Nelson et al. (2008) reported that the soil C sequestration rate was 1.4–2.9 Mg ha^{-1} yr^{-1} in 0–60 cm soil layer after grassland restoration in south-central Saskatchewan, Canada; Wu et al. (2010) estimated that the soil C sequestration rate was nearly 0.6 Mg ha^{-1} yr^{-1} in the 0–30 cm soil layer following 9 years of fencing in the alpine meadow of the Qinghai-Tibetan Plateau; And Qiu et al. (2013) reported an accumulation of 1.68–4.40 Mg C ha^{-1} yr^{-1} in 0–80 cm soil depths in GE grassland compared to grazed grasslands on the Loess Plateau. However, the rates are higher than some reports yet. For example, in South Africa, Talore et al. (2016) found that long term (> 75 years) GE had significant improved soil C stock in the 0–30 cm, as the C sequestration rate of 0.13 Mg ha^{-1} yr^{-1}. In Norway, Speed et al. (2014) have reported that the soil C sequestration rate was 0.26 Mg ha^{-1} yr^{-1} in the top 22–29 cm following 12 years of GE. Despite the rate of soil C sequestration after GE in china's grassland is not peak in the global, China's grassland also showed significant increase in soil C stock, this synthesis supports the view that GE is an effective approach to promote ecosystem C sequestration for grasslands in China.

4.3. Factors effects on C sequestration

4.3.1. Age of GE

In our study, the rates of changes in soil C stock (excluding the observations with the rates were less than zero) and plant biomass C significantly decreased with the years of GE ($r_{AGBC} = -0.360$, $p < 0.01$ and $r_{BGBC} = -0.257$, $p < 0.01$, Table 1), which showed an *Exponential Decay* trend since grazing exclusion (Figs. 4 and 6, Appendix Fig. S1). Moreover, the rate tends to reach equilibrium at the late stage, indicating that age plays a major role in shaping the trajectory of C dynamics. This result is consistent with the finds of Hu et al. (2016)'s study, which suggested that GE leads to temporal changes in ecosystem C pools within a short period, but does not affect long-term C dynamics. However, when we used the whole dataset collected to analysis, we found that the rates of soil C stock change had non-significant difference among different recovery ages since GE and the rate of soil C changes had no significant correlation with age ($p > 0.05$) (Table 1, Fig. 5). This suggested that age of GE isn't a critical factor to influence on the rate of soil C stock change in the regional scale of China. Soil C saturation with age and litter input changes maybe the potential reason to explain soil C stock changes (Frouz, 2017). Moreover, soil C stock in the early stage (< 3 yr) had non-significant changes ($p > 0.05$) (Figs. 4 and 5), this may be due to livestock manure inputting increased SOC and trampling increased soil BD in the grazed grassland led to soil C stock in the early period after GE had no significant difference with grazed grassland. And our study found that a significant increase in soil C stock after > 3 years of GE. These results suggests that it might take > 3 years of GE for increase in soil C storage to be significantly appreciable in China's grassland. In addition, we found that the dynamics was consistent between BGBC and soil C stocks following with GE, indicating that soil processes had correlated with plant dynamics due to plant community has an affected on soil processes (Li et al., 2009). More roots input to the soils is a main reason to lead to soil C stock increasing through vegetation recovery (Prietzel and Bachmann, 2012), and more above- and belowground biomass in GE grassland also accelerating more roots input to the soils. In our study, we also found that AGBC changes reached a steady state (when the rate at the equilibrium point) first, followed by BGBC, and then soil C (Appendix Fig. S1), indicating that the increase of BGBC lags behind the accumulation of AGBC, and the increase of soil C stock lags behind the accumulation of biomass C stock after GE. These results are consistent with the expectation that changes in soil C stock lag behind changes in vegetation biomass C, as plant biomass is the major source of soil C inputs. These results were similar with Hu et al.'s study, but Hu et al. (2016) have reported that plant biomass and SOC concentrations not reported the plant biomass C and soil C stocks.

4.3.2. Climate

Climate may affect C accumulation through those biotic processes associated with both the productivity of vegetation and decomposition of organic matter. The study showed that the AGBC and BGBC both had significant positive correlations with the MAP ($p < 0.01$), but they hadn't significant correlations with the MAT ($p > 0.05$) (Table 1). This indicates that GE can sequestrate more C under wetter climatic conditions and plant biomass C is mainly determined by MAP rather than the MAT in China's grassland. Many studies have found that plant productivity tend to be faster and higher under wetter than under drier conditions in the grassland ecosystem worldwide (Bai et al., 2004; Luyssaert et al., 2007; Ma et al., 2008). However, the influence of precipitation varies across different regions: in North America precipitation explains 90% of the variation in grassland aboveground biomass (Sala et al., 1988), whereas in Inner Mongolia it is 43–57% for temperate grasslands (Ma et al., 2008), only 18% for alpine grasslands on the Tibetan Plateau (Yang et al., 2009). However, the findings of this study imply that with the increase of MAP, the enhancement of plant photosynthesis is greater than ecosystem respiration, which promoted the increase of biomass C pools for the grasslands ecosystems in China (Hu et al., 2016). In addition, the MAT rather than MAP had significantly positive effect on the rate of C changes in the top 30 cm soil layers ($p < 0.05$) (Tables 1, 2), indicate that high temperature promoted soil C sequestration after GE in China's grassland. Previous studies reported that high temperature can improve microbial activity in the top soils, and then increase soil respiration making soil C output into atmosphere (Luo and Zhou, 2010), meanwhile, higher microbial activity promoted the decomposition of biomass residues (litters, dead roots, etc.) leading to the increase of the organic C input into the soil (Anderson et al., 2008). As a result soil C inputs more than outputs following with GE, resulting in soil C sequestration enrichment in the regions with higher temperature. This may be the potential mechanisms support this increase of soil C stock in higher temperature regions. We also found that in the deeper soil of 30–100 cm, the rate of soil C changes had no significant correlation with both MAP and MAT ($p > 0.05$) (Table 1). This imply that the deeper soil C sequestration maybe determined by other factors, such as soil pH, soil microbe, and roots rather than climate factor.

4.3.3. Grassland type and soil properties

Vegetation type is a key factor to effect C sequestration of ecosystem (Mcsherry and Ritchie, 2013; Deng et al., 2014a, 2014b). Prietzel and Bachmann (2012) have reported that different species with different plant traits can impact on retentions of soil C, for example, influence on releasing nutrients to soil via mineralization (Mueller et al., 2012). In our study, we found the grassland dominated by grass species had

Table 3
Effects of grassland types (grass and forb) on C sequestration following with GE.

Grassland types	ABGC ($g\ m^{-2}\ yr^{-1}$)	BGBC ($g\ m^{-2}\ yr^{-1}$)	Soil C changes ($Mg\ ha^{-1}\ yr^{-1}$)			
			0–10 cm	10–20 cm	20–30 cm	30–100 cm
Grass	10.98 ± 2.87a	32.03 ± 17.46a	0.27 ± 0.12a	0.24 ± 0.12a	0.19 ± 0.08a	0.09 ± 0.04a
Forb	3.71 ± 1.39b	20.32 ± 43.37a	0.24 ± 0.21a	0.16 ± 0.17a	0.14 ± 0.14a	0.07 ± 0.08a

Note: The different letters indicate significant difference between the two grassland types at 0.05 level ($p < 0.05$). Values are mean ± CI (95%).

higher rate of C change in all C pool components than the grassland dominated by forb species (Table 3). In China's grassland, the dominant grass species have *Leymus chinensis*, *Stipa grandis*, *S. capillata*, *S. bungeana*, *Leymus secalinus*, *Kobresia tibetica*, *K. humilis*, *Pennisetum centrasiaticum*, etc., and the dominant grass species have *Allium polyrhizum*, *Serratula centauroides*, *Seriphidium transiliense*, *Caragana microphylla*, *Agriophyllum squarrosum*, *Trifolium repens*, *Androsace erecta*, *Artemisia scoparia*, *A. halodendron*, *A. capillaries*, *A. frigid*, etc. (Appendix Dataset S1). Previous studies reported that grasses dominant grasslands have greater productivity and faster turnover rate of fine roots than forbs dominant grasslands (Gallego et al., 2004; Wu et al., 2009), and plant productivity and roots turnover are two important driver to C sequestration in plant and soil (Matamala et al., 2003; Deng et al., 2014b), which caused the results of our study (Table 3). Especially in the AGBC, our results showed that grass dominant grasslands had much higher rate of C change than the forb dominant grasslands (Table 3), also indicated that grasses have more greater aboveground productivity than forbs.

Soil C changes strongly coupled with soil N change have reported by many studies (Yang et al., 2011; Wang et al., 2014; Deng and Shangguan, 2017; Hu et al., 2016) also, in our present study, we found that a significant linearly positive correlation between the rates of soil C change and the rates of soil N change was found in grasslands with GE. Furthermore, C—N interactions are very important in determining whether the C sink in land ecosystems can be sustained over the long term (Luo et al., 2006; Deng and Shangguan, 2017), and soil C—N coupling relationships were also closely related to age (Deng and Shangguan, 2017). Luo et al. (2004) have reported that N dynamics are a key factor in the regulation of long-term terrestrial C sequestration. In our study, we found that the C/N ratios at GE sites was significantly higher ($p < 0.05$) in the late stage (> 20 years) compared to the early recover stage (< 20 years), and soil C/N ratios at grazing exclusion sites were significantly higher compared with grazing sites in the late recovery stage (i.e., > 20 years) (Fig. 8). This implies that GE had more effects on soil C accumulation than soil N and the accretion of N could not meet the demand of C increase at the later stage in the GE site. As Luo et al. (2004) reported that N probably progressively becomes progressively more limiting as C accumulates in one ecosystem if the N total does not change in the ecosystem, especially in the later stage of vegetation restoration (Luo et al., 2004). Therefore, for a long term (i.e., > 20 years), the increase of soil N will reduce N limitation and might be an important mechanism of the continuous C sequestration during the period of GE. However, the C/N ratios had no significant difference in the early stages (< 20 years) after GE, and the ratios at grazing sites had been no significant difference (Fig. 8). This suggested that soil C and N show similar trends under grazing sites and in the early years of GE and soil C dynamics are closely coupled with N dynamics which may be a reason why the C/N ratios had no significant difference in a relative short period (i.e., < 20 years) between GE sites and grazing sites (Fig. 8). In addition, the rates of soil C stock change were significant decreased from surface soil (0–30 cm) to deeper soil (30–100 cm) (Fig. 7). Previous studies reported that increased organic matter (litter, dead roots, mycorrhizae, and exudates) input resulting from vegetation biomass to the soil leads to SOC increases through vegetation recovery (Nelson et al., 2008; Prietzel and Bachmann, 2012), as well as decreased erosion are probably the main factors contributing to the sequestration of soil C (Nelson et al., 2008; Zhou et al., 2011). Furthermore, root biomass increases is larger in top soil layers than that in the deeper soil layers compared to GE sites to grazing site is the reason surface soil had higher rate of soil C change than deeper soils (Zhu et al., 2016).

4.4. Implications of GE for grassland management

Our results indicated that the grazing exclusion in a region of severely degraded grassland had a positive effect on C accumulation in both biomass and soil in China. To assess the importance of our findings to the global C cycle, we extrapolated our findings across the whole of China (Appendix Table S1). Assumed China has implement comprehensive 'Returning Grazing Land to Grassland' Project, we roughly estimate that the C stock in China grassland ecosystems could increase by up to 0.21 pg yr^{-1} in the above- and belowground vegetation C pool and 0–100 cm soil C pool (Appendix Table S1). The value suggested China's grassland has a large C sequestration ability under the condition of GE. However, a cessation of grazing is in conflict with the policy goal of increasing food production in China, so completely GE is not possible. To better play C sequestration ability of China's grassland, we suggest the complementary application of more active restoration techniques, such as control the intensity of grazing (Gan et al., 2012), rotational or seasonally grazing (Pei et al., 2008; Wang et al., 2011), and so on. Similar management recommendations might be applied in other grassland ecosystems with similar histories and patterns of soil degradation. For example, in the upland grasslands of Scotland (Dennis et al., 2008), the African savannas (Ogada et al., 2008). Furthermore, despite the distinct trajectories (the rate at the equilibrium point), all the C pools (both plant biomass C and Soil C) reach equilibrium after > 15 years of GE (Figs. 4 and 6). To our knowledge, this is the second study followed by Hu et al. (2016) that simultaneously quantifies the dynamics and the duration for C pools to reach steady state for grasslands with GE. The findings of this study have valuable implications for C sequestration through GE. With the information on the duration and dynamics of C sequestration before reaching the steady state, the C sequestration potential for grassland ecosystems can be evaluated.

4.5. Uncertainty analysis

Compared to other meta-analyses or synthesis, our synthesis features a relatively larger number of studies (n = 118), which offers the most accurate estimate on C sequestration following GE across the whole China. Strict accuracy is limited due to the uneven distribution of data collected in each age group. Additionally, many of the studies have no long term observations and consequently, these measurements may add to the uncertainty. Furthermore, we were unable to evaluate several other potentially important factors, such as soil pH, historical grazing practices and grazing intensities before GE, type of grazing management (rotational vs. continuous), and wild vs. domestic grazers, because they were not measured in most of the studies we surveyed. Therefore, considerable knowledge gaps about the effects of grazing on C sequestration in grassland ecosystem still exist and suggest major areas of further research. In addition, for a few sites, nonsignificant change or even decrease of soil C has been observed as a result of GE, particularly in the first 3 years (Appendix Dataset S1). Reasons for the decrease may be historical grazing practices and grazing intensities before GE (Shrestha and Stahl, 2008). As there are insufficient data on grazing intensity over time for most sites in our synthesis, the influence of historical grazing practices on soil C dynamics during the period of GE could not be elaborated. However, despite these limitations, our synthesis did reveal several interesting and informative patterns that reflect the importance of considering the environmental and biotic context of grazing in management decisions designed to help mitigate greenhouse gases and store soil C.

5. Conclusions

With the recovery age > 27 years under the China's ‘Returning Grazing Land to Grassland’ Project. Most sites had a positive impact of GE on vegetation and soil C stock except some individual cases, indicating that GE is an effective management practice to restore degraded grasslands and improve C stock (0.21 pg yr^{-1}). Soil C stock rates and vegetation biomass C changes showed an *Exponential Decay* trend since GE, and the AGBC changes reached a steady state (when the rate at the equilibrium point) first, followed by BGBC, and then soil C. Plant biomass C is mainly determined by MAP rather than the MAT in China's grassland, but it was just the opposite for the soil C in top 30 cm soil depths. For the deeper soils, C sequestration maybe determined by soil pH, soil microbe, roots, etc. rather than climate factor. Moreover, soil N is a key factor in the regulation of soil C sequestration since long term GE (> 20 years), suggesting that increased soil N supply to grasslands with GE at the latter recovery stage may enhance ecosystem C sequestration capacity. The findings of this study have valuable implications for C sequestration through GE. With the information on the duration and dynamics of C sequestration before reaching the steady state, the C sequestration potential for grassland ecosystems can be evaluated.

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.earscirev.2017.08.008.

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